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Stream buffers ameliorate the effects of timber harvest on amphibians in the Cascade Range of Southern Washington, USA

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ABSTRACT

We addressed the efficacy of stream-side buffers in ameliorating the effects of clearcut timber harvest on Cascade torrent salamanders (*Rhyacotriton cascadae*), coastal/Cope's giant salamanders (*Dicamptodon tenebrosus*/*D. copei*), coastal tailed frogs (*Ascaphus truei*), and water temperature regimes in the Cascade Range in southern Washington. Forty-one streams in 4 categories were sampled; streams in clearcuts with and without buffers, streams in 35+ year old second-growth forest, and streams in unharvested forest (150+ years old). Tailed frog and Cascade torrent salamander densities were 2–7-fold lower ($P < 0.05$), respectively, in streams in managed forests than in streams in unharvested forest. In addition, both these species were less abundant ($P < 0.05$) in unbuffered streams than streams with buffers or in second-growth forest. In contrast, giant salamander densities were 5–50% greater ($P > 0.05$) in managed streams than unharvested, being greatest in unbuffered and second-growth streams. We used the differences in density estimates of unbuffered streams and unharvested streams to define an ecologically important effect size for each species and then compared the mean effect size and 95% confidence intervals of contrasts between managed stream categories to assess buffer effectiveness. Buffers had a positive ecologically important effect on the density of torrent salamanders and tailed frogs, but had an ecologically negative effect on giant salamanders. Water temperatures were similar among stream categories. However, Cascade torrent salamanders were nearly absent from streams where temperatures were $\geq 14\,^{\circ}$C for ≥ 35 consecutive hours. Issues that need further study include effective buffer width and longitudinal extent, and confirmation of the water temperature threshold we identified.

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1. Introduction

Much of the range of stream-associated amphibians in the Pacific Northwest occurs on private industrial timberlands, and the effects of timber harvest on these species has been a concern for many years (Kroll, 2009). Steele et al. (2002, 2003) examined the abundance of Cascade torrent salamanders (*Rhyacotriton cascadae*) and giant salamanders (*Dicamptodon* spp.) in streams traversing forest stands ranging from recent clearcuts to 94 years of age in managed forests of the Cascade Range of southern Washington. They found no relationship between forest age and giant salamander captures. However, Cascade torrent salamanders were least abundant in streams in forests 0–24 years old, most abundant in streams in forests 25–64 years old, and intermediate in abundance in streams in older forest. The findings of Steele et al. (2003) for Cascade torrent salamanders begged the question of the efficacy of stream-side buffers in mitigating the short-term effects of clearcut harvest in headwater streams; the subject of our study.

Presumed increase in water temperature associated with forest management are often cited as an important factor in stream amphibian distribution and abundance (Hawkins et al., 1988; Welsh and Lind, 1996; Steele et al., 2003; Olsen et al., 2007). In addition, Steele et al. (2003) reported a flat trend in the relationship between forest age and water temperature. However, most water temperature data are limited to a single estimate for each stream taken at the time of sampling for amphibians. Continuous monitoring of water temperature may provide better insight to the relationships between water temperature regimes and amphibian metrics.

Steele's et al. (2002, 2003) findings prompted our study, which was designed to assess the effectiveness of stream-side buffers left during clearcut harvest operations on the abundance of Cascade torrent salamanders, tailed frogs, and giant salamanders. We also collected more definitive data on water temperature regimes. At

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the time of our data collection (during summer 2000), few studies addressing these issues had been published and buffers were not required on streams that did not support fish. Although policy changes have been made since then, studies addressing buffer efficacy relative to maintenance of species and water temperatures in small streams remain depauperate (see Olsen et al., 2007).

2. Study area and methods

The study area was described in detail by Steele et al. (2002, 2003), occurring along the west slope of the Cascade Range in southern Washington from the Columbia River Gorge, north to the Lewis River basin in stream reaches under a variety of harvest scenarios. The majority of the area was managed for even-aged timber production, featuring Douglas-fir (*Pseudotsuga menziesii*), by private corporations and the Washington Department of Natural Resources (DNR). U.S. Forest Service lands occurred at higher elevations on the east side of the study area.

To assess the efficacy of stream buffers, we sampled headwater streams in 4 categories: (1) streams in recent clearcuts ($\leq$10 years old) with buffers, (2) streams in recent clearcuts that were not buffered (however, regulations prohibited operating ground-based equipment or application of herbicides near streams), (3) streams in second-growth plantations $\geq$35 years old, and (4) streams in unharvested forest. A sample size target of 12 or more streams per category was based on a power analysis ($\alpha = 0.05$, $\beta = 0.80$) using the difference in Cascade torrent salamander captures in 0–24 year old stands and 25–94 year old stands reported by Steele et al. (2003). At the time of site selection, there were no regulatory requirements to buffer streams on private lands that did not support fish. However, wildlife and green leave-tree requirements were sometimes met by buffering headwater streams, particularly in harvest units that did not contain streams with fish. In addition, DNR had just begun to implement a habitat conservation plan that required buffering headwater streams. Due to the limited number of buffered headwater streams in the study area, it was not possible to randomly select streams for sampling and we effectively conducted a complete census of buffered streams in the area. We then found streams in the other 3 categories that matched (Cochran, 1983), as closely as possible, the aspect, gradient, basin size, and elevation of the buffered streams within the shortest geographic distance.

During June–August 2001 we sampled 12 buffered, 10 unbuffered, 10 second growth, and 9 unharvested streams. Additional streams that met the matching criteria were not available in the later 3 categories. The buffers ranged from 5 to 23 m wide on each side of the stream. Trees in the buffers were $\geq$45 years old and the stand was clearcut beyond the buffer. The unbuffered category had no trees $\geq$10 years old within 20 m of either side of the stream. The second-growth category had a minimum of 200 m of $\geq$35 year old trees on each side of the stream and had been cut to the stream edge when previously harvested. Managed stands (unbuffered, buffered, and second-growth) were primarily on Longview Timberlands, LCC properties and the DNR. The unharvested category had streams surrounded by forests that lacked evidence of historic or recent logging (i.e., stumps or roads). Unharvested stands ranged from 150 to 300+ years old, were located on U.S. Forest Service lands, and had traits characteristic of the effects of stand-replacing fires (i.e., multi-cohort stands with large snags and downed trees); the best documented fires occurred in 1900–1950 (Felt, 1977). Stand age was determined through date of planting or tree coring.

A 45–55 m long perennial reach of each stream near its origin was selected as the sample unit. The top of the sample reach began where the stream channel was $\geq$0.25 m wide (bank-full width). Six, 2 m long plots separated by at least 3 m were sampled for amphibians in each stream, except those in unharvested forests, where time constraints limited sampling to only 3 plots per stream. The first plot was randomly located within the first 10 m of the downstream portion of each reach and subsequent plots were then systematically spaced at 3–5 m intervals. All managed stream reaches began below a road and the unharvested stream reaches were above roads.

In each plot the entire wetted width of the stream channel was intensively searched for amphibians. A screen was secured at both top and bottom of each plot prior to searching. All of the wood and rocks were removed from the plot, as possible during hand-sampling, and the remaining substrate was raked with a garden tool. A small net was held directly below the area being searched. Periodically, the screens were searched for amphibians. In addition, each side (0.5 m) of the stream was then raked with the garden tool and amphibians found were caught by hand. All captured amphibians were identified to species, except coastal and Cope's giant salamanders which were difficult to distinguish in the field so they were combined and are referred to as giant salamanders. The maximum and mean wetted width of each in-stream plot was measured to the nearest 10 cm.

Stream aspect was measured using a compass, gradient was estimated using a clinometer, and elevation was estimated using a global positioning system (GPS) unit and verified using geographic information system (GIS) coverage of Washington. Aspect, gradient, and elevation were verified from digital 7.5 min topographic maps.

Water temperature regimes were recorded from 24 July to 9 October 2000 using Onset Tidbit® data loggers, programmed to record at 1 h intervals. Loggers placed in the stream were housed in a PVC pipe T-joint, placed at the bottom of each reach, secured to a stake driven into the substrate. The overall mean, maximum, minimum and standard deviation were calculated for water temperatures. To estimate potential water temperature thresholds that may influence stream occupancy patterns, the number of consecutive hours that a stream was at a specific temperature (in 1 °C increments, starting at 8 °C, up to the maximum recorded) was calculated. These data were not collected for the unharvested streams.

Amphibian densities were estimated for each plot by dividing the number of animals captured by the area of a plot (2 m $\times$ average wetted width) plus 1 to account for the stream-side area searched and averaged across plots. This approach assumes that the probability of capturing all individuals in a plot is 1.0 and that capture probabilities are consistent across plots and across streams. We did not determine detection probabilities in this study, but MacCracken et al. (2009) reported detection probabilities $\geq$0.9 for 3 species of torrent salamander, 2 species of giant salamander, and coastal tailed frogs. Nonetheless, our density estimates are biased low to an unknown extent.

Density estimates had highly skewed distributions (Shapiro–Wilks test, $P < 0.05$) and transformations ($\log_e$, square-root, rank) were unsuccessful in normalizing the data. We analyzed the amphibian density data with the effect size and confidence interval approach outlined by Di Stefano (2004). We used this approach to test for differences in density among the stream categories and to assess the efficacy of stream-side buffers. In this approach, the mean raw effect size and the 95% confidence interval is calculated for each stream category contrast (e.g., the difference between unbuffered and second-growth streams). Effect size estimates were also skewed, so we used non-parametric bootstrap re-sampling (2000 iterations, with a re-sampling $n = 12$) to estimate the mean and 95% confidence interval (CI) for each contrast.

We predicted that amphibian densities would be lowest in unbuffered streams; followed by buffered streams and second-growth – which should be about the same; and greatest in unharvested streams. We assessed that prediction by comparing the mean effect size and confidence intervals for each stream category contrast. If the mean and CI did not include zero, then the contrast was statistically different ($P < 0.05$).

To assess buffer efficacy, we compared the effect size and CI of contrasts between unbuffered streams and second-growth, and buffered streams and second-growth to an ecologically important effect size (EIES) (Di Stefano, 2004). We defined the EIES for each species as the difference in density between streams in clearcuts without buffers and streams in unharvested stands. This EIES was assumed to represent a worst case scenario and served as the benchmark in assessing the stream category contrasts that addressed buffer efficacy. Interpretation of those contrasts were as follows: (1) if the mean and CI do not overlap zero or the EIES, then the relationship is statistically significant and ecologically important; (2) if the mean and CI overlap zero, but not the EIES, then the relationship is not statistically significant, but is ecologically important; and (3) if the mean and CI overlap both zero and the EIES, then the relationship is not statistically significant and the data are insufficient to determine ecological importance (Di Stefano, 2004). Interpretation of all other possible effect size scenarios follows the logic presented above (Di Stefano, 2004). The analyses above were conducted with SYSTAT v. 12.0.

Classification trees (Breiman et al., 1984; De'ath and Fabricius, 2000) were used to examine relationships between stream occupancy by amphibians and water temperature regimes. S-Plus 2000 for Windows was used to generate classification trees. Node deviance was set at 0.01, 10-fold cross-validation was repeated 50 times for each tree, and the mean number of splits was used for the final pruning of each tree. The pseudo R^2 values are reported for each tree, which was calculated as $((D_o - D_r)/D_o)$ where D_o is the original deviance and D_r is the residual deviance.

3. Results

Cascade torrent salamanders ($n = 303$) were found in 22 of 32 (69%) streams sampled. Cascade torrent salamander density was 2–7-fold lower ($P < 0.05$) in managed streams than unharvested, and 3-fold lower ($P < 0.05$) in streams without buffers than in streams with buffers and in second-growth, which differed by only 2% (Table 1). Buffering streams had a statistically non-significant ($P > 0.05$), but ecologically important (EIES; Di Stefano, 2004) positive effect on torrent salamander densities (Fig. 1).

Giant salamanders ($n = 223$) were observed in 16 of the streams sampled (50%). The abundance of giant salamanders varied by 16–25% among managed streams and was greatest in unbuffered and second-growth streams (Table 1). However, giant salamander abundance was 150% lower ($P < 0.05$) in unharvested streams compared to the mean of the managed streams (Table 1). Buffers had a negative effect on giant salamander density ($P > 0.05$), that was ecologically important (Fig. 1).

Tailed frogs ($n = 36$) were found in 10 of the streams (31%). Fifty-eight percent of buffered streams, 48% of streams in second-growth, and no unbuffered streams were occupied. Density of tailed frogs was 125–165% lower ($P < 0.05$) in managed than unharvested streams and none were found in unbuffered streams (Table 1). However, densities differed by only 50% between buffered and second-growth streams. Buffers had a significant ($P < 0.05$), and ecologically important, positive effect on tailed frog abundance (Fig. 1).

Water temperature trees for tailed frogs and giant salamanders were not supported through cross-validation. However, the classification tree for Cascade torrent salamanders segregated streams into 2 groups based on the number of consecutive hours that the

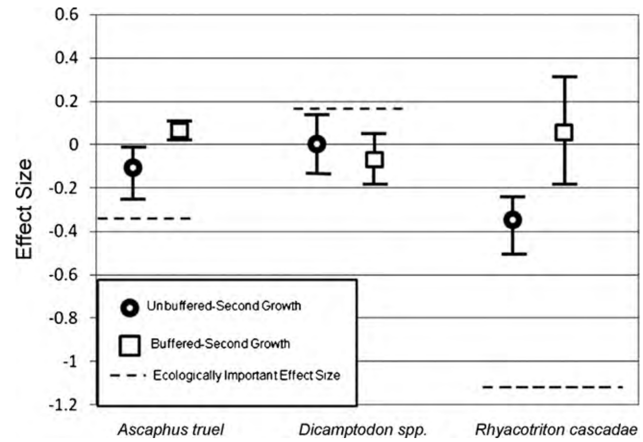

Fig. 1. Mean raw effect size (symbol), and 95% confidence interval (bars) for contrasts in the abundance tailed frogs (*Ascaphus truei*), giant salamanders (*Dicamptodon* spp.), and Cascades torrent salamanders (*Rhyacotriton cascadae*) between 3 stream categories (in clearcuts, unbuffered; in clearcuts, buffered; and in 35+ year old second-growth forests) in the Cascade Range of southern Washington. Dashed horizontal lines are ecologically important effect sizes (EIES, unbuffered streams−streams in unharvested forest). If the mean and CI do not include zero or the EIES, then the effect is both statistically significant and ecologically important, if the mean and CI include zero and the EIES, then the effect is not statistically significant nor ecologically important, and if the mean and CI include zero, but not the EIES, then the effect is not statistically significant, but ecologically important. The interpretation of all other possible relationships follows the same logic.

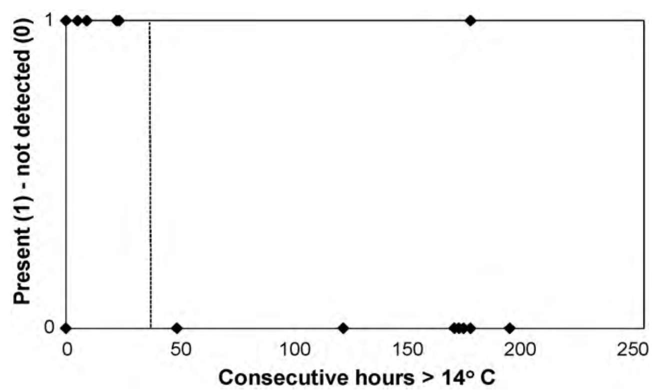

Fig. 2. Occurrence of Cascade torrent salamanders (*Rhyacotriton cascadae*) (1 = present; 0 = not detected) in relation to the number of consecutive hours where the water temperature of a stream exceeded 14 °C. The vertical dashed line indicates the value (35 h) of the primary split in the classification tree.

stream temperature was >14 °C (pseudo $R^2 = 0.89$). Cascade torrent salamanders were present in 17 of 19 (90%) of the streams with <35 consecutive hours ≥14 °C, but in only 1 of 8 (13%) streams with >35 consecutive hours of temperatures ≥14 °C (Fig. 2). In addition, 5 buffered streams exceeded the 14 °C thresholds for an average of 111 h (95% CI = 26–195), 5 unbuffered streams for 98 h (26–170), and 3 second-growth streams for 82 h (14–178).

4. Discussion

The efficacy of riparian buffers in terms of maintaining headwater stream amphibians in clearcut forests is supported by our study. Cascade torrent salamander abundance was about 3-fold lower in streams in clearcuts without buffers than in buffered and second-growth streams. Tailed frogs appear to be patchily distributed, but locally abundant in this area (J.G. MacCracken, personal observation). The absence of tailed frogs in the unbuffered streams suggests that they were either extirpated from logged sites or were not present in those streams before logging. However, general accounts

Table 1
Mean (95% confidence interval) density (individuals/m^2) of stream amphibians in 4 stream categories (buffered in clearcuts, unbuffered in clearcuts, in unharvested second-growth [>35 years old], and in unharvested [>150 years old] forests) in the Cascade Range of southern Washington in 2001.

Species	Stream category			
	Buffered	Unbuffered	Second-growth	Unharvested
Tailed frog (*Ascaphus truei*)	0.39(−0.1, 0.7)	0	0.2(0.0, 0.5)	0.7(−0.1, 1.4)
Giant salamanders (*Dicamptodon* spp.)	0.2(0.0, 0.5)	0.3(0.0, 0.6)	0.3(−0.1, 0.7)	0.2(−0.1, 0.5)
Cascade torrent salamander (*Rhyacotriton cascadae*)	0.5(−0.1, 1.1)	0.2(0.0, 0.4)	0.6(0.1, 1.0)	1.5(0.3, 2.6)

of tailed frog occurrence (Nussbaum et al., 1983) and more recent studies (Dupuis and Friele, 2006; Hayes et al., 2007; Kroll et al., in press) indicate that basin size, stream length, and flow are major determinants of tailed frog abundance and occupancy. Our sampling design that targeted stream reaches near the origin was likely biased against larval tailed frogs, thus our findings for this species have limited inference. In addition, giant salamander abundances differed by only 50% among the managed stream categories. Our density estimates for Cascades torrent salamanders and giant salamanders are consistent with the findings of Steele et al. (2002, 2003).

The differences in the density of Cascade torrent salamanders and tailed frogs among the stream categories in the managed forests and the lack of CI overlap with the EIES indicates that buffering streams mitigated the short-term effects of clearcut logging. In contrast, the giant salamander data suggests that buffering streams had an ecologically important, but negative effect on those species.

Specification of an EIES can be difficult and arbitrary, but is necessary to place management effects in context or to set management goals (Di Stefano, 2004). In most cases, data are not available to empirically define an EIES and expert opinion may have to suffice. Our use of the contrast between streams in clearcuts without buffers and unharvested forest is not without problems, but characteristics of unharvested forests are often invoked as a benchmark reference condition. Information on species vital rates (e.g., survival, reproduction, emigration, immigration) could be used to derive a more realistic EIES, but this type of data for these species is largely unavailable (Kroll, 2009).

Since the inception of this study, 4 studies have been published that address the effects of buffers on torrent salamanders, tailed frogs, or giant salamanders, and most authors concluded that buffers mitigate the effects of logging to varying degrees (Dupuis and Steventon, 1999; Stoddard and Hayes, 2005; Johnston and Frid, 2002). Stoddard and Hayes (2005) found that the presence of a 46 m wide buffer was a good predictor of the occurrence of tailed frogs, and southern and Columbia torrent salamanders (*Rhyacotriton variegatus* and *Rhyacotriton kezeri*, respectively). In contrast to our study, Stoddard and Hayes also reported a positive relationship between Pacific giant salamanders (=coastal giant salamander), and the presence of stream buffers. Johnston and Frid (2002) tracked the movements of Pacific giant salamanders and found that they stayed closer to the stream in clearcut sites without buffers, compared to buffered streams. Tailed frogs occurred in higher densities in streams with old growth buffers (5–60 m wide) relative to clearcuts (Dupuis and Steventon, 1999). In contrast, a study on the Olympic Peninsula, Washington, that included 7 buffered streams, found fewer amphibians in buffered (14–32 m wide) streams than clearcut or second-growth streams (Raphael et al., 2002). These mixed results emphasize the conclusion of Kroll (2009) that both abiotic and biotic factors interact with management actions in complex ways to influence the abundance of stream amphibians in the Pacific Northwest.

A number of studies have suggested that water temperature is an important factor affecting the distribution and abundance of lotic amphibians in the Pacific Northwest (Bury and Corn, 1988; Hawkins et al., 1988; Welsh and Lind, 1996; Wahbe and Bunnell, 2003; Steele et al., 2003; Olsen et al., 2007). However, conclusions about preferred and threshold temperatures are based on data of very limited temporal and spatial scope and are usually obtained by placing a thermometer in the stream at the time and point of capture (Diller and Wallace, 1996; Dupuis and Steventon, 1999; Steele et al., 2003; Russell et al., 2005). These "spot" measurements have limited inference due to daily and seasonal fluctuations. Temperature data loggers programmed to record data at relatively short intervals (e.g., every hour as in this study) provide a time series that can be used to define potential thresholds, based not only on specific temperatures, but also duration. We found that streams that had >35 consecutive hours of temperatures $\geq$14 °C had few to no Cascade torrent salamanders, and that stream management category did not influence water temperature regimes. Steele et al. (2003) also found no relationship between stream water temperature and forest age and concluded that headwater temperature regimes were largely governed by groundwater inputs in this area. Optimal temperature regimes for Cascade torrent salamanders are unknown, but the thresholds we found may represent the point at which physiological and behavioral adaptations preclude other important activities, such as foraging and reproduction. However, persistence at higher temperatures for longer periods is possible, given the single individual we found in a stream that exceeded the defined regime, and the fact that the lethal temperature for torrent salamanders in laboratories studies ranged from about 23–28 °C (Brattstrom, 1963; Bury, 2008).

5. Management implications

The results of this study suggest that buffering headwater streams reduces the impacts of clearcut logging on Cascade torrent salamanders, and larval tailed frogs. Giant salamanders appear to be able to tolerate, and possibly benefit from changes to stream environments due to clearcut logging. Issues that remain to be investigated include buffer width and longitudinal extent. Buffers sampled in this study ranged from 20 to 46 m wide and amphibian abundance was weakly related ($r = -0.40$) to buffer width. In addition, those buffers were continuous and current Washington regulations prescribe a patch buffer system that covers $\geq$50% of the length of headwater streams. The efficacy of the patch buffer system needs examination.

Previous studies have associated torrent salamanders with low water temperature, but we are unaware of any studies that have estimated potential exposure thresholds for this species. This finding could be useful in establishing temperature limits for management of headwater streams, but needs further confirmation. Temperature thresholds are likely 'species specific' and may differ among watersheds, landforms, and stream characteristics (Dunham et al., 2007).

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