

Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington – Phase 3 (Fifteen Years after Harvest)

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Executive Summary

To be written after the rest of the report has been developed.

Introduction

Washington State enacted the Forests and Fish Law in July 2001 (WFPB, 2001). This was largely motivated by the listing, and potential further listings, of salmonid populations in Washington State as either endangered or threatened under the federal Endangered Species Act (ESA; US Fish and Wildlife Service, USFWS 1999), and the identification of hundreds of stream segments with water quality problems under the §303(d) of the federal Clean Water Act (CWA). The Forests and Fish Law, negotiated among federal, state, tribal and county governments, and private forest landowners, was intended to improve and protect riparian habitat on non-federal forestlands in Washington State (hereafter, Forest Practices rules; USFWS 1999). Forest Practices rules were designed to develop biologically sound and economically practical solutions to meet four focal Performance Goals: (1) provide compliance with the ESA for aquatic and riparian-dependent species; (2) restore and maintain riparian habitat to support a harvestable supply of fish; (3) meet the requirements of the CWA for water quality, and; (4) keep the timber industry economically viable in the state of Washington.

At the time of Forest Practices negotiations few studies had addressed the efficacy of riparian buffers along non-fish-bearing, perennial “headwater” streams (or Type Np Waters). However, these small streams comprise more than 65% of the total stream length on forestlands in western Washington (Rogers & Cooke, 2007). Furthermore, existing studies tended to be retrospective (e.g., Bisson et al., 2002; Raphael et al., 2002) or lack the statistical power needed to fully inform Forest Practices effects on aquatic resources of interest (e.g., Jackson et al., 2001; O'Connell et al., 2000). The objective of the Type N Experimental Buffer Treatment Study in Hard Rock Lithologies (hereafter, Hard Rock Study) was to evaluate the effectiveness of the current westside riparian management zone (RMZ) rules for Type Np Waters in maintaining key aquatic conditions and processes affected by Forest Practices. This study was intended to address the key question (WADNR, 2006, FPHCP, Appendix N):

Will the rules produce forest conditions and processes that achieve Resource Objectives as measured by the Performance Targets, while taking into account the natural spatial and temporal variability inherent in forest ecosystems? ¹

In the Hard Rock Study, we compared unharvested references to the current Forest Practices buffer prescription (FP treatment) and to experimental treatments that did not retain a riparian buffer in the RMZ (0% treatment) and that retained a riparian buffer throughout the entire RMZ (100% treatment). We provided information relevant to evaluating whether these riparian buffer prescriptions met the Performance Goals to provide compliance with the ESA for aquatic and riparian-dependent species and met the requirements of the CWA for water quality. We also evaluated whether buffer prescriptions met the Resource Objectives (i.e., key aquatic conditions and processes affected by Forest Practices) for large wood inputs, organic inputs, and hydrology from the Forest Practices Habitat Conservation Plan (FPHCP; WADNR, 2006, Appendix N). In

¹ Each Resource Objective consists of (1) a Functional Objective, or broad statement of objectives for the major watershed functions potentially affected by Forest Practices, and (2) a series of Performance Targets, or measurable criteria defining specific, attainable target forest conditions and processes.

addition, we provided data and the analyses needed by the Washington State Department of Ecology to help determine compliance with water quality standards. The study commenced in 2006 and included up to three years of pre-harvest data collection depending on the response variable. Treatments were implemented over a period of 14 months in 2008 and 2009. Post-harvest data were collected for up to 15 years following harvest. Post-harvest sampling frequency and duration depended on the response variable. Results for Phase I of the study, comparing the response among treatments up to three years following harvest, were reported in McIntyre et al. (2018). Results for the Phase II of the study, comparing the response among treatments up to 11 years following harvest, were reported in McIntyre et al. (2021).

Though the original study supported only two years of post-harvest sampling, significant responses to harvest for some variables (e.g., stream temperature) led the Forest Practices Board to support continued post-harvest monitoring beyond those two years in a Phase II effort. Continued monitoring in Phase II allowed us to evaluate trajectories of response variables that changed immediately after harvest, such as for stream temperature, and to detect potential lag effects for those for which a significant response was not detected in the two years following harvest (e.g., stream-associated amphibians).

Results from Phase II suggested a delayed decline in larval Coastal Tailed Frog densities (-65% to -93%) 7- and 8-years post-harvest in the 100%, FP and 0% treatments that were not apparent in the two years post-harvest (i.e., Phase I). There was also a delayed negative response for torrent salamanders estimated in the FP treatment (-65%). In response to the Phase II amphibian results, the Adaptive Management Program supported continued monitoring to evaluate trends in stream-associated amphibian densities through 15-years post-harvest. This Phase III effort will inform whether amphibian densities at study sites stabilized, continued to decline, or recovered over time.

Pacific Northwest headwater streams support stream-associated amphibian abundances that are greater than in larger, fish-bearing river systems (see Richardson & Danehy, 2007). Fish densities decline in smaller streams, offering amphibians a refuge from fish predators common in higher-order streams (Richardson & Danehy, 2007). In fact, stream-associated amphibians often replace fish as the dominant vertebrate predators in and along headwater streams (Burton & Likens, 1975; Bury et al., 1991). In headwaters of the Pacific Northwest, aquatic amphibians are estimated to be ten times more abundant than salmonid fishes (Bury et al., 1991).

Stream-associated amphibian species may be uniquely adapted to the physical conditions of headwater streams (Kiffney et al., 2003). Some of the specific headwater habitat attributes important to amphibians, such as substrate composition (Dupuis & Steventon, 1999; Grialou et al., 2000; Stoddard & Hayes, 2005) and water temperature (Bury, 2008; Pollett et al., 2010), are affected by timber harvest and associated activities (Araujo et al., 2013; Grizzel & Wolff, 1998; Jackson et al., 2001; Janisch et al., 2012; Johnson & Jones, 2000; Moore et al., 2005). Stream-associated amphibians may be particularly predisposed to large variations in population size or local extirpation because of disturbance, including timber harvest (Bury & Corn, 1988; Fagan, 2002). Once extirpated, opportunities for recolonization from adjacent headwater streams may be restricted by larger downstream reaches (Lowe & Bolger, 2002; Richardson & Danehy, 2007) or gaps in overhead canopy (Cecala et al., 2014) that form barriers to dispersal.

Amphibian populations have experienced declines in local abundances and range contractions because of disease, competition with introduced species, and habitat degradation and conversion (Sparling et al., 2001; Stuart et al., 2004). Some studies have concluded that stream-associated amphibians are sensitive to forestry practices. For example, Corn and Bury (1989) found that Coastal Tailed Frogs occurred with higher frequency in unlogged watersheds. Steele et al. (2003) reported reduced numbers of Cascade Torrent Salamander (*Rhyacotriton cascadae*) in young forests (i.e., recent clearcuts to 24-year old) compared with mature forests (i.e., 25 to 60 years old). Jackson et al. (2007) found that giant salamander and Coastal Tailed Frog populations declined in the several years immediately following timber harvest. Olson and Ares (2022) found reduced densities of giant and torrent salamanders five years after a second forest thinning. Conversely, others have not detected a correlation between amphibian abundance and forestry activities, including for Coastal Giant Salamander and Coastal Tailed Frog (Murphy & Hall, 1981; O'Connell et al., 2000).

Amphibians are often considered among the vertebrate groups most susceptible to environmental modification and, because of their limited dispersal abilities, dual life histories, and explicit microhabitat and physiological requirements (Lawler et al., 2010; Welsh & Ollivier, 1998), are frequently preferred for monitoring environmental conditions (Wake, 1991). One of three Overall Performance Goals for the Forest Practices Habitat Conservation Plan (FPHCP) is to support the long-term viability of designated stream-associated amphibians, including Coastal Tailed Frog (*Ascaphus truei*); and Olympic (*Rhyacotriton olympicus*), Columbia (*R. kezeri*) and Cascade (*R. cascadae*) Torrent Salamanders (hereafter, FP-designated amphibians; Schedule L-1). One Resource Objective is to provide conditions that sustain FP-designated amphibian population viability within occupied sub-basins.

Though Coastal and Cope's Giant Salamanders (*Dicamptodon tenebrosus* and *D. copei*, respectively) are not FP-designated amphibians, we included them in our study because Cope's Giant Salamander is one of only two instream-breeding amphibian species distributed throughout our entire study area and, for this reason, was included in the amphibian genetics component of the study (Spear et al., 2011; Spear et al., 2019). Since Cope's and Coastal Giant Salamanders are extremely difficult to differentiate in the field (Foster & Olson, 2014; Good, 1989; Nussbaum, 1970, 1976) we included both species. Like the other FP-designated amphibians, changes in giant salamander populations reflect changes in the environment.

There is substantial uncertainty regarding the effectiveness of the FPHCP buffer strategy for Type Np streams relative to the long-term viability of stream-associated amphibians. To address these uncertainties, we used a basin-scale approach to compare changes in stream-associated amphibian densities in response to clearcut timber harvest with alternative riparian buffer treatments including no buffering (0% treatment), partial buffering using the FPHCP prescription (FP treatment) and complete buffering (100% treatment). Though Phase I and Phase II efforts included genetic and stable isotope monitoring to complement our demographic monitoring, the current Phase III effort included only an evaluation of stream-associated amphibian demographic response to treatment.

Study Sites

Site Selection

The inclusion of stream-associated amphibian species as a response variable placed important constraints on site selection for the Hard Rock Study (**Table 1**). Six of the seven Forest Practices (FP)-designated amphibians occur exclusively ($n = 5$) or largely ($n = 1$) in westside forestlands of Washington State. We selected sites in western Washington that supported Coastal Tailed Frog (*Ascaphus truei*) and Olympic, Columbia, and Cascade Torrent Salamanders (*Rhyacotriton olympicus*, *R. kezeri*, and *R. cascadae*).² Although Coastal (*Dicamptodon tenebrosus*) and Cope's (*D. copei*) Giant Salamanders are not FP-designated amphibians, we included them in the study because they co-occur with FP-designated species throughout the study area and Cope's Giant Salamander, along with the Coastal Tailed Frog, was appropriate for evaluating amphibian genetic responses (Spear et al., 2019). The site selection process is outlined in detail in McIntyre et al. (2009).

We limited site selection to the three westside physiographic regions with the greatest number of FP-designated amphibians (Olympic Mountains, Willapa Hills and Southern Cascades south of the Cowlitz River; Jones et al., 2005). We limited sites to those less than 1,067 m (3,500 ft) and 1,219 m (4,000 ft) elevation in the Olympic and South Cascade physiographic regions, respectively, because FP-designated amphibians rarely occur above 1,219 m (4,000 ft) elevation in Washington State and the upper elevation limit declines with increasing latitude (Dvornich et al., 1997). We did not impose an upper elevation limit in the Willapa Hills because the maximum elevation (Boisfort Peak: 948 m [3,110 ft]) is within the range of all amphibian species. We limited sites to those with a slope between 5% and 50% (3 and 27 degrees) to encompass the range of stream gradients within which FP-designated amphibians are typically found (Adams & Bury, 2002). We included only sites composed of competent lithology, or those that could potentially be competent depending on weathering and age (as identified by Patrick Pringle, formerly with WADNR), because some FP-designated amphibians tend to occur more frequently on these types of lithology (Dupuis et al., 2000; Wilkins & Peterson, 2000). Finally, since Coastal Tailed Frogs rarely reproduce in small first-order basins in western Washington (Hayes et al., 2006), we restricted site selection to include second-order streams (Strahler, 1952); however, we later relaxed the stream order criteria to include first- to third-order streams to obtain the desired number of study sites.

To maximize the influence of the buffer treatments and to reduce confounding effects we designed the study so that harvest units would encompass the entire Type N basin when possible. We also wanted harvest unit size to represent operational forest practices (McIntyre et al., 2009). Interviews with landowners revealed that the typical minimum unit size was about 12 ha (30 ac); maximum harvest unit size is limited by Forest Practices to 49 ha (120 ac; WFPB, 2001). Thus,

² The remaining three Forest Practices-designated amphibians not covered in our study include the Rocky Mountain Tailed Frog (*A. montanus*), and Dunn's (*Plethodon dunnii*) and Van Dyke's (*P. vandykei*) Salamanders. Rocky Mountain Tailed Frog could not be included because it occurs exclusively in southeastern Washington, an area not included in our study. The two plethodons were not included because they breed and lay eggs on land, and have no free-living (i.e., aquatic) larval stage. Thus, they require different sampling techniques than the focal species in this study.

sites were limited to basins within that range.³ Subsequently, we relaxed the criterion to include basins up to 54 ha (133 ac) to obtain the desired number of study sites. To ensure that downstream fish response⁴ was not confounded by other management activities, we required at least 75 m (246 ft) of stream below the upstream extent of fish distribution (F/N break) that lacked an incoming tributary.

Inclusion of study sites relied on commitments that landowners manage them according to treatment specifications (i.e., harvest layout and timing). We requested that landowners commit to completing timber harvest and associated buffer treatments between April 2008 through March 2009. We limited sites to those with at least 70% of the basin area with stands between 30 and 80 years of age at the time of harvest, because the average minimum stand age at harvest is 30 years and harvest of stands over 80 years is infrequent in Washington State. Finally, because multiple ownership of the same study site would greatly complicate the coordination and implementation of treatments, we limited sites to those for which more than 80% of the Type N basin had a single landowner.

Selection of study sites began in June 2004 and continued through August 2006. We used a Geographic Information System (GIS) in ArcMap (ESRI, 2004) to identify Type Np basins meeting geographic range, elevation, stream gradient, lithology and stream order site selection criteria (see McIntyre et al. 2009). We conducted on-site surveys to validate lithology type, stream gradient and stand age. For those meeting site selection criteria, we conducted surveys to establish amphibian occupancy. On-site electrofishing surveys were conducted between December 2005 and June 2006 to verify the location of the F/N break (WFPB, 2002). Field surveys revealed inaccuracies in the hydrology layer used to determine stream order, so we relaxed our criteria to include a few first- and third-order sites for which we had already determined FP-designated amphibian presence.

Table 1. Site selection criteria and associated limits by category for the Hard Rock Study, 2004–2006.

Category	Criterion	Limit
FP-designated amphibian presence	Geographic range	Olympic Mountains, Willapa Hills, and South Cascade south of the Cowlitz River physiographic regions of Washington State
	Elevation	<1,067 m (3,500 ft) for the Olympic region <1,219 m (4,000 ft) for the South Cascade region No limit for the Willapa Hills region
	Stream gradient	5–50% (3–27 degrees)

³ Unless an exception is granted after review by an interdisciplinary science team.

⁴ Downstream fish response was only included through the two years following harvest and included only 6 of the 17 study sites. Results are reported in McIntyre et al. 2018.

Category	Criterion	Limit
	Lithology	Competent (or any lithology that could potentially be competent, i.e., potentially producing long-lasting large clasts or coarse grain sizes)
	Stream order	Second-order stream basins
Fish presence	Stream network	Minimum of 75 m (246 ft) of stream between the F/N break and nearest downstream tributary intersection
Landowner/operational considerations	Type N basin size	12–49 ha (30–120 ac)
	Stand age	30–80 years old
	Harvest timing	Buffer treatments: harvest Apr 2008–Mar 2009; References: no harvest
	Area owned	>80% owned by single landowner

Experimental Treatments

We established four treatments: three buffer treatments with clearcut harvest and riparian buffers of variable length, and a reference (i.e., control) with no timber harvest (**Figure 1**):

- 1) **Reference** (REF: unharvested reference with no timber harvest activities within the entire study site during the study period,
- 2) **100% treatment** (100%): clearcut harvest with a no-harvest riparian leave-tree buffer (i.e., two-sided 50-ft [15.2-m]) throughout the RMZ,
- 3) **Forest Practices treatment** (FP: clearcut harvest with current Forest Practices no-harvest riparian leave-tree buffer (i.e., two-sided 50-ft [15.2-m]) along $\geq 50\%$ of the RMZ, and
- 4) **0% treatment** (0%): clearcut harvest with no riparian leave-tree buffer retained within the RMZ.

Clearcut harvest was applied throughout the Type Np basin in sites with a riparian buffer treatment and, except for the length of the riparian buffer in the RMZ, harvest followed Forest Practices rules. Buffer width of 50 ft (15.2 m) is the horizontal distance from the bankfull channel. In all treatments, a 30-ft (9.1-m) equipment limitation zone (ELZ) was maintained along all Type Np and Ns (i.e., seasonal) Waters (WAC 222-30-021(2)), and no harvest activities were conducted on any potentially unstable slopes (WAC 222-16-050 (1)(d)). In the 100% and FP treatment sites, RMZ buffers were required for the five categories of sensitive sites WAC 222-

16-010): side-slope⁵ and headwall⁶ seeps, headwater springs⁷, Type Np intersections⁸ and alluvial fans⁹. Riparian buffers on headwall and side-slope seeps require a 50-ft (15.2-m) no-harvest buffer around the outer perimeter of the perennially saturated area. Riparian buffers on Type Np intersections and headwater springs require a 56-ft (17.1-m) radius no-harvest buffer centered on the feature. No harvest is allowed within alluvial fans.

We identified all Type Np and Ns Waters and the locations of all sensitive sites according to Forests and Fish rules. All features were mapped in the field using Trimble Global Positioning Systems (GPS), which were differentially corrected using Pathfinder Office software and integrated into GIS (ArcMap).

The buffered length of the streams in FP treatment sites was determined by FP rules, which require a two-sided, 50-ft (15-m) wide buffer along a minimum of 50% of the length of the Type Np stream. Non-fish-bearing streams <1,000 ft (305 m) and ≥1,000 ft require a minimum of 300-ft (91-m) and 500-ft (152-m) length riparian buffer, respectively, located directly upstream of the F/N break, with additional riparian buffers centered on sensitive sites. All study sites were ≥1,000 ft (305 m), requiring a minimum 500-ft (152-m) length buffer. The configuration of the riparian buffer on a Type Np Water is subject to stream dendritic patterns and the number and location of sensitive sites. To determine the configuration at our sites, we located sensitive sites in the field 12 June to 1 November 2006. The application of FP rules at the three FP treatment sites resulted in riparian buffer lengths of 55%, 62%, 73% and 97%. In addition, due to regulatory and/or logistic constraints (e.g., buffers required on unstable slopes and downstream fish-bearing waters), 2 to 15% of the basin area was not harvested in four riparian buffer treatment sites (specifically, OLYM-100%, WIL-100%-1, WIL-0%-2, and CASC-0%).

Site Identification and Blocking

Though 35,957 Type Np basins were identified within our geographic scope of interest (Olympic Mountains, Willapa Hills and Southern Cascades physiographic regions), only 17 basins remained for inclusion in our study after selection criteria were applied and landowner and timber harvest constraints were considered. Sites consisted of first-, second- and third-order Type Np stream basins located in managed second-growth forests on private, state, and federal forestlands across western Washington. Stands were 30 to 80 years old and dominated by Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*). Sites were in areas dominated by competent lithology types (largely basaltic) with average Type Np channel gradients ranging from 14 to 34% and catchment areas ranging from 12 to 54 ha (30 to 133 ac). Cumulative stream lengths ranged from 325 to 2,737 m (1,066 to 8,980 ft; **Table 2**). Sites were located along tributaries of the Clearwater, Humptulips and Wishkah Rivers in the Olympic physiographic region (n = 4); the North, Willapa, Nemah, Grays, and Skamokawa Rivers, and

⁵ A seep with perennial water at or near the surface throughout the year, located within 100 ft (30.5 m) of a Type Np Water, on side-slopes greater than 20%, connected to the stream channel via overland flow, and characterized by loose substrate and fractured bedrock, excluding muck.

⁶ A seep with perennial water at or near the surface throughout the year, located at the toe of a cliff or other steep topographical feature at the head of a Type Np Water, connected to the stream channel via overland flow and characterized by loose substrate and/or fractured bedrock.

⁷ A permanent spring at the head of a perennial channel and coinciding at the uppermost extent of perennial flow.

⁸ The intersection of two or more Type Np Waters.

⁹ An erosional landform consisting of a cone-shaped deposit of water-borne, often coarse-sized, sediments.

Smith Creek in the Willapa Hills physiographic region (n = 10); and the Washougal River and Trout Creek in the South Cascade physiographic region (n = 3; **Figure 2**).

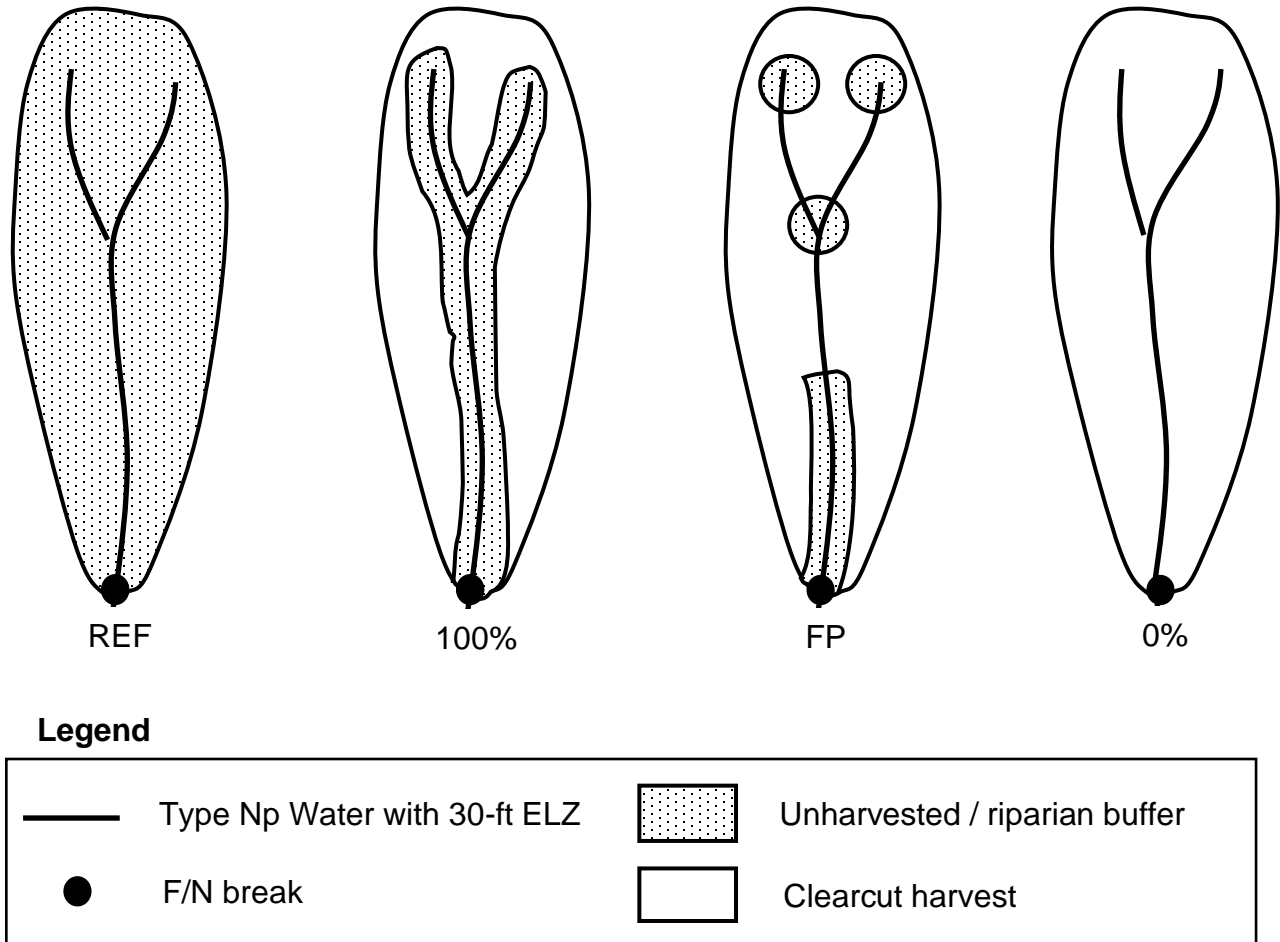


Figure 1. Schematic of the four experimental treatments included in the Hard Rock Study. Treatments included unharvested references (REF) and sites receiving a clearcut harvest with one of three, two-sided 50-ft (15.2 m) buffer treatments along the Type Np Water riparian management zone (RMZ): 100% of the stream length buffered (100%), $\geq 50\%$ of the stream length buffered (Forest Practice, FP), and no buffer (0%). FP and 100% treatments include 56-ft (17.1-m) radius buffers around Type Np intersections and the uppermost extent of perennial flow. All streams are protected by a two-sided 30-ft (9.1-m) equipment limitation zone (ELZ).

We blocked (grouped) study sites geographically within each physiographic region (i.e., Olympic, Willapa Hills, and South Cascade) to minimize variability (e.g., regional differences) and assigned sites within each block to one of the four treatments. In the Phase I and Phase II analyses, we included five blocks: one block in the Olympics, three blocks in the Willapa Hills, and one block in the South Cascades. Under the original study design, the intent was to have each of the four treatments (i.e., three buffer treatments and unharvested reference) represented in each block. However, in application this did not work as designed (see McIntyre et al. 2018

for details). Nonetheless, we originally randomly assigned treatments based on the premise of one treatment per each of the five proposed study blocks. In practice, we ended up with some incomplete blocks and some blocks with more than one treatment type represented. As such, in the current analysis, we simplified the blocking to be based on physiographic region only to control for regional variation, including the fact that the three species of torrent salamanders are distributed regionally with distributions that do not overlap between regions (**Table 2**). Study site codes are based on the geographic block and treatment.

While original treatment assignment was random when possible; we were unable to assign some treatments to particular sites. For example, unharvested references were assigned only to public ownership lands because private landowners would not agree to exclude sites from harvest for the duration of the study. Conversely, federal regulations prevented application of buffer treatments on National Forest sites. As a result, only state forestlands (Washington Department of Natural Resources) were available for the full complement of treatments.

Given these constraints, we randomized treatment assignments within blocks to the extent possible, as follows:

Olympic: Treatments were randomly assigned to the four sites in this physiographic region, yielding a single a block with one each of the four treatments (OLYM).

Willapa Hills: Ten sites were available in the Willapa Hills region. Eight were distributed across the coastal region; two were located south and east of these. The original assignment of treatments considered overall study objectives including the evaluation of fish response in the downstream Type F Waters included only in Phase I. As such, we created two blocks, each with four sites, from the coastal region. Of these, only five sites (four on state forestland and one on private land) were suitable for evaluating downstream fish response. To ensure one complete block representative of all treatments was available for downstream fish response evaluation, we assigned treatments to sites as follows: First, the site on private land was assigned a buffer treatment. Of the four state-owned sites in the coastal Willapas, two were randomly chosen as unharvested reference sites and randomly assigned to one of the two coastal Willapa Blocks. The remaining two state-owned sites and the private site suitable for evaluating fish response were randomly assigned to the three buffer treatments to complete assignment in the block within which downstream fish response was evaluated.

The remaining coastal state-owned reference site was grouped with the remaining three coastal sites, which were randomly assigned to one of three buffer treatments to form a second block in the coastal Willapa Hills. Due to unfavorable economic conditions, harvest of the FP treatment site in this block was postponed, so it served as a second reference in this block until harvest in January 2016.

For the two remaining sites in the Willapa Hills, located south and east of the eight coastal sites, one was assigned the reference treatment due to biological constraints (presence of marbled murrelet habitat) and the other was assigned the 100% buffer treatment due to slope instability. In Phase III, all Willapa sites were consolidated into a single block. Two reference sites in the Willapa block were harvested in 2020 and were excluded from the Phase III analysis. **South Cascade:** Three sites were included in the South Cascade (CASC)

block. One was in the Gifford Pinchot National Forest and could only be assigned the reference treatment. We assigned buffer treatments randomly to the two remaining sites, FP and 0%.

For Phases I and II, reference and treatment sites were distributed across federal, state and private timberlands as follows: two references located on national forestlands, three on state lands, and one on private land; three 100% treatment sites on state lands and one on private land; two FP treatment sites on state lands and one on private land; and two 0% treatment sites on state lands and two on private lands (**Table 2**). References located on federal national forestlands may have been subjected to a different management history, including extent and frequency of harvest; however, their inclusion as references still allows us to account for temporal variation of forested stands in western Washington in the absence of active timber harvest. Overall, four references were located on state and private lands actively managed for timber production. For Phase III, one of the Willapa references located on private land was harvested during the winter of 2015 and was included in subsequent analysis as an FP site. Two additional references located on state land were harvested in 2020 and excluded from the Phase III analysis.

Table 2. Treatments, site codes and physical characteristics of study sites used in the Hard Rock Study. Type Np Length is the cumulative length of all perennial, non-fish-bearing tributaries in the study basin. Bankfull Width is the mean of the mainstem channel in the pre-harvest period. * sites not included in Post 14 & 15 treatment response. ^ site included as FP treatment in Phase III report because it was harvested in 2016.

Site Code	Ownership	Basin Area (ha[ac])	Type Np Length (m [ft])	Elevation (m [ft])	Stream Gradient (%)	Lithology	Bankfull Width (m [ft])	Aspect
OLYM-REF	USFS	54 (133)	2,737 (8,980)	163 (535)	18	Basalt flows and flow breccias	2.6 (8.5)	N
WIL-REF-1*	Private	16 (41)	816 (2,677)	228 (748)	18	Basalt flows and flow breccias	1.2 (3.9)	SE
WIL-REF-2*	State	12 (30)	589 (1,932)	200 (656)	19	Basalt flows and flow breccias	1.3 (4.3)	SW
WIL-REF-3	State	37 (92)	2,513 (8,245)	241 (791)	14	Basalt flows	1.7 (5.6)	SW
CASC-REF	USFS	50 (122)	1,080 (3,543)	601 (1,972)	21	Tuffs and tuff breccias	2 (6.6)	N
OLYM-100%	State	28 (68)	1,949 (6,394)	72 (236)	27	Tectonic breccia	2 (6.6)	NE
WIL-100%-1	Private	26 (65)	1,257 (4,124)	22 (72)	21	Basalt flows and flow breccias	1.8 (5.9)	SW
WIL-100%-2	State	31 (76)	1,029 (3,376)	198 (650)	18	Basalt flows and flow breccias	1.9 (6.2)	SW
WIL-100%-3	State	23 (58)	1,359 (4,459)	351 (1,152)	19	Basalt flows	2.1 (6.9)	SE

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OLYM-FP	Private	17 (41)	1,070 (3,510)	277 (909)	25	Basalt flows and flow breccias	1 (3.3)	SE
WIL-FP-1	State	15 (37)	325 (1,066)	197 (646)	19	Basalt flows and flow breccias	1.3 (4.3)	SW
WIL-FP-2^	State	19 (48)	653 (2,142)	183 (600)	34	Basalt flows and flow breccias	1.9 (6.2)	W
OLYM-0%	Private	13 (32)	637 (2,090)	233 (764)	31	Basalt flows and flow breccias	1.6 (5.2)	W
WIL-0%-1	Private	28 (69)	1,525 (5,003)	87 (285)	16	Terraced deposits	1.9 (6.2)	NE
WIL-0%-2	State	17 (42)	933 (3,061)	159 (522)	21	Basalt flows	2.4 (7.9)	E
CASC-0%	State	14 (36)	420 (1,378)	438 (1,437)	29	Andesite flows	1.7 (5.6)	SE

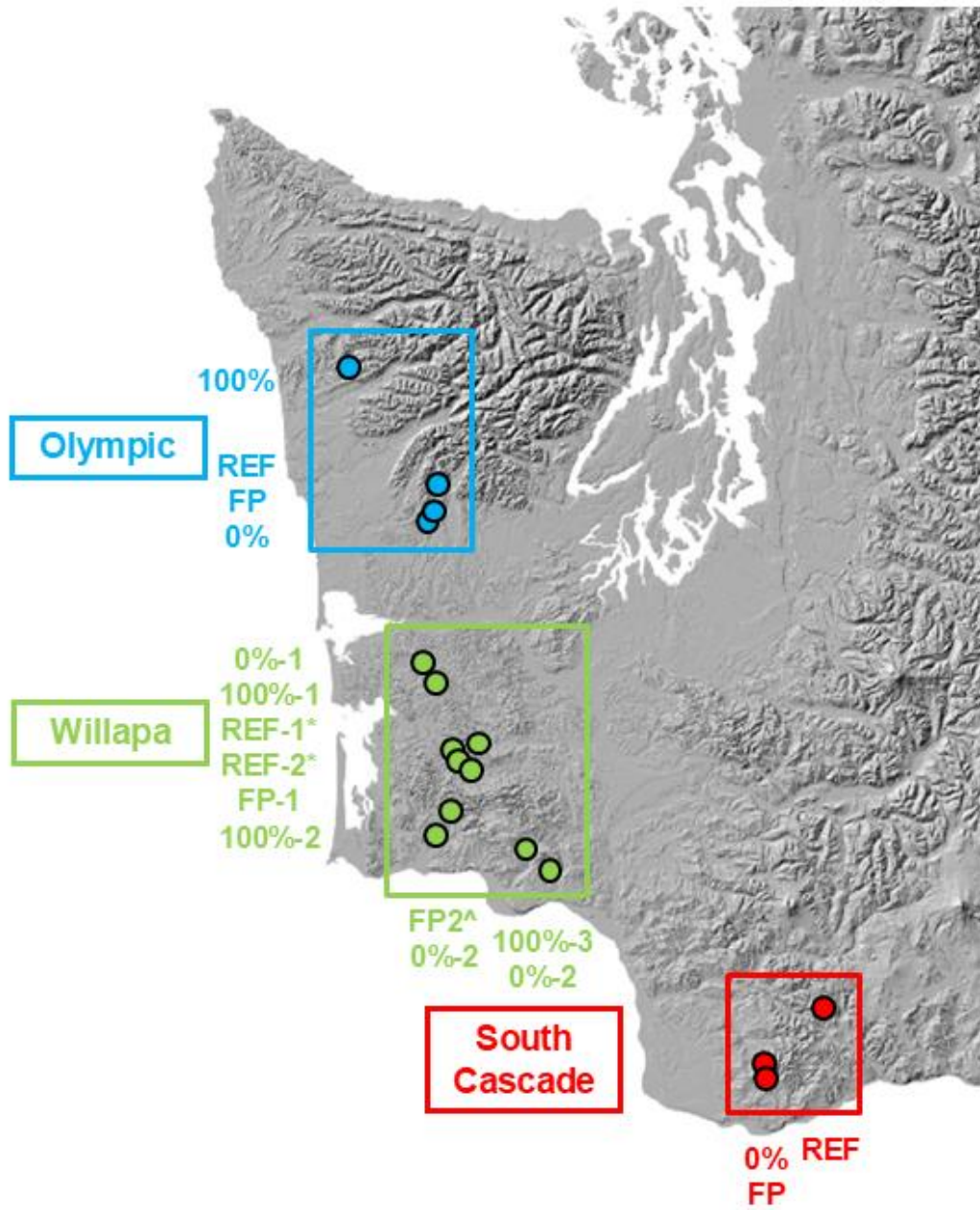


Figure 2. Distribution of study sites and treatments for the Hard Rock Study, 2006–2023. Sites are grouped (blocked) geographically (color coded). REF is the reference treatment (unharvested control) and 100%, FP, and 0% are the 100%, Forest Practices ($\geq 50\%$) and 0% riparian buffer treatments, respectively.

Methods

Study Timeline

Pre-harvest sampling across all study sites began in 2006. Harvest timing and duration varied among study sites. Harvest at the first site to be treated began in July 2008. Harvest was completed at most treated sites by August 2009. (**Table 3**). Two references were harvested in 2020 and were excluded from the Phase III analysis. The WIL-FP-2 site was originally assigned the FP treatment, but harvest was delayed until January 2016, between the Post 7 & 8 sample years. For the Phase II analysis, we included it as a reference and did not include data reflecting the post-harvest state in the statistical analysis. We included it as a fourth FP treatment for Post 14 & 15. In the current analysis we included the data from the pre-harvest period and then included data collected after harvest in 2016 as Post 1, and sampling in 2022 and 2023 as Post 7 & 8. This decision produces a more balanced design (four replicates of the FP treatment).

Table 3. Harvest timeline and periods of analysis. * indicates sites not included in Post 14 & 15 treatment response. ^ indicates WIL-FP-2 was harvested in 2016 and was included as an FP treatment in Phase III.

Site Code	2006	2007	2008	2009	2010	2015	2016	2022	2023		
Period	Pre-harvest Period			Phase I		Phase II		Phase III			
OLYM-REF	Pre 3, Pre 2, Pre 1			Post 1 & Post 2		Post 7 & Post 8		Post 14 & Post 15			
WIL-REF-1*								-			
WIL-REF-2*											
WIL-REF-3										Post 14 & Post 15	
CASC-REF											
OLYM-100%						Post 1 & Post 2		Post 7 & Post 8			
WIL-100%-1											
WIL-100%-2											
WIL-100%-3											
OLYM-FP											
WIL-FP-1											
WIL-FP-2^					Pre-harvest		Post 1	Post 7 & Post 8			
CASC-FP											
OLYM-0%											
WIL-0%-1					Post 1 & Post 2		Post 7 & Post 8		Post 14 & Post 15		
WIL-0%-2											
CASC-0%											

I Unanticipated Disturbance Events

Disturbance is a normal, even integral part of the long-term dynamics of natural and managed forests (Dale et al., 2005). Disturbance processes in Pacific Northwest forests include avalanches, debris-flows, disease, fire, flooding, insects, volcanic activity and wind (Agee, 1993; Fetherston et al., 1995; Franklin et al., 2002). With 17 study sites and data collected over 17 years, it is not surprising that disturbance other than timber harvest impacted some study sites over the course of investigation. Two major disturbances occurred during the study: an extensive windthrow event in December 2007 that affected multiple study sites, and a wildfire in October 2009 that affected two buffer treatment sites in the South Cascade block (see McIntyre et al. 2018, **Chapter 4** – *Unanticipated Disturbance Events*). In response to the December 2007 windthrow event, we collected data in an additional pre-treatment year (summer 2008) so that estimates of amphibian densities reflected the true variation for the full pre-treatment study period, including post-windthrow and pre-harvest. The fire was extinguished with water from fire engines and helicopter bucket drops by 14 October 2009. No bulldozers or fire retardants were used, and the fire had no impact on future management.

Scope of Inference

The temporal scope of inference is the fifteen years post-harvest. The spatial scope of inference is limited to Type Np basins dominated by competent lithologies, which comprise approximately 29% of western Washington FPHCP-covered lands (P. Pringle, personal communication, September 2005, formerly Washington Department of Natural Resources). The spatial scope of the study reflects other constraints as well, including those associated with basin size, stand age, and the presence of stream-associated amphibians (see **Section 2-4**. Site Identification and Blocking). Results should be applied with caution to Type N streams outside the selection criteria. A similar study on sites representing more erodible, soft-rock lithologies is also in progress. In combination, the two studies will allow for broader inferences about FP rule effectiveness.

In FP treatment sites, buffer lengths ranging from 55 to 73% of the non-fish-bearing stream length exceeded the minimum required under Forest Practices rules. This may contribute to greater similarity between the responses in the 100% and FP treatments compared to that in the 0% treatment. This study was designed to evaluate responses to buffer length; however, the same rules that influenced buffer *length* in the FP treatment sites also affected buffer *width* in some 100% treatment sites. Specifically, in some 100% treatment sites, unstable slopes required buffers wider than the 50 ft minimum, which may have reduced effects of harvest (see McIntyre et al. 2018, Chapter 3 – *Management Prescriptions*).

Three aspects of this study create a strong base of inference. First, the geographic scope is large, encompassing multiple sites in western Washington and the southern Cascade Range. Second, the duration of the study exceeds that of most other large-scale studies of forest practices effectiveness in the Pacific Northwest. It includes two to three years of pre-harvest sampling and as many as nine years of post-harvest sampling. In contrast, the current FP prescription for Type Np Waters is based on little research and monitoring. Finally, we use a BACI design, capitalizing on pre- and post-harvest data to distinguish between responses to treatments and other sources of temporal variation.

Amphibian Sampling and Density Estimation

Data were collected at 17 study sites consisting of Type N headwater basins located in competent lithologies (largely basaltic) across western Washington. We evaluated the response of amphibian densities and body condition among reference and treatment sites in a BACI-designed study (see Chapter 2–*Study Design* in this report). We compared amphibian populations in Type Np reference basins (n = 6) to the response in basins with clearcut harvest and one of three riparian buffer treatments in the RMZ: 100% treatment (two-sided riparian buffer along the entire length of the Type Np stream network; n = 4), FP treatment (two-sided riparian buffer along at least 50% of the Type Np stream length, according to current Forest Practices Rules; n = 3), and 0% treatment (clearcut harvest to the stream edge with no riparian buffer; n = 4).

We used two standard amphibian sampling methods: light-touch (conducted at systematically identified locations throughout the entirety of the Type N stream network), and rubble-rouse (restricted to the 200 m stream reach immediately upstream of the F/N break, i.e., the point of last known fish use). We conducted light-touch and rubble-rouse amphibian surveys diurnally between 0700 and 1900 hours during the summer low-flow period, generally July through October.

Light-touch Sampling

Researchers commonly use light-touch methods (Lowe & Bolger, 2002) for headwater amphibians in the Pacific Northwest to establish occupancy or abundance (Quinn et al., 2007; Russell et al., 2004; Steele et al., 2003). A modified light-touch sampling was used to provide count data over an extensive area of the stream network. We conducted stream network-wide light-touch surveys in Pre 3, Pre 2, Pre 1, Post 1, Post 2, Post 7, and Post 8. We actively searched for amphibians as we sampled from down- to upstream, turning all moveable surface substrates small cobble-sized or larger (≥ 64 mm) and within the ordinary high-water mark (WFPB, 2001). We returned substrates to their original position and took care to preserve in-channel structures (e.g., steps). We sampled all study reaches, including those lacking surface water flow, from the F/N break and upstream to each PIP (i.e., uppermost point of perennial flow).

We conducted light-touch sampling along a subset the stream channel network that included the contiguous 200 m (656 ft) of stream immediately upstream of the F/N break, as well as additional reaches located throughout the remainder of the stream channel network. For basins with a cumulative stream length less than 800 m, we surveyed a minimum of 50% of the stream length. For basins with a cumulative stream length greater than 800 m, we surveyed a minimum of 25% of the stream length. Additional reaches were surveyed in 20 m (66 ft) stream segments (i.e., two consecutive 10 m [33 ft] sample reaches, hereafter, sample intervals) distributed throughout the remainder of the mainstem channel (i.e., upstream of the contiguous 200 m sample reach) and spaced 20 m apart for shorter streams and 60 m apart for longer streams. In Pre 1, light-touch sampling was restricted to the 200 m upstream from the F/N break and to the 30-m long plots used for the estimation of detection probability (see **Section 0. Detection Estimation**).

Detection Estimation

Starting in Pre 1, we incorporated a multi-pass light-touch sampling methodology in 30-m long plots (hereafter, detection plots). We sampled these plots in addition to the standard light-touch surveys of sample intervals, though detection plot locations sometimes overlapped with the locations of sample intervals. This approach allowed us to adjust our amphibian light-touch counts for detection probability, accounting for spatial and annual variation in detection in our estimates of stream network-wide amphibian abundance (McIntyre et al., 2012). We chose a 30-m plot length to maximize the likelihood of detecting focal amphibian taxa (Quinn et al., 2007).

We randomly located detection plots and stratified plots by buffer type (buffered, unbuffered, reference) and stream order (first- and second-/third-order; Strahler 1952; **Table 4**). We established new plot locations each year. In some instances, we were not able to sample the entire 30-m plot length (e.g., due to obstructions); however, we required at least 15 m of surveyed length for each plot. We surveyed each detection plot on three separate occasions, concurrent with our stream network-wide light-touch surveys. Our goal was to conduct repeat surveys on consecutive days. One day was considered enough time to reduce the possibility of a behavioral response that would impact amphibian detectability on subsequent surveys, while minimizing the chance of amphibian movement into or out of the plot between surveys. In some cases, more than 1 day did fall between repeat visits. However, we met our goal for the majority of passes (>75%), and no more than 8 days passed between surveys for any plot and year (<3 % of passes). One sampler conducted each survey and to reduce bias repeat surveys were conducted by different samplers. We counted animals and returned them to the channel at their location of capture. We included the animals detected during our first visit in our summaries of individuals encountered during stream network-wide light-touch sampling. We recorded stream temperature at the beginning of the plot (accuracy $\pm 1^\circ\text{C}$).

Table 4. The number of 30-m detection plots sampled by treatment, buffer and year. All plots in Pre 1 reflect reference conditions since buffer treatments had not yet been applied.

Buffer Type	Pre-harvest	Post-harvest					
	Pre 1	Post 1	Post 2	Post 7	Post 8	Post 14	Post 15
Reference	37	20	24	21	17	20	22
Buffered	0	27	19	21	24	17	18
Unbuffered	0	18	13	14	16	15	13

Obstructed Reach Rubble-rouse Sampling

In Post 1 & 2 and Post 7 & 8, we were not able to sample some stream reaches that were obstructed by downed trees or logging slash that prevented access to the stream or made it impossible to see under cover objects. During these years, we conducted a more intensive rubble-rouse sampling on a subset of stream meters when 5% or more of the total stream network length for a basin was obstructed. This allowed us to account for densities in obstructed reaches in our stream network-wide estimates of amphibian abundance (See Density Estimation section and

McIntyre et al. 2018, Chapter 15 – *Stream-associated Amphibians*). In Post 14 & 15 no sites had 5% or greater obstructed stream length.

Animal Processing

During both light-touch and rubble-rouse sampling, we captured amphibians by hand or with a dip net and identified each to species and life stage: larva (including individuals undergoing metamorphosis for Coastal Tailed Frog), neotene (for giant salamanders) or post-metamorph. We considered giant salamanders neotenic when they were >50 mm snout-vent length, had a shovel or rectangular shaped head, protruding eyes, and short, bushy gills. We considered salamanders post-metamorphs if they lacked external gills and a tail fin. We measured snout-vent and total lengths to the nearest 1 mm, weighed them using OHAUS® 120 g hand-held scales (rubble-rouse sampling only), and released them at the point of capture. We followed animal handling guidelines for the use of live amphibians in field research (Beaupre et al., 2004). To minimize the risk of spreading infectious diseases we sanitized all sampling and personal equipment that came into contact with amphibians or streams when traveling between watersheds.

We collected small tissue samples for some amphibians for use in genetic diversity (Coastal Tailed Frog and giant salamanders only; Spear et al., 2011; 2019) and stable isotope analyses in Phases I and II (McIntyre et al., 2018; 2021). We also collected tissue samples for giant salamanders for the purpose of genetic differentiation between the species. We collected tissue samples from individuals as they were encountered until minimum sample sizes were met (target samples sizes ranged from 10 samples for stable isotopes analysis and 40 for the genetic analysis). After that point we collected tissue samples from the first individual encountered in each 10-m sample interval so that samples were distributed equally throughout the stream network. We collected tail tissue from salamanders and Coastal Tailed Frog larvae and toe clips from post-metamorphic Coastal Tailed Frogs. We did not collect tissue from animals with injuries (e.g., missing part of tail or limb). We used sterilized dissecting scissors to remove tissue and placed samples in 1.5-ml ethanol-filled sample vials. Animals were immediately released at the point of capture. Samples were kept on ice for transport from the field to the lab, where they were immediately placed in a freezer.

Species Observations

We summarized amphibian species observations by site and year, since not all taxa were detected in every site or year. We did not include animals from the 3-m obstructed rubble-rouse plots since we conducted those surveys only in the post-harvest period. We will note observations that confirm occupancy for a species in the rare case that it was detected only in obstructed plots or incidentally.

Density Estimation

We calculated Coastal Tailed Frog densities for larvae and post-metamorphs separately due to differences in body structure, physical requirements, and diet. We considered individuals in the process of metamorphosis to be larvae. We combined the counts of Coastal and Cope's Giant Salamander for analysis because of differentiating and hybridization considerations between the two species (Spear et al., 2011). We also combined the three species of torrent salamanders into a single group for analysis because the range of each single species by itself only spans a small

number of study sites. This assumes that ecology and response to disturbance among torrent salamander species is similar, an assumption based on the fact that the species were only relatively recently identified as distinct (Good & Wake, 1992) and the three species use habitats similarly (Jones et al., 2005).

We used a modified double-sampling design (Pollock et al., 2002) whereby we estimated stream network-wide density by applying detection probability estimates derived from a subset of 30-m detection plots to animal counts collected throughout the study site using the light-touch method. To do this, we delineated reaches throughout the entirety of each study site, so that the entire stream length of every study site from the F/N break and upstream to the PIP along every tributary was assigned to one combination of two covariates, which included stream order (first-order or second-/third-order) and buffer type (reference, buffered, or unbuffered). Hereafter, we refer to these reaches as single-pass reaches. The upstream and downstream limits of each single-pass reach were defined as the point at which either one of the two covariates changed (e.g., went from first- to second-order or from buffered to unbuffered). The number of single-pass reaches at a site ranged from 2 to 23.

We field-verified the stream order (Strahler, 1952) for each single-pass plot by walking the channel network one time in the pre- (2006) and one time in the post-harvest (2010) period. We obtained stream temperature for each single-pass plot from the StowAway TidbiT thermistors (Onset Computer Corporation, Bourne, Massachusetts) used for stream temperature monitoring or from handheld thermometers. Temperature sensors were spaced from the F/N break to the PIP on the mainstem channel as well as on side tributaries, just upstream from the confluence with the mainstem. Data were collected at 30-minute intervals. During Phase I and II, we calculated stream temperature for each single-pass plot as the average temperature recorded by the nearest sensor during the period between 0800 and 1700 hours on the day, or days, that sampling occurred. During Phase 3, we used handheld thermometers to obtain stream temperature in the plot at the time of sampling. The purpose of stream temperature data collection during Phase 3 was to enable us to adjust detection and density estimates by temperature.

We calculated stream network-wide amphibian density for each study site and year as a linear density (count/30 m) in five steps: (1) estimating detection probability at the 30-m detection plot level (Royle, 2004); (2) dividing observed counts in all single-pass reaches by the detection probability estimated for each different combination of covariates (stream order, stream temperature and buffer type); (3) calculating the mean density within a site for each combination of stream order and buffer type by adding all adjusted counts and dividing by the total stream length for each combination, then normalizing to 30 m; (4) calculating the stream network-wide weighted mean of adjusted single-pass reach-level densities based on total stream lengths for each stream order and buffer type combination; and (5) adjusting linear density to incorporate the mean density from 3-m obstructed plots, when applicable, and based on the obstructed length by site and post-harvest year. The constituent habitat types included as sampling strata were stream order, buffer type, and obstructed/unobstructed reach.

We used data obtained from the detection plots to estimate detection probabilities using the *N*-mixture model approach of Royle (2004). Specifically, we used a Poisson mixing distribution and a log-link function for the abundance model and a logit-link function for the detection

model. We note that, unlike in the post-harvest analysis, we did not perform adjustments for detection probability to our counts for tailed frogs (steps 1 and 2 above). Zero counts in several basins led to unstable estimates of detection probability. Therefore, adjustments for detection probability were only performed for torrent and giant salamanders. The mean model (i.e., the model for the expected value) for torrent salamander and giant salamander abundance included covariates for stream order, year, buffer type, and the buffer type \times year interaction, along with a basin-specific random intercept. The detection model for these two taxa contained covariates for stream order, stream temperature, year and buffer type. In the abundance model, buffer type was defined by the post-harvest state and was constant across all years (i.e., reference, buffered and unbuffered for all single-pass reaches located in the reference, 100% and 0% treatments, respectively, and buffered or unbuffered for plots located in the FP treatment). The interaction term (buffer type \times year) accounted for the buffer treatment application. For the detection model, buffer type for all study sites was defined as a reference condition during the pre-harvest period but took the post-harvest state during the post-harvest period.

We fit all N -mixture models within a Bayesian framework using the WinBUGS (Spiegelhalter et al., 2003) software package called from R (R Development Core Team, 2010) using package R2WinBUGS (Sturtz et al., 2005). We assessed convergence using the Gelman-Rubin statistic (Gelman et al., 2004) and visual inspection of the chains and used posterior predictive checks to check for consistency between the model and the data.

We used estimates obtained from the N -mixture model in detection plots to predict detection probabilities for all single-pass plots, across all basins and years, using the appropriate covariate data. We accounted for the uncertainty in the detection probability estimates in our adjusted density estimates (McIntyre et al. 2018, Chapter 15 – Stream-Associated Amphibians, Appendix 15-A). We did not have the replicated count data for Pre 3 and Pre 2 needed to estimate detection probability, so we based estimates for detection probabilities for those years on data collected in Pre 1. We justified this approach based on the fact that: (1) all pre-harvest years are in the reference state; (2) relevant covariate data were collected during Pre 3 and Pre 2; and (3) detection probability estimates for Post 1 & 2 were close for all species. We conducted a sensitivity analysis by fitting the Before-After Control-Impact (BACI) model without Pre 3 and Pre 2 data and comparing results to the full analysis. Across all species, the results were sufficiently similar that we felt comfortable including the Pre 3 and Pre 2 data, which provided better precision on our estimates due to larger sample sizes.

We calculated estimates of amphibian linear density from the adjusted single-pass plot-level abundance values by considering the adjusted counts as coming from a stratified random sample. The constituent habitat types included as sampling strata were stream order, buffer type, and obstructed/unobstructed reach. We estimated the length of the obstructed stratum separately for all post-harvest years. We calculated separate estimates for each basin by year. We calculated the amphibian linear density for stratum h in basin i in year j as follows:

$$\tilde{N}_{ijh} = C \cdot \frac{\sum_k \tilde{N}_{ijhk}}{\sum_k c_{ijhk}} \quad (\text{Eq. 1})$$

where: k indexes plot,

\tilde{N}_{ijk} is the adjusted plot abundance,
 c_{ijk} is the plot length, and
 $C = 30$ m.

We calculated the weighted abundance estimate for basin i in year j as follows:

$$\tilde{N}_{ij} = \sum_h w_{ijh} \cdot \tilde{N}_{ijh} \quad (\text{Eq. 2})$$

where: $w_{ijh} = l_{ijh}/l_{ij}$, with l_{ijh} = stratum network length, and
 l_{ij} = total stream network length.

Statistical Analysis

We designed this study to evaluate differences in the magnitude of change (post-harvest – pre-harvest) among treatments at the site scale. We evaluated the effect of clearcut timber harvest with three variable-length riparian-buffer treatments relative to an unharvested control (reference). We used a Before-After Control-Impact (BACI) design whereby we established baseline conditions across study sites, implemented harvest at buffer treatment sites and monitored the response after harvest. The BACI design allowed us to compare harvested sites to their pre-harvest baseline conditions and unharvested references. An advantage of this design is that it controls for the effect of large-scale temporal variation (e.g., annual environmental variability) by establishing relationships between the control (i.e., unharvested reference) and impact (i.e., buffer treatment) sites in the pre- versus post-harvest periods (Smith, 2002), allowing us to adjust for environmental variation when estimating the impact of forest practices on post-harvest responses.

Randomization during site selection, when possible, helped ensure that there was not a systematic bias in the comparison of treatment effects; however, with smaller sample sizes there may be some bias in the sites to which treatments were assigned by chance.

The statistical models used for the analysis of the BACI design include a blocking term, which groups sites geographically to increase precision, and a year term to account for inter-annual environmental variability. The model error term represents experimental error, which captures several sources of variation, including within-site sampling variability, measurement error, site \times time interaction, and site \times treatment interaction. The latter two terms correspond to the variation in the year effect by basin and the variation in treatment effect by basin, respectively. Other sources of variation are also included in the experimental error.

As with many ecological studies, our statistical analysis was limited by sample size, variability among plots, sites and blocks, and missing replicates of some treatments in some blocks. Marginally significant effect ($0.05 < P < 0.15$) sizes would likely be significant with greater replication, leading to greater confidence in our interpretations. For these reasons, we set α and β

at 0.1 *a priori* (e.g., Underwood, 1997; Welsh & Ollivier, 1998), including for stream-associated amphibian density. Interpretation of results consider the relatively small sample sizes, the effect sizes, and variability associated with response variables. Hence, understanding the overall pattern of responses, rather than focusing on a single P-value associated with any one result, is an integral part of appropriately evaluating our results.

We used generalized linear mixed effects models to evaluate the pre- versus post-harvest changes for each treatment type (McDonald et al., 2000). The analysis focused on estimating mean treatment effects in each of three post-harvest time periods: 1-2 years, 7-8 years, 14-15 years. For each response, the models contained block and site random effects, as well as fixed effects for year to account for interannual variation. The models were further parameterized with terms for all combinations of treatment and post-harvest period. Post-hoc contrasts were used to estimate treatment effects for each post-harvest period. We examined pairwise contrasts for six combinations of references and buffer treatments, namely: REF vs. 0%, REF vs. FP, REF vs. 100%, 0% vs. FP, 0% vs. 100%, and FP vs. 100%.

The analyses of density produce results on the natural log (ln) scale. We exponentiated the difference in the natural logs of post- and pre-harvest values to give an estimate of the proportional change in density on its original scale. Therefore, a back-transformed result equal to 1 equates to no change in the average pre- and post-harvest estimates. A value between 0 and 1 equates to a result in the post-harvest period that is less than the average in the pre-harvest period. A value greater than 1 equates to a result in the post-harvest period that is more than the average in the pre-harvest period. For example, estimates of 0.5 and 1.5 equate to a 50% decrease and a 50% increase from pre- to post-harvest, respectively. We present estimates in the text of the results for estimates for which the 95% credible interval does not include 1.

In cases where low amphibian counts led to numeric instability in maximum likelihood estimates from the GLMM, we fit the model using Bayesian methods. All Bayesian models were fit using JAGS (Plummer, 2003) called from the R programming environment. We specified Gaussian priors for all parameters, and performed sensitivity checks to verify that conclusions were consistent across a range of vague priors. Posterior mean estimates, contrasts, and 95% credible intervals were used to summarize results from all Bayesian analyses. We note that P-values are not available from the Bayesian analysis.

Basin-level density estimates for both torrent and giant salamanders were adjusted for imperfect detection (*reference where this was described*) using estimates of detection probability from fitted N-mixture models (Royle, 2004). We propagated detection probability uncertainty into our generalized linear mixed model analysis using multiple imputation (Little & Rubin, 2019). Specifically, we used the following steps to account for this uncertainty:

1. Draw a sample s from the posterior distribution of the fitted N-mixture model.
2. Calculate detection probabilities using sample s and covariate data for each single-pass light touch sample.
3. Adjust observed counts by dividing by the calculated detection probabilities in step 2; aggregate the adjusted counts to obtain basin-wide density estimates, by year.
4. Fit the generalized linear mixed model to the basin-wide density estimates in step 3 and record contrast estimates and standard errors.

5. Repeat steps 1-4 *S* times.
6. Calculate the mean of the squared standard error over the *S* samples for each contrast; calculate the variance of contrast mean estimates over the *S* samples. Sum these two quantities.
7. Calculate the sample average over all *S* contrast mean estimates.

The square root of the sum in step 6 is an estimate of contrast standard error that incorporates both experimental error and uncertainty in the estimated detection probability. This value was used to calculate 95% confidence intervals. Due to the use of multiple imputation, we do not report p-values for either the torrent or giant salamander results. The generalized linear mixed model in step 4 was fit in R using the glmmPQL function in package MASS (Venables & Ripley, 2002).

Results

Summary of Amphibian Species Observations

In the Phase I and II efforts, we made 21,194 amphibian observations using light-touch and rubble-rouse techniques in the lower Np reach, of which 98% were focal amphibians (McIntyre et al., 2021). As a part of the Phase III effort, we made an additional 4,818 observations for focal amphibians through our light-touch and triple pass efforts. Of those, 480 were Coastal Tailed Frog, 2,951 were torrent salamanders, and 1,387 giant salamanders. In the pre-treatment period, we detected Coastal Tailed Frog in 15 of 17 sites, and torrent and giant salamanders in all 17 sites (**Table 5**). In Post 14 & 15, we detected Coastal Tailed Frog in 10 of 15 sites, torrent salamanders in 13 of 15 sites, and giant salamanders in all 15 sites that were included in the Phase III comparison (**Table 5**).

Table 5. Focal amphibian taxa detected during stream network-wide light-touch for all study sites and periods (pre-harvest, post-harvest [Post 1 & 2; Post 7 & 8, and Post 14 & 15]). Filled circles (●) indicate where a focal taxa was detected for a site and period and empty circles (○) where a taxa was not detected. * indicates sites not included in Post 14 & 15 treatment response. ^ indicates WIL-FP-2 was harvested in 2016 and was included as an FP treatment in Phase III.

Site Code	Coastal Tailed Frog				Giant Salamanders				Torrent Salamanders			
	Pre	Post 1 & 2	Post 7 & 8	Post 14 & 15	Pre	Post 1 & 2	Post 7 & 8	Post 14 & 15	Pre	Post 1 & 2	Post 7 & 8	Post 14 & 15
OLYM-REF	●	●	●	●	●	●	●	●	●	●	●	●
WIL-REF-1*	●	●	●	*	●	●	●	*	●	●	●	*
WIL-REF-2*	●	●	●	*	●	●	●	*	●	●	●	*
WIL-REF-3	●	●	●	●	●	●	●	●	●	●	●	●
CASC-REF	●	●	●	●	●	●	●	●	●	●	●	●
OLYM-100%	○	●	●	●	●	●	●	●	●	●	●	●

WIL-100%-1	●	●	●	●	●	●	●	●	●	●	●	●
WIL-100%-2	●	●	●	●	●	●	●	●	●	●	●	●
WIL-100%-3	●	●	●	●	●	●	●	●	●	●	●	●
OLYM-FP	●	●	●	○	●	●	●	●	●	○	●	○
WIL-FP-1	●	●	●	○	●	●	●	●	●	●	●	●
WIL-FP-2^	●	●	●	●	●	●	●	●	●	●	●	●
CASC-FP	●	○	○	○	●	●	●	●	●	●	●	●
OLYM-0%	●	○	●	●	●	●	●	●	●	●	○	○
WIL-0%-1	●	●	●	●	●	●	●	●	●	●	●	●
WIL-0%-2	●	●	●	○	●	●	●	●	●	●	●	●
CASC-0%	○	○	○	○	●	●	●	●	●	●	●	●

Density

Coastal Tailed Frog Larvae

Mean annual larval tailed frog densities ranged from 0.0 to 3.1 in the pre-harvest period, 0.0 to 4.5 in Post 1 & 2, 0.0 to 1.2 in Post 7 & 8, and 0.0 to 2.3 in Post 14 & 15 (**Figure 3**). We found evidence that treatments differed in the magnitude of change over time (**Table 6; Figure 4; Table 7**).

In Post 1 & 2, we estimated the between-treatment comparison for the 100% treatment and the reference to be 1.61 (approximate 95% credible interval: 1.08, 2.41) or in other words a +61% (approximate 95% credible interval +8%, +141%) change in mean density compared to pre-harvest period after controlling for temporal changes in the reference. Likewise, for the FP treatment we estimated a +72% (approximate 95% credible interval +9%, +171%) change in density compared to the change in the reference.

In Post 7 & 8, we estimated a -58% (-82%, -1%), -94% (-99%, -66%), and -75% (-93%, -8%) change in density in the 100%, FP, and 0% treatments, compared with the change in the reference.

In Post 14 & 15, we estimated a -71% (-86%, -41%), -95% (-99%, -68%), and -70% (-86%, -37%) change in density in the 100%, FP, and 0% treatments, compared with the change in the reference.

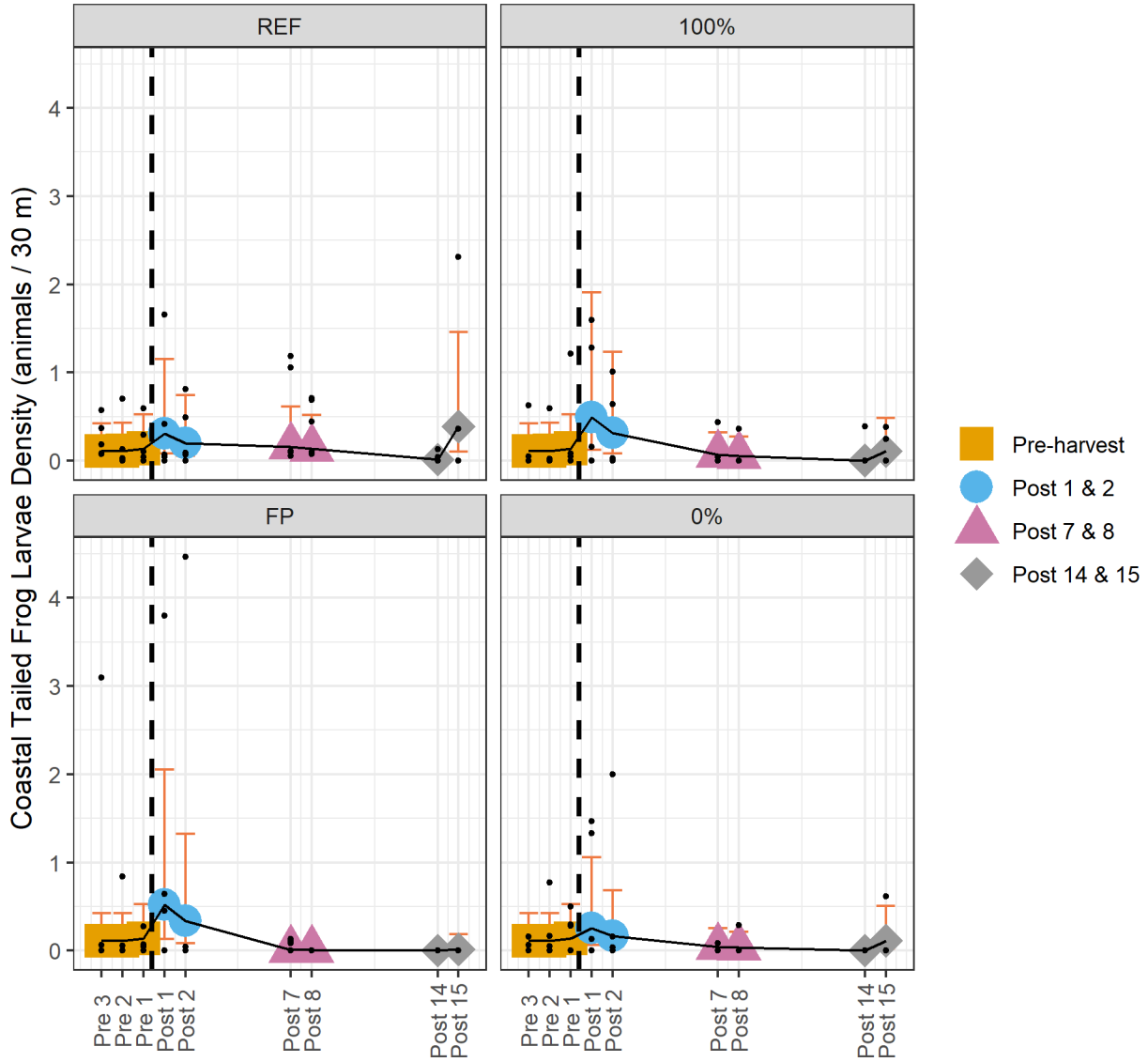


Figure 3. Mean larval Coastal Tailed Frog density (animals/30 m) by sample year. Vertical colored lines show approximate 95% credible intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 6. The within-treatment estimate of the proportional change and 95% credible intervals (CI) for mean larval Coastal Tailed Frog density between the pre-harvest period and Post 1 & Post 2, Post 7 & Post 8 and Post 14 & Post 15.

Treatment	Estimate (CI)		
	Post 1 & 2	Post 7 & 8	Post 14 & 15
REF	2.03 (1.69, 2.44)	1.22 (0.97, 1.54)	0.58 (0.37, 0.91)

100%	2.58 (2.09, 3.18)	0.79 (0.51, 1.23)	0.31 (0.18, 0.55)
FP	2.66 (2.08, 3.41)	0.31 (0.12, 0.83)	0.13 (0.05, 0.36)
0%	1.85 (1.34, 2.56)	0.61 (0.32, 1.19)	0.32 (0.18, 0.56)

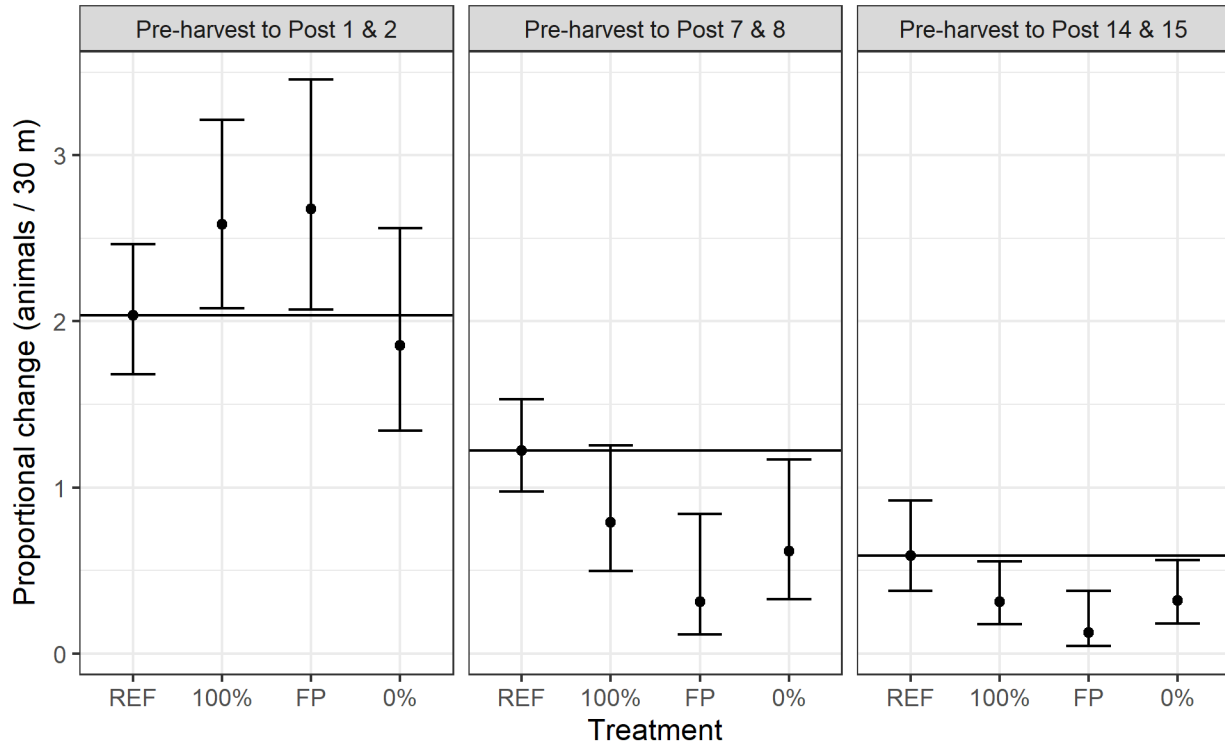


Figure 4. The within-treatment estimate of the proportional change and approximate 95% credible intervals for mean larvae Coastal Tailed Frog density between the pre-harvest and post-harvest periods Post 1 & 2, Post 7 & 8 and Post 14 & 15. A horizontal line placed at the reference treatment (REF) value indicates the estimated temporal change under reference conditions.

Table 7. The between-treatment comparison of the proportional change and approximate 95% credible intervals (CI) of the estimates for mean larval Coastal Tailed Frog density. Contrasts with credible intervals that do not overlap 1 are bolded. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	Estimate (CI)		
	Post 1 & 2	Post 7 & 8	Post 14 & 15
100% vs. REF	1.61 (1.08, 2.41)	0.42 (0.18, 0.99)	0.29 (0.14, 0.59)
FP vs. REF	1.72 (1.09, 2.71)	0.06 (0.01, 0.44)	0.05 (0.01, 0.32)
0% vs. REF	0.83 (0.44, 1.58)	0.25 (0.07, 0.92)	0.30 (0.14, 0.63)
0% vs. FP	0.48 (0.23, 1.02)	3.87 (0.38, 39.02)	6.12 (0.81, 46.26)
0% vs. 100%	0.52 (0.25, 1.05)	0.60 (0.13, 2.78)	1.04 (0.37, 2.88)

FP vs. 100% 1.07 (0.60, 1.91) 0.16 (0.02, 1.27) 0.17 (0.02, 1.25)

Coastal Tailed Frog Post-metamorphs

Mean annual post-metamorphic tailed frog densities ranged from 0.0 to 2.2 in the pre-harvest period, 0.0 to 2.5 in Post 1 & 2, 0.0 to 0.7 in Post 7 & 8, and 0.0 to 1.7 in Post 14 & 15 (**Figure 5**). We found evidence that treatments differed in the magnitude of change over time (**Table 8; Figure 6; Table 9**).

In Post 1 & 2, we estimated the between-treatment comparison for the 100% treatment and the reference to be 0.37 (approximate 95% credible interval: 0.20, 0.68) or in other words a -63% (approximate 95 % credible interval -80%, -32%) change in mean density compared to pre-harvest period after controlling for temporal changes in the reference. Likewise, for the comparison of the 0% treatment and the reference we estimated a +343% (approximate 95 % credible interval +79%, +993%) change in density. We also estimated a +522% (+66%, +2229%) change in density in the 0% treatment compared with the FP treatment and a +1112% (+354, +3137%) change in density in the 0% treatment compared to the 100% treatment after adjusting for pre-harvest differences among the treatment sites.

In Post 7 & 8, we estimated a -88% (-96%, -64%) and -91% (-98%, -63%), change in density in the 100% and FP treatments compared with the change in the reference.

In Post 14 & 15, we estimated a -98% (-99.6%, -93%), -97% (-99.6%, -82), -85% (-97%, -36%) change in density in the 100%, FP, and 0% treatments, compared with the temporal change in the reference. We also estimated a +781% (+16%, +6601%) change in density in the 0% treatment compared with the change in the 100% treatment.

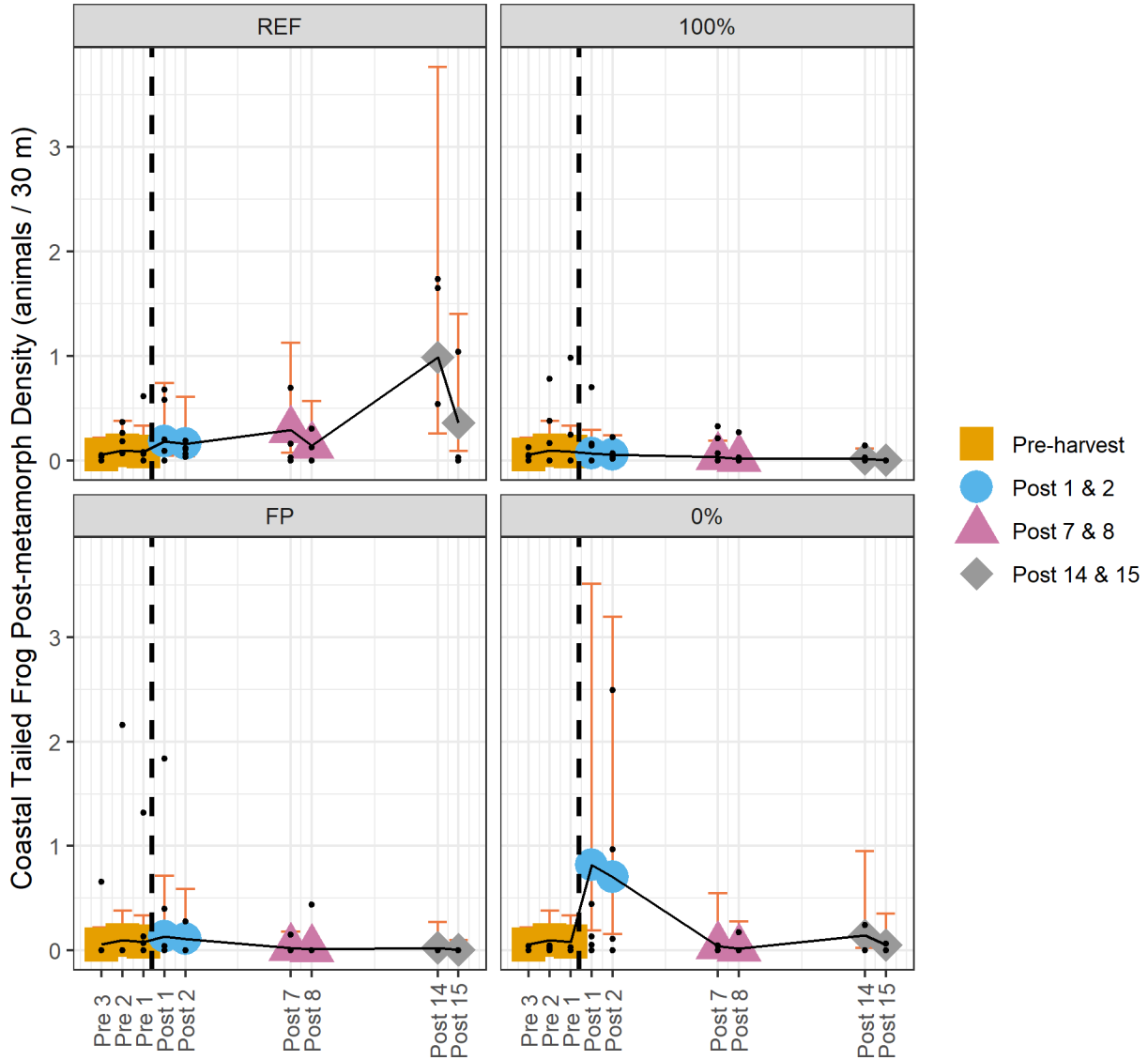


Figure 5. Mean post-metamorphic Coastal Tailed Frog density (animals/30 m) by sample year. Vertical colored lines show approximate 95% credible intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 8. The within-treatment estimate of the proportional change and approximate 95% credible intervals (CI) for mean post-metamorphic Coastal Tailed Frog density between the pre-harvest period and post-harvest periods Post 1 & 2, Post 7 & 8 and Post 14 & 15.

Treatment	Estimate (CI)		
	Post 1 & 2	Post 7 & 8	Post 14 & 15
REF	2.19 (1.49, 3.22)	2.6 (1.79, 3.77)	7.65 (5.57, 10.51)
100%	1.33 (0.94, 1.87)	0.91 (0.52, 1.6)	0.99 (0.48, 2.04)

FP	1.85 (1.02, 3.34)	0.78 (0.36, 1.68)	1.28 (0.49, 3.35)
0%	4.62 (2.93, 7.27)	0.98 (0.31, 3.09)	2.93 (1.37, 6.26)

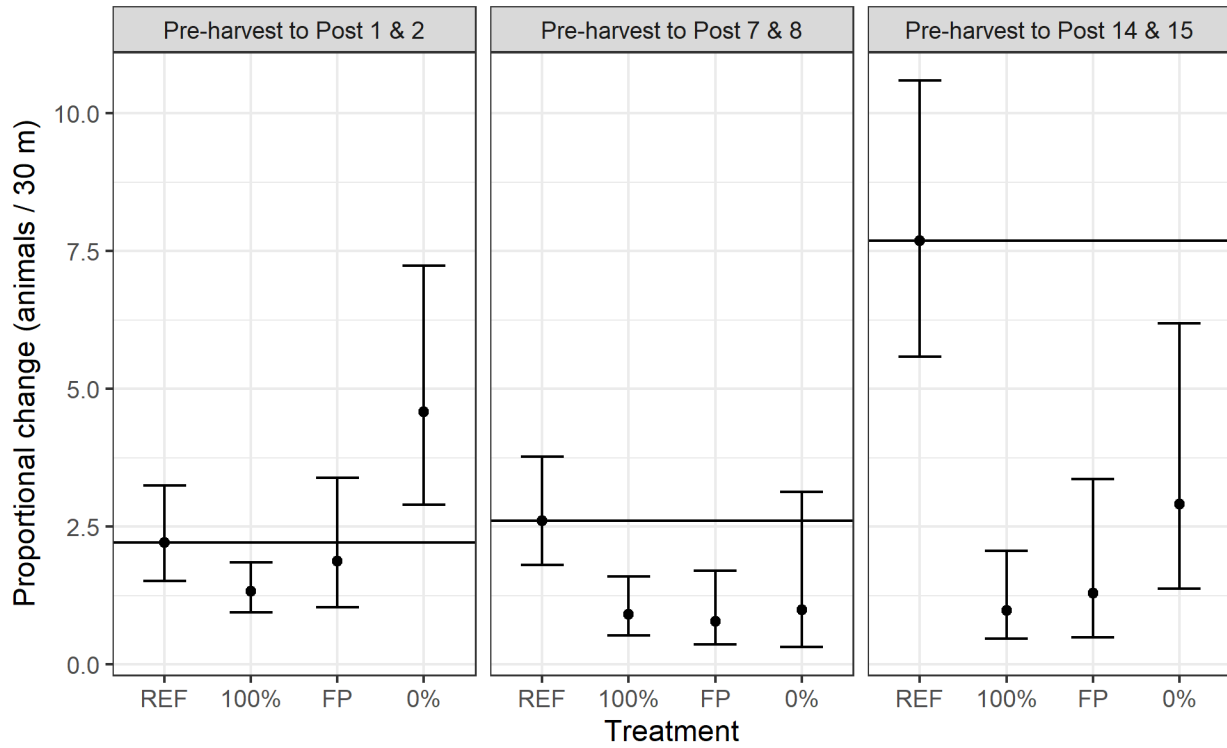


Figure 6. The within-treatment estimate of the proportional change and approximate 95% credible intervals for mean post-metamorphic Coastal Tailed Frog density between the pre-harvest and post-harvest periods Post 1 & 2, Post 7 & 8 and Post 14 & 15. A horizontal line placed at the reference treatment (REF) value indicates the estimated temporal change under reference conditions.

Table 9. The between-treatment comparison of the proportional change and approximate 95% credible intervals (CI) of the estimates for mean post-metamorphic Coastal Tailed Frog density. Contrasts with credible intervals that do not overlap 1 are bolded. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	Estimate (CI)		
	Post 1 & 2	Post 7 & 8	Post 14 & 15
100% vs. REF	0.37 (0.20, 0.68)	0.12 (0.04, 0.36)	0.02 (0.004, 0.07)
FP vs. REF	0.71 (0.25, 2.05)	0.09 (0.02, 0.37)	0.03 (0.004, 0.18)
0% vs. REF	4.43 (1.79, 10.93)	0.14 (0.02, 1.31)	0.15 (0.03, 0.64)
0% vs. FP	6.22 (1.66, 23.29)	1.59 (0.11, 22.11)	5.26 (0.49, 56.32)
0% vs. 100%	12.12 (4.54, 32.37)	1.17 (0.10, 13.14)	8.81 (1.16, 67.01)
FP vs. 100%	1.95 (0.61, 6.25)	0.74 (0.13, 4.33)	1.67 (0.16, 17.54)

Torrent Salamander

Mean annual torrent salamander densities ranged from 0.0 to 69.0 in the pre-harvest period, 0.0 to 171.0 in Post 1 & 2, 0.0 to 117.0 in Post 7 & 8, and 0.0 to 19.7 in Post 14 & 15 (**Figure 7**). We found evidence that treatments differed in the magnitude of change over time (**Table 10; Figure 8; Table 11**).

In Post 1 & 2, we estimated the between-treatment comparison for the 100% treatment and the reference to be 2.96 (approximate 95% credible interval: 1.42, 6.18) or in other words a +196% (approximate 95 % credible interval +42%, +518%) change in mean density compared to pre-harvest period after controlling for temporal changes in the reference. Likewise, we estimated a +130% (+19%, +343%), +187% (+36%, +502%), and -75% (-93%, -8%) change in density in the 100%, FP, and 0% treatments, compared with the change in the reference.

In Post 7 & 8, we estimated a -58% (-79%, -15%) change in density in the FP treatment compared with the change in the 100% treatment.

In Post 14 & 15, we estimated a -88% (-98%, -38%) and a -80% (-95%, -18%) change in density in the FP and 0% treatments compared with the change in the reference.

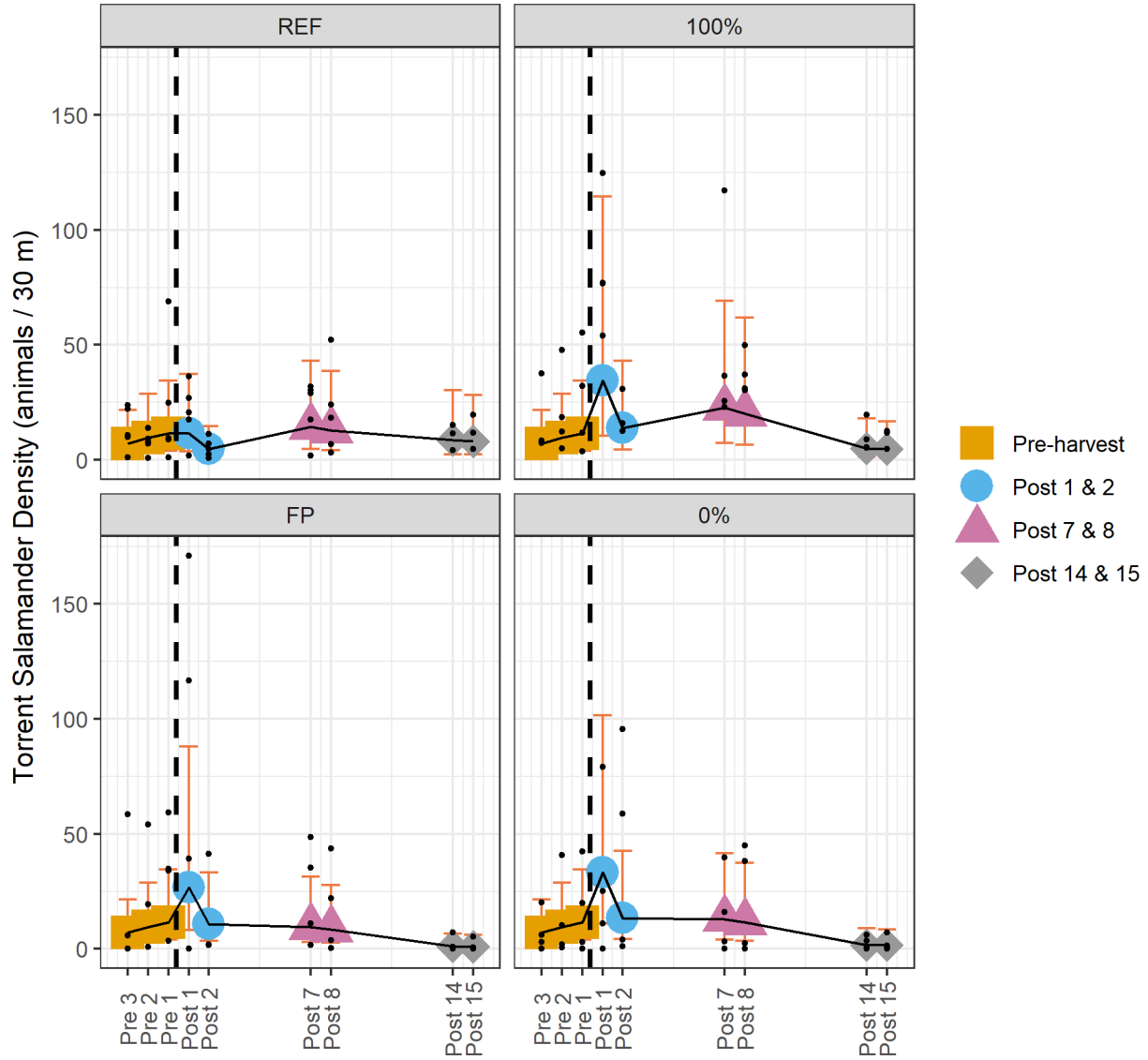


Figure 7. Mean Torrent Salamander density (animals/30 m) by sample year. Vertical colored lines show approximate 95% credible intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 10. The within-treatment estimate of the proportional change and 95% credible intervals (CI) for mean torrent salamander density between the pre-harvest period and post-harvest periods Post 1 & 2, Post 7 & 8 and Post 14 & 15.

Treatment	Estimate (CI)		
	Post 1 & 2	Post 7 & 8	Post 14 & 15
REF	0.80 (0.43, 1.48)	1.46 (0.86, 2.49)	0.90 (0.43, 1.88)
100%	2.36 (1.28, 4.37)	2.30 (1.31, 4.03)	0.52 (0.24, 1.14)
FP	1.83 (1, 3.37)	0.97 (0.50, 1.9)	0.11 (0.02, 0.51)

0% 2.29 (1.27, 4.13) 1.32 (0.68, 2.56) 0.18 (0.05, 0.67)

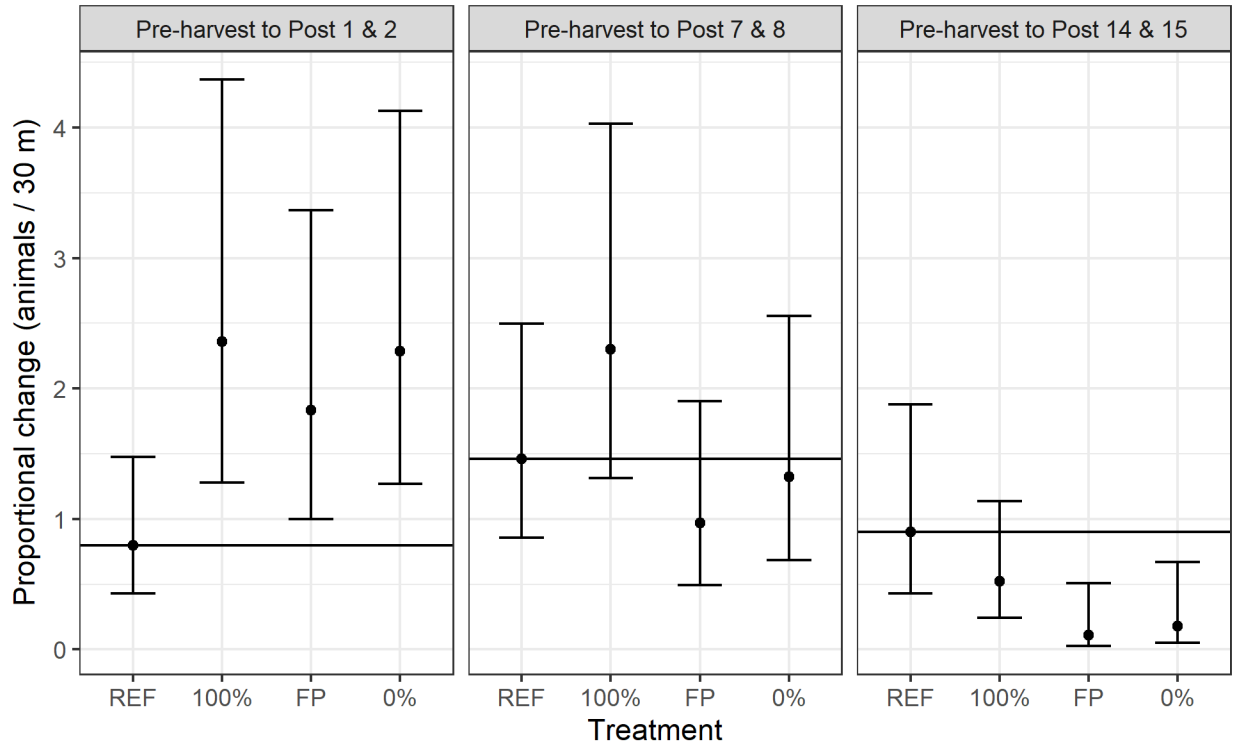


Figure 8. The within-treatment estimate of the proportional change and approximate 95% credible intervals for mean torrent salamander density between the pre-harvest and post-harvest periods Post 1 & 2, Post 7 & 8 and Post 14 & 15. A horizontal line placed at the reference treatment (REF) value indicates the estimated temporal change under reference conditions.

Table 11. The between-treatment comparison of the proportional change and 95% credible intervals (CI) of the estimates for mean torrent salamander density. Contrasts with credible intervals that do not overlap 1 are bolded. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	Estimate (CI)		
	Post 1 & 2	Post 7 & 8	Post 14 & 15
100% vs. REF	2.96 (1.42, 6.18)	1.57 (0.82, 3.02)	0.58 (0.23, 1.48)
FP vs. REF	2.30 (1.19, 4.43)	0.66 (0.32, 1.39)	0.12 (0.02, 0.62)
0% vs. REF	2.87 (1.36, 6.02)	0.90 (0.43, 1.89)	0.20 (0.05, 0.82)
0% vs. FP	1.25 (0.58, 2.67)	1.36 (0.58, 3.19)	1.63 (0.24, 11.21)
0% vs. 100%	0.97 (0.45, 2.09)	0.57 (0.27, 1.23)	0.35 (0.08, 1.44)
FP vs. 100%	0.78 (0.43, 1.41)	0.42 (0.21, 0.85)	0.21 (0.04, 1.07)

Giant Salamander

Mean annual giant salamander densities ranged from 0.2 to 33.9 in the pre-harvest period, 0.2 to 21.3 in Post 1 & 2, 1.6 to 54.4 in Post 7 & 8, and 0.0 to 13.0 in Post 14 & 15 (**Figure 9**). We found evidence that treatments differed in the magnitude of change over time (

Table 12; Figure 10; Table 13).

In Post 1 & 2, we estimated the between-treatment comparison for the FP treatment and the reference to be 0.35 (approximate 95% credible interval: 0.17, 0.72) or in other words a -65% (approximate 95 % credible interval -83%, -28%) change in mean density compared to pre-harvest period after controlling for temporal changes in the reference. Likewise, we estimated a +266% (+78%, +649%) change in density in the 0% treatment compared to the FP treatment and a -62% (-92%, -26%) change in density in the FP treatment compared to the 100% treatment after adjusting for pre-harvest differences among the treatment sites.

In Post 7 & 8, we estimated a -53% (-77%, -7%) change in density in the FP treatment compared with the change in the reference.

In Post 14 & 15, we estimated a -81% (-94%, -43%) change in density in the FP treatment compared with the change in the reference and a -76% (-92%, -26%) change in density in the FP treatment compared with the change in the 100% treatment.

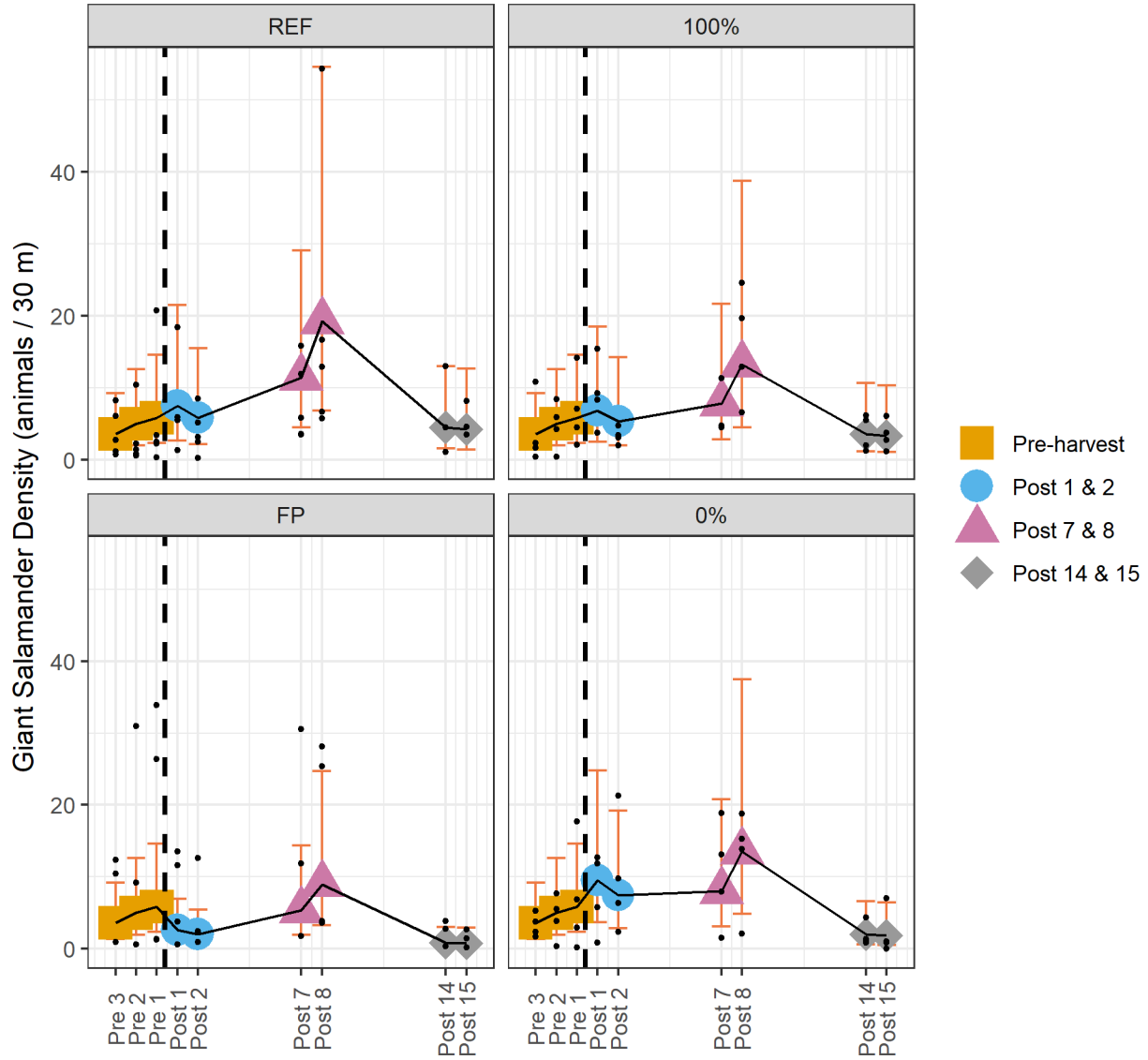


Figure 9. Mean giant salamander density (animals/30 m) by sample year. Vertical colored lines show approximate 95% credible intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 12. The within-treatment estimate of the proportional change and 95% credible intervals (CI) for mean giant salamander density between the pre-harvest period and post-harvest periods Post 1 & 2, Post 7 & 8 and Post 14 & 15.

Treatment	Estimate (CI)		
	Post 1 & 2	Post 7 & 8	Post 14 & 15
REF	1.41 (0.72, 2.76)	3.15 (1.69, 5.87)	0.93 (0.44, 1.95)

100%	1.28 (0.66, 2.50)	2.16 (1.09, 4.31)	0.74 (0.34, 1.60)
FP	0.49 (0.25, 0.94)	1.47 (0.78, 2.74)	0.18 (0.06, 0.48)
0%	1.79 (0.98, 3.26)	2.21 (1.14, 4.27)	0.41 (0.17, 1.02)

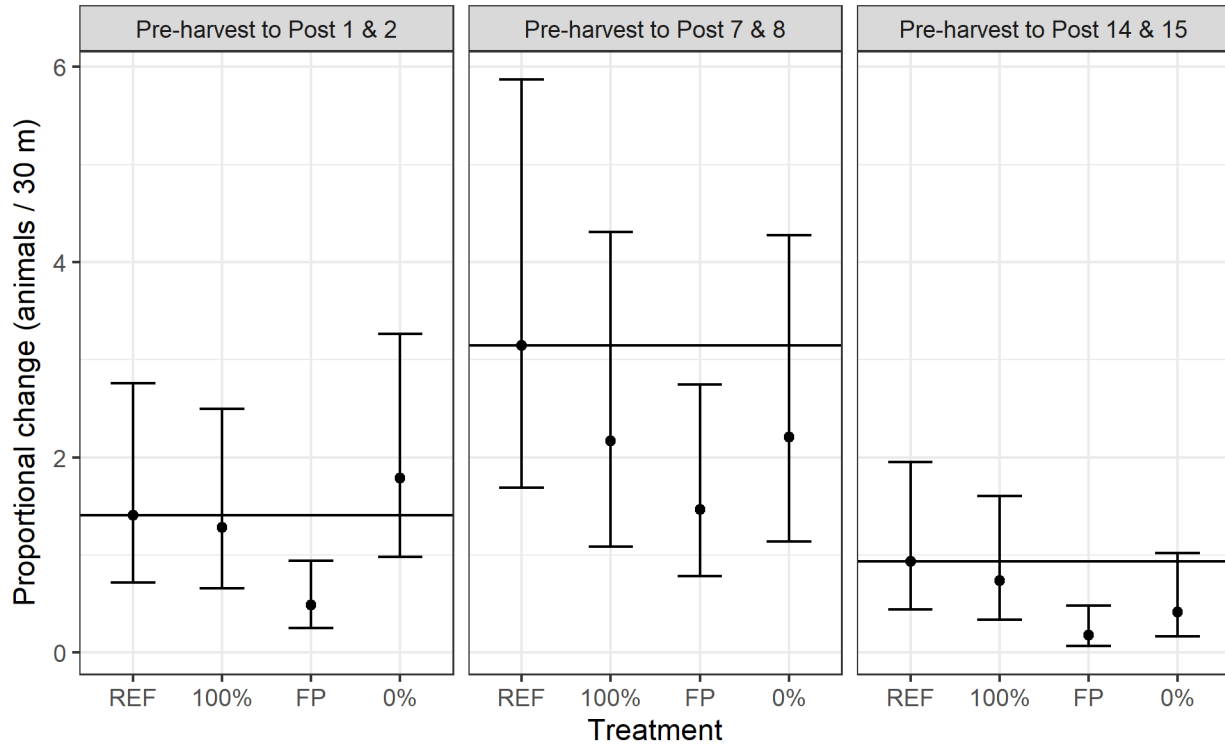


Figure 10. The within-treatment estimate of the proportional change and approximate 95% credible intervals for mean giant salamander density between the pre-harvest and post-harvest periods Post 1 & 2, Post 7 & 8 and Post 14 & 15. A horizontal line placed at the reference treatment (REF) value indicates the estimated temporal change under reference conditions.

Table 13. The between-treatment comparison of the proportional change and 95% credible intervals (CI) of the estimates for mean giant salamander density. Contrasts with credible intervals that do not overlap 1 are bolded. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	Estimate (CI)		
	Post 1 & 2	Post 7 & 8	Post 14 & 15
100% vs. REF	0.91 (0.43, 1.94)	0.69 (0.33, 1.42)	0.79 (0.33, 1.87)
FP vs. REF	0.35 (0.17, 0.72)	0.47 (0.23, 0.93)	0.19 (0.06, 0.57)
0% vs. REF	1.27 (0.60, 2.68)	0.70 (0.35, 1.42)	0.44 (0.17, 1.19)
0% vs. FP	3.66 (1.78, 7.49)	1.50 (0.73, 3.12)	2.34 (0.68, 7.98)

0% vs. 100%	1.39 (0.67, 2.90)	1.02 (0.46, 2.25)	0.56 (0.20, 1.57)
FP vs. 100%	0.38 (0.18, 0.80)	0.68 (0.37, 1.25)	0.24 (0.08, 0.74)

Water Temperature

Maximum 7-day average daily maximum (7-DADMax) stream temperature ranged from 9.8 to 16.3 during the summer of 2023 (**Table 14**).

Table 14. Maximum 7-day average daily maximum (7-DADMax) for Post 15 recorded near the F/N break. * asynchronous harvest year (see **Table 3**). Prior to 2023, water temperature monitoring was conducted by Washington State Department of Ecology (McIntyre et al., 2021, Chapter 4 – *Stream Temperature and Cover*). * indicates sites not included in Post 14 & 15 treatment response. ^ indicates WIL-FP-2 was harvested in 2016 and was included as an FP treatment in Phase III.

Site Code	2006 Pre 3	2007 Pre 2	2008 Pre 1	2009 Post 1	2010 Post 2	2015 Post 7	2016 Post 8	2023 Post 15
OLYM-REF	-	11.7	11.8	12.8	11.5	12.4	12.3	14.3
WIL-REF-1	13.3	12.4	12.5	13.4	12.3	13.2	13.6	*
WIL-REF-2	13.2	12.4	13.2	14.6	13.4	13.8	13.6	*
WIL-REF-3	9.2	9.5	9.2	8.9	9.5	10	10.5	9.8
CASC-REF	13.9	13.5	13	15.3	12.4	14.4	14.3	14
OLYM-100%	14.9	13.4	13.4	15	13.7	14.5	14.3	14.9
WIL-100%-1	13	12.1	12.3	14.3	13.3	13	13	12.9
WIL-100%-2	12.7	12	12.4	14.3	13.2	12.6	12.4	11.6
WIL-100%-3	-	14.6	15.5	19.6	16	16.1	15.3	16.3
OLYM-FP	11.1	10.5	10.9	12.4	11.2	12.3	12	11.6
WIL-FP-1	11.2	10.3	11.2	14.1	12.7	13.7	13.6	12.1
WIL-FP-2^	13	12.2	12	13.1	12	13.4	-	14 (Post 8)
CASC-FP	12.2	11.7	12.1	12.7	12.1	12.5	12.9	13.9
OLYM-0%	10.4	9.9	9.8	-	11.6	10.6	10.5	10.2
WIL-0%-1	12	11.5	11.7	17.5	15.8	12.9	12.8	14.2
WIL-0%-2	14.1	13.3	-	18.6	15.2	14.7	14.8	15.6
CASC-0%	15.1	15	16.1	-	19.5	18.4	17.4	13.2

Discussion/Conclusions

Initially, in Post 1 & 2, we observed evidence of increased Coastal Tailed Frog larval densities in the 100% and FP treatments and increased post-metamorph density in the 0% treatment compared to pre-harvest period after controlling for temporal changes in the reference. However,

by Post 7 & 8 we estimated declines in all buffer treatments. There was no evidence of recovery for the species by Post 14 & 15, when we estimated a continued decline in Coastal Tailed Frog densities in all buffer treatments. Specifically, we estimated a -71%, -95% and -70% decline in stream network-wide larval density in the 100%, FP and 0% treatments, respectively.

For post-metamorphic Coastal Tailed Frog, we estimated a decline in density for the 100% treatment in Post 1 & 2 compared to pre-harvest period after controlling for temporal changes in the reference, but large increases in density in the 0% treatment. However, like our results for larvae, by Post 7 & 8 there was no evidence for an increase in density for any treatment, and there was evidence of a decline in both the 100% and FP treatments. By Post 14 & 15 we had evidence of declines across all buffer treatments, with estimated declines of -98%, -97% and -85% in the 100%, FP and 0% treatments, respectively. Importantly, unlike larvae, post-metamorphic tailed frogs are not restricted to the stream channel, and the declines we estimated do not account for terrestrial individuals or their movements. Changes in riparian conditions may have influenced the proportion of terrestrial individuals versus those that stayed in- or near-stream. Matsuda and Richardson (2005) suggested the possibility of higher post-metamorphic mortality or increased movements and dispersal in clearcut sites.

The results from our pre-harvest genetic evaluation revealed that levels of genetic diversity were high in Coastal Tailed Frog, with little evidence of genetic clustering beyond region, and that effective population sizes were large (Spear et al., 2011), implying high levels of connectivity and movement of Coastal Tailed Frogs between drainages. It is possible that following upland harvest, tailed frog post-metamorphs moved overland into adjacent basins, and/or downstream into an unimpacted reach. The decline we observed in Coastal Tailed Frog at the basin level may be temporary, e.g., if animals successfully immigrate back into study streams to breed when conditions become more favorable. However, the evaluation of post-harvest effects through Post 14 & 15 is representative of the likely full life span for the species, which is estimated to be as much as 15 to 20 years (Daugherty & Sheldon, 1982).

Similar to our results for the response of Coastal Tailed Frogs to buffer treatments, we had evidence of increased torrent salamander densities only in Post 1 & 2, in this case for all three buffer treatments compared to pre-harvest period after controlling for temporal changes in the reference. However, in Post 7 & 8, there was no longer evidence for increased densities in any buffer treatment, and instead we had evidence of an estimated 58% decline in the FP treatment. In Post 14 & 15, we estimated an 88% and 80% decline in density for the FP and 0% treatments, respectively.

We suspect that the increase in torrent salamander densities we estimated in Post 1 & 2 may have been at least partially attributable to the presence of stream reaches covered by dense accumulations of in-channel slash and windthrow, or wood-obstructed reaches, and the way that we accounted for animal densities in these reaches for our estimates of stream network-wide abundance (McIntyre et al., 2018). In Post 1 & 2, we found high densities of torrent salamanders in wood-obstructed reaches. However, the elevated density we observed for torrent salamanders in these reaches did not persist in Post 7 & 8. In fact, we had evidence of a 34% decline in torrent salamander density in wood-obstructed reaches between Post 1 & 2, and Post 7 & 8 (McIntyre et al., 2021). In Post 14 & 15, wood obstructed reaches were so uncommon across all sites that we

did not even have to adjust our basin-wide counts by densities in these reach types (see Methods).

Contrary to findings for both Coastal Tailed Frog and torrent salamanders, we had no evidence of an increase in stream network-wide giant salamander density for any treatment or in any period compared to pre-harvest period after controlling for temporal changes in the reference. In Post 1 & 2, we found evidence of a 65% decline in giant salamander density in the FP treatment. We estimated a similar 53% decline in the FP treatment in Post 7 & 8. In Post 14 & 15 giant salamander density was estimated to decline by 81% in the FP treatment.

We are aware of limited experimental timber management effectiveness studies that have evaluated the response of stream-associated amphibians to upland timber harvest. In a similar experimental study, Jackson et al. (2007) concluded that clearcut timber harvest without riparian buffers had an immediate negative effect on Coastal Tailed Frog populations, that giant salamanders were sensitive to the immediate impacts of upland harvest but that the negative impacts were short-lived (e.g., three years or less), and that torrent salamanders were not greatly affected by timber harvest. However, that study evaluated only the three years following harvest and study findings for Coastal Tailed Frog were based on limited observations.

In another BACI-designed study in western Washington, O’Connell et al. (2000) observed no difference in larval tailed frog densities among variable width buffers. However, this study only monitored amphibian densities for two years post-harvest. The short-term efforts of many experimental timber harvest studies may be limited in their ability to detect a treatment response for stream-associated amphibians. In fact, had we relied on the results from the first two years post-treatment, we may have erroneously concluded a positive effect of timber harvest for some taxa and buffer treatments. Effects of silvicultural treatments on amphibians, particularly those with relatively long lifespans such as the species included in this study, may not be realized until many years after treatment (Hawkes & Gregory, 2012).

Our research findings are consistent with an increasing body of evidence concluding the negative effects of timber harvest on stream-associated amphibians. We are aware of two experimental studies that monitored stream-associated amphibian response to timber management over a longer period following harvest and had sufficient data from which to draw conclusions. The longest ongoing effort we are aware of is a long-term research effort by Olson and Ares (2022) to evaluate the response of multiple aquatic species to upland timber thinning with variable width no-entry riparian buffers (~ 70 m, 6 m, and a variable-width buffer with a 15-m minimum width) and a wider thin-through managed buffer in eight study sites in western Oregon. Olson and Ares (2022) reported a delayed negative response developing 10 years after upland thinning of second-growth forest, and additional effects five years after a second upland-forest thinning. Five years after the second thinning, higher densities of Coastal Giant Salamander and torrent salamanders (including Cascade and Southern Torrent Salamanders *R. variegatus*) were detected in no-entry ~70 m wide riparian buffers, compared with lower densities for these species in the other buffers (narrower, and thin-through). Unfortunately, a species-specific statistical analysis was not possible for Coastal Tailed Frog as a part of this effort due to low and variable samples sizes (Olson et al., 2014). In another study, Hawkes and Gregory (2012) evaluated tailed frog post-metamorphs and in riparian and upland areas, finding that relative abundance declined in clearcut upland habitats 2- and 10-years following timber harvest.

Our findings are also consistent with several retrospective observational studies that have concluded that tailed frog is less abundant in stands with a history of timber harvest (Ashton et al., 2006; Hawkes & Gregory, 2012; Stoddard & Hayes, 2005; Welsh & Lind, 2002) and another that found that tailed frog occupancy was positively associated with stand age (Kroll et al., 2008). However, other retrospective studies have concluded a lack of effect of clearcut harvest or stand age on Coastal Tailed Frogs (Matsuda & Richardson, 2005; Richardson & Neill, 1998). We cannot say with certainty why the findings from these latter studies differ from our own. However, Richardson and Neill (1998) evaluated occupancy rather than density, so declines in density would not have been noted. Another possibility is that both these studies were conducted in sites located farther north than our study area, in British Columbia, Canada. It is possible that the response of these species to timber harvest varies with latitude, i.e., the species may respond differently depending on the location within its geographic range (Hayes & Quinn, 2015). Associations with old-growth or late-seral stands may be strongest in the southern range of the distribution (Gilbert & Allwine, 1991), a correlation that is likely to be further intensified by climate change.

Not all evaluations have concluded a negative response of stream-associated amphibians to timber harvest. Conclusions from retrospective studies evaluating the impacts of forest management on torrent and giant salamanders have been inconsistent. Several studies concluded that torrent salamanders occur in lower abundances in managed stands compared with old-growth stands (Bury et al., 1991; Corn & Bury, 1989; Russell et al., 2005). However, some researchers detected no relationship between torrent salamander occupancy or relative abundance and stand age (Jackson et al., 2007; Olson et al., 2014). Still others have found that torrent salamander numbers and occupancy were greatest in mid-rotation stands (Russell et al., 2004). In retrospective efforts that included giant salamander populations related to stand age, Ashton et al. (2006) observed an increased relative abundance in streams in late-seral forests, and Kroll et al. (2008) found a positive association between giant salamander occupancy and stand age. Conversely, other evaluations have failed to find a relationship between giant salamander abundance (not adjusted for detection) and stand age (Bury et al., 1991; Leuthold et al., 2012). Finally, others concluded that the response of giant salamanders to timber harvest was site dependent, e.g., populations in low gradient channels being more likely to respond negatively to timber management (Corn & Bury, 1989; Murphy & Hall, 1981). However, unlike experimental BACI studies, retrospective efforts cannot account for historic patterns of occupancy or abundance at study sites, and most retrospective studies have not attempted to account for detection probability in their statistical comparisons. Both considerations should be taken into account when interpreting the findings from retrospective efforts.

To maximize our ability to detect changes in abundance and increase the certainty of our conclusions, we adjusted our counts from light-touch sampling for the probability of detection. This allowed us to control for the possibility that treatment may confound our ability to detect amphibians. Occupancy, density and abundance estimates adjusted for detection can be used to confidently compare populations through time and space (Ficetola et al., 2018; Guillera-Arroita et al., 2014; MacKenzie & Kendall, 2002; Mazerolle et al., 2007; McIntyre et al., 2012), and the statistical methods we used to adjust amphibian density have been validated in other amphibians studies (Chelgren et al., 2011; McKenny et al., 2006; Price et al., 2011). We surveyed study sites with an intensity that surpasses the intensity of sampling in many other similar studies, with a minimum of 50% of the stream channel network sampled.

Nevertheless, low counts for Coastal Tailed Frogs, especially in the 0% treatments, led to wide confidence intervals and numerically unstable model fits. As such, we were unable to account for detection probabilities in our analysis for Coastal Tailed Frog larvae and post-metamorphs.

Despite that issue, the consistency in our study findings across Phase II and Phase III and across larval and post-metamorphic life stages for Coastal Tailed Frog bolsters confidence in our result.

Low counts in later post-harvest years were almost certainly related to decreased densities at these sites. This conclusion was supported by the fact that additional intensive sampling efforts in Post 7 & 8 (i.e., kick-net and nocturnal surveys) designed to increase tailed frog tissue samples for use in genetic and stable isotopes analyses failed to find numbers of frogs that would suggest our systematic sampling was somehow less effective in this later sample period.

Implications of Forest Management Activities

While many research efforts have revealed a positive relationship between stream-associated amphibian populations and stand age (Ashton et al., 2006; Pollett et al., 2010; Stoddard & Hayes, 2005; Welsh et al., 2005; Welsh & Lind, 2002), it is likely that forest age alone does not determine amphibian species' occupancy and abundance. Rather, occupancy and abundance is intrinsically linked to microclimate and microhabitat conditions that tend to vary in relation to forest age (Diller & Wallace, 1994; Welsh, 1990). Amphibians have been associated with stream and riparian conditions including stream temperature, overstory canopy, primary productivity, wood loading, sediment retention, flow dynamics, stream and bank morphology, and nutrients, all metrics that likely impact occupancy and abundance at the microhabitat level.

The mechanistic links between timber harvest and riparian stands, wood loading, channel characteristics, stream temperature and cover, discharge, sediment and nutrients have been well documented in the literature (e.g., Moore et al., 2005; Richardson & Béraud, 2014; Yeung et al., 2017). Results for stream-associated amphibians, however, appear somewhat more complex. This is due in part to the fact that these species are long-lived. As such, response of these species in the short-term would largely reflect movement in or out of study sites. Longer-term impacts reflect the additional influence of timber harvest on reproduction and onsite survival. Our study was designed to evaluate treatment effects, not the mechanisms behind potential changes in amphibian abundance. However, because Phases I and II (McIntyre et al. 2018; 2021) also evaluated changes in stream temperature, overstory canopy, primary productivity, wood loading, sediment retention, flow dynamics, stream and bank morphology, and nutrients, we can consider the mechanisms behind the changes we observed in amphibian densities. For a thorough evaluation of potential relationships between changes in microclimate and microhabitat conditions relative to amphibian response, see McIntyre et al., 2021, Chapter 9 – *Stream-associated Amphibians*.

The relationship between reductions in overstory canopy and stream-associated amphibians is complex. Increased light and stream temperatures have been associated with increased instream primary productivity (Kiffney et al., 2003), which may have beneficial consequences for stream-associated amphibians either directly (for grazing Coastal Tailed Frogs; Kiffney & Richardson, 2001) or indirectly, through increased macroinvertebrate prey availability (Hawkins et al., 1983). Conversely, increased sunlight and/or stream temperature can cause a shift in the species composition of periphyton away from diatoms (Beschta et al., 1987), the primary food source for larval tailed frogs (Altig & Brodie, 1972; Nussbaum et al., 1983), which could have negative

consequences if food availability is limited. As a part of Phase I, we detected no changes in biofilm or periphyton in harvested sites in the post-harvest period (McIntyre et al., 2018, Chapter 13 – *Biofilm and Periphyton*). Consistent with these findings, our analysis of stable isotopes (McIntyre et al., 2021, Chapter 8 – *Stable Isotopes* in this report) failed to find evidence that harvest in the RMZ resulted in a change in the primary energy source supporting food webs in our small streams. Overall, our results are not consistent with findings for larger and wider stream channels where canopy modification increases trophic support from autotrophic sources (Kaylor & Warren, 2017). Based on our lack of evidence for a change in instream primary production in the post-harvest period we do not believe that the stream-associated amphibian response we observed was related to change in periphyton production. However, we did not evaluate periphyton species composition and do not know if the proportion of nutritious diatoms in the periphyton matrix changed as a function of treatment.

All focal amphibians have been found to utilize cool waters or avoid areas with higher stream temperatures (Bury, 2008; de Vlaming & Bury, 1970; Karraker et al., 2006; Pollett et al., 2010). We observed an increase in July–August daily maximum stream temperatures in all buffer treatments relative to the reference (mean increase of as much as 1.1, 1.1 and 3.8°C in the seven-day average daily maximum temperature response for the 100%, FP, and 0% treatments, respectively, across all post-harvest years), and only the 100% treatment did not differ statistically from the reference nine years post-harvest (McIntyre et al., 2021, Chapter 4 – *Stream Temperature and Cover*). The increased stream temperatures we observed in all buffer treatment streams may have affected movement or reproductive success over time, especially for Coastal Tailed Frogs and torrent salamanders.

Treatment-related inputs of wood may have impacted habitat quality by increasing the retention of fine sediments, which can negatively affect amphibian occurrence and density (Diller & Wallace, 1996, 1999; Dupuis & Steventon, 1999; Hawkins et al., 1983; Stoddard & Hayes, 2005; Welsh & Lind, 1996; Welsh & Ollivier, 1998). We observed an increase in fine and sand substrates in all buffer treatments in the Post 7 & 8, though the increase was not statistically significant in the 100% treatment (McIntyre et al., 2021, Chapter 7 – *Stream Channel Characteristics*). Fine sediment can modify grazing surfaces and availability of retreats for Coastal Tailed Frog larvae (Gomi et al., 2001; Hassan et al., 2005; Jackson & Sturm, 2002; Maxa, 2009), which are specialized periphyton grazers that preferentially select smooth, exposed rocks for grazing and daytime retreats (Altig & Brodie, 1972).

Timber harvest may impact stream-associated amphibian movement, stream-network wide or between drainages, altering emigration or immigration (Dupuis & Steventon, 1999; Pollett et al., 2010; Stoddard & Hayes, 2005; Vesely & McComb, 2002). Movement may decline with an increasing density of log jams (Wahbe & Bunnell, 2001). Stream-associated amphibians have been found to resist movement across even relatively small (i.e., 13-m) gaps in stream channel riparian canopy (Cecala et al., 2014), and researcher has shown that post-metamorphic amphibians travel farther away from streams in old-growth forests than in recent clearcuts (Fagan, 2002; Grant et al., 2007). If instream and/or terrestrial environments are unfavorable for movement, isolating amphibian populations or limiting opportunities for immigration by individuals from outside the area, then the population may decline through time.

Notably, however, we had evidence of high levels of gene flow among sites for Coastal Tailed Frogs and Coastal Giant Salamanders in both the pre- and post-harvest periods (Spear et al., 2019). Genetic structure is likely influenced by surrounding basins in addition to site-level treatment effects, providing some support for the hypothesis that site-level declines in densities for these species may be mediated by immigration back into the impacted area over time. However, changes in genetic diversity in response to a disturbance are often not detected until several generations post-impact (Hoban et al., 2013). Furthermore, Cope's Giant Salamander had much more restricted levels of gene flow overall, although there was genetic connectivity among nearby sites. Finally, we did not include the three species of torrent salamanders in our genetic investigation of treatment impacts. However, one genetic study found that the Columbia Torrent Salamander had a more restricted geographic range and significantly lower average within-population genetic diversity than another closely related torrent salamander species and that reduced gene flow reflected habitat fragmentation and inbreeding (Emel et al., 2019).

Pre-treatment occupancy of stream-associated amphibians in our study sites, located within forested stands with a history of prior timber management activities, does provide evidence of continued occupancy of previously harvested stands throughout our study area to date. Occupancy has continued under historic timber harvest practices and continues now, which may cause some readers to speculate that their continued persistence is guaranteed, since current forest practices are more protective than historical practices. However, we do have strong evidence of a decline in amphibian abundance for some species and treatments in Post 7 & 8 that continued Post 14 & 15, with no evidence of recovery for any species in any treatment between those post-harvest periods.

Research supports the conclusion that riparian buffers can ameliorate the impacts of timber harvest on stream-associated amphibians (Jackson et al., 2007; Olson et al., 2014; Russell et al., 2004). However, the ameliorating effects of riparian buffers depend on the size and extent of the riparian buffers (Olson & Ares, 2022). We detected substantial declines in density for Coastal Tailed Frog larvae and post-metamorphs across all buffer configurations evaluated, for torrent salamanders in the FP and 0% treatments, and for giant salamanders in the FP treatment. Considering these results in combination leads us to conclude that even the most protective buffer evaluated in this study (i.e., the 100% buffer) was inadequate to meet the Overall Performance Goals to *not significantly impair the capacity of aquatic habitat to support the long-term viability of other covered species* (i.e., the FP-designated amphibians; FP HCP, Schedule L-1, Appendix N). In their efforts to evaluate riparian buffer effects on stream-associated amphibians, Olson and Ares (2022) found evidence of long-term negative effect with upland forest thinning and variable-width riparian buffers with a minimum 15-m width. Similar to our study, the mechanism behind the change was unclear, however, the authors conclude that either lag-time or cumulative effects of factors associated with treatments were developing many years after harvest (Olson & Ares, 2022).

Recommendations

The broad distribution of our study sites gave us a unique and important opportunity to better understand the impacts of forest management actions on stream-associated amphibians in

occupied basins across western Washington. Coupling our amphibian demographic study with an evaluation of genetic structure (Spear et al., 2011; 2019) allowed us to interpret our basin-scale amphibian responses in context of the larger landscape-scale at which these species appear to operate. Nonetheless, we observed a substantial negative response to timber harvest in Post 7 & 8 that continued through Post 14 & 15. Considering the result of our demographic evaluation in combination with our previous genetic efforts, we believe an effort to evaluate the status of FP-designated amphibians at broader scales throughout western Washington as part of the Extensive Monitoring Program is warranted. The results to date provide evidence of a negative and sustained effect of upland timber harvest on stream-associated amphibians in hard rock lithologies. However, without a landscape effort to evaluate occupancy throughout western Washington we are unable to evaluate the long-term consequences at broader spatial scales. Understanding landscape trends will complement our understanding of FP-designated amphibians at the scale of harvest unit.

Coastal Tailed Frog Extensive Status Project (CMER Work Plan)

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