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Riparian buffers and forest thinning: Effects on headwater vertebrates 10 years after thinning

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ABSTRACT

We monitored instream vertebrate and stream-bank-dwelling amphibian counts during a stand-scale experiment of the effect of riparian buffer width with upland forest thinning in western Oregon, USA using a before/after/control methodology. We analyzed animal counts along 45 streams at 8 study sites, distributed from the foothills of Mount Hood to Coos Bay, Oregon using data collected pre-treatment and during the first decade post-treatment. We examined the role of four types of stream buffers in explaining the variability in post-treatment animal counts. We built separate linear regression models for stream-bank and instream animals, examining species and species-assemblages of specific interest. Stream-bank models addressed all amphibians, the subset of all terrestrial-breeding amphibians, *Plethodon dunni*, and *Plethodon vehiculum*, which were the two most abundant stream-bank species. Instream models were examined for all vertebrates, the subset of all stream-breeding amphibians, *Dicamptodon tenebrosus*, and *Rhyacotriton* species. All bank and instream models considered buffer treatment, survey area, stream width, pre-treatment count, and number of days post-treatment as possible explanatory variables. Instream models also considered survey method: hand sampling or electrofishing. Along banks there was support for a negative effect of the two narrowest buffers in the all-species model and the terrestrial-breeding amphibian assemblage model, and for an apparent negative effect of the narrowest buffer in the *P. dunni* model. Nevertheless, *P. dunni* were retained as one of the most common species along stream banks throughout the 10-years of our post-treatment monitoring. Instream, complex interactions among covariates in the model precluded determination of consistently positive or negative effects of buffers on animal counts. This is the first study to test the riparian reserve widths of the US federal Northwest Forest Plan, and it is encouraging that we documented no negative effects of those buffers with upland thinning in headwater drainages. Narrower buffers appeared to pose a risk to stream bank animals. Nevertheless, with our moderate thinning regime and in treatments with all buffer widths, species occurrences were retained through time. The joint buffers-with-thinning treatments appear to be relatively benign and may be reconciled by the designed long-term habitat restoration benefits of the thinning-and-buffer prescriptions. Mixed widths of buffers might be considered to hedge uncertainties, and balance the socioeconomic and ecological benefits of thinning in riparian areas with risks to some species.

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1. Introduction

Legal mandates for biodiversity protection are drivers of riparian forest protection worldwide. In the United States, broad concerns for stream-and-riparian dependent species were raised in the 1960s (e.g., Everest and Reeves, 2007). Subsequently, three federal laws provided impetus for development of forest riparian protection guidelines: the Clean Water Act of 1972 (http://www.epa.gov/lawsregs/laws/cwa.html; accessed 27 February 2013); the Endangered Species Act of 1973 (ESA; http://www.fws.gov/laws/lawsdigest/esact.html; accessed 27 February 2013); and the National Forest Management Act of 1976 (NFMA; http://www.fs.fed.us/emc/nfma/index.htm; accessed 27 February 2013). In the European Union, the Water Framework Directive (WFD; 2000) protects running waters through river-basin management plans. Indicators of success include protection of species of interest, conservation of biodiversity through designation of protected areas, and species diversity (http://www.eea.europa.eu/data-and-maps/indicators/; accessed 10 June 2013). The Habitats Directive (1992), which established the Natura 2000 network, a

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network of sites to protect Europe's most valuable and threatened species and habitats, provides additional legislative incentive for protection of biodiversity in forest-riparian areas across Europe. The Environment Protection & Biodiversity Conservation Act in Australia and The River Law in Japan provide similar legal frameworks supporting the protection of riparian forest biodiversity.

Riparian forest biodiversity protections in the US have been widely applied to the maintenance and restoration of habitat conditions for ESA-listed species, the rarest or most threatened species in dire need of protection. Secondarily, US regulations have been used to forestall perceived threats or disturbances that degrade habitat conditions that might lead a species with sensitive or concern status towards an ESA-listing proposal, especially on federally managed lands (e.g., Suzuki and Olson, 2007). The National Forest Management Act of 1976 specified the need to maintain viable populations of existing native and desired nonnative vertebrate species, hence extending protections to all species in this subphylum on National Forest lands.

Together, the content of these US laws was used to develop the federal Northwest Forest Plan (NFP) in the Pacific Northwest (USDA and USDI, 1993, 1994). The NFP was developed as a response to curtail the trends toward ESA-listing of late-successional and old-growth forest-dependent species in the region. Over 1000 forest-dependent species were considered during NFP development (Thomas et al., 1993, 2006; USDA and USDI, 1993, 1994). Expanded streamside riparian reserves were mitigation measures of the aquatic conservation strategy included in the NFP to provide for aquatic- and riparian-dependent species persistence (Reeves et al., 2006). In particular, both salmonids and amphibians were of high concern regionally (e.g., Stouder et al., 1996; Olson, 2009), and figured prominently in NFP riparian reserve design (Hohler et al., 2001; Olson and Burnett, 2013; Reeves, 2006; USDA and USDI, 1996a,b).

Salmonids and amphibians can be considered 'biodiversity indicators' (Holthausen and Sieg, 2007), representative of larger aquatic and riparian communities. Their population trends may reflect recent anthropogenic disturbances to forested aquatic–riparian habitats as well as patterns of disturbance over the last century (Harding et al., 1998; Walter and Merritts, 2008). In forested landscapes, these taxa often rely on physical habitat attributes such as down wood, microclimate conditions (e.g., cool temperatures) that may be limited in some areas, water availability in certain seasons, and clean water resources without excessive erosion or stream sedimentation. To maintain or restore these habitat elements for aquatic species, protective stream buffers of various widths have been designed (e.g., Gregory et al., 1991; Naiman and Décamps, 1997; Naiman et al., 2000; USDA and USDI, 1994). Prior to 1993, 30-m riparian zones were retained along fish-bearing streams on US federal lands west of the Cascade Range. After reassessment of scientific advances in 1993, the NFP expanded these areas for management consideration to the current 90–145 m interim riparian reserve zone along each side of fish-bearing streams, and added an interim riparian reserve zone of up to 45–70 m along non-fish bearing streams (these widths correspond to the distance of one or two site-potential tree heights – the tallest height a tree can grow at a site, a metric used for provision of down wood, shading and other attributes; e.g., Cissel et al., 2006; Everest and Reeves, 2007; USDA and USDI, 1994, 1996a). These are the widest riparian buffers in the region (Olson et al., 2007).

Most legal mandates for protection of freshwater biota have a scientific basis related to watershed ecology, and some of these considerations originate in small streams. Headwater streams can comprise up to ~80% of forested stream networks in the Pacific Northwest (Gomi et al., 2002), and affect the development of downstream aquatic habitat conditions such as down wood (Reeves, 2006; Reeves et al., 2003), sediment (Benda and Cundy, 1990; Benda and Dunne, 1997a,b; Rashin et al., 2006), and invertebrate prey (Wipfli and Gregovich, 2002). However, until relatively recently, stream buffers were not mandated on small headwater streams, and most managed forests in their first forest-harvest rotations were clearcut without headwater stream protection. For example, on federal forest lands west of the Cascade Range, buffers in small fishless streams have been implemented only as of 1994 (USDA and USDI, 1994). Although protections are provided to many forested headwaters now, small-stream management approaches vary considerably with land ownership, and some headwaters lack protection (Olson et al., 2007). How much headwater protection is warranted is still a point of controversy because few studies have reported on the natural resource values and effects of protection on species and habitats in this uppermost portion of the stream network. Although several recent advances have increased our understanding of small stream characterizations (Kroll et al., 2008; Janisch et al., 2011) and stream–riparian species and habitat responses to forest management effects (De Groot et al., 2007; Hawkes and Gregory, 2012; Jackson et al., 2007; Janisch et al., 2012; Kreutzweiser and Capell, 2001; Leuthold et al., 2012; Raphael et al., 2002; Rykken et al., 2007a,b; Stoddard and Hayes, 2005; Vesely and McComb, 2002; Wilk et al., 2010; Wilkins and Peterson, 2000), there continues to be a need for baseline information on the effects of alternative stream–riparian protective measures, particularly given the variety of site conditions, forest practices, species, and habitats across the landscape. Our study contributes to narrowing these knowledge gaps by being the first relatively long-term and spatially extensive experimental study examining the effects on aquatic and semi-aquatic amphibians of alternative headwater stream buffer widths in managed forests after upland thinning.

In 1994, we initiated our riparian buffer study as part of the larger Density Management Study of Western Oregon (Cissel et al., 2006). The aim of this overarching framework was to examine upland forest density management approaches to accelerate development of late-successional forest characteristics in managed federal forests. We monitored instream vertebrates and stream-bank amphibians as part of the riparian component, using a before/after/control methodology. The riparian buffers were specifically designed in response to the interim riparian reserve widths identified in the NFP, extending one and two site-potential tree-height widths on each side of streams (Hohler et al., 2001; USDA and USDI, 1994). We examined those widths in addition to two narrower buffers within the upland density management "moderate retention" prescription (Cissel et al., 2006). Previously, we reported on pre-treatment vertebrate assemblages found in and along streams at 12 sites (Olson and Weaver, 2007), headwater assemblages occurring in unmanaged old-forest 'reference' sites near one of our treatments (Sheridan and Olson, 2003; also plant assemblages, Sheridan and Spies, 2005), post-treatment responses of instream and bank assemblages 1–2 years after thinning at 11 sites (Olson and Rugger, 2007), and responses of upland salamanders at selected sites 1–2 years post-treatment at 2 sites (Rundio and Olson, 2007) and 5–6 years post-treatment at 3 sites (Kluber et al., 2008). Animal responses to buffers have been variable among our different studies, with no dominant or consistent pattern emerging in the first years after thinning. We propose that a longer-term response might be easier to detect and of greater ecological importance for these relatively long-lived animals.

Herein, we examine instream vertebrates (fish and amphibians) and streambank amphibians throughout a 15-year time period, 1995–2010, including a 10-year post-treatment time span at our western Oregon study sites. Our objective was to assess whether there was an effect of buffer width on instream and streambank communities during the first decade after upland forest thinning. We built the simplest possible statistical model to explain the

variability in post-treatment animal counts after accounting for pre-treatment counts. We examined models for 8 data subsets: (1) all amphibians occurring along stream banks; (2) the subset of terrestrial-breeding amphibians along banks – these animals are semi-aquatic or upland-associated; (3) all vertebrates (fish and amphibians) within wetted stream channels; (4) the subset of stream-breeding amphibians in streams, whose larvae are aquatic; and single-species models for the most abundant species in our two habitats; (5) instream *Dicamptodon tenebrosus* Baird and Girard (coastal giant salamander); (6) instream *Rhyacotriton* species (torrent salamander species); (7) bank-dwelling *Plethodon dunni* Bishop (Dunn's salamander), and (8) bank-dwelling *Plethodon vehiculum* Cooper (western red-backed salamander).

2. Materials and methods

2.1. Study sites

Our 10-year post-treatment study analyzed data from 8 managed forest sites on Bureau of Land Management (BLM) lands in the western Cascade and Coast Ranges of Oregon (Fig. 1 and Table 1). Implementation variances among sites have constrained our study over time, resulting in 8 sites and 45 stream reaches (segments of streams) being analyzed here, whereas 11 sites and 68 reaches were analyzed previously (1–2 years post-harvest; Olson and Rugger, 2007). Sites were located within the *Tsuga heterophylla* [Raf.] Sarg. (western hemlock) vegetation zone (Franklin and Dyrness, 1988), and were chosen to be representative of young managed forest stands within BLM administrative units of the area (Olson et al., 2002). Sites were previously clearcut without stream buffers, and stands were naturally regenerated (430–600 trees per hectare, tph) and had aged 30–40 years (6 sites) and 70 years (2 sites) at the start of the study (Cissel et al., 2006). Experimental timber harvest occurred at all sites between 1997 and 2000, usually within a 2-year time window per site, and reduced tree density to about 200 tph at all sites. Note that Perkins Creek, an older stand which had been thinned 20 years earlier to 250 tph, was an exception. This was its second-entry thinning and it was thinned to 100–150 tph. The thinning prescription in our study included some 0.1–0.4-ha gaps and leave islands (Cissel et al., 2006), except at Callahan Creek, an older unthinned stand, that was thinned with no such upland heterogeneity.

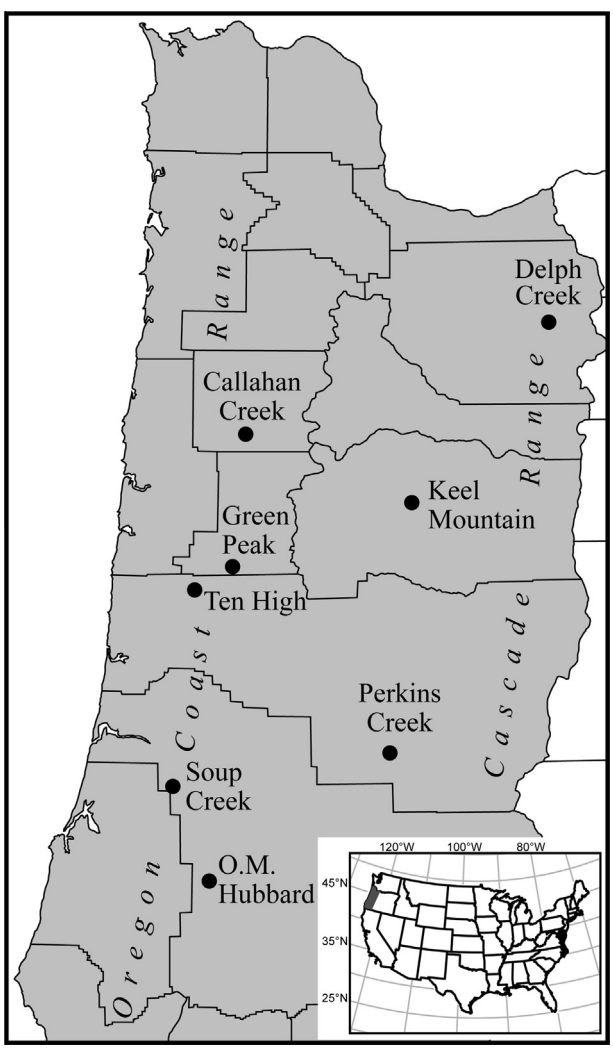

Fig. 1. Study sites in western Oregon, USA examining the responses of instream vertebrates and stream-bank amphibians to experimental treatments of different streamside riparian buffer widths with upland forest thinning.

A total of 45 stream reaches were included in the study (Table 1). Treatment reaches were protected by one of four riparian buffer widths: streamside retention ("streamside buffer," 6 m), variable width ("variable buffer," 15 m minimum buffer), one-tree height (~70 m buffer), or two-tree height (~145 m buffer). However, stream-to-ridge distance constraints precluded the implementation of the two wider buffers on some streams (Fig. 2); we aimed for about 60 m of thinned upland forest between buffers and ridges. Number of stream reaches within study units also constrained implementation of all buffer treatments at each site. Consequently, there were fewer replicates of the one- and two-tree height buffers. Data from those two treatments were combined as "wide" buffers for analyses. At least one reach at each site was left unthinned as a reference (control) treatment. Stream reach lengths were approximately two site-potential tree heights long, 110–150 m. Assignment of treatments to stream reaches was done randomly whenever possible.

2.2. Field data collection

Surveys were conducted in and along our 45 stream reaches during 1 year pre-treatment and at approximately years 1, 2, 5, and 10 post-treatment during the spring rainy seasons. A modified Hankin and Reeves (1988) approach was used in which we first conducted habitat surveys, followed by animal sampling (Olson and Rugger, 2007; Olson and Weaver, 2007). Reaches were classified as slow- (pools) and fast-water (riffles/cascades) habitat "units," moving upstream from the bottom-most end of the reach. For the purpose of our analyses herein, we measured stream width and length per unit. We later calculated the average width among all units in the reach, weighted by the sum of all unit lengths (reach length), to obtain the reach-level width.

For animal sampling, we subsampled 10 units per reach. We eliminated ~15 m from the bottom- and top-most reach areas from our pool of units for sampling, and then calculated the proportion of slow- and fast-water habitat unit types in the remaining reach that could be sampled. Units with a surplus of overhanging vegetation or instream down wood that obscured access and visibility could not be sampled and were omitted. We sampled units in proportion to their frequency in the reach. The first unit sampled was randomly chosen among the first five units up from the excluded section at the downstream end of the reach. The remaining sampled units were systematically chosen along the upstream reach such that every *n*th unit was sampled, where n = number of units by type (slow or fast water) divided by the remaining units.

Table 1
Number of reaches used for analyses of vertebrate responses to thinning and buffers, by study site and riparian buffer treatment, in western Oregon, USA. Number of reaches also displayed by search method used for instream data collection.

Site	Unthinned control	1–2 Tree height (70–145 m)	Variable width (15 m min.)	Streamside retention (6 m)	Total
Riparian buffer width					
Callahan creek	1	2	2	2	7
Delph creek	1	0	2	1	4
Green peak	2	1	1	1	5
Keel mountain	2	2	1	1	6
North soup creek	2	1	1	1	5
O.M. Hubbard	1	2	2	0	5
Perkins creek	2	0	2	2	6
Ten high	1	1	3	2	7
Total no. reaches	12	9	14	10	45
Hand search	12	4	9	9	34
Electrofishing	0	5	5	1	11

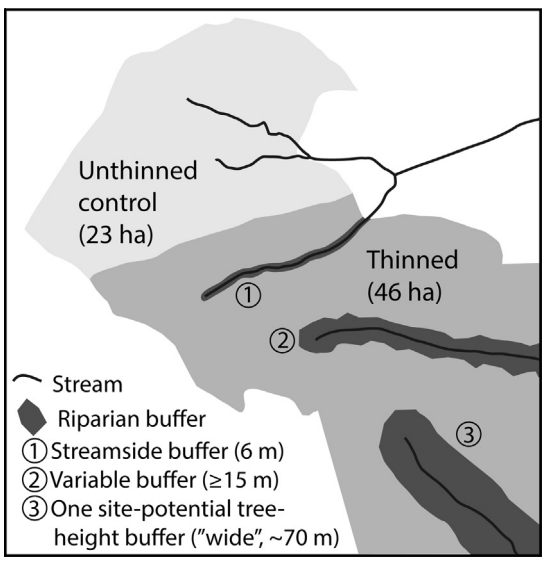

Fig. 2. Schematic diagram of the layout of a study site for our western Oregon riparian buffer width study. At this site, Green Peak, the headwater stream network geometry permitted implementation of only three of four riparian buffer widths within the thinned forest upland, plus two stream reaches extending into the unthinned control forest stand. We were unable to implement the widest buffer of our overall study at this site, which extended two site-potential tree heights on each side of stream reaches.

Instream and bank sampling was conducted separately at each unit. Instream units were sampled either by hand (34 reaches) or by electrofishing (11 reaches), dependent upon whether or not fish were present in the reach (Table 1). Fish counts could not be reliably achieved by hand samples, and although amphibians may have been hidden in substrates and undersampled by electrofishing, our consistent efforts among units and reaches likely reduced bias. We sampled entire slow-water units and 2-m unit lengths of fast-water units by electrofishing. Upper and lower unit boundaries were blocked with nets, and we made two to five passes with the electrofisher and dip nets until we achieved a 75% reduction in captures of all species from the first pass, or captured no animals in a pass. We stopped at the fifth pass if those criteria were not achieved. Hand sampling was conducted in fishless reaches. Animals were captured with dip nets placed downstream from cover objects and were 'herded' into nets by hand as cover objects were moved and replaced, in a 'soft-touch' approach. For both types of instream surveys, surveyors moved from downstream to upstream, placing captures in buckets as they proceeded through the unit. Captures were identified to species and returned to units. Instream survey areas were summed per reach for analyses.

Bank surveys were conducted for 5 min on each side of the stream, within 2 m of the water's edge, at the same units that were identified for instream sampling. Bank surveyors looked in, on, and under substrates, and had flexibility to move up or downstream to use the full 5 min of survey time in an effective way. For example, if a unit's bank was totally bedrock with no cover objects, they could shift their search to an adjacent unit with substrate or down wood that served as cover for animals. Animals were placed in plastic bags upon capture, and were identified and returned to their location of capture after the search. Animal handling times were not counted toward search times. The area surveyed was estimated at each unit. Bank survey areas were summed per reach for analyses.

2.3. Statistical analyses

We calculated the number of days before and after thinning to fine-tune calculations of time-since-thinning beyond the estimated 1, 2, 5, and 10 year sampling windows. Using this approach allowed a more accurate reflection of time since thinning of each unit within which a stream reach was located. We were able to account for variable access to study sites, field crew scheduling, and the synchrony of the completion of thinning with the spring survey season using this approach. For example, data collected 1-year post-treatment ranged from 1 to 669 days after the last date of thinning (Table 2) because some sites could be sampled immediately post-thinning and other sites could not be entered for almost 2 years due to continued logging nearby. Pre-treatment data were collected 649–1245 days before upland forest thinning and data collected during the final sampling period was collected 2967–3880 days (~8–10½ years) post-treatment (Table 2).

Prior to analysis, pre-treatment counts, post-treatment counts, and total survey area were standardized to ten sampling units along each stream reach. For example, if data showed that there were only seven usable sampling units along a reach instead of 10, the pre-treatment count, post-treatment count, and total survey area were multiplied by 10/7 as a way of estimating the total amounts. This was done to ensure that the summary metrics used

Table 2
Timing of post-treatment data used in analyses of the effects of thinning and buffers on instream and bank vertebrates in western Oregon, USA.

Survey	Years	No. days before or after thinning
1-Year pre-treatment	1995–1998	649–1245, before
1-Year post-treatment	1998–2003	1–669, after
2-Years post-treatment	1999–2004	345–1042, after
5-Years post-treatment	2002–2006	1468–2117, after
10-Years post-treatment	2007–2010	2967–3880, after

for each reach were based on ten sampling units. These anomalies of the data were relatively rare, occurring on ~6% of sampled reaches.

Due to the complementary distributions of the two *Rhyacotriton* species among sites, with *R. variegatus* Stebbins and Lowe occurring in the Coast Range and *R. cascadae* Good and Wake occurring in the Cascade Range, they were combined to one taxon for analysis. These two torrent salamander species are ecologically similar (Jones et al., 2005) and have been recognized as separate species relatively recently through genetic analyses (Good and Wake, 1992).

We analyzed 8 subsets of the data to determine if we could observe effects of buffer width on animal assemblages or on the most common species, by habitat type – streambank or instream. These 8 subsets were instream vertebrates, instream stream-breeding amphibians, instream *D. tenebrosus*, instream *Rhyacotriton* species, bank amphibians, bank terrestrial-breeding amphibians, bank *P. dunni*, and bank *P. vehiculum*. Our analysis goal was to find the most parsimonious model to explain variability in post-treatment animal counts. Considerable pre-treatment variability in animal counts was apparent among reaches and sites. Hence, we used animal count from pre-treatment data collections as a covariate in all analyses. Consequently our models examined post-treatment counts over time after accounting for pre-treatment counts. In addition to buffer treatment, days since treatment, and pre-treatment counts, we considered the area surveyed and average stream width as potential predictor variables in all models. We were able to model the raw animal counts rather than catch per unit effort or relative abundance because we applied a standard search effort at each sampling unit, further standardized counts and search area to 10 sampling units per stream reach, and included search area as a possible covariate in our models. For instream models, we also assessed the effect of survey method (hand sampling or electrofishing) (Table 1).

We used a linear mixed model (assuming Gaussian errors) with square root transformations applied to both the pre- and post-treatment counts. Log transformations, which are more commonly used, showed a greater degree of non-constant variance in the residual plots and require adjustments for zero values. Residual and normal probability plots were used to assess the validity of assuming Gaussian errors with constant variance. Using square-root transformed data, these plots showed no extreme departures from normality or constant variance. We chose Bayesian Information Criteria (BIC) to evaluate our models because we were searching for the most parsimonious model that would explain our data. For the bank analyses, stream reaches were modeled with a compound symmetry covariance structure, meaning one overall variance parameter (residual variance) and one covariance parameter for observations on the same reach. For the instream analyses, a compound symmetry covariance structure was again shown to be adequate; however, separate covariance structures were specified for each survey method (hand search or electroshock). Covariance structures were chosen based on BIC values. Due to the unbalanced study design of treatments with repeated measures, the Kenward-Rogers degrees of freedom adjustment was used in all analyses, as suggested by Littell et al. (2006). All analyses were completed using PROC MIXED in SAS, version 9.2 (SAS Institute Inc., Cary, NC, USA). We report *p*-values for all estimated coefficients and use the term "significant" to refer to *p*-values less than 0.1.

We note that count data are often analyzed using generalized linear models with a log-link function (i.e., Poisson or negative binomial regression). However, these data were overdispersed relative to the Poisson distribution. Poisson regression, allowing for overdispersion, led to convergence issues for some of our more complicated models (highly parameterized with complex covariance structures). Using a linear mixed model (assuming Gaussian errors) with square root transformations allowed us to accurately model overdispersed data within a relatively robust modeling framework.

In order to find the most parsimonious model for estimating effects of buffer treatments on animal counts, all possible models were considered with the following stipulations: (1) buffer treatment and pre-treatment count were included in every model, (2) models that included interaction terms must also have included the main effects of the interaction and all lower-order interactions, and (3) the highest-order interaction terms must have had a *p*-value < 0.05. We then selected the simplest model (smallest number of parameters) that was within 10 BIC units of the lowest BIC value.

3. Results

Overall, we captured 5771 individuals of 15 fish and amphibian species during our study of headwater streams and stream banks, inclusive of the cottid taxon as a group and each of the two *Rhyacotriton* salamander species found. Although many species were detected in small numbers and hence would be considered incidental captures, the overall biodiversity of instream and bank habitats in our western Oregon managed forest sample was relatively high. We detected all but one (*Ambystoma macrodactylum* Baird) of the native forest-occurring amphibian species with Oregon ranges coincident with our study sites (Jones et al., 2005). Species detected and classifications of amphibian species to breeding assemblage (terrestrial, stream or pond) are provided in Tables 3 and 5.

3.1. Stream bank

Stream bank surveys yielded 1158 individuals of 12 amphibian species over the course of the study (Table 3). The terrestrial-breeding yet semi-aquatic *P. dunni* was the most-common species detected, found at all study sites. The terrestrial-breeding *P. vehiculum* was the second most-abundant species along banks, but did not occur at some sites. Pond-breeding amphibians were incidental captures on stream banks, as were some terrestrial- and stream-breeding amphibians.

The stream bank analyses (Table 4) suggest an overall buffer effect within the all-amphibians ($p = 0.032$), terrestrial-breeding amphibians (0.088), and *P. dunni* ($p = 0.071$) models, after adjusting for pre-treatment count. For the all-amphibians and terrestrial-breeding amphibian models, the two narrower buffers, 6 m and 15 m, had significant negative effects on post-treatment animal counts after ~10 years. For *P. dunni*, the streamside buffer, 6 m, had a significant negative effect on ~10-year post-treatment counts. There was no evidence of an overall buffer effect within the *P. vehiculum* species group, after adjusting for pre-treatment count ($p = 0.684$).

Box-and-whisker plots (Fig. 3) show patterns in the transformed counts for each species or group over time. Similar count patterns are visually evident among the all-species, terrestrial-breeding amphibians, and *P. dunni* box plots (Fig. 3, white boxes): (1) an increase can be seen in post-treatment year 1 compared to pre-treatment in the control reaches, wide buffers, and variable-buffer reaches and (2) a decrease is apparent in post-treatment year 1 compared to pre-treatment in the streamside-buffer reaches. Of note, two zero median counts for *P. dunni* are shown after this initial decrease in the streamside-buffer reaches, during years 2 and 10 post-treatment – these likely were dominant contributing factors to the apparent negative effect of the streamside buffers on *P. dunni* counts. Despite over 1000 captures overall, relatively low occurrences of bank captures at the reach scale (Fig. 3) are a consideration in interpreting these results, with a median of zero captures for *P. vehiculum* (the second-most abundant species) in three post-treatment measurement periods in the unthinned

Table 3
Stream bank amphibian counts by breeding assemblage (terrestrial, stream, or pond) at 8 study sites in western Oregon, USA. Total post-treatment count combining years 1, 2, 5, and 10 (pre-treatment count); ANFE = *Aneides ferreus* Cope, clouded salamander; BAWR = *Batrachoseps wrighti* Bishop, Oregon slender salamander; ENES = *Ensatina eschscholtzii* Gray, Ensatina; PLDU = *Plethodon dunni* Bishop, Dunn's salamander; PLVE = *P. vehiculum* Cooper, western red-backed salamander; ASTR = *Ascaphus truei* Stegneger, coastal tailed frog; DITE = *Dicamptodon tenebrosus* Baird and Girard, coastal giant salamander; RHspp = *Rhyacotriton* species, torrent salamanders; AMGR = *Ambystoma gracile* Baird, northwestern salamander; PSRE = *Pseudacris regilla* Baird and Girard, Pacific treefrog, RAAU = *Rana aurora* Baird and Girard, northern red-legged frog; TAGR = *Taricha granulosa* Skilton, rough-skinned newt.

Breeding assemblage and species	Callahan creek	Delph creek	Green peak	Keel Mt.	No. soup creek	O.M. Hubbard	Perkins creek	Ten high	Total
Terrestrial									
ANFE	0 (0)	0 (0)	1 (0)	1 (0)	0 (0)	3 (0)	4 (0)	1 (0)	10 (0)
BAWR	0 (0)	1 (2)	0 (0)	16 (1)	0 (0)	0 (0)	0 (0)	0 (0)	17 (3)
ENES	0 (0)	4 (0)	2 (0)	16 (2)	0 (0)	11 (1)	2 (1)	8 (0)	43 (4)
PLDU	47 (12)	4 (0)	61 (25)	32 (5)	86 (23)	46 (8)	146 (30)	31 (7)	453 (110)
PLVE	39 (10)	0 (0)	54 (37)	0 (0)	49 (20)	19 (7)	3 (0)	37 (14)	201 (88)
Stream									
ASTR	2 (2)	0 (0)	9 (0)	2 (0)	1 (0)	0 (0)	0 (0)	6 (0)	20 (2)
DITE	2 (0)	2 (4)	5 (2)	1 (3)	2 (0)	11 (1)	1 (0)	4 (0)	28 (10)
RHspp	3 (1)	6 (3)	22 (4)	7 (2)	0 (1)	17 (2)	0 (0)	10 (3)	65 (16)
Pond									
AMGR	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	2 (0)	0 (0)	2 (0)
PSRE	2 (0)	1 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	3 (0)
RAAU	13 (0)	2 (0)	1 (0)	1 (0)	1 (0)	0 (0)	0 (0)	1 (0)	19 (0)
TAGR	2 (0)	0 (0)	21 (2)	1 (0)	16 (4)	4 (1)	1 (0)	0 (1)	45 (8)
Unknown									
Salamander	0 (0)	0 (0)	1 (5)	0 (0)	0 (0)	1 (0)	0 (0)	2 (0)	4 (5)
Frog	0 (0)	0 (0)	0 (0)	2 (0)	0 (0)	0 (0)	0 (0)	0 (0)	2 (0)
Total	110 (25)	20 (9)	177 (75)	79 (13)	155 (48)	112 (20)	159 (31)	100 (25)	912 (246)

Table 4
Stream bank amphibian models examining the effects of riparian buffer width and pre-treatment animal counts on post-treatment animal counts 10 years after forest thinning, using data from 45 reaches within 8 study sites in western Oregon, USA. Final models selected using BIC. Riparian buffer treatment is a fixed effect; reach is treated as a random effect.

Model	Independent variable	*F*-test df	Coefficient estimate	Standard error	*P*-value
All amphibians	Intercept		1.30	0.21	<0.001
	Buffer treatment	3/40			0.032
	1–2 Tree height		−0.24	0.24	0.306
	Variable width		−0.49	0.21	0.026
	Streamside retention		−0.65	0.23	0.007
	Pre-treatment count	1/40	0.483	0.069	<0.001
Terrestrial-breeding amphibians	Intercept		1.02	0.20	<0.001
	Buffer treatment	3/40			0.088
	1–2 Tree height		−0.16	0.23	0.481
	Variable width		−0.41	0.21	0.053
	Streamside retention		−0.53	0.22	0.023
	Pre-treatment count	1/40	0.511	0.068	<0.001
Plethodon dunni	Intercept		0.62	0.17	<0.001
	Buffer treatment	3/40			0.071
	1–2 Tree height		0.10	0.22	0.660
	Variable width		−0.13	0.20	0.528
	Streamside retention		−0.48	0.21	0.031
	Pre-treatment count	1/40	0.577	0.073	<0.001
Plethodon vehiculum	Intercept		0.15	0.11	0.192
	Buffer treatment	3/40			0.684
	1–2 Tree height		0.09	0.15	0.546
	Variable width		−0.08	0.13	0.529
	Streamside retention		−0.03	0.14	0.832
	Pre-treatment count	1/40	0.545	0.049	<0.001

control reaches, and in 8 of 24 (1/3; Fig. 4) of the reaches with thinning and buffers in the post-treatment measurement periods.

3.2. Instream

Surveys conducted in streams detected 4613 individuals of 13 taxa over the entire time span of the study (Table 5). *D. tenebrosus*, a stream-breeding amphibian (Table 3), was the most consistently captured animal, followed by the *Rhyacotriton* species, another stream breeder which was detected at all but one site. Tailed frogs and two fish taxa were abundant at some sites. Incidental captures of many species were apparent.

Instream models of animal counts yielded more complex relationships than the bank models (Table 6). Models of instream species counts consisted of combinations of additive (main) effect variables and multiplicative (interaction) effects. Such complexity makes it difficult, if not impossible, to isolate the contributions of each explanatory variable on its own. For example, although buffer was significant in many models, the buffer interactions with other variables make its effect on amphibian counts challenging to decipher. Buffer effects varied with survey method (hand search or electroshock), pre-treatment count, and average stream width.

The instream box-and-whisker plots of captures (Fig. 4) display greater counts of instream animals as compared to bank animals. A

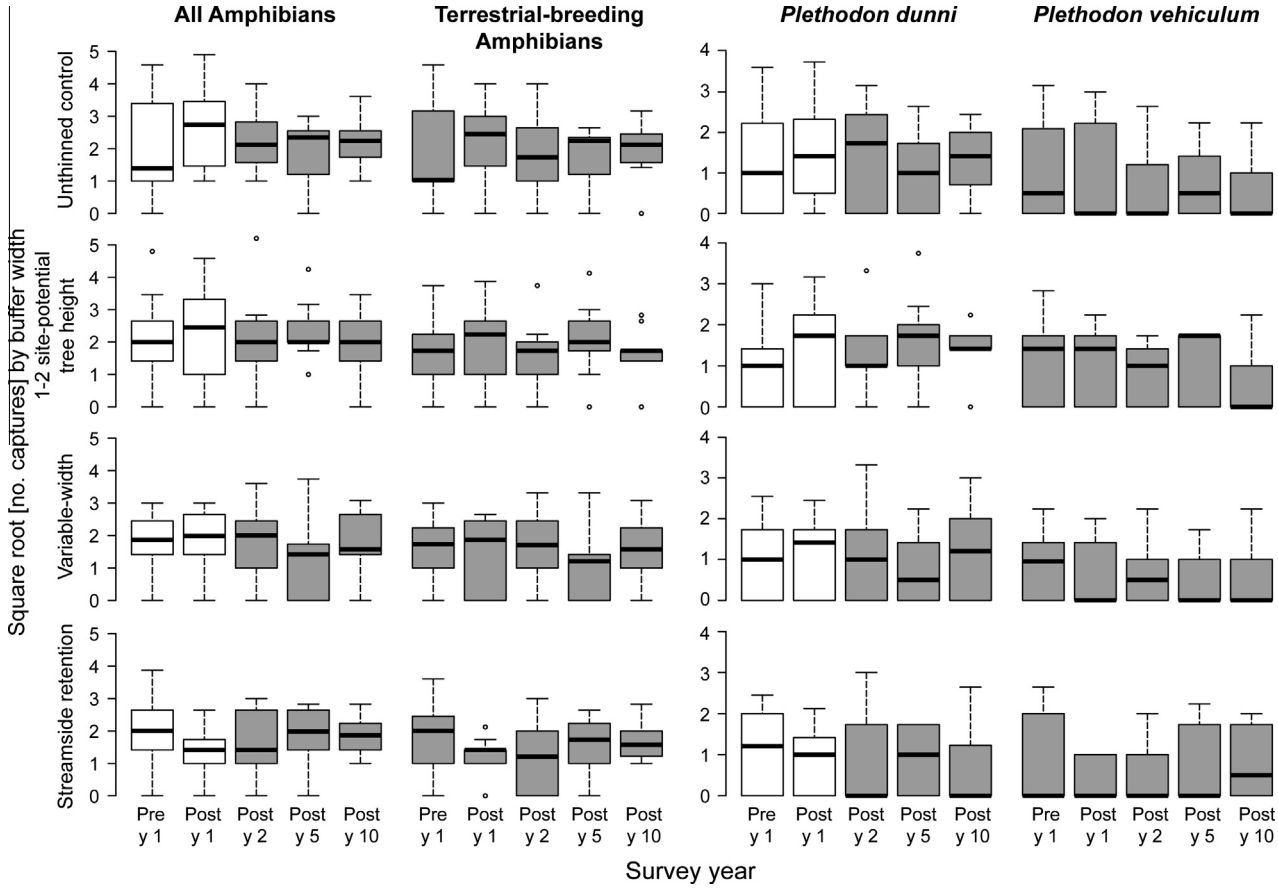

Fig. 3. Box-and-whisker plots of animals captured during stream bank surveys at 8 sites in western Oregon, USA. Results from three buffer treatments with upland forest thinning and an unthinned control buffer is shown: streamside retention (6 m); variable-width (15 m minimum); and 1–2 site potential tree height ("wide" buffer, 70–145 m). Terrestrial-breeding amphibians (Table 3) were separated as a group; PLDU = *Plethodon dunni*, Dunn's salamander; PLVE = *P. vehiculum*, western red-backed salamander; Line in box = median; box = 25–75th percentile of the data; whisker = 1.5 times the interquartile range; dot = outlier point.

pattern of increasing counts over time is evident in stream reaches with the wide buffer, except for *Rhyacotriton* species. *D. tenebrosus* counts in the wide buffer initially decreased in years 1 and 2 post-treatment and then increased in years 5 and 10, indicating a possible initial negative effect and then a rebounding positive effect of wide buffers on that species, suggesting a longer-term beneficial management effect. The few *Rhyacotriton* salamanders found among sites may make pattern detection difficult for that species. Median reach counts were never zero for *Rhyacotriton*, providing evidence for their persistence at treated reaches.

The instream data show that the control and streamside-buffer reaches had remarkably fewer animals pre-treatment and throughout the time span of the study, a pattern that did not appear for the banks. This implicates a potential inadvertent bias in buffer placement within sites, which could be related to animal counts. If the wide-buffer treatments were located in larger subdrainages, simply to fit them within the area available, and more animals occurred in larger drainage basins, then an underlying pattern of overall buffer treatment effects could be obscured. However, our inclusion of stream width in our models is a proxy measure for drainage area (correlation = 0.82; GIS-estimated drainage areas per study reach ranged from 2.4 to 175.7 ha), and hence it is at least partially accounted for in our analyses.

4. Discussion

All study species were present up to a decade following forest thinning along headwater streams, and thus timber harvest did not cause clear, severe, persistent declines or local extirpations of amphibians and fish at our sites. However, we documented a reduction in counts of stream-bank amphibians following upland harvest. These reductions were strongest at sites with the narrowest riparian buffers. Instream animal counts showed no reductions over the decade-long study. With caveats due to the specific context of our study, and potential issues of low sample size, occupancy, and detection probability that are explained below, we found that: (1) the 5 and 10-year effects of combined thinning and our narrowest buffers (6 m and ~15 m) had an apparent negative effect on the stream bank assemblage, with the narrowest buffer affecting Dunn's salamanders (*P. dunni*) in particular; (2) no adverse effects were observed on instream vertebrates from any buffer width with our upland thinning regime; (3) a nonsignificant trend for increasing *D. tenebrosus* counts was evident in streams with the wide buffers; and (4) pre-treatment counts explained variation in post-treatment numbers in both our instream and bank samples, lending support for the importance of local conditions. Our study validates the efficacy of the one- and two-site potential tree height riparian reserves of the Northwest Forest Plan to retain species in headwater streams. We did not detect adverse effects of these relatively wide stream–riparian buffers on fish and amphibian counts in our study within ~10 years of their installation, and there was a suggestion of a possible positive effect on *D. tenebrosus* counts of these wide buffers. Hence, our study supports the concept of streamside buffers as an effective management tool at the reach scale for maintaining fish and amphibians within managed headwaters.

Context is important when considering forest-riparian protection zones, with integral factors including: location, site history,

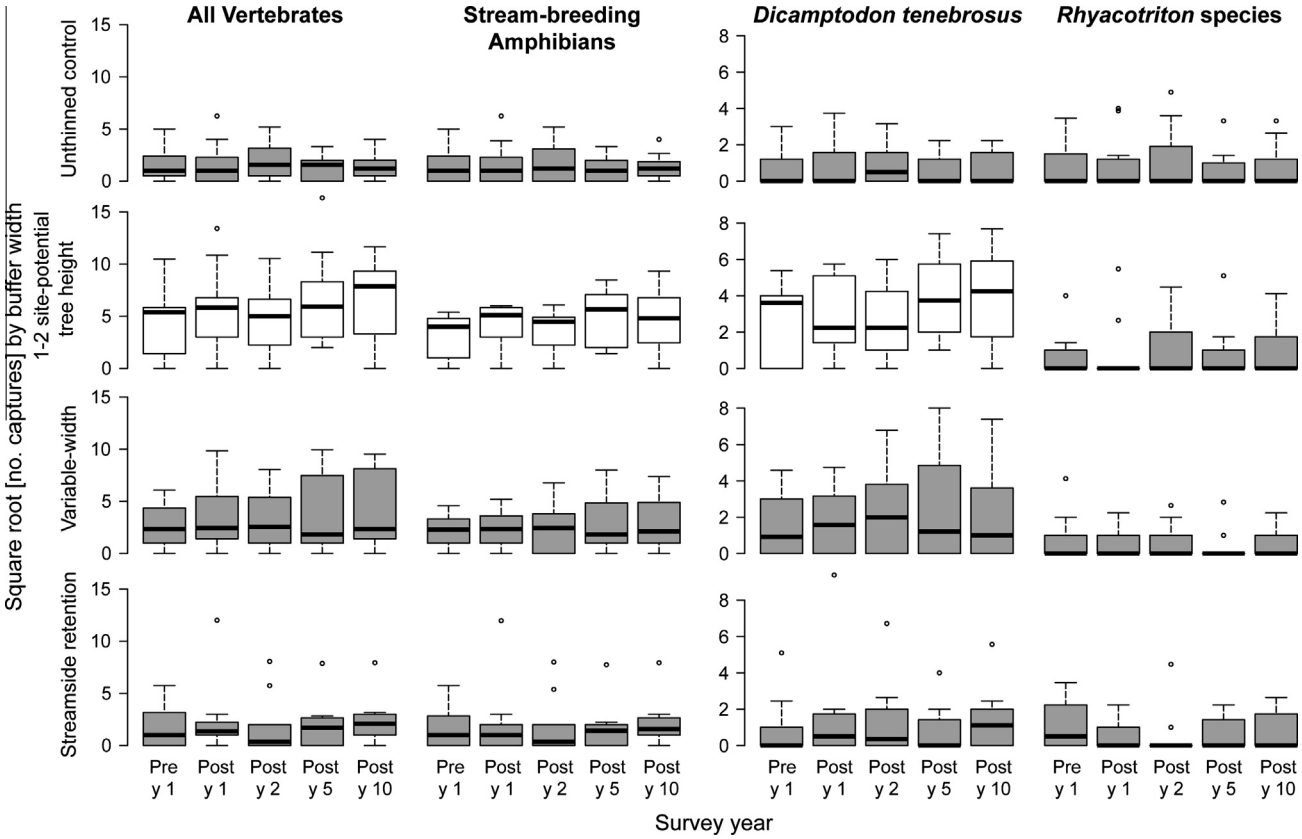

Fig. 4. Box-and-whisker plots of animal captures from instream surveys at 8 sites in western Oregon, USA. Results from three buffer treatments with upland forest thinning and an unthinned control buffer is shown: streamside retention (6 m); variable-width (15 m minimum); and 1–2 site potential tree height ("wide" buffer, 70–145 m). Stream-breeding amphibians (Table 5) were separated as a group; DITE = *Dicamptodon tenebrosus*, coastal giant salamander; RHspp = *Rhyacotriton* species, torrent salamanders; Line in box = median; box = 25–75th percentile of the data; whisker = 1.5 times the interquartile range; dot = outlier point.

Table 5
Instream vertebrate counts (amphibians are listed by breeding assemblage: terrestrial, stream, or pond) at 8 study sites in western Oregon, USA. Post-treatment count from combined years 1, 2, 5, and 10 (pre-treatment count); ONCL = *Oncorhynchus clarkii clarkii*, coastal cutthroat trout; Trout 0+ = age 0 + years trout, species undetermined; Cottid = sculpin species; ANFE = *Aneides ferreus*, clouded salamander; BAWR = *Batrachoseps wrighti*, Oregon slender salamander; ENES = *Ensatina eschscholtzii*, Ensatina; PLDU = *Plethodon dunni*, Dunn's salamander; PLVE = *P. vehiculum*, western red-backed salamander; ASTR = *Ascaphus truei*, coastal tailed frog; DITE = *Dicamptodon tenebrosus*, coastal giant salamander; RHspp = *Rhyacotriton* species, torrent salamanders; PSRE = *Pseudacris regilla*, Pacific treefrog, RAAU = *Rana aurora*, northern red-legged frog; TAGR = *Taricha granulosa*, rough-skinned newt.

Species	Callahan creek	Delph creek	Green peak	Keel Mt.	N. soup creek	O.M. Hubbard	Perkins creek	Ten high	Total
Fish									
ONCL	333 (37)	2 (11)	0 (0)	149 (42)	0 (0)	0 (0)	35 (16)	0 (0)	519 (106)
Trout 0+	195 (0)	0 (2)	0 (0)	0 (0)	23 (0)	0 (0)	24 (0)	0 (0)	242 (2)
Cottid	802 (121)	79 (3)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	881 (124)
Terrestrial									
ANFE	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	1 (0)	0 (0)	1 (0)
BAWR	0 (0)	0 (0)	0 (0)	1 (0)	0 (0)	0 (0)	0 (0)	0 (0)	1 (0)
ENES	0 (0)	0 (0)	1 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	1 (0)
PLDU	2 (0)	0 (0)	5 (0)	1 (0)	3 (2)	7 (1)	0 (0)	2 (1)	20 (4)
PLVE	0 (1)	0 (0)	0 (0)	0 (0)	0 (0)	0 (1)	0 (0)	1 (1)	1 (3)
Stream									
ASTR	257 (13)	0 (0)	21 (4)	53 (0)	1 (0)	1 (0)	0 (0)	86 (14)	419 (31)
DITE	484 (100)	183 (22)	78 (29)	430 (45)	15 (0)	18 (4)	247 (13)	94 (14)	1549 (227)
RHspp	5 (1)	13 (4)	183 (44)	5 (1)	9 (9)	38 (19)	0 (0)	98 (16)	351 (94)
Pond									
PSRE	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (0)	0 (1)	0 (1)
RAAU	2 (0)	1 (0)	0 (0)	0 (0)	1 (0)	0 (0)	0 (0)	0 (0)	4 (0)
TAGR	0 (0)	0 (0)	5 (0)	4 (0)	10 (3)	3 (0)	0 (3)	1 (0)	23 (6)
Unknown									
Frog	0 (0)	0 (0)	0 (0)	3 (0)	0 (0)	0 (0)	0 (0)	0 (0)	3 (0)
Total	2080 (273)	278 (42)	293 (77)	646 (88)	62 (14)	67 (25)	307 (32)	282 (47)	4015 (598)

Table 6
Instream animal models examining the effects of riparian buffer width and pre-treatment animal counts on post-treatment animal counts 10 years after forest thinning, using data from 45 reaches within 8 study sites in western Oregon, USA. Response variable is square-root transformed. Buffer treatment is a fixed effect; reach is treated as a random effect. Final models selected using BIC. DF = degrees of freedom.

Model	Independent variable	*F*-test DF	Coefficient estimate	Standard error	*P*-value
All vertebrates	Intercept		−13.9	10.1	0.178
	Buffer treatment	3/59.6			0.042
	1–2 Tree height		11.8	10.6	0.274
	Variable width		21.3	10.3	0.047
	Streamside retention		0.92	0.89	0.304
	Hand search (yes/no)	1/34	13.5	10.1	0.451
	Buffer × hand search	2/40.8			0.011
	1–2 Tree height		−11.1	10.6	0.303
	Variable width		−21.1	10.3	0.048
	Streamside retention				
	Pre-treatment count	1/29.5	0.72	0.12	<0.001
	Buffer × pre-treatment count	3/28.2			0.039
	1–2 Tree height		0.35	0.19	0.076
	Variable width		−0.19	0.20	0.344
	Streamside retention		−0.22	0.19	0.256
	Stream width	1/33.6	13.7	6.8	0.032
	Buffer × stream width	3/66.5			0.127
	1–2 Tree height		−12.1	6.9	0.090
	Variable width		−15.1	6.9	0.035
	Streamside retention		−0.9	1.1	0.415
	Hand search × stream width	1/34.6	−12.4	6.8	0.136
	Buffer × hand search × stream width	2/47.8			0.051
	1–2 Tree height		11.6	7.0	0.108
	Variable width		14.8	6.9	0.038
	Streamside retention				
Stream-breeding amphibians	Intercept		4.1	1.2	0.003
	Buffer treatment	3/17.3			0.174
	1–2 Tree height		−3.0	1.1	0.021
	Variable width		−2.7	1.1	0.032
	Streamside retention		−0.28	0.25	0.283
	Hand search (yes/no)	1/30.2	−4.1	1.1	0.001
	Buffer × hand search	2/11.5			0.015
	1–2 Tree height		3.8	1.1	0.006
	Variable width		2.4	1.1	0.054
	Streamside retention				
	Pre-treatment count	1/29.1	0.674	0.067	<0.001
	Stream width	1/94.2	0.84	0.32	0.010
Dicamptodon tenebrosus only	Intercept		−20.5	8.8	0.026
	Buffer treatment	3/54.3			0.028
	1–2 Tree height		22.6	9.1	0.019
	Variable width		26.1	9.0	0.007
	Streamside retention		−0.55	0.54	0.310
	Hand search (yes/no)	1/33.2	20.5	8.8	0.199
	Buffer × hand search	2/36.7			0.016
	1–2 Tree height		−22.7	9.1	0.019
	Variable width		−26.4	9.0	0.006
	Streamside retention				
	Pre-treatment count	1/26.9	0.591	0.065	<0.001
	Stream width	1/31.5	15.5	5.8	0.005
	Buffer × stream width	3/58.7			0.011
	1–2 Tree height		−15.2	6.0	0.016
	Variable width		−17.2	5.9	0.007
	Streamside retention		0.94	0.73	0.200
	Hand search × stream width	1/31.9	−14.9	5.8	0.084
	Buffer × hand search × stream width	2/40.8			0.016
	1–2 Tree height		16.2	6.0	0.011
	Variable width		17.8	5.9	0.005
	Streamside retention				
Rhyacotriton species	Intercept		0.10	0.18	0.596
	Buffer treatment	3/20.5			0.463
	1–2 Tree height		−0.08	0.21	0.702
	Variable width		0.06	0.20	0.772
	Streamside retention		−0.26	0.27	0.339
	Pre-treatment count	1/32.6	0.90	0.13	<0.001
	Buffer × pre-treatment count	3/31.5			<0.001
	1–2 Tree height		0.30	0.18	0.092
	Variable width		−0.42	0.17	0.021
	Streamside retention		−0.40	0.19	0.042

forest management activities and objectives, and species of consideration for protection. The context of our study includes it being the first thinning entry to managed stands of moist coniferous forest in headwaters that had been previously clearcut with no ripar-

ian buffers. The forest structure and faunal assemblages that we examined at our sites are representative of much of the federal forest landscape in western Oregon, and also may have application to many conditions southward along the north coastal zone of California and northward into western Washington and British Columbia.

Due to potential lag-time effects on long-lived amphibians with relatively low fecundity, continued monitoring could reveal effects on animal counts over longer time periods. For example, it may take up to 6 years for a young torrent salamander to reach reproductive maturity and a coastal giant salamander may live >20 years (Jones et al., 2005), hence the time of our study may only begin to reflect implications of upland forest thinning with streamside buffers on long-term survival, reproduction, or population and metapopulation dynamics. Due to the life history traits of many amphibians such as their associations with moist habitats, their relatively low movement patterns and site fidelity, Blaustein et al. (1994) called for collection of multi-site, long-term monitoring data at the temporal scale of at least one complete turnover of a population, in conjunction with experiments, to assess the responses of amphibian populations to a change in conditions. Of note for our study, we assessed the stability of count data over time rather than population performance. The additional context to note for our study is that the sites were previously clearcut, so that if the headwater aquatic vertebrate assemblages are not resilient to the disturbances of those earlier timber harvests, the system may not have yet stabilized from that initial disturbance. Relative to the current timber harvest with buffers, however, the assemblages appear relatively resilient.

4.1. Stream bank

Along stream banks in the narrowest stream buffer examined (6 m), we found that counts of the most frequently captured species, *P. dunni*, were reduced during the decade after thinning. We found this negative effect of 6-m buffers was echoed in models with the larger data sets of all terrestrial-breeding amphibians and all amphibians, which may have been driven by the *P. dunni* counts alone. Additionally, in the all-amphibians and terrestrial-amphibians models, significant negative effects on post-treatment counts resulted from the variable buffer (a minimum of 15-m wide). Upon examination of the count data used in analyses (Fig. 3), animal count reductions are evident between the pre-treatment year and the first post-treatment year for data collected in the 6-m streamside retention buffer, except for *P. vehiculum*. These decreasing trends contrast with the increasing counts between these same 2 years in the other treatments. In the streamside retention buffer, the *P. dunni* count declined further to a median of zero in year 2 post-treatment, which was repeated in year ~10 post-treatment (Fig. 3). Although *P. dunni* median counts could be described as low and variable in the reaches with the other treatments, they did not reach zero. An adverse effect of the narrowest stream buffer with upland thinning is implicated.

A reduced post-treatment count of a plethodontid salamander such as *P. dunni* could arise from other factors including altered behavior or habitat use, such as remaining underground during daylight hours, and low occupancy and detection probability. Whereas we have no observations to support altered behavior or ecology, in a case study at our Green Peak study site, we examined salamander occupancy and detection probability both along stream banks and instream using the same search methods as those used in this study (L. Biga, D. Olson, and A. Blaustein, Oregon State University and US Forest Service, unpubl. data). We found *P. dunni* occupancy ranged from 0.45 to 0.85, and detection probability ranged 0.3–0.45. In contrast, using our instream hand-sampling methodology, occupancy was considerably higher at 0.83 for *D. tenebrosus* and 0.90 for *R. variegatus*, and detection probabilities ranged from 0.35 to 0.53 for *D. tenebrosus* and was 0.65 for *R. variegatus*. However, there was no pattern of occupancy and detection probability with buffer treatment at Green Peak. Hence, whereas the lower occupancy and detection probability of *P. dunni* may affect our findings, we have no *a priori* support for these factors being linked to treatment. Occupancy and detection biases could affect pre-treatment counts as well. Kroll et al. (2008) found that occupancy patterns of amphibians in small streams varied with stream substrate, gradient, and stand age (factors that were not addressed in our analyses). We agree with Kroll et al. (2008, 2009) that low occupancy casts a shadow of uncertainty when interpreting abundance metrics, such as our count data, but we also believe that studies monitoring amphibian counts over time can provide useful ecological insights and should not be dismissed (Welsh et al., 2009).

Interestingly, in earlier studies of upland salamanders at our study sites, effects of thinning on animal captures were detected at only one site, immediately after the harvest. A 40% reduction of *P. vehiculum* and *Ensatina eschscholtzii* Gray was found within the thinned area at one of two sites in years 1–2 post-treatment (Rundio and Olson, 2007). In a later follow-up study in years 5–6 post-treatment at 3 sites (Kluber et al., 2008), no effect of thinning was found on upland captures of these two species. Kluber et al. (2008) found 60% of their amphibian captures occurred within 15 m of streams, and consequently suggested that thinning with buffers inclusive of vital habitat such as subsurface rock and legacy down wood could protect amphibians in managed headwater forests. Complementing these Oregon findings, Hawkes and Gregory found no effect of clearcut timber harvest with stream buffers on upland *P. vehiculum* and *E. eschscholtzii* 10-years post-harvest in western Washington (Hawkes and Gregory, 2012).

Species persistence after timber harvest is an important point of emphasis in these studies because on managed forest lands, the combined goals of timber harvest for wood production and maintenance or restoration of ecological values and processes are being balanced. Our forest system is not retained as a no-entry reserve, yet in many regards the ecological resiliencies of the system elements are being tested. Can we pose a short-term risk to system elements, such as a species like *P. dunni*, for a short-term gain in wood production for socioeconomic benefits or for longer-term gains in ecological restoration because near-stream conditions could be improved? Our finding of a reduction of *P. dunni* in our 6-m buffer stream reaches raises a concern about the extent of a potential adverse effect of conducting partial timber harvests close to streams for some aquatic-dependent fauna. *P. dunni* is a semi-aquatic salamander that does not require flowing or standing water for any part of its life history; it is a terrestrial breeder that is commonly found along stream banks, especially along small streams (Storm, 2005; Sheridan and Olson, 2003).

Our finding for *P. dunni* may help formulate hypotheses for follow-up studies. In particular, at our Oregon sites, is the downward trend real and how do the animals continue to respond to the treatments over time? In Washington, this stream-bank species is listed with sensitive status due to rarity and threat concerns (http://wdfw.wa.gov/conservation/endangered/All/; accessed 27 February 2013). *P. dunni* is a peripheral species in north-coastal California, where its distribution extends over the border from Oregon. Populations at these north and south range limits may be more vulnerable to threats, with the northward extent of the range occurring largely on Washington industrial forest lands that have generally narrower headwater stream buffer guidance and more intensive timber harvest practices (Olson et al., 2007; Suzuki and Olson, 2007). To the south, potential habitat changes from climate change are a concern, such as increasing mean temperatures. Several studies have provided support for stream buffer widths greater

than 6 m to retain cool stream–riparian microclimate conditions from the warming, drying edge effects of forestry activities (e.g., relative to clearcut harvests: 45–300 m, Brosofske et al., 1997; 30 m, Rykken et al., 2007a; relative to the same buffers and thinning treatment that we examined here: >15 m, Anderson et al., 2007), and might be considered to forestall climate change effects, especially to the south if those warming trends are more severe in that region.

4.2. Instream

A buffer effect was found in our instream models, however, due to interacting factors, we cannot assign a positive or negative effect to buffers, nor pull out a specific buffer width that was important in explaining our animal count data. Examination of the count data (Fig. 3) reveals one visible trend of increasing counts over time for the wide buffers, which extended upslope ~70–145 m on each side of streams. This general trend of increasing counts holds for *D. tenebrosus* (Fig. 3), the most common salamander in our streams. Although it is somewhat unsatisfying to have inconclusive results relative to buffers and instream animals, we can say with confidence that a negative effect of buffers does not appear to be supported, and a pattern exists for this one species of increasing numbers with the widest buffers.

Although the majority of biodiversity concerns with timber harvest are relative to known or potential adverse effects (e.g., amphibians: Bury and Corn, 1988; deMaynadier and Hunter, 1995), some studies have found that aquatic-dependent species have either no response to harvest or have exhibited increased abundances in response to harvest. For example, no effect of timber harvest on *D. tenebrosus* was reported in a paired watershed study in western Oregon (Leuthold et al.,2012), no effect of clearcutting and buffers on upland terrestrial salamanders was reported in western Washington (Hawkes and Gregory, 2012; however, they reported a negative effect on coastal tailed frogs, *Ascaphus truei* Stejneger, in uplands), no effects of logging on headwater coastal cutthroat trout populations was found in British Columbia, Canada (De Groot et al., 2007), some positive effects of harvest on aquatic life stages of pond-breeding amphibians were found in other US regions (Semlitsch et al., 2009), and several studies found positive responses of fish or aquatic macroinvertebrates to logging in the Pacific Northwest (e.g., Murphy and Hall, 1981; Murphy et al., 1981, 1986; Bisson and Sedell, 1984). As stated earlier, context may be extremely important for examining harvest effects on aquatic resources. For example, in the De Groot et al. (2007) study, they suggested that their findings of no effect were likely attributable to the careful logging approaches used that did not disturb stream habitats, and the relatively cooler climate that coincided with their post-treatment years.

Our findings similarly suggest that the combined buffer and thinning treatments applied around our study streams were relatively benign influences on instream animals, with a potential positive effect observed on *D. tenebrosus*. Although the inference of our findings is to our study sites, our site conditions are highly reflective of managed federal forests in western Oregon and Washington. Physical habitat conditions may be of paramount importance for stream–riparian associated vertebrates, generally, and while vegetation management activities, *per se*, may have less effect, their effects on physical habitats might be a continued focus of research. In our study, we are continuing to analyze post-treatment patterns on water availability and down wood inputs, with water and structure being potentially critical elements of headwater vertebrate habitat.

Low and variable sample sizes were issues with two instream amphibian species of particular interest in our headwater study, *Rhyacotriton* species and *A. truei*, and both fish taxa, coastal cutthroat trout and sculpins, which precluded a more in-depth analysis. Both amphibians have headwater associations (Jones et al., 2005; Olson and Weaver, 2007) and are Oregon state sensitive species (http://orbic.pdx.edu/documents/2010-rte-book.pdf; accessed 27 February 2013). Interestingly, *A. truei* numbers increased greatly over the course of our project, with only 31 individuals captured pre-treatment at all sites combined and 419 animals captured post-treatment. This suggests that our treatments may have a positive effect on *A. truei*. These captures were largely instream larvae, supporting increased reproduction. A single clutch of eggs may include 60 eggs (Brown, 2005), so that the numbers we detected might reflect only a few more clutches within the stream. Such fluctuating counts could reflect a variable life history with boom and bust years of reproduction, or differential dispersal or distribution patterns among years. Relative to *Rhyacotriton* species, we included instream analyses because they were found at 7 of our 8 sites – but our results are inconclusive relative to buffer widths, which could be due to low sample sizes. Yet their persistence at sites speaks to either their potential resiliency to the treatments we imposed, or the benign nature of those treatments for this salamander taxon. Similarly, for fish, persistence at sites is quite apparent. The most abundant fish were sculpins at Callahan Creek, and they showed a fluctuating pattern of captures over time: 121 captures pre-treatment; 206 in year ~1 post-treatment; 118 in year ~2; 284 in year ~5; and 194 in year ~10. More research is warranted on the effects of forest management and stream buffers on all of these species, especially given the importance of site specific contexts for understanding potential effects of harvest.

5. Conclusion

The goal for biodiversity in managed forests differs significantly among US land ownerships in the Pacific Northwest (Suzuki and Olson, 2007). Some land-use objectives rate biodiversity protection very high, such as on federally administered lands, whereas other land uses prioritize socioeconomic objectives over species concerns. Our findings over a 10-year post-treatment period monitoring amphibian counts in old clearcuts after subsequent thinning with riparian buffers attests to the relative resiliency of the headwater fauna to habitat management. Our counts of animals within instream and stream bank assemblages reflect fairly persistent levels of the chief headwater-associated species, with abundant counts most evident for *D. tenebrosus*, and other species at some sites. For many species with low or patchy occurrences, headwater streams and stream banks may not be their primary habitats, hence their apparently incidental occurrences are not surprising. Documentation of headwater-species persistence at our managed forest sites is encouraging for a landscape that has been previously clearcut without consideration of headwater natural resource values. Forest managers with objectives to maintain or restore late-successional forest conditions, such as the overarching goal of our thinning and buffer treatments (Cissel et al., 2006), might consider small-scale risks an acceptable trade-off for long-term ecological gains. In our study, the greatest apparent risk is of the narrowest buffer (6 m) to stream bank assemblages, and *P. dunni* in particular.

Based on these findings, avenues of consideration to future managers include: (1) acknowledge the potential risk to some species and manage riparian forests to within 6 to 15 m of streams to meet forest management priorities or (2) in a more precautionary context, manage the potential risk to species, and expand protective zones beyond a 6–15 m minimum width. Microclimate considerations in headwater streams and riparian areas (e.g., Anderson et al., 2007), in addition to results of other studies (reviewed in Ol-

son et al. (2007)), have provided support to retain buffers >15 m wide on each side of stream reaches.

However, a one-size-fits-all approach to riparian forest management may not meet both socioeconomic and ecological objectives at the stand or broader landscape or watershed scale. A mixed approach with calculated short-term risks at sites for long-term gains within watersheds could be considered to design a win–win solution to enhance socioeconomic benefits such as wood production and longer-term ecological restoration, and address biodiversity protection. The aquatic conservation strategy of the NFP is a regional approach to retain aquatic-dependent resources, and its objectives were intended to be attained at larger spatial scales, with periodic unfavorable conditions within any particular watershed (Reeves et al., 2006). Hence a calculated risk such as implementing a 6- to 15-m riparian buffer at a stream reach with upland restoration forestry, especially when other stream reaches within the drainage are managed more conservatively to hedge uncertainties, seems consistent with current policy for federal lands within the NFP area. Targeting aquatic–riparian species protections at known or suspected biodiversity or species hotspots, while considering areas of intrinsic potential for high quality habitat (salmonids: Burnett et al., 2007), habitat refugia such as cool, moist north-facing slopes (e.g., Suzuki et al., 2008; Shoo et al., 2011), and identified aquatic–riparian connectivity corridors (Olson and Burnett, 2013) may help refine a mixed-bag approach of riparian buffer widths in managed forests. This mixed-buffer approach remains a conceptual design, and monitoring would be needed to validate its efficacy in retaining biodiversity and other aquatic–riparian values, especially with an uncertain future of anthropogenic and natural disturbances, to forests and to forest-dependent species.

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