Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington – Phase 2 (Nine Years after Harvest)

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Cooperative Monitoring
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The Landscape and Wildlife Advisory Group (LWAG) and
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Washington State Forest Practices Board
Forest Practices Adaptive Management Program
Washington State Department of Natural Resources
Olympia, Washington

July 2021 **CMER #2021.07.27**

Washington State Forest Practices Adaptive Management Program

The Washington State Forest Practices Board (FPB) has established an Adaptive Management Program (AMP) by rule in accordance with the Forests & Fish Report (FFR) and subsequent legislation. The purpose of this program is to:

> *Provide science-based recommendations and technical information to assist the FPB in determining if and when it is necessary or advisable to adjust rules and guidance for aquatic resources to achieve resource goals and objectives. The board may also use this program to adjust other rules and guidance.* (Forest Practices Rules, WAC 222-12-045(1)).

To provide the science needed to support adaptive management, the FPB established the Cooperative Monitoring, Evaluation and Research (CMER) committee as a participant in the program. The FPB empowered CMER to conduct research, effectiveness monitoring, and validation monitoring in accordance with WAC 222-12-045 and Board Manual Section 22.

Report Type and Disclaimer

This technical report contains scientific information from research or monitoring studies that are designed to evaluate the effectiveness of the forest practices rules in achieving one or more of the Forest and Fish performance goals, resource objectives, and/or performance targets. The document was prepared for the Cooperative Monitoring, Evaluation and Research Committee (CMER) and was intended to inform and support the Forest Practices Adaptive Management program. The project is part of the Type N Riparian Effectiveness Program and was conducted under the oversight of the Landscape and Wildlife Scientific Advisory Group (LWAG) and the Riparian Scientific Advisory Group (RSAG).

This document was reviewed by CMER and was assessed through the Adaptive Management Program's independent scientific peer review process. CMER has approved this document for distribution as an official CMER document. As a CMER document, CMER is in consensus on the scientific merit of the document. However, any conclusions, interpretations, or recommendations contained within this document are those of the authors and may not reflect the views of all CMER members.

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Full Reference

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EXECUTIVE SUMMARY

Aimee McIntyre, Marc Hayes, William Ehinger, Stephanie Estrella, Reed Ojala-Barbour, Greg Stewart, Dave Schuett-Hames, and Timothy Quinn

Headwater streams are largely understudied relative to their frequency in the landscape, constituting approximately 65% of the total stream length on forestlands in western Washington. We evaluated the effectiveness of riparian forest management prescriptions in maintaining key aquatic conditions and processes affected by Forest Practices for small non-fish-bearing (Type N) headwater stream basins underlain by competent, "hard rock" lithologies (i.e., volcanic or igneous rocks) in western Washington (see Chapter 1 – *Introduction* in this report). We compared current prescriptions to two alternatives, one with longer riparian leave-tree buffers and one with no buffers. We looked at the magnitude, direction (positive or negative), and duration of change for riparian-related inputs and response of instream and downstream components. We evaluated riparian processes affecting in-channel wood recruitment and loading, stream temperature and shade, discharge, suspended sediment export, nutrient export, channel characteristics, and stable isotopes. To evaluate biological response, we selected stream-associated amphibians as a key response variable because they are one of the important biotic resources for protection in non-fish-bearing streams. The results of this study are intended to inform the efficacy of current Forest Practices (FP) rules, including how landowners can continue harvesting wood resources while protecting important headwater habitats and associated species, and meeting resource objectives outlined in the FP Habitat Conservation Plan (FP HCP; Schedule L-1, Appendix N).

We used a Before-After Control-Impact (BACI) study design with blocking to examine how key aquatic resources, conditions and processes responded to riparian buffer treatments. We collected a minimum of two years of pre-harvest data from 2006 until harvest began in 2008, and post-harvest data from 2009 (one year post-harvest) through 2016 or 2017, depending on the response variable (i.e., up to nine years post-harvest; see Chapter 2 – *Study Design* in this report). Study sites included 17 Type N stream basins located in managed second-growth conifer forests across western Washington in three physiographic regions (Olympic Mountains, Willapa Hills and Southern Cascades). Sites were restricted to Type N basins ranging from 12 to 54 ha (30 to 133 ac) underlain by relatively competent lithologies, primarily volcanic flow rocks and breccias, and that were known to support Coastal Tailed Frog (*Ascaphus truei*) and Olympic, Columbia, or Cascade Torrent Salamanders (*Rhyacotriton olympicus*, *R. kezeri*, or *R. cascadae*).

We evaluated four experimental treatments, including an unharvested **Reference** (i.e., withheld from harvest; n = 6) and three alternative riparian buffer treatments with clearcut harvest: **100% treatment** (a two-sided 50-ft [15.2-m] riparian buffer along the entire Riparian Management Zone [RMZ; n = 4]); **FP treatment** (a two-sided 50-ft [15.2-m] riparian buffer along at least 50% of the RMZ, consistent with the current Forest Practices buffer prescription for Type N streams [n = 3]); and **0% treatment** (clearcut harvest throughout the entire RMZ [n = 4]). The timber harvests and associated riparian buffer treatments were implemented between October 2008 and August 2009.

In-channel wood plays an important functional role in Pacific Northwest streams, influencing channel morphology and hydraulics, storage and routing of sediment and organic matter, aquatic habitat, aquatic communities, and food resources. We found that harvest of timber in and adjacent to streamside riparian forests directly affected tree mortality, tree fall rates, wood recruitment to streams, and in-channel wood loading (pieces/m; see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report). The greatest post-harvest change in stand structure occurred in the 0% treatment and the unbuffered portions of the FP treatment where all riparian trees were removed during harvest. A pulse of large wood (≥10 cm [4 in] diameter) was recruited to streams adjacent to unbuffered reaches during harvest, with very little additional large wood recruitment in the following eight years. Windthrow was the dominant tree mortality agent in riparian buffers, with the highest mortality in the first two years post-harvest. The highest tree mortality rates and greatest reductions in density and basal area occurred in the FP treatment, where cumulative mortality eight years post-harvest was 51% and 56% of initial basal area and 50% and 68% of initial density for the RMZ and PIP buffers, respectively, compared to a cumulative mortality of 16% and 9% of basal area and 20% and 15% of initial density for the RMZ and PIP in the reference. Windthrow-associated tree fall in riparian buffers increased large wood recruitment to channels in the 100% and FP treatments. However, we found that most recruited trees (>80%) were suspended above the active stream channel. From two to eight years post-harvest, in-channel large wood loading (mean pieces per linear stream meter) continued to increase in the FP treatment, remained relatively stable in the 100% treatment, and decreased in the 0% treatment. The greatest increase in in-channel small wood loading (<10 cm [4 in] diameter) was in the 0% treatment in the two years post-harvest, and was comprised largely of logging slash from timber harvest of the streamside trees. Small wood loading continued to increase in the FP and 0% treatments through five years post-harvest and then declined in all buffer treatments.

Riparian vegetation is an important source of organic matter and macroinvertebrates, nutrients, and cool water to downstream reaches. The shading that this vegetation provides is a dominant control on stream temperature, which in turn is a critical environmental condition for many aquatic organisms and biological processes. Riparian shade decreased and water temperature increased in all buffer treatments after harvest (see Chapter 4 – *Stream Temperature and Cover* in this report). Canopy closure decreased by less than 10% in the 100% treatment but declined 32% in the FP treatment and 87% in the 0% treatment by three years post-harvest. After nine years, canopy closure returned to pre-harvest levels in the 100% treatment, but remained 15% and 27% below pre-harvest values at the FP and 0% treatments, respectively. The seven-day average temperature response increased in the 100% treatment by 1.1°C in the year immediately following harvest but returned to pre-harvest temperatures in the three years post-harvest. In the FP treatment, the temperature response ranged from +0.5 to +1.2°C and changed little throughout the post-harvest period, possibly from the ongoing loss of buffer trees to windthrow. In the 0% treatment, the temperature response was nearly 4°C in the first year after harvest but then steadily declined to a 0.8°C increase by nine years post-harvest. The greatest change in temperature occurred during the July–August period, but temperatures were also elevated in spring and fall at most locations. Substantial (>1.0°C) temperature responses within the harvest unit were attenuated downstream of the harvest unit where the stream had flowed through 100 m or more of unharvested forest or buffers wider than 50 ft. The primary driver of post-harvest temperature increases appeared to be loss of riparian cover. However, there was evidence that basin aspect may have influenced the magnitude of change; and in one locality, hyporheic flow may have mitigated higher temperature within a well-shaded downstream reach.

Changes in stream discharge, and in the sediment loads that are carried by those flows, have been long recognized as common, but highly variable, responses to timber harvest. We measured discharge and suspended sediment export in eight of the study sites, with four sites each (one of each buffer treatment) in the Olympic and Willapa Hills physiographic regions. Total water yield increased in all buffer treatment sites, but treatment effects varied with buffer treatment and climate, with sites receiving the most rainfall (i.e., Olympic block) and the greatest proportion of watershed area harvested (FP and 0% treatments) exhibiting the largest increases (see Chapter 5 – *Stream Discharge, Turbidity, and Suspended Sediment Export* in this report). On average, this study affirms prior literature reports that show discharges increasing 1 to 18 mm/yr for each percent of the watershed harvested, albeit with much variability as a function of buffer treatment, climate, and precipitation. Relative changes in flow were greatest for baseflows and median discharge, but specific discharges increased for all flows up to the 30-day recurrence interval in the FP and 0% treatment sites. Late summer discharge decreased in both 100% treatment sites through eight years post-harvest, presumably because of increased evapotranspiration rates in the residual vegetation during periods of little rainfall. In contrast, harvest did not change the magnitude of suspended sediment export regardless of buffer treatment. Over 11 years of the study, turbidity readings were very low over 95% of the time. Both turbidity and suspended sediment concentration increased with increasing discharge, typically during late fall and early winter storm events, and then rapidly declined. The basins appear to be supply-limited, with the quantity of exported sediment restricted to the quantity of sediment delivered to the stream from the adjacent uplands, and so additional flow (especially non-peak flow) has little ability to affect sediment transport. While discharge increased in all treatments after harvest, suspended sediment export events were episodic, poorly correlated with discharge, and not synchronized across all sites, suggesting that export magnitudes are unrelated to harvest.

Nutrients exported to downstream receiving waters may increase primary productivity, leading to decreases in instream dissolved oxygen from decomposition. Because the watersheds of western Washington drain to the sensitive, confined marine waters of Puget Sound, Grays Harbor, and Willapa Bay, nutrient loading is a potential environmental concern. We measured mean total nitrogen (N) and nitrate-N concentrations for nine years post-harvest in the same eight sites used in our discharge and suspended sediment export components of the study (see Chapter 6 – *Nitrogen Export* in this report). Nitrogen export increased in all treatment sites immediately after harvest, with variable increases among sites ranging from less than 10% to more than three-fold. The estimated change was greatest in the 0% treatment, intermediate in the FP treatment, and lowest in the 100% treatment. Controlling for treatment type, the increases corresponded to the proportion of the watershed harvested. Only the 0% treatment differed statistically from the other treatments. At seven and eight years post-harvest, the eight sites displayed no consistent response in nitrogen concentration or export to the buffer treatments or to the proportion of the watershed harvested: total-N export had declined from their immediate post-harvest levels at three sites and increased slightly at three sites, while nitrate-N export declined from post-harvest levels at four sites and increased slightly at two sites. Only one site, however, had recovered to pre-harvest export rates by eight years post-harvest.

Changes to wood recruitment and loading, stream flow and sediment transport from timber harvest can result in changes to physical stream channel characteristics, particularly channel dimensions and substrate sediment materials. Measurements in the 17 study basins in the years immediately pre-harvest, and again one, two, five and eight years post-harvest, showed some systematic patterns (see Chapter 7 – *Stream Channel Characteristics* in this report). In the two years post-harvest, we estimated a decline in stream wetted and bankfull widths in the 0% treatment compared to the pre-harvest period after controlling for temporal changes in the reference. This pattern continued through eight years post-harvest. We also measured a post-harvest increase in the proportion of the stream channel dominated by fine sediment substrates in the 0% treatment in the two years post-harvest, which was still evident eight years post-harvest. A similar increase was also estimated for the FP treatment eight years post-harvest, but not in the other sample years. Finally, we estimated an increase in the proportion of the channel rise attributed to in-channel steps in the 0% treatment in all post-harvest sampling years. We suspect that post-harvest changes in stream channel characteristics, primarily in the 0% treatment, can be attributed at least in part to post-harvest increases in in-channel small and large wood recruitment and loading. Wood recruitment and loading increased in all buffer treatments as logging slash and windthrown trees from unharvested RMZs entered the stream channel during and after timber harvest. This may explain the decrease we observed in stream wetted and bankfull widths, despite the observed increase in flows that would normally encourage increases in channel width.

Stable isotope ratios are especially useful for identifying shifts in trophic system organization due to canopy modification, which other researchers have associated with an increase in the contribution of algae to the trophic support of streams. Samples of biofilm, litterfall, instream detritus, macroinvertebrates, and amphibian tissue were collected during the pre-harvest period and one, two and eight years post-harvest (see Chapter 8 – *Stable Isotopes* in this report). Not every group was sampled for all sites in all years (see Tables 8-1 and 8-2 for sample sizes). We found limited and inconsistent differences in carbon (^{13}C) and nitrogen (^{15}N) isotopic signatures among treatments; however, stable isotope signals suggested that the organic matter sources supporting stream biofilm did not appreciably change in response to the buffer treatments. Though we did not detect a notable difference in the biofilm isotopic values between the pre- and post-harvest period, we estimated a decrease in mean $\delta^{13}C$ for giant salamander larvae in the FP treatment and an increase in the 100% treatment in the two years post-harvest. Over this same period, we estimated a decrease in mean $\delta^{15}N$ for gatherer invertebrates in the FP and 0% treatments. We found no evidence of an increase in algal content in the biofilm, thus challenging the hypothesis that canopy modification might increase trophic support from autotrophic sources. Our $\delta^{13}C$ versus $\delta^{15}N$ comparison of stable isotope data were also used to characterize stream-associated amphibian diet. Results indicated that Coastal Tailed Frog larvae were ingesting primarily biofilm. The post-metamorphic Coastal Tailed Frogs, torrent salamanders and giant salamanders, however, all exhibited stable isotope values that suggested a diet of aquatic predators and shredders, and terrestrial spiders.

Amphibians have experienced declines in local abundance and range contractions as a result of habitat loss and degradation, disease, and competition with introduced species. Stream-associated amphibians are frequently the dominant vertebrates in and along non-fish-bearing headwater streams. We observed the largest post-harvest response for Coastal Tailed Frog (see Chapter 9 – *Stream-associated Amphibians* in this report). In the two years post-harvest we estimated an increase in larval Coastal Tailed Frog density in the FP treatment compared to the pre-harvest period, after controlling for temporal changes in the reference; however, by eight years post-harvest we estimated substantial declines in larval density in all buffer treatments. In the two years post-harvest, post-metamorphic tailed frog density declined in the 100% treatment but increased in the 0% treatment. However, by eight years post-harvest we again estimated substantial declines in density in the 100% and FP treatments, whereas the change in density in the 0% treatment no longer differed from that of the reference. We estimated an increase in torrent salamander density in the 0% treatment in the two years post-harvest; by eight years post-harvest this increase was no longer evident in the 0% treatment although we estimated a decline in the FP treatment. Finally, for giant salamanders we estimated an initial decline in density in the FP treatment in the two years post-harvest, however, by eight years post-harvest we had no evidence of a difference for any treatment. Our study was designed to evaluate treatment effects, not the mechanisms behind potential changes in amphibian abundance. However, stream temperature, overstory canopy, wood loading, sediment retention, flow dynamics, stream morphology, and nutrients all have been associated with amphibian abundance, and changes documented in these metrics following timber harvest likely explain some or all of the changes in amphibian abundances.

In summary, the greatest effects of alternative buffer treatments were observed in riparian stand condition, large wood recruitment and in-channel wood loading, stream shade and temperature, stream channel characteristics, and stream-associated amphibian densities (see *Appendix A* in this report). The 100% treatment was generally the most effective in minimizing changes from pre-harvest conditions, the FP was intermediate, and the 0% treatment was least effective. The collective effects of timber harvest were most apparent in the 0% treatment in the two years immediately post-harvest. For many metrics, the magnitude of harvest-related change observed for a given treatment diminished over time. The one clear exception to this generality was for the stream-associated amphibians. For these species, treatment effects were largely not evident in the two years post-harvest except for declines in giant salamander density in the FP treatment. However, substantial negative declines were recorded for Coastal Tailed Frog density in the eight years post-harvest, including for larvae in all buffer treatments and post-metamorphic individuals in the 100% and FP treatments. We also estimated a decline in torrent salamander density in the FP treatment in the eight years post-harvest. Continued monitoring of the amphibian response to treatment is strongly recommended to expand on our understanding of the long-term impacts of timber harvest and variable length buffers on stream-associated amphibians.

CHAPTER 1 - INTRODUCTION

Aimee McIntyre, Marc Hayes, William Ehinger, Stephanie Estrella, Reed Ojala-Barbour, Greg Stewart, Dave Schuett-Hames, and Timothy Quinn

1-1. 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.

Few studies had addressed the efficacy of riparian buffers along non-fish-bearing, perennial “headwater” streams (or Type Np Waters) at the time of Forest Practices negotiations. However, these small streams comprise more than 65% of the total stream length on forestlands in western Washington (Rogers and Cooke 2007). Furthermore, existing studies tended to be retrospective (e.g., Bisson *et al.* 2002; Raphael *et al.* 2002) or lack the power needed to fully inform Forest Practices for aquatic resources of interest (e.g., O'Connell *et al.* 2000; Jackson *et al.* 2001). 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 2005, 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

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[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.

from the Forest Practices Habitat Conservation Plan (FPHCP; WADNR 2005, Appendix N). In 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. Post-harvest data were collected for up to nine years following harvest. Post-harvest sampling frequency and duration depended on the response variable. Results comparing the response among treatments up to three years following harvest were reported in McIntyre and colleagues (2018).

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. Continued monitoring 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 through nine years post-harvest are reported herein.

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CHAPTER 2 - STUDY DESIGN

Aimee McIntyre, Marc Hayes, William Ehinger, Stephanie Estrella, Reed Ojala-Barbour, Greg Stewart, Dave Schuett-Hames, and Timothy Quinn

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2-1. RESOURCE OBJECTIVES AND RESPONSE VARIABLES

We designed this study to evaluate whether current westside riparian management zone (RMZ) prescriptions for non-fish-bearing, perennial (Type Np) waters maintained key aquatic conditions and process affected by Washington State Forest Practices. Resource Objectives for watershed functions affected by forest practices include:

1) **Heat/Water Temperature:** Provide cool water by maintaining shade, groundwater temperature, flow, and other watershed processes controlling stream temperature.

2) **Large Wood/Organic Inputs:** Develop riparian conditions that provide complex habitats for recruiting large wood and litter.

3) **Sediment:** Provide clean water and substrate and maintain channel-forming processes by minimizing to the maximum extent practicable the delivery of management-induced coarse and fine sediment to streams (including timing and quantity) by protecting stream bank integrity, providing vegetative filtering, protecting unstable slopes, and preventing the routing of sediment to streams.

4) **Hydrology:** Maintain surface and groundwater hydrologic regimes (magnitude, frequency, timing, and routing of stream flows) by disconnecting road drainage from the stream network, preventing increases in peak flows causing scour, and maintaining the hydrologic continuity of wetlands.

Selected response variables were related to Resource Objectives and included riparian vegetation, wood recruitment and loading, stream temperature, discharge, nutrient export, litterfall and detritus export, turbidity, channel characteristics, periphyton, macroinvertebrate export, stream-associated amphibian demographics, downstream fish, and stable isotopes (**Table 2-4**). Stream-associated amphibians were selected as a key response variable because stakeholder scientists that participated in negotiations leading to the development of current Forest Practices rules identified them as one of the important biotic resources for protection in non-fish-bearing (Type N) Waters (USFWS 1999).

2-2. SITE SELECTION CRITERIA

The inclusion of stream-associated amphibian species as a response variable placed important constraints on site selection (**Table 2-1**). 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*).[1] Although Coastal (*Dicamptodon tenebrosus*) and Cope's (*D. copei*) Giant Salamanders are not FP-designated amphibians, they were included in the study for two reasons: (1) they co-occur with designated species throughout the study area; and (2) Cope's Giant Salamander, along with the Coastal Tailed Frog, occurs throughout the study area and was appropriate for evaluating amphibian genetic responses (Spear *et al.* 2019). The site selection process is outlined in detail in McIntyre and colleagues (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 and 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 and 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, sites were limited to basins within that range.[2] 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 response[3] 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.

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[1] The remaining three Forest Practices-designated amphibians not covered in our study include the Rocky Mountain Tailed Frog (*A. montanus*), and Dunn's (*Plethodon dunni*) 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.

[2] Unless an exception is granted after review by an interdisciplinary science team.

[3] Downstream fish response was only included through the two years following harvest. Results are reported in the previously published report, McIntyre and colleagues 2018.

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 (**Table 2-1**). 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.

2-3. 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 2-1**):

1) **Reference** (REF, n = 6): unharvested reference with no timber harvest activities within the entire study site during the study period,

2) **100% treatment** (100%, n = 4): 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, n = 3): clearcut harvest with current Forest Practices no-harvest riparian leave-tree buffer (i.e., two-sided 50-ft [15.2-m]) along ≥50% of the RMZ, and

4) **0% treatment** (0%, n = 4): clearcut harvest with no riparian leave-tree buffer retained within the RMZ.

Table 2-1. 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)
	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

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-slope[4] and headwall[5] seeps, headwater springs[6], Type Np intersections[7] and

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[4] 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.

[5] 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.

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

[7] The intersection of two or more Type Np Waters.

alluvial fans[8]. 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). We created maps displaying Type Np and Ns Waters and locations of sensitive sites, headwalls, and road crossings (**Appendix 2A**; **Appendix Figures 2-1** through **2-16**).

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% and 73%. 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, the 100% treatments in the Olympic and Willapa 2 blocks, and the 0% treatments in the Willapa 2 and South Cascade blocks; **Table 2-2**).

2-4. 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 2-2**). 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 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 2-2**).

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

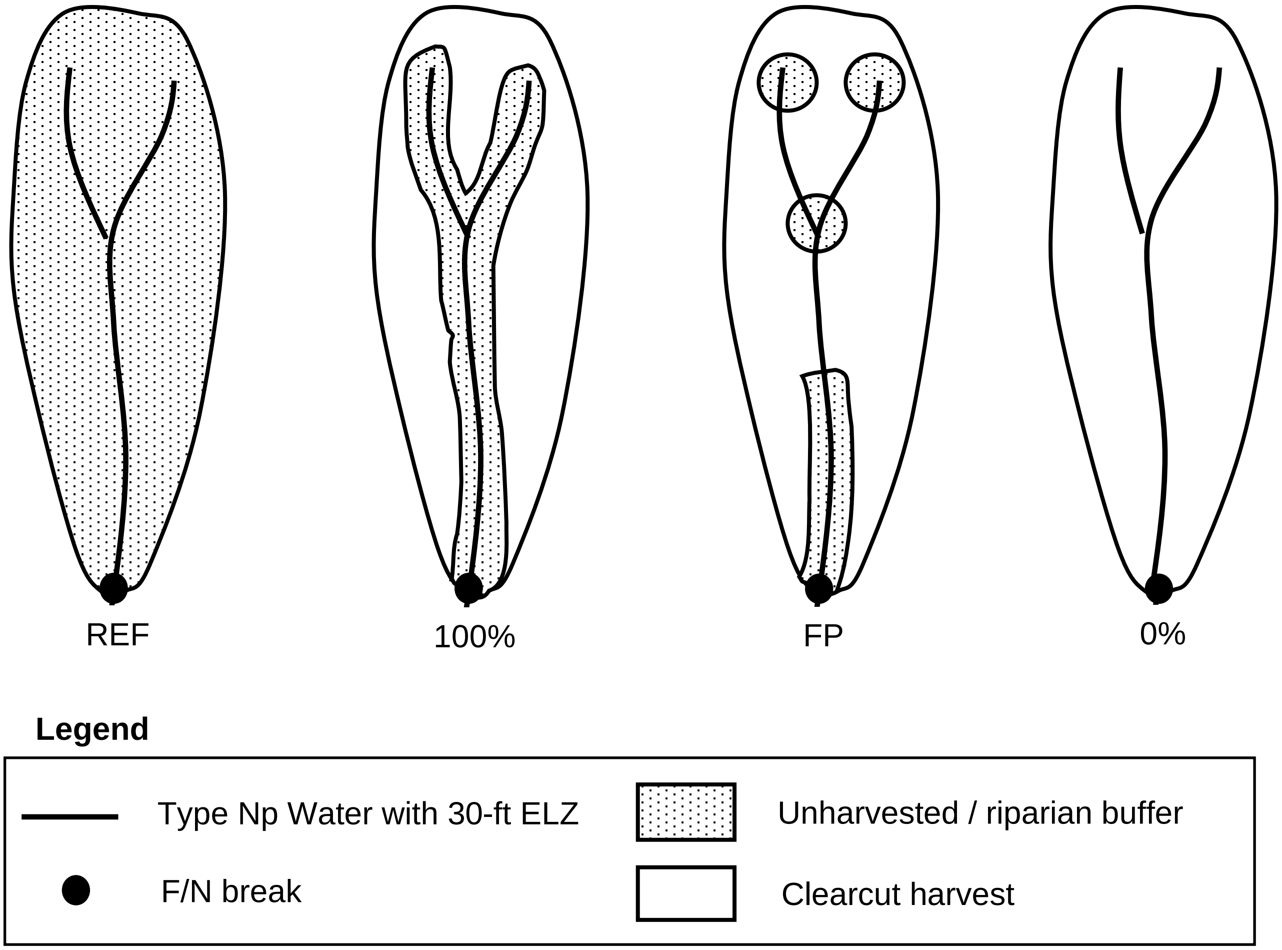

Figure 2-1. Schematic of the four experimental treatments included in the Hard Rock Study. Treatments included unharvested references (REF) and sites receiving a clearcut harvest with one of three, two-sided 50-ft (15.2 m) buffer treatments along the Type Np Water riparian management zone (RMZ): 100% of the stream length buffered (100%), ≥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. Assignment of treatments was random when possible; however, 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. In addition, physical constraints (including a lack of suitable low-gradient reaches for flume installation and/or inaccessibility due to snow in winter and spring) limited measurement of downstream exports to eight of the sites. Study site codes are based on the geographic block and treatment (**Table 2-2**).

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

1) **Olympic:** Treatments were randomly assigned to the four sites in this physiographic region, yielding a single block (OLYM). All four sites were suitable for assessing export variables.

2) **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. 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 responses (i.e., they had the required 75 m downstream reach necessary for evaluating fish response, which was included only in the evaluation through Post 2, see McIntyre et al. 2018). To ensure one complete block representative of all treatments was available to evaluate the downstream fish response, we assigned treatments to sites as follows. First, the site on private land was assigned a buffer treatment. Of the four state-owned sites, two were randomly chosen as unharvested reference sites and randomly assigned to one of the two coastal Willapa Blocks, Willapa 1 (WIL1) and Willapa 2 (WIL2). 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 WIL1 block. All sites in WIL1 were suitable for assessing export variables.

   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 in the WIL2 block. Due to unfavorable economic conditions, harvest of the FP treatment site in the WIL2 block was postponed, so it served as a second reference in this block.[9] None of these sites were included in the assessment of export variables or downstream fish response.

   The two sites located south and east of the eight coastal sites were grouped as the Willapa 3 (WIL3) block. One was assigned the reference treatment due to biological constraints (presence of marbled murrelet habitat); the other was assigned the 100% buffer treatment due to slope instability. Neither was included in the assessment of export variables.

3) **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%. None of the sites in this block were included in the assessment of export variables.

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[9] This second WIL2 reference site was harvested as an FP treatment in 2016 between the Post 7 and Post 8 sample years, after which it was treated differently in analyses depending on the response metric. See individual chapters for details.

Ultimately, 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 2-3**). 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.

Table 2-2. 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. Mainstem Length is the length of the mainstem tributary. Bankfull Width is the mean of the mainstem channel in the pre-harvest period.

Block	Treatment	Study Site Codes	Basin Area (ha [ac])	Type Np Length (m [ft])	Mainstem Length (m [ft])	Elevation (m [ft])	Stream Gradient (%)	Lithology	Bankfull Width (m [ft])	Aspect
Olympic	Reference	OLYM-REF	54 (133)	2,737 (8,980)	1387 (4,551)	163 (535)	18	Basalt flows and flow breccias	2.6 (8.5)	N
	100%	OLYM-100%	28 (68)	1,949 (6,394)	689 (2,260)	72 (236)	27	Tectonic breccia	2.0 (6.6)	NE
	Forest Practices	OLYM-FP	17 (41)	1,070 (3,510)	223 (7,32)	277 (909)	25	Basalt flows and flow breccias	1.0 (3.3)	SE
	0%	OLYM-0%	13 (32)	637 (2,090)	323 (1,060)	233 (764)	31	Basalt flows and flow breccias	1.6 (5.2)	W
Willapa 1	Reference	WIL1-REF	12 (30)	589 (1,932)	467 (1,532)	200 (656)	19	Basalt flows and flow breccias	1.3 (4.3)	SW
	100%	WIL1-100%	31 (76)	1,029 (3,376)	564 (1,850)	198 (650)	18	Basalt flows and flow breccias	1.9 (6.2)	SW
	Forest Practices	WIL1-FP	15 (37)	325 (1,066)	325 (1,066)	197 (646)	19	Basalt flows and flow breccias	1.3 (4.3)	SW
	0%	WIL1-0%	28 (69)	1,525 (5,003)	524 (1,719)	87 (285)	16	Terraced deposits	1.9 (6.2)	NE
Willapa 2	Reference 1[1]	WIL2-REF1	19 (48)	653 (2,142)	636 (2,087)	183 (600)	34	Basalt flows and flow breccias	1.9 (6.2)	W
	Reference 2	WIL2-REF2	16 (41)	816 (2,677)	375 (1,230)	228 (748)	18	Basalt flows and flow breccias	1.2 (3.9)	SE
	100%	WIL2-100%	26 (65)	1,257 (4,124)	797 (2615)	22 (72)	21	Basalt flows and flow breccias	1.8 (5.9)	SW

Block	**Treatment**	**Study Site Codes**	**Basin Area (ha [ac])**	**Type Np Length (m [ft])**	**Mainstem Length (m [ft])**	**Elevation (m [ft])**	**Stream Gradient (%)**	**Lithology**	**Bankfull Width (m [ft])**	**Aspect**
	0%	WIL2-0%	17 (42)	933 (3,061)	745 (2,444)	159 (522)	21	Basalt flows	2.4 (7.9)	E
Willapa 3	Reference	WIL3-REF	37 (92)	2,513 (8,245)	1,342 (4,403)	241 (791)	14	Basalt flows	1.7 (5.6)	SW
	100%	WIL3-100%	23 (58)	1,359 (4,459)	984 (3,228)	351 (1,152)	19	Basalt flows	2.1 (6.9)	SE
South Cascade	Reference	CASC-REF	50 (122)	1,080 (3,543)	800 (2,625)	601 (1,972)	21	Tuffs and tuff breccias	2.0 (6.6)	N
	Forest Practices	CASC-FP	26 (64)	822 (2,697)	526 (1,726)	450 (1,476)	16	Andesite flows	1.5 (4.9)	E
	0%	CASC-0%	14 (36)	420 (1,378)	420 (1,378)	438 (1,437)	29	Andesite flows	1.7 (5.6)	SE

[1] WIL2-REF1 was originally assigned the FP treatment, but harvest was delayed. It was used as a second reference in the Willapa 2 block until its harvest in 2016. Subsequently, it was not included in analyses of most responses, except for stream temperature and cover (see **Section 2-5**. Study Timeline in this report).

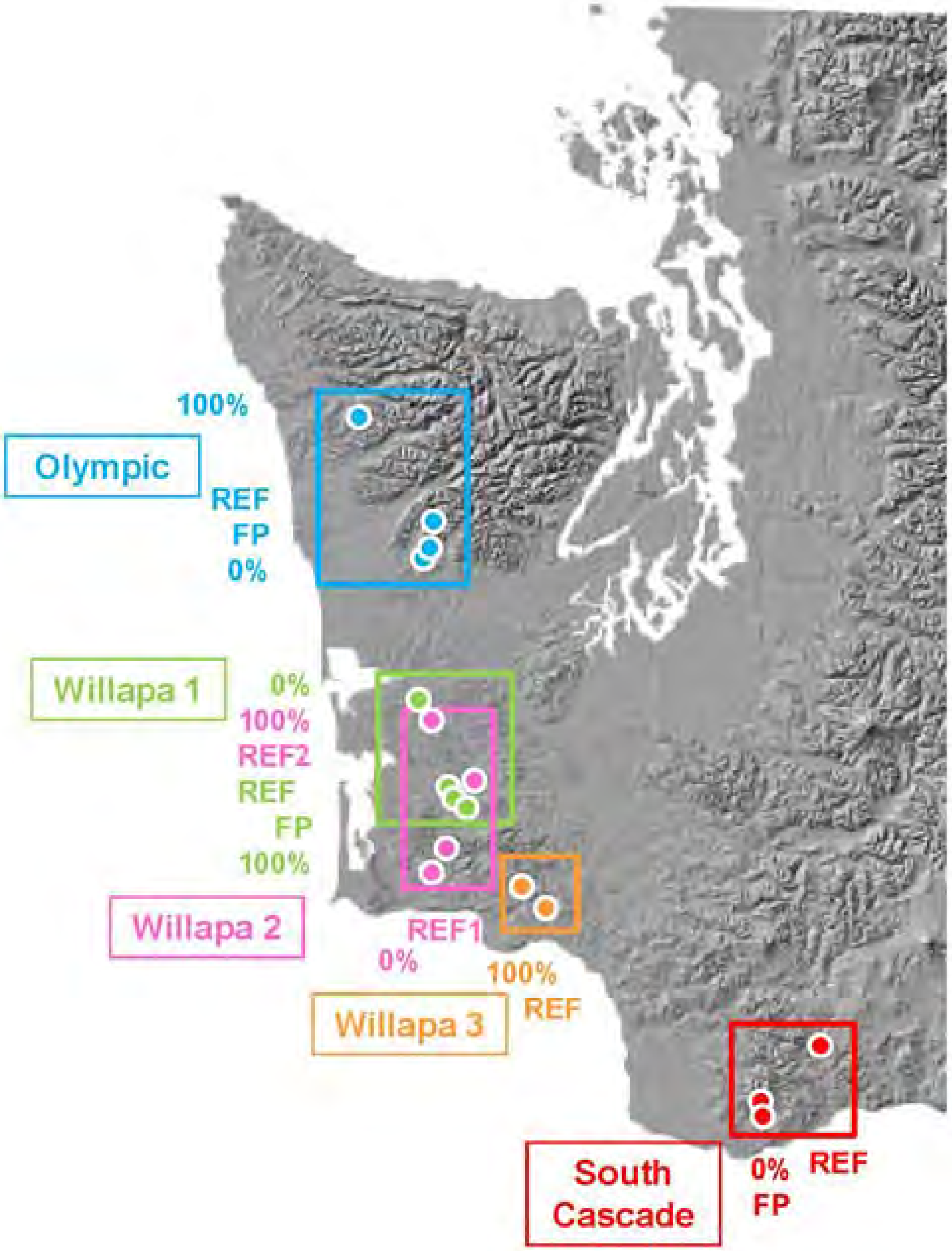

Figure 2-2. Distribution of study sites and treatments for the Hard Rock Study, 2006–2017. Sites are grouped (blocked) geographically (color coded). REF is the reference treatment (unharvested control) and 100%, FP, and 0% are the 100%, Forest Practices (≥50%) and 0% riparian buffer treatments, respectively.

Table 2-3. Distribution of reference and treatment sites included in the Hard Rock Study across landowners by block. The Willapa 2 block had two references and was missing an FP treatment until 2016 (see **Section 2-5**. Study Timeline); the Willapa 3 block only had a reference and a 100% treatment; and the CASC block did not have a 100% treatment.

Block	REF	100%	FP	0%
OLYM	National Forest	State	Private	Private
WIL1	State	State	State	Private
WIL2	State (1) & Private (1)	Private	NA	State
WIL3	State	State	NA	NA
CASC	National Forest	NA	State	State

2-5. STUDY TIMELINE

Pre-harvest sampling began in 2006 and continued through 2008 (i.e., Pre 3, Pre, 2, and Pre 1), although data were not collected for all response variables in all pre-harvest years (**Table 2-4**). Harvest timing and duration varied among study sites. Harvest of the first site began in July 2008 and harvest of the last site was completed in August 2009. Average duration of harvest was four months (see McIntyre *et al.* 2018, Chapter 3 – *Management Prescriptions*). Post-harvest sampling began in 2009 (Post 1) and continued for two to nine years depending on the response variable and timing of harvest. Some metrics, including stream temperature, discharge, and turbidity, were monitored continuously, while others were sampled in discrete periods (**Table 2-4**). Pre- and post-harvest years for stream temperature, discharge, turbidity, and nutrient export depended on the harvest dates and varied among sites. Litterfall input and detritus export, biofilm and periphyton, macroinvertebrate export, and downstream fish were only evaluated for two years following harvest. These two-year responses are discussed in McIntyre and colleagues (2018).

The WIL2-REF1 site was originally assigned the FP treatment, but harvest was delayed until January 2016, between the Post 7 and Post 8 sample years. Consequently, we included it as a reference for most response variables, and did not include it in its post-harvest state during analysis. After 2016, we included it as a fourth FP treatment (WIL2-FP) for stream temperature and cover, but only for two years of post-harvest response. This decision reflected the desire for a more balanced design (four replicates of each buffer treatment) and more information on response of stream temperature to current Forest Practices rules. This substitution was possible because we were able to collect stream temperature continuously for a full two years after harvest of the site. However, this was not possible for other responses for which post-harvest data collection would have been limited to a single year.

Table 2-4. Timing and duration of measurements for each response variable included in the Hard Rock Study for pre-harvest and post-harvest sample years. Shading represents years in which the response variable was sampled. Variables in italics (litterfall input and detritus export, biofilm and periphyton, macroinvertebrate export, downstream fish, and amphibian genetics) were only evaluated for two years after harvest. Results for these variables are presented in McIntyre and colleagues (2018). The number of sites and blocks and the timing for sampling for the Stable Isotopes response was variable; see Chapter 8 – *Stable Isotopes* in this report.

				Pre-harvest Year				**Post-harvest Year**								
Response Variable	**# of Sites**	**# of Blocks**	**Sample Months**	**Pre 3**	**Pre 2**	**Pre 1**		**Post 1**	**Post 2**	**Post 3**	**Post 4**	**Post 5**	**Post 6**	**Post 7**	**Post 8**	**Post 9**
Stand Structure & Tree Mortality	17	5	Apr-Sep		▨	▨	**Harvest**	▨	▨			▨			▨	
Large Wood Recruitment	17	5	May-Sep		▨	▨		▨	▨			▨			▨	
Wood Loading & Cover	17	5	Apr-Jun	▨	▨			▨	▨			▨			▨	
Stream Temp & Cover	17	5	Year round		▨	▨		▨	▨	▨	▨	▨	▨	▨	▨	▨
Discharge & Turbidity	8	2	Year round		▨	▨		▨	▨	▨	▨	▨	▨	▨	▨	
Nutrient Export	8	2	Year round		▨	▨		▨	▨					▨	▨	
Stream Channel Characteristics	17	5	Apr-Jun	▨	▨	▨		▨	▨			▨			▨	
Litterfall Input & Detritus Export	8	2	Year round		▨	▨		▨	▨							
Biofilm & Periphyton	17	5	Jun-Sep	▨	▨	▨		▨	▨							
Macroinvertebrate Export	8	2	Year round	▨	▨	▨		▨	▨							
Stable Isotopes	*	*	*	▨	▨	▨		▨	▨						▨	
Amphibian Demographics	17	5	Jul-Oct	▨	▨	▨		▨	▨					▨	▨	
Amphibian Genetics	17	5	Jul-Nov		▨	▨								▨	▨	
Downstream Fish	6	2	Jul & Oct	▨	▨	▨		▨	▨							

* Number of sites, blocks and sample timing depended on sample period and group (e.g., biofilm, macroinvertebrate group, stream-associated amphibian)

2-6. 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 11 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.

During the pre-harvest sample years, a series of storms occurring 1-4 December 2007 caused extensive windthrow (i.e., trees were uprooted or sustained severe trunk damage) throughout western Washington. These storms resulted in significant damage to forestlands along the Washington coast from Naselle to north of Hoquiam. To ensure that our pre-harvest data reflected the range of disturbances across study sites, we added additional sampling in 2008, prior to harvest. We assessed the extent and severity of windthrow at all sites based on aerial photos taken in March and April 2008 and on field data (counts of downed trees within the bankfull channel; see McIntyre *et al.* 2018, **Chapter 4** – *Unanticipated Disturbance Events* for methods related to these two evaluations).

Field data were consistent with interpretation of the aerial photographs. Both indicated that study sites with the most windthrow were in the Willapa 1 and Willapa 2 blocks. Aerial photos indicated major damage in all sites in the Willapa 1 block and two of four sites in the Willapa 2 block (**Table 2-5**). Photos were not available for the two sites in the Willapa 3 block. Field assessments indicated the greatest damage in the WIL1-REF and WIL1-100% sites based on the number of downed trees over the stream channel and per 10 m of stream length (**Table 2-5**). However, WIL1-FP had a greater proportion of the Type Np Water length impacted than did WIL1-100%.

Regardless of the metric, among blocks windthrow severity was greatest in sites located in the Willapa 1 and 2 blocks, comparatively moderate in the Olympic block and minimal in the Willapa 3 and South Cascade blocks. Severity of windthrow was more similar among sites within than between blocks. Although pre-harvest structural variation among blocks incorporates this variability in wind damage, analyses and interpretations—especially for stand structure and wood—require careful consideration of the timing, severity, distribution of damage among and within blocks. We discuss the possible consequences of pre-treatment disturbance for each response variable where appropriate.

Although the most catastrophic wind storms during this study occurred prior to treatment, the area also experienced storm- and hurricane-force winds (e.g., 55-73.9 mph and >74 mph, respectively) after treatment (**Table 2-6**).

Table 2-5. Results of field and aerial photo evaluations of windthrow for study sites included in the Hard Rock Study. Field data include total number of downed trees along the entire Type Np Water length (Downed Trees), proportion of Type Np Water length with one or more downed trees (Type Np Water Impacted), and average number of downed trees per 10-m stream interval (Downed Trees/10 m). The aerial photo evaluation included proportion of study site (from F/N break) and flume area (from flow monitoring equipment, when applicable). Aerial photos were not available for WIL3-REF or WIL3-100% sites.

		Field Evaluation			Aerial Photo Evaluation	
Block	**Treatment**	**Downed Trees**	**Type Np Water Impacted**	**Downed Trees/10 m**	**Total Area Impacted**	**Flume Area Impacted**
OLY	REF	109	0.15	0.34	0.00	0.00
	100%	116	0.18	0.56	0.00	0.00
	FP	86	0.17	0.74	0.00	0.00
	0%	23	0.16	0.34	0.00	0.00
WIL1	REF	380	0.76	6.13	0.41	0.48
	100%	458	0.44	4.24	0.48	0.52
	FP	105	0.54	2.19	0.27	0.28
	0%	209	0.23	1.28	0.06	0.06
WIL2	REF1	151	0.49	1.72	0.10	NA
	REF2	223	0.42	2.62	0.08	NA
	100%	190	0.36	1.16	0.00	NA
	0%	207	0.54	1.99	0.00	NA
WIL3	REF	39	0.10	0.15	Aerial photos not available	
	100%	18	0.09	0.13	Aerial photos not available	
CASC	REF	62	0.24	0.33	0.00	NA
	FP	3	0.03	0.03	0.00	NA
	0%	1	0.02	0.02	0.00	NA

Table 2-6. Number of days with storm- or hurricane-force winds in the area of study sites included in the Hard Rock Study during the study period, 2006-2017 (data from Astoria, Hoquiam and Portland weather stations). Storm- and hurricane-force winds are defined as those 55-73.9 mph and >74 mph, respectively.

Period[1]	**Wind Strength**	**Weather Station**		
		Astoria	**Hoquiam**	**Portland**
Pre 3, Pre 2, Pre 1	Storm-force	8	10	0
	Hurricane-force	2	1[2]	0
Post 1, Post 2	Storm-force	9	10	0
	Hurricane-force	0	0	5
Post 3 through Post 8	Storm-force	30	27	5
	Hurricane-force	4	0	0

[1] Pre 3, Pre 2, Pre 1 = May 2006 through September 2008; Post 1, Post 2 = October 2008 through September 2010; Post 3 through Post 8 = October 2010 through September 2016

[2] Weather station was out of service due to high winds on 3-4 December 2007

In October 2009, a wildfire burned portions of two study sites, CASC-FP and CASC-0%, previously harvested November 2008 through March 2009. 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. Site visits on 22 October 2009 revealed that 47% of the basin was affected by fire at the CASC-0% site and 23% at the CASC-FP site. The fire crossed the stream in the CASC-0% site and approximately 200 m of stream length contained charred logs and wood. The fire did not cross the stream at the CASC-FP site. Riparian buffers were not greatly impacted in the CASC-FP site, though the fire did come within feet of the stream in several places. The fire did not directly affect sensitive sites in either site. We consider the consequences of this fire when interpretating responses to treatments but, given its limited spatial extent and severity, it is unlikely to have had a confounding effect.

2-7. STATISTICAL ANALYSIS APPROACH

We designed this study to evaluate differences in the magnitude of change (post-harvest – pre-harvest) among treatments at the site scale. Analyses evaluated the following, generalized null hypothesis:

$$\Delta T_{\mathrm{REF}} = \Delta T_{100\%} = \Delta T_{\mathrm{FP}} = \Delta T_{0\%} \qquad \textbf{(Eq. 2-1)}$$

where ΔT_{REF} is the change in the reference, and $\Delta T_{100\%}$, ΔT_{FP}, and $\Delta T_{0\%}$ are the changes in the 100%, FP and 0% buffer treatments, respectively.

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 determine whether post-harvest differences among treatments are associated with forest practices or environmental variation.

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 × time interaction, and site × treatment interaction. The latter two terms correspond to the variation in the year effect by basin, and the variation in treatment effect by basin. Other sources of variation are also included in the experimental error.

As different response variables may have different sampling constraints or statistical properties (e.g., continuous vs. count), the statistical methods varied slightly among response variables. Each chapter details the statistical approach used within the BACI design. We present the units of measurement and their equivalents (English or metric).

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 effects ($0.05 < P < 0.15$) 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 for some variables *a priori* (e.g., Underwood 1997; Welsh and Ollivier 1998; **Table 2-7**). 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.

Table 2-7. Alpha (α) level used for each response category to evaluate the statistical significance of the period × treatment contrast for a difference among treatments in the Hard Rock Study.

Response Category	Alpha (α)
Stand Structure and Tree Mortality	0.10
Wood Recruitment and Loading	0.10
Stream Temperature and Cover	0.05
Nitrogen Export	0.05
Stream Channel Characteristics	0.10
Stable Isotopes	0.10
Stream-associated Amphibians	0.10

2-8. SCOPE OF INFERENCE

The temporal scope of inference is the nine 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.

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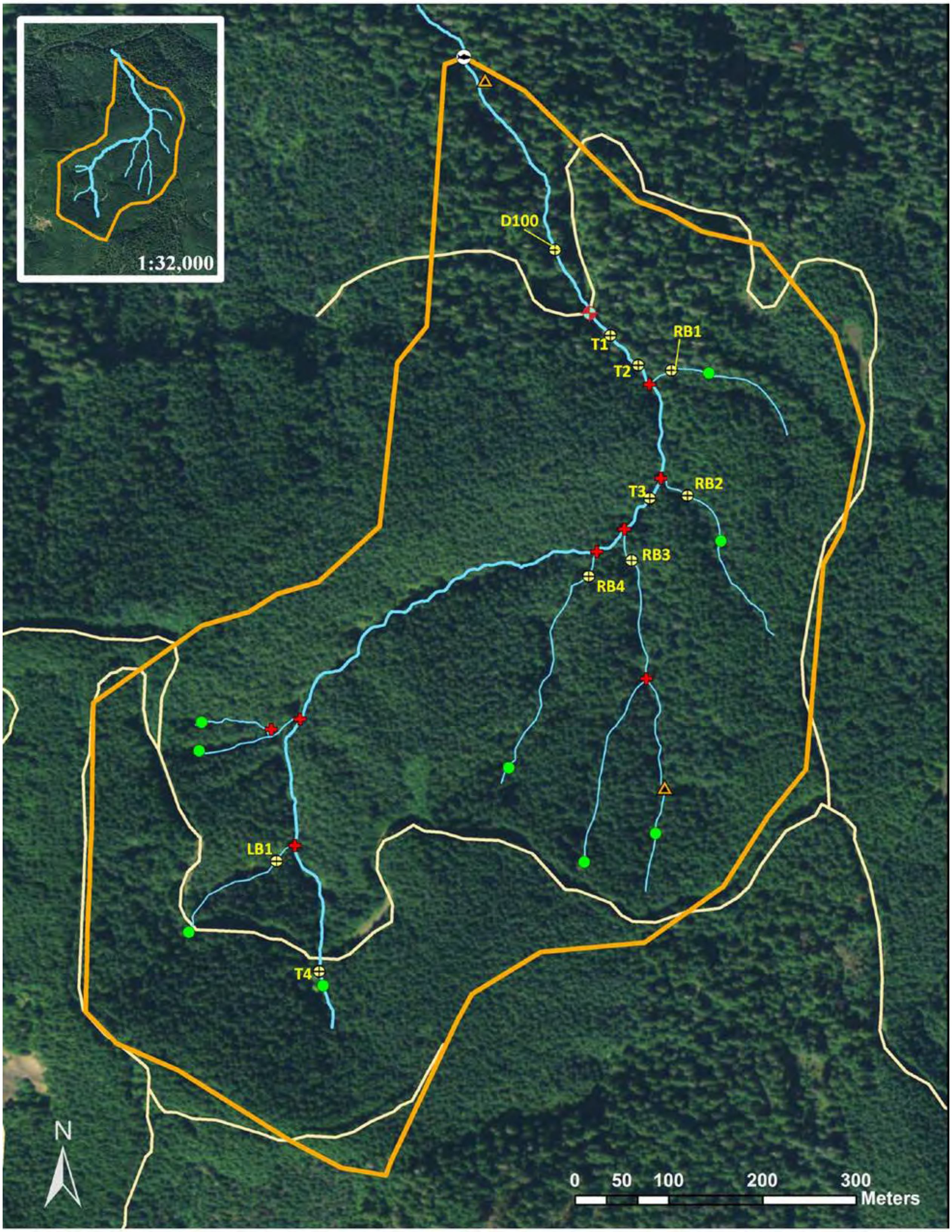
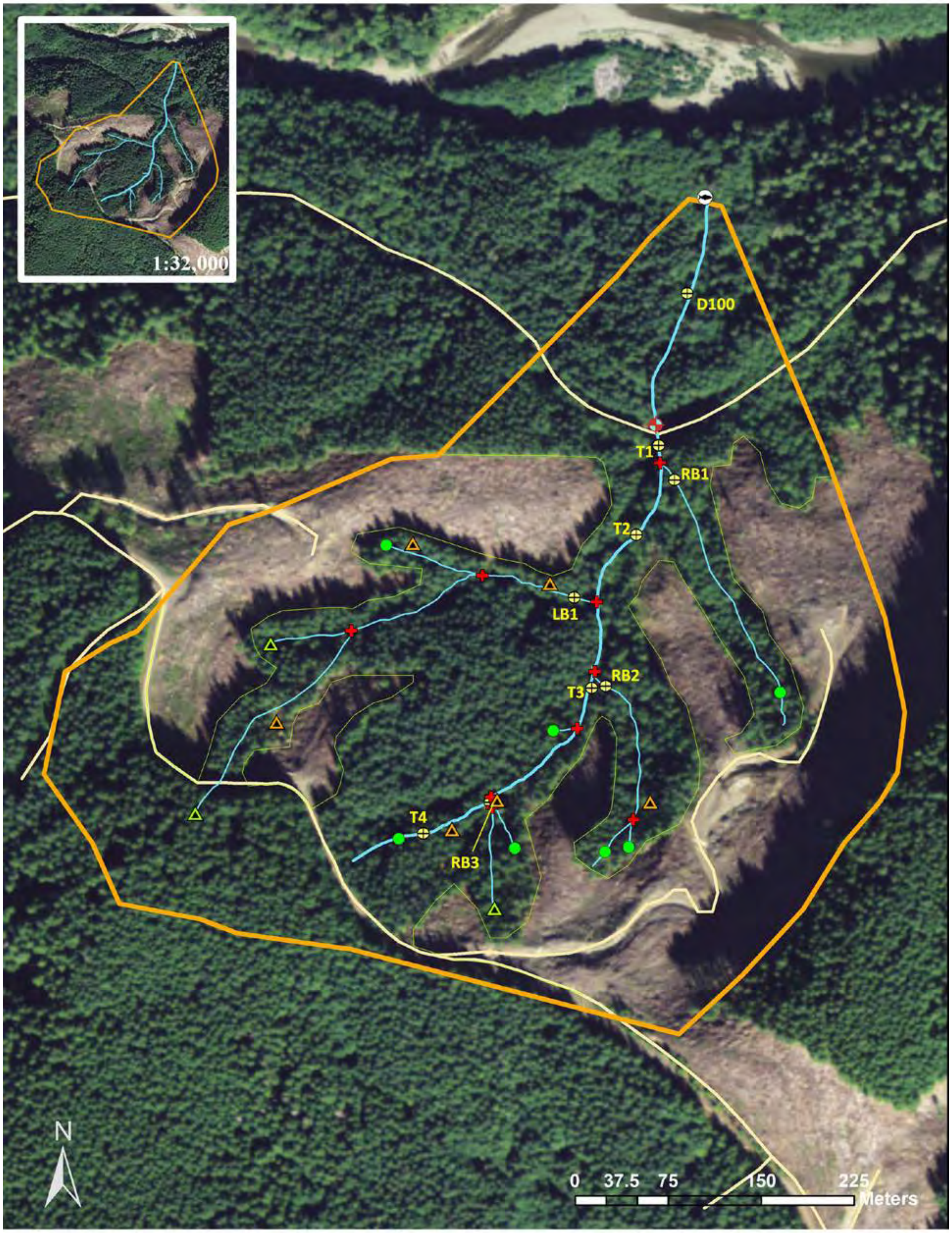
2-10. APPENDIX 2A – AERIAL OVERVIEW OF STUDY SITES IN THE YEAR FOLLOWING HARVEST

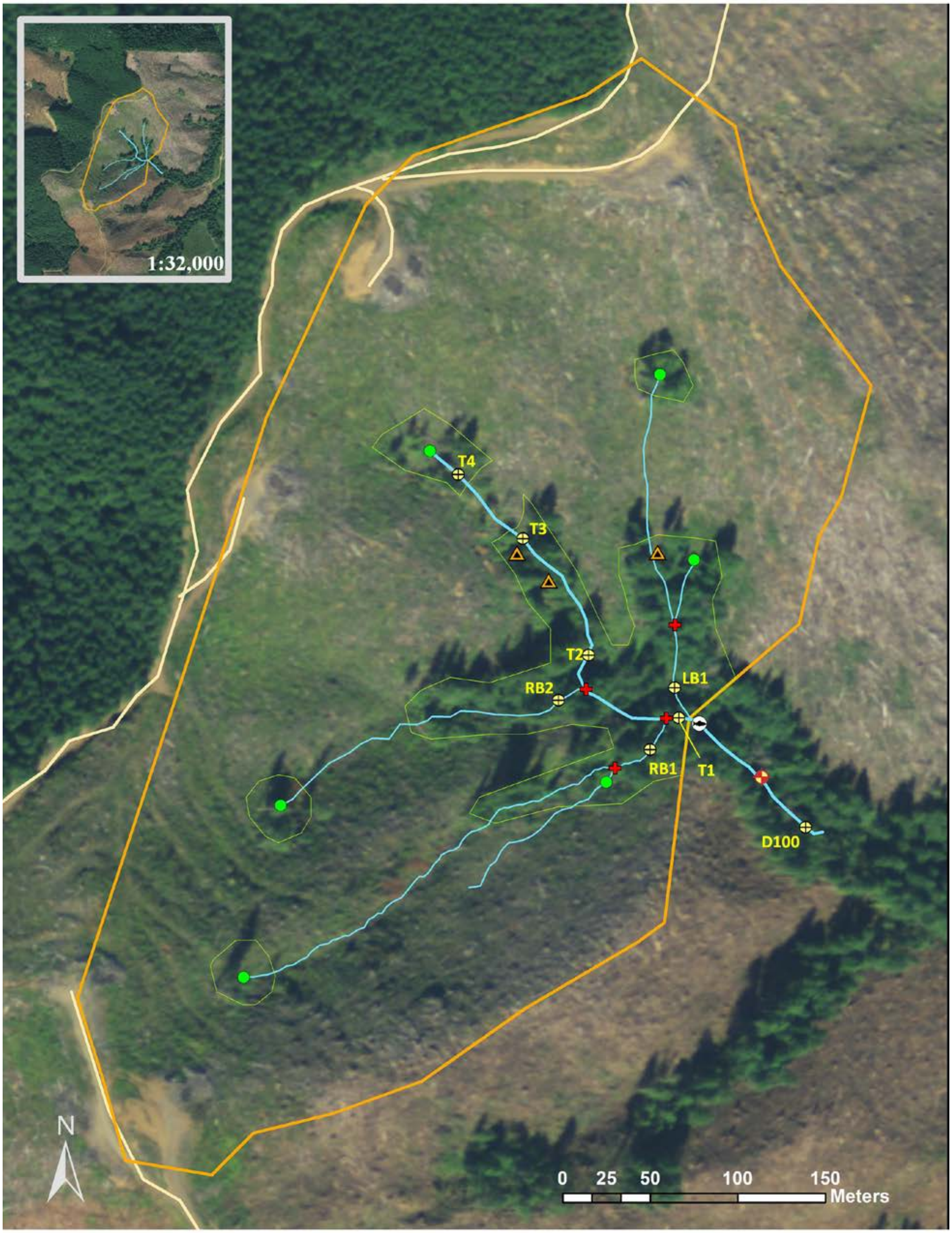
Appendix Figure 2-1. Study basin map legend for Type N Waters, sensitive sites and other features for sites included in the Hard Rock Study. *Note: the Olympic block 100% treatment (OLYM-100%) map displays the NAIP 2009 orthophoto as its base layer.



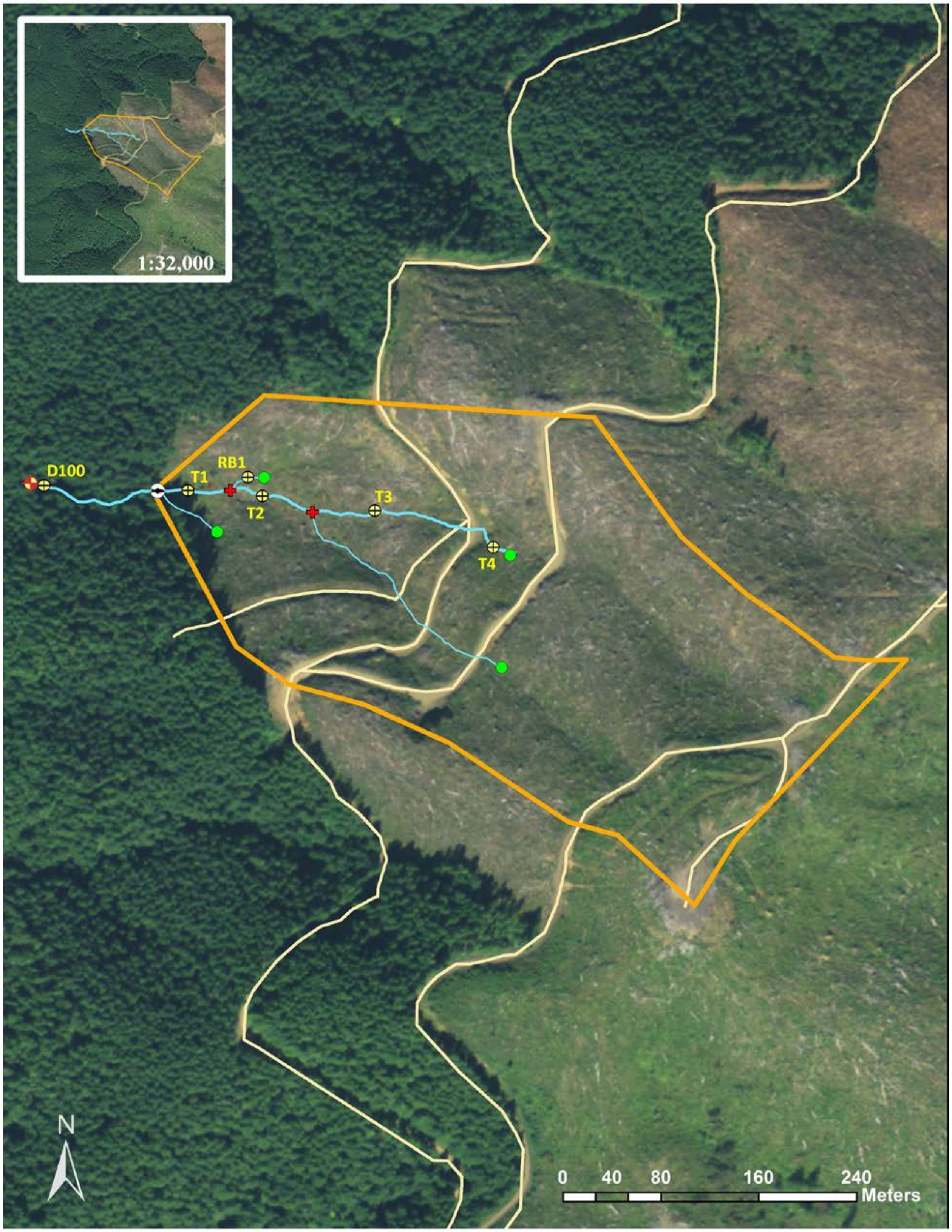
Appendix Figure 2-2. Olympic block reference (OLYM-REF).



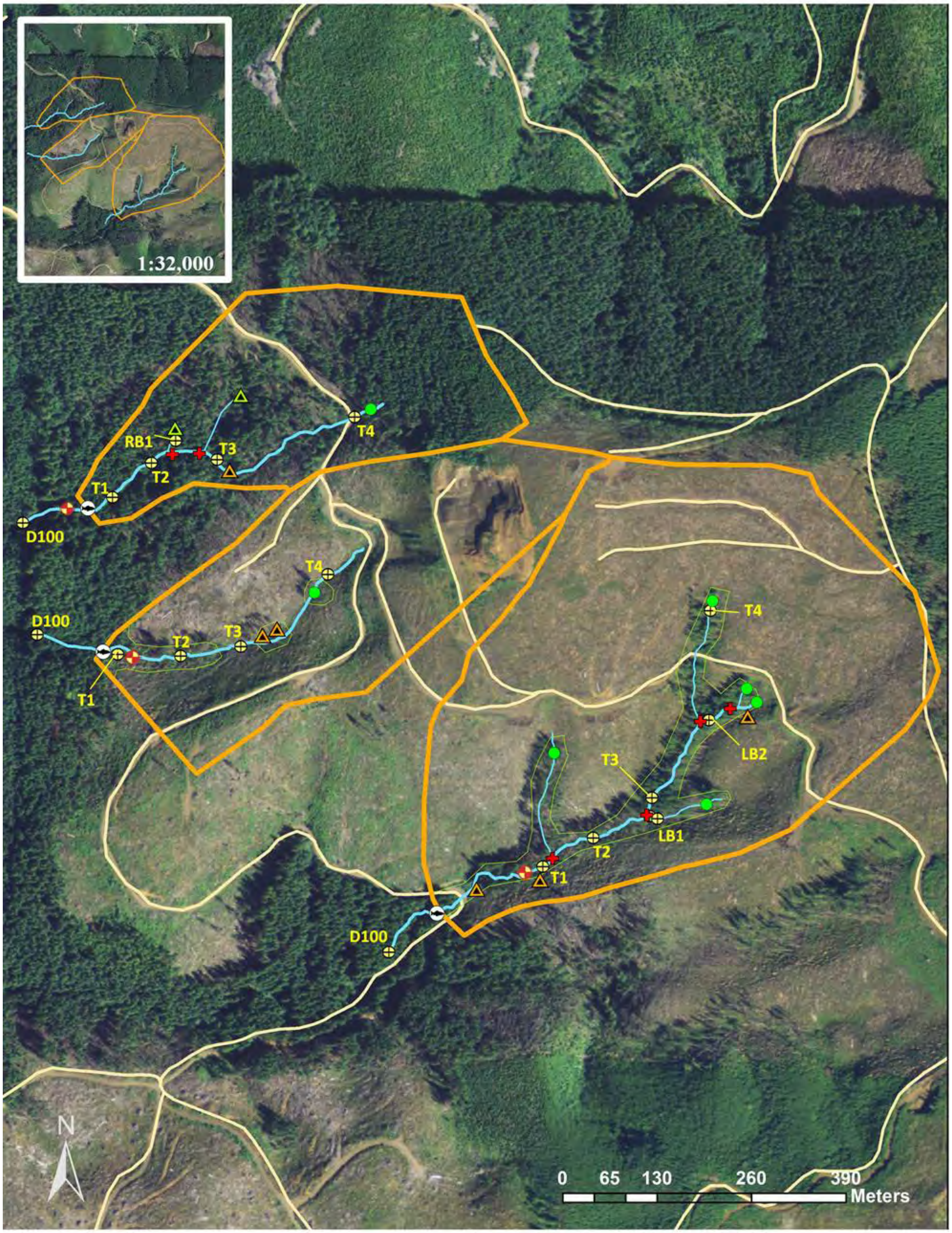
Appendix Figure 2-3. Olympic block 100% treatment (OLYM-100). Base layer is the NAIP 2009 orthophoto.



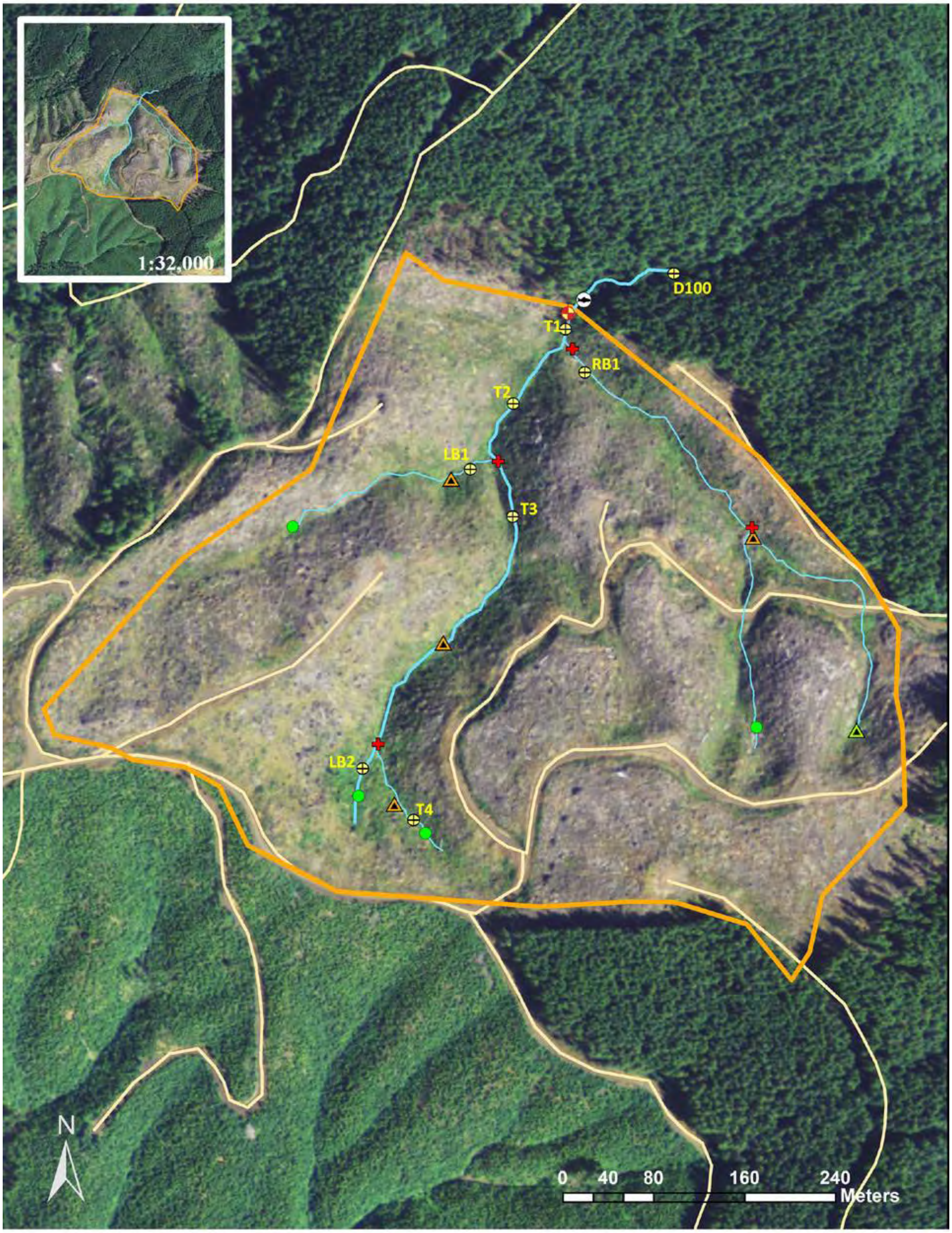
Appendix Figure 2-4. Olympic block FP treatment (OLYM-FP).



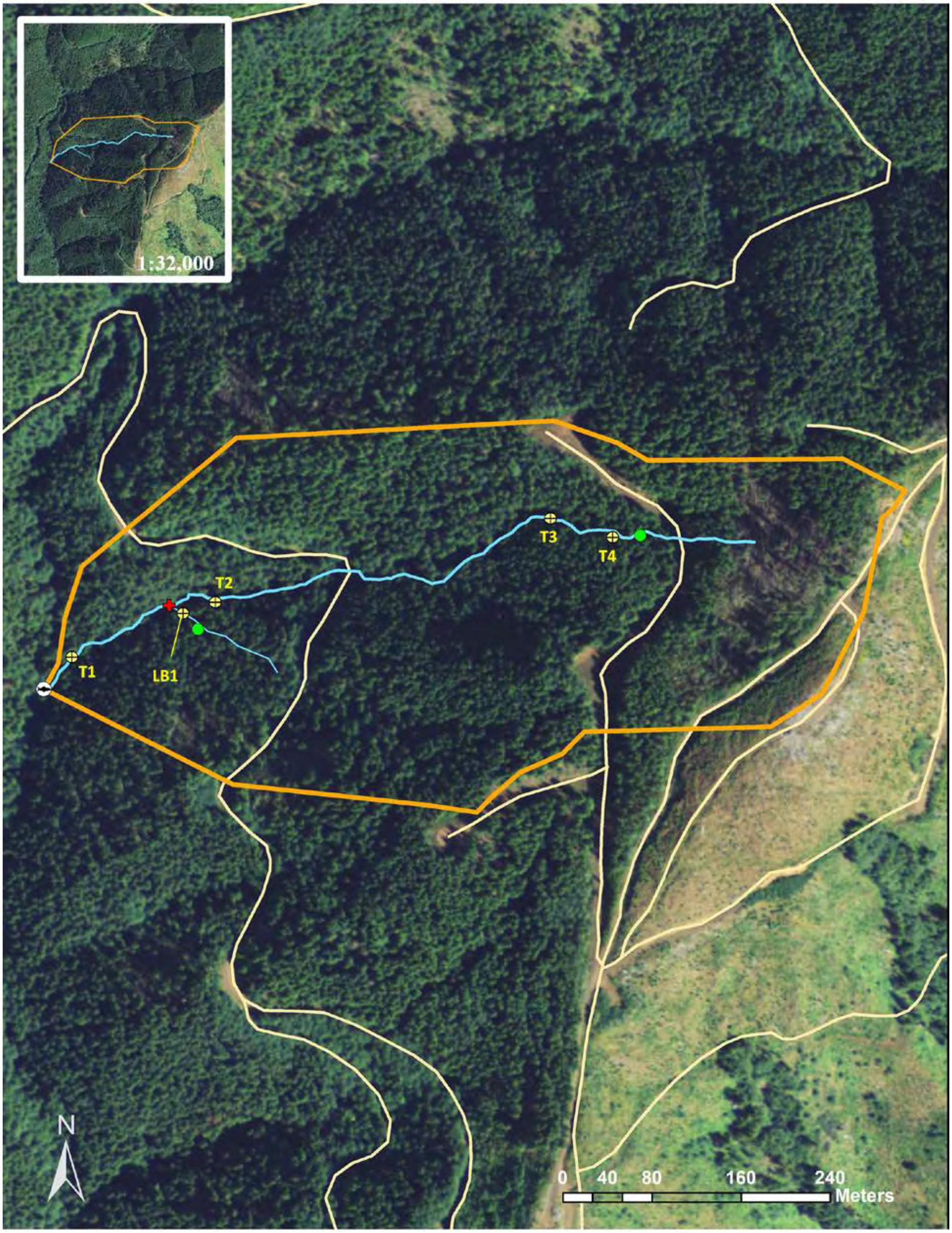
Appendix Figure 2-5. Olympic block 0% treatment (OLYM-0).



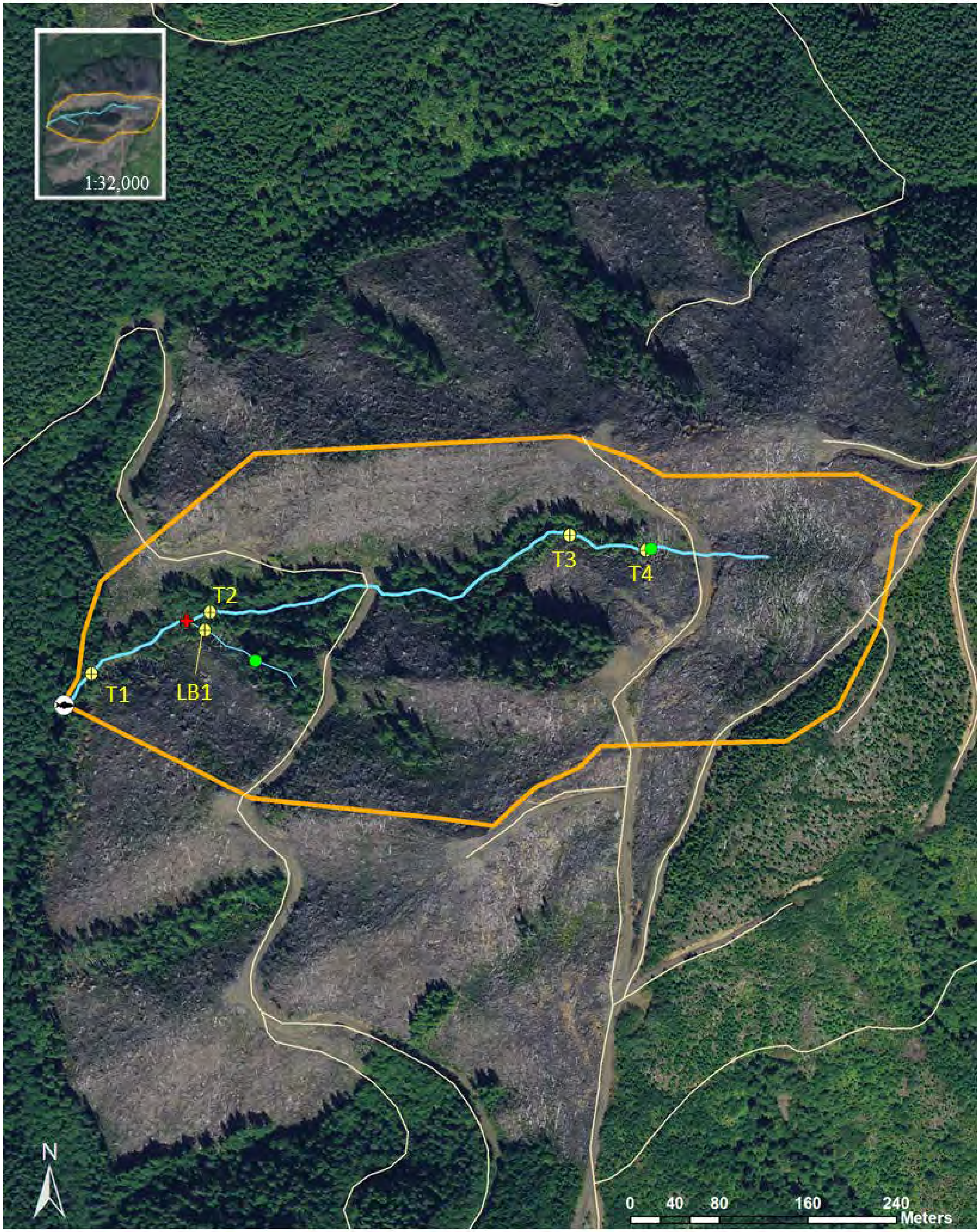
Appendix Figure 2-6. Willapa 1 block (from north to south): reference (WIL1-REF), FP treatment (WIL1-FP) and 100% treatment (WIL1-100%).



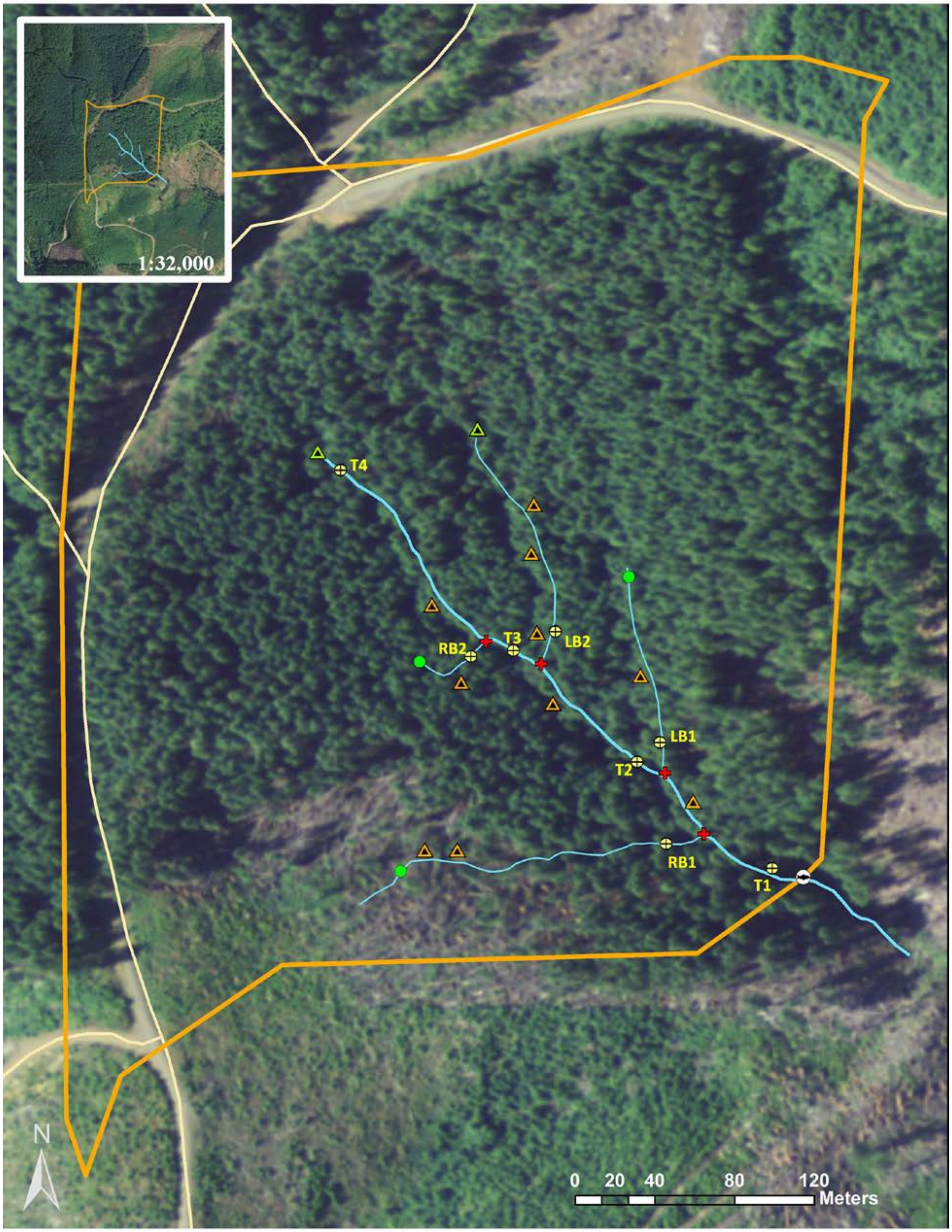
Appendix Figure 2-7. Willapa 1 block 0% treatment (WIL1-0%).



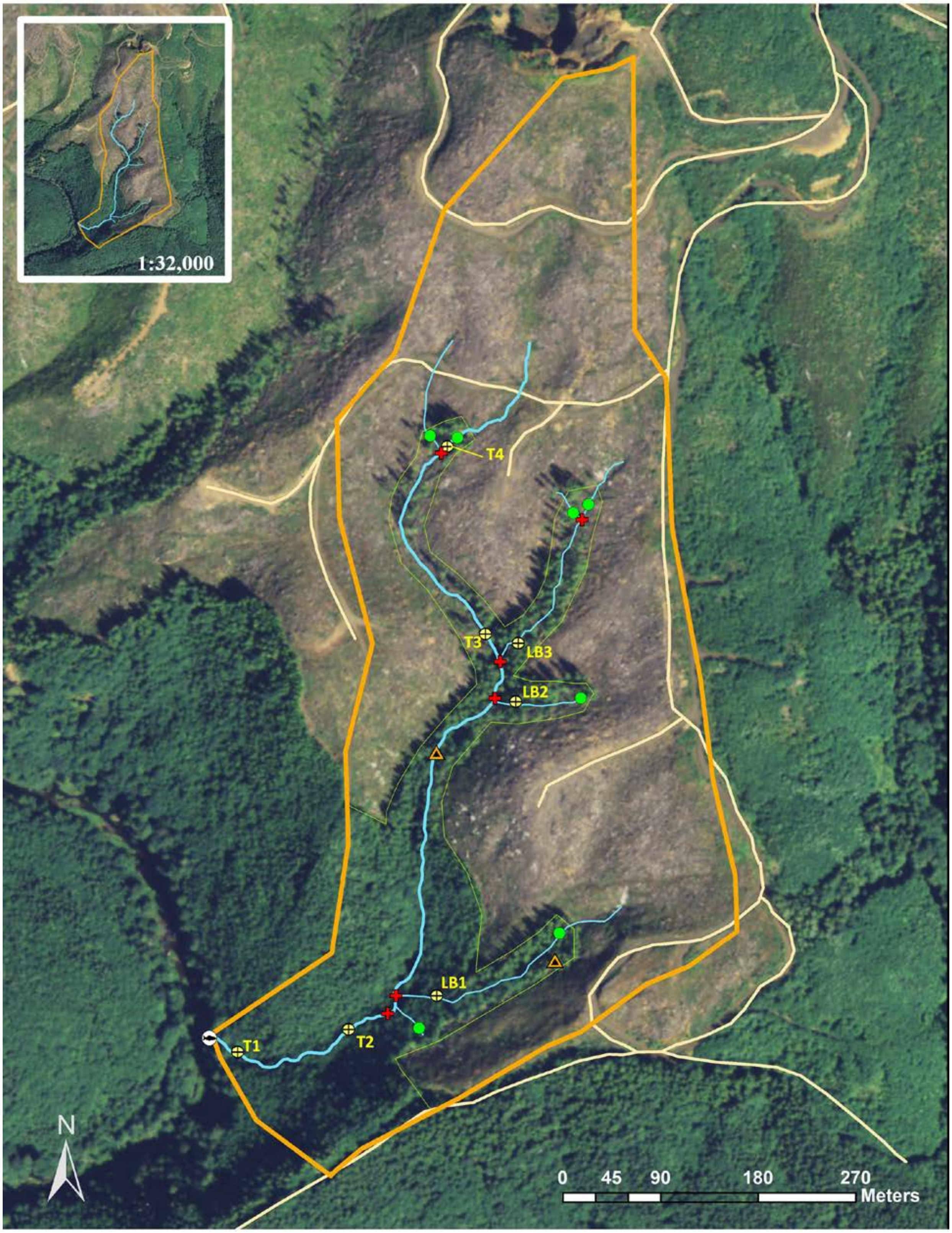
Appendix Figure 2-8. Willapa 2 block reference 1 (WIL2-REF1). This site was harvested in 2016, becoming the WIL2-FP (see **Appendix Figure 2-9**).



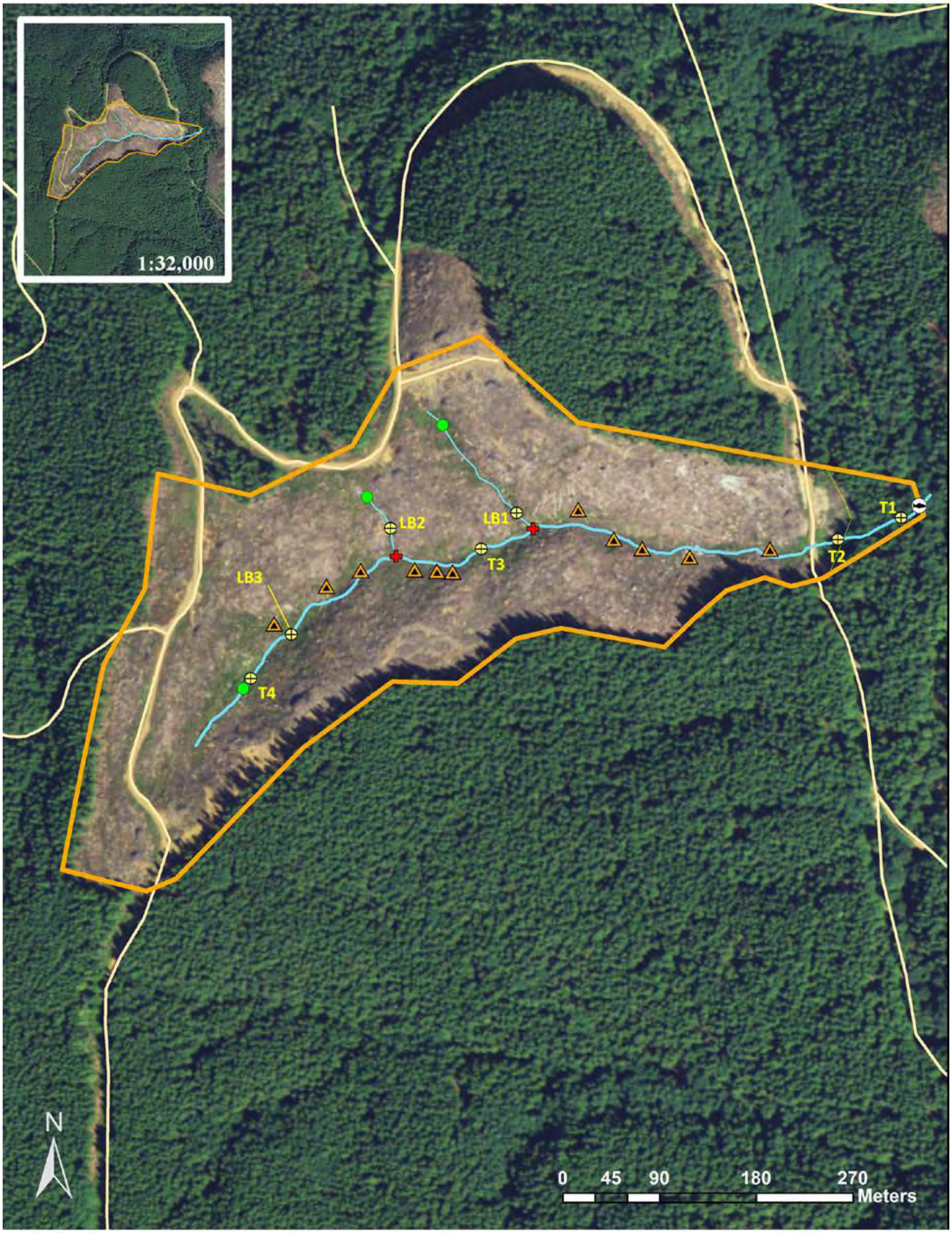
Appendix Figure 2-9. Willapa 2 block FP treatment (WIL2-FP), harvested in 2016 (previously the WIL2-REF1, see **Appendix Figure 2-8**).



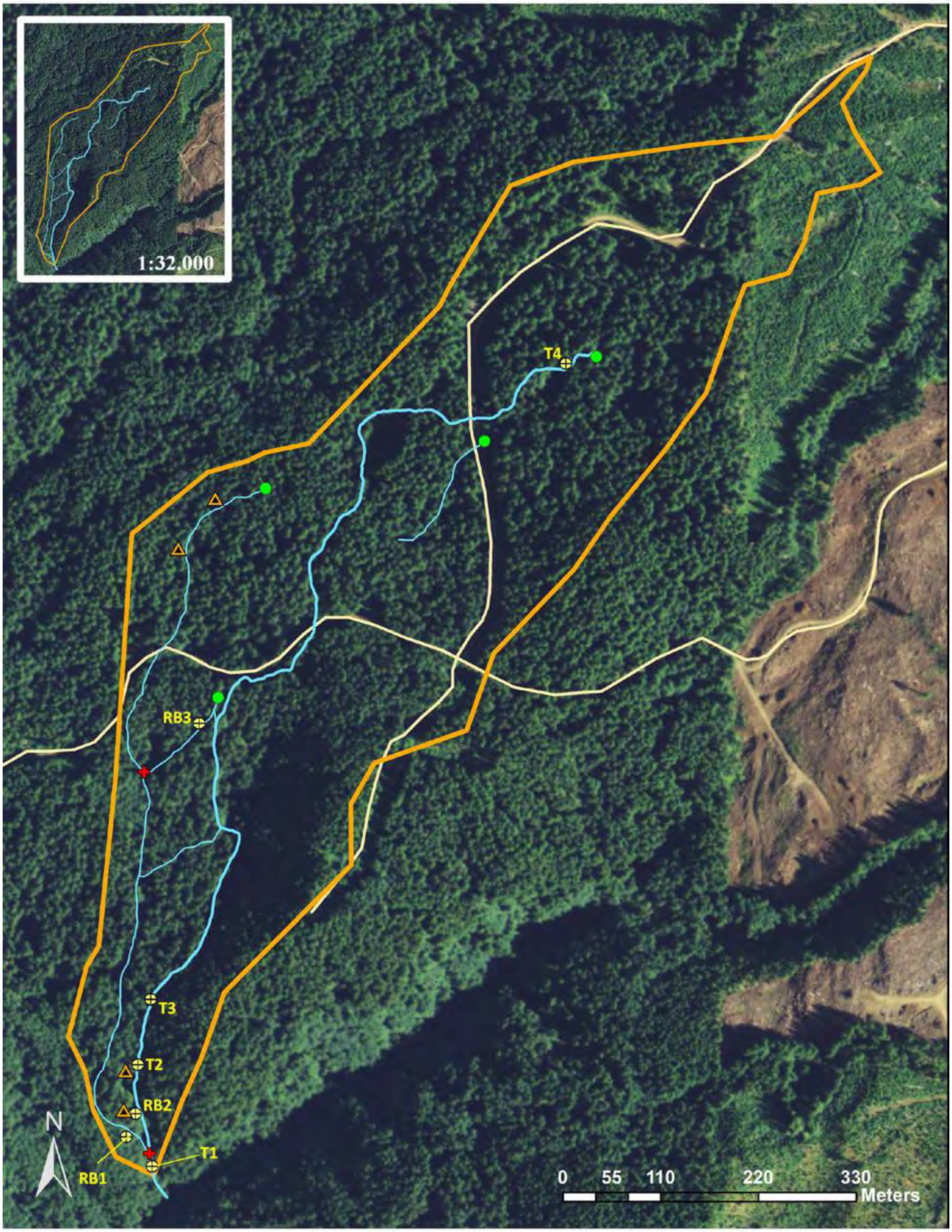
Appendix Figure 2-10. Willapa 2 block reference 2 (WIL2-REF2)



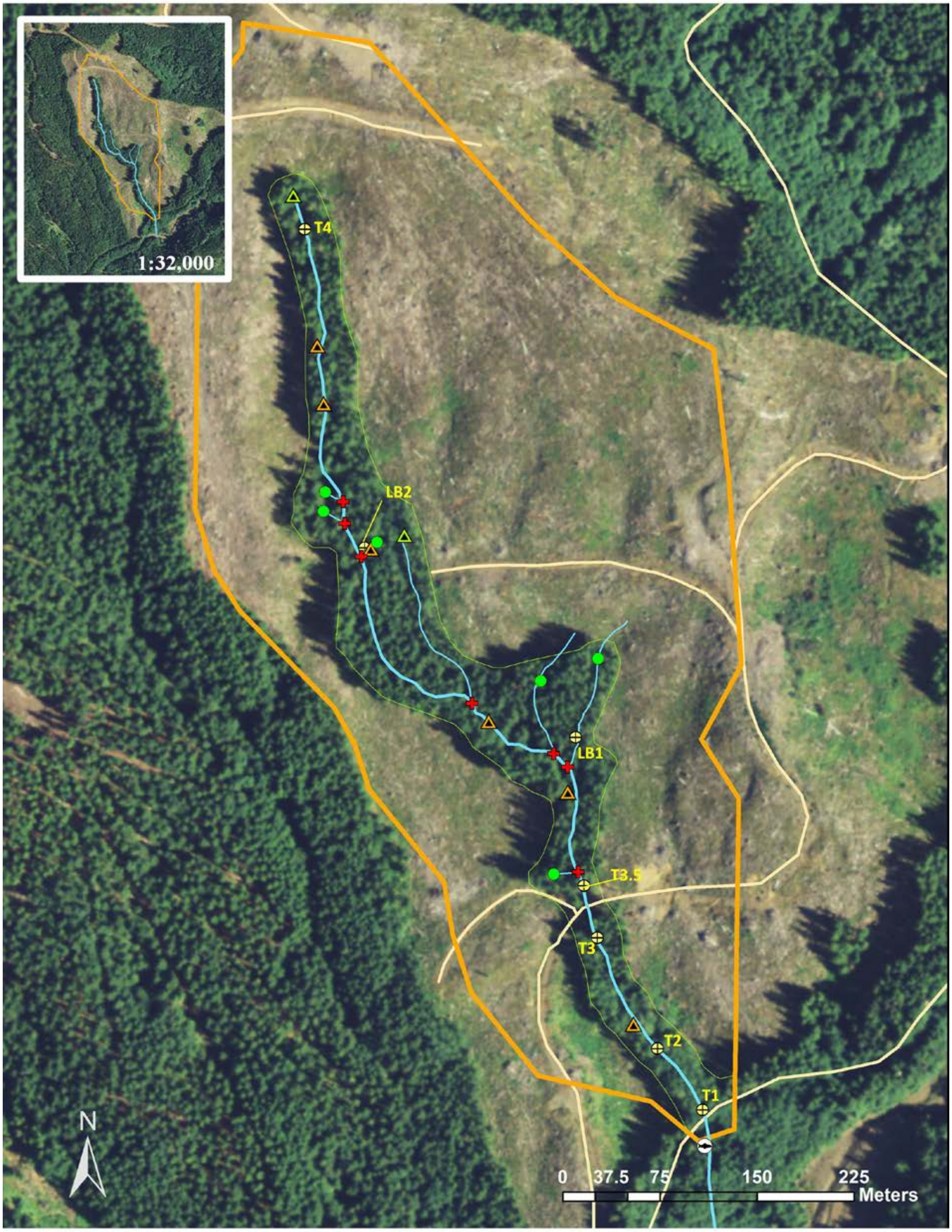
Appendix Figure 2-11. Willapa 2 block 100% treatment (WIL2-100%).



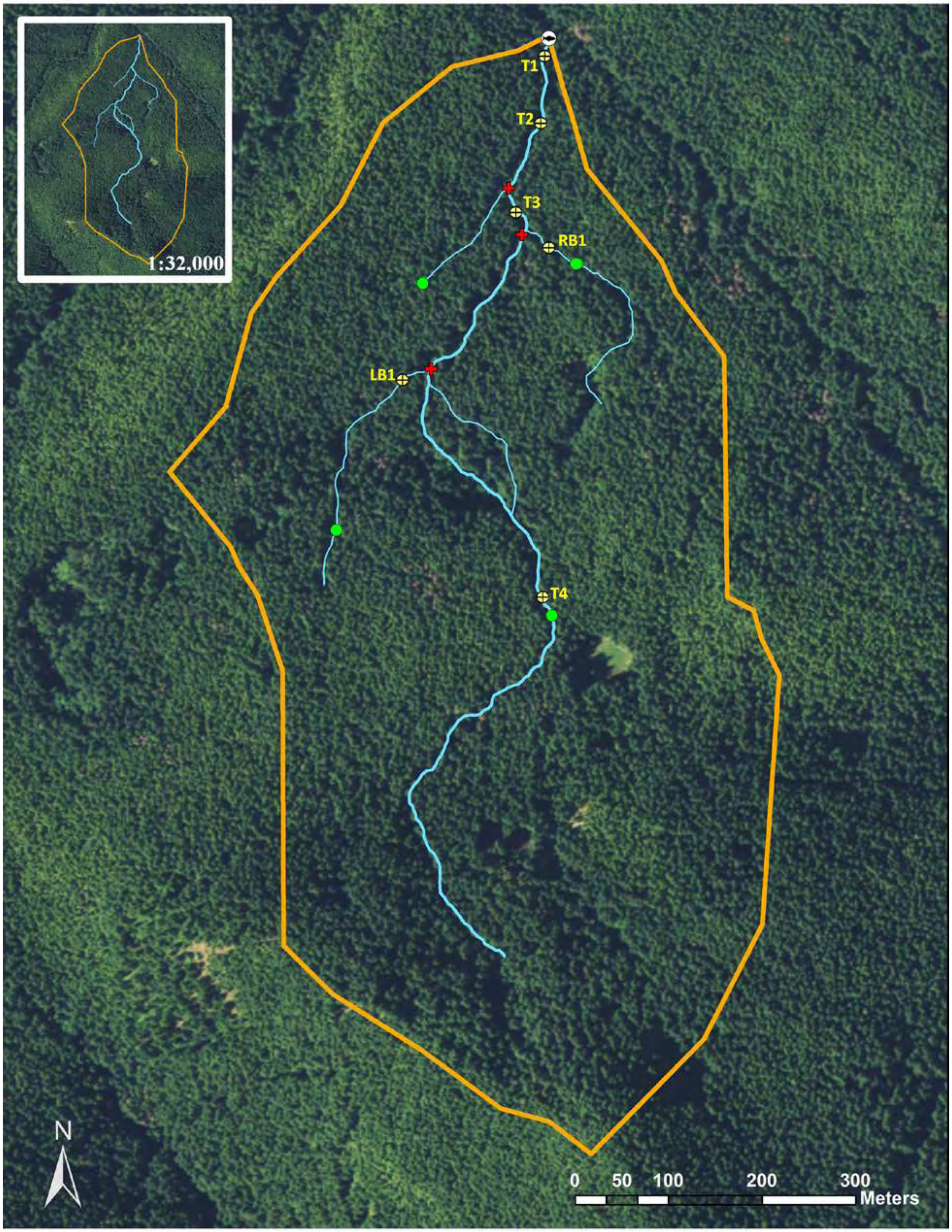
Appendix Figure 2-12. Willapa 2 block 0% treatment (WIL2-0%).



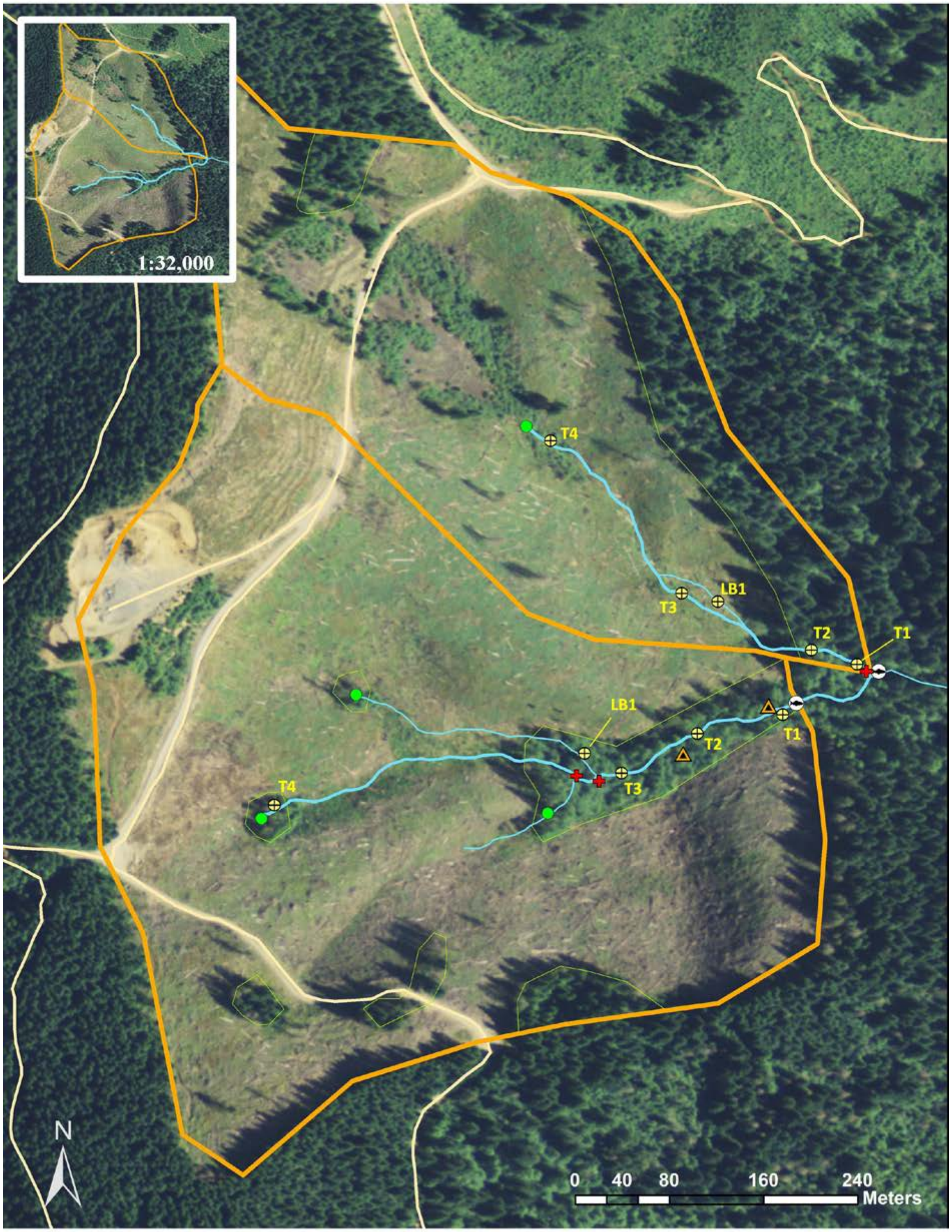
Appendix Figure 2-13. Willapa 3 block reference (WIL3-REF).



Appendix Figure 2-14. Willapa 3 block 100% treatment (WIL3-100%).



Appendix Figure 2-15. South Cascade block reference (CASC-REF).



Appendix Figure 2-16. South Cascade block (from north to south): 0% treatment (CASC-0%) and FP treatment (CASC-FP).

CHAPTER 3 - STAND STRUCTURE, TREE MORTALITY, WOOD RECRUITMENT AND LOADING

Dave Schuett-Hames, Aimee McIntyre, Reed Ojala-Barbour, and Greg Stewart

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3-1. ABSTRACT

We evaluated tree mortality, change in stand structure, large wood recruitment and wood loading in response to a range of riparian management strategies for clearcut timber harvest on non-fish-bearing, perennial (Np) streams in western Washington. The treatments differed in the proportion of the Np stream network with 15.2-m-wide buffers, including the 0% treatment (no buffers), the Forest Practices (FP) treatment (minimum of 50% of Np network buffered) and the 100% treatment (entire Np network buffered). A reach-scale analysis compared the response in two buffer types, the stream-adjacent riparian management zone (RMZ) buffers and circular buffers established around uppermost points of perennial flow (PIPs) located at the uppermost point of perennial flow. Treatment and reference basins were grouped into five geographic blocks. Data were collected for two years prior to and eight years after harvest. General linear mixed-effect models were used for between-treatment and treatment-reference comparisons.

Prior to harvest, most sites had dense riparian stands of 30- to 60-year-old Douglas-fir and western hemlock, except in the South Cascades block where tree density was lower and broadleaf species more abundant. A storm with hurricane-force winds hit coastal southwestern Washington during the pre-harvest period, causing extensive, but patchy, mortality and wood input into the streams in two blocks.

The greatest change in stand structure occurred in the 0% treatment and unbuffered portions of the FP treatment where clearcut harvest removed the riparian trees. A pulse of logging debris entered adjacent streams during harvest, but there was little wood input in the following 8 years. Among buffered RMZs, mortality and change in stand structure were greatest in FP treatment RMZ buffers (FPB RMZs) that lost 51% of initial basal area—2 and 4 times greater loss than in the 100% and reference RMZs, respectively. Cumulative change in basal area was significantly greater in the FPB RMZs than in the 100% treatment or reference RMZs. However, change was not significantly different in the 100% and reference RMZs. Cumulative mortality in FPB, 100% and reference PIPs was 56%, 43%, and 9%, respectively. There was not a significant difference in cumulative change in basal area between FPB and 100% PIPs, but both had significantly greater change than the reference PIPs.

In general, post-harvest changes in stand structure in the 100% treatment were intermediate between FPB and reference; more similar to the reference in the RMZs, but more similar the FPB in the PIPs. Post-harvest tree mortality rates were highest during the first two years post-harvest, then decreased over time. Windthrow was the dominant mortality agent in both 100% and FPB RMZ and PIP buffers. Large wood input appeared greater in FPB and 100% RMZs and PIPs than in the corresponding reference reaches, but the contrasts were not statistically significant. The post-harvest increase in mortality and wood input and reduction in density and basal area in the FPB RMZs and PIPs are consistent with the results of the same prescriptions in the earlier Westside Type N Buffer Characteristics, Integrity and Function Study.

Changes in the amount and characteristics of in-channel wood differed among treatments. In the two years post-harvest there was a significant increase in the number of small wood pieces (<10 cm diameter) over pre-harvest levels in all buffer treatments. Values increased in the 100%, FP and 0% treatments by an average of 58%, 69% and 176%, respectively. Small wood loading

increased in the FP and 0% treatments through post-harvest year 5, then declined in all treatments. The number of large wood pieces also increased in the two years post-harvest. Average increases among 100%, FP and 0% treatments were 66%, 44% and 47%, respectively. From post-harvest year 2 to 8, large wood continued to increase in the FP treatment, remained relatively stable in the 100% treatment, and decreased in the 0% treatment. Both large and small wood pieces frequently provided in-channel functions, including step formation, bank stability and hydraulic roughness. The proportion of stream channel length covered with newly recruited wood in post-harvest year 2 was greatest in the 0% treatment (0.48), intermediate in the FP and 100% treatments (0.41 and 0.36, respectively), and lowest in the reference (0.31). By post-harvest year 8, channel wood cover appeared to have stabilized in the FP treatment but had decreased in the 100% and 0% treatments.

Our ability to detect treatment effects was limited by substantial variation among (and within) FP and 100% treatment sites—variation related to pre-harvest stand structure, wood loading, and post-harvest mortality. Variation in stand structure 8 years post-harvest has implications for future stand development, wood recruitment and wood loading. Sites with a greater density of Douglas-fir and western hemlock that experienced little mortality should progress through the competitive-exclusion phase of forest development as single-cohort stands. Where post-harvest wind disturbance created canopy openings, a second cohort of shade-tolerant conifers may develop, creating a more complex stand structure. Based on simulation modeling in the literature, we expect wood loading in unbuffered RMZs to remain low in the short term, but to oscillate in the longer term in response to periodic inputs of logging debris from future harvest. We expect greater wood input and loading in the buffered reaches, with temporal patterns determined by stand characteristics and the magnitude and timing of disturbance.

3-2. INTRODUCTION

Wood plays an important functional role in Pacific Northwest streams (Bilby and Bisson 1998), influencing channel morphology and hydraulics, storage and routing of sediment and organic matter, aquatic habitat, aquatic communities, and food resources (Harmon *et al.* 1986; Bisson 1987; Curran and Wohl 2003; Montgomery *et al.* 2003; Wilcox *et al.* 2011). The abundance and characteristics of in-channel wood are a function of the interaction between input and depletion processes (Benda and Sias 2003). In small, tightly confined headwater stream channels, tree fall from streamside forests and input of upslope trees by mass wasting, snow avalanches, and forest harvest are major sources of wood, while bank erosion is less important (May and Gresswell 2003a). Due to limited transport capacity, large pieces tend to persist over time, but small wood is less persistent due to more rapid decay and greater susceptibility to downstream transport (Wallace *et al.* 2000; Scherer 2004). Wood loading in small headwater streams is highly variable (Jackson and Sturm 2002; Hassan *et al.* 2005). Compared to larger channels, small streams tend to have a greater abundance of functional pieces (Bilby and Ward 1989; Gomi *et al.* 2006) and smaller pieces because stream power is inadequate to transport wood efficiently (Bilby and Ward 1991; Jackson and Sturm 2002). Wood pieces lodge and accumulate in narrow headwater channels, forming obstructions that accumulate sediment and debris, and steps that dissipate energy (Bilby and Ward 1991; Gomi and Sidle 2003; Maxa 2009). Even small wood can play short-term functional roles in smaller stream channels (Gomi *et al.* 2001; Jackson and Sturm 2002; Maxa 2009).

The nature and timing of disturbance processes have a strong influence on wood input and loading (Spies *et al.* 1988; Bragg 2000). Mortality and recruitment of individual or small groups of trees from streamside forests that occurs gradually over time in the absence of major disturbance provides a stable supply of wood that gradually increases with stand age (Hedman *et al.* 1996; Warren *et al.* 2009). Episodic disturbances such as wind, fire, insect outbreaks, landslides or snow avalanches result in large inputs of wood at irregular intervals (Benda *et al.* 2003) resulting in temporal variation in wood loading (Bragg 2000). Episodic debris flows have a profound impact on wood loading in headwater channels, scouring wood from steep sections and depositing accumulations downstream where the gradient lessens (Benda 1990).

Wood input and loading are typically highest during the first few decades after disturbance, as newly killed trees recruit to the channel, augmenting pre-disturbance loading. This period of peak wood abundance is followed by an extended period of attrition as in-channel wood is depleted and little new wood is recruited. In the absence of further major disturbance, wood loading increases as the forest matures and wood is recruited from chronic mortality associated with suppression and small-scale disturbance (Spies *et al.* 1988; Bragg 2000).

Prior to widespread harvest in the 20th century, western Washington was largely covered with coniferous forests skewed towards older ages (Franklin and Hemstrom 1981). Forests progressed through successional stages following periodic disturbance from fire, wind, and flooding (Agee 1993; Edmonds *et al.* 2005) and wood was typically abundant in channels adjacent to older forests (Spies *et al.* 1988; Murphy and Koski 1989; Bilby and Ward 1991). Widespread timber harvest had profound effects on wood recruitment and loading in stream channels (Murphy and Koski 1989; Bilby and Ward 1991). Clearcut harvest adjacent to streams results in immediate input of logging debris (Jackson *et al.* 2001). It also increases the potential for mass wasting of unstable slopes (May 2001) and for debris flows (Nakamura *et al.* 2000; May 2001). Long-term effects include a reduction in large wood input (Beechie *et al.* 2000) and loading for many decades until new forests establish (Beechie *et al.* 2000; Bragg 2000; Meleason *et al.* 2003).

Concern about the effects of timber harvest on aquatic resources led to changes in management practices, including retention of streamside trees to provide shade and a source of future wood recruitment (Bisson 1987). The riparian management strategy in Washington State's Forest Practices Habitat Conservation Plan (FPHCP) includes a partial buffering strategy for non-fish-bearing, perennial (Type Np) streams (WADNR 2006). A riparian management zone (RMZ) is required on at least 50% of the stream length in each Type Np basin, while the remaining stream length may be clearcut to the channel edge. The buffering strategy was intended to provide enough large wood recruitment to create, restore and maintain riparian and aquatic habitats.

Harvest on state and private lands in western Washington is now occurring in dense, young forests dominated by western hemlock or Douglas-fir (NOAA and USFWS 2006). Newly established riparian buffers in exposed locations adjacent to clearcut harvest units are susceptible to disturbance from wind (Ruel *et al.* 2001; Beese *et al.* 2019). In the absence of disturbance, growth and density-dependent mortality will contribute increasing amounts of wood to the adjacent stream (Liquori 2000; NOAA and USFWS 2006). Past studies of riparian buffers on headwater streams in western Oregon and Washington have documented varying effects of wind, including elevated rates of mortality in a substantial proportion of sites (Grizzel and Wolff 1998; Burton *et al.* 2016; McIntyre *et al.* 2018; Schuett-Hames and Stewart 2019). Windthrow provides immediate input of wood to streams (Bahuguna *et al.* 2010) but reduces stand density and potential for future recruitment (Martin and Grotefendt 2007). This mortality can also affect the future development of the riparian buffer, depending on density of surviving trees and regeneration processes (Franklin *et al.* 2002).

Scientific uncertainty surrounds the potential for the FPHCP buffering strategy for Type Np streams to influence tree mortality rates; future stand development; and the sources, magnitude, and characteristics of wood input, persistence, and loading. To address these uncertainties, we compared changes in stand structure, tree mortality, and wood input and loading among and within Type Np headwater basins with differing proportions (0 to 100%) of stream length buffered.

3-3. METHODS

Data were collected in 17 Type Np headwater basins located in competent lithologies (largely basaltic) across western Washington. Using a BACI design (see Chapter 2–*Study Design* in this report), we compared changes in riparian stand structure, tree mortality, large wood recruitment to the channel, in-channel wood loading, and channel wood cover among reference and treatment sites. Specifically, we compared unharvested Type Np reference basins (n = 6) to 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), Forest Practices (FP) treatment (two-sided riparian buffer along at least 50% of the Type Np stream length, according to current FP rules; n = 3), and 0% treatment (clearcut harvest to the stream edge with no riparian buffer; n = 4). Our objective was to quantify the magnitude and duration of change in stand structure, tree mortality, wood recruitment and in-channel wood loading following harvest.

3-3.1. DATA COLLECTION

3-3.1.1. Stand Structure, Tree Mortality and Wood Recruitment

We evaluated two riparian management strategies defined by FPHCP for Type Np streams in western Washington. Riparian Management Zones are 15.2-m (50-ft) wide on both sides of Type Np streams. Perennial initiation points (PIPs) are sensitive sites located at the uppermost point of perennial flow surrounded by a circular buffer of 17.1 m (56 ft) radius. Riparian Management Zones present a challenge to sampling due to their shape (narrow linear features) and variability in stand structure, both parallel and perpendicular to the stream. We sampled RMZs with strip plots (Marquardt *et al.* 2010) — rectangular plots of 15.2 × 30.5 m (50 × 100 ft) oriented with the long axis perpendicular to, and bisected by, the stream channel. Plots were established systematically along each stream at intervals that varied with basin size. At sites with <1,524 m (5,000 ft) of RMZ (n = 11), plots were spaced every 30.5 m (100 ft) to sample ~50% of the RMZ. At sites with >1,524 m of RMZ (n = 6), plots were spaced every 45.7 m (150 ft) to sample ~33% of the RMZ. Areas were not sampled where it was not possible to apply riparian buffer treatments due to management constraints (e.g., potential for mass wasting or presence of adjacent fish-bearing (Type F) streams). PIP plots sampled the entire 17.1 m (56-ft) radius PIP buffer. The number of PIPs sampled varied among sites and treatments due to differences in basin size and tributary density. PIPs were not sampled in the Willapa 3 (WIL3) block (**Table 3-1**).

Data were collected between May and September of each sample year. Pre-harvest data were collected during summer 2007 (Pre 2) and 2008 (Pre 1), with one exception. The WIL3 sites were not sampled in 2007. Rather, 2007 stand conditions were reconstructed from data on standing and fallen trees in 2008. Post-harvest data were collected in 2010 (Post 2), 2013 (Post 5) and 2016 (Post 8). In each plot, we sampled all standing trees ≥10.2 cm (4 in) in diameter at breast height (dbh). We recorded the condition (live or dead), species, and diameter for each tree. Live trees were marked by tree crayon at each survey. Dead trees were marked with tree paint so that newly dead trees could be identified in subsequent surveys. Where possible, we recorded the mortality agent (i.e., wind, erosion, suppression, fire, insects, disease, or physical damage). At randomly selected plots, regenerating trees were tallied in six circular subplots (1.1 m [3.7 ft] radius) spaced at perpendicular distances of 3.0, 7.6 and 12.2 m (10, 25 and 40 ft) from the channel edge on both sides of the stream. Separate tallies were made by species for seedlings (≥ 15 cm [6 in] tall, < 2.5 cm [1 in] dbh) and saplings (≥ 2.5 to < 10.2 cm [4 in] dbh).

Table 3-1. Number of RMZ and PIP plots by site, block, and treatment.

Block	Treatment	RMZ Plots	PIP Plots
OLYM	REF	44	3
	100%	20	2
	FP	21	3
	0%	10	0
WIL1	REF	14	2
	100%	20	2
	FP	7	1
	0%	21	5
WIL2	REF1	16	2
	REF2	15	1
	100%	29	2
	0%	20	3
WIL3	REF	37	0
	100%	25	0
CASC	REF	28	3
	FP	14	3
	0%	9	1

We tallied and measured all fallen trees that originated within the plot boundaries. Each tree that had fallen since the last measurement was marked with tree paint to avoid subsequent resampling. Fallen trees were classified as uprooted (toppled with roots attached) or broken (i.e., snapped along the bole). If the portion that remained standing was at least 1.4 m (4.5 ft) tall it was treated as a standing tree; the remaining portion was treated as fallen if the diameter was ≥10.2 cm (4 in) at the large end. We recorded species, dbh, and horizontal distance to channel edge from the rooting location. Fallen trees that reached the channel were recorded as one of three recruitment classes: bankfull (intruding into the bankfull channel), spanning (extending over the channel supported on both sides), or suspended (extending over the channel but supported on only one side).

We tallied pieces of large wood that recruited to the channel and recorded the length and mid-point diameter for the in- and over-channel portions of each piece. Pieces had to originate from trees within a plot, fall into or over the bankfull channel, and meet the size criteria of Gomi and colleagues (2001) for headwater streams: ≥10.2 cm (4 in) in diameter and ≥0.5 m (1.6 ft) in length.

3-3.1.2. Channel Wood Loading and Cover

We sampled channel wood loading from April through October in each sample year: 2006 (Pre 3), 2007 (Pre 2), 2009 (Post 1), 2010 (Post 2), 2013 (Post 5), and 2016 (Post 8). To evaluate channel wood loading, we sampled the contiguous 200 m (656 ft) of stream immediately upstream of the F/N break (i.e., the point of last known fish use), and additional portions of the mainstem channel depending on its length. Specifically, for sites with a mainstem length <300 m (984 ft; n = 1) we sampled the entirety of the mainstem; for lengths of 300 to 800 m (984 to 2,625 ft; n = 10) we sampled a minimum of 50% of the remaining mainstem (above the contiguous 200 m); and for lengths >800 m (2,625 ft; n = 6) we sampled a minimum of 25% of the remaining mainstem. Where we subsampled, we did so in alternating 20 m (66 ft) stream segments comprising two consecutive 10 m (33 ft) sample reaches (hereafter, sample intervals). These segments were systematically distributed along the mainstem channel above the 200-m (656-ft) contiguous reach sampled above the F/N break.

Within each sample interval, we tallied all wood pieces (≥2 cm [~1 in] average diameter and ≥10 cm [4 in] long) in or directly over the stream within the bankfull channel using the methodology developed by Veldhuisen and colleagues (2007). We tallied wood pieces in each of two diameter classes based on average diameter, following Schuett-Hames and colleagues (2012). The classes were small (2–10 cm) and large (≥10 cm). We also assigned each piece to a functional category based on its apparent in-stream function (**Table 3-2**). Wood loading in Post 1 includes wood recruited during harvest as well as recruitment from a pre-harvest storm event in December 2007 (see Chapter 2 – *Study Design*, **Section 2-6** *Unanticipated Disturbance Events* in this report).

We evaluated channel wood cover by estimating cover of newly recruited wood in the post-harvest period (Post 2, Post 5, and Post 8). Recruited wood included all small and large wood originating from fallen or cut trees, branches or twigs. We estimated wood cover visually to the nearest 10% in consecutive 10-m sample intervals over the area of the bankfull channel for the entire stream length, including mainstem and all tributaries.

We did not sample on several occasions. We did not sample the OLYM-REF in 2006 because it had not yet been approved for inclusion in the study. We did not sample the WIL2-REF1 in 2009 because it was initially intended as an FP treatment; when it was not harvested on time, we included it as a second reference in the WIL2 block. In 2016, when this same site was harvested, we did not include it in our Post 8 sample because it could no longer serve as a reference (see Chapter 2 – *Study Design*, **Section 2-5** *Study Timeline* in this report).

Table 3-2. Stream functional categories for wood pieces. Categories were hierarchical, with each piece classified as the highest function to which it contributed (e.g., a piece that contributed to hydraulic roughness, bank stability, and step formation was classified as “step formation”).

Functional Category	Functional Hierarchy	Description
Step formation	1	Contributes step formation
Bank stability	2	Contributes to stream bank stability
Hydraulic roughness	3	Creates hydraulic roughness
Loose	4	Loose, not anchored in the channel
Spanning	5	Spans part or all of the channel

When wood from in-channel slash and/or windthrow obstructed the channel so that we could not effectively sample without disturbing or removing it, we recorded the length and locations of these reaches (hereafter, wood-obstructed reaches; **Figure 3-1**). The length of obstruction ranged from 2 m (7 ft) to 144 m (472 ft) in a given site and year. When they comprised >5% of the sampled stream length, we undertook a modified sampling approach within the wood-obstructed reaches. We randomly located 3-m long plots parallel to the stream for destructive sampling—a length that could be completely censused while minimizing in-channel disturbance both within and adjacent to the plot. The number of destructively sampled plots was proportional to the length of the stream channel obstructed; values ranged from 1 to 6 plots per site per year (**Table 3-3**). To fully census each plot, we used hand tools (e.g., handsaws, clippers) to remove wood from the bankfull channel down to the streambed. As with the standard methodology, we tallied all pieces by diameter and functional categories. Once completed, we returned all wood to the bankfull channel. Although destructive samples had the potential to influence piece number, length and transport potential for wood pieces sampled, the proportion of the stream sampled with this method was minimal compared to the overall channel length sampled (maximum of 18 m for a site and year). However, to minimize the potential for bias, we did not resample wood-obstructed plots.

Figure 3-1. Examples of a) an unobstructed stream reach in which the standard sampling protocol could be applied, and b) an obstructed reach in which a modified sampling approach was used in some study sites during the Hard Rock Study in the post-harvest period. Photo credits: a) Frithiof Teal Waterstrat, b) Aimee P. McIntyre.

Table 3-3. The number of plots sampled destructively in wood-obstructed reaches at each site in each post-harvest sample year.

Block	Treatment	Post 1	Post 2	Post 5	Post 8
OLYM	REF	0	0	0	0
	100%	0	0	0	0
	FP	0	2	0	3
	0%	6	6	6	3
WIL1	REF	0	0	2	0
	100%	3	1	2	0
	FP	4	4	3	0
	0%	4	4	3	0
WIL2	REF1[1]	0	0	0	-
	REF2	0	0	0	0
	100%	0	3	0	2
	0%	3	4	5	4
WIL3	REF	0	0	0	0
	100%	0	0	0	0
CASC	REF	0	0	0	0
	FP	2	2	0	0
	0%	0	0	0	0

[1] WIL2-REF1 was harvested between Post 7 and Post 8 and was not sampled for wood loading in Post 8 (see Chapter 2 – *Study Design*, **Section 2-5** *Study Timeline* in this report).

3-3.2. ANALYSIS

The partial buffering strategy in the FP treatment created two distinct RMZ conditions: reaches with leave-tree buffers (hereafter FPB) and unbuffered reaches lacking trees (**Figure 3-2**). Consequently, in analyses of stand structure, mortality, and wood recruitment, we included only those treatments or reaches within a treatment in which trees were retained: REF, 100% and FPB.

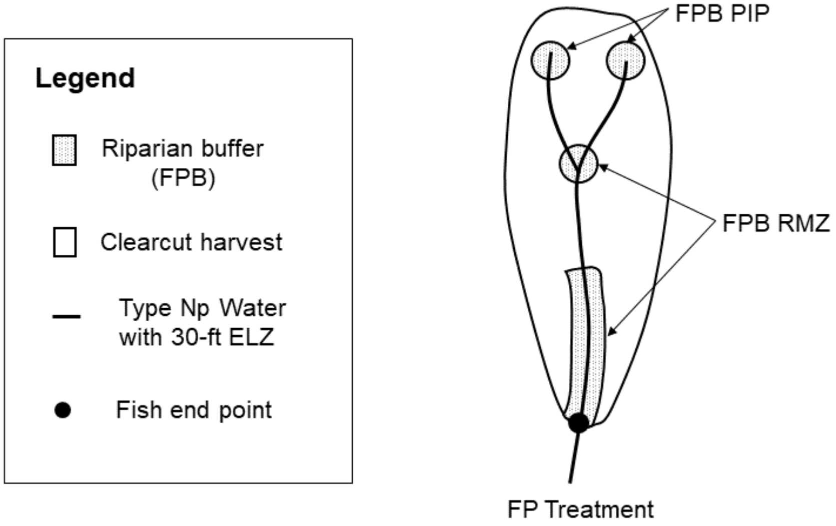

Figure 3-2. Simplified schematic of the Forest Practices (FP) treatment for the Type N Study, including riparian management zone (RMZ) reaches with buffers (FPB), and unbuffered reaches clearcut to the stream channel. Sensitive site buffers were also placed around tributary junctions (i.e., Type Np intersections) and uppermost points of perennial flow (PIPs).

Stand structural metrics were calculated for each RMZ and PIP plot for each pre and post-harvest sampling date. These included live density, basal area, percent live conifer basal area, quadratic mean diameter (Curtis and Marshall 2000), and relative density (Curtis 1982). RMZ and PIP plot values were averaged by site, and site means were averaged to generate treatment means. For each site we computed the percentage of regeneration plots with seedlings or saplings in Post 5 and Post 8.

For each RMZ and PIP plot we computed changes in density and basal area, tree mortality and ingrowth, and recruitment of fallen trees and large wood pieces. Changes were computed as (1) cumulative values for each of four pre- or post-harvest intervals (Pre 2 to Pre 1, and Pre 1 to Post 2, Post 5 and Post 8) and (2) as annualized rates over four distinct intervals (Pre 2 to Pre 1, Pre 1 to Post 2, Post 2 to Post 5 and Post 5 to Post 8 using a compounding formula (Sheil *et al.* 1995). Changes in live density and basal area were computed as initial minus final values. Ingrowth was expressed as the number of new trees achieving a dbh of at least 10.2 cm (4 in) between surveys. Tree mortality was expressed the percentage of live trees or live basal area that died over each interval.

For each plot, fallen trees and large wood pieces that recruited to the channel were summed for each interval. We calculated the combined in- and over-channel volume for each newly recruited large wood piece as a cylinder based on length and mid-point diameter. RMZ and PIP plot values were then averaged by site and site means averaged to generate treatment means.

For wood loading, we calculated the mean number of small and large pieces of wood per linear meter of stream for all wood (irrespective of function; **Table 3-1**) and functional wood (pieces categorized as step formation, bank stability or hydraulic roughness) for each site and year. To account for the presence of obstructed reaches, we generated separate means for unobstructed and obstructed reaches, then computed a weighted mean (pieces/m_{weight}) based on the proportions of stream length in the unobstructed and obstructed condition (**Eq. 3-1**).

$$pieces/m_{weight} = (pieces/m_{unobs} * \text{proportion unobstructed stream length}) + (pieces/m_{obs} * \text{proportion obstructed stream length}) \qquad \textbf{(Eq. 3-1)}$$

For channel wood cover we calculated the mean cover among sample intervals for each site and year.

3-3.2.1. Statistical Analysis

To detect differences among treatments, we used statistical models to make comparisons among the treatments and reference. RMZ and PIPs were analyzed separately for change in live basal area, tree mortality (% basal area) and large wood recruitment (pieces/ha of buffer). Separate models were run for each of three intervals: Pre 1 to Post 2, Post 5 or Post 8. We evaluated the null hypothesis:

$$T_{REF} = T_{100\%} = T_{FPB} \qquad \textbf{(Eq. 3-2)}$$

where: T_{REF}, $T_{100\%}$ and T_{FPB}, are values for reference, 100%, and FPB reaches, respectively.

Analysis of channel wood loading evaluated the generalized null hypothesis:

$$\Delta T_{REF} = \Delta T_{100\%} = \Delta T_{FP} = \Delta T_{0\%} \qquad \textbf{(Eq. 3-3)}$$

where: ΔT_{REF} is the change (post-harvest − pre-harvest) in the reference, and $\Delta T_{100\%}$, ΔT_{FP}, and $\Delta T_{0\%}$ are the changes in the 100%, FP and 0% treatments, respectively. We present the between-treatment comparison of change for the pre-treatment mean (Pre 3 and Pre 2) and Post 1 and 2 mean, pre-treatment and Post 5, and pre-treatment and Post 8.

Analysis of post-harvest channel wood cover, evaluated the null hypothesis:

$$T_{REF} = T_{100\%} = T_{FP} = T_{0\%} \qquad \textbf{(Eq. 3-4)}$$

where: T_{REF}, $T_{100\%}$, T_{FP}, and $T_{0\%}$ are post-harvest rates in the reference, 100%, FP and 0% treatments, respectively.

We used generalized linear mixed effects models (GLMM; McDonald *et al.* 2000) in SAS with the GLIMMIX procedure (except for channel wood cover, which used the MIXED procedure; SAS Institute Inc. 2013). Mixed models account for correlation associated with hierarchical nesting, as with the nesting of plots within sites, and sites within blocks. GLMM can be used to fit data that derive from non-normal distributions using monotonic link transformations. An added benefit of mixed models is that they accommodate missing data if those data are missing at random (SAS Institute Inc. 2013).

All models included treatment as a fixed effect and block as a random effect. For channel wood loading under the BACI design, the models incorporated site as a random effect with year and a treatment × year interaction term as fixed effects. For the RMZ and PIP analyses of change in live basal area, % mortality and large wood recruitment the random effect included site nested within block. We assumed that random effects were normally distributed (SAS Institute Inc. 2013).

We generally estimated model parameters using Restricted Maximum Likelihood. Channel wood cover was log-transformed prior to analysis to obtain an approximately Gaussian error distribution. All other responses used the appropriate link function in a GLMM (**Table 3-4**). Percent mortality was treated as a binary event/trial and analyzed using a binomial distribution and log link using restricted pseudo-likelihood.

We determined the covariance matrix for the fixed-effect parameter estimates and denominator degrees of freedom for t and F tests according to the method of Kenward and Roger (1997), which is recommended for unbalanced designs. We ran standard diagnostics to verify that model assumptions (e.g., normality and homoscedasticity of residuals) were met.

We evaluated the null hypothesis with a Wald-type test using linear contrasts of the model fixed effects. We constructed contrasts to test the difference in mean response for pre- and post-harvest periods, where period corresponded to all years in either the pre- or post-harvest condition. When the period × treatment contrast or treatment F-test had a P-value ≤0.1, we examined pairwise contrasts to test for differences among the combinations of references and buffer treatments. If the P-value for the period × treatment contrast or treatment F-test was >0.1, we did not report test results for these terms.

Table 3-4. The SAS Procedure and distribution/link for each response variable, where n is the number of subjects included in the analysis (e.g., sites). LW is large wood and SW is small wood.

Response Variable	Distribution/Link	Subjects (*n*)
Δ live basal area/ha-RMZ	Gaussian/Identity	13
Δ live basal area/ha-PIP	Gaussian/Identity	11
% mortality (basal area)-RMZ	Binomial/Logit	13
% mortality (basal area)-PIP	Binomial/Logit	11
LW recruitment (pieces/ha)-RMZ	Poisson/Log	13
LW recruitment (pieces/ha)-PIP	Poisson/Log	11
SW loading total pieces	Poisson/Log	17
SW loading functional pieces	Poisson/Log	17
LW loading total pieces	Poisson/Log	17
LW loading functional pieces	Poisson/Log	17
Channel wood cover	Gaussian/NA	17

For channel wood cover, we predicted that the proportion of the stream channel covered by newly recruited wood may differ by treatment and number of years since harvest, so we conducted three analyses, one for each post-harvest sample year. We hypothesized that a large amount of wood would enter the stream channel immediately post-harvest in the unbuffered FP and 0% treatment RMZs (reflected in Post 2 sampling), reflecting large increases in new wood cover in these treatments relative to the reference and 100% treatment. However, wood recruitment in these unbuffered reaches in subsequent years would be minimal, lacking a source of new wood recruitment from the clearcut RMZ. Alternatively, wood recruitment as a result of harvest was expected to be minimized by the RMZ buffers in the 100% and FP treatments immediately post-harvest, while future wood recruitment was expected to be greater in these buffered reaches compared with the unbuffered reaches of the FP and 0% treatments over time (as reflected in Post 5 and Post 8).

Both reference and treated sites were affected by the 2007 windstorm, which fortunately preceded treatment implementation (i.e., harvest). Because blocks were impacted differently, the blocking term helps to account for this variation.

3-4. RESULTS

3-4.1. PRE-HARVEST STAND STRUCTURE AND WOOD RECRUITMENT

Prior to harvest, riparian forests were typically dense, young (30- to 60-year old) second-growth stands. Among sites, live density ranged from 213 to 953 trees/ha (86 to 386 trees/ac) in RMZs and from 120 to 759 trees/ha (49 to 307 trees/ac) in PIPs. Basal area ranged from 26 to 73 m^2/ha (114 to 318 ft^2/ac) in the RMZs and from 12.2 to 69.6 m^2/ha (53.2 to 303.0 ft^2/ac) in the PIPs (**Appendix Table 3-1**). Stands in the WIL1, WIL2, WIL3 and OLYM blocks were dominated by western hemlock (*Tsuga heterophylla*) and Douglas-fir (*Pseudotsuga menziesii*). Western hemlock dominated the basal area in RMZs at all WIL1 and WIL2 sites, three of four OLYM sites, and one of two WIL3 sites. The pattern was similar for PIPs. Mean density and basal area were lower in the CASC sites, where Douglas-fir and red alder (*Alnus rubra*) were dominant (**Appendix Table 3-1** and **Appendix Table 3-2**).

Blocks differed in pre-harvest mortality and structural change (**Appendix Table 3-3**). A December 2007 storm with hurricane-force winds caused extensive mortality in the WIL1 and WIL2 blocks (~20% and ~10% of the initial basal area, respectively). Mortality was much lower (<2% of live basal area) in the OLYM, WIL3, and CASC blocks located outside the storm path (**Figure 3-3**; left panel). Consequently, pre-harvest structural changes were greater at the WIL1 and WIL2 blocks (basal area declines of ~10 and ~ 5 m^2/ha [44 and 22 ft^2/ac], respectively) than at the OLYM, WIL3 and CASC blocks (little change in basal area; **Figure 3-3**; right panel). Variability in mortality was especially high in WIL1 RMZs and WIL1 and WIL2 PIPs. Large wood recruitment followed a similar pattern among blocks: higher and variable at WIL1 and WIL2, lower at OLYM and WIL3, and lowest at CASC.

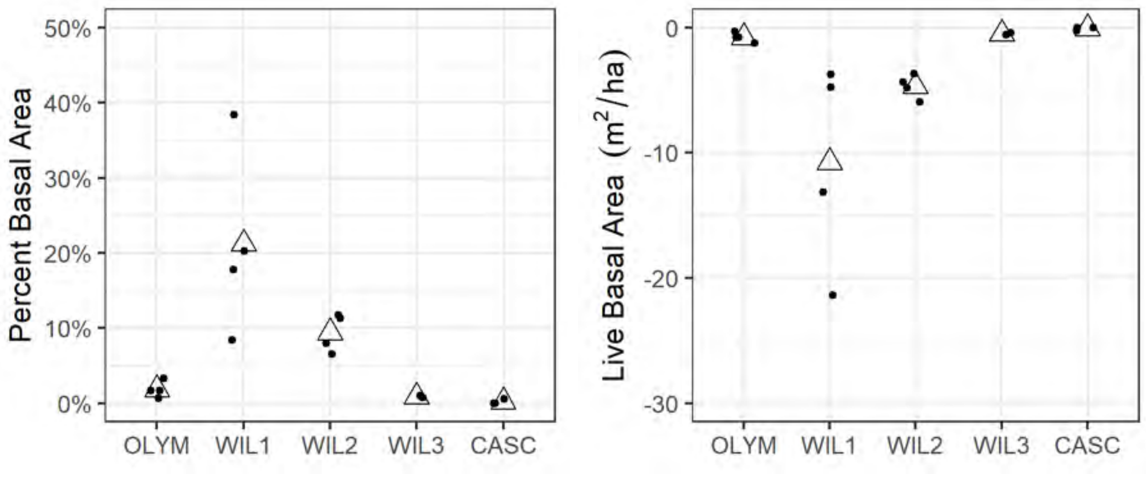

Figure 3-3. Pre-harvest mortality as a percentage of initial live basal area (left panel) and change in live basal area/ha prior to harvest (Pre 2 to Pre 1) for RMZs (right panel). Dots are site means and triangles are block averages computed from site means.

3-4.2. STAND RESPONSE

3-4.2.1. Change in Stand Structure

Stand structure in year 8 reflected a combination of pre-treatment structure, effects of harvest, and change over the post-harvest interval. The greatest changes were in 0% and unbuffered portions of FP reaches where nearly all trees were removed. Given the absence of trees in these areas, we focus on the post-harvest responses of the reference (REF), 100%, and buffered portions of the FP treatments (FPB).

In the 100% and FPB RMZs, density, basal area, and RD decreased over the post-harvest interval (**Figure 3-4**; **Appendix Table 3-4**). The magnitude of change was greatest in FPB, where density, basal area and RD decreased by 59%, 55% and 54%, respectively. For the same variables, reductions in the 100% RMZs were 30%, 14%, and 17%, respectively (**Figure 3-5; Appendix Table 3-5**). In contrast, stand structure in the reference RMZs was more stable, with a 17% decrease in density and little change in basal area or RD. The smaller reductions in the 100% RMZs made stand structure more similar to the reference RMZs in Post 8, while the differences between the FPB and reference RMZs increased over time. There was little variability in change in density or basal area among reference, in contrast to the substantial variability among 100% and FPB (**Appendix Table 3-5**). Change in live basal area did not differ statistically between 100% and REF RMZs for any interval although the differences increased over time (**Table 3-5**). The FPB–REF contrast was not significant in the first interval, but it was in subsequent intervals as the magnitude of change in FPB RMZs increased over time. The FPB–100% contrast was not significant until the last interval when basal area stabilized in the 100% treatment but continued to decline in FPB. The dominant species remained unchanged in the REF, 100% and FPB RMZs over the 8-year interval, and the proportion of basal area by species changed by less than 10%.

Table 3-5. Between-treatment comparisons (estimate, 95% confidence interval, and P-value) of the cumulative change in live basal area (m^2/ha) in RMZs and PIPs between Pre 1 and the Post 2, Post 5, or Post 8 samples. Estimates with P-values ≤0.10 are in bold font.

	Pre 1–Post 2		Pre 1–Post 5		Pre 1–Post 8	
Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
			RMZs			
100% vs. REF	-2.9 (-16.9, 11.0)	0.655	-6.0 (-20.0, 8.0)	0.366	-6.8 (-20.8, 7.1)	0.306
FPB vs. REF	-10.2 (-25.5, 5.2)	0.173	**-16.1 (-31.4, -0.8)**	**0.041**	**-21.1 (-36.4, -5.8)**	**0.011**
FPB vs. 100%	-7.2 (-23.8, 9.3)	0.359	-10.1 (-26.7, 6.4)	0.207	**-14.3 (-30.8, 2.3)**	**0.085**
			PIPs			
100% vs. REF	-8.2 (-21.7, 5.4)	0.208	**-16.2 (-29.7, -2.6)**	**0.024**	**-22.0 (-35.6, -8.5)**	**0.005**
FPB vs. REF	**-13.7 (-27.3, -0.2)**	**0.047**	**-18.7 (-32.3, -5.2)**	**0.012**	**-23.8 (-37.3, -10.2)**	**0.003**
FPB vs. 100%	-5.6 (-20.7, 9.6)	0.433	-2.6 (-17.7, 12.5)	0.711	-1.7 (-16.9, 13.4)	0.804

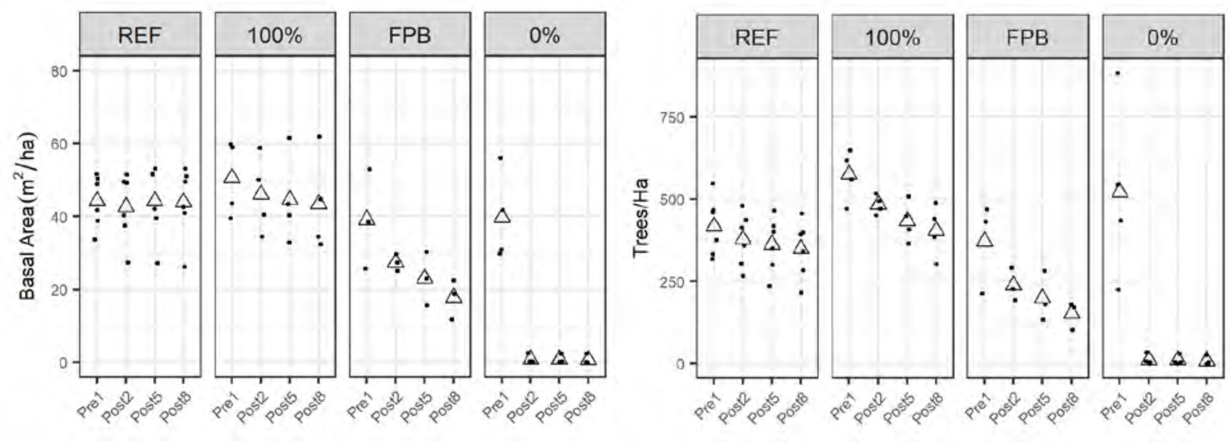

Figure 3-4. Live basal area (left panel) and density (right panel) in RMZs. Dots are site means and triangles are treatment averages computed from site means.

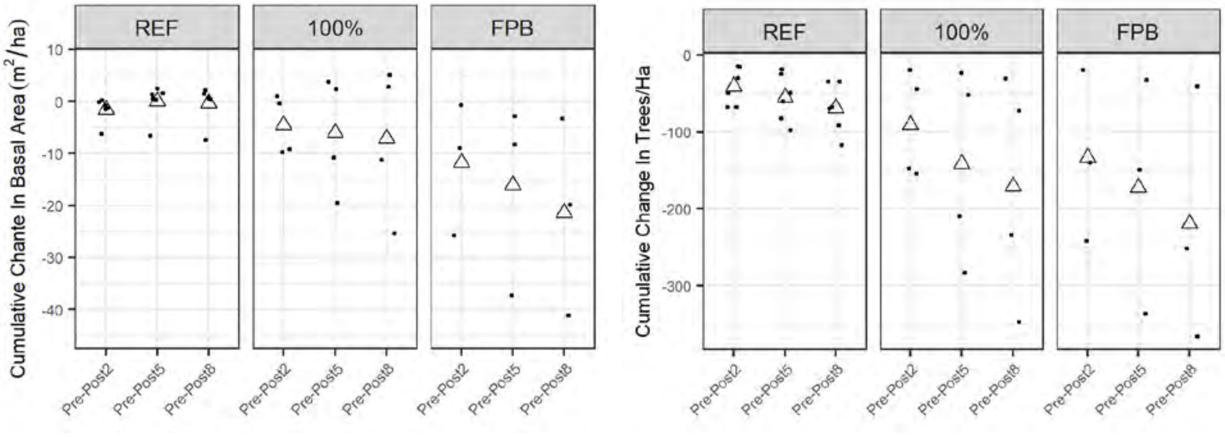

Figure 3-5. Cumulative change in live basal area (left panel) and density (right panel) for RMZs. Dots are site means and triangles are treatment averages computed from site means.

Among PIPs, stand structure changed substantially by Post 8 in the 100% and FPB PIPs while there was little change in the reference. The magnitude of change was greatest in the FPB where density, basal area and RD decreased by 70%, 53% and 58%, respectively. Declines were somewhat lower in the 100%, i.e., 51%, 38% and 42%, respectively. In the reference, density decreased by 14% while basal area increased slightly (**Appendix Table 3-4** and **Appendix Table 3-5**). Variability among sites was greater in the 100% and FPB than in the reference. The FPB–REF contrast was significant for all intervals. The 100%–REF contrast was not significant in the first interval, but was in later intervals as the magnitude of change increased in the 100% PIPs. There were no significant differences between FPB and 100% PIPs for any interval (**Table 3-5**).

3-4.2.2. Tree Mortality

Tree mortality was the primary driver of post-harvest structural change in both RMZs and PIPs. In the RMZs, mortality over the Pre 1 to Post 8 interval as a percentage of pre-harvest basal area was lower in the reference (16.1%) than in the 100% (24.3%) and FPB (50.8%) (**Figure 3-6**; **Appendix Table 3-5**). The FPB–REF contrast was not significant for the Pre 1 to Post 2 interval, but it was in subsequent intervals as mortality in FPB increased relative to the reference (**Table 3-6**). The remaining contrasts—100% vs. REF and FPB vs. 100%—were not significant for any interval. It appears that high variability in mortality among 100% and FPB sites reduced the power of the tests to detect differences. The pattern was similar for mortality as a percentage of live tree count.

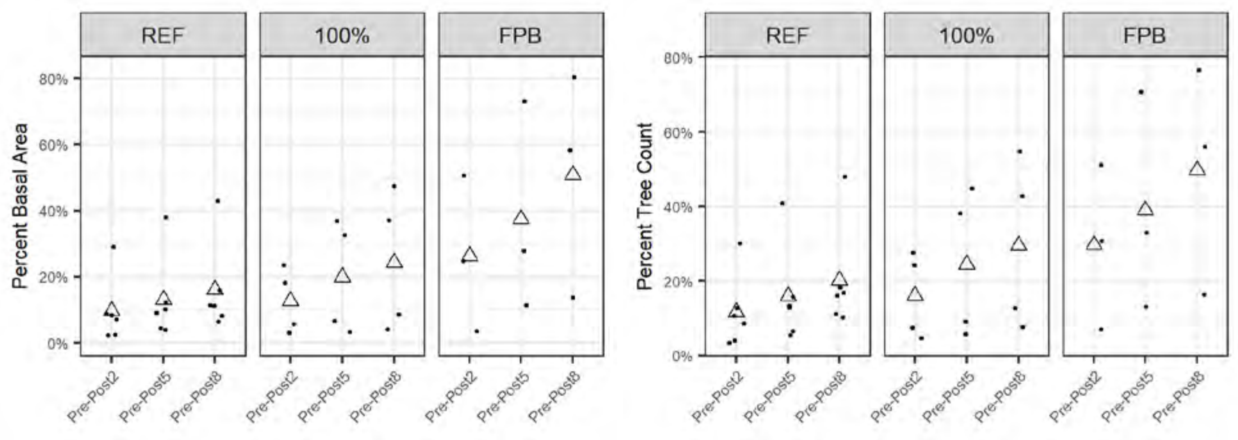

Figure 3-6. Cumulative tree mortality as a percentage of pre-harvest live basal area (left panel) and tree count (right panel) in RMZs between Pre 1 and the Post 2, Post 5, and Post 8 samples. Dots are site means and triangles are treatment averages computed from site means.

Treatment differences were more pronounced in the PIPs. Cumulative mortality over the Pre 1 to Post 8 interval was 9.4%, 43.0% and 56.4% for the reference, 100% treatment and FPB PIPs, respectively (**Appendix Table 3-5**). The pattern was similar, but values higher, for mortality as a percentage of live tree count. The REF–FPB contrast for mortality as a percentage of pre-harvest live basal area was significant for all intervals (**Table 3-6**). The 100%–REF contrast was not significant during the initial interval but was in subsequent intervals as mortality increased over time. There were no significant differences in mortality between 100% and FPB PIPs.

Table 3-6. Between-treatment comparisons (estimate, 95% confidence intervals, and P-value) of cumulative tree mortality (percent of live basal area) in RMZs and PIPs between Pre 1 and the Post 2, Post 5, or Post 8 samples. Estimates with P-values ≤0.10 are in bold font.

	Pre 1–Post 2		**Pre 1–Post 5**		**Pre 1–Post 8**	
Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
			RMZs			
100% vs. REF	-0.40 (-0.77, -0.12)	0.584	-0.35 (-0.73, -0.10)	0.400	-0.36 (-0.73, -0.10)	0.432
FPB vs. REF	-0.19 (-0.58, -0.04)	0.103	**-0.17 (-0.54, -0.03)**	**0.073**	**-0.10 (-0.41, -0.02)**	**0.023**
FPB vs. 100%	-0.26 (-0.71, -0.05)	0.262	-0.28 (-0.72, -0.05)	0.290	-0.17 (-0.59, -0.03)	0.102
			PIPs			
100% vs. REF	-0.23 (-0.67, -0.04)	0.191	**-0.18 (-0.59, -0.03)**	**0.098**	**-0.14 (-0.51, -0.02)**	**0.054**
FPB vs. REF	**-0.06 (-0.30, -0.01)**	**0.010**	**-0.08 (-0.36, -0.01)**	**0.017**	**-0.07 (-0.32, -0.01)**	**0.012**
FPB vs. 100%	-0.17 (-0.63, -0.02)	0.123	-0.29 (-0.77, -0.05)	0.349	-0.31 (-0.79, -0.05)	0.406

There was a temporal pattern to mortality in 100% and FPB RMZs and PIPs. Annual rates of mortality as percentage of live basal area and density were highest in the first two years after harvest, then decreased. Stand structure showed a similar trend, with changes in density, basal area and relative density (RD) in the 100% and FPB RMZs and PIPs declining over time (**Appendix Table 3-6**).

Wind/physical damage was the primary cause of mortality in both the RMZ and PIP buffers. In the 100% treatment it accounted for 78% and 90% of the loss of basal area, respectively; in FPB it accounted for 78% and 65% of the loss. Wind accounted for a smaller proportion of mortality in reference RMZ and PIPs (52% and 43%, respectively).

3-4.2.3. Regeneration and Ingrowth

Natural regeneration of tree seedlings and saplings was widespread in the reference, 100%, and FPB RMZs and PIPs (**Appendix Table 3-4**). Regeneration was dominated by conifers; western hemlock was the most common species at most sites. The mean proportion of RMZ regeneration plots with conifer regeneration in Post 8 was greatest in 100% (38%), lower in references (26%), and lowest in FPB (21%) RMZs. Among PIPs the proportion of plots with conifer regeneration was greater in 100% and FPB PIPs (53 and 47%, respectively) than in references (23%). Post-harvest ingrowth (new trees reaching the 10-cm dbh threshold) averaged less than 11 trees/ha in the REF, 100% and FPB RMZs and PIPs (**Appendix Table 3-5**).

In the harvested portions of RMZs (0% and unbuffered FP reaches) regeneration derived from planted seedlings and natural recruitment. At Post 8, 42% of plots in the 0% treatment and 33% of plots in unbuffered FP reaches had conifer regeneration. No ingrowth occurred through year 8, however trees planted after harvest were growing rapidly and nearing the 10-cm dbh threshold.

3-4.3. LARGE WOOD RECRUITMENT

Large wood recruitment to the channel was greater in the 100% and FPB RMZs than in the reference for each pre- to post-harvest interval (**Appendix Table 3-5; Figure 3-7**). Over the Pre 1 to Post 8 interval, mean recruitment of large wood volume was two to nearly three times greater in 100% and FPB RMZs than in the references. However, none of the treatment contrasts differed for any post-harvest interval (**Table 3-7**), likely due to the large variation among 100% and FPB sites in the first interval (**Figure 3-7**).

The pattern was similar in the PIPs (**Appendix Table 3-5**). Variability was much greater PIPs than in the reference in the first interval (**Table 3-7**), but not subsequently as large wood recruitment declined.

Annual recruitment rates for fallen trees and large wood followed patterns similar to those of mortality. Rates were greatest during the first two years, then decreased (**Appendix Table 3-6**). During the Pre 1–Post 2 interval, recruitment rates for large wood volume were three and two times greater in 100% and FPB RMZs than in the reference, respectively. The differences were greater in the PIPs, where 100% and FPB rates were 26 and 19 times the reference.

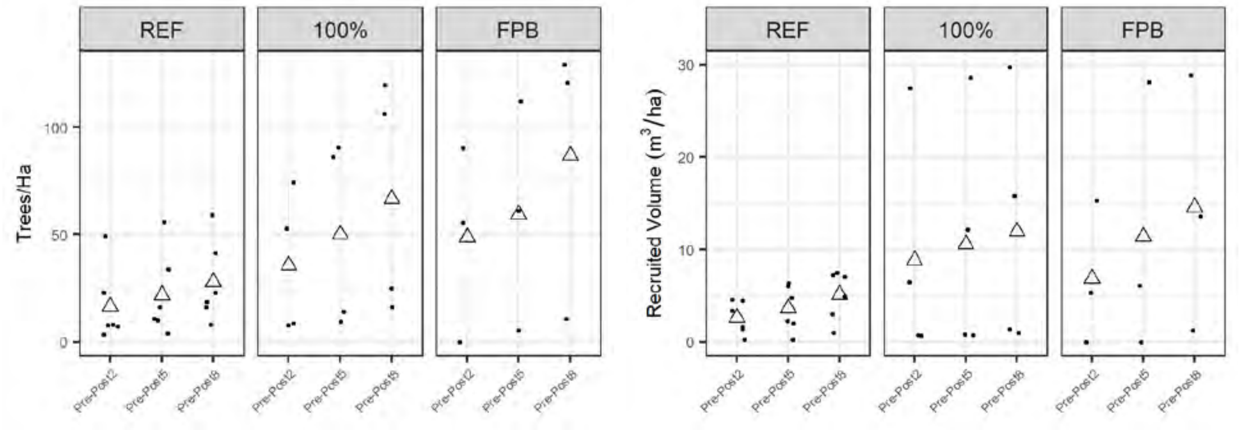

Figure 3-7. Cumulative recruitment of fallen trees (left panel) and large wood by volume (right panel) in RMZs between Pre 1 and the Post 2, Post 5, or Post 8 samples. Dots are site means and triangles are treatment averages computed from site means.

Table 3-7. Between-treatment comparisons (estimate, 95% confidence interval, and P-value) of cumulative large wood recruitment (pieces/ha) in RMZs and PIPs between Pre 1 and the Post 2, Post 5 and Post 8 samples. Estimates with P-values ≤0.10 are in bold font.

	Pre 1–Post 2		**Pre 1–Post 5**		**Pre 1–Post 8**	
Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
			RMZs			
100% vs. REF	-0.59 (-2.67, -0.13)	0.452	-0.53 (-2.39, -0.12)	0.368	-0.48 (-2.18, -0.11)	0.304
FPB vs. REF	-0.50 (-2.68, -0.09)	0.384	-0.53 (-2.85, -0.10)	0.425	-0.38 (-2.03, -0.07)	0.230
FPB vs. 100%	-0.85 (-5.19, -0.14)	0.850	-1.01 (-6.13, -0.17)	0.989	-0.80 (-4.80, -0.13)	0.784
			PIPs			
100% vs. REF	-0.09 (-0.61, -0.01)	**0.019**	-0.22 (-1.47, -0.03)	0.103	-0.30 (-1.98, -0.05)	0.176
FPB vs. REF	-0.18 (-1.29, -0.02)	**0.082**	-0.48 (-3.22, -0.07)	0.402	-0.77 (-5.09, -0.12)	0.753
FPB vs. 100%	-2.07 (-16.95, -0.25)	0.445	-2.15 (-17.61, -0.26)	0.418	-2.52 (-20.62, -0.31)	0.330

In general, fewer than half of the trees that fell during the post-harvest interval recruited to the channel. The proportion was greater in the REF (49%) than in the 100% and FPB RMZs (42% and 45%, respectively). Of those that reached the channel, >80% were spanning or suspended. The proportion that intruded into the bankfull channel was higher in the REF (19%) than in the 100% or FPB RMZs (14 and 15%, respectively).

3-4.4. CHANNEL WOOD LOADING AND COVER

3-4.4.1. Small Wood Loading

Small wood was abundant in headwater study sites. Mean small wood loading (pieces/m of channel length) ranged from 1.7 to 8.2 in the pre-harvest period, from 3.5 to 42.1 in Post 1 and Post 2, and from 6.8 to 58.7 and 4.2 to 26.3 in Post 5 and Post 8, respectively (**Figure 3-8**). Treatments differed significantly in the magnitude of change over time (P <0.001; **Table 3-8**; **Table 3-9; Figure 3-9**). We observed an increase in small wood loading across all treatments in Post 1 and Post 2, with the greatest increase in the 0% treatment (176%; P <0.001), compared with the pre-harvest period and after controlling for temporal changes in the references. The increase in the 0% treatment was also greater (64%; P = 0.02, and 74%; P = 0.10) than in the 100% and FP treatments, after adjusting for pre-harvest differences. Small wood loading continued to increase in the FP and 0% treatments through Post 5 (68%; P = 0.09, and 142%; P <0.001), after controlling for temporal changes in the references (**Table 3-8**), and the values for these treatments were also greater than in the 100% treatment, after adjusting for pre-harvest differences. In Post 8, we observed a decline in small wood loading across all buffer treatments and did not find clear evidence of differences in the change among treatments (P >0.10 for all comparisons).

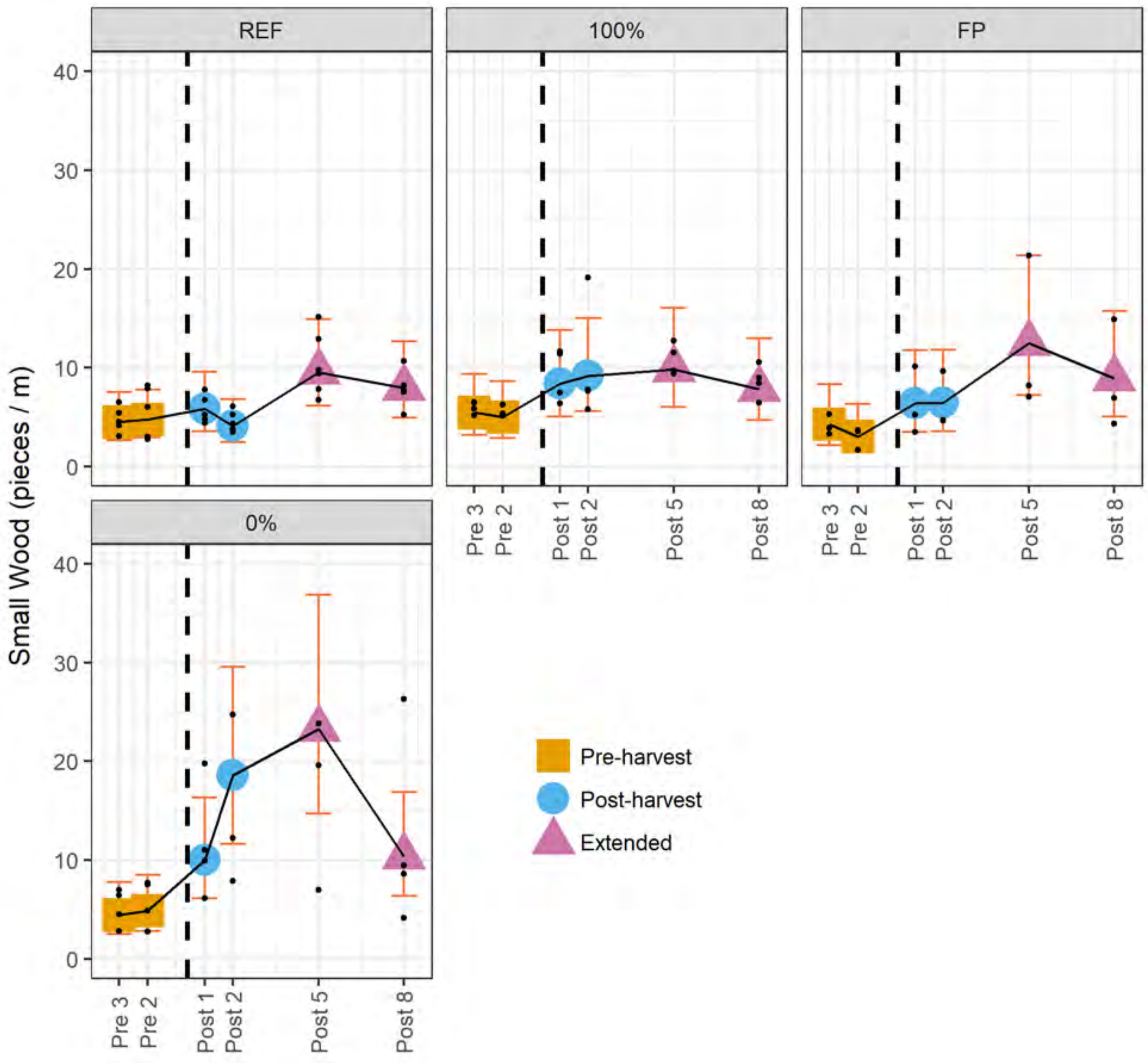

Figure 3-8. Mean small wood pieces/m of channel length by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. To ensure a y-axis scale that highlights the variability and is consistent among panels, outliers are not shown for one 0% site (WIL2-0%) in both Post 2 (42.1 pieces/m) and Post 5 (58.7 pieces/m). Site means are dots; treatment means are colored symbols.

Table 3-8. The within-treatment estimate of the proportional change and 95% confidence intervals (CI) for small wood pieces between the pre-harvest and Post 1 and Post 2, Post 5, and Post 8 samples.

	Post 1 & 2	Post 5	Post 8
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.06 (0.75, 1.50)	2.06 (1.47, 2.88)	1.71 (1.16, 2.52)
100% (n = 4)	1.68 (1.20, 2.34)	1.88 (1.29, 2.76)	1.50 (1.00, 2.24)
FP (n = 3)	1.79 (1.10, 2.92)	3.45 (2.11, 5.65)	2.47 (1.46, 4.19)
0% (n = 4)	2.92 (2.13, 4.01)	4.99 (3.62, 6.89)	2.22 (1.53, 3.24)

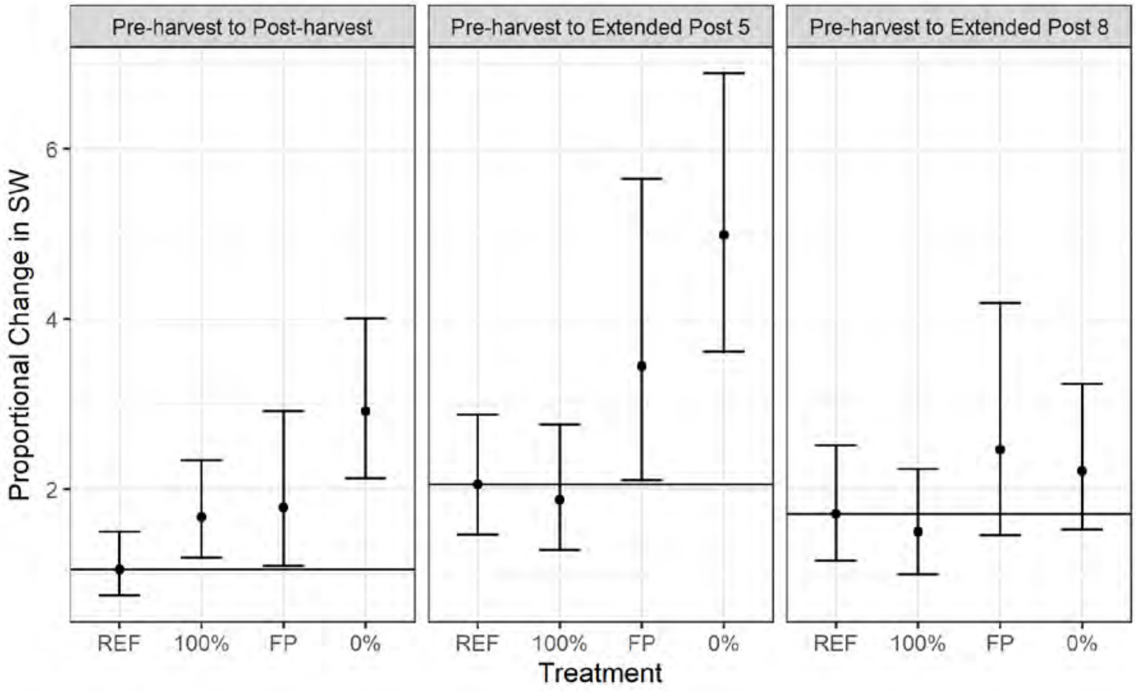

Figure 3-9. The within-treatment estimate of proportional change and 95% confidence intervals for small wood (SW) pieces between the pre-harvest and post-harvest (Post 1 and Post 2), Post 5, and Post 8 samples. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 3-9. The between-treatment comparison of the proportional change, 95% confidence intervals (CI) and P-values (P) for small wood pieces between the pre-harvest and Post 1 and Post 2, Post 5, and Post 8 samples. Estimates with P-values ≤0.10 are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	**Post 1 & 2**		**Post 5**		**Post 8**	
	Estimate (CI)	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
100% vs. REF	**1.58** (0.98, 2.57)	**0.06**	0.91 (0.55, 1.52)	0.73	0.87 (0.50, 1.53)	0.63
FP vs. REF	**1.69** (0.93, 3.08)	**0.09**	**1.68** (0.92, 3.04)	**0.09**	1.44 (0.75, 2.78)	0.27
0% vs. REF	**2.76** (1.72, 4.42)	**<0.001**	**2.42** (1.52, 3.86)	**<0.001**	1.30 (0.76, 2.23)	0.34
0% vs. FP	**1.64** (0.91, 2.93)	**0.10**	1.45 (0.80, 2.61)	0.21	0.90 (0.47, 1.72)	0.75
0% vs. 100%	**1.74** (1.10, 2.77)	**0.02**	**2.65** (1.61, 4.36)	**<0.001**	1.49 (0.86, 2.58)	0.16
FP vs. 100%	1.07 (0.59, 1.93)	0.83	**1.83** (0.98, 3.41)	**0.06**	1.65 (0.85, 3.21)	0.14

Small wood frequently provided in-channel function, contributing to step formation, bank stability and hydraulic roughness. For small wood that provided these functions, mean loading (pieces/m of channel length) ranged from 1.0 to 5.1 in the pre-harvest period, 1.5 to 16.6 in Post 1 and Post 2, and 3.1 to 24.0 and 2.5 to 17.2 in Post 5 and Post 8, respectively (**Figure 3-10**). Treatments differed in the magnitude of change over time (P <0.01; **Table 3-10**; **Table 3-11; Figure 3-11**). In Post 1 and Post 2, we estimated a 69% (P = 0.03), 72% (P = 0.07) and 131% (P <0.001) increase for the 100%, FP and 0% treatments, respectively, compared with the pre-harvest period and after controlling for temporal changes in the references. Functional small wood loading continued to increase in the FP and 0% treatments through Post 5, with an estimated 75% (P = 0.06) and 136% (P <0.001) increase compared to the pre-harvest period, after controlling for temporal changes in the references. The estimates for the FP and 0% treatments were also greater than the 100% treatment, after adjusting for pre-harvest differences. The response of functional small wood loading differed from that of the total in that we continued to observe treatment differences in Post 8. Estimates for the FP and 0% treatments, which did not differ from the reference, were greater than for the 100% treatment, after adjusting for pre-harvest differences.

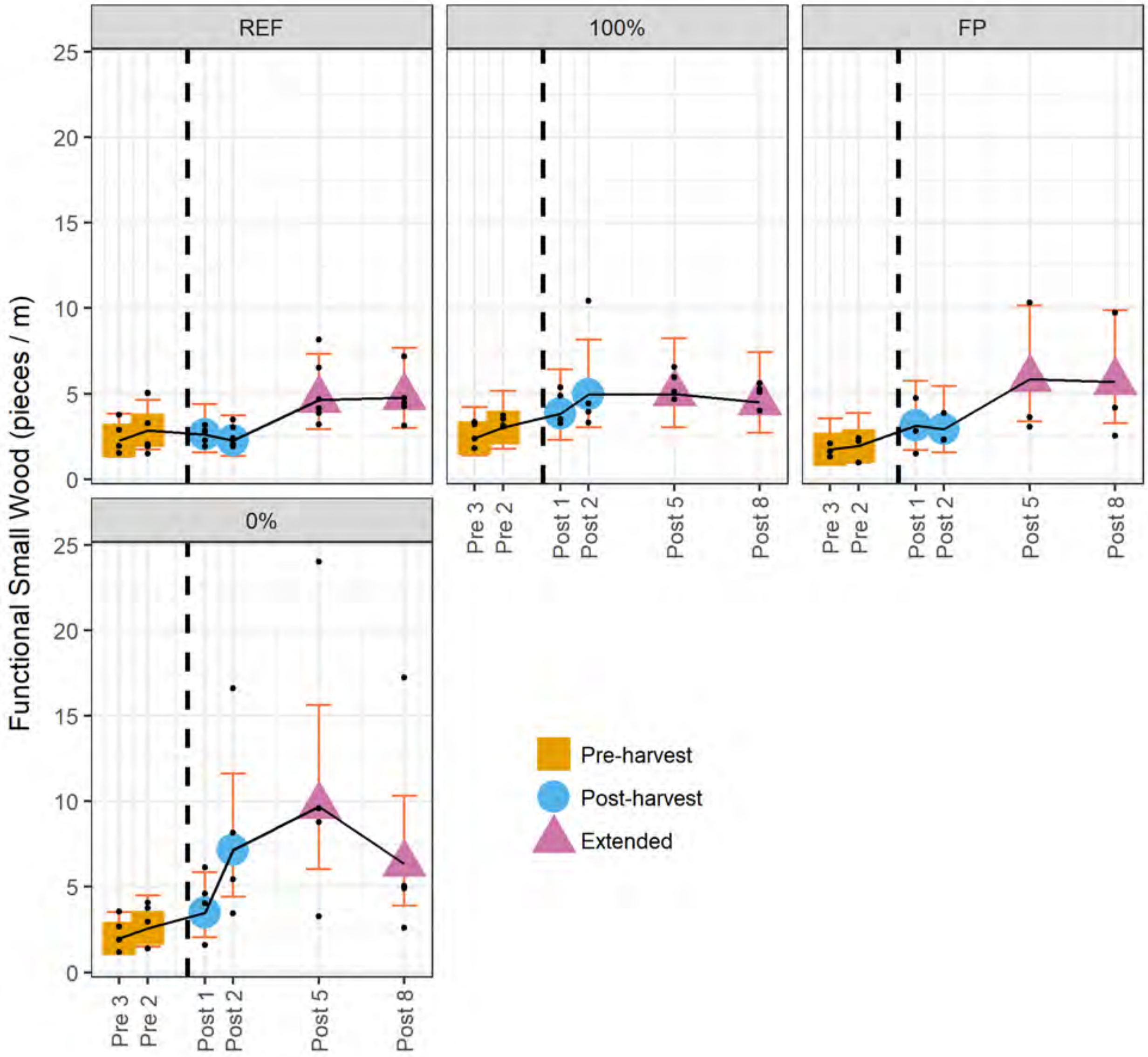

Figure 3-10. Mean functional small wood pieces/m of channel length by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended incldues Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 3-10. The within-treatment estimate of the proportional change and 95% confidence intervals (CI) for functional small wood pieces between the pre-harvest and Post 1 and Post 2, Post 5, and Post 8 samples.

Treatment	Post 1 & 2	Post 5	Post 8
	Estimate (CI)	Estimate (CI)	Estimate (CI)
REF (n = 6)	0.96 (0.68, 1.34)	1.82 (1.32, 2.53)	1.89 (1.32, 2.70)
100% (n = 4)	1.62 (1.17, 2.25)	1.86 (1.28, 2.69)	1.66 (1.14, 2.44)
FP (n = 3)	1.65 (1.01, 2.69)	3.19 (1.95, 5.22)	3.09 (1.89, 5.08)
0% (n = 4)	2.21 (1.58, 3.08)	4.30 (3.09, 5.99)	2.80 (1.96, 4.01)

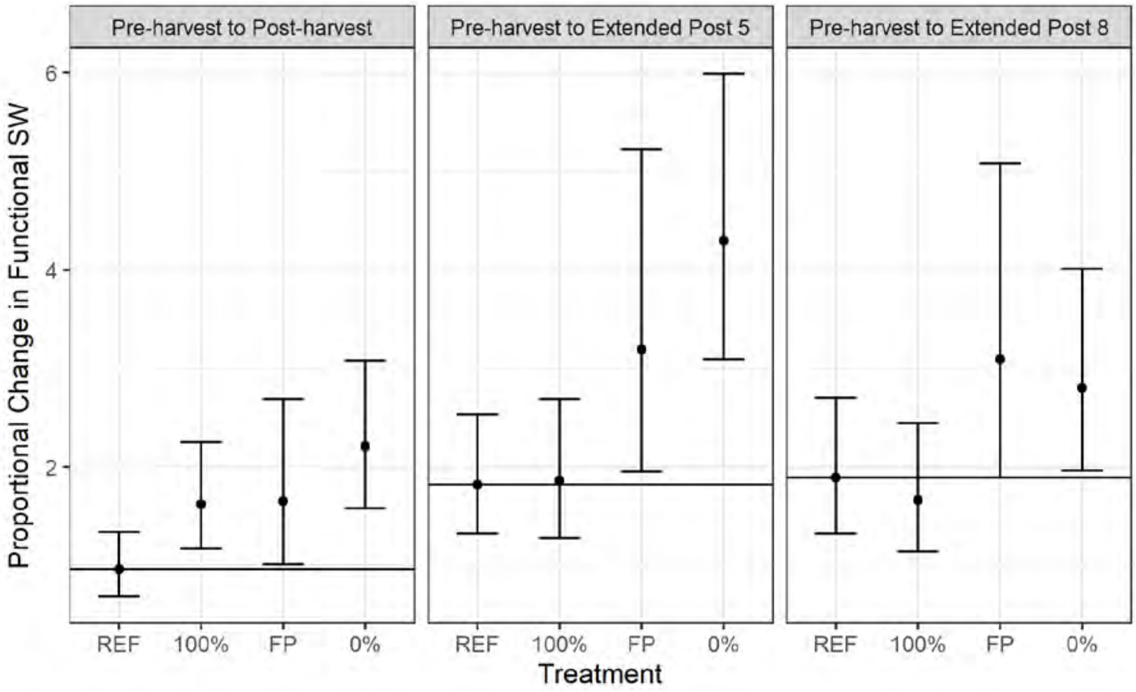

Figure 3-11. The within-treatment estimate of proportional change and 95% confidence intervals for functional small wood (SW) pieces between the pre-harvest and post-harvest (Post 1 and Post 2), Post 5, and Post 8 samples. A horizontal line is placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 3-11. The between-treatment comparison of the proportional change, 95% confidence intervals (CI) and P-values (P) for functional small wood pieces between the pre-harvest and Post 1 and Post 2, Post 5, and Post 8 samples. Estimates with P-values ≤0.10 are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	**Post 1 & 2**		**Post 5**		**Post 8**	
	Estimate (CI)	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
100% vs. REF	**1.69** (1.05, 2.70)	**0.03**	1.02 (0.62, 1.67)	0.95	0.88 (0.52, 1.49)	0.63
FP vs. REF	**1.72** (0.95, 3.11)	**0.07**	**1.75** (0.97, 3.16)	**0.06**	1.64 (0.89, 3.02)	0.11
0% vs. REF	**2.31** (1.43, 3.71)	**<0.001**	**2.36** (1.48, 3.75)	**<0.001**	1.48 (0.89, 2.46)	0.13
0% vs. FP	1.34 (0.74, 2.43)	0.33	1.35 (0.74, 2.44)	0.32	0.91 (0.49, 1.67)	0.75
0% vs. 100%	1.36 (0.85, 2.18)	0.19	**2.32** (1.41, 3.81)	**<0.001**	**1.68** (1.00, 2.84)	**0.05**
FP vs. 100%	1.02 (0.56, 1.83)	0.95	**1.72** (0.93, 3.18)	**0.08**	**1.86** (0.99, 3.47)	**0.05**

3-4.4.2. Large Wood Loading

Large wood was also abundant in headwater study sites. Mean large wood loading (pieces/m of channel length) ranged from 0.5 to 2.2 in the pre-harvest period, from 0.6 to 4.7 in Post 1 and Post 2, and from 0.8 to 4.7 and 0.9 to 3.7 in Post 5 and Post 8, respectively (**Figure 3-12**). Treatments differed significantly in the magnitude of change over time (P <0.01; **Table 3-12**; **Table 3-13**; **Figure 3-13**). We observed a 66% (P <0.001), 44% (P = 0.05) and 47% (P = 0.01) increase in mean large wood density in the 100%, FP and 0% treatments, respectively, in Post 1 and Post 2 compared with the pre-harvest period and after controlling for temporal changes in the references. We continued to see an increase in the FP (42%; P = 0.08) and 0% (32%; P = 0.8) treatments in Post 5 after controlling for temporal changes in the references (**Table 3-12**). In Post 8, only the FP treatment differed from the reference in its proportional increase (41%; P = 0.09).

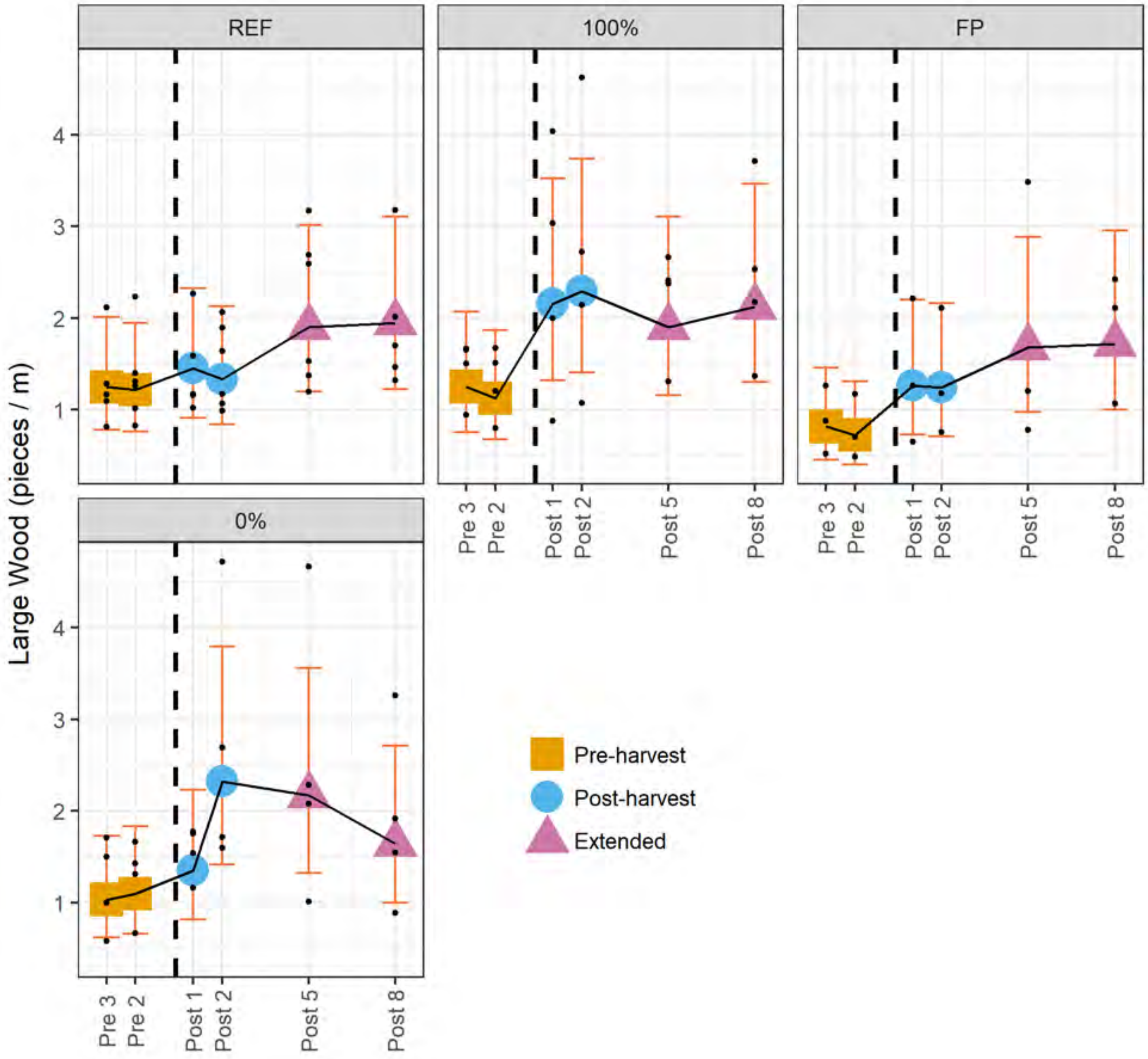

Figure 3-12. Mean large wood pieces/m of channel length by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 3-12. The within-treatment estimate of the proportional change and 95% confidence intervals (CI) for large wood pieces between the pre-harvest and Post 1 and Post 2, Post 5, and Post 8 samples.

Treatment	Post 1 & 2	Post 5	Post 8
	Estimate (CI)	Estimate (CI)	Estimate (CI)
REF (n = 6)	1.13 (0.93, 1.38)	1.54 (1.25, 1.89)	1.58 (1.26, 1.99)
100% (n = 4)	1.87 (1.54, 2.29)	1.60 (1.26, 2.04)	1.79 (1.42, 2.26)
FP (n = 3)	1.63 (1.21, 2.21)	2.18 (1.57, 3.04)	2.24 (1.61, 3.10)
0% (n = 4)	1.66 (1.34, 2.06)	2.04 (1.61, 2.59)	1.54 (1.19, 2.00)

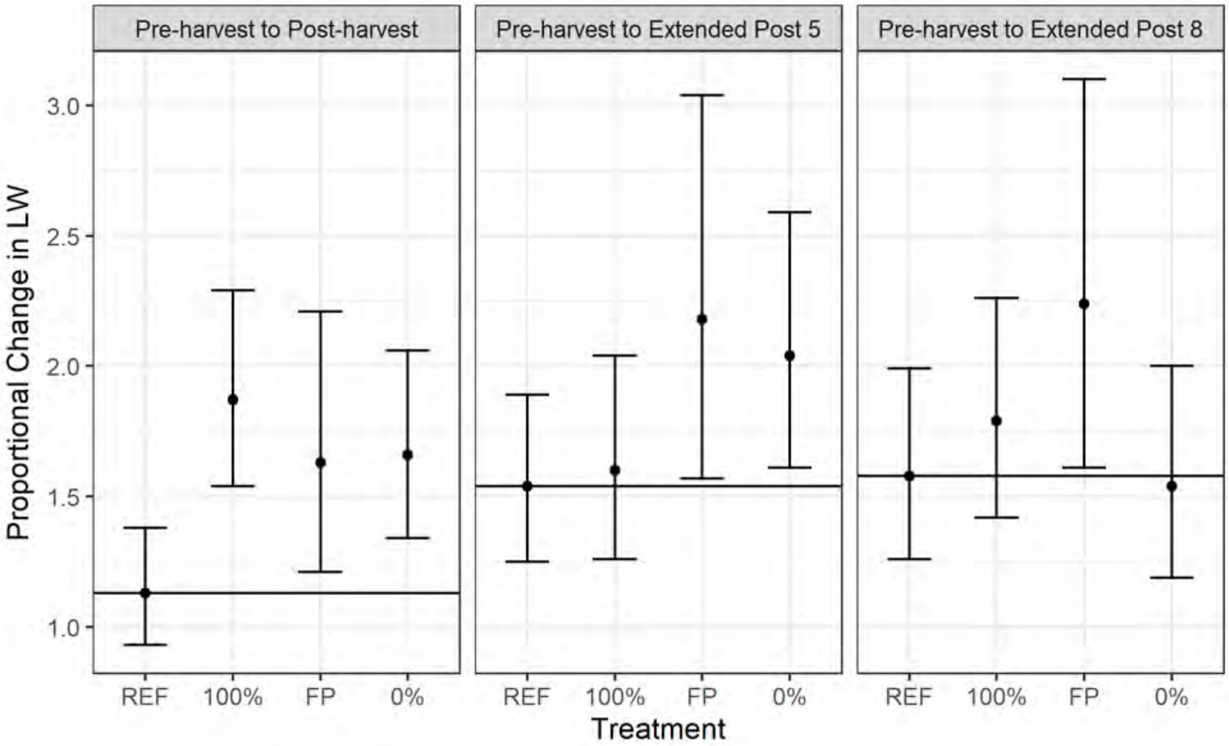

Figure 3-13. The within-treatment estimate of proportional change and 95% confidence intervals for large wood (LW) pieces between the pre-harvest and post-harvest (Post 1 and Post 2), Post 5 and Post 8 samples. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 3-13. The between-treatment comparison of the proportional change, 95% confidence intervals (CI) and P-values (P) for large wood pieces between the pre-harvest and Post 1 and Post 2, Post 5, and Post 8 samples. Estimates with P-values ≤0.10 are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	**Post 1 & 2**		**Post 5**		**Post 8**	
	Estimate (CI)	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
100% vs. REF	**1.66 (1.25, 2.20)**	**<0.001**	1.04 (0.76, 1.43)	0.81	1.13 (0.82, 1.57)	0.45
FP vs. REF	**1.44 (1.01, 2.07)**	**0.05**	**1.42 (0.96, 2.09)**	**0.08**	**1.41 (0.95, 2.11)**	**0.09**
0% vs. REF	**1.47 (1.10, 1.97)**	**0.01**	**1.32 (0.97, 1.82)**	**0.08**	0.98 (0.69, 1.38)	0.89
0% vs. FP	1.02 (0.70, 1.48)	0.92	0.93 (0.62, 1.40)	0.74	**0.69 (0.46, 1.05)**	**0.08**
0% vs. 100%	0.89 (0.66, 1.19)	0.41	1.27 (0.91, 1.79)	0.16	0.86 (0.61, 1.22)	0.40
FP vs. 100%	0.87 (0.61, 1.25)	0.44	1.36 (0.91, 2.05)	0.13	1.25 (0.84, 1.87)	0.27

Large wood also provided in-channel function in headwater sites, contributing to step formation, bank stability and hydraulic roughness. Mean functional large wood loading (pieces/m of channel length) ranged from 0.4 to 2.1 in the pre-harvest period, from 0.5 to 3.6 in Post 1 and Post 2, and from 0.7 to 4.2, and 0.7 to 2.8 in Post 5 and Post 8, respectively (**Figure 3-14**). Treatments differed significantly in the magnitude of change over time (P <0.01; **Table 3-14**; **Table 3-15; Figure 3-15**). Functional large wood loading increased by 61% (P <0.01), 37% (P = 0.09) and 33% (P = 0.06) in the 100%, FP and 0% treatments, respectively, compared with the pre-harvest period and after controlling for temporal changes in the references. In Post 5, only the 0% treatment differed from the reference, with an estimated 35% (P = 0.06) increase over the pre-harvest period (**Table 3-14**). Functional large wood loading in the 0% treatment also differed from the 100% treatment (i.e., 35% increase; P = 0.08), after adjusting for pre-harvest differences. In Post 8, we did not find clear evidence of differences in the change among treatments (P >0.10 for all comparisons).

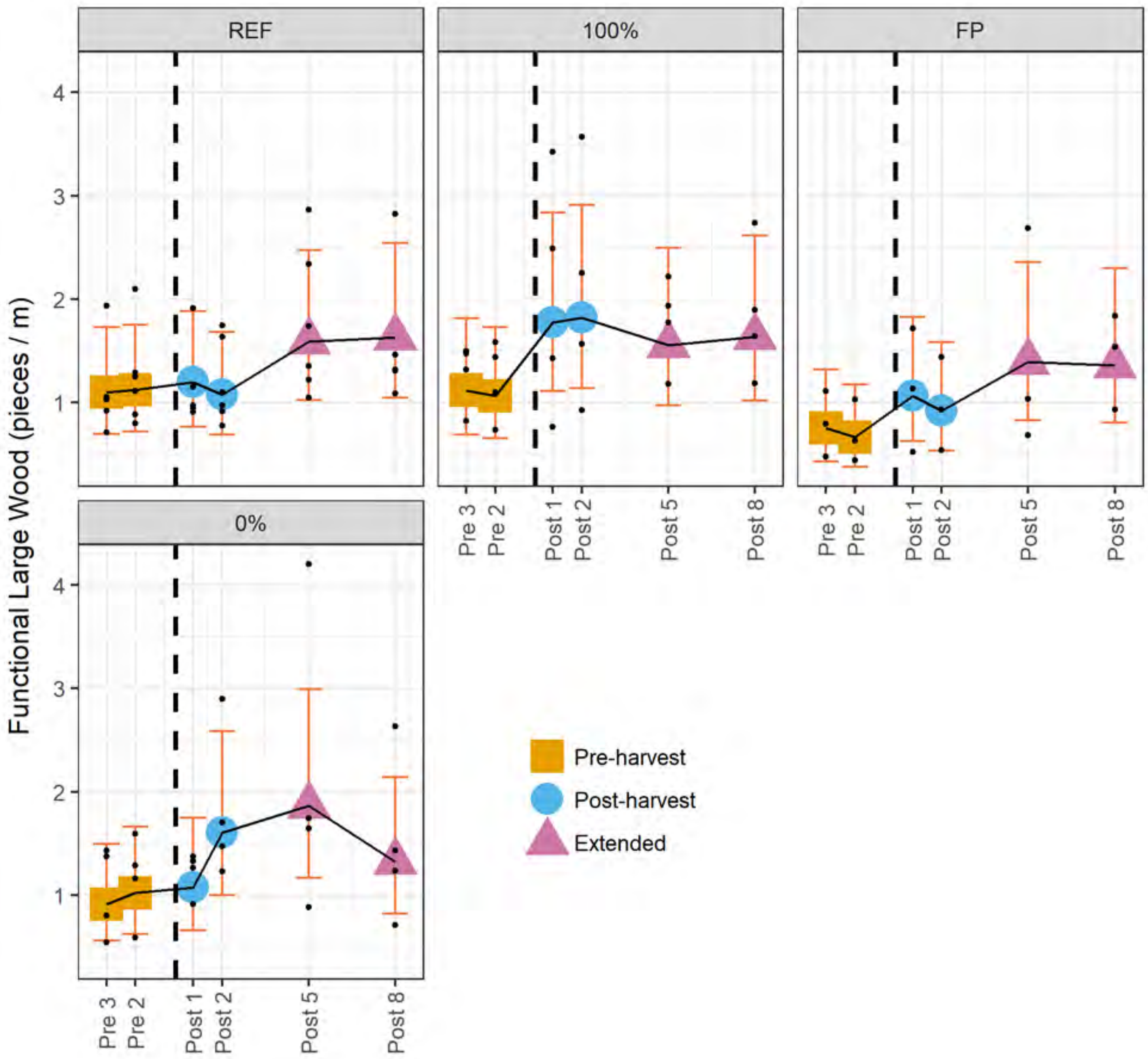

Figure 3-14. Mean functional large wood pieces/m of channel length by sample year (where pre-harvest includes Pre 3 & Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 3-14. The within-treatment estimate of the proportional change and 95% confidence intervals (CI) for functional large wood pieces between the pre-harvest and Post 1 and Post 2, Post 5, and Post 8 samples.

Treatment	Post 1 & 2	Post 5	Post 8
	Estimate (CI)	Estimate (CI)	Estimate (CI)
REF (n = 6)	1.02 (0.84, 1.25)	1.44 (1.17, 1.77)	1.47 (1.17, 1.85)
100% (n = 4)	1.65 (1.35, 2.01)	1.43 (1.12, 1.82)	1.50 (1.18, 1.90)
FP (n = 3)	1.40 (1.03, 1.90)	1.97 (1.41, 2.74)	1.92 (1.38, 2.69)
0% (n = 4)	1.36 (1.09, 1.70)	1.93 (1.52, 2.45)	1.37 (1.06, 1.78)

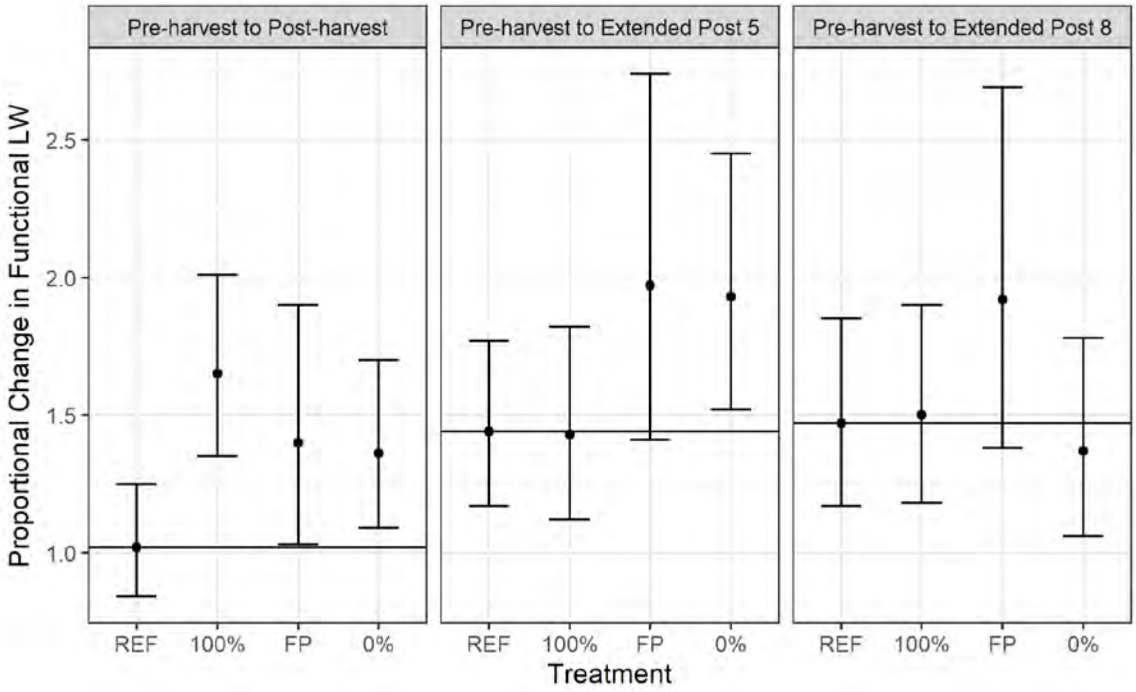

Figure 3-15. The within-treatment estimate of proportional change and 95% confidence intervals for functional large wood (LW) pieces between the pre-harvest and post-harvest (Post 1 and Post 2), Post 5 and Post 8 samples. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 3-15. The between-treatment comparison of the proportional change, 95% confidence intervals (CI) and P-values (P) for functional large wood pieces between the pre-harvest and Post 1 and Post 2, Post 5, and Post 8 samples. Estimates with P-values ≤0.10 are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	Post 1 & 2		Post 5		Post 8	
	Estimate (CI)	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
100% vs. REF	**1.61** (1.22, 2.13)	**<0.01**	0.99 (0.72, 1.37)	0.97	1.02 (0.73, 1.42)	0.91
FP vs. REF	**1.37** (0.95, 1.97)	**0.09**	1.37 (0.93, 2.03)	0.11	1.31 (0.87, 1.96)	0.19
0% vs. REF	**1.33** (0.99, 1.79)	**0.06**	**1.35** (0.98, 1.84)	**0.06**	0.93 (0.66, 1.32)	0.69
0% vs. FP	0.97 (0.67, 1.42)	0.89	0.98 (0.65, 1.48)	0.93	0.71 (0.47, 1.09)	0.12
0% vs. 100%	0.83 (0.61, 1.11)	0.20	**1.35** (0.96, 1.90)	**0.08**	0.92 (0.64, 1.31)	0.62
FP vs. 100%	0.85 (0.59, 1.22)	0.37	1.38 (0.92, 2.08)	0.12	1.28 (0.85, 1.94)	0.23

3-4.4.3. Channel Wood Cover

The proportion of stream channel surface area length covered with newly recruited wood in the post-harvest period did not differ significantly among treatments (P = 0.22, 0.20 and 0.35 for Post 2, Post 5 and Post 8, respectively; **Table 3-17; Figure 3-16**; **Figure 3-17**). Site means ranged from 0.18 to 0.73 in Post 2, from 0.16 to 0.54 in Post 5, and from 0.15 to 0.60 in Post 8 (**Figure 3-16**). However, we did note a pattern, with the greatest Post 2 wood cover in the 0% treatment (0.48), moderate values in the FP and 100% treatments (0.41 and 0.36, respectively), and the lowest value in the reference (0.31; **Table 3-16**). In Post 5, the trend was similar. By Post 8, the greatest mean wood cover was in the FP treatment (0.43).

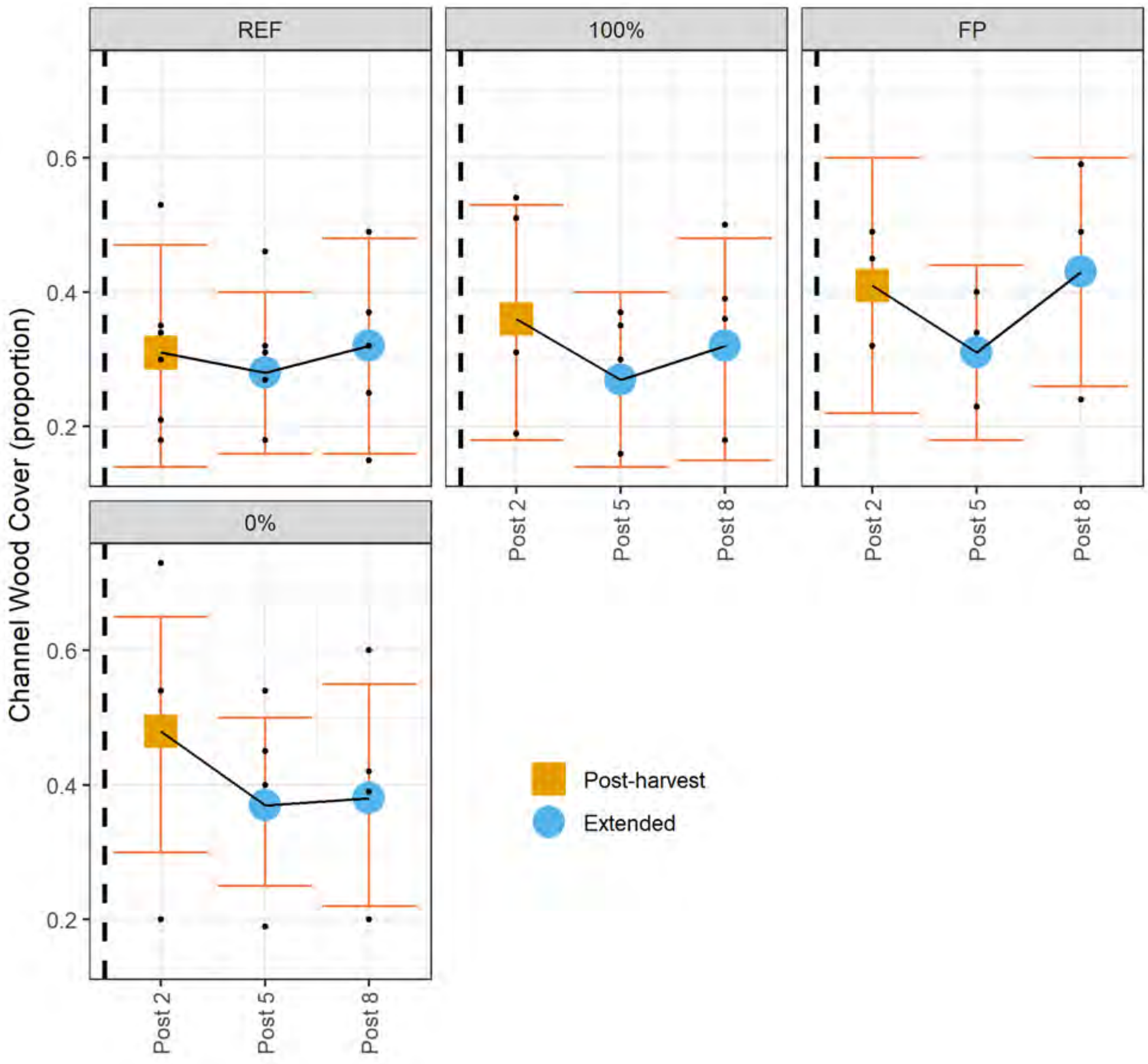

Figure 3-16. Mean proportion of the stream channel length covered by wood by sample year. Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 3-16. The estimate and 95% confidence intervals (CI) for the proportion of the stream channel length covered with wood at the Post 2, Post 5, and Post 8 samples.

Treatment	Post 2	Post 5	Post 8
	Estimate (CI)	Estimate (CI)	Estimate (CI)
REF (n = 6)	0.31 (0.14, 0.47)	0.28 (0.16, 0.40)	0.32 (0.16, 0.48)
100% (n = 4)	0.36 (0.18, 0.53)	0.27 (0.14, 0.40)	0.32 (0.15, 0.48)
FP (n = 3)	0.41 (0.22, 0.60)	0.31 (0.18, 0.44)	0.43 (0.26, 0.60)
0% (n = 4)	0.48 (0.30, 0.65)	0.37 (0.25, 0.50)	0.38 (0.22, 0.55)

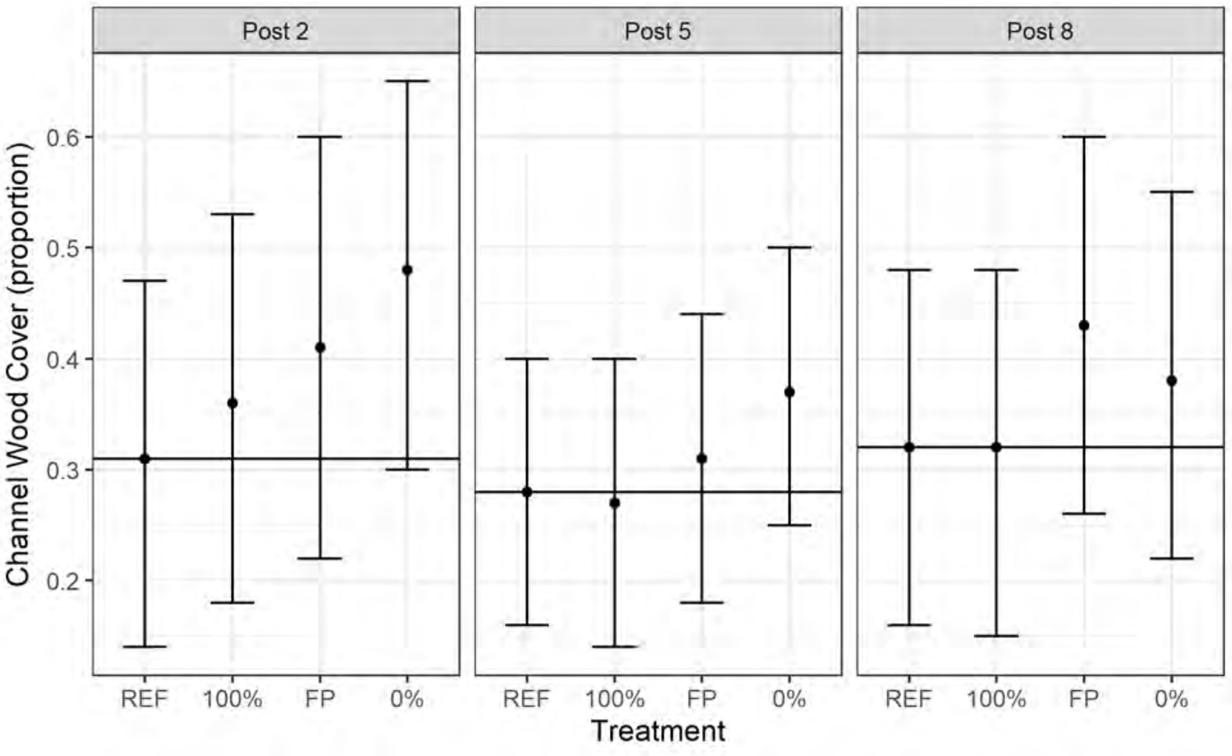

Figure 3-17. The estimate and 95% confidence intervals for the proportion of the stream channel length covered with newly recruited wood at the Post 2, Post 5, and Post 8 samples. A horizontal line placed at the reference treatment value indicates the mean reference condition.

Table 3-17. The between-treatment comparison and 95% confidence intervals (CI) for the proportion of the stream channel length covered with newly recruited wood at the Post 2, Post 5, and Post 8 samples. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	Post 2	Post 5	Post 8
	Estimate (CI)	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	0.05 (-0.18, 0.28)	-0.01 (-0.15, 0.13)	0.00 (-0.18, 0.18)
FP vs. REF	0.10 (-0.16, 0.37)	0.03 (-0.13, 0.19)	0.11 (-0.09, 0.32)
0% vs. REF	0.17 (-0.06, 0.40)	0.09 (-0.04, 0.23)	0.07 (-0.12, 0.25)
0% vs. FP	0.07 (-0.20, 0.34)	0.06 (-0.10, 0.23)	-0.05 (-0.25, 0.16)
0% vs. 100%	0.12 (-0.14, 0.38)	0.11 (-0.05, 0.26)	0.06 (-0.14, 0.26)
FP vs. 100%	0.05 (-0.24, 0.34)	0.04 (-0.13, 0.21)	0.11 (-0.11, 0.33)

3-5. DISCUSSION

3-5.1. STAND STRUCTURE AND WOOD INPUT

3-5.1.1. Magnitude and Duration of Post-harvest Response

Changes in stand structure and wood recruitment differed in response to three riparian management strategies for Type Np streams in western Washington, corresponding to the proportion of the stream network with buffers.

Clearcut harvest to the edge of the stream in 0% treatment and unbuffered reaches of the FP treatment produced the most dramatic changes in structure and wood recruitment. Removal of the existing stand put clearcut reaches on an entirely new trajectory by reinitiating succession, delivering logging debris to the channel, and reducing future wood recruitment potential.

There were also differences between the treatments with 13.2 m wide no-harvest buffers. Over the 8-year post-harvest period, declines in tree density and basal area were greater in the buffered (FPB) reaches of partially buffered FP treatment sites than in the entirely buffered 100% treatment or unharvested references, but declines in the latter two were not significantly different. In the PIP buffers, declines in live basal area were similar in the FPB and 100% PIPs and significantly greater than in the references. Post-harvest mortality did not change species' dominance in 100% or FPB RMZs, although one PIP shifted from primarily western hemlock to Douglas-fir. Effects of treatments on large wood input were equivocal. Although mean recruitment of fallen trees and input of large wood (pieces and volume) into the RMZs increased as the proportion of stream length buffered decreased, differences were not significant. Among PIPs, significant short-term differences between 100% and FPB buffers and the reference did not persist over longer intervals.

The results of this study are largely consistent with those of previous studies of mortality and changes in stand structure in buffers on Type Np streams in western Washington (Grizzel and

Wolff 1998; Jackson *et al.* 2001; Schuett-Hames and Stewart 2019). Post-harvest mortality and change in stand structure in the FPB RMZs of the Type N Buffer Characteristics, Integrity and Function (BCIF) study (Schuett-Hames and Stewart 2019) were greater than in the reference RMZs and intermediate between the FPB and 100% RMZ values in this study, while large wood recruitment was similar. Among PIPs, cumulative mortality in the FPB PIPs in the BCIF study was similar to FPB PIPs in this study, but changes in stand structure and cumulative large wood recruitment volume in the BCIF study were greater than in either the 100% treatment or FPB PIPs in this study (**Table 3-18**).

Table 3-18. Comparison of changes in mean live basal area (m^2/ha) and density (trees/ha), cumulative tree mortality (% of pre-harvest live basal area) and cumulative large wood recruitment to the channel (volume/ha) in RMZ and PIP buffers for treatments in this study and the BCIF study through five years post-harvest. The Buffer Characteristics, Integrity and Function Study (BCIF) sample size is 13 RMZs and 3 PIPs.

Study	Treatment	RMZ	PIP
Change in Live Basal Area (m^2/ha)			
This Study	100%	-6.0	-14.9
This Study	FPB	-16.1	-17.6
BCIF	FPB	-10.1	-30.0
Change in Live Density (trees/ha)			
This Study	100%	-142	-215
This Study	FPB	-172	-201
BCIF	FPB	-78	-280
Cumulative Mortality (% live basal area)			
This Study	100%	19.9	34.0
This Study	FPB	37.5	47.6
BCIF	FPB	27.2	48.0
Cumulative Large Wood Recruitment (m^3/ha)			
This Study	100%	10.6	6.6
This Study	FPB	11.4	4.0
BCIF	FPB	10.5	23.9

Most mortality driving changes in stand structure in this study was due to windstorms (see Chapter 2 – *Study Design*, **Section 2-6** *Unanticipated Disturbance Events* in this report). Wind damage is common in streamside buffers and cut-block edges in this region after harvest of the adjacent stand (Mitchell 1995; Grizzel and Wolff 1998; Harris 1999; Mitchell and Rowan 2002; Busby *et al.* 2006; Liquori 2006; Martin and Grotefendt 2007; Rollerson *et al.* 2009). Trees embedded within large blocks of contiguous forest are protected from exposure to wind and do not develop characteristics that increase wind-resistance—as do open-grown stems that have tapered crowns, low centers of gravity, and strong stems and buttressed roots (Harris 1999). Consequently, trees subject to sudden exposure within retained buffers have little wind-resistance, particularly if they are not deeply rooted.

Beese and colleagues (2019) determined that wind-related mortality was lower in larger leave patches in coastal forests of Vancouver Island where windstorms cause extensive post-harvest mortality. We did not observe lower mortality in the PIPs of the 100% treatment (relative to those in FPB), despite continuity with adjacent RMZ buffers. High mortality in PIP buffers is consistent with factors that increase susceptibility to wind, including small patch size, large fetch distances, and upper-slope positions where wind can accelerate (Ruel *et al.* 2001; Rollerson *et al.* 2009). These risk factors may override any benefits of connectivity to adjacent RMZ buffers.

We observed decreases in rates of tree mortality, changes in stand structure, and in-channel large wood input in both the 100% and FPB reaches over the eight year post-harvest interval. In both RMZs and PIPs, mortality rates were highest during the first two years, then decreased. Nevertheless, by year eight, mortality rates were still two and four times the reference rate for 100% and FPB RMZs, respectively, and over eight times the reference rate for the corresponding PIPs. As a consequence, stand structures in the buffered RMZs and PIPs have continued to diverge over time from reference sites.

Patterns of elevated, but gradually declining rates of wind-related mortality in the buffered reaches of these treatments are consistent with those of previous studies of FPB RMZ and PIP buffers on Type Np streams in western Washington (Schuett-Hames *et al.* 2012). They are also consistent with trends in mortality along the edges of cut-blocks and in variable-retention units (Mitchell *et al.* 2001; Mitchell and Rowan 2002; Busby *et al.* 2006; Urgenson *et al.* 2013). Elevated mortality following partial harvest reflects rapid loss of trees that are least wind-resistant (Busby *et al.* 2006). Mortality then declines as more wind-resistant trees adjust to greater exposure through wind pruning of branches or changes in stem or root system architecture (Harris 1999; Busby *et al.* 2006). However, intense windstorms may elevate mortality rates again (Ruel *et al.* 2001).

3-5.1.2. Variability in Post-harvest Response

One notable aspect of the post-harvest response was the much greater variability among 100% and FPB sites than among reference sites. This reduced our ability to detect statistical differences, despite large differences in treatment means. This variability is also problematic for resource managers because it makes it difficult to reliably predict responses to management.

Initial stand structures may have varied based on prior harvest history, and there may be some correlation between landowners (private, state, federal) and initial stand structure. Initial stand structures that varied by owner may have had added variability in our analysis of stand structure and wood loading. However, most site to site variation appears to be associated with variability in wind-related mortality. In a study of tree mortality on federal forestlands in the Pacific Northwest, Reilly and Spies (2016) characterized the severity of mortality among forest stands as (1) chronic (<5% of live trees/year, typically due to endogenous causes); partial stand replacement (5−25%/year, typically due to exogenous causes including wind damage); or (3) stand replacement (>25%/year, typically driven by fire). In this study, mortality rates at most 100% treatment sites were at chronic levels (71% vs. 36% of FPB reaches), whereas mortality in the FPB reaches most often manifested as partial stand replacement (48% vs. 24% of 100% treatment sites). The frequency of stand replacement events was also higher in FPB (16% vs. 5% of 100% treatment sites).

Variation in wind damage likely reflects differences in exposure and other regional or site conditions. Susceptibility of buffers to wind damage is determined by climate, topography, stand structure and management history (Mitchell *et al.* 2001). Variation in the frequency and magnitude of windstorms create regional differences in disturbance regime (Kramer *et al.* 2001). At a local scale, susceptibility to wind damage is affected by factors that increase exposure to wind such as fetch distance (Scott and Mitchell 2005; Rollerson *et al.* 2009); topography that concentrates or accelerates wind (Moore 1977; Harris 1999; Ruel *et al.* 2001; Rollerson *et al.* 2009; Mitchell 2013); management factors such as edge or buffer orientation or patch size (Mitchell *et al.* 2001; Rollerson *et al.* 2009); edaphic factors that affect rooting strength such as soil depth and moisture (Moore 1977; Harris 1999; Mitchell 2013); and stand or species characteristics that affect wind-resistance (e.g., species composition, density, age, and height-diameter ratio (Harris 1999; Scott and Mitchell 2005; Rollerson *et al.* 2009; Mitchell 2013). The highest mortality rates in the 100% and FPB RMZ buffers occurred in WIL1, WIL2 and OLYM—coastal blocks where storm- and hurricane-force windstorms were frequent during the post-harvest interval (see Chapter 2 – *Study Design*, **Section 2-6** *Unanticipated Disturbance Events* in this report). Although several WIL1 and WIL2 sites experienced elevated pre-harvest mortality due to the December 2007 storm event, the two sites with the highest post-harvest mortality had relatively low (<5%) mortality in the pre-harvest period.

3-5.2. CHANNEL WOOD LOADING AND COVER

We observed large increases in in-channel wood following harvest, with the greatest increase in small wood loading—176% relative to the reference—occurring in the 0% treatment in the two years after harvest. Over the same period, we also saw marked increases in the FP and 100% treatments (69 and 58%, respectively). Similarly, large wood loading increased in all buffer treatments (range of 44 to 66%) in the two years following harvest. These increases in small and large wood derive from logging slash (Gomi *et al.* 2001; Jackson *et al.* 2001), windthrow from riparian buffers (Chen *et al.* 1995; Reid and Hilton 1998; May and Gresswell 2003b; Schuett-Hames *et al.* 2012), and possibly fragmentation and movement of wood from adjacent hillslopes (Harmon *et al.* 1986). As observed by Jackson and colleagues (2001), in-channel wood loading was greatest adjacent to clearcut reaches of the RMZ. In contrast, retention of a riparian buffer in the 100% and FP treatments greatly reduced input of logging debris into the stream channel (Jackson *et al.* 2001; Maxa 2009; Schuett-Hames *et al.* 2012). The lack of a difference in large wood counts among riparian buffer treatments suggests that the number of pieces from harvest-related inputs and inputs from windthrow were similar. However, our data indicate that about 75% of large wood pieces recruited from 100% and FPB buffers consisted of stems with attached rootwads, while pieces recruited from logging debris consisted of broken stems and tops. Given that large wood recruitment to the RMZ in the reference declined five-years post-harvest, the increase in total and functional large wood pieces may seem surprising. We suspect that the increase may be the result of fragmentation of trees that fell in and over the channel during the December 2007 (pre-harvest) windstorm (see Chapter 2 – *Study Design*, **Section 2-6** *Unanticipated Disturbance Events* in this report).

Five years following harvest, the increase in small and large wood pieces in the FP and 0% treatments was greater than the change in the reference and 100% treatment. At this point changes in small and large wood pieces did not differ between 100% treatment and reference, although the lack of a difference stemmed as much from an increase in the reference as from a decline in the 100% treatment. By eight years post-harvest, we were unable to detect a difference in small wood among treatments. At this point, large wood counts remained elevated in the FP treatment relative to the change in the reference, while large wood in the 0% treatment declined to levels that no longer differed from that of the reference. Previous studies have shown various effects of harvest on in-channel large wood loading. Ralph and colleagues (1994) found no difference in piece counts between unharvested and previously harvested sites in western Washington. On the other hand, in southern Alaska, Gomi and colleagues (2001) observed greater loading in streams adjacent to 37-year-old stands than in old growth—a result of wood input from past logging of steep headwater streams. Finally, Bilby and Ward (1991) found that large wood loading was lower in previously logged than in older forests in southwestern Washington. Given the rapid changes in wood loading observed in this study, it is likely that at least some of this variation relates to time since harvest and to differences in management practices at the time of harvest.

Although small wood is not typically considered in studies of wood loading in headwater streams (Hassan *et al.* 2005), we recorded large inputs of small wood following harvest. However, much of the small wood in our study was depleted by eight years following harvest. Since decay rate is partly a function of surface area (Aumen *et al.* 1983), small wood should decay more quickly than large wood. Depletion was most likely through decay and downstream transport (Bilby *et al.* 1999; Wallace *et al.* 2000; Hyatt and Naiman 2001).

In contrast to small wood, it is more difficult to explain why large wood loading declined between years five and eight years post-harvest in the 0% treatment, but was stable in the FP treatment. Other studies suggest that small streams tend to retain wood that is otherwise transported in larger streams (Bilby and Ward 1989; Bilby and Bisson 1998; Gomi *et al.* 2006; Maxa 2009). Because wood transported by streams is typically shorter than bankfull width (Lienkaemper and Swanson 1987; Millard 2001), our lack of a minimum length criterion makes it impossible to distinguish between long less mobile pieces and short easily transported pieces that could elucidate the decline in the 0% treatment over time. One likely explanation is that the FP treatment was comprised of buffered and unbuffered stream reaches, while the 0% treatment was entirely unbuffered. Large wood in buffered reaches of the FP treatment tended to include a greater proportion of large stems, some with rootwads, from windthrow in the adjacent buffer. Conversely, large wood in unbuffered reaches tended to be broken stems and tops that may have been more prone to transport than many large wood pieces in buffered reaches. Our findings are consistent with those of Bilby and Ward (1991) who observed a loss of in-channel large wood in western Washington following harvest. However, we cannot make direct comparisons with their work because they considered only longer pieces (>2 m). It is unclear why large wood loading remained greater in the FP treatment than in the reference or other buffer treatments: the difference may be related to the greater rates of windthrow from the FP buffer.

Functions provided by in-channel wood depend on characteristics of the wood and stream channel (Gomi *et al.* 2001; Maxa 2009). All sizes of wood play important roles in headwater streams (Bilby and Bisson 1998; Gomi *et al.* 2001; Jackson and Sturm 2002; Maxa 2009), contributing to step formation, bank stability, and in-channel roughness (Harmon *et al.* 1986; Bilby and Ward 1989; Gomi *et al.* 2002; Hassan *et al.* 2005). Temporal and treatment-related patterns of change in functional small and large wood loading were very similar to those of total small and large wood. Piece counts increased in all buffer treatments after harvest and continued to increase in FP and 0% treatments until year five; however, functional large wood in the FP treatment did not differ significantly from the reference or 100% treatment. We suspect that the increase in FP and 0% treatments was due to fragmentation of larger pieces and input from wood suspended above the channel (Nakamura and Swanson 1993). By post-harvest year eight, functional wood loading did not differ between buffer treatments and the reference. In addition to quantity, the function of individual pieces plays an important role in structuring the physical habitat in headwater streams.

Initially, in-channel wood may shade the channel (Jackson *et al.* 2001; Kibler *et al.* 2013), provide refuge for fish and amphibians (Grialou *et al.* 2000; Rundio and Olson 2007), and influence invertebrate assemblages (Anderson *et al.* 1978). This may prove especially important shortly after harvest, when overstory and understory cover are lowest (Gravelle and Link 2007; Janisch *et al.* 2012; Rex *et al.* 2012; Kibler *et al.* 2013). Although not statistically significant, cover from in-channel wood tended to increase in the two years following harvest in proportion to the level of buffer removal (0% > FP > 100%). Consistent with temporal trends in wood loading, wood cover in the 100% and reference treatments were nearly identical five years post-harvest and cover eight years post-harvest was greatest in the FP treatment. As with wood loading, we speculate that the elevated cover in the FP treatment was associated with continued windthrow from the riparian buffers.

Our long-term observations suggest that levels of small and large wood pieces in buffer treatments were equal to or greater than those in the reference. However, for the 0% treatment, a combination of declining tree density and a lack of source wood in the developing forest suggest further declines in the future, as existing wood decays or is transported downstream. Although differences in size-class definitions make direct comparisons difficult, results from Maxa (2009) are consistent with this prediction: recently harvested stands had more large wood (0.54 pieces/m >10 cm diameter) than did older, 50- to 70-year-old, second growth stands (0.36 pieces/m).

Wood in small headwater streams is highly retentive, physically obstructing sediment transport and forming steps (Hogan *et al.* 1998; Gomi and Sidle 2003; Lancaster *et al.* 2003; May and Gresswell 2003b). If wood recruitment and loading continue to decline in the 0% treatment, sediment storage capacity in these headwater streams may decline (e.g., Benda *et al.* 2005), with consequences for a diversity of aquatic organisms.

3-5.3. IMPLICATIONS FOR STAND DEVELOPMENT, WOOD INPUT AND WOOD LOADING

Classic models of stand development describe a predictable chronology of structural stages following stand-replacing disturbance (Oliver 1980; Franklin *et al.* 2002). Stands proceed through initiation, canopy closure, competitive exclusion, and maturation stages, culminating in a more complex structure that develops through vertical and horizontal diversification (Oliver 1980; Franklin *et al.* 2002; Reilly and Spies 2016). This model may be appropriate to forests experiencing single, stand-replacing disturbance events (e.g., clearcut harvest or catastrophic fire), but it does not account for situations in which regeneration is poor or patchy (Lutz and Halpern 2006; Donato *et al.* 2012) or where stands are subject to periodic or episodic, low- to moderate-intensity disturbance (e.g., from fire, wind, snow loading, insects, or disease; Weisberg 1998; Sinton *et al.* 2000; Tepley *et al.* 2013; Reilly and Spies 2016; Meigs *et al.* 2017). In these situations, stands can develop along alternate pathways that vary with initial conditions and interact with climate, soils, and the type, frequency and intensity of disturbance (Weisberg 1998; Donato *et al.* 2012; Tepley *et al.* 2013; Warren *et al.* 2016).

Lack of seed sources or unfavorable climatic or environmental conditions can result in patchy or low-density conifer regeneration following stand-replacement events (Beach and Halpern 2001; Lutz and Halpern 2006). If tree density is too sparse to achieve canopy closure, shrub and broadleaf species can persist, competing with shade-intolerant conifers for space and light. The characteristics of stands that develop under these open-canopy conditions may include clumped or widely spaced trees, multiple canopy layers, and greater diversity of tree species (Donato *et al.* 2012).

Low to moderate severity disturbances can profoundly affect stand development (Tepley *et al.* 2013; Meigs *et al.* 2017). Mortality of overstory trees can leave gaps that allow light to penetrate to the forest floor, initiating new cohorts of trees (Warren *et al.* 2016)—typically shade-tolerant species such as western hemlock, provided that seed sources are present (Sinton *et al.* 2000). Repeated disturbance can initiate multiple cohorts. Moderate-level disturbances from wind and disease do not typically eliminate advance regeneration; rather they promote growth release and eventual gap filling (Warren *et al.* 2016; Meigs *et al.* 2017). Fallen trees created by these disturbances serve as substrates for regeneration of shade-tolerant species such as western hemlock (Christy and Mack 1984; Harmon and Franklin 1989) and, in riparian areas, contribute large wood to adjacent streams.

In this study, variation in initial structure, buffer treatment and subsequent disturbance contributed to variation in stand structure and composition, with implications for future stand development and in-stream wood. In unbuffered reaches, nearly all trees were removed during harvest. Adjacent channels received a pulse of logging debris consisting of mostly small diameter tops, branches, and broken stems. These sites have been replanted, primarily with Douglas-fir, and are expected to be harvested at periodic intervals in the future. Repeat harvest drastically reduces present and future wood input to the stream, thus models predict rapid and prolonged decreases in large wood loading over time as legacy wood is lost to attrition (Beechie *et al.* 2000; Bragg 2000; Meleason *et al.* 2003). Low wood loading will continue unless streamside forests are restored or there is additional wood input from upslope processes such as mass wasting.

Differences in initial stand structure and post-harvest mortality among buffered reaches have placed stands on different trajectories with implications for wood input and loading. Where post-harvest disturbance and mortality remained low (mortality rates <5%/year), stands have attained canopy closure and are undergoing competitive exclusion. Deep shade inhibits regeneration (Warren *et al.* 2016), so these stands should continue to develop as single cohorts of a mix of Douglas-fir and western hemlock (Oliver 1980). However, higher levels of light at the edges of buffers may promote regeneration of shade-tolerant conifers. In the absence of major disturbance, these stands should provide a relatively continuous source of wood to the adjacent channel as individual trees or small groups of trees die from competition or small-scale disturbance. In this scenario, models predict an increase in wood load over time as input exceeds depletion (Bragg 2000).

In contrast, stand trajectories differ where pre-harvest density was low (red alder dominated reaches in the Cascade block), pre-harvest mortality was high (PIPs in one WIL1 site), or post-harvest mortality exceeded 5% per year (partial stand replacement levels of Reilly and Spies 2016). In these situations, relative density is below 35, under-utilizing the growing potential of the site (Drew and Flewelling 1979). This occurred in 20 and 92% of RMZ plots and half of the PIP plots in the 100% treatment and FPB reaches, respectively. Natural regeneration, primarily western hemlock with some Douglas-fir and red alder, is occurring in the Willapa 1, Willapa 2, Willapa 3 and Olympic block RMZs. These stands appear to be recruiting a second cohort of shade-tolerant trees, creating a two age-class structure. Regeneration is lower in the FPB RMZ in the Cascade block, with a higher proportion of red alder characteristic of the open-canopy developmental pathway.

Post-harvest wind damage in these stands created an initial pulse of in-channel wood. Mortality has continued at many sites although rates are decreasing. Future storms are likely to cause additional mortality in sites with topographic or other features that increase susceptibility to wind damage (Harcombe *et al.* 2004). Although early mortality has reduced recruitment potential in the near term, these riparian buffers retain live trees and snags that will contribute some wood until the new cohort matures (Bragg 2000). Consequently, with an episodic, moderate-intensity disturbance regime, wood input and loading would be dynamic, alternating between periods of input and subsequent decline as stands recover. In the long term, episodic mortality can provide as much or more wood than chronic mortality (if trees are not harvested), with oscillating periods of high to low loading (Bragg 2000).

The FPHCP partial-buffering strategy is altering the structure of riparian forests adjacent to headwater streams across large areas of western Washington. Harvest in unbuffered reaches and variation in structure and mortality in adjacent RMZ and PIP buffers is creating a mosaic of young to older stand structures. Thus, this strategy is increasing stand structural diversity rather than homogenizing it. As trees in unharvested buffers grow and mature, stand structure should become more complex, both horizontally and vertically (Franklin *et al.* 2002), enhancing riparian habitats for wildlife associated with older forests. Chronic or episodic input of wood should increase temporal and spatial variability in channel wood loading, with a trend toward greater loading over time. Clearly, the eight-year, post-harvest timeframe of this study is too short to capture the dynamics of stand development, wood input and channel wood loading that occur over decades or centuries. Further research and monitoring are needed to understand how the changing mosaic of stand structures adjacent to headwater streams will develop and function over the long term, and to document the long-term effects of the partial-buffering strategy on wood input and loading.

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3-7. APPENDIX 3A. DATA TABLES

Appendix Table 3-1. Pre-harvest (Pre 2) stand characteristics by site.

Plot Type	Block	Treatment[1]	Density (trees/ha)	Basal Area (m²/ha)	Basal Area % Conifer	Dominant Species Code[2]	Dominant Species %[3]
RMZ	OLYM	REF	555	42.1	99	TSHE	48
RMZ	OLYM	100%	491	40.4	84	TSHE	68
RMZ	OLYM	FPB	461	40.0	80	TSHE	56
RMZ	OLYM	FPU	502	46.4	98	TSHE	63
RMZ	OLYM	0%	560	42.7	78	PSME	50
RMZ	WIL1	REF	554	55.0	96	TSHE	95
RMZ	WIL1	100%	800	73.0	99	TSHE	81
RMZ	WIL1	FPB	487	54.7	100	TSHE	96
RMZ	WIL1	FPU	344	44.0	100	TSHE	100
RMZ	WIL1	0%	485	34.6	92	TSHE	80
RMZ	WIL2	REF	536	56.3	99	TSHE	82
RMZ	WIL2	REF	543	43.7	90	TSHE	45
RMZ	WIL2	100%	678	48.0	83	TSHE	70
RMZ	WIL2	0%	953	59.8	96	TSHE	81
RMZ	WIL3	REF	325	49.5	89	TSHE	45
RMZ	WIL3	100%	568	59.7	89	PSME	51
RMZ	CASC	REF	378	52.0	89	PSME	88
RMZ	CASC	FPB	213	25.8	3	ALRU	96
RMZ	CASC	FPU	251	26.8	9	ALRU	89
RMZ	CASC	0%	225	31.0	67	PSME	70
PIP	OLYM	REF	514	43.6	100	TSHE	73
PIP	OLYM	100%	382	47.9	98	TSHE	77
PIP	OLYM	FP	390	47.9	100	TSHE	79
PIP	WIL1	REF	541	54.3	90	TSHE	84
PIP	WIL1	100%	759	69.0	100	TSHE	68
PIP	WIL1	FP	579	65.5	100	TSHE	99
PIP	WIL1	0%	319	33.3	100	TSHE	92
PIP	WIL2	REF	612	69.6	100	TSHE	81
PIP	WIL2	REF	601	60.7	100	TSHE	87
PIP	WIL2	100%	617	55.0	100	TSHE	92
PIP	WIL2	0%	619	43.3	93	TSHE	75

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[1] FPB = buffered FP treatment RMZs reaches; FPU = unbuffered FP treatment RMZ reaches.

[2] ALRU=red alder (*Alnus rubra*); PSME=Douglas-fir (*Pseudotsuga menziesii*); TSHE=western hemlock (*Tsuga heterophylla*).

[3] Percent of live basal area.

Plot Type	Block	Treatment[1]	Density (trees/ha)	Basal Area		Dominant Species	
				(m^2/ha)	% Conifer	Code[2]	%[3]
PIP	CASC	REF	364	51.0	92	PSME	91
PIP	CASC	FP	277	34.3	9	ALRU	88
PIP	CASC	0%	120	12.2	84	PSME	84

Appendix Table 3-2. Initial (Pre 2) live structural characteristics of RMZ and PIP forest reaches by block. Values are the averages of site means with standard deviations in parentheses. QMD is the quadratic mean diameter. PIPs were not sampled in the Willapa 3 (WIL3) block.

Block	Density (trees/ha)	Basal Area (m^2/ha)	% Conifer Basal Area	QMD (cm)	Relative Density
			RMZ		
OLYM	521 (42)	42.0 (1.1)	87.1	32.7 (1.3)	51 (1.4)
WIL1	571 (159)	53.5 (15.7)	96.7	35.3 (3.0)	63 (17.9)
WIL2	677 (195)	51.9 (7.4)	91.8	32.2 (3.6)	64 (10.3)
WIL3	447 (172)	54.6 (7.2)	88.6	41.4 (5.4)	60 (12.1)
CASC	277 (87)	36.4 (13.7)	53.3	42.1 (3.1)	39 (14.4)
			PIP		
OLYM	434 (80)	46.3 (9.2)	99.5	37.1 (5.5)	53 (7.9)
WIL1	477 (252)	47.9 (21.6)	97.4	36.2 (2.4)	62 (17.7)
WIL2	615 (81)	54.9 (14.2)	97.0	33.6 (4.5)	65 (13.7)
CASC	292 (95)	38.3 (15.2)	54.9	40.4 (5.0)	41 (15.2)

Appendix Table 3-3. Change in live stand structure, tree mortality and large wood (LW) recruitment to the channel during the pre-harvest interval (Pre 2 to Pre 1). Values are the averages of site means with standard deviations in parentheses. PIPs were not sampled in the WIL3 block.

Block	Change in Stand Structure		Mortality		LW Recruitment	
	Density (trees/ha)	Basal Area (m^2/ha)	% of Stems	% of Basal Area	Fallen (trees/ha)	LW (pieces/ha)
			RMZ			
OLYM	-16.3 (7.7)	-0.8 (0.4)	3.0 (1.8)	1.9 (1.1)	14.9 (9.6)	17.8 (13.0)
WIL1	-112.9 (88.6)	-10.8 (8.2)	21.2 (13.0)	21.2 (12.6)	57.5 (53.8)	60.4 (55.4)
WIL2	-69.3 (8.6)	-4.7 (0.9)	10.7 (3.1)	9.4 (2.5)	45.5 (11.7)	49.6 (13.2)
WIL3	-6.8 (1.4)	-0.5 (0.1)	1.1 (0.2)	0.9 (0.2)	6.9 (2.4)	8.2 (1.8)
CASC	-0.5 (0.9)	-0.1 (0.2)	0.3 (0.4)	0.2 (0.4)	0.8 (1.3)	1.3 (2.2)
			PIP			
OLYM	-5.5 (10.1)	-0.5 (0.9)	1.4 (2.6)	0.8 (1.4)	5.5 (10.1)	10.9 (23.4)
WIL1	-98.3 (105.5)	-9.6 (11.3)	21.8 (21.1)	20.8 (21.2)	31.7 (41.3)	31.7 (41.3)
WIL2	-88.8 (57.6)	-7.4 (6.8)	13.6 (8.3)	12.4 (9.3)	42.3 (25.7)	43.7 (29.2)
CASC	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	1.6 (4.1)	3.1 (8.3)

Appendix Table 3-4. Mean live stand live structural characteristics before harvest (Pre 1) and at each post-harvest sample (Post 2, Post 5, and Post 8) in the reference, 100% treatment, and FPB reaches. For both RMZs and PIPs, values are averages of site means with standard deviations in parentheses.

Year	**RMZ**			**PIP**		
	REF	**100%**	**FPB**	**REF**	**100%**	**FPB**
Basal Area (m^2/ha)						
Pre 1	44.2 (7.3)	50.5 (10.4)	39.1 (13.6)	47.6 (10.3)	51.7 (12.0)	40.2 (7.8)
Post 2	42.6 (9.3)	46.0 (10.7)	27.4 (2.2)	47.8 (10.2)	43.5 (13.8)	25.2 (8.4)
Post 5	44.2 (10.1)	44.5 (12.2)	23.0 (7.3)	48.7 (10.7)	36.8 (17.1)	22.5 (9.0)
Post 8	44.0 (9.9)	43.4 (13.5)	17.7 (5.4)	49.6 (10.8)	31.9 (19.5)	18.8 (7.2)
Density (trees/ha)						
Pre 1	418 (89.4)	575 (78.1)	371 (138.8)	447 (96.2)	517 (212.6)	323 (70.5)
Post 2	377 (81.8)	484 (28.1)	238 (50.1)	416 (99.0)	408 (171.9)	156 (35.0)
Post 5	363 (83.8)	433 (60.8)	199 (76.7)	391 (105.2)	302 (136.1)	122 (41.1)
Post 8	349 (87.1)	404 (80.0)	152 (42.6)	383 (93.4)	251 (136.5)	98 (37.8)
Quadratic Mean Diameter (cm)						
Pre 1	37.8 (5.3)	34.2 (2.9)	37.6 (2.8)	37.1 (5.2)	36.6 (5.1)	40.0 (4.0)
Post 2	38.9 (5.1)	35.6 (2.9)	38.0 (2.7)	38.6 (5.1)	37.4 (5.7)	44.8 (4.1)
Post 5	40.3 (5.1)	37.5 (2.4)	38.9 (1.5)	40.3 (5.2)	39.3 (4.8)	48.2 (6.1)
Post 8	41.2 (4.9)	38.1 (2.6)	37.5 (2.8)	41.0 (5.2)	39.5 (4.1)	49.4 (7.0)
Relative Density						
Pre 1	49.8 (6.5)	60.0 (10.8)	44.6 (15.6)	54.1 (10.1)	59.7 (15.5)	44.1 (8.3)
Post 2	47.7 (7.9)	53.6 (10.3)	30.5 (3.4)	53.3 (10.2)	49.4 (15.4)	25.9 (7.9)
Post 5	49.1 (7.8)	50.9 (12.2)	27.1 (6.5)	53.3 (11.0)	40.5 (18.0)	22.4 (8.5)
Post 8	48.4 (7.9)	49.8 (13.1)	20.7 (4.0)	53.8 (10.7)	34.8 (20.3)	18.4 (6.9)
Percentage of Plots with Conifer Regeneration						
Post 5	18.8 (16.7)	31.0 (11.6)	10.3 (13.8)	14.8 (26.5)	49.8 (31.5)	35.3 (26.1)
Post 8	26.0 (22.2)	37.8 (16.9)	20.7 (21.5)	22.4 (31.8)	52.5 (28.5)	41.3 (31.1)

Appendix Table 3-5. Post-harvest change in live basal area and density, mortality in percentage of basal area and percentage of stems, ingrowth density, and recruiting fallen trees, large wood pieces and volume for the reference, 100% treatment, and FPB reaches. For both RMZs and PIPs, values are the averages of site means with standard deviations in parentheses.

Interval	RMZ			PIP		
	REF	**100%**	**FPB**	**REF**	**100%**	**FPB**
Change in Live Basal Area (m^2/ha)						
Pre 1–Post 2	-1.6 (2.3)	-4.5 (5.7)	-11.8 (12.7)	0.2 (2.3)	-8.1 (9.0)	-15.0 (7.3)
Pre 1–Post 5	0.0 (3.3)	-6.0 (11.1)	-16.1 (18.5)	1.1 (3.2)	-14.9 (14.3)	-17.6 (8.1)
Pre 1–Post 8	-0.3 (3.5)	-7.1 (14.1)	-21.4 (19.0)	2.1 (3.5)	-19.8 (18.4)	-21.4 (8.6)
Change in Live Density (trees/ha)						
Pre 1–Post 2	-41 (24.4)	-91 (69.4)	-133 (111.3)	-31 (38.4)	-109 (107.5)	-167 (85.7)
Pre 1–Post 5	-55 (31.2)	-142 (125.0)	-172(153.1)	-56 (54.9)	-215 (162.0)	-201 (89.8)
Pre 1–Post 8	-69 (32.1)	-171 (146.5)	-219 (165.1)	-64 (56.5)	-266 (201.7)	-225 (96.1)
Mortality (% of pre-harvest basal area)						
Pre 1–Post 2	9.7 (9.9)	12.7 (9.9)	26.3 (23.5)	4.1 (4.6)	17.2 (17.7)	38.3 (16.4)
Pre 1–Post 5	13.0 (1 2.7)	19.9 (17.3)	37.5 (31.8)	7.9 (8.3)	34.0 (26.2)	47.6 (16.6)
Pre 1–Post 8	16.1 (13.6)	24.3 (21.2)	50.8 (34.0)	9.4 (8.9)	43.0 (32.9)	56.4 (16.3)
Mortality (% of pre-harvest stems)						
Pre 1–Post 2	11.7 (9.8)	16.1 (11.6)	29.7 (22.1)	6.5 (8.6)	19.0 (17.4)	48.7 (18.6)
Pre 1–Post 5	15.9 (12.9)	24.5 (19.9)	39.0 (29.3)	12.5 (11.9)	38.5 (25.1)	60.0 (19.4)
Pre 1–Post 8	20.1 (14.0)	29.5 (22.9)	49.7 (30.6)	15.2 (12.0)	49.0 (31.0)	68.0 (19.0)
Ingrowth (trees/ha)						
Pre 1–Post 2	2.2 (3.5)	5.9 (5.0)	1.2 (2.1)	0.0 (0.0)	1.8 (4.5)	0.0 (0.0)
Pre 1–Post 5	4.5 (5.2)	7.4 (6.2)	3.0 (5.2)	1.0 (3.3)	1.8 (4.5)	1.6 (4.1)
Pre 1–Post 8	7.2 (6.7)	10.7 (9.9)	5.4 (9.3)	6.0 (13.3)	7.3 (8.9)	4.7 (5.8)
Fallen Tree Recruitment (trees/ha)						
Pre 1–Post 2	16.5 (17.5)	35.8 (33.1)	48.7 (45.6)	4.0 (7.4)	40.1 (32.9)	29.7 (30.1)
Pre 1–Post 5	21.8 (19.5)	50.0 (44.3)	59.4 (53.3)	11.9 (11.4)	49.2 (29.9)	32.8 (31.5)
Pre 1–Post 8	27.8 (19.0)	66.7 (53.7)	86.8 (66.0)	19.9 (15.3)	69.2 (49.2)	40.6 (35.0)
Large Wood Recruitment (pieces/ha)						
Pre 1–Post 2	20.3 (16.6)	38.6 (33.6)	55.1 (50.5)	4.0 (7.4)	43.7 (34.6)	31.2 (31.2)
Pre 1–Post 5	26.7 (19.8)	56.0 (47.6)	66.5 (57.9)	12.9 (11.8)	54.6 (31.7)	37.5 (35.6)
Pre 1–Post 8	34.3 (20.7)	76.5 (59.8)	99.9 (73.6)	25.8 (27.3)	80.1 (53.4)	46.8 (42.2)
Wood Recruitment Volume (m^3/ha)						
Pre 1–Post 2	2.6 (1.8)	8.8 (12.7)	6.9 (7.8)	0.2 (0.4)	5.2 (4.9)	3.8 (4.4)
Pre 1–Post 5	3.6 (2.5)	10.6 (13.2)	11.4 (14.8)	1.0 (1.7)	6.6 (5.7)	4.0 (4.4)
Pre 1–Post 8	5.1 (2.6)	12.0 (13.7)	14.6 (13.8)	1.7 (2.2)	7.8 (5.9)	6.9 (8.4)

Appendix Table 3-6. Annualized rates of mortality and input of fallen trees, large wood pieces and large wood volume for the Pre 1–Post 2, Post 2–Post 5, and Post 5–Post 8 intervals for the reference, 100% treatment, and FPB reaches. For both RMZs and PIPs, values are the averages of site means with standard deviations in parentheses.

Interval	**RMZ**			**PIP**		
	REF	**100%**	**FPB**	**REF**	**100%**	**FPB**
Mortality (% of pre-harvest basal area)						
Pre 1–Post 2	5.9 (7.3)	7.1 (6.0)	16.8 (16.6)	2.0 (2.3)	9.7 (10.2)	20.9 (11.2)
Post 2–Post 5	2.5 (4.2)	4.4 (4.9)	12.8 (18.7)	1.2 (1.3)	8.1 (7.7)	5.0 (4.5)
Post 5–Post 8	1.5 (1.6)	4.3 (4.5)	11.3 (9.1)	0.5 (0.4)	7.6 (9.2)	5.3 (5.3)
Mortality (% of pre-harvest stems)						
Pre 1–Post 2	6.8 (7.2)	8.9 (6.9)	18.2 (15.0)	3.2 (4.4)	10.3 (10.3)	28.3 (12.9)
Post 2–Post 5	2.9 (4.6)	5.6 (5.9)	11.9 (18.1)	2.1 (1.9)	9.7 (6.7)	8.4 (5.8)
Post 5–Post 8	2.1 (2.1)	4.7 (4.4)	8.9 (7.2)	0.9 (0.7)	8.8 (9.3)	7.6 (7.7)
Fallen Tree Recruitment (trees/ha)						
Pre 1–Post 2	8.2 (8.7)	17.9 (16.6)	24.3 (22.8)	2.0 (3.7)	20.0 (16.4)	14.8 (15.0)
Post 2–Post 5	1.8 (1.3)	4.7 (4.7)	3.6 (3.1)	2.6 (2.4)	3.0 (2.7)	1.0 (2.8)
Post 5–Post 8	2.0 (0.7)	5.5 (4.3)	9.1 (11.8)	2.6 (4.3)	6.7 (7.8)	2.6 (4.1)
Large Wood Recruitment (pieces/ha)						
Pre 1–Post 2	10.2 (8.3)	19.3 (16.8)	27.6 (25.2)	2.0 (3.7)	21.8 (17.3)	15.6 (15.6)
Post 2–Post 5	2.1 (1.8)	5.8 (6.7)	3.8 (2.9)	3.0 (2.7)	3.6 (5.6)	2.1 (5.5)
Post 5–Post 8	2.5 (1.4)	6.8 (5.1)	11.1 (13.7)	4.3 (9.6)	8.5 (8.5)	3.1 (5.3)
Wood Recruitment Volume (m^3/ha)						
Pre 1–Post 2	1.3 (0.9)	4.4 (6.4)	3.4 (3.9)	0.1 (0.2)	2.6 (2.5)	1.9 (2.2)
Post 2–Post 5	0.3 (0.4)	0.6 (0.9)	1.5 (2.4)	0.3 (0.5)	0.5 (0.6)	0.1 (0.2)
Post 5–Post 8	0.5 (0.6)	0.5 (0.5)	1.1 (1.3)	0.2 (0.5)	0.4 (0.3)	1.0 (2.2)

CHAPTER 4 - STREAM TEMPERATURE AND COVER

William Ehinger, Stephanie Estrella, and Greg Stewart

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4-1. ABSTRACT

We used a Before-After Control-Impact study design to estimate the changes in riparian cover and stream temperature after timber harvest in non-fish-bearing headwater streams in western Washington. Each site was an entire non-fish-bearing stream basin. The study included six no-harvest reference sites and 11 sites that received a clearcut harvest with one of three riparian buffer treatments. The treatments were a 50-ft (15.2-m) wide buffer along each side of the perennial stream for 100% of its length (100%), a 50-ft (15.2-m) wide buffer along at least 50% of its length (Forest Practices-FP), and no buffer (0%). Harvested sites were also given unstable slope buffers so some buffer widths exceeded 50 ft (15.2 m), especially in the 100% treatment. We monitored for two years pre-harvest and nine years post-harvest.

Riparian shade decreased post-harvest in all treatments relative to unharvested reference sites. Decreases in canopy closure were less than 10 percentage points in the 100% treatment. Canopy closure declined 32 percentage points by Post 3 in the FP treatment, due to ongoing windthrow, and was still 15 points below pre-harvest levels at Post 9. Canopy closure in the 0% treatment declined by 85 percentage points by Post 3 and was 27 points below pre-harvest at Post 9.

None of the three buffer treatments were effective at preventing increases in summer water temperature within the non-fish-bearing stream. In terms of the magnitude and duration of the temperature increase, the 100% treatment was most effective with an increased seven-day average temperature response of 1.1°C in the first two years post-harvest followed by a decline to pre-harvest temperatures in Post 3. The FP treatment was less effective with temperature responses ranging from 0.5–1.2°C that changed little over the post-harvest period. The 0% treatment was least effective with an increase of 3.8°C in Post 1 that declined to 0.8°C in Post 9. The consistently elevated FP treatment temperature response was likely a function of the ongoing loss of shade due to windthrow in the buffered portion of two FP sites. However, there was evidence that site aspect and hyporheic exchange were factors at some individual sites.

Spring and fall temperatures were elevated at nearly all location in all sites with the mean monthly temperature response usually less than 1.0°C. There was evidence that temperature increases within the harvest unit decreased rapidly below the harvest unit through approximately 100 m of wider (than 50 ft) buffers downstream.

4-2. INTRODUCTION

Non-fish-bearing “headwater” (Type N) streams comprise more than 65% of the total stream length on industrial forestlands in western Washington (Rogers and Cooke 2007). These streams provide important subsidies of organic matter and macroinvertebrates (Wipfli *et al.* 2007), nutrients (Alexander *et al.* 2007), and cool water to downstream reaches. Stream temperature is an important determinant in many biological processes that may affect these subsidies and the growth and survival of aquatic biota (Wehrly *et al.* 2007; Friberg *et al.* 2013), many of which have narrow thermal tolerances for specific life stages (Richter and Kolmes 2005).

Stream temperature is a function of the water temperature entering the reach and energy exchanges between the stream and its surroundings (see Moore *et al.* 2005b). Radiative exchanges include direct and diffuse shortwave radiation inputs and long-wave radiation exchange with the surrounding atmosphere, vegetation, and terrain. In forested streams, shade provided by riparian vegetation attenuates incoming shortwave radiation and is the most important determinant of summer stream temperature (Brown 1969; Johnson and Jones 2000; Danehy *et al.* 2005; Groom *et al.* 2011). There are several other pathways for heat exchange in the stream environment. Latent heat exchange is associated with the evaporation or condensation of water and varies with the vapor pressure gradient between the water and the overlying air and turbulent exchange of the overlying air. Sensible heat exchange between the water and overlying air depends upon the temperature difference between the two and turbulent exchange. Bed heat exchange can occur when radiative energy is absorbed by the stream bed then transferred back to the water or, by conduction of heat from the water, to the stream bed or, via flow, into bed sediments. Estimates of latent and sensible exchange in forested environments are typically less than 10% of net radiation (Brown 1969; Johnson 2004; Moore *et al.* 2005a) while estimates of bed heat exchange are 10% of net radiation for a step-pool stream (Moore *et al.* 2005a) to 25% in a bedrock channel (Brown 1969). Ground water inflow in summer is usually cooler than stream water and can moderate diurnal and seasonal temperature (Webb and Zhang 1999). Hyporheic exchange of water between the stream and the underlying substrate typically moderates surface water temperature extremes and can be an important factor in local and reach-scale temperatures in headwater streams (Johnson 2004; Moore *et al.* 2005a).

Early studies of the direct effects of forest harvest on stream temperature demonstrated large decreases in shade and increases in summer stream temperature up to 11.6°C after unregulated harvesting with no buffers (Brown and Krygier 1970; Harris 1977; Feller 1981; Holtby and Newcombe 1982) and a long-term correlation between increasing stream temperatures and forest harvest (Beschta and Taylor 1988). These provided much of the initial justification for rules requiring riparian buffer zones along fish-bearing streams (Richardson *et al.* 2012). Moore and colleagues (2005a) reported more modest temperature increases of 2.5 to 5.0°C from a review of studies of harvest impacts following contemporary forest practices and more recent studies have generally confirmed this (Kibler *et al.* 2013; Bladon *et al.* 2018; Reiter *et al.* 2020). Moore and colleagues (2005b) suggested that much of the variability in results among studies was likely due to differences in buffer width, forest management within the buffer, and length of stream harvested. They also noted that other site-specific factors play a role. For example, studies have shown that stream width and depth, flow velocity and volume, length of surface flow (Janisch *et al.* 2012), subsurface hydrology (Story *et al.* 2003), upstream hydrology (Gomi *et al.* 2006), site aspect and elevation (Beschta *et al.* 1987; Isaak and Hubert 2001; Poole and Berman 2001; Moore *et al.* 2005b), geologic setting (Bladon *et al.* 2018), stream substrate size (Janisch *et al.* 2012), and distance downstream from a disturbance (Cole and Newton 2013) influence stream temperature response.

An explicit goal of the Forest Practices Habitat Conservation Plan (FPHCP; WADNR 2005) is to meet water quality standards (e.g., stream temperature) within non-fish-bearing streams as well as downstream in fish-bearing waters. Current Washington Forest Practices rules, based on the Forests and Fish Law (WFPB 2001), expanded riparian buffer requirements to include a 50-ft (15.2-m) wide, two-sided buffer along at least 50% of the length of perennial, non-fish-bearing headwater streams. The FPHCP assumed that 50 ft wide buffers would retain 50 to 75% of shade, measured as angular canopy density, and that temperature increases within buffered reaches "…are expected to be small". The FPHCP also assumes recovery of stream temperature to pre-harvest levels within unbuffered stream reaches would "likely be rapid" based on observations by Caldwell and colleagues (1991) and Summers (1982) of rapid understory vegetation regrowth. The FPHCP's general assumption of stream heating and cooling as it flows through unbuffered and buffered reaches is supported by Burton and Likens (1973) who noted rapid increases and decreases in stream temperature as water flowed through alternating unbuffered and buffered stream reaches. Heating and cooling has been observed in other studies of forested streams (Caldwell *et al.* 1991; Storey and Cowley 1997; Keith *et al.* 1998; Story *et al.* 2003; Wilkerson *et al.* 2006; Gravelle and Link 2007), however, energy budget models suggest this is due to inputs of cooler water rather than shading (Brown *et al.* 1971; Story *et al.* 2003; Garner *et al.* 2014). The assumption that stream temperature would rapidly return to pre-harvest levels is largely untested, although MacDonald and colleagues (2003), Gomi and colleagues (2006), and Rex and colleagues (2012) indicate that temperatures could remain elevated for at least five years.

Our objective was to estimate the effects of clearcut timber harvest with three different buffer treatments on riparian cover and water temperature. We assessed the degree to which each buffer treatment affects shade and the effects of shade reduction, if any, on stream temperature, both year round and during the critical summer period. Specific questions, taken from the first report (McIntyre *et al.* 2018), were:

1) What was the magnitude of change in riparian cover relative to the unharvested reference sites following timber harvest in each of the three buffer treatments?

2) What was the change in the daily maximum stream temperature following harvest within the non-fish-bearing streams?

3) What was the effect of each buffer treatment on the seven-day average daily maximum stream temperature at the boundary between non-fish-bearing and fish-bearing portions of the stream?

4-3. METHODS

4-3.1. STUDY SITES

The 17 study sites were perennial non-fish-bearing (Type Np) watersheds (WAC 222-16-030), delineated by the upstream extent of fish presence (fish/non-fish or F/N break) and included first-, second-, and third-order stream basins located in western Washington State. Sites were located along the Clearwater River, Humptulips River, and Wishkah River in the Olympic physiographic region; the North River, Willapa River, Nemah River, Grays River, Skamokawa River, and Smith Creek in the Willapa Hills physiographic region; and the Washougal River and Trout Creek in the South Cascade physiographic region. Study sites consisted of 30- to 80-year old managed Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*)-dominated second-growth forests on private, state, and federal forestlands. Sites were dominated by competent lithology types with average Type Np channel gradients ranging from 14% to 34% and catchment areas ranging from 12 to 49 ha (**Table 4-1**). We present site-selection criteria in McIntyre and colleagues (2018, Chapter 2 – *Study Design*).

Table 4-1. Study site characteristics. Trmt = treatment; Elev = elevation; BFW = bankfull width.

Block	Trmt	Elev (m)	Lithology	Stream Gradient (%)	Aspect	Type Np Stream Length (m)	BFW (m)	Area (ha)
OLYM	REF	163	Basalt flows and flow breccias	18	N	2737	2.6	44
	100%	72	Tectonic breccia	27	NE	1949	2.1	28
	FP	277	Basalt flows and flow breccias	25	SE	1070	1.1	17
	0%	233	Basalt flows and flow breccias	31	W	637	1.5	13
WIL1	REF	200	Basalt flows and flow breccias	19	SW	589	1.4	12
	100%	198	Basalt flows and flow breccias	18	SW	1029	2.1	31
	FP	197	Basalt flows and flow breccias	19	SW	325	1.5	15
	0%	87	Terraced deposits	16	NE	1525	1.8	28
WIL2	REF2	228	Basalt flows and flow breccias	18	SE	816	1.3	16
	100%	22	Basalt flows and flow breccias	21	SW	1257	1.9	26
	FP	183	Basalt flows and flow breccias	34	W	653	1.9	19
	0%	159	Basalt flows	21	E	933	2.2	17
WIL3	REF	241	Basalt flows	14	SW	2513	1.8	37
	100%	351	Basalt flows	19	SE	1359	2.2	23
CASC	REF	601	Tuffs and tuff breccias	21	N	1080	2	49
	FP	450	Andesite flows	16	E	822	1.5	26
	0%	438	Andesite flows	29	SE	420	1.6	14

4-3.2. EXPERIMENTAL TREATMENTS

The four experimental treatments included in the study are:

1) **Reference (REF):** No timber harvest during the study period.

2) **100% treatment (100%):** Clearcut harvest with a two-sided 50-ft (15.2-m) riparian buffer along the entire perennial stream length.

3) **Forest Practices treatment (FP):** Clearcut harvest with a current Forest Practices two-sided 50-ft (15.2-m) riparian buffer along at least 50% of the perennial stream length (**Figure 4-1**).

4) **0% treatment (0%):** Clearcut harvest with no riparian buffer.

The riparian management zone for Type Np and non-fish-bearing seasonal (Type Ns) waters in western Washington also includes a two-sided, 30-ft (9.1-m) wide equipment limitation zone (WAC 222-30-021(2)) to limit the amount of ground disturbance near the stream. Timber harvest on potentially unstable slopes or landforms that have the potential to deliver sediment or debris to a public resource, or that has the potential to threaten public safely, require an environmental checklist in compliance with the State Environmental Policy Act (SEPA; WAC 222-16-050(1)(d)), so harvest in these areas is generally avoided. In this study, no harvest activities were conducted on any potentially unstable slopes, regardless of buffer treatment, and all treatments included the equipment limitation zone.

The final buffer delineation by the landowners often occurred one or more years after we initiated the study, just prior to harvest, resulting in some inconsistencies between the intended and actual buffer layouts at several sites. The primary reason for differences was the addition of buffers on unstable slopes. For example, we intended the 100% treatment to be a continuous two-sided 50-ft (15.2-m) wide riparian buffer; however:

- In the OLYM-100% site, unstable slope buffers resulted in a contiguous unharvested buffer that merged across much of the drainage network leaving only one tributary, RB1, with a 50-ft (15.2-m) buffer along the entire length (see **Appendix Figure 2-3** in this report).

- In the WIL2-100% site, unstable slope buffers left the lower 150 m of stream unharvested and buffers much wider than 50 ft (15.2 m) between 150 m and 380 m above the F/N break (see **Appendix Figure 2-11** in this report).

The WIL2-FP site was buffered along the entire stream length because of unstable slope buffers (see **Appendix Figure 2-9** in this report).

We intended the 0% treatment to have no buffer; however:

- In the CASC-0% site, the F/N break (T1) was located 85 m downstream and into the fish-bearing buffer on the downstream tributary (see **Appendix Figure 2-16** in this report).

- In the WIL2-0% site, no harvest occurred along the 50 m of stream immediately above the F/N break (T1; see **Appendix Figure 2-12** in this report).

Timber harvest occurred in 2008 or 2009 at eleven of the 12 buffer treatment sites. The last site was harvested January to June 2016 as an FP treatment. In McIntyre and colleagues (2018), which reported study results from 2006 to 2011, we included this site as an unharvested reference (e.g., WIL2-REF1). This chapter on stream temperature and cover treats this site as the fourth FP treatment site, WIL2-FP, even though the entire stream length was buffered. We based this decision on the need for more information regarding application of the full range of the FP treatment in western Washington. This is the only chapter that includes this site as an FP treatment. The remaining chapters include the site as a second reference in the Willapa 2 block, consistent with McIntyre and colleagues (2018), and do not include data for this site in its post-harvest state.

OLYM-0% was harvested in July to August 2009 and, therefore, off limits to field crews during peak summer stream temperatures. Riparian cover measurements were not affected because there was time after harvest in early September to make the measurements prior to leaf fall. However, we likely missed maximum stream temperatures in 2009, so the first year post-harvest is 2010. There are only eight years of post-harvest data (2010-17), compared to nine years (2009-17) at most sites other than WIL2-FP (noted above).

4-3.3. RIPARIAN COVER

We calculated four metrics of riparian cover. Two metrics were calculated from hemispherical canopy photos taken approximately one meter above the water surface.

- Canopy and Topographic Density (CTD): defined as the percentage of the photograph obscured by vegetation or topography.

- Effective Shade: defined as one minus the ratio of total (direct plus diffuse radiation) below canopy radiation to total above canopy radiation (Stohr and Bilhimer 2008). We used the mean value for the entire seventh solar month.

Effective shade was included because it is best proxy for the actual change in incident shortwave radiation resulting from the buffer treatments, and CTD for comparison with other studies.

We measured canopy closure using a spherical densiometer at 1 meter (CC-1m) above the water surface and at the water surface (CC-0m). Canopy closure at 1-m is a well-accepted and commonly measured metric within the forest industry and CC-0m was included to capture shade provided by low vegetation and debris.

Hemispherical canopy photos were taken for the first seven years of the study, from 2007 to 2013, and then discontinued for budgetary reasons. As a result, effective shade and CTD were calculated only for 2007 to 2013. Measurements of canopy closure at CC-1m were taken from 2007 to 2017 and CC-0m from 2008 to 2017.

4-3.3.1. Hemispherical Canopy Photos

We took hemispherical photographs 1 m above the water surface using a Nikon Coolpix 995 digital camera with a FC-E8 fisheye lens. The camera was set to manual function for a wide-angle lens and medium resolution (Stohr and Bilhimer 2008). Images were analyzed using HemiView Canopy Analysis Software, v. 2.1 (Delta-T Devices, LTD, Cambridge, UK) set to the default simple solar model, which assumes clear sky theoretical shortwave radiation for the site's specific latitude. We took photographs at 10 stations per site, except in the OLYM-0% site, where only nine stations were established due to a change in the mapped location of the uppermost point of perennial flow after the study began. Stations were equally spaced longitudinally along the mainstem channel with the furthest downstream station located at a randomly assigned distance between zero and 50 m upstream from the F/N break, and the last station located at the previously identified uppermost point of perennial flow. The distribution of sampling locations ensured that measurements were collected within buffered and unbuffered reaches from channel initiation to Type N basin outlet. We used the same locations each year unless a station was inaccessible due to slash or windthrown trees. In these cases, photos were taken at, or within 2 m of, the original station. Each photo was scrutinized to ensure that over- and underexposed areas of the photo were correctly interpreted by the software. For example, brightly lit hillsides that were incorrectly classified as sky or overly dark skies classified as cover were darkened or lightened, respectively, until correctly interpreted.

Photographs were taken between 6 June and 9 August annually from 2007 through 2013 in most of the study sites, with the following exceptions:

- In 2009, photographs in the OLYM-0% were taken on 30 September shortly after harvest but before deciduous leaf fall.

- In 2009, photographs were not taken in WIL2-FP.

It was not always possible to take photographs during ideal lighting conditions (i.e., early morning, dusk, or overcast skies) to avoid glare from the sun, relatively dark sky, or relatively bright vegetation. We edited glare (blacked out), dark blue sky (lightened), and brightly lit vegetation (darkened) using Adobe Photoshop CS3 v. 10.0.1 software (Adobe Systems Inc.) prior to running the calculations in HemiView. Of the nearly 700 photographs taken, only five photos, taken in 2008 at WIL1-0%, had too much glare for a satisfactory analysis in HemiView. Retaking these photographs was not an option because harvest had already taken place so we elected to exclude all of the 2008 photo-derived data from this site in the analysis rather than include values from only half of the measurement stations.

To calculate CTD and effective shade, for which we used the mean value for the entire seventh solar month (22 June–21 July), we calculated the following values for all photographs in each year using the HemiView software:

- Monthly direct above canopy radiation (MJ/m^2, DirAb),
- Monthly direct below canopy radiation (MJ/m^2, DirBe),
- Monthly diffuse above canopy radiation (MJ/m^2, DifAb),
- Monthly diffuse below canopy radiation (MJ/m^2, DifBe), and
- Proportion of photograph that is visible sky (VISSKY).

4-3.3.2. Canopy Closure

We used a spherical densiometer (Lemmon 1956) to measure canopy closure at the same time and location as we took canopy photos. We took measurements mid-channel (facing upstream, downstream, right bank, and left bank) and at two heights: 1 m above the water surface to assess the overhead riparian cover, and at the water surface to include cover provided by low understory vegetation, instream woody debris, and logging slash (Werner 2009). For the measurement at the water surface, we slid the densiometer into position, taking care to minimize disturbance of any overhanging slash or vegetation. Overall, 17% of the stream length was obstructed by substantial amounts of logging slash. Where slash was so dense that we could not see the water surface or stream substrate, we assigned a value of 100% cover. This rarely occurred because slash could be introduced into the stream channel only in the unbuffered portions of the FP treatment and in the 0% treatment while buffered portions of the 100%, FP, and the entire REF sites were not affected.

In 2017, we measured CC-1m and CC-0m during the leaf off period (29 November 2017–13 March 2018) for comparison with summer values.

4-3.3.3. Correlation among Riparian Cover Metrics

Canopy photos were discontinued in 2013 due to budget constraints and only densiometer-based measurements were collected over the course of the entire study. Although photo-based measurements are less variable (OWEB 2000), we chose to rely on densiometer-based measurements because of the high degree of correlation between the metrics (Kelley and Kruger 2005). Pearson correlation coefficients and Bonferroni-adjusted p-values were calculated to evaluate the utility of relying solely on canopy closure to track riparian cover after harvest.

4-3.4. STREAM TEMPERATURE

We measured water temperature at 30-minute intervals using TidbiT thermistors (Onset Computer Corporation, Bourne, Massachusetts). We used narrow range HOBO StowAway TidbiT dataloggers (-5°C to 37°C range, 0.2°C stated accuracy) from 2006 to 2012 and then replaced them with HOBO TidbiT v2 dataloggers (-20°C to 70°C, 0.21° stated accuracy) in Fall 2012. At each location, we installed a TidbiT where there was sufficient water depth and flow to keep it submerged and stable substrate to prevent loss of the sensor during high flows (Schuett-Hames *et al.* 1999). We attached TidbiTs to iron rebar driven into the streambed. We used zip ties to suspend the TidbiTs in the water column and leaned woody debris on the rebar to protect the sensor from direct sunlight and detection (vandalism). Portions of these streams were very shallow (<3 cm), especially near channel initiation, and some sensors were installed very near the streambed surface. The likely effect of being positioned near the streambed, if any, was that in areas of upwelling, extremes in water temperature may be dampened by the influx of cooler subsurface flow.

We monitored at least four locations along the perennial stream length in each site. We based locations on the conceptual layout of buffers in the FP treatment (**Figure 4-1**). The intent was to measure water temperature at multiple locations along the main perennial channel to capture temperature changes throughout the Type Np stream. The harvest layout was done by the landowners shortly before harvest and these sometimes varied from the initial survey in 2005-2007. The result is that the monitoring locations were not precisely located with respect to the final FP buffers. We monitored comparable locations along the mainstem channel in the other treatments. We installed TidbiTs in all perennial tributaries near the confluence with, but above the influence of, the mainstem channel. Our convention for labeling these locations was RB (right bank) or LB (left bank) facing downstream and numbered beginning at the F/N break. The relatively high density of monitoring locations was intended to describe spatial variability within the Type Np stream as it flowed through buffered and unbuffered reaches and to provide redundancy in the event of missing data (e.g., in case of missing data at the F/N break, we used the next location upstream).

To monitor downstream temperature recovery in Type F Waters, we also monitored a location downstream from the harvest unit in six buffer treatment sites with at least 100 m of stream flowing through a fish-bearing stream buffer with no perennial tributaries. The D100 locations were added at the request of CMER reviewers and were intended to determine whether stream temperature in the Type F stream below the harvest unit was affected, independent of inflowing tributaries. The 100 m minimum distance was a compromise between the study authors and the CMER reviewers. Only six treatment sites presented the opportunity to add a D100 site either because the other treatment sites fed immediately into a higher order Type F stream or because of a tributary confluence just below the F/N junction (**Table 4-2**).

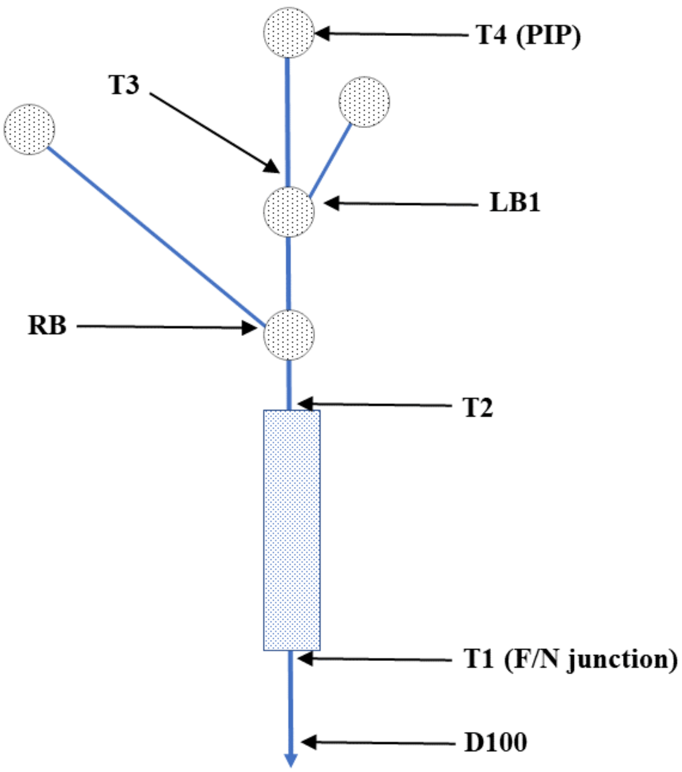


Table 4-2. Name and location of temperature monitoring stations.

Name	Location
T4	At or near the perennial initiation point (PIP)
T3	Mainstem within unbuffered portion of FP stream
T2	Mainstem within Type N buffer above F/N junction
T1	At F/N junction
D100	Mainstem within Type F buffer below F/N junction
LB,RB	Left bank and right bank tributaries of Type N stream

We monitored air temperature along the main stream channel at the same locations as water temperature. We used wide range HOBO StowAway TidbiT dataloggers (-20°C to 50°C range, 0.2°C stated accuracy) from 2006 to 2012 and then replaced them with HOBO TidbiT v2 dataloggers (-20°C to 70°C, 0.21° stated accuracy) in Fall 2012. We placed sensors 1 m above the ground, adjacent to the stream channel and protected from direct sunlight. These data were used to identify periods when the water temperature sensor was not fully submerged.

We downloaded temperature data each spring and fall using Onset Optic Shuttles (Onset Computer Corporation, Bourne, Massachusetts). TidbiTs were downloaded onsite and immediately reinstalled. At each download, we verified the serial number and recorded the sensor's status (submerged or exposed to air), the time of download, whether the TidbiT successfully relaunched, and whether the TidbiT was replaced. We graphically compared all water temperature data to nearby air temperature records to identify changes in the water-air temperature relationship that may indicate a sensor was not fully submerged. In addition, we used field records from the riparian cover surveys and riparian vegetation surveys to identify specific time periods when a TidbiT was not submerged so these data could receive special scrutiny. We flagged all suspect records in the database and excluded them from the analyses.

Prior to use, all TidbiTs passed a calibration check where they were compared to a National Institute of Standards and Technology (NIST) thermometer in an ice bath and in an ambient water bath (~18°C). We did not use TidbiTs that deviated by more than 0.2°C from the NIST thermometer. In 2010, we replaced all TidbiTs manufactured before 2007 and ran them through the same calibration check. Only eight of 182 narrow range TidbiTs failed. The magnitude of the differences never exceed 0.45°C. Based on the large proportion of sensors that passed the post-deployment calibration checks and the small deviation from the NIST thermometer seen in those that failed the calibration check, we believe any effect of sensor drift on the study results is very small relative to the magnitude of temperature change.

4-4. ANALYSIS

4-4.1. RIPARIAN COVER

We used generalized linear mixed-effects models (GLMM) that incorporate both fixed and random effects for hypothesis testing. GLMM can be used to fit data that derive from non-normal distributions with monotonic link transformations. In matrix form, this model can be represented as (**Eq. 4-1**):

$$Y = X\beta + Z\gamma + \epsilon \qquad \textbf{(Eq. 4-1)}$$

where: X is a vector of observations,

β is vector of unknown fixed-effects parameters,

Z is a random effects design matrix with a specified covariance structure,

γ is a vector of unknown random-effects parameters, and

ϵ is a vector of independent and identically distributed Gaussian random errors.

Our analyses evaluated the generalized null hypothesis (**Eq. 4-2**):

$$\Delta S_{\mathrm{REF}} = \Delta S_{100\%} = \Delta S_{\mathrm{FP}} = \Delta S_{0\%} \qquad \textbf{(Eq. 4-2)}$$

where: ΔS_{REF} is the change (post-harvest minus pre-harvest) in riparian cover in the reference sites, and $\Delta S_{100\%}$, ΔS_{FP}, and $\Delta S_{0\%}$ are the post-harvest change in the 100%, FP, and 0% treatments, respectively.

We conducted statistical analyses using the GLIMMIX procedure in SAS 9.4 (SAS 2013). We estimated model parameters using restricted pseudo-likelihood, beta distribution, and the logit link function for all shade metrics except CC-1m (**Table 4-3**), where the model was estimated using maximum likelihood (Method = Laplace) to get the model to converge. Site was included as a random effect. Fixed effects were treatment, period (e.g., Pre-harvest, Post 1, Post 2,…Post 9), and the treatment × period interaction. We initially included block as a random effect but dropped it because the associated variance estimate was zero (i.e., block did not explain any additional variation in the dependent variables). We determined the covariance matrix for the fixed-effect parameter estimates and denominator degrees of freedom for t and F tests according to the method of Kenward and Roger (1997), which is recommended for imbalanced designs, except for CC-1m which used the containment method to calculate degrees of freedom. (Kenward-Rogers option is not available when using Method = Laplace.) We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals and found no evidence of either.

Pairwise comparisons were used to estimate the effect size for each buffer treatment relative to the reference treatment in each post-harvest year where (**Eq. 4-3**):

$$\text{Effect size} = (\text{Trmt}_{i,\,\text{Post}\,j} - \text{Trmt}_{i,\,\text{Pre}}) - (\text{REF}_{\text{Post}\,j} - \text{REF}_{\text{Pre}}) \qquad \textbf{(Eq. 4-3)}$$

where: REF = reference treatment
Trmt = buffer treatment
i = buffer treatment (100%, FP, or 0%)
Pre = pre-harvest
Post = post-harvest
j = year post-harvest

Although the analyses were done using the Distribution = Beta and Link = logit, the effect sizes are presented in tables as percentages to better relate to the measured shade values. These were calculated using **Equation 4-3** and the least squares means transformed from Beta space to percentages. We did not adjust the P-values for multiple comparisons but focused on the overall pattern of riparian cover reduction and recovery in the post-harvest years.

Table 4-3. The SAS procedure, estimation technique, and distribution/link used in the analysis of buffer treatment effects for all responses. CC-1m and CC-0m = canopy closure at 1m and the water surface; CTD = canopy and topographic density; 7DTR = seven-day average temperature response at the buffer treatment location or F/N junction. ML = maximum likelihood with Laplace approximation; RPL = residual pseudo-likelihood; RML = restricted maximum likelihood.

Response Variable	SAS Procedure	Est. Technique	Distribution/Link
CC-1m	GLIMMIX	ML (Laplace)	Beta/Logit
CC-0m	GLIMMIX	RPL	Beta/Logit
Effective Shade	GLIMMIX	RPL	Beta/Logit
CTD	GLIMMIX	RPL	Beta/Logit
7DTR-Buffer treatment	GLIMMIX	RML	Gaussian/Identity
7DTR-F/N junction	GLIMMIX	RML	Gaussian/Identity

We examined the correlation among the four stream cover metrics using Pearson correlation coefficients and the associated, uncorrected probabilities using SYSTAT v13 statistical software (Systat Software Inc 2009).

4-4.2. STREAM TEMPERATURE

We addressed objectives two and three by calculating a daily temperature response for each monitoring location in the buffer treatment streams, then describing the magnitude and pattern of seasonal and temporal temperature changes using the mean monthly temperature response. Monthly means were less affected than seven-day averages by short-term fluctuations in the model or in stream temperature and allowed the calculation of more precise confidence intervals.

We used the maximum seven-day average TR in July–August to estimate the buffer treatment effect on the summer daily maximum temperature and included tables of the seven-day average daily maximum temperature for direct comparison with Washington State's not-to-exceed water quality criteria.

4-4.2.1. Calculation of Daily Temperature Response (TR)

We used generalized least squares regression to calculate a daily temperature response to account for the autocorrelation among residuals of the daily temperature values. We calculated daily temperature response at each location in each treatment stream using an approach similar to that advocated by Watson and colleagues (2001) and modified by Gomi and colleagues (2006). This calculation involves two steps:

Step 1. We used a generalized least squares (GLS) regression of treatment vs. reference daily maximum temperature using the pre-harvest period data (**Eq. 4-4**).

$$y_t = \beta_0 + \beta_1 x_t + \beta_2 x_t^2 + \sin(2\pi t/365) + \cos(2\pi t/365t) + \varepsilon_t \qquad \textbf{(Eq. 4-4)}$$

where: y_t is the maximum temperature in the treatment site on day t,
x_t is the maximum temperature in the reference site on day t,
β_0, β_1, and β_2 are the estimated regression coefficients,
$\sin(2\pi t/365)$ and $\cos(2\pi t/365)$ are terms to account for seasonal variability, and
ε_t is an error term modeled with an autoregressive moving average (ARMA) process.

ARMA models (Pinheiro and Bates 2000) are the combination of an autoregressive (AR) model in which the current observation is expressed as a linear function of previous observations plus a homoscedastic white noise term (**Eq. 4-5**):

$$\varepsilon_t = \phi_1 \varepsilon_{t-1} + \cdots + \phi_p \varepsilon_{t-p} + a_t \qquad \textbf{(Eq. 4-5)}$$

where: ε_t is an error term on day t,
ε_{t-p} is an error term p days before,
ϕ_p is the autocorrelation coefficient at lag p, and
a_t is white noise centered at 0 and assumed to be independent of previous observations;

and a moving average (MA) model in which the error in the current observation is expressed as a series of correlated noise terms (**Eq. 4-6**):

$$\varepsilon_t = \theta_1 a_{t-1} + \cdots + \theta_q a_{t-q} + a_t \qquad \textbf{(Eq. 4-6)}$$

where: ε_t is an error term on day *t*,
a_{t-q} is the noise term q days before, and
θ_q is the correlation coefficient at lag q.

The combined ARMA model is therefore (**Eq. 4-7**):

$$\sum_{i=1}^{p} \phi_i \varepsilon_{t-1} + \sum_{j=1}^{q} \theta_j a_{t-j} + a_t \qquad \textbf{(Eq. 4-7)}$$

The parameters of the ARMA model were determined during the GLS regression, which was conducted using the gls function from the Linear and Nonlinear Mixed-effects Models (nlme) package by Pinheiro and colleagues (2012) in 64-bit R 2.15 (R Development Core Team 2012). We began with a lag one autoregressive term and examined the model residuals for autocorrelation, homoscedasticity, and normality (partial autocorrelation plots for autocorrelation, plot of residuals vs. time and residuals vs. predicted values for heteroscedasticity, and Q-Q plots for normality). This process was repeated with an AR term one order higher (up to lag six) until there was no significant ($P < 0.05$) autocorrelation through lag 10 and the residuals were homoscedastic, relative to the predicted value and to time, and were approximately normally distributed.

If these conditions were not met with a lag six AR term, then we repeated the sequence with an MA term equal to one. If no suitable model was found using all combinations of AR terms (one through six) and MA terms (one or two) then the process was repeated using data from a different location within the same reference site. For example, locations high in the watershed tended to be better correlated with similarly-placed reference locations.

The square of the correlation coefficient (r^2) describes the proportion of the dependent variable's variance explained by an ordinary least squares regression model. The standard calculation of r^2 is not appropriate to GLS, so we estimated a coefficient of determination (pseudo R^2) based on likelihood-ratios (Magee 1990) (**Eq. 4-8**):

$$\text{Pseudo } R^2{}_{LR} = 1 - \exp({}^{-2}/_{n} * (\log Lik(x) - \log Lik(0))) \qquad \textbf{(Eq. 4-8)}$$

where: log$Lik(x)$ is the log-likelihood from the fitted model, and
log$Lik(0)$ is the log-likelihood from the null model (i.e., intercept only).

Pseudo R^2 is interpreted in the same manner as r^2, with pseudo $R^2 = 0$ indicating that the model explains none of the variation and pseudo $R^2 = 1$ indicating the model explains all the observed variation. We performed the extraction of log-likelihoods and calculation of R^2 using routines in the R MuMIn package (Barton 2012), and incorporated the ARMA correlation structure into the null model so that pseudo R^2 reflects the adequacy of the prediction model.

All GLS regressions exhibited significant lag one or greater autocorrelation in the residuals. Autoregressive lag terms in final models for daily maximum stream temperature ranged from one to four. Twenty-six locations required lag one, 27 required lag two, 14 required lag three, and three required lag four AR terms. The MA term was needed in only seven locations. Pseudo R^2 values varied widely with low values usually occurring at locations with relatively short (<10–100 m) reaches of contiguous surface flow immediately above the monitoring location. This typically occurred in one of two situations: either the monitoring location was near the upper limits of perennial flow or the channel was dry some portion of the year immediately above the monitoring location. The monitoring locations used in the Buffer Treatment effects analysis had pseudo R^2 values ranging from 0.398 in the CASC-0% site to 0.930 in the WIL1-100% site. Over all locations pseudo R^2 ranged from 0.372 to 0.930 with a median value of 0.699.

Step 2. We calculated the daily temperature response (TR) as the observed temperature minus the predicted temperature using **Equation 4-4** (**Eq. 4-9**).

$$TR = (y_t - \hat{y}_t) \qquad \textbf{(Eq. 4-9)}$$

where: y_t is the observed temperature on day *t*, and
$\hat{y}_t$ is the predicted temperature on day *t*.

We did not use the WIL1-REF and WIL3-REF sites as a reference in any of the regressions because of poor model fit. In the WIL1-REF site, the heterogeneous distribution of the residuals may have been due to the windthrow caused by the December 2007 storm. The WIL3-REF site had many isolated reaches with no surface flow during the summer and the daily maximum stream temperatures were low and temporally stable relative to all other sites resulting in a non-linear relationship and heterogeneous distribution of residuals. We paired the WIL1, WIL2, and WIL3 buffer treatment sites with the WIL2-REF2 site. All OLYM and CASC treatment sites were paired with the reference from the same block.

4-4.2.2. Calculation of Mean Monthly Temperature Response (MMTR)

We used the gls function within the nlme package in R to estimate the mean monthly temperature response (MMTR) and 95% confidence intervals, using the daily TR values calculated above, for each month in the post-harvest years. We included an ARMA term in the model to account for the autocorrelation present and the weights = VarIdent option because the variance differed among months (**Eq. 4-10**).

$$y_i = \beta_j + \varepsilon_{ij} \qquad \textbf{(Eq. 4-10)}$$

where: y_i is the daily temperature response,
β_j are the monthly mean responses for months j=1…12,
ε_{ij} are the errors.

The errors are modeled using an ARMA correlation structure as described in **Equations 4-5**, **4-6**, and **4-7**, above.

Each month was allowed to have a different error variance (**Eq. 4-11**):

$$Var(\varepsilon_{ij}) = \sigma^2 \delta^2{}_{ij} \qquad \textbf{(Eq. 4-11)}$$

where: σ^2 is the variance parameter, and

$\delta^2{}_{i,j=2\ldots12}$ representing the ratio of the standard deviations between jth month and the first month (Pinheiro and Bates 2000).

The large number of MMTR (i.e., 12 months x nine post-harvest years at multiple locations per each of 12 harvested sites) increased the likelihood of Type II errors so it is inappropriate to emphasize any single monthly estimate. Rather, we focused on patterns in the direction, magnitude, seasonality, and post-harvest trajectory of the monthly estimates.

4-4.2.3. Stationarity of Reference Sites and Sensitivity of the Method

The use of a reference site assumes that in the absence of harvest the treatment and reference conditions are correlated and that this relationship does not change over the course of the study (i.e., is stationary). If this relationship changes (e.g., due to the reference basin changing over time), then spurious changes may be detected in the treatment sites.

We used the same method described above to fit a regression model of daily maximum stream temperature between several unharvested sites to assess the stability of the relationship over time. WIL2-REF2 was the designated 'reference' site and WIL2-FP, prior to harvest in 2016, and OLYM-REF were the 'treatment' sites. We used the first two years of data to calibrate the model (July 2006 to June 2008 at WIL2-FP; September 2006 to August 2008 at OLYM-REF) and the remaining data set (2008 to 2015 at WIL2-FP; 2008 to 2017 at OLYM-REF) to calculate post-calibration MMTR values. The CASC-REF site was not included because it is too far from WIL2-REF2, approximately 150 km, to be considered a viable reference.

4-4.2.4. Statistical Analysis of Buffer Treatment Effect on Temperature

We used the maximum seven-day average TR (7DTR) during July–August to estimate human-caused change in maximum summer stream temperature. These 7DTR values, one value per year per site for each pre-harvest and each post-harvest year, were used in the analyses described below.

The analyses evaluated the generalized null hypothesis (**Eq. 4-12**):

$$\Delta 7DTR_{100\%} = \Delta 7DTR_{\mathrm{FP}} = \Delta 7DTR_{0\%} \qquad \textbf{(Eq. 4-12)}$$

Where: $\Delta 7DTR_{100\%}$, $\Delta 7DTR_{\mathrm{FP}}$, and $\Delta 7DTR_{0\%}$ are the difference (post- minus pre-harvest) in 7DTR in the 100%, FP, and 0% treatments, respectively.

We used GLMMs that incorporate both fixed and random effects for hypothesis testing. In matrix form, this model is (**Eq. 4-13**):

$$Y = X\beta + Z\gamma + \epsilon \qquad \textbf{(Eq. 4-13)}$$

where: Y is Δ7DTR,
X is a vector of 7DTR observations,
β is a vector of unknown fixed-effects parameters,
Z is a random effects design matrix with a specified covariance structure,
γ is a vector of unknown random-effects parameters, and
ϵ is a vector of independent and identically distributed Gaussian random errors.

The analysis was run using a Gaussian distribution/identity link. Site was included as a random effect and fixed effects were treatment, period, and the treatment × period interaction. We initially included block as a random effect but dropped it because the variance estimate associated with block was zero (i.e., block did not explain any additional variation in the dependent variables). We determined the covariance matrix for the fixed-effect parameter estimates and denominator degrees of freedom for *t* and *F* tests following the method of Kenward and Roger (1997), which is recommended for unbalanced designs. We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals.

The Δ7DTR for each buffer treatment was estimated using (**Eq. 4-14**):

$$\Delta 7\text{DTR} = 7\text{DTR}_{i,\text{Post}\,j} - 7\text{DTR}_{i,\text{Pre}} \qquad \textbf{(Eq. 4-14)}$$

where: Pre = pre-harvest
Post = post-harvest
i = buffer treatment (100%, FP, or 0%)
j = year post-harvest

Differences among the treatments were estimated as:

$$(7\text{DTR}_{100\%,\ \text{Post}\,j} - 7\text{DTR}_{100\%,\ \text{Pre}}) - (7\text{DTR}_{\text{FP},\ \text{Post}\,j} - 7\text{DTR}_{\text{FP},\ \text{Pre}}) \qquad \textbf{(Eq. 4-15)}$$

$$(7\text{DTR}_{100\%,\ \text{Post}\,j} - 7\text{DTR}_{100\%,\ \text{Pre}}) - (7\text{DTR}_{0\%,\ \text{Post}\,j} - 7\text{DTR}_{0\%,\ \text{Pre}}) \qquad \textbf{(Eq. 4-16)}$$

$$(7\text{DTR}_{\text{FP},\ \text{Post}\,j} - 7\text{DTR}_{\text{FP},\ \text{Pre}}) - (7\text{DTR}_{0\%,\ \text{Post}\,j} - 7\text{DTR}_{0\%,\ \text{Pre}}) \qquad \textbf{(Eq. 4-17)}$$

where: buffer treatment = 100%, FP, or 0%
Pre = pre-harvest
Post j= post-harvest year

Estimates of the mean effect size are tabulated and those with p-values <0.05 indicated. We chose 95% CI by convention and we elected not to adjust for multiple comparisons. Reviews conducted through the Washington Department of Natural Resource’s Adaptive Management Program involve many stakeholder groups who often request additional analyses. We chose not to adjust for multiple comparisons so that reported p-values in successive revisions would not change simply due to a request for additional comparisons or because of the ongoing analysis of additional years’ data. The combination of a small number of replicates in each treatment and a large number of pairwise comparisons does increase the likelihood of Type II error; therefore, we focused on the direction, magnitude, and patterns of the effects across the treatments and time rather than any specific pairwise comparison.

Forest policy stakeholders were interested in the buffer treatment effects on water temperature where it discharged into fish-bearing waters. As noted earlier, some sites were not harvested as planned and the harvest unit did not always extend to the F/N break. As a result, we ran the analysis on two different overlapping sets of monitoring locations:

1) F/N break analysis — We used data from the location at or nearest the F/N break to evaluate the effects of the actual harvest on stream temperature where the stream discharges to fish-bearing waters. This location is at the F/N break in all sites except the WIL1-100% site, where we used the next location upstream because of missing temperature data at the F/N break (**Table 4-4**).

2) Buffer Treatment analysis — We used data from the location in each site that best represented the intended buffer treatment to isolate the effects of that buffer treatment on stream temperature. Eight of the 12 sites used the same location as the F/N break analysis. In the remaining four sites (OLYM-100%, WIL2-100%, WIL2-0%, and CASC-0%), we used other locations farther upstream where the actual riparian buffer matched the buffer treatments.

Note that the buffer treatment effects for the FP treatment are the same for both analyses because the monitoring locations were the same.

Table 4-4. Temperature monitoring location within each treatment basin that is nearest the F/N break, that best represents the intended buffer treatment, and whether a site was monitored downstream of the harvest unit. Notes describe the reason why the two monitoring locations differ. Trmt = treatment; F/N = F/N analysis locations; BT = buffer treatment analysis locations; US = upstream location used in the longitudinal analysis; DS = downstream location below harvested stream reach. (See site maps in **Appendix 2A** in this report.)

Trmt	Block	F/N	BT	US	DS	Notes
100%	OLYM	T1	RB1	-	-	Wider buffers due to unstable slopes.
	WIL1	T2	T2	T2	D100	Missing data at T1.
	WIL2	T1	LB3	T2	T1	Unstable slopes buffer. Missing data at T3.
	WIL3	T1	T1	-	-	
FP	OLYM	T1	T1	-	-	
	WIL1	T1	T1	T1	D100	
	WIL2	T1	T1	-	-	
	CASC	T1	T1	-	-	
0%	OLYM	T1	T1	T1	D100	
	WIL1	T1	T1	T1	D100	
	WIL2	T1	T3	-	-	No harvest on lower 50 m. Missing data at T2.
	CASC	T1	T3	T3	T1	T1 within fish-bearing buffer.

4-4.2.5. Relationship of Temperature Change to Shade, Aspect, and Discharge

We examined the relationship between post-harvest temperature change, July MMTR, and the four riparian cover metrics using Pearson correlation coefficients and the associated, uncorrected probabilities using SYSTAT v13 statistical software (Systat Software Inc 2009). We first calculated the correlation coefficients across sites using the site-level mean for the riparian cover metrics for each post-harvest year to examine the strength of the relationship over time. We then calculated the Pearson correlation coefficient of July MMTR with CC-1m and CC-0m at each individual site using the Buffer Treatment locations except in OLYM-100% and WIL2-100% where we used the F/N break location (T1) and plotted all three variables over time.

We examined the temperature response by buffer type using box plots of July MMTR by four buffer types–greater than 50 ft, 50 ft, the 56 ft buffer used for perennial initiation points (PIP), and no buffer–for each post-harvest year. In addition, we graphically examined the relationship of aspect and temperature response using plots of July MMTR vs. aspect for each post-harvest year. We calculated Pearson correlation coefficients to evaluate the relationship between discharge and July MMTR at the six harvested sites where discharge was monitored.

4-4.2.6. Longitudinal Patterns in Temperature Response

We graphically examined July MMTR, CC-1m, buffer location and type, and presence/absence of surface flow (surveyed in 2010) along the main Type N channel for patterns of post-harvest warming. All sites, except WIL2-FP, and all post-harvest years were plotted. WIL2-FP was omitted because of the paucity of locations with sufficient data and the limited number of post-harvest years.

We examined the effects of harvest on water temperature downstream of the harvest unit at the six sites with monitoring locations below the harvested stream reach by comparing the July MMTR at the location within the Type N channel nearest the bottom of the harvested reach (the upstream location) with the monitoring location below the harvest unit (the downstream location) (**Table 4-4**). These downstream locations had much wider riparian buffers because they were:

- Within a fish-bearing stream reach (WIL1-100%, WIL1-FP, WIL1-0%, OLYM-0%) or
- Within the Type N channel and:
    - Within an unstable slope buffer (WIL2-100%), or
    - Within the buffer of an adjacent Type F stream (CASC-0%).

Several aspects of downstream temperature effects were examined. First, was stream temperature elevated after harvest at the downstream locations (after flowing through unharvested riparian buffers much wider than the prescribed 50-ft Type N buffers)? Second, if so, how many years after harvest was stream temperature elevated? Finally, was the magnitude of warming less at the downstream location compared to the upstream location (i.e., was the buffer treatment effect lessened by the wider buffer downstream)? The magnitude of warming was calculated as the difference between July MMTR at the downstream and upstream locations. July MMTR at the upstream location, the downstream location, and the differences were tabulated for each site.

4-5. RESULTS

4-5.1. RIPARIAN COVER

4-5.1.1. Correlation Among Metrics

The three riparian cover metrics measured one meter above the water surface were strongly correlated to each other with correlation coefficients of at least 0.950 and P <0.05 (**Table 4-5**) and exhibited a linear relationship. Correlations with CC-0m were weaker, (0.667–0.714) and were non-linear. The non-linearity was especially pronounced at locations where logging slash in the channel resulted in high CC-0m values but low values for the metrics measured above the slash. Overall, this suggests CC-1m is an adequate surrogate for effective shade for the long-term analyses, although we acknowledge it is a less precise measure.

Table 4-5. Pearson correlation coefficients for riparian cover metrics. Bonferroni-adjusted p-values <0.001. CC-1m = canopy closure at 1m; CC-0m = canopy closure at the water surface; CTD = canopy and topographic density; Eff Shade = effective shade.

	Eff Shade	**CTD**	**CC-0m**
CC-1m	0.950	0.962	0.714
Eff Shade		0.973	0.667
CTD			0.677

4-5.1.2. Buffer Treatment Effect

All sites were well-shaded pre-harvest with mean values of 94%, 95%, 94%, and 89% for CC-1m, CC-0m, CTD, and effective shade, respectively, with very little variability among sites (**Table 4-6**; **Figure 4-2**) before harvest. Values of CC-1m and CC-0m were very similar pre-harvest and throughout the study in the REF sites reflecting the paucity of low vegetative cover (<1m) and instream debris.

Riparian cover declined after harvest in all buffer treatments reaching a minimum around Post 4. The treatments, ranked from least to most change, were REF, 100%, FP, and 0% for all metrics and across all years. The P-value for the treatment × period interaction term in the GLMM analyses for all riparian cover metrics was <0.0001 indicating significant differences among the treatments post-harvest (**Table 4-7**). Least squares means with 95% confidence intervals are listed in **Table 4-8**. Pairwise comparisons of each treatment by post-harvest year are presented in **Appendix Tables 4-1** through **4-4**. The pattern of post-harvest decreases in CC-1m, CTD, and effective shade were very similar, although the specific values differed, especially for the FP and 100% treatments. Canopy closure at 1m decreased by as much as 10 percentage points post-harvest in the 100% treatment but no significant (P <0.05) differences were noted. The FP and 0% treatments declined by as much as 32 and 87 points, respectively, post-harvest and remained 15 and 27 points, respectively, below pre-harvest levels after nine years. Effective shade results closely matched those of CC-1m in magnitude with decreases of 11, 36, and 74 points in the 100%, FP, and 0% treatments. Significant post-harvest decreases were noted for all treatments and all years reflecting the greater precision of hemispherical photo analysis over densiometer measurements. The magnitude of change in the CTD results tended to be slightly less for the FP and 0% treatments than the CC-1m and effective shade results and indicated significant decreases of up to 7, 20, and 50 percentage points post-harvest.

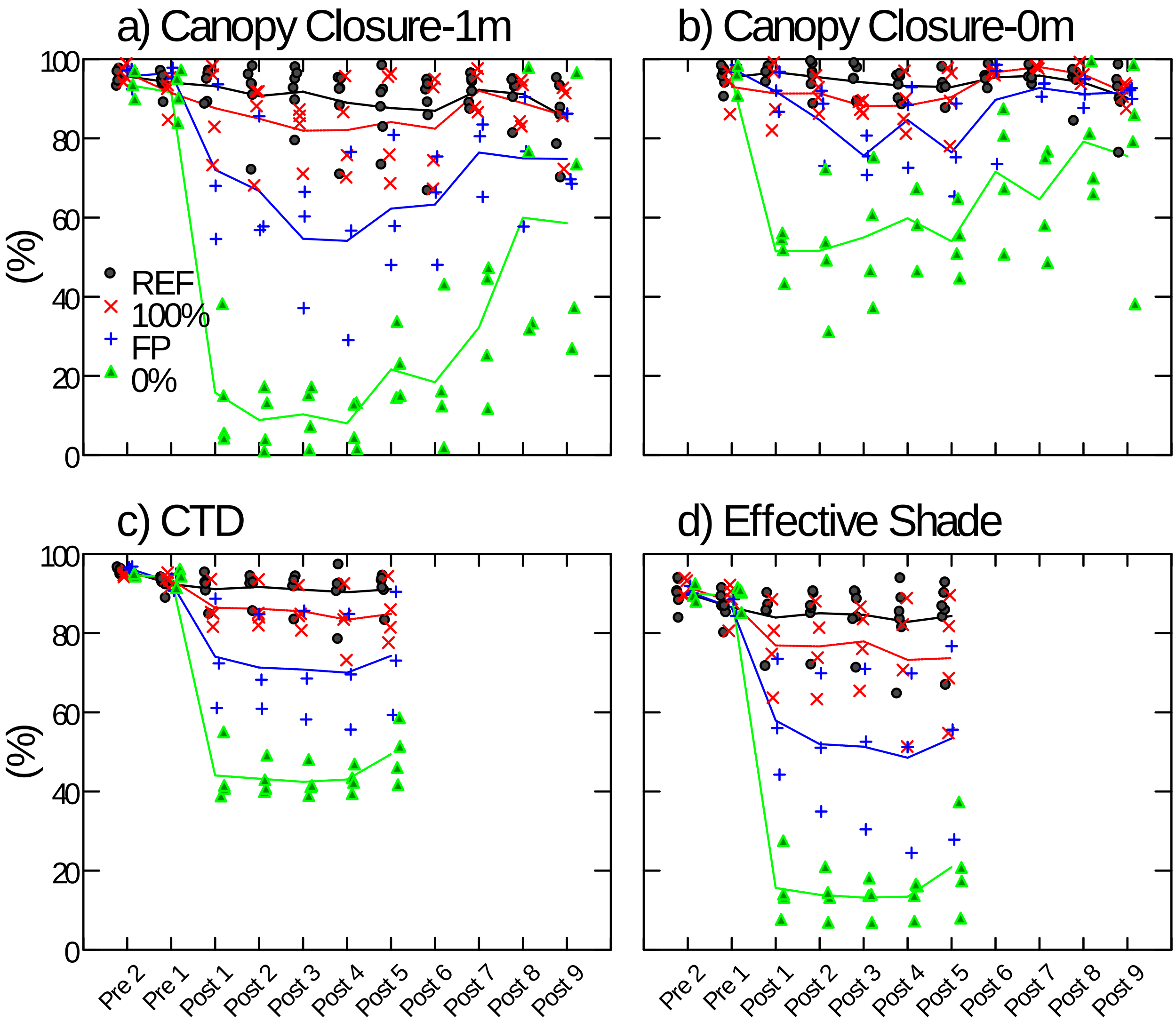


Figure 4-2. Mean values for riparian cover metrics by treatment and period (line) and by site (symbols). We did not measure canopy closure-0m until 2008 (Pre 1). Also, we were unable to calculate canopy and topographic density (CTD) and effective shade for WIL1-0% in 2008. Hemispherical photos for estimating CTD and effective shade were not collected after Post 5.

Table 4-6. Mean values for riparian cover metrics by treatment and period. Hemispherical photos for estimating canopy and topographic density and effective shade were not collected after Post 5. CC-1m = canopy closure at 1 m; CC-0m = canopy closure at 0 m; CTD = canopy and topographic density; Eff Shade = effective shade. Sample sizes: REF= 4 sites; 100% = 4 sites; FP = 4 sites for Pre–Post 2, otherwise 3 sites; 0% = 4 sites.

Treatment	Period	CC-1m	CC-0m	CTD	Eff Shade
REF	Pre	95	95	94	88
	Post 1	93	97	91	84
	Post 2	91	95	92	85
	Post 3	92	94	91	85
	Post 4	89	93	90	83
	Post 5	88	93	91	84
	Post 6	87	95		
	Post 7	92	96		
	Post 8	91	94		
	Post 9	85	90		
100%	Pre	94	93	94	90
	Post 1	88	91	86	77
	Post 2	85	91	86	77
	Post 3	82	88	86	78
	Post 4	82	88	83	73
	Post 5	84	90	85	74
	Post 6	82	97		
	Post 7	92	98		
	Post 8	89	96		
	Post 9	86	91		
FP	Pre	96	98	95	89
	Post 1	72	92	74	58
	Post 2	67	85	71	52
	Post 3	55	76	71	51
	Post 4	54	85	70	49
	Post 5	62	76	74	53
	Post 6	63	90		
	Post 7	76	93		
	Post 8	75	91		
	Post 9	75	92		
0%	Pre	92	95	94	90
	Post 1	16	51	44	16
	Post 2	9	52	43	14
	Post 3	10	55	43	13
	Post 4	8	60	43	14
	Post 5	22	54	50	21
	Post 6	19	71		
	Post 7	33	65		
	Post 8	61	80		
	Post 9	59	75		

Table 4-7. Results of the generalized linear mixed-effects model for riparian cover metrics. Significant (P <0.05) treatment × period interaction terms indicate pre- to post-harvest differences among treatments. The analysis was run using the GLIMMIX procedure, Beta distribution, and logit link. CTD = canopy and topographic density; Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Metric	Effect	Num DF	Den DF	F Value	Pr > F
Canopy Closure-1m	Treatment	3	13.4	42.07	<0.0001
	Period	9	126	34.54	<0.0001
	Treatment × Period	27	126	10.85	<0.0001
Canopy Closure-0m	Treatment	3	12.7	23.59	<0.0001
	Period	9	110	11.12	<0.0001
	Treatment × Period	27	109	4.17	<0.0001
CTD	Treatment	3	12	38.31	<0.0001
	Period	5	73.9	147.45	<0.0001
	Treatment × Period	15	73.9	37.46	<0.0001
Effective Shade	Treatment	3	12	32.39	<0.0001
	Period	5	74	133.08	<0.0001
	Treatment × Period	15	74	32.68	<0.0001

Table 4-8. Estimated change for riparian cover metrics based on pairwise comparisons using the generalized linear mixed-effects model analyses. Least squares means were converted to percent and the changes was calculated per Equation 4-3. Values with P <0.05 are in **bold type**. CC-1m = canopy closure at 1 m; CC-0m = canopy closure at 0 m; CTD = canopy and topographic density; Eff Shade = effective shade.

	CC-1m			**CC-0m**			**CTD**			**Eff Shade**		
Year	**100%**	**FP**	**0%**	**100%**	**FP**	**0%**	**100%**	**FP**	**0%**	**100%**	**FP**	**0%**
Post 1	-4	**-17**	**-83**	-6	-7	**-46**	**-5**	**-16**	**-49**	**-7**	**-27**	**-71**
Post 2	-5	**-22**	**-86**	-4	**-13**	**-45**	**-6**	**-20**	**-50**	**-8**	**-34**	**-73**
Post 3	-10	**-32**	**-87**	-7	**-18**	**-41**	**-6**	**-19**	**-50**	**-7**	**-35**	**-74**
Post 4	-6	**-28**	**-79**	-5	-8	**-34**	**-7**	**-20**	**-49**	**-10**	**-36**	**-72**
Post 5	-4	**-24**	**-85**	-4	**-19**	**-41**	**-7**	**-16**	**-43**	**-11**	**-32**	**-65**
Post 6	-3	**-20**	**-70**	1	-6	**-22**						
Post 7	1	**-18**	**-73**	1	-4	**-32**						
Post 8	-5	**-12**	**-62**	0	-6	**-18**						
Post 9	-3	**-15**	**-27**	-2	-2	**-18**						

4-5.1.3. Within Site Variability and Relationship with Tree Mortality

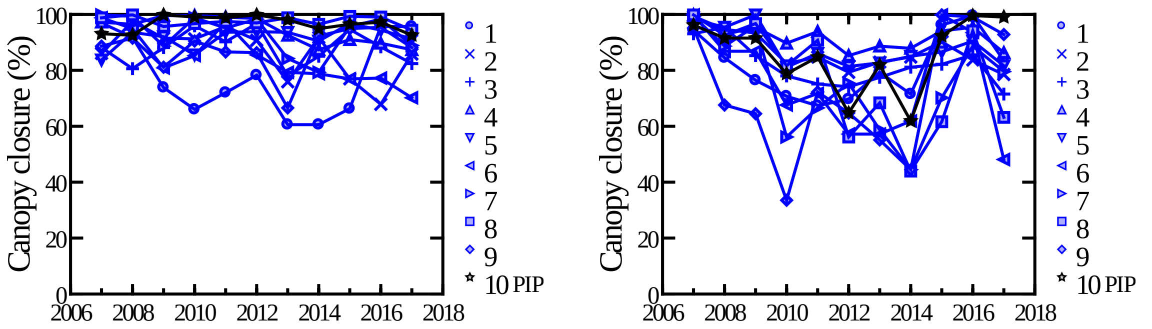
Riparian cover was consistently high, >70% at nearly all locations within each unharvested site and within the buffered portions of the harvested sites, except those sites with high tree mortality due to windthrow (**Table 4-9**; **Figures 4-3** through **4-6**). Of the unharvested sites WIL1-REF had the highest rates of riparian tree mortality (4.6–38.4%), nearly all due to windthrow, and the lowest and most variable CC-1m values in unharvested sites over the course of the study (**Figure 4-3**). (Note that tree mortality surveys at WIL2-FP were done prior to harvest in 2016 and so this site is categorized as unharvested in **Table 4-9** and **Figure 4-3**.) The 100% treatment sites with little windthrow, OLYM-100% and WIL3-100%, had uniformly high CC-1m throughout the study with few values less than 80% (**Figure 4-4**). In contrast, WIL1-100%, where riparian tree mortality ranged from 7.7–17.8%, and WIL2-100%, where mortality ranged from 8.0–14.3%, had lower and more variable CC-1m over the post-harvest period. Of the three FP sites where vegetation surveys were done after harvest, CC-1m within the buffered reach remained consistently high only at CASC-FP, where riparian tree mortality was low (0.0–2.8%) (**Figure 4-5**). Tree mortality was high at both OLYM-FP (1.4–18.3%) and WIL1-FP (8.4–34.6%) and CC-1m within the buffer decreased over the first three to four years post-harvest and the variability was high, similar to that seen in other sites with high post-harvest tree mortality (e.g., WIL1-REF, WIL1-100%, and WIL2-100%).

In the 0% sites and in the unbuffered reaches of FP sites, CC-1m decreased after harvest at all locations that were not shaded by nearby fish-bearing stream buffers (e.g., location 1 at all 0% sites) or by shrubs or uncut, nonmerchantable trees (e.g., location 10-PIP at CASC-0%) (**Figures 4-5** and **4-6**). CC-1m began increasing four to six years after harvest.

Table 4-9. Tree mortality presented as percentage of basal area lost during each of four time periods; Pre-harvest (Pre), Immediate post-harvest (IPH) to Post 2, Post 2 to Post 5, and Post 5 to Post 8. Pre-harvest (Pre) windthrow occurred from 2007 to harvest; Immediate post-harvest (IPH-Post 2) from harvest through spring 2010; Post 2-5 from fall 2010 through spring 2013; and Post 5-8 from fall 2013 through spring 2016. All measurements at WIL2-FP were taken prior to harvest in 2016 and therefore are included with the unharvested sites.

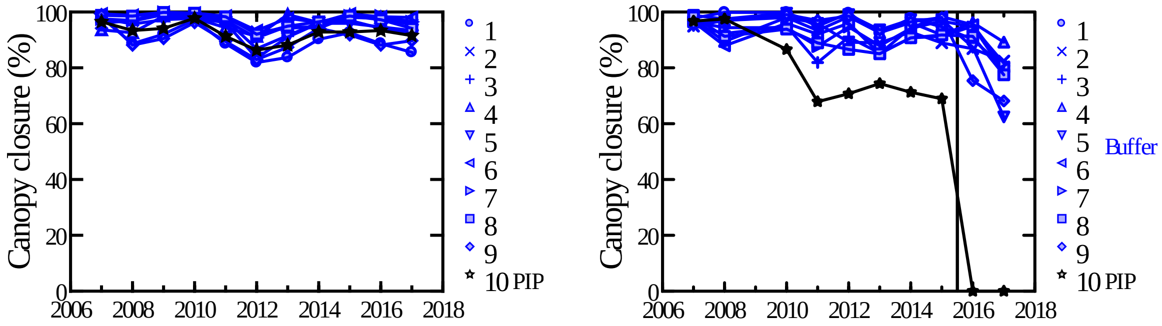
Treatment	Site	Pre	IPH-Post2	Post2-5	Post5-8
Unharvested	OLYM-REF	0.7%	4.3%	0.6%	0.4%
	WIL1-REF	38.4%	20.4%	11.0%	4.6%
	WIL2-REF2	11.3%	4.5%	1.3%	1.4%
	WIL2-FP*	11.7%	3.7%	1.0%	0.8%
	WIL3-REF	0.8%	1.3%	0.5%	1.3%
	CASC-REF	0.6%	1.1%	0.6%	0.6%
100% buffer	OLYM-100%	1.7%	1.5%	0.1%	0.2%
	WIL1-100%	17.8%	9.7%	8.9%	7.7%
	WIL2-100%	8.0%	14.3%	8.3%	8.7%
	WIL3-100%	1.0%	2.8%	0.3%	0.6%
FP buffer	OLYM-FP	3.3%	14.2%	1.4%	18.3%
	WIL1-FP	8.4%	34.6%	34.4%	14.5%
	CASC-FP	0.0%	1.8%	2.8%	1.0%
0% buffer Pre-harvest only	OLYM-0%	1.7%			
	WIL1-0%	20.2%			
	WIL2-0%	6.5%			
	CASC-0%	0.0%			

*Vegetation surveys in WIL2-FP were done before harvest in 2016 and so represent an unharvested condition.



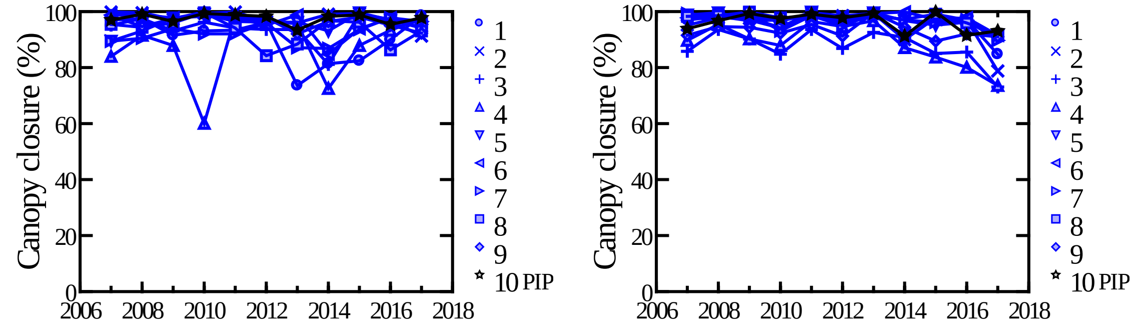
a) OLYM-REF

b) WIL1-REF



c) WIL2-REF

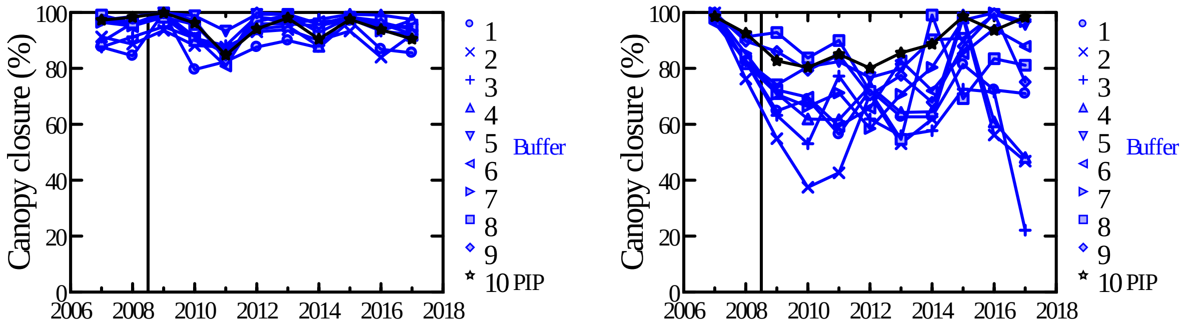
d) WIL2-FP



e) WIL3-REF

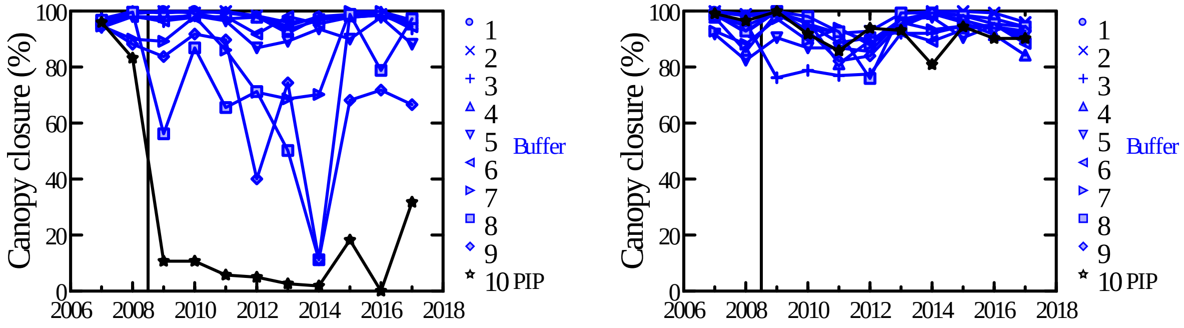
f) CASC-REF

Figure 4-3. Canopy closure at individual locations within the reference (REF) sites with perennial initiation point (PIP) locations in black. Vertical solid line separates pre- and post-harvest. Measurements were made at ten equidistant locations along the main channel between the Type F/N break (location 1) to the PIP (location 10). All measurements at WIL2-FP were taken prior to harvest in 2016 and therefore are included with the unharvested sites.



a) OLYM-100%

b) WIL1-100%



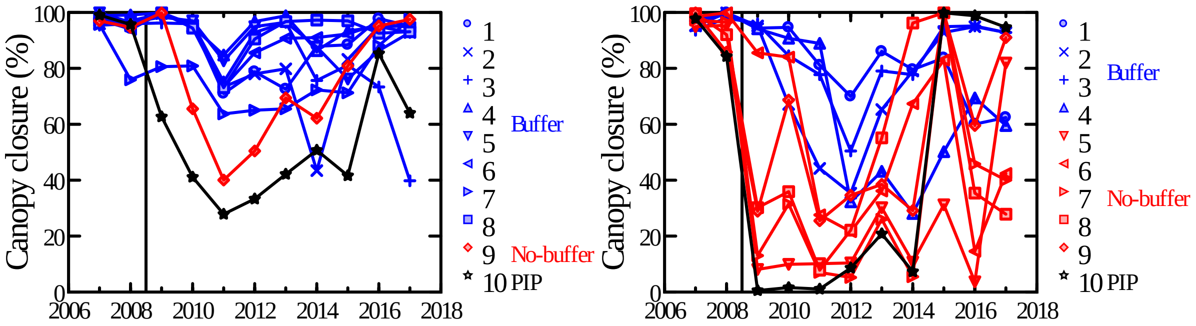
c) WIL2-100%

d) WIL3-100%

Figure 4-4. Canopy closure at individual locations within the 100% treatment sites with perennial initiation point (PIP) locations in black. Vertical solid line separates pre- and post-harvest. Measurements were made at ten equidistant locations along the main channel between the Type F/N break (location 1) to the PIP (location 10).

a) OLYM-FP

b) WIL1-FP



c) CASC-FP

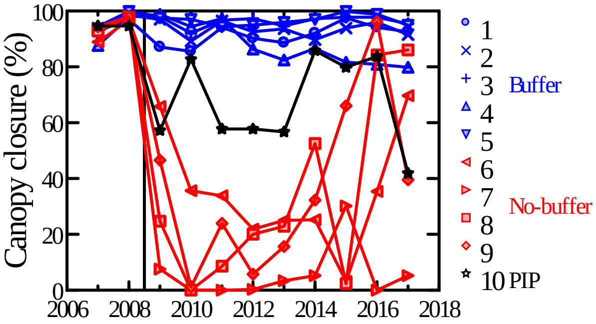
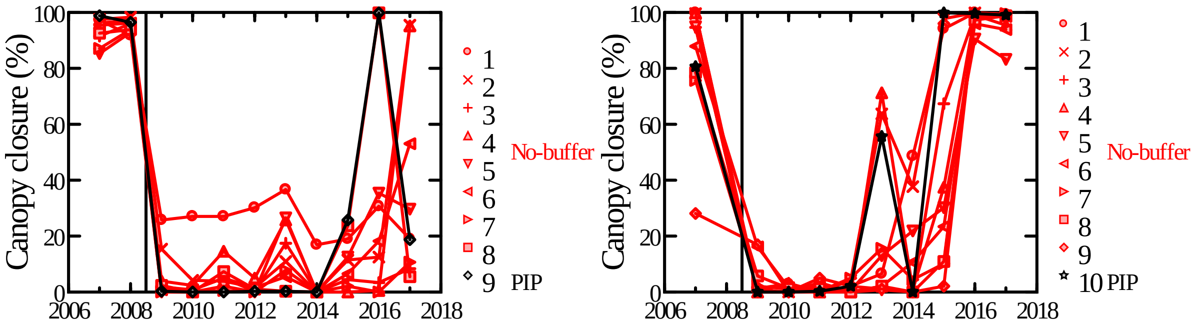


Figure 4-5. Canopy closure at individual locations within the FP treatment sites at buffered (blue), unbuffered (red), and perennial initiation point (PIP; black) locations. Vertical solid line separates pre- and post-harvest. Measurements were made at ten equidistant locations along the main channel between the Type F/N break (location 1) to the PIP (location 10).

a) OLYM-0%

b) WIL1-0%



c) WIL2-0%

d) CASC-0%

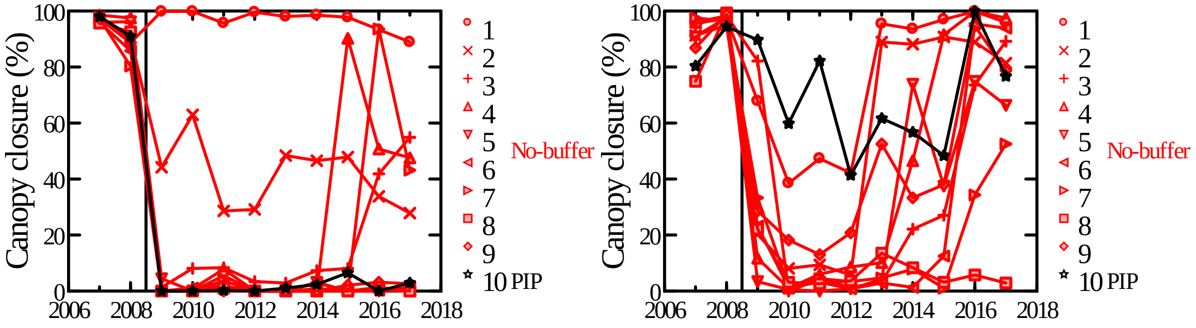


Figure 4-6. Canopy closure at individual locations within the 0% treatment sites with perennial initiation point (PIP) locations in black. Vertical solid line separates pre- and post-harvest. Measurements were made at ten equidistant locations along the main channel between the Type F/N break (location 1) to the PIP (location 10).

4-5.1.4. Stream Cover vs. Buffer Width and Season

Overlapping unstable slope and sensitive site buffers on top of prescribed 50-ft wide riparian buffers resulted in much wider buffers along portions of the OLYM-100% and WIL2-100% sites. We categorized all canopy closure locations based on the buffer width—greater than 50 ft wide, 50 ft wide, 56 ft diameter circle at the PIP, and no buffer—then plotted them over time to illustrate the effect of these unstable slope buffers on shade within the 100% treatment. Mean pre-harvest CC-1m values exceeded 90% for all buffer types prior to harvest reflecting the uniformity of canopy closure across and within sites (**Figure 4-7**). After harvest, mean CC-1m in buffers greater than 50 ft changed little, exceeding 80% in all years. Mean CC-1m in the 50-ft buffers decreased for four years after harvest to 77% then increased to more than 80% by Post 9. PIP buffers decreased to 40% by Post 3 due to windthrow, but increased to approximately 70% by Post 9. The unbuffered locations declined from 95% pre-harvest to a median of 5% in Post 2, but reached 58% by Post 9.

Both CC-1m and CC-0m were lower during leaf-off than in the leaf-on period with the greatest differences in the FP and 0% treatments (**Table 4-10**). Mean leaf-off CC-1m was 7 percentage points lower in the REF and 100% treatments, 20 points lower in the FP treatment, and 40 points lower in the 0% treatment. Similarly, differences in CC-0m were 8, 7, 23, and 36 percentage points lower in the REF, 100%, FP, and 0% treatments, respectively. All sites were conifer-dominated pre-harvest, so the dominant source of shade after leaf-fall was the remaining conifers in the REF and 100%, regenerating conifers and branches of regenerating deciduous vegetation in the 0%, and a combination of the remaining conifers, regenerating conifers, and deciduous branches in the FP.

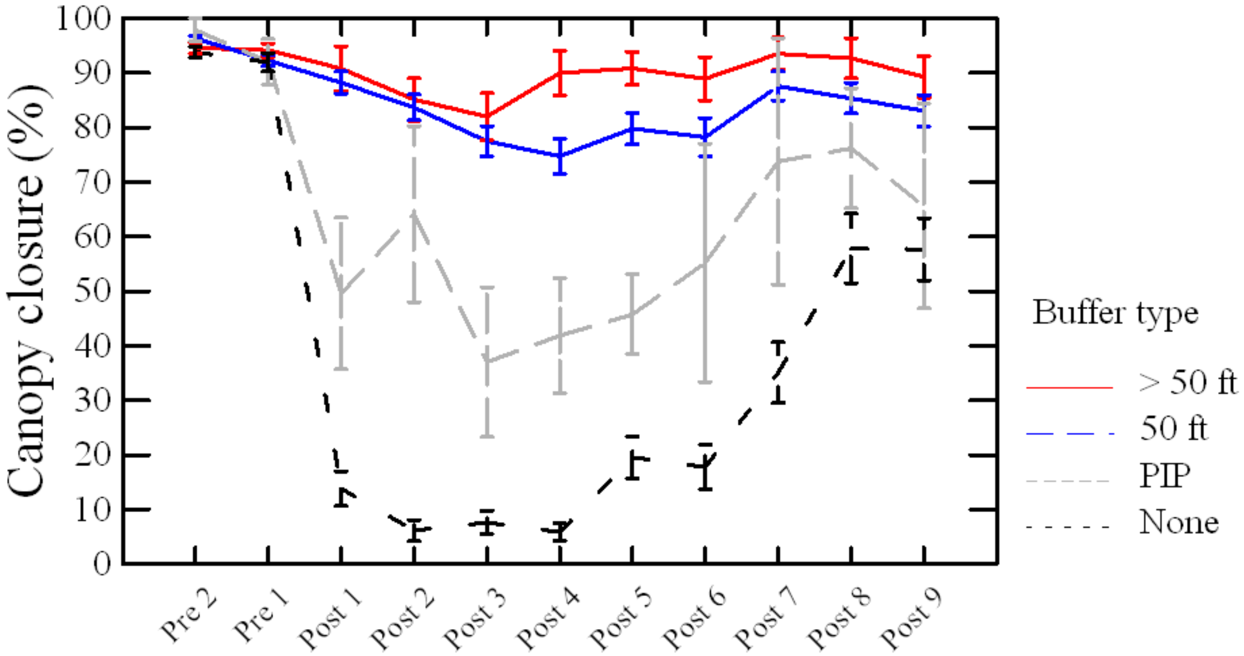


Figure 4-7. Canopy closure (%) at 1-m with standard errors plotted by buffer category over time. Red = greater than 50 ft wide (n = 78); blue = 50 ft wide (n = 43); gray = 56 ft diameter perennial initiation point (PIP; n = 3); black = no buffer (n = 45).

Table 4-10. Mean canopy closure (%) values for leaf-on (summer 2017) and leaf off (November 2017 to March 2018) periods.

	CC-1m			CC-0m		
Treatment	**Leaf-on**	**Leaf-off**	**Difference**	**Leaf-on**	**Leaf-off**	**Difference**
REF	85	78	-7	89	81	-8
100%	82	75	-7	88	81	-7
FP	71	50	-20	84	62	-23
0%	56	16	-40	73	37	-36

4-5.2. STREAM TEMPERATURE

The study streams were cool with pre-harvest maximum 7-day average daily maximum water temperature (7DADM) ranging from 9.2 to 16.1°C compared to 7.2 to 23.4°C in a random sample of Type N streams (Washington State Department of Ecology 2019). Maximum daily temperatures are plotted in **Figures 4-8** through **4-11**. **Table 4-11** shows the maximum July–August 7DADM, number of days available for the calculation, and the difference between each post-harvest year and the mean pre-harvest 7DADM. The interannual variability in the REF sites' 7DADM illustrates the value of controlled, experimental studies. The within treatment mean post–pre-harvest difference in the REF treatment never exceeded 1.0°C. In contrast, mean within treatment difference in the 100% was 2.4°C in 2009 (Post 1) but never exceeded 1.0°C in later years. The mean difference in the FP treatment exceeded 1.0°C immediately after harvest then again in 2014–2016 (Post 6–9) while in the 0% treatment the mean difference was 5.3°C initially, then decreased over time to near, but never below, 0.9°C.

4-5.2.1. Stationarity of the Reference Sites

WIL2-FP, prior to harvest, was stationary for the first six years post-calibration. The median TR value across the first six post-calibration years was 0.01°C, 90% of the TR values were between −0.53°C and 0.62°C, and only five of 210 post-calibration 1–6 MMTR values were greater than 0.5°C in absolute value with $P < 0.05$, ranging from −0.7–0.8°C (**Table 4-12**). The daily TR values began drifting higher in 2014, post-calibration 7, when nearly all MMTR values exceeded 0.5°C (**Figure 4-12**).

The OLYM-REF site remained stationary throughout the course of the study. Median post-calibration daily TR was -0.01°C and 90% of the TR values were between -0.72°C and 0.68°C. There were 18 of 196 post-calibration MMTR values greater than 0.5°C in absolute values, six negative, and 12 positive. There may be minimal drift beginning around 2015, post-calibration 9, as shown by elevated MMTR values for January through March of that year. However, MMTRs are 0.4°C or less for other months that year and within the range observed for that month in earlier post-calibration years.

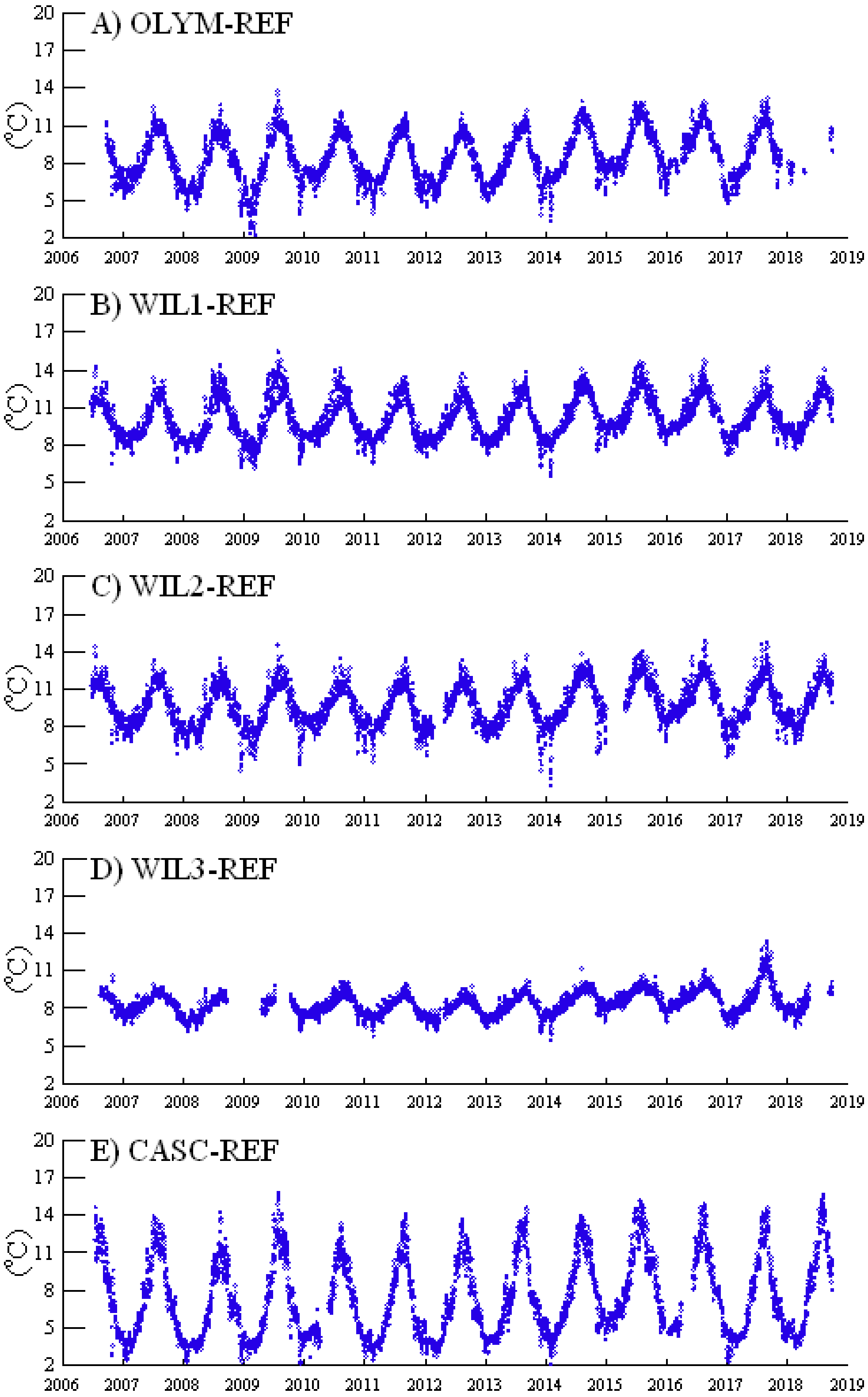


Figure 4-8. Maximum daily stream temperature over time in the reference (REF) sites.

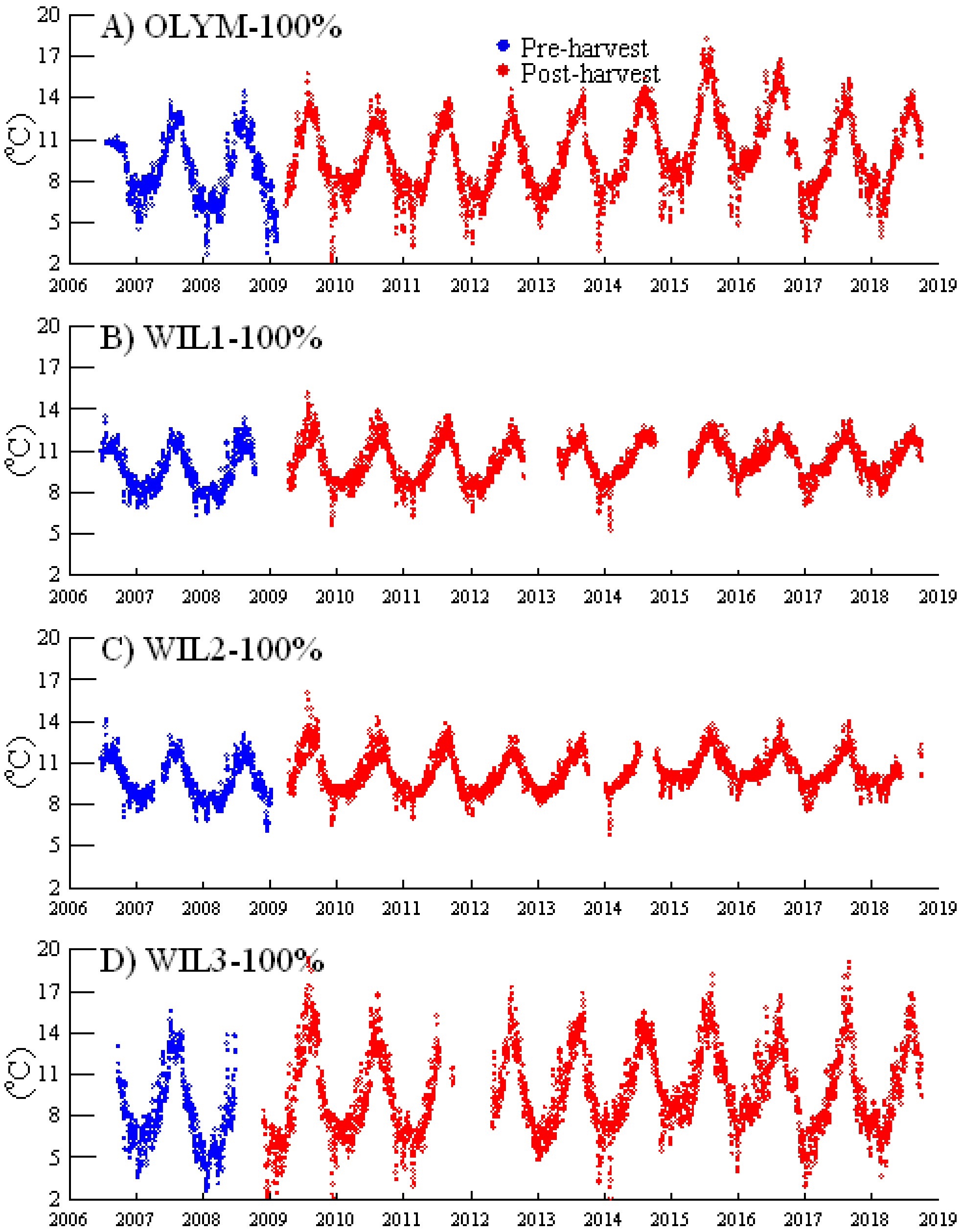


Figure 4-9. Maximum daily stream temperature over time in the 100% treatment sites.

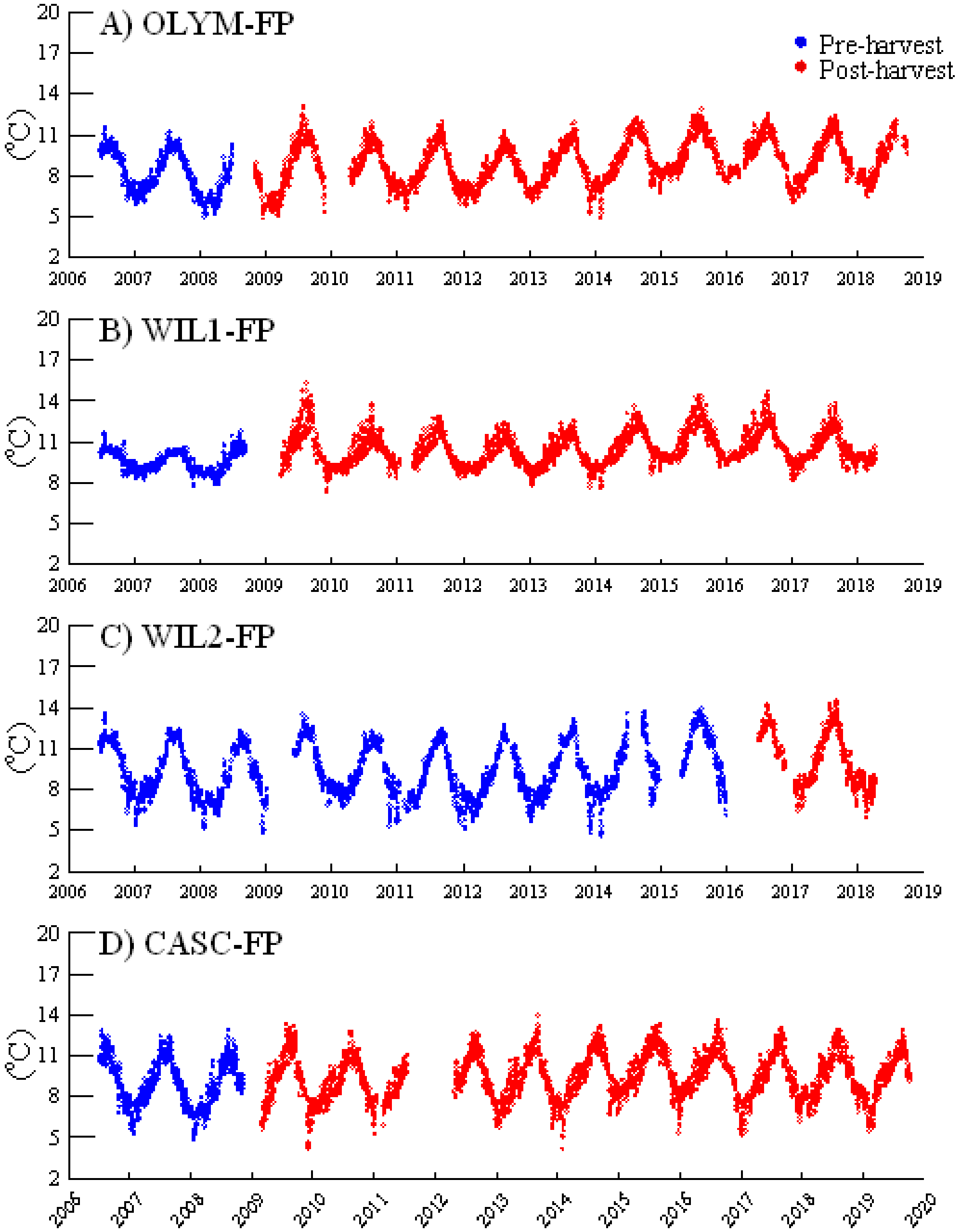


Figure 4-10. Maximum daily stream temperature over time in the FP treatment sites.

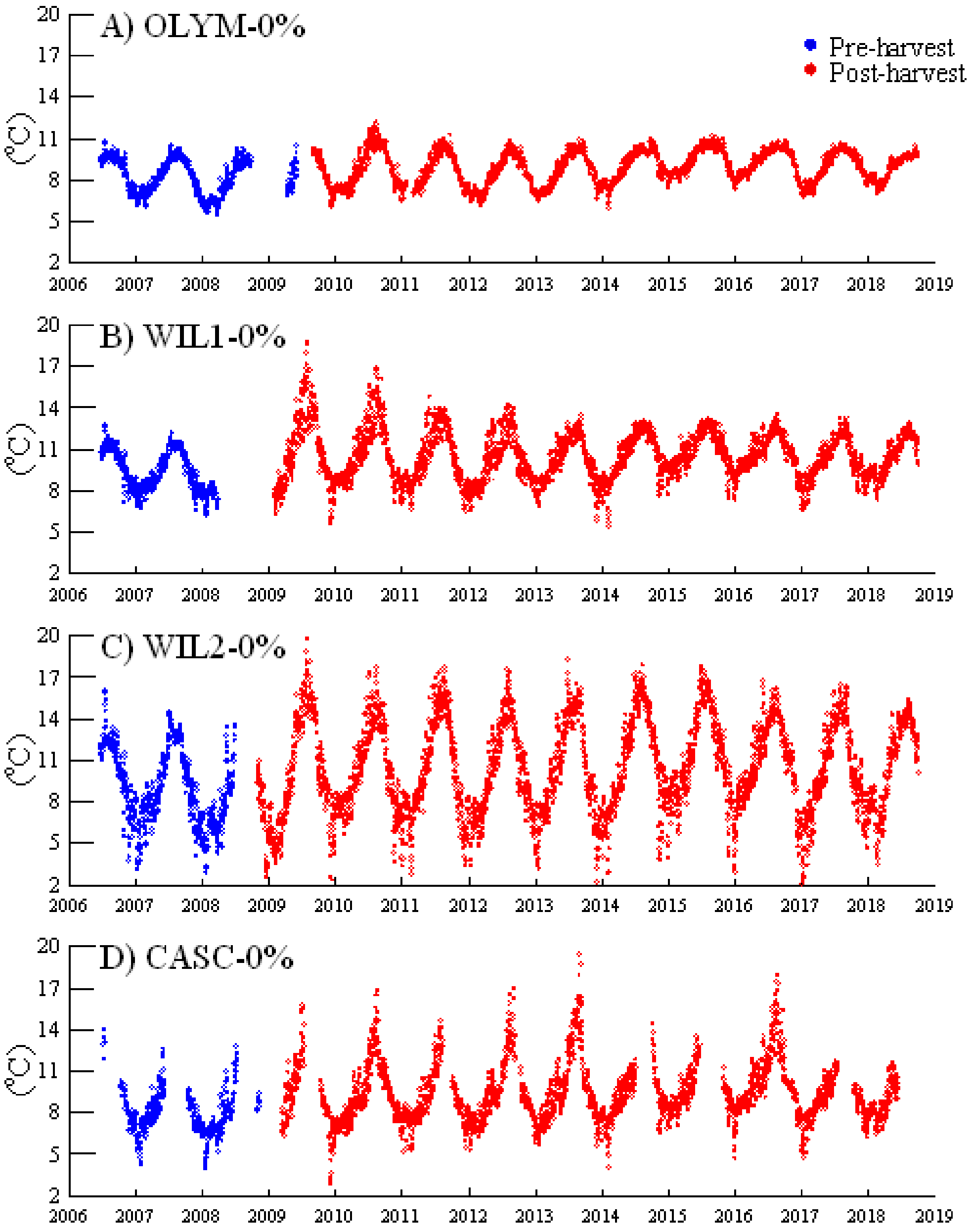


Figure 4-11. Maximum daily stream temperature over time in the 0% treatment sites.

Table 4-11. Maximum 7-day average daily maximum (7DADM) temperature (°C) for July–August for each site and year, number of observations (N), and difference (Diff) between post-harvest 7DADM and the mean pre-harvest 7DADM. Shading indicates post-harvest period.

Treatment	Block	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
REF	OLYM		11.7	11.8	12.8	11.5	11.3	11.3	11.0	12.0	12.4	12.3	12.2
	N		62	62	62	62	62	62	62	62	62	62	62
	Diff				1.0	-0.3	-0.5	-0.4	-0.7	0.3	0.7	0.5	0.5
	WIL1	13.2	12.4	13.2	14.6	13.4	12.8	12.8	12.4	13.2	13.8	13.6	12.9
	N	62	62	62	62	62	62	62	62	62	62	62	62
	Diff				1.7	0.4	-0.1	-0.2	-0.6	0.2	0.8	0.6	-0.1
	WIL2	13.3	12.4	12.5	13.4	12.3	12.3	12.3	12.2	12.7	13.2	13.6	13.3
	N	62	62	62	62	62	62	62	62	62	62	62	62
	Diff				0.6	-0.4	-0.4	-0.4	-0.5	-0.1	0.5	0.8	0.6
	WIL3	9.2	9.5	9.2	8.9	9.5	9.3	9.0	9.3	9.8	10.0	10.5	12.0
	N	4	62	62	3	62	62	62	62	62	62	62	62
	Diff				-0.4	0.2	0.1	-0.3	0.0	0.5	0.7	1.2	2.7
	CASC	13.9	13.5	13.0	15.3	12.4	12.9	13.1	13.3	13.4	14.4	14.3	13.7
	N	37	62	62	62	62	62	62	62	62	62	62	62
	Diff				1.8	-1.0	-0.6	-0.4	-0.2	0.0	0.9	0.9	0.3
Mean w/in treatment diff					**1.0**	**-0.2**	**-0.3**	**-0.3**	**-0.4**	**0.2**	**0.7**	**0.8**	**0.8**
100%	OLYM	14.9	13.4	13.4	15.0	13.7	11.8	13.7	13.9	14.4	14.5	14.3	14.7
	N	44	62	62	62	62	23	62	62	62	62	62	62
	Diff				1.2	-0.2	-2.1	-0.2	0.0	0.6	0.6	0.5	0.8
	WIL1	12.7	12.0	12.4	14.3	13.2	13.0	12.6	11.9	12.1	12.6	12.4	12.5
	N	62	62	62	62	62	62	62	62	62	62	62	62
	Diff				1.9	0.9	0.6	0.2	-0.5	-0.2	0.2	0.1	0.2
	WIL2	13.0	12.1	12.3	14.3	13.3	12.8	12.2	12.2	12.2	13.0	13.0	12.6
	N	62	62	62	62	62	62	62	62	21	62	62	62
	Diff				1.8	0.8	0.3	-0.3	-0.3	-0.3	0.5	0.5	0.1
	WIL3		14.6	15.5	19.6	16.0	13.5	16.0	14.4	14.7	16.1	15.3	16.7
	N		62	62	62	62	20	62	62	62	62	62	62
	Diff				4.6	1.0	-1.6	1.0	-0.6	-0.3	1.0	0.3	1.7
Mean w/in treatment diff					**2.4**	**0.6**	**-0.7**	**0.2**	**-0.3**	**-0.1**	**0.6**	**0.3**	**0.7**
FP	OLYM	11.1	10.5	10.9	12.4	11.2	11.2	10.8	10.8	11.8	12.3	12.0	11.8
	N	62	62	62	62	62	62	62	62	62	62	62	62
	Diff				1.6	0.3	0.4	0.0	0.0	0.9	1.4	1.1	1.0
	WIL1	11.2	10.3	11.2	14.1	12.7	12.5	11.9	11.8	12.7	13.7	13.6	12.8
	N	62	62	62	62	62	62	52	62	62	62	62	62
	Diff				3.2	1.8	1.6	1.0	0.9	1.8	2.8	2.7	1.9
	WIL2	13.0	12.2	12.0	13.1	12.0	12.1	12.2	12.2	12.5	13.4	13.5	13.8
	N	62	62	62	62	62	62	62	62	3	62	62	62
	Diff				0.6	-0.5	-0.4	-0.3	-0.2	0.0	0.9	1.1	1.3
	CASC	12.2	11.7	12.1	12.7	12.1	10.3	12.0	12.0	12.2	12.5	12.9	12.1
	N	44	62	62	62	62	21	62	62	62	62	62	62
	Diff				0.7	0.1	-1.7	0.0	0.0	0.2	0.5	0.9	0.1
Mean w/in treatment diff					**1.8**	**0.8**	**0.1**	**0.3**	**0.3**	**1.0**	**1.6**	**1.4**	**1.1**
0%	OLYM	10.4	9.9	9.8		11.6	10.7	10.6	10.4	10.4	10.6	10.5	10.4
	N	62	62	62		62	62	62	62	62	62	62	62
	Diff					1.6	0.7	0.5	0.4	0.4	0.6	0.5	0.4
	WIL1	12.0	11.5	11.7	17.5	15.8	13.6	13.4	12.4	12.8	12.9	12.8	12.6
	N	62	62	62	62	62	62	62	62	62	62	62	62
	Diff				5.8	4.1	1.8	1.7	0.7	1.0	1.2	1.0	0.8
	WIL2	14.1	13.3		18.6	15.2	14.9	14.7	14.4	14.6	14.7	14.8	15.0
	N	62	62		62	62	62	62	62	62	13	62	62
	Diff				4.9	1.5	1.2	1.0	0.7	0.9	1.0	1.0	1.3
	CASC	15.1	15.0	16.1		19.5	17.1	19.2	17.4	18.0	18.4	17.4	17.4
	N	39	62	62		62	21	62	62	62	62	62	62
	Diff					4.1	1.7	3.8	2.0	2.5	3.0	2.0	1.9
Mean w/in treatment diff					**5.3**	**2.8**	**1.3**	**1.8**	**0.9**	**1.2**	**1.5**	**1.1**	**1.1**

Table 4-12. Mean monthly temperature response (MMTR) values for reference sites listed by location and calibration year. Blue shading = MMTR <-0.5°C and P <0.05. Brown shading = MMTR >0.5°C and P <0.05.

Site	Location	Calib	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL2-REF	T4	Pre 2	0.0	0.1	0.1	-0.1	0.1	-0.3	-0.1	-0.1	0.2	0.3	0.1	0.0
		Pre 1	-0.2	-0.3	-0.2	-0.2	-0.2	-0.2	0.0	0.0	-0.1	0.1	0.1	-0.1
		Post 1	-0.2	-0.2	-0.1	-0.3	-0.2	-0.2	0.0	0.0	-0.1	0.3	0.0	-0.1
		Post 2	0.0	0.1	0.2	0.1	0.1	0.1	-0.2	-0.2	-0.3	0.0	0.1	0.1
		Post 3	0.1	0.0	0.0	0.0	-0.1	-0.1	0.0	0.0	-0.1	-0.3	0.1	0.0
		Post 4	-0.1	-0.1	-0.2	-0.1	-0.1	0.0	0.0	0.1	0.1	0.1	0.1	0.0
		Post 5	-0.1	-0.1	-0.1	0.0	0.0	0.0	-0.1	-0.2	-0.1	0.0	0.1	0.0
		Post 6	0.4	0.4	0.3	0.3	0.2	0.1	0.1	0.1	1.2	0.4	0.4	0.5
		Post 7	0.7	0.7	0.7	0.6	0.5	0.5	0.5	0.3	0.6	0.9	0.7	0.7
	T3	Pre 2	0.0	0.4	0.2	0.0	0.0	0.0	0.0	0.2	0.6	0.1	-0.2	-0.1
		Pre 1	0.0	0.0	-0.1	-0.3	-0.3	-0.5	0.3	0.3	0.1	0.0	0.1	0.1
		Post 1	-0.1	-0.2	-0.1	-0.2	-0.3	-0.2	-0.4	0.1	0.1	-0.1	-0.1	0.0
		Post 2	0.1	0.1	0.1	0.0	-0.1	-0.3	0.4	0.4	0.4	-0.2	-0.2	-0.1
		Post 3	0.2	0.2	0.2	0.0	-0.1	-0.2	-0.2	0.1	0.0	-0.1	0.0	0.1
		Post 4	0.2	0.2	0.1	0.0	-0.1	-0.3	-0.3	-0.1	0.2	-0.1	0.2	0.1
		Post 5	0.1	0.2	0.1	0.0	0.0	-0.2	-0.3	0.1	0.3	0.1	0.0	0.0
		Post 6	0.4	0.6	0.4	0.3	0.1	0.1	0.2	0.3	0.1	0.1	0.2	0.3
		Post 7	0.7	0.7	0.6	0.5	0.5	0.5	0.6	0.8	0.9	0.5	0.4	0.6
	T1	Pre 2	0.3	0.3	0.1	0.0	0.0	0.0	0.2	0.2	0.1	0.1	-0.4	0.2
		Pre 1	-0.2	0.0	0.0	-0.1	-0.4	-0.4	0.2	0.1	0.1	-0.2	-0.2	-0.1
		Post 1	0.0					0.2	-0.2	0.0	-0.1	-0.3	-0.3	-0.2
		Post 2	0.3	0.1	0.2	0.1	-0.1	0.1	0.1	0.3	0.3	0.2	0.2	0.3
		Post 3	-0.2	0.3	0.0	-0.4	-0.7	-0.4	-0.1	-0.1	0.0	-0.2	-0.2	-0.1
		Post 4	-0.2	0.1			-0.6	-0.7	-0.2	-0.1	-0.3	-0.3	-0.4	-0.3
		Post 5	-0.1	0.4	0.3	0.3	-0.1	-0.1	0.0	-0.1	0.0	-0.2	-0.2	-0.1
		Post 6	0.5	0.1	0.2	0.3	0.4	0.8	0.4	0.3	0.1	-0.2	-0.1	0.5
		Post 7				0.5	0.5	0.5	1.1		0.8	1.0	1.2	0.5
OLYM-REF	T2	Pre 2	0.2	0.4	0.5	0.2	0.2	0.1	0.2		0.3	0.0	0.0	0.1
		Pre 1	-0.3	-0.3	-0.2	-0.4	-0.1	-0.2	0.1	0.0	0.1	0.0	0.0	0.0
		Post 1	-0.6	-1.2	-0.5	-0.5	-0.2	0.1	0.4	0.4	-0.3	-0.1	-0.1	-0.2
		Post 2	0.6	0.6	0.3	0.0	-0.1	-0.3			0.0	-0.1	0.0	-0.2
		Post 3	0.1	-0.1	-0.3	-0.4	-0.5	-0.4	-0.1	-0.2	-0.1	0.0	-0.3	0.1
		Post 4	-0.1	0.1	-0.5	-0.3	-0.5	-0.8	-0.2	-0.1	-0.2	-0.3	-0.3	-0.2
		Post 5	-0.3	0.0	0.0	0.0	-0.2	-0.4	-0.4	-0.3	-0.4	-0.4	-0.2	-0.4
		Post 6	0.3	0.4	0.2	0.1	0.0	-0.2	0.0	-0.1	0.0	-0.2	0.0	-0.1
		Post 7	0.5	0.7	0.4	0.2	0.4	0.4	0.3	0.4	0.4	0.1	0.2	0.3
		Post 8	0.5	0.8	0.6	0.4	0.3	0.4	0.3	0.4	0.2	0.0	0.1	0.4
	T1	Pre 2	0.2	0.4	0.4	0.2	0.2	0.1	0.2		0.2	0.0	0.1	0.1
		Pre 1	-0.4	-0.3	-0.2	-0.4	-0.1	-0.2	0.0	0.0	0.1	0.0	0.0	-0.1
		Post 1	-0.6	-1.1	-0.5	-0.5	-0.1	0.1	0.3	0.4	-0.3	-0.1	-0.1	-0.1
		Post 2	0.6	0.5	0.2	0.0	-0.2	-0.3			0.0	-0.1	0.0	-0.1
		Post 3	0.2	0.0	-0.2	-0.4	-0.5	-0.4	0.0	-0.1	-0.2	-0.1	-0.2	0.2
		Post 4	0.0	0.3			-0.4	-0.7	-0.2	0.0	-0.1	-0.2	-0.2	-0.1
		Post 5	-0.2	0.0	0.1	0.0	-0.2	-0.4	-0.4	-0.3	-0.3	-0.3	-0.1	-0.3
		Post 6	0.4	0.5	0.2	0.1	0.0	-0.2	-0.1	-0.2	0.0	-0.2	0.1	-0.1
		Post 7	0.7			0.0	0.4	0.4	0.3	0.4	0.3	0.1	0.3	0.4
		Post 8	0.1	0.2	0.0	0.2	0.0	0.1	-0.1	0.0	0.0	0.1	0.2	-0.2
		Post 9	0.6	0.8	0.6	0.4	0.3	0.4	0.3	0.4	0.2	0.0	0.2	0.4

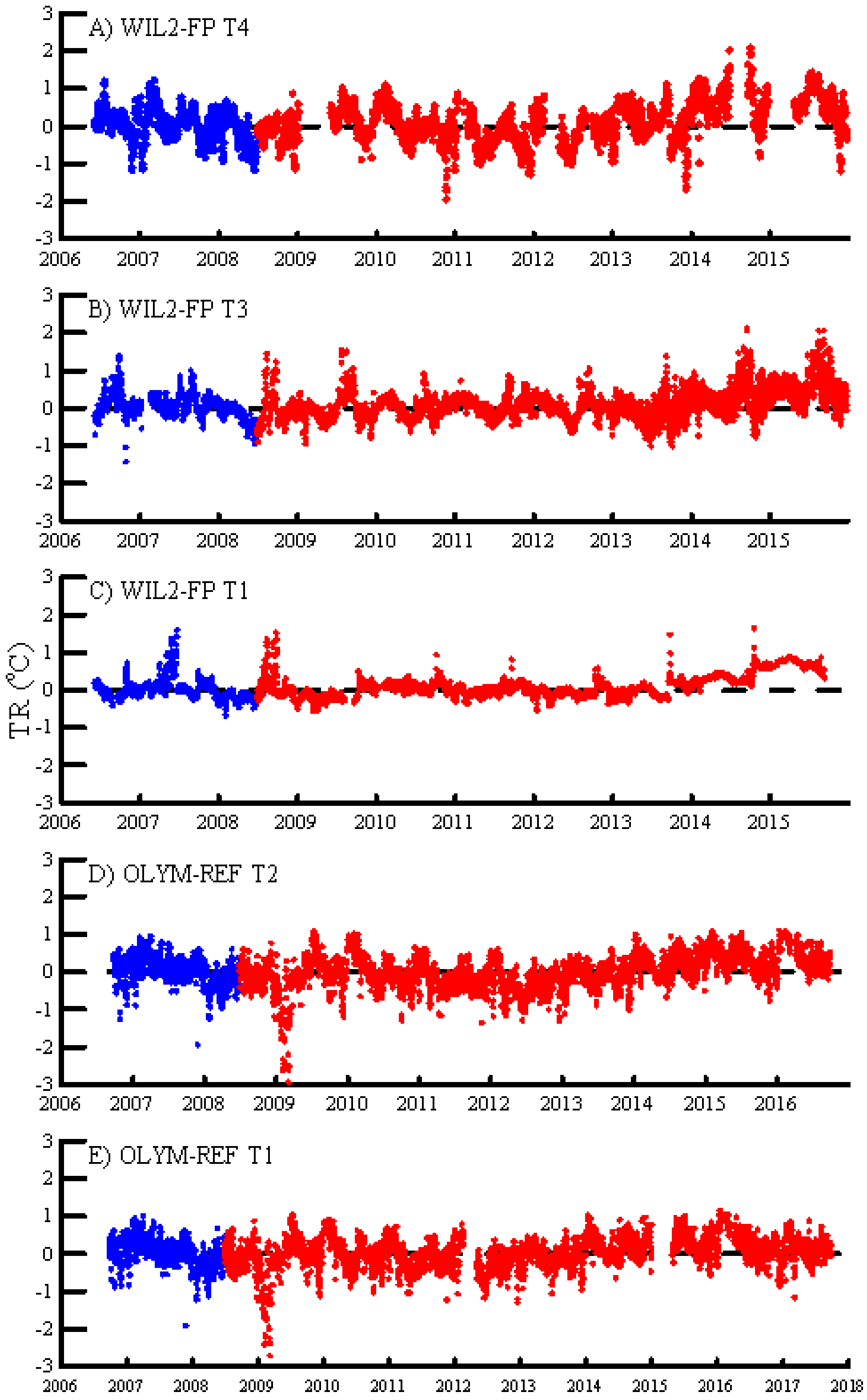


Figure 4-12. Daily temperature response (TR) plotted over time for monitoring locations in WIL2-FP (prior to harvest) and OLYM-REF. Blue = calibration period; Red = post calibration.

4-5.2.2. Seasonal and Temporal Temperature Response by Site

Stream temperature increased post-harvest at most locations within all 12 harvested sites and remained elevated in the FP and 0% treatments over much of the nine years post-harvest. **Appendix Tables 4-5** through **4-16** tabulate the MMTRs for all locations in all sites across all years sorted by treatment, block, location within each site, and treatment year. Significant (P <0.05) MMTRs are shaded blue, if negative and less than −0.5°C, or color-coded (pink through brown) by magnitude of change if positive and greater than 0.5°C. There were 6,297 MMTRs calculated cumulatively across all locations in the harvested sites and post-harvest years. Of these, 98.5% were positive. Fifty-three percent were positive, greater than 0.5°C, and P <0.05, while only 0.8% of the total were less than −0.5°C and P <0.05. Of the 3,337 significant responses greater than 0.5°C, 34% exceeded 1°C, 7% exceeded 2°C, and 2% exceeded 3°C. In comparison, there were 438 MMTRs calculated for the post-calibration years in all of the REF-REF comparisons. Of these 45.9% were positive and 54.1% were negative. Only 7.8% had P <0.05 and exceeded 0.5°C and 2.1% were less than −0.5°C and significant.

Temperature responses varied by treatment, by season, and over the years. We use examples from the buffer treatment locations to illustrate the seasonal responses for spring (April, May), summer (July, August) and fall (October, November) below.

Summer MMTR values in the 100% treatment showed an initial increase immediately after harvest followed by a steady decrease toward pre-harvest conditions over time (**Figure 4-13**). Summer MMTR was elevated at all four 100% treatment sites for the first few years after harvest, but decreased to near zero by Post 3 to Post 6 at all but the OLYM-100% site. Summer MMTR was usually less than 1.0°C except at WIL3-100%, where the July MMTR was 2.3°C in Post 1. Summer MMTR remained elevated at Post 9 at only the OLYM-100% site, but was less than 1.0°C.

Summer MMTR response varied among the four FP sites. MMTR values were very similar at the OLYM-FP and WIL1-FP sites where MMTR was elevated and changed little over the post-harvest years (**Figure 4-14**). Estimates of MMTR at OLYM-FP and WIL1-FP were 0.5 and 1.8°C, respectively, in Post 1, decreased to a minimum of 0.1 and 0.5°C, respectively, at Post 4, then increased to near Post 1 values through Post 9. CASC-FP was unique among all harvested sites in that summer MMTR did not change (P >0.05) post-harvest. MMTR was consistently <0.5°C and P >0.05.

Summer MMTR in the 0% treatment sites was initially approximately 2.0°C higher than either the 100% or FP sites and remained elevated throughout most of the post-harvest period at all sites (**Figure 4-15**). Summer MMTRs were consistently higher than spring or fall but MMTRs were elevated most of the year at all but the CASC-0% site. Over the course of the study, the 0% sites' summer MMTR decreased consistently toward zero, similar to the 100% treatment. However, unlike the 100% sites, all of the 0% sites were still elevated (P >0.05) at Post 9.

Spring and fall MMTRs were elevated at all sites in all treatments over most of the post-harvest period but usually by less than 1.0°C.

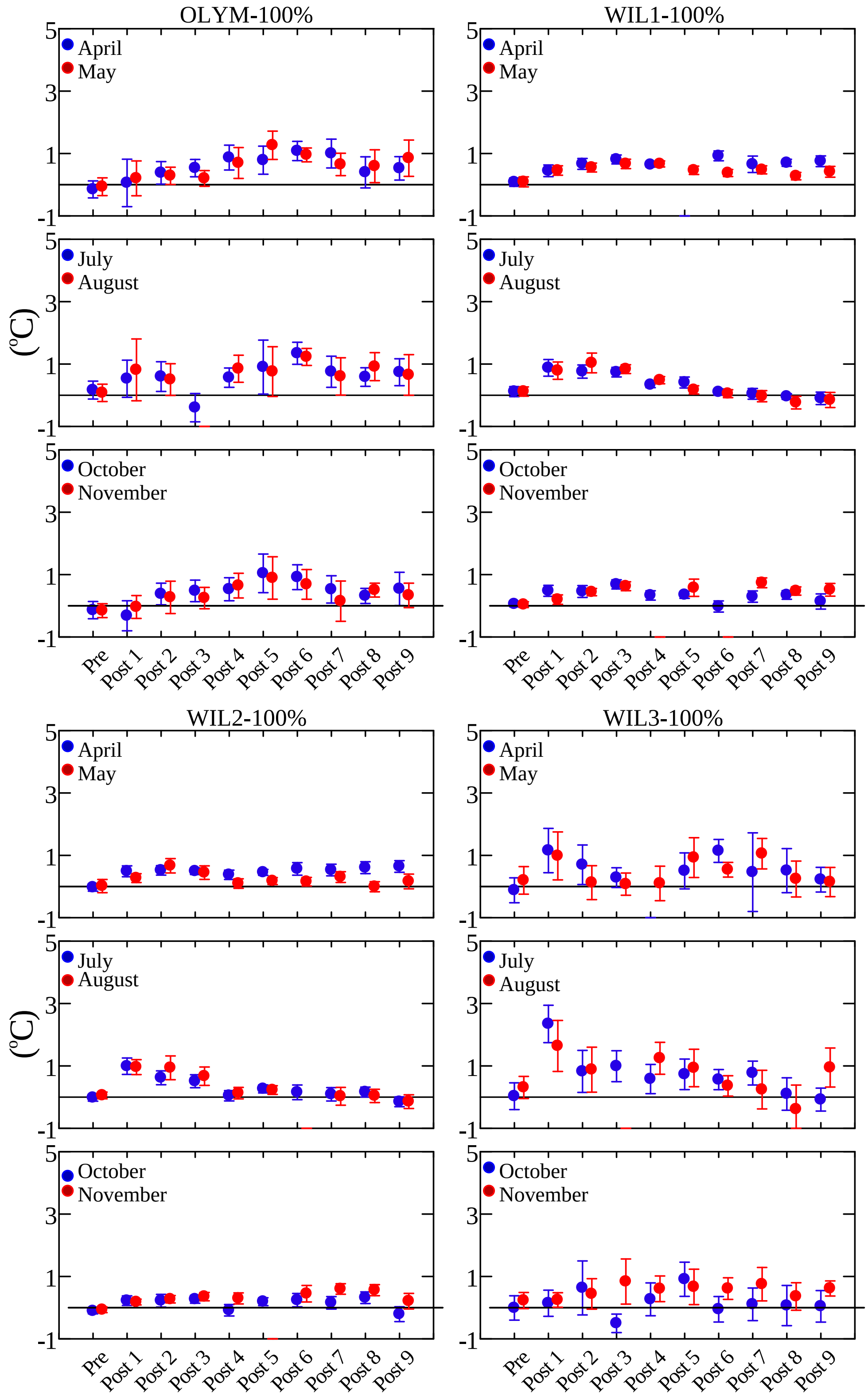


Figure 4-13. Mean monthly temperature response (MMTR) with 95% confidence intervals for spring, summer, and fall in the 100% treatment sites.

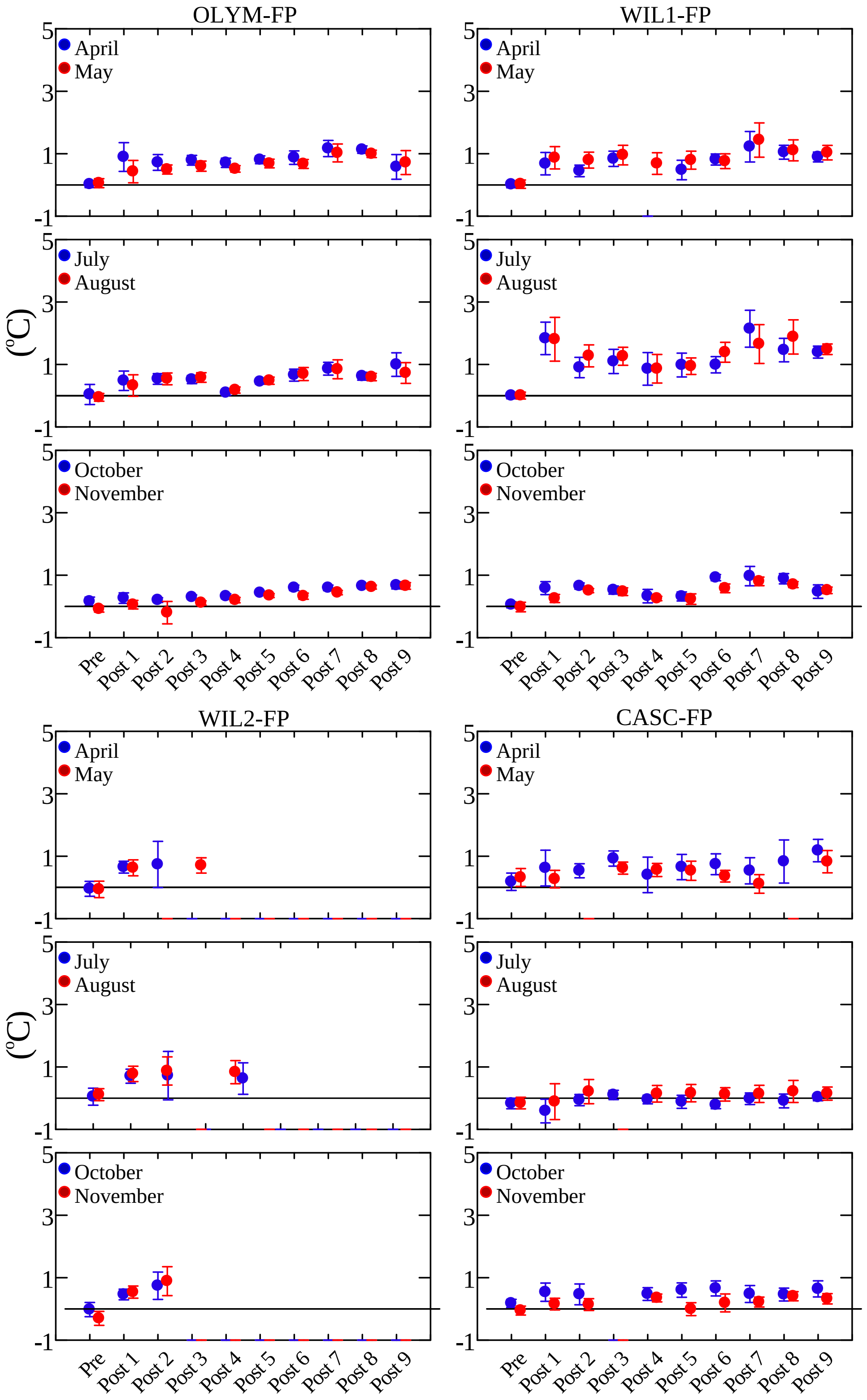


Figure 4-14. Mean monthly temperature response (MMTR) with 95% confidence intervals for spring, summer, and fall in the FP treatment sites. WIL2-FP had only one full year of post-harvest data.

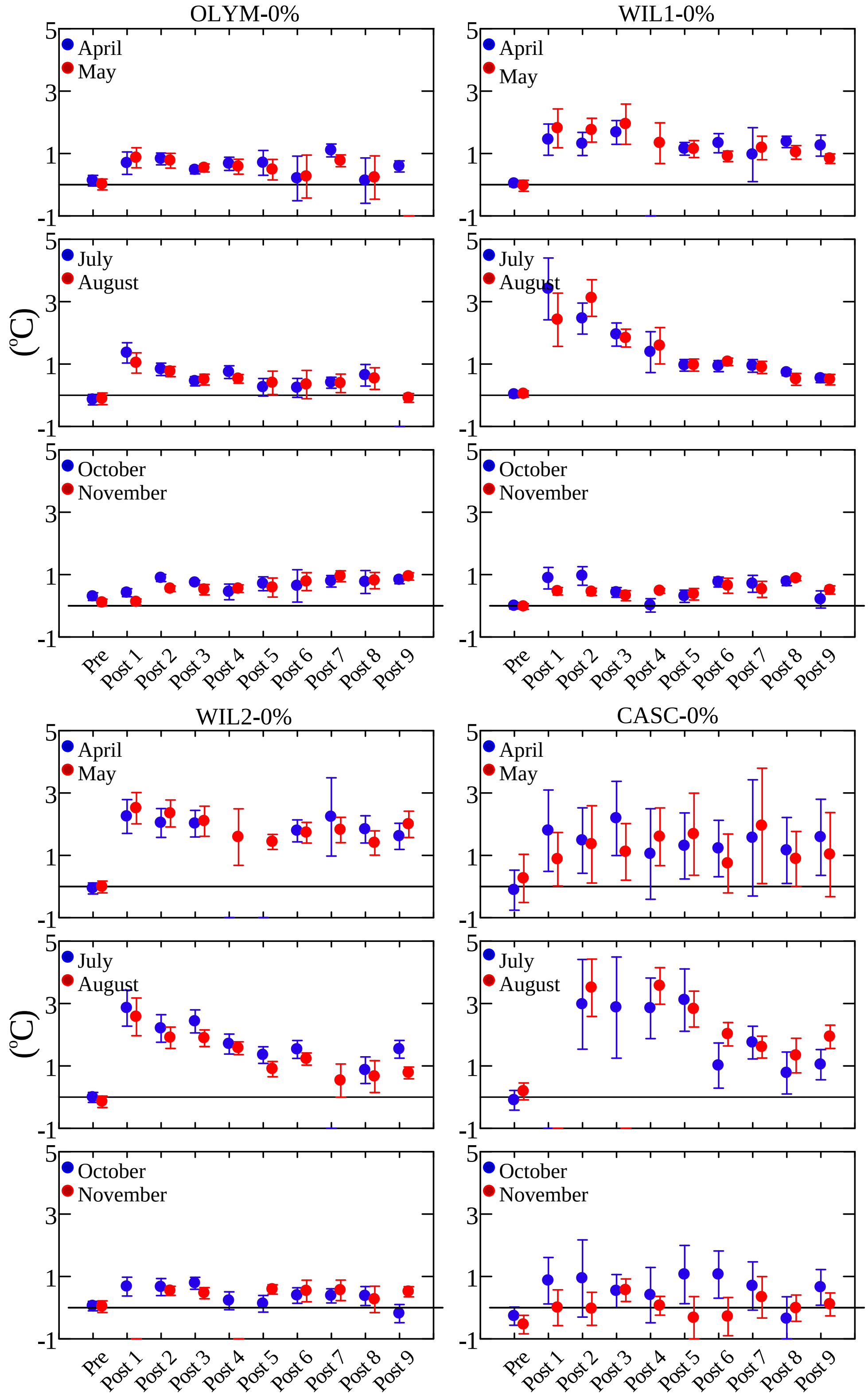


Figure 4-15. Mean monthly temperature response (MMTR) with 95% confidence intervals for spring, summer, and fall in the 0% treatment sites.

4-5.2.3. Relationship between Temperature, Riparian Cover, Buffer Width, Aspect, and Discharge

We evaluated the relationship between July MMTR and the post-harvest change in riparian cover across all sites for each post-harvest year (**Table 4-13**). In three out of the first four post-harvest years there was, at least, a weak (r <−0.48) negative correlation between July MMTR and the change in riparian cover based on each of the four shade metrics. The correlation was generally weaker (−0.4< r and P >0.10) after Post 4, except for Post 9 (−0.6< r <−0.4). However, there were only eight data pairs available for Post 9, compared to ten to twelve for the other years, which affected the correlation coefficient and p-value.

The correlation coefficient of July MMTR with both shade metrics across the post-harvest years varied greatly among sites and among treatments (**Table 4-14**; **Figures 4-17** through **4-19**). For example, the WIL2-100%, WIL1-0%, and CASC-0% had significant (P <0.05) negative Pearson correlations with at least one canopy closure metric. However, WIL1-FP had a significant positive correlation between MMTR and CC-1m. P-values for the correlations at all other sites were >0.05.

In the first few years after harvest median July MMTR tended to be lowest at locations with wider (>50 ft) buffers, slightly higher in 50 ft buffers, and highest in unbuffered locations (**Figure 4-16**). This pattern held through Post 4, after which there was considerable overlap among the three buffer widths. This overlap reflected a combination of much lower MMTRs at the unbuffered locations and only a slight decrease at the buffered locations.

Table 4-13. Pearson correlation coefficients and p-values between July mean monthly temperature response (MMTR) and the post-harvest change in riparian cover, as measured by the four metrics. The analysis was done separately for each post-harvest year. P-values were not corrected for multiple comparisons. Number of samples for CC-1m and CC-0m are the same as are n-values for effective shade and CTD. CC-1m = canopy closure at 1m; CC-0m = canopy closure at the water surface; CTD = canopy and topographic density.

Year		CC-1m		CC-0m			Effective Shade		CTD	
	n	r	P-value	r	P-value	n	r	P-value	r	P-value
Post 1	12	−0.487	0.108	−0.079	0.818	11	−0.439	0.176	−0.497	0.120
Post 2	12	−0.704	0.016	−0.589	0.073	11	−0.673	0.023	−0.682	0.021
Post 3	11	−0.538	0.088	−0.551	0.099	11	−0.528	0.095	−0.548	0.081
Post 4	12	−0.646	0.008	−0.794	0.006	11	−0.724	0.010	−0.753	0.008
Post 5	11	−0.374	0.258	−0.250	0.489	11	−0.363	0.273	−0.357	0.281
Post 6	10	−0.091	0.803	−0.239	0.536					
Post 7	11	−0.055	0.871	−0.210	0.416					
Post 8	11	−0.173	0.633	−0.119	0.761					
Post 9	8	−0.467	0.243	−0.579	0.176					

Table 4-14. Pearson correlation coefficients and p-values between July mean monthly temperature response (MMTR) and canopy closure for each site. Values of n apply to both CC-1m and CC-0m. CC-1m = canopy closure at 1m; CC-0m = canopy closure at the water surface.

		CC-1m		CC-0m	
Site	n	R	P-value	R	P-value
OLYM-100%	9	0.176	0.651	0.177	0.659
WIL1-100%	9	−0.487	0.184	−0.642	0.062
WIL2-100%	9	0.114	0.787	**−0.642**	**0.020**
WIL3-100%	9	0.220	0.569	0.140	0.719
OLYM-FP	9	0.210	0.587	0.147	0.704
WIL1-FP	9	**0.713**	**0.017**	0.504	0.167
CASC-FP	9	−0.260	0.500	0.026	0.948
OLYM-0%	8	0.169	0.689	−0.391	0.338
WIL1-0%	9	−0.653	0.056	**−0.707**	**0.033**
WIL2-0%	8	−0.646	0.084	−0.200	0.653
CASC-0%	8	**−0.870**	**0.004**	**−0.711**	**0.048**

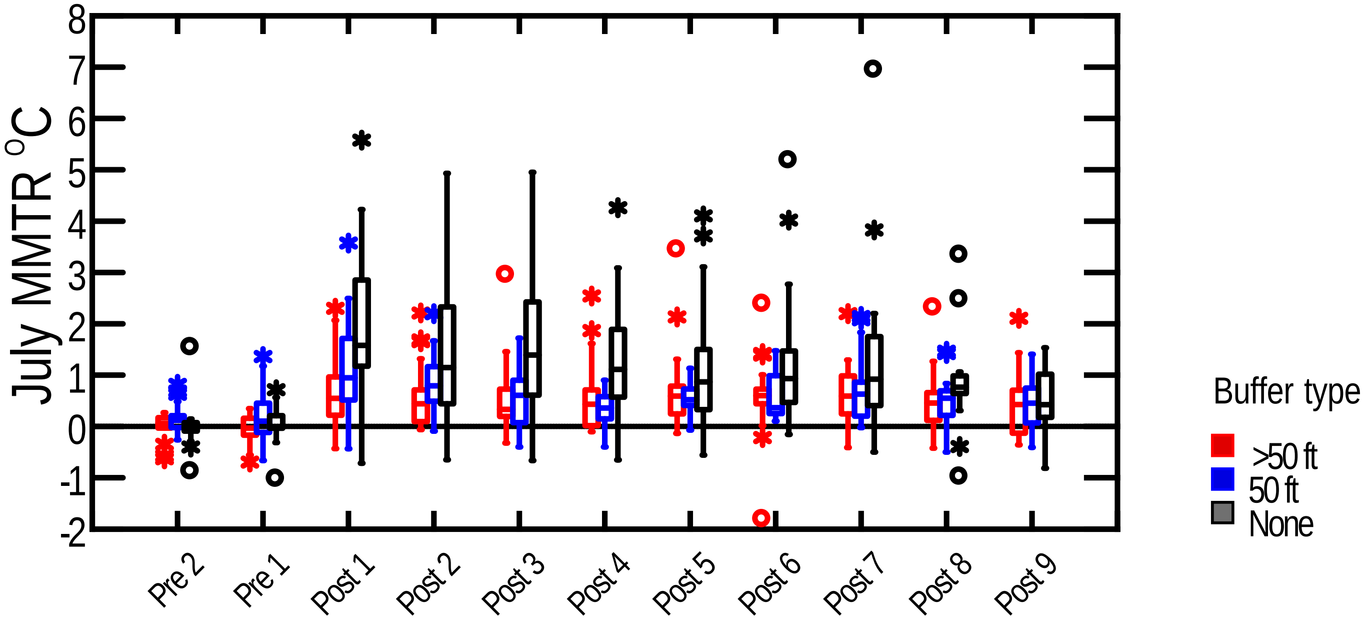


Figure 4-16. Box plots of July mean monthly temperature response (MMTR) plotted by buffer type and treatment year. Red = greater than 50-ft wide (n = 15-21); blue = 50-ft wide (n = 18-25); black = no buffer (n = 18-23). The box represents the 75th, 50th, and 25th percentiles. Whiskers show the range of values within 1.5 times the interquartile range of the 75th and 25th percentiles. Asterisks are values from 1.5 and 3 times the interquartile range. Circles are greater than 3 times the interquartile range.

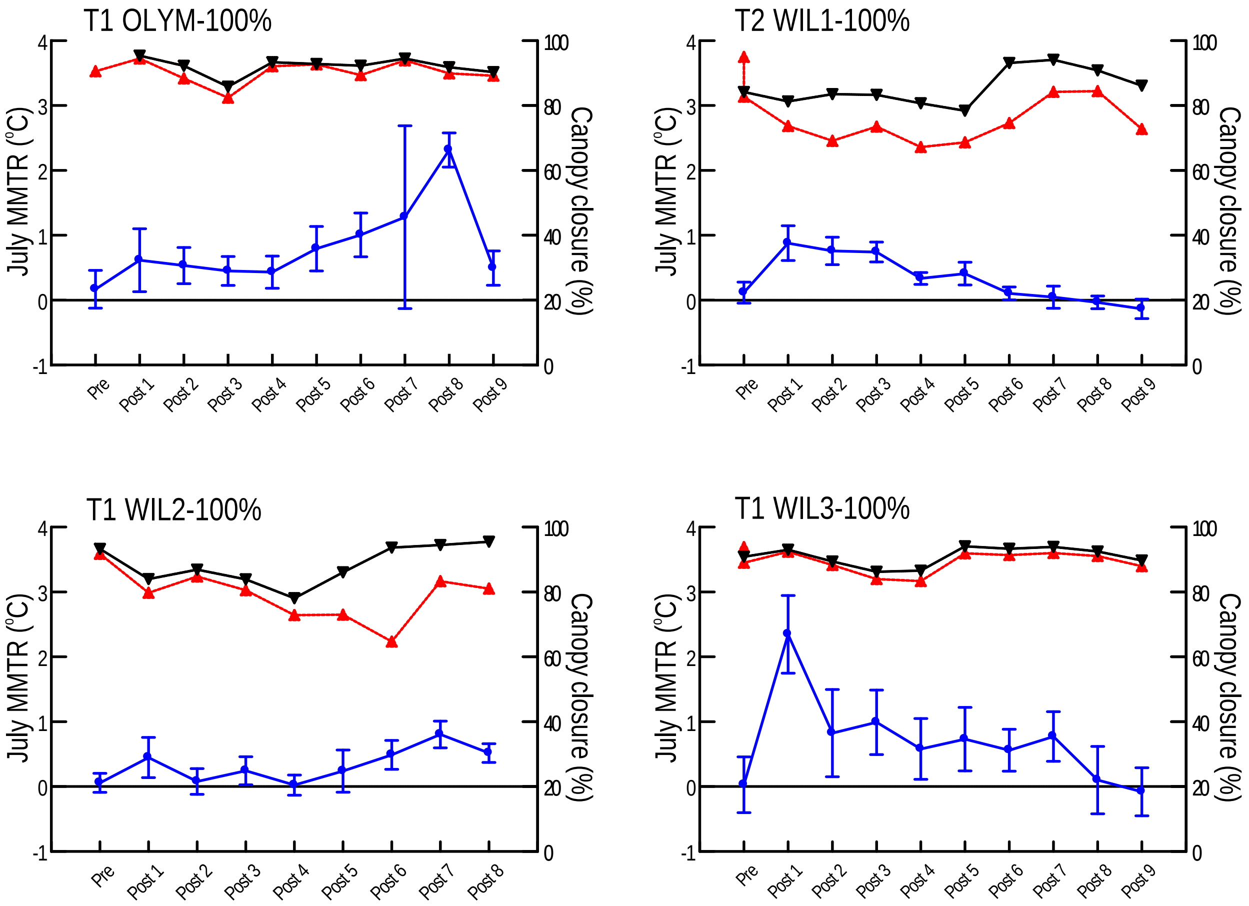


Figure 4-17. July mean monthly temperature response (MMTR) with 95% confidence intervals (blue, left y-axis), mean canopy closure-1m (red, right y-axis), and mean canopy closure-0m (black, right y-axis) vs. treatment year in the 100% treatment sites.

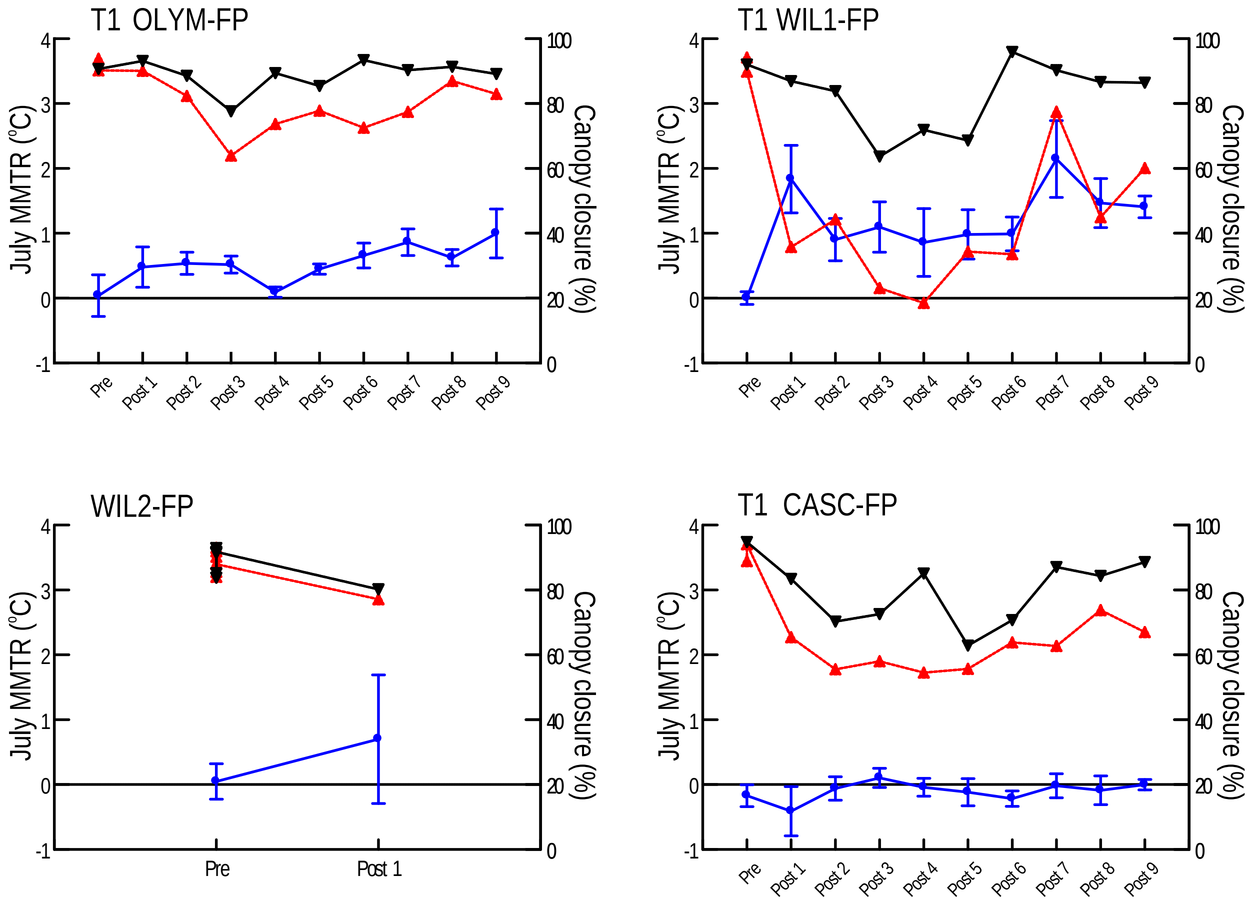


Figure 4-18. July mean monthly temperature response (MMTR) with 95% confidence intervals (blue, left y-axis), mean canopy closure-1m (red, right y-axis), and mean canopy closure-0m (black, right y-axis) vs. treatment year in the FP treatment sites.

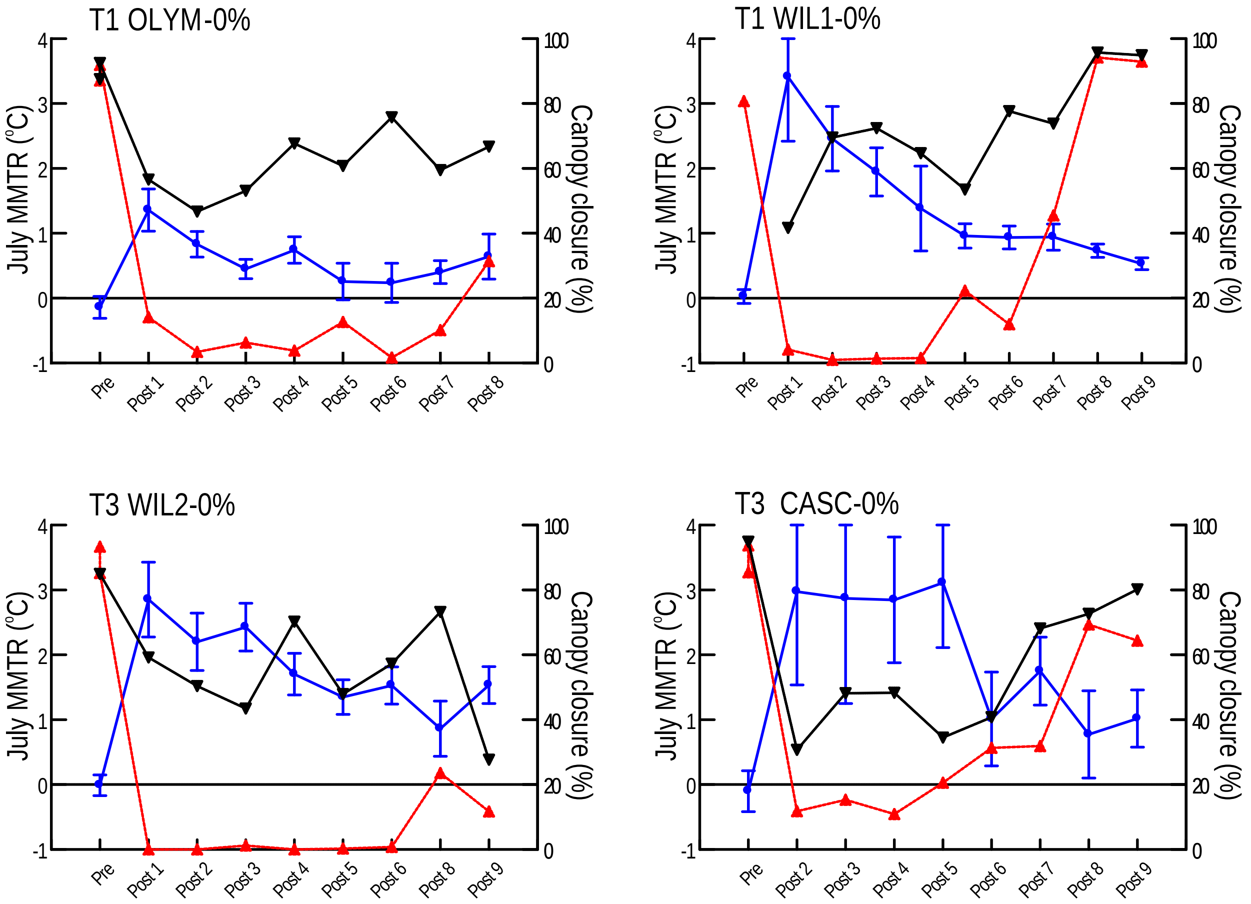


Figure 4-19. July mean monthly temperature response (MMTR) with 95% confidence intervals (blue, left y-axis), mean canopy closure-1m (red, right y-axis), and mean canopy closure-0m (black, right y-axis) vs. treatment year in the 0% treatment sites.

We examined plots of July MMTR by site aspect for each post-harvest year for evidence of whether a southerly aspect influenced the summer temperature response. In the first five post-harvest years and in Post 7 the highest MMTR in each treatment was nearly always the site with a southern (SE or SW) aspect (**Figure 4-20**). The one exception was in Post 1 when we could not calculate MMTR at the 0% site with a southeastern aspect (CASC-0%). The WIL1-FP site with a southwest aspect was consistently the warmest FP site all nine post-harvest years.

As noted in Chapter 5 (*Stream Discharge, Turbidity, and Suspended Sediment Export* in this report), summer low flows increased after harvest in the FP and 0% treatments but were lower in the 100% treatment sites over most of the post-harvest period. However, we observed no significant correlation between July MMTR and either mean July discharge or the post-harvest difference in discharge. However, we monitored discharge at only six harvested sites, so it was less likely that subtle relationships would be detected.

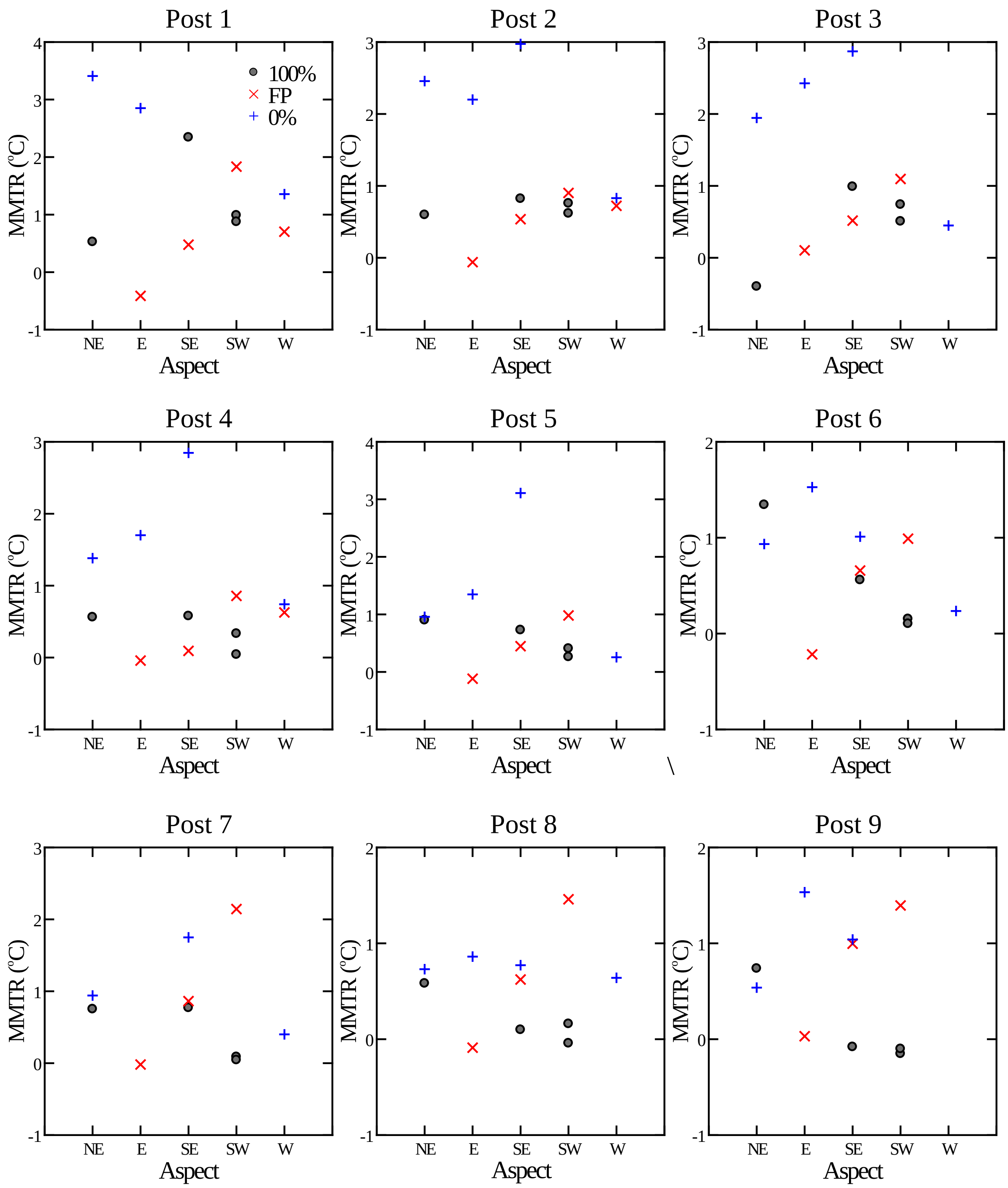


Figure 4-20. July mean monthly temperature response (MMTR) plotted by stream aspect for each post-harvest year.

Mid-summer surveys showed that the proportion of the stream network with no surface water ranged from 0 to 51% pre-harvest and showed no consistent change post-harvest (**Table 4-15**). Sites with less than 10% dry reaches (e.g., WIL2-REF2, WIL2-100%, WIL3-100%, WIL1-FP, WIL1-0%, WIL2-0%) tended to change little across surveys, while most sites with more than 10% dry reaches reached a minimum in 2010. The changes in surface flow do not correspond well with the changes in summer low flow observed at the sites where discharge was measured. For example, in two of the four sites where summer low flows increased, OLYM-FP and OLYM-0%, the percent of channel with no surface water increased. In contrast, the percent with no surface flow increased in the two 100% sites, where summer low flows decreased after harvest. There was no significant correlation (P >0.05) between MMTR and the length of contiguous, wetted channel above the monitoring location nor with the proportion of the Type Np channel with no surface flow.

There was no consistent relationship between July MMTR and the percent of stream buffered or the length of stream buffered (**Figure 4-21**). WIL1-FP had the highest MMTR and the shortest absolute length of stream buffered, but this represented a moderate percentage of stream. CASC-FP had the lowest MMTR and the lowest percentage of stream buffered, but represented a moderate length of stream. The OLYM-FP and WIL2-FP had similar MMTR values and similar length buffers, but differed widely in the percent of stream buffered.

Table 4-15. Proportion of Type Np channel with no mid-summer surface flow. Shaded cells indicate post-harvest surveys. Trmt = treatment.

Trmt	Block	2006	2010	2015	2016
REF	OLYM	0.14	0.05	0.10	0.08
	WIL1	0.35	0.18	0.24	0.26
	WIL2	0.00	0.01	0.01	0.00
	WIL2*	0.23	0.13	0.22	
	WIL3	0.45	0.31	0.42	0.42
	CASC	0.35	0.21	0.44	0.37
100%	OLYM	0.07	0.20	0.33	0.10
	WIL1	0.10	0.13	0.14	0.14
	WIL2	0.01	0.01	0.01	0.02
	WIL3	0.00	0.00	0.00	0.05
FP	OLYM	0.21	0.04	0.42	0.38
	WIL1	0.04	0.00	0.00	0.00
	CASC	0.32	0.05	0.28	0.24
0%	OLYM	0.51	0.46	0.63	0.34
	WIL1	0.03	0.05	0.09	0.12
	WIL2	0.02	0.03	0.02	0.04
	CASC	0.26	0.00	0.14	0.04

*WIL2-FP is included with reference sites because all surveys were done before harvest in 2016.

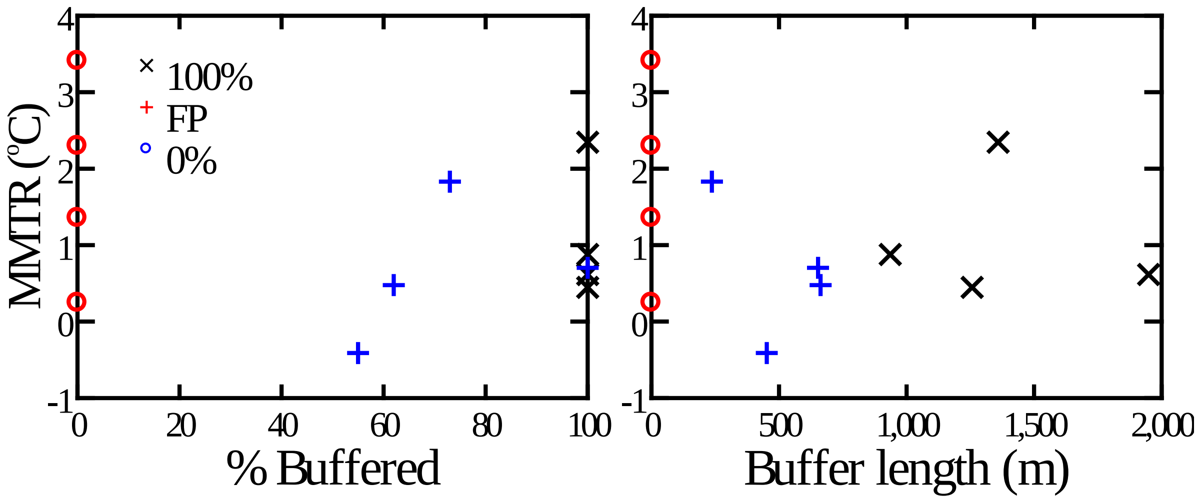


Figure 4-21. July mean monthly temperature response (MMTR) plotted against the percentage of stream buffered and the length of stream buffered.

4-5.2.4. Longitudinal Patterns in Temperature Response

Longitudinal patterns in the point estimates of July MMTR along the main channel were variable among treatments and often within treatments even when considering canopy closure and surface flow. For example, July MMTR in the OLYM-100% site was generally less than 1.0°C and varied little longitudinally or over time (**Figure 4-22**). In contrast, in the early post-harvest years in WIL1-100% and WIL2-100% July MMTR was greater than 1.0°C in the upper watershed T4 and T3 locations, then decreased in value moving downstream (**Figures 4-23** and **4-24**). July MMTR within the WIL3-100% site exceeded 2.0°C at all locations in Post 1, then gradually decreased until by Post 4 all locations were less than 1.0° (**Figure 4-25**). By four years post-harvest nearly all MMTR values in the 100% sites were, and remained through Post 9, less than 1.0°C.

Within the three FP sites, MMTR at T3, which was within or immediately below the unbuffered reach, was consistently greater than 1.0°C immediately after harvest. When MMTR at T3 exceeded 1.0°C in the OLYM-FP (**Figure 4-26**) and CASC-FP (**Figure 4-28**) sites, downstream MMTR values within the 50 ft wide buffer were consistently and often substantially lower (i.e., non-overlapping 95% confidence intervals). In contrast, MMTR changed little through the harvest unit in WIL1-FP (**Figure 4-27**) from locations T3 through T1 but was consistently lower below the harvest unit at D100 within the fish-bearing stream buffer. The CASC-FP site was unique in that MMTR at the F/N junction (T1) was near zero and varied little over the course of the study in spite of consistent, substantial warming noted within the unbuffered reach at T3 (**Figure 4-28**).

In the 0% treatment, MMTR in the OLYM site tended to be lower than in the other 0% sites and less than 1.0°C at most locations in most years after Post 1 (**Figure 4-29**). There was usually little difference in MMTR between T2 and T1 and little difference between T1 and D100, except when T1 was greater than 1.0°C. In those cases, MMTR was lower at D100. In the other three

0% sites, July MMTR tended to be higher in the middle portion of the harvested reach than at either end, often exceeding 3.0° in the early post-harvest years and usually greatest at T2, well above the F/N junction (**Figures 4-30**, **4-31**, and **4-32**). MMTR tended to decrease at all locations in these sites over time.

At several sites MMTR was lower below stream reaches with no surface discharge. July MMTR at T4 in OLYM-100% (**Figure 4-22**), situated within a reach with only isolated pools, was consistently lower than further downstream in Post 2–9. Locations T4 and T3 in WIL1-0%, located below subsurface reaches, tended to have lower MMTR than further downstream (**Figure 4-30**) and CASC-FP, mentioned above. However, the pattern is not consistent across sites.

There was a negative difference between July MMTR downstream of the harvest unit and the upstream locations at nearly every site/year combination where upstream (harvest unit boundary) July MMTR was greater than 0.5°C (i.e., warming had occurred in the harvest unit) (**Table 4-16**). Cooling was greatest (difference >1.0°C) downstream of the harvest unit where upstream warming was highest (upstream MMTR >1.5°C). July MMTR at the downstream location was rarely greater than 0.5°C in the two 100% treatment sites with downstream monitoring, suggesting little or no warming downstream of the harvest unit over the course of the study. However, MMTRs were elevated over much of the post-harvest period at both the upstream and downstream sites in the one FP site and the three 0% treatment sites reflecting the higher MMTR observed at the harvest unit boundary.

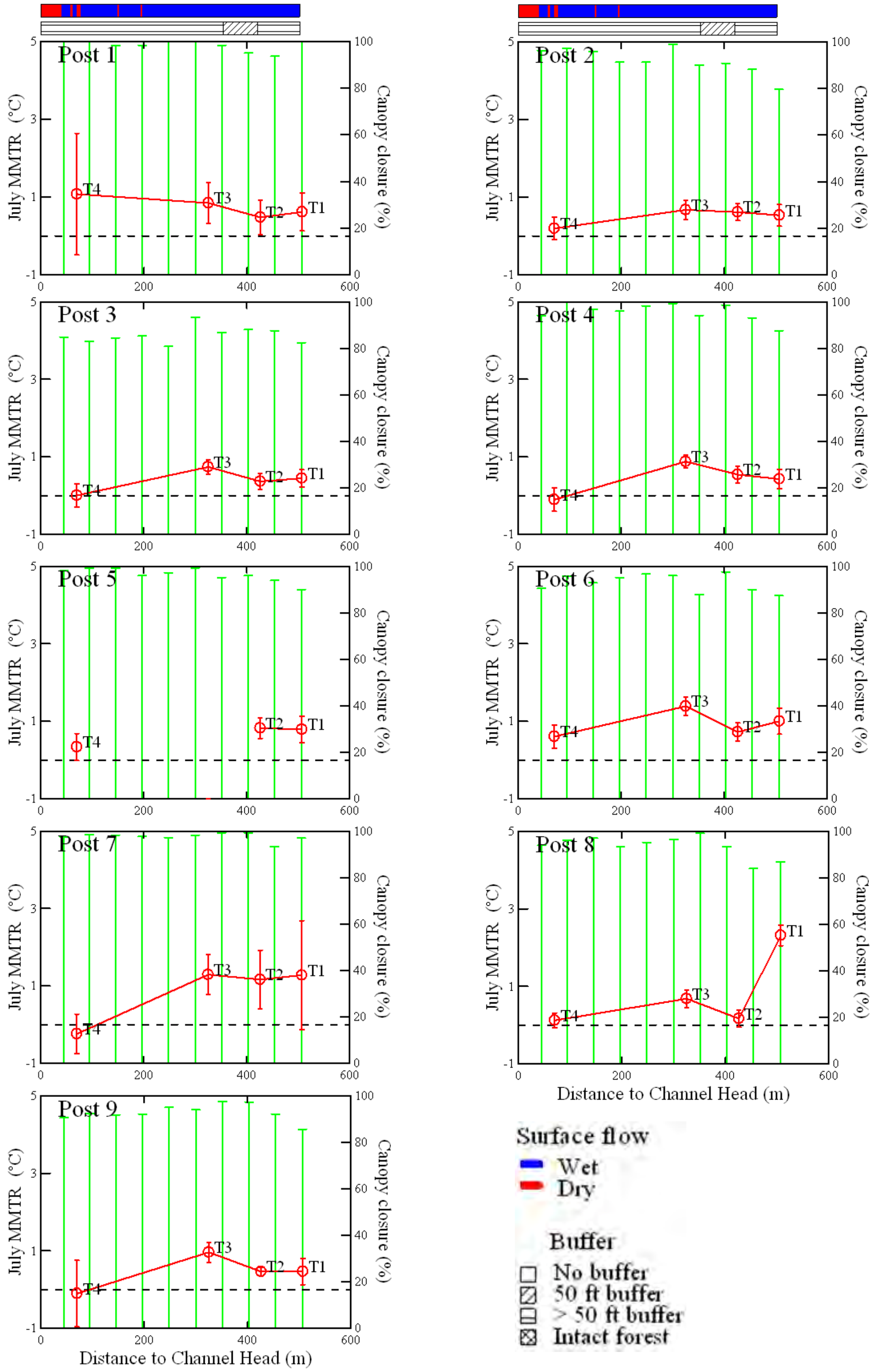


Figure 4-22. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at OLYM-100% plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

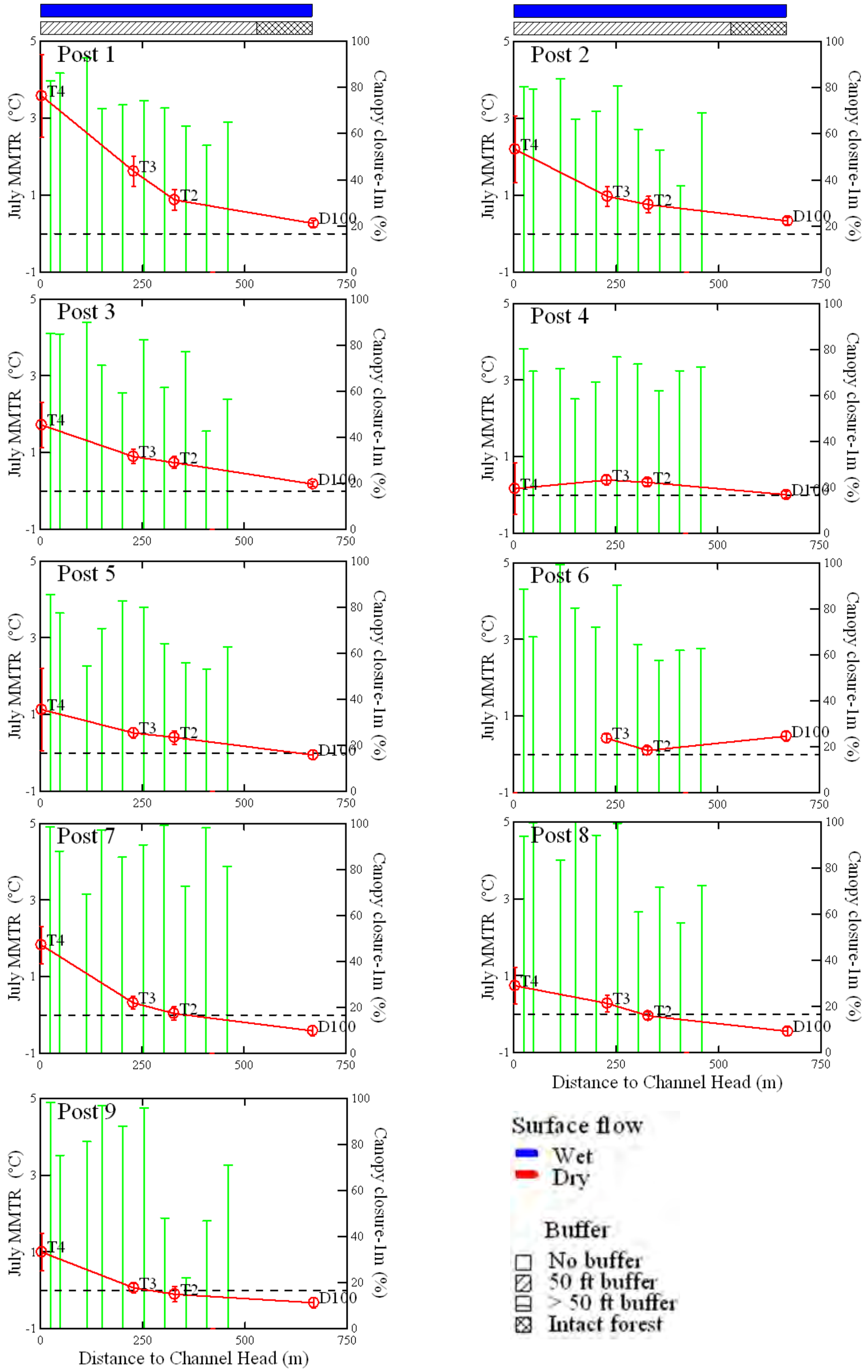


Figure 4-23. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at WIL1-100% plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

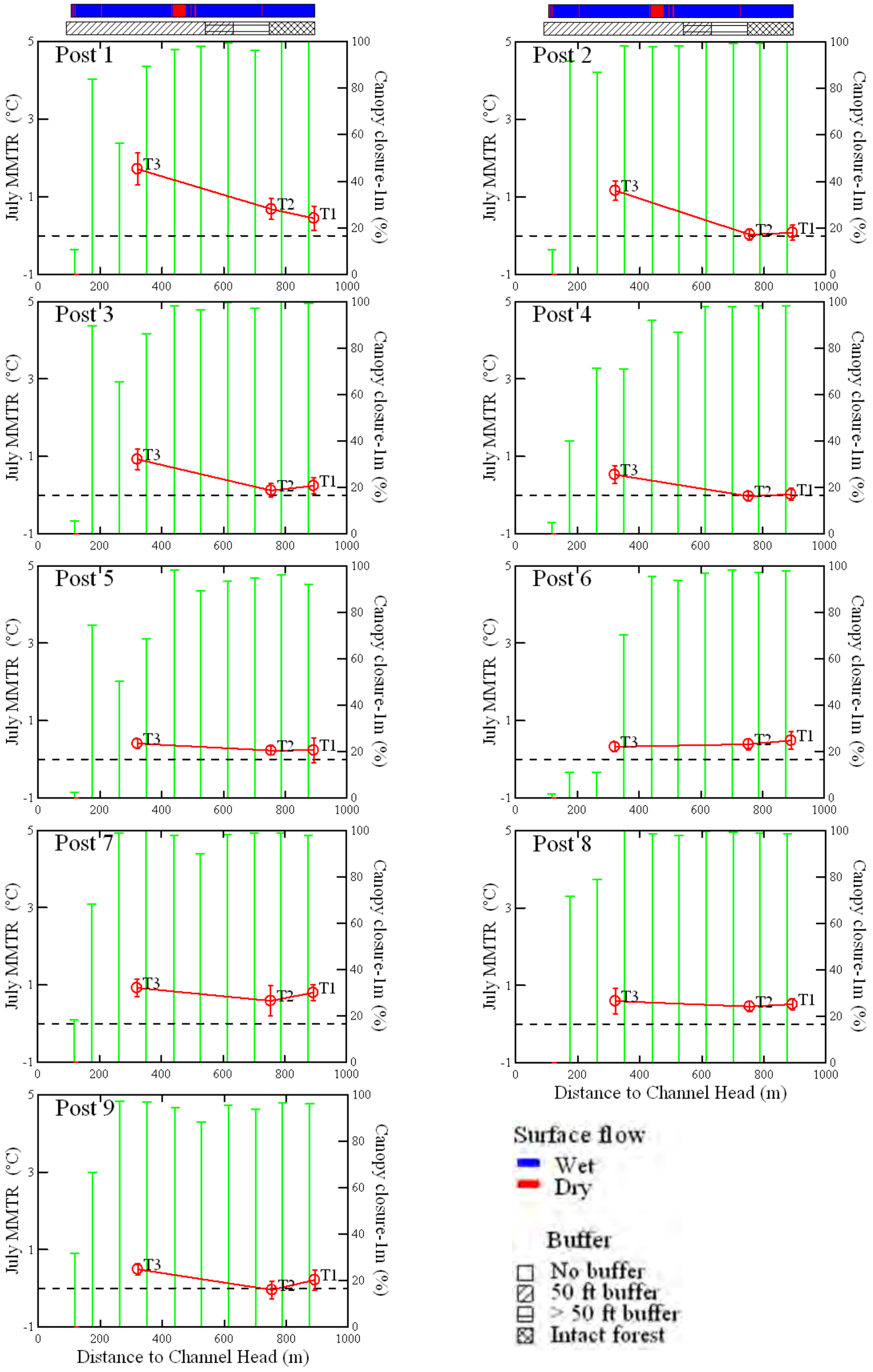


Figure 4-24. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at WIL2-100% plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

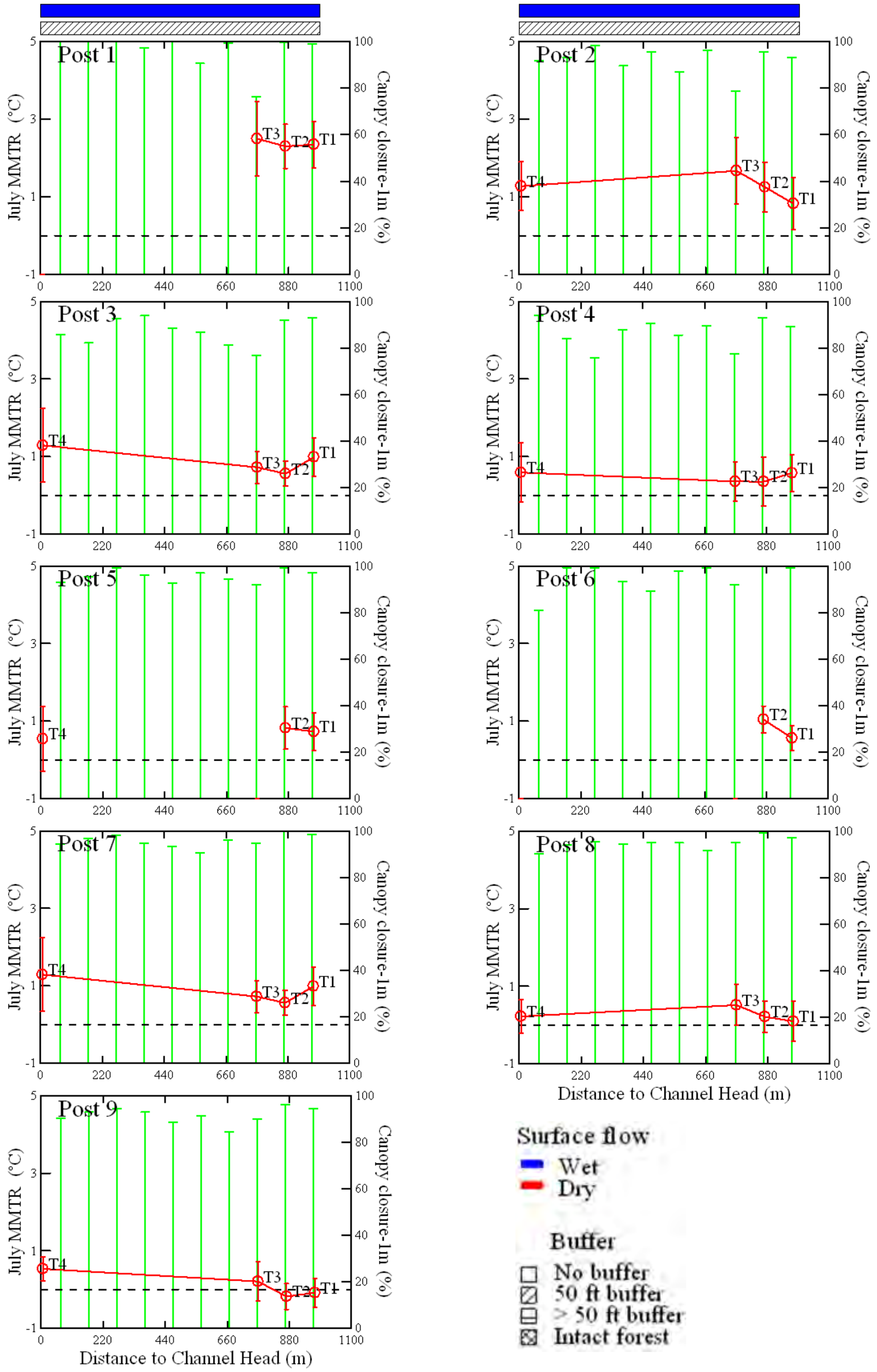


Figure 4-25. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at WIL3-100% plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

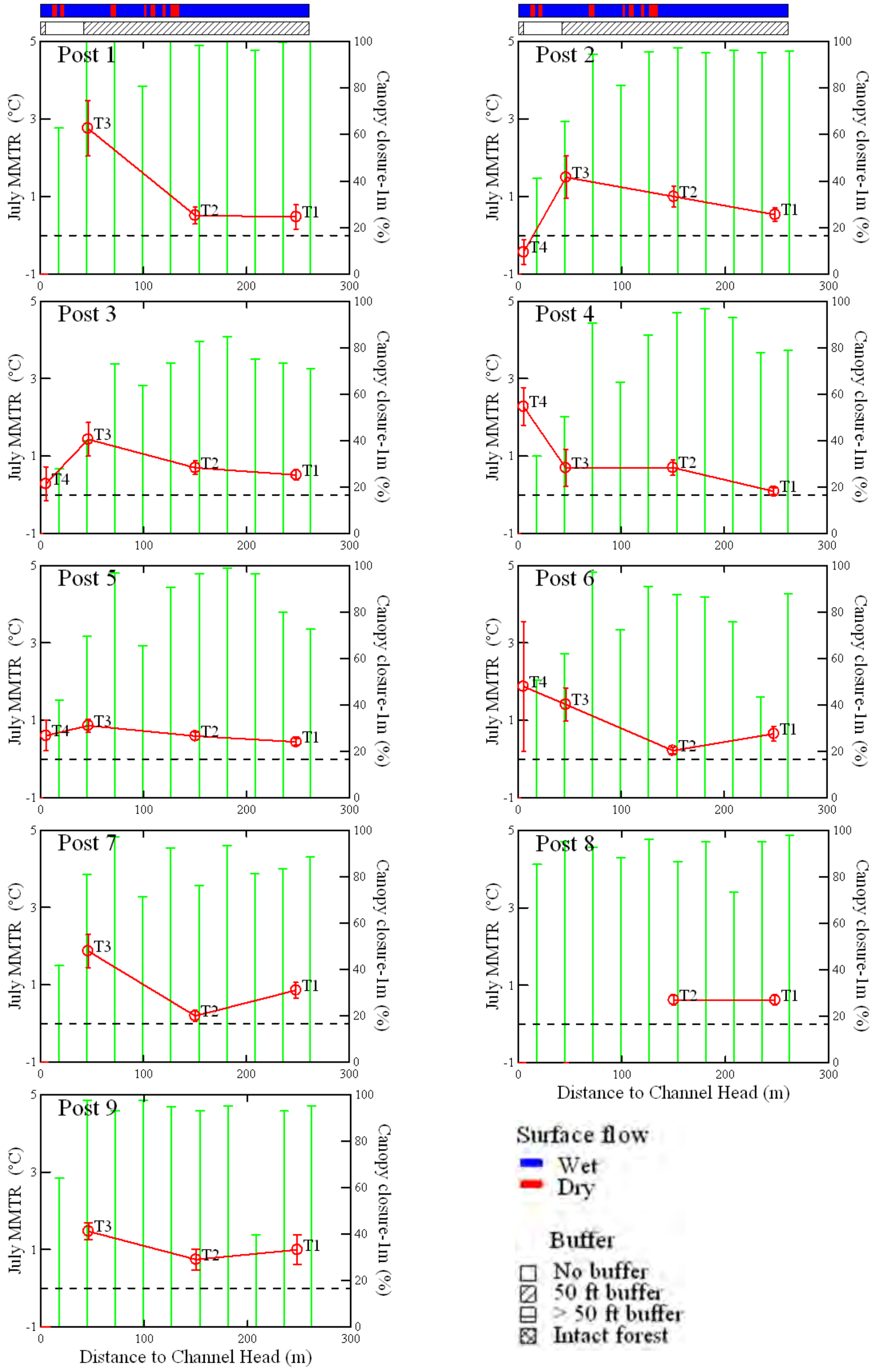


Figure 4-26. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at OLYM-FP plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

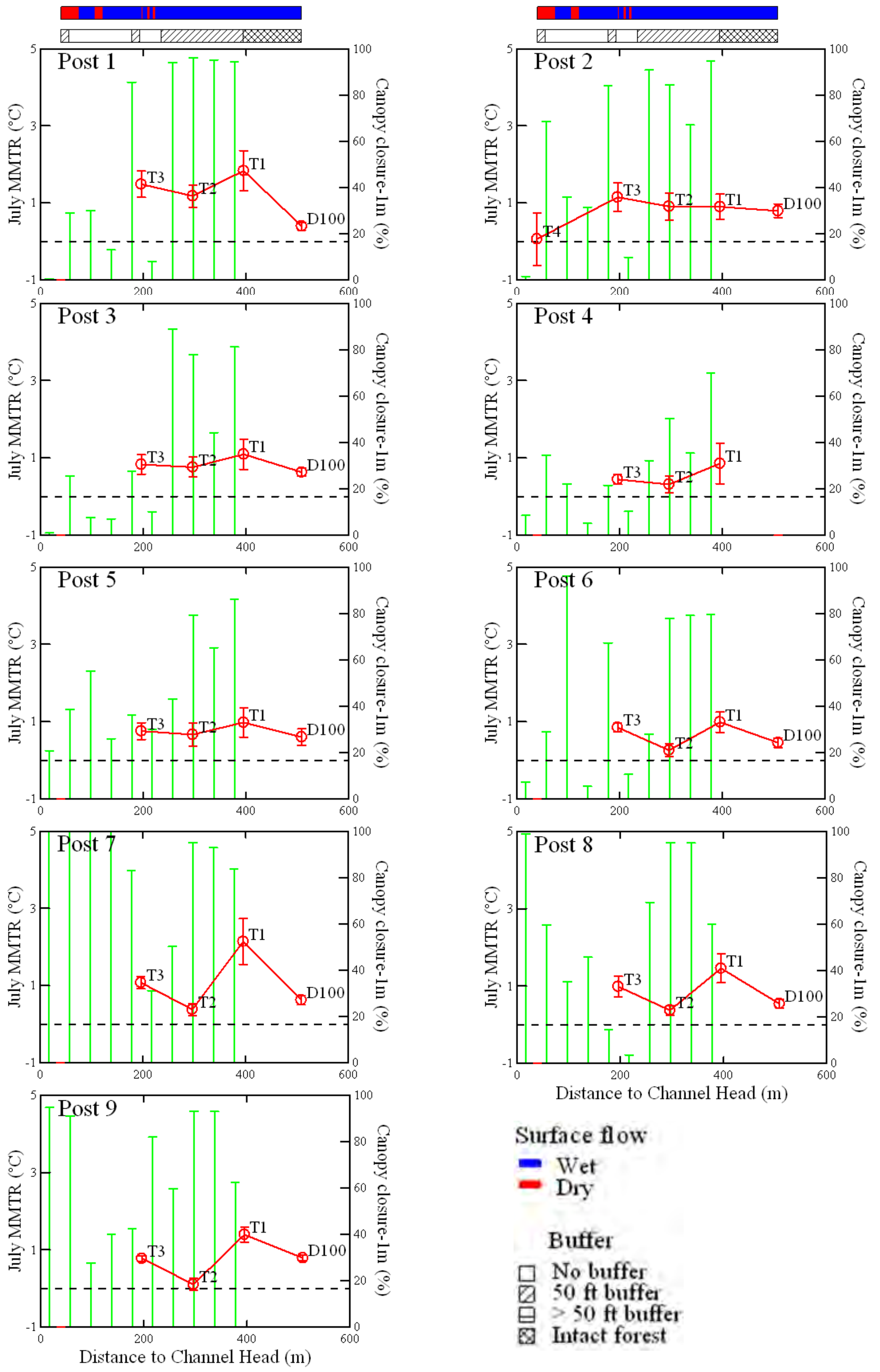


Figure 4-27. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at WIL1-FP plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

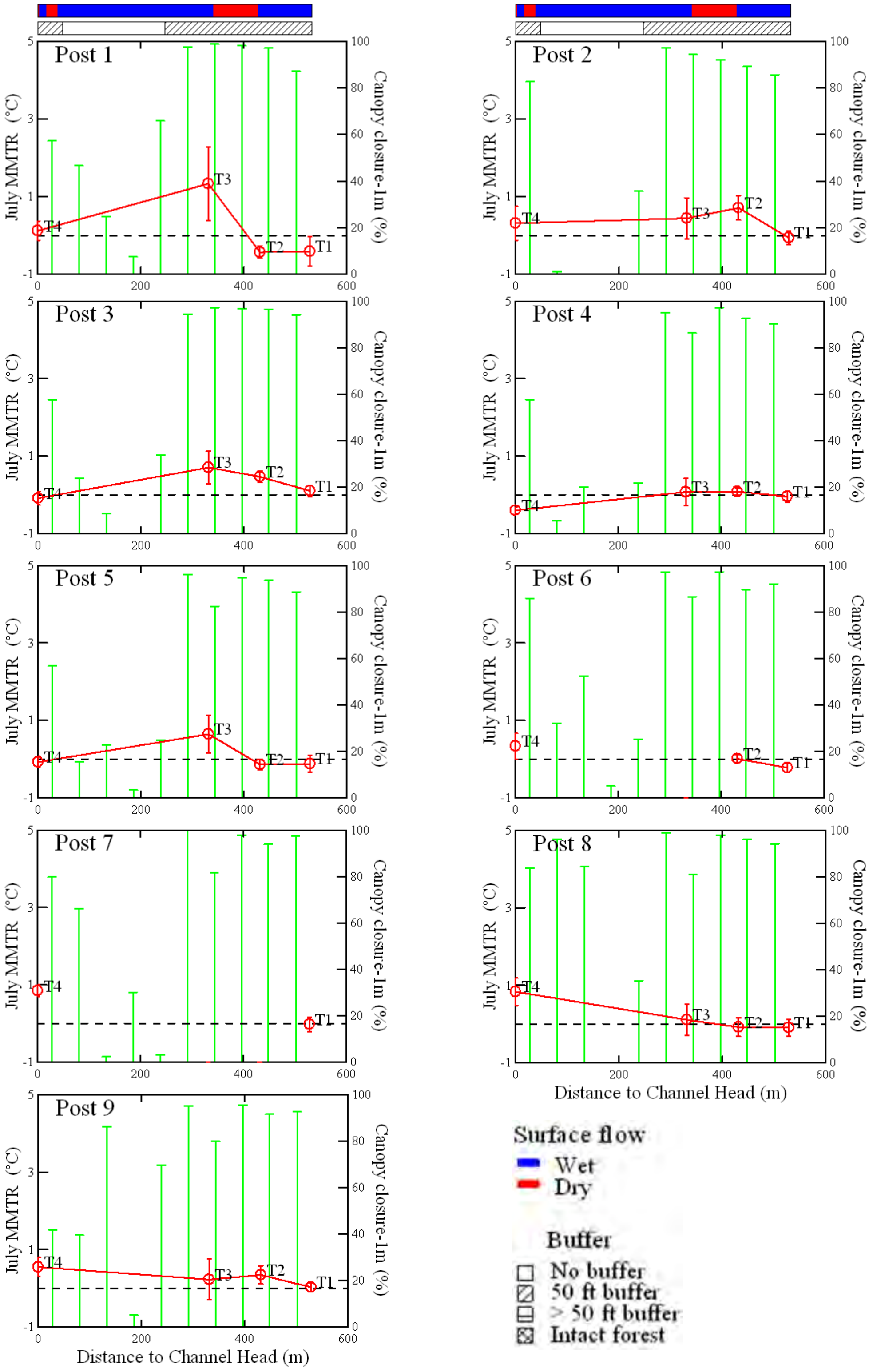


Figure 4-28. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at CASC-FP plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

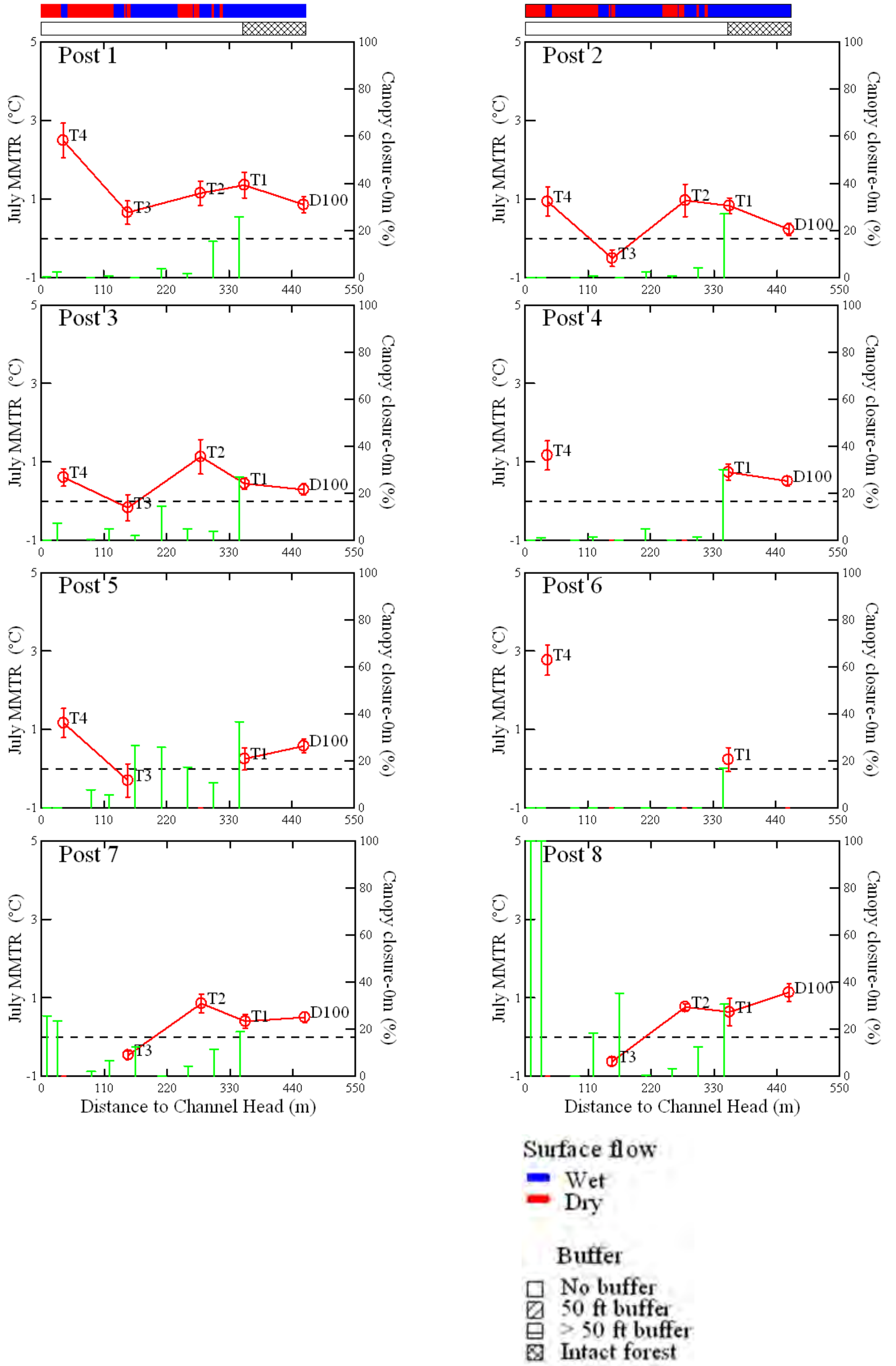


Figure 4-29. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at OLYM-0% plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

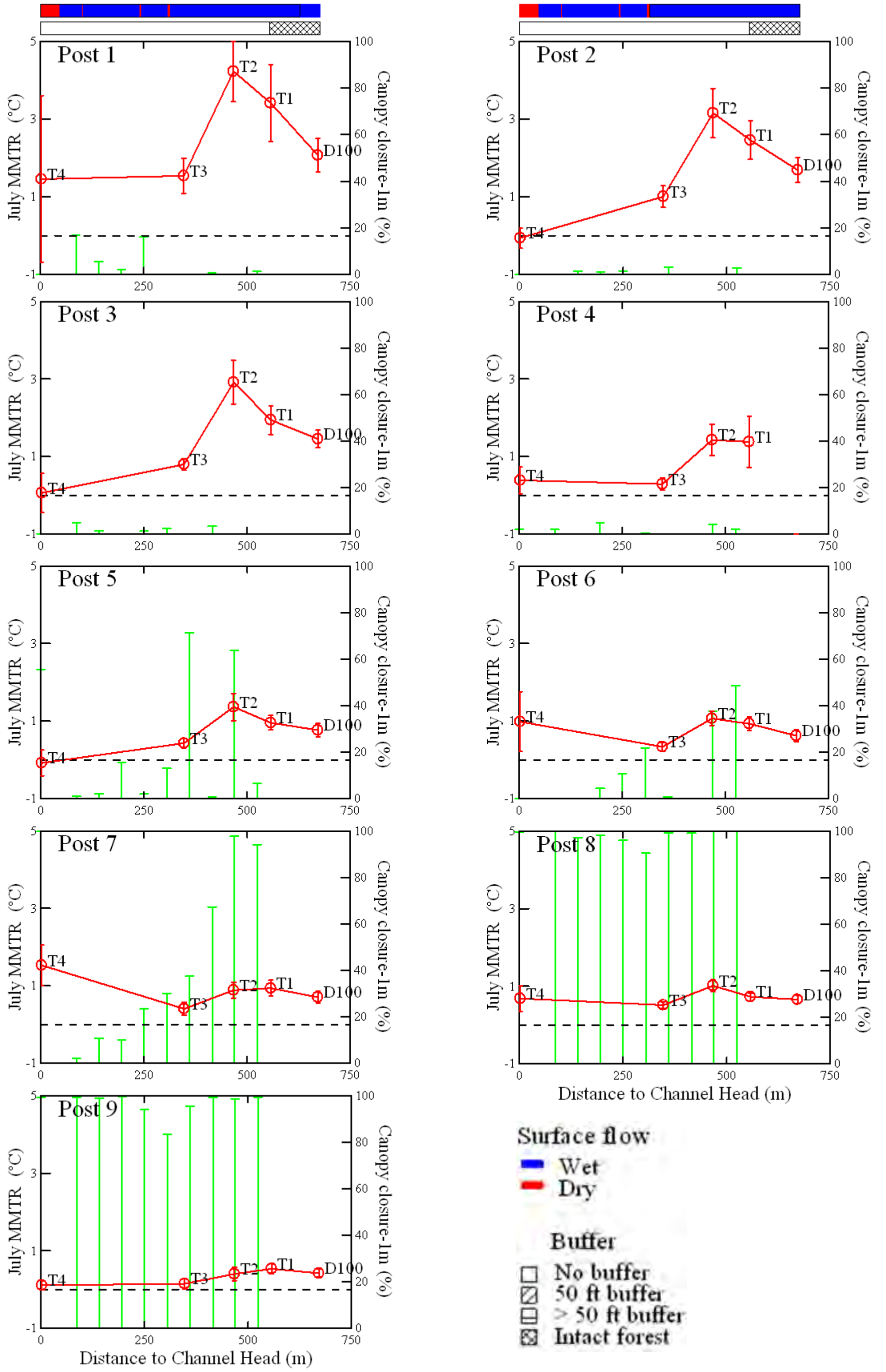


Figure 4-30. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at WIL1-0% plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

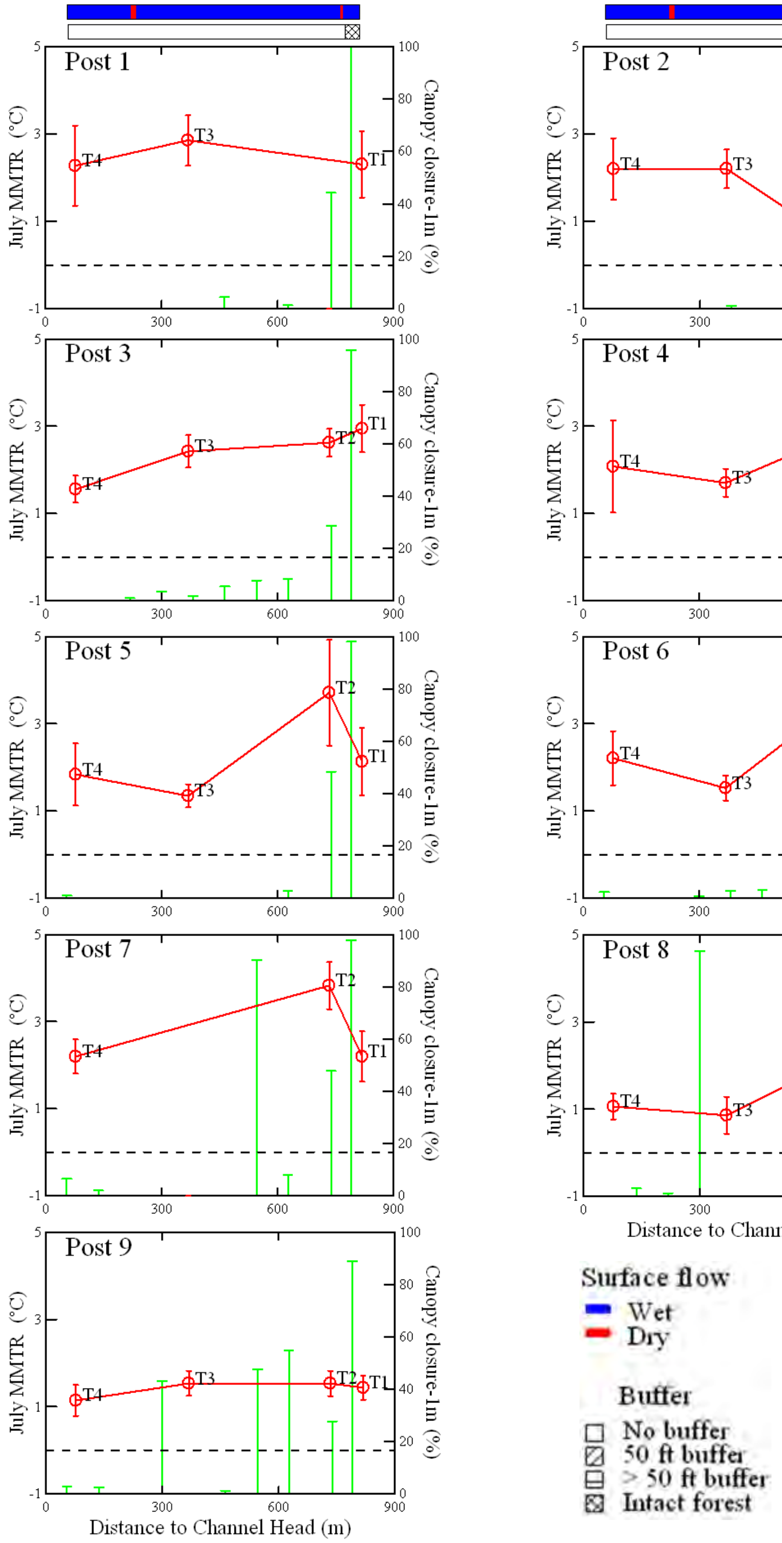


Figure 4-31. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at WIL2-0% plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

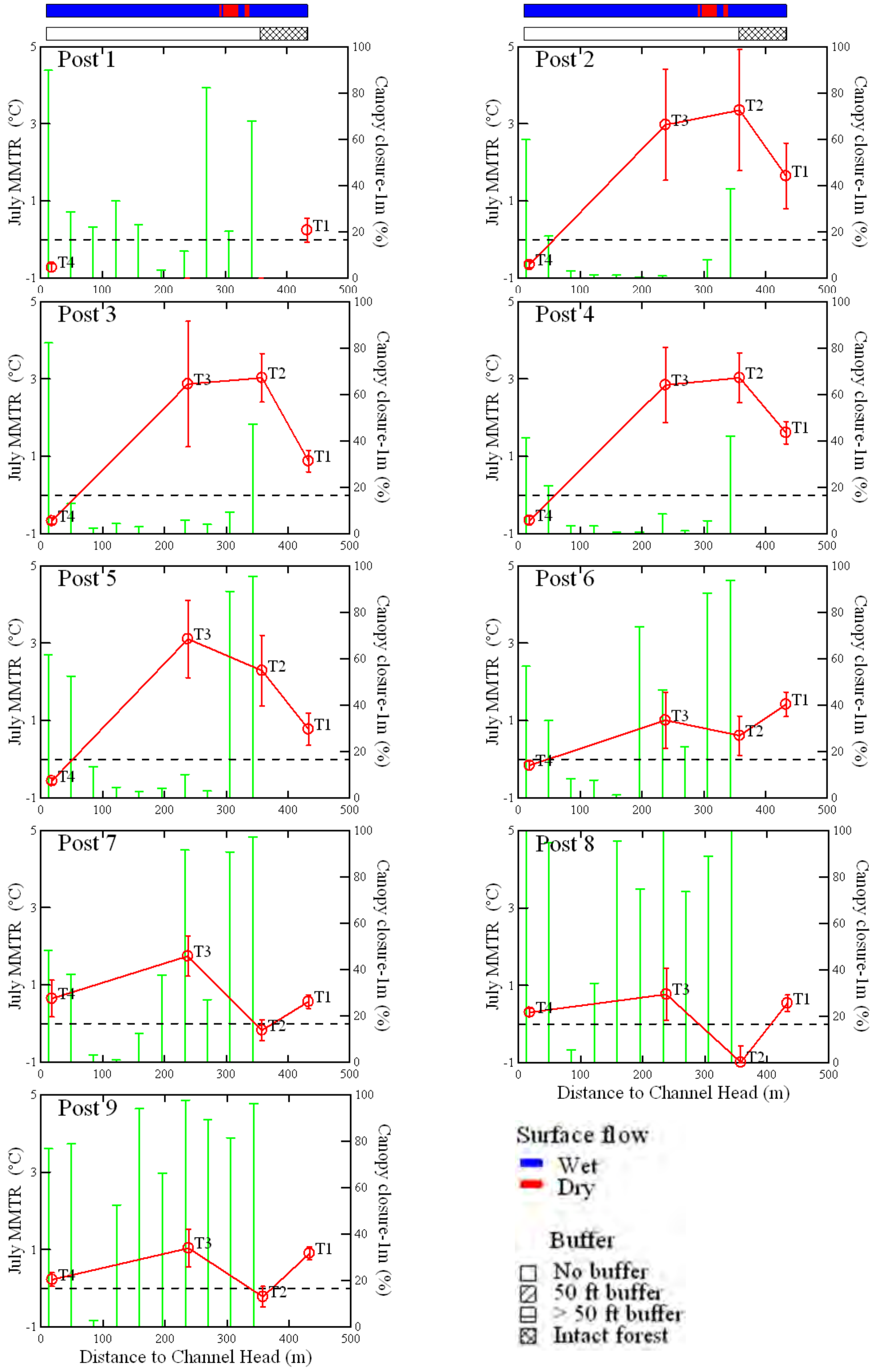


Figure 4-32. July mean monthly temperature response (MMTR) with 95% confidence intervals (red-left axis) and canopy closure-1m (green bars-right axis) at CASC-0% plotted by distance along the main stream channel. Presence of surface flow was collected in 2010. Dry reaches may have isolated pools where temperature was measured. Intact forest buffer type was below the harvest. Horizontal dashed line is MMTR equal zero.

Table 4-16. July mean monthly temperature response (MMTR, °C) at the bottom of the harvest unit (Up), downstream of the harvest unit within a wider riparian buffer (Down), and the difference between them (Diff; Down minus Up). A negative difference indicates less warming downstream within the wider buffer after harvest than upstream. Shaded values indicate MMTR >0.5°C and P <0.05.

	WIL1-100%			**WIL2-100%**			**WIL1-FP**		
	Up	**Down**	**Diff**	**Up**	**Down**	**Diff**	**Up**	**Down**	**Diff**
Post 1	0.9	0.3	-0.6	0.7	0.4	-0.2	1.8	0.4	-1.4
Post 2	0.8	0.3	-0.4	0.0	0.1	0.0	0.9	0.8	-0.1
Post 3	0.7	0.2	-0.6	0.1	0.2	0.1	1.1	0.6	-0.5
Post 4	0.3	0.0	-0.3	0.0	0.0	0.0	0.9		-0.9
Post 5	0.4	0.0	-0.5	0.2	0.2	0.0	1.0	0.6	-0.4
Post 6	0.1	0.5	0.4	0.4	0.5	0.1	1.0	0.5	-0.5
Post 7	0.0	-0.4	-0.5	0.6	0.8	0.2	2.1	0.6	-1.5
Post 8	0.0	-0.4	-0.4	0.5	0.5	0.1	1.5	0.5	-0.9
Post 9	-0.1	-0.3	-0.2	0.0	0.2	0.3	1.4	0.8	-0.6

	OLYM-0%			**WIL1-0%**			**CASC-0%**		
	Up	**Down**	**Diff**	**Up**	**Down**	**Diff**	**Up**	**Down**	**Diff**
Post 1	1.4	0.9	-0.5	3.4	2.1	-1.3			
Post 2	0.8	0.2	-0.6	2.5	1.7	-0.8	3.4	1.7	-1.7
Post 3	0.4	0.3	-0.2	1.9	1.5	-0.5	3.0	0.9	-2.1
Post 4	0.7	0.5	-0.2	1.4			3.0	1.6	-1.4
Post 5	0.3	0.6	0.3	1.0	0.8	-0.2	2.3	0.8	-1.5
Post 6	0.2			0.9	0.6	-0.3	0.6	1.4	0.8
Post 7	0.4	0.5	0.1	0.9	0.7	-0.2	-0.2	0.6	0.7
Post 8	0.6	1.1	0.5	0.8	0.7	-0.1	-1.0	0.5	1.5
Post 9				0.5	0.4	-0.1	-0.2	0.9	1.1

4-5.2.5. Buffer Treatment Effects

4-5.2.5.a. Buffer treatment locations

The GLMM ANOVA showed significant period and treatment × period interactions (P <0.05) (**Table 4-17**). The Δ7DTR was elevated (P <0.05) in the first two years post-harvest in all three buffer treatments (**Table 4-18**; **Figure 4-33**). In the 100% treatment, Δ7DTR was significantly elevated in Post 1 (1.1°C) and Post 2 (1.1°C), then was near zero through Post 9. The initial response in the FP treatment was similar to the 100% treatment, with values of 1.1°C and 0.9°C in Post 1 and Post 2, respectively, but Δ7DTR remained elevated for most of the post-harvest period. In the 0% treatment, Δ7DTR was significantly (P <0.05) elevated for all years, from a high of 3.8°C in Post 1 declining to a minimum of 0.8°C in Post 9.

The 100% and FP treatment responses were not significantly different from each other except in Post 8, when the FP was 1.1°C higher (**Table 4-18**), however, Δ7DTR in the FP treatment was consistently higher. The 0% treatment was higher than the 100% and FP treatment in nearly all years, but $P < 0.05$ only during some of the Post 1–6 years. The pairwise comparisons with 95% confidence intervals are presented in **Appendix Table 4-17**.

Table 4-17. Results of the generalized linear mixed-effects model for temperature response at the F/N break and at the buffer treatment locations. Significant ($P < 0.05$) treatment × period interaction terms indicate pre- to post-harvest differences among treatments. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Metric	Effect	Num DF	Den DF	F Value	Pr > F
Δ7DTR F/N break	Treatment	2	9.13	1.59	0.2561
	Period	9	94.3	10.17	<0.0001
	Treatment × Period	18	94.3	2.62	0.0014
Δ7DTR Buffer Treatment	Treatment	2	9.19	2.24	0.1614
	Period	9	93.3	15.89	<0.0001
	Treatment × Period	18	93.2	3.67	<0.0001

Table 4-18. Estimated mean seven-day average temperature response measured at the F/N break and at the buffer treatment locations. Bold type indicates a significant ($P < 0.05$) change from pre-harvest. Subscripts next to the estimate list treatments where $P < 0.05$ for that year's estimate. Estimates and 95% confidence intervals are tabulated in **Appendix Tables 4-17** and **4-18**.

	F/N break			**Buffer Treatment**		
Year	**100%**	**FP**	**0%**	**100%**	**FP**	**0%**
Post 1	**0.9**$_{0\%}$	**1.1**$_{0\%}$	**3.2**	**1.1**$_{0\%}$	**1.1**$_{0\%}$	**3.8**
Post 2	0.5$_{0\%}$	**0.9**$_{0\%}$	**2.7**	**1.1**$_{0\%}$	**0.9**$_{0\%}$	**3.0**
Post 3	0.6$_{0\%}$	0.8	**1.9**	0.3$_{0\%}$	**0.8**$_{0\%}$	**2.4**
Post 4	0.6$_{0\%}$	0.5$_{0\%}$	**1.8**	0.5$_{0\%}$	0.5$_{0\%}$	**2.0**
Post 5	0.4$_{0\%}$	0.5	**1.6**	0.4$_{0\%}$	0.5$_{0\%}$	**1.6**
Post 6	0.6	0.9	**1.2**	0.2$_{0\%}$	**0.9**	**1.3**
Post 7	**1.0**	**1.2**	**1.4**	0.3	**1.2**	**1.2**
Post 8	0.6	**1.2**	**0.9**	0.1$_{FP}$	**1.2**	**1.0**
Post 9	0.3	0.8	0.9	0.3	**0.9**	**0.8**

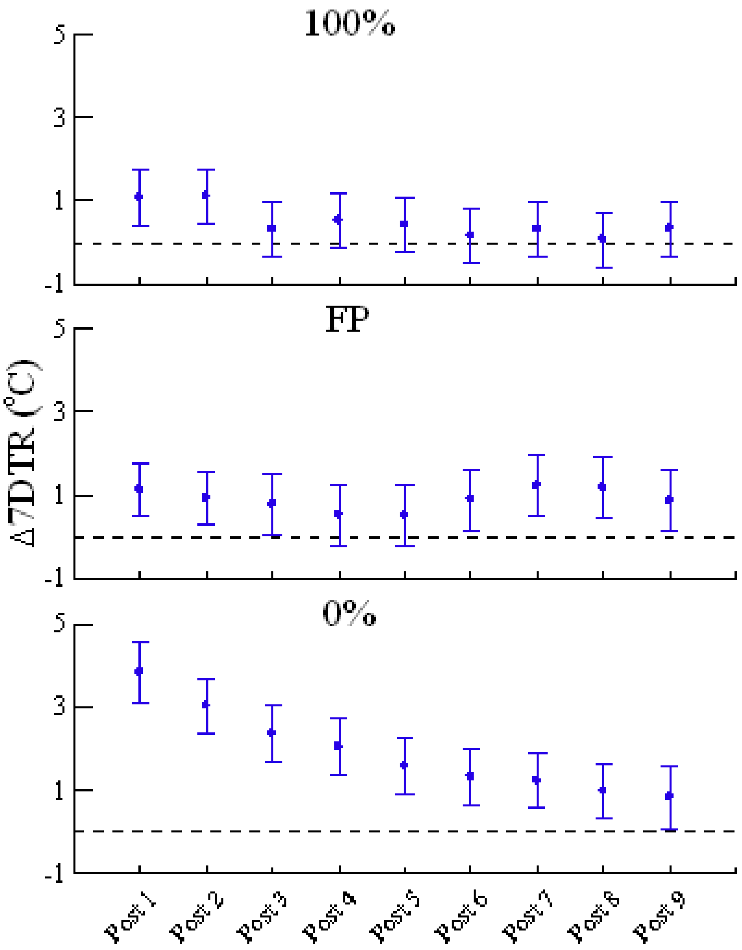


Figure 4-33. Pairwise comparisons of each post-harvest year to the pre-harvest period for the seven-day average temperature response (Δ7DTR) measured at the buffer treatment locations. Estimated means and 95% confidence intervals are presented. Data are shown in **Appendix Table 4-17**.

4-5.2.5.b. F/N break locations

Overall, results at the F/N break were similar to those at the Buffer Treatment locations but slightly lower in magnitude. The Δ7DTR was positive in all years for all treatments (**Table 4-18**; **Figure 4-34**) indicating higher water temperatures after harvest in all treatments. In the 100% sites, Δ7DTR ranged from 0.3°C to 1.0°C but only Post 1 (0.9°C) and Post 7 (1.0°C) were significantly ($P < 0.05$) higher than pre-harvest. The Δ7DTR in the FP treatment ranged from 0.5°C to 1.2°C but $P < 0.05$ only in Post 1 (1.1°C), Post 2 (0.9°C), Post 7 (1.2°C), and Post 8 (1.2°C). The Δ7DTR in the 0% treatment was 3.2°C in Post 1 then declined steadily to 0.9°C in Post 9. Only Post 9 was not significant with $P = 0.059$.

The 100% and FP treatments had similar responses across all years. Although the mean FP treatment response was slightly higher than the 100% in eight of the nine post-harvest years, the pairwise comparisons showed no significant ($P < 0.05$) differences in any year (**Appendix Table 4-18**). The Δ7DTR was higher in the 0% treatment than in either the 100% or FP treatments in nearly all years with $P < 0.05$ in the comparisons with the FP treatment in Post 1, Post 2, and Post 4 and with the 100% treatment in the first five of the nine post-harvest years.

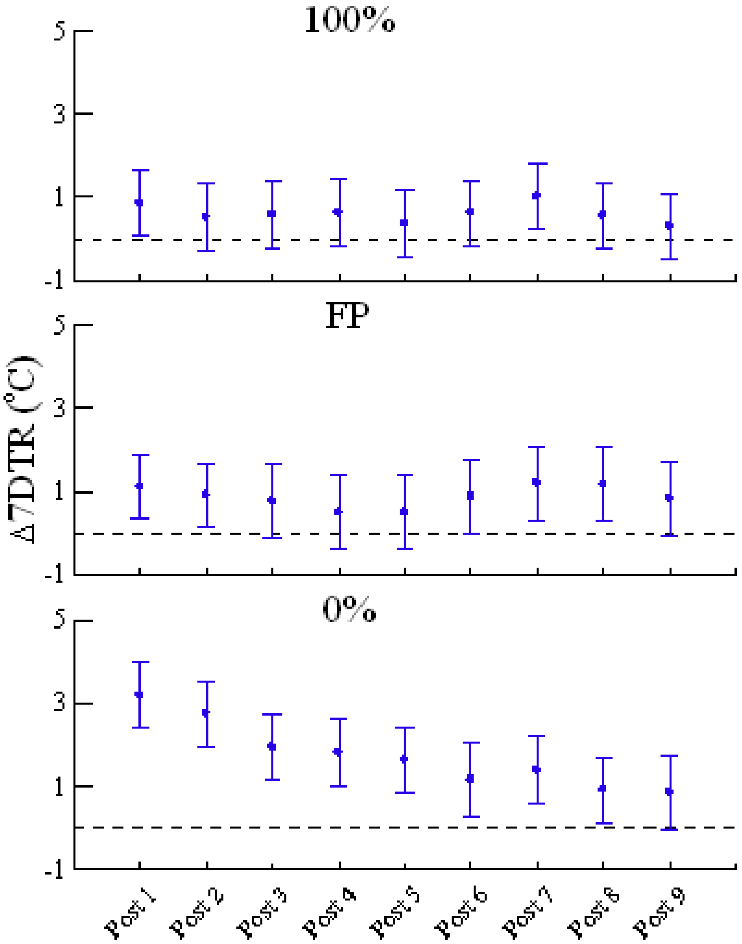


Figure 4-34. Pairwise comparisons of each post-harvest year to the pre-harvest period for the seven-day average temperature response (Δ7DTR) measured at the F/N break locations. Estimated means and 95% confidence intervals are presented. Data are shown in **Appendix Table 4-18**.

4-6. DISCUSSION

4-6.1. RIPARIAN COVER

Riparian cover declined after harvest in all buffer treatments. The initial decrease due to the harvest was followed by loss of overstory shade due to ongoing windthrow in the remaining unharvested buffers. The net effect of harvest and windthrow on the 100% and FP treatments was that mean cover reached a minimum around four years post-harvest. None of the buffer treatments showed appreciable increases in cover until five years post-harvest. CC-1m was still 15 and 27 percentage points below pre-harvest levels in the FP and 0% treatments, respectively (**Table 4-8**).

Our shade measurements before and immediately after harvest were similar to those of other studies from the Pacific Northwest with similar width buffers on similar-size streams in mature forests. Janisch and colleagues (2012) reported mean pre-harvest CTD of 94%, compared to our 94%. Mean CTD immediately after harvest for their continuous 50-ft buffer, patch cut buffer, and clearcut (no buffer) treatments were 86%, 75%, and 53%. Post 1 mean CTD in our analogous 100%, FP, and 0% treatments was 86%, 74%, and 44%. Schuett-Hames and Stewart (2019 and unpublished data) measured canopy closure at one, three, five, and ten years after harvest in Type N reaches in western Washington with either no buffer or a 50-ft (15.2-m) no-cut buffer. Canopy closure in their clearcut category, analogous to our 0% treatment, was 12%, 14%, 37%, and 72% at one, three, five, and ten years after harvest, respectively, compared with 14%, 8%, 20%, and 58% at Post 1, Post 3, Post 5, and Post 9 in the unbuffered stream reaches of this study (**Figure 4-7**). Schuett-Hames and colleagues (2012) reported canopy closure within the 50-ft (15.2-m) buffer of 76%, 81%, 81%, and 90% at one, three, five, and ten years after harvest, respectively. This is very similar to the mean in our reaches with a 50-ft buffer; 87%, 77%, 80%, and 83% at Post 1, Post 3, Post 5, and Post 9, respectively. Rex and colleagues (2012) reported little recovery of shade (-3% to 17%) over three or four years after harvest, although this study was conducted in a sub-boreal forest, and MacDonald and colleagues (2003) reported little or no recovery of shade five years after harvest. This is consistent with our FP and 0% treatments, which showed a decline through the first four years after harvest (**Table 4-8**; **Figure 4-2**).

The FPHCP (WADNR 2005) assumed that rapid regrowth of understory vegetation along with shade provided by logging debris would minimize temperature increases after harvest and allow for quick return to pre-harvest conditions. The FPHCP assumptions were based largely on Summers (1982) who sampled stands across a range of years after clearcut harvest with no buffers and estimated angular canopy density would take 14 years to recover to 84%. Summers' (1982) empirical data showed canopy closure reaching 17% to 89% by nine or ten years after harvest and 79% to 84% by 15 years after harvest in his *Tsuga heterophylla* zone sites (lower elevation Cascades and Coast Range in the Willamette Valley, OR). Schuett-Hames and colleagues (2012 and unpublished data) measured canopy closure ten years after harvest of 90% in 50-ft wide buffers and 71% along unbuffered stream reaches consistent with Summers (1982). Our mean CC-1m in the 0% treatment was 59% at Post 9, within the range sampled by Summers (1982) and is likely on a trajectory to attain the projected 84% by year 14.

Tree mortality through windthrow was the primary cause of continued loss of riparian cover within the buffered reaches in the post-harvest period. Cumulative loss in basal area through Post 8 (the last year sampled) was 16%, 24%, and 51% in the REF, 100%, and FP buffers, respectively (see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report). This level of tree mortality is higher than Schuett-Hames and Stewart (2019 and unpublished data) where ten-year cumulative mortality was 11% and 33% in the unharvested reference sites and within 50 ft wide buffers, respectively. The relative ranking of mortality (highest in the FP, lowest in REF treatments) is consistent with Rollerson and colleagues (2009) who observed greater windthrow with decreasing area of stand/buffer. The slow rate of shade recovery, which did not begin to occur until five years after harvest (**Table 4-6**; **Figure 4-2**), may have been due to low rates of tree regeneration observed at five and eight years after harvest in all of our Hard Rock Study buffer treatment sites (see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report).

Mean riparian cover within the 100% treatment was higher than expected due to the presence of extensive unstable slope buffers within the OLYM and WIL2 sites resulting in buffers much wider than the 50 ft intended. The net effect was lower estimates of the loss in riparian cover than expected if a uniform 50 ft width buffer had been applied.

4-6.2. STREAM TEMPERATURE

4-6.2.1. Reliability of Temperature Data Analysis Methods

We expected that mean daily TR values from the stationarity analyses for the post-calibration period would be centered on zero, the MMTR values equitably split between positive and negative responses, and both the magnitude of MMTR values and any trend in TR over the course of the study would be small, relative to those calculated for the buffer treatment sites. These expectations were largely met for the first six post-calibration years in WIL2-FP and for the entire nine post-calibration years at OLYM-REF. An upward drift in daily TR in 2014 was seen in the WIL2-FP site and not at the OLYM-REF site suggesting that the change in the temperature relationship between the two WIL2 sites was due to a change at WIL2-FP rather than at WIL2-REF2 (**Figure 4-12**). Only 6.4% of reference site MMTRs exceeded 0.5°C in magnitude (14 positive, 9 negative) with $P < 0.05$ over the first six years of post-calibration at WIL2-FP plus the nine post-calibration years at OLYM-REF (**Table 4-12**). This indicates the WIL2-REF2 and OLYM-REF sites were stationary and suggests that the analysis method is capable of reliably detecting changes in MMTR of 0.5 to 1.0°C based on (1) the mean and distribution of the TR values and (2) the magnitude and distribution of MMTR values from OLYM-REF and from the first six years post-calibration at WIL2-FP. However, we caution the reader to focus on the overall pattern of temperature response across treatments, sites, and years rather than on single MMTR values.

Detecting buffer treatment effects on the Δ7DTR is a matter of both effect size (magnitude and direction of change) and variability in the effect size (consistency of effect) among the sites within each treatment. **Appendix Tables 4-17** and **4-18** suggest the minimum detectable treatment effect is approximately 0.8°C.

4-6.2.2. Buffer Treatment Effects

None of the three buffer treatments in our study was successful in preventing significant increases in July–August maximum daily stream temperature. However, there were substantial differences among treatments in their performance in terms of the magnitude and the duration of the temperature changes. The 100% and FP treatments were similar in the magnitude of the initial temperature response of approximately 1.0°C in the first two years after harvest but temperature soon thereafter returned to pre-harvest levels only in the 100% treatment while remaining elevated (Δ7DTR >0 and P <0.05) in the FP treatment over most of the study (**Figure 4-33**). Although there was no statistical difference (P >0.05) between the 100% and FP treatment effects in most of the post-harvest years (**Appendix Table 4-17**), the FP treatment effect was greater than zero (P <0.05) for seven of the nine years and equal to or greater than the 100% treatment effect for eight of the nine years suggesting that the FP treatment was less effective than the 100% treatment over the course of the study. The 100% and 0% treatments' trajectories were similar in that the temperature response decreased over time toward pre-harvest levels. However, the 0% warmed much more than the other treatments initially and was still elevated after nine years. The FP treatment was unique in that the temperature response varied little (0.5–1.2°C) over the post-harvest years and, like the 0% treatment, remained elevated nine years after harvest (**Table 4-18**).

The magnitude of temperature response immediately after harvest was consistent with studies of similar-size streams and buffers. There were three recent studies using similar methods on similar-size streams with which we can directly compare our initial temperature response. In a study that included sites in both marine sedimentary and basalt lithologies, Janisch and colleagues (2012) reported that mean July–August temperature increased by 0.61°C and 1.06°C in treatments analogous to our 100% and FP treatments, respectively. This is very similar to the 0.9°C and 1.1°C change in the 100% and FP buffer treatments at the F/N break in our study. Increases in mean July–August temperature of 1.5°C (Janisch *et al.* 2012) and 1.7°C (Gomi *et al.* 2006) have also been reported in unbuffered streams, lower than the 3.2°C increase we observed in the 0% treatments. Guenther and colleagues (2014) observed mean July–August temperature increases of 1.64 to 3.00°C at different locations within a partial retention harvest that resulted in a 14% decrease in canopy closure, comparable to our FP treatment. However, their stream had no harvest along the uppermost stream reach and a greater loss in riparian cover near the bottom of the harvest unit, the inverse of our FP treatment (little or no buffer in the upper reach and 50-ft [15.2-m] buffer in the lower portion), which may have affected the outcome. Bladon and colleagues (2018) reported increases in the median 7DADM (seven-day average daily maximum) temperature of 3.4°C and 3.9°C at two unbuffered sites underlain by resistant basalt lithologies, 0.6°C and 1.0°C from sites with buffers of 11 m and 12 m, respectively, underlain by friable lithologies, 0.8°C from a site with a 17 m buffer in mixed lithologies, and 2.4°C and 3.3°C from sites with buffers 8 m wide along 25% and 60% of the stream length, respectively, underlain by mixed lithologies. Although the analyses differed and their lithologies ranged from 100% resistant to 100% friable, the results from their unbuffered sites are similar to those of our 0% treatment and their sites with 11-17 m wide buffers are similar to both our 100% and FP treatments. Gravelle and Link (2007) observed temperature changes ranging from 0.5–3.6°C in three, south-facing, non-fish-bearing streams in the first year after clearcut harvest with no buffer. By their fourth year, temperatures at all sites declined and ranged from -1.5°C below pre-harvest to 1.4°C above. Stream substrate was mostly sand–large gravel-sized, similar to our sites.

The trajectory of post-harvest temperature response in the FP treatment differed markedly from both the 100% and 0% treatments. Summer stream temperature remained elevated in the FP treatment in most post-harvest years while both the 100% and 0% treatments decreased substantially after harvest; to pre-harvest conditions in the 100% and to less than 1.0°C in the 0% (**Table 4-18**). The temperature response over time at the individual sites within the 100% and 0% treatments was generally consistent among sites. July–August MMTR values decreased in the 100% treatment to near zero at three of the four sites within a few years (**Figure 4-13**) and in the 0% treatment MMTR steadily declined in all four sites (**Figure 4-15**). In contrast, summer MMTR response over time in the FP sites varied among the FP sites with July–August MMTR still elevated at Post 9 at the OLYM and WIL1 sites while the CASC site never showed significant summer warming after harvest (**Figure 4-14**). Gomi and colleagues (2006) observed elevated temperatures at least four years after harvest in similar sized streams with no buffers or a 10 m wide buffer in British Columbia and stream temperatures in subboreal watersheds after harvest remained elevated at least five years after harvest (MacDonald *et al.* 2003; Rex *et al.* 2012). The differences in response in the FP treatment sites were likely due to a combination of factors, including: increased riparian cover due to regrowth, especially in the unbuffered portion of the stream, and decreases in overstory cover in the buffered portions due to ongoing windthrow; site-specific differences in the proportion of stream with surface water and its location relative to the riparian buffers; stream aspect; and harvest-induced changes in discharge.

The loss of riparian cover was the dominant factor in the increased summer stream temperatures observed in the first four years after harvest. This is supported by the buffer treatment effects analysis where the 100% and 0% treatments, with the least and greatest loss of cover, respectively, exhibited the least and greatest temperature responses, respectively; by the lower July MMTR observed in wider buffers (**Figure 4-16**), which typically provided more shade (**Figure 4-7**); and by the negative correlations between July MMTR and the shade metrics in the first four post-harvest years (**Table 4-13**), when the temperature responses were greatest. The relative importance of cover declined over time as the mean riparian cover in all treatments began to increase in Post 5, as suggested by the weaker correlations between temperature change and riparian cover in the last five years of the study (**Table 4-13**), and the inconsistent, and usually poor, correlations between MMTR and riparian cover seen at the site scale (**Table 4-14**). We expected to see a negative correlation between July MMTR and canopy closure at the individual sites as the temperature response decreased and riparian shade increased over time, but this was the case in only three sites (**Table 4-14**). However, a correlation between MMTR and either CC-1m or CC-0m would have been difficult to detect at sites with a very small change in temperature (e.g., WIL1-100%, OLYM-FP, CASC-FP, OLYM-0%) or in canopy closure (e.g., OLYM-100%, WIL2-100%) compared to those with a greater change (e.g., WIL1-0% or CASC-0%). Perhaps a more precise measure of riparian cover (i.e., effective shade rather than canopy closure) would have been more effective at detecting these underlying relationships, but the similarity of the correlation coefficients between July MMTR and both CC-1m and effective shade in **Table 4-13** do not support this.

Although, the lack of site-scale correlation between temperature response and riparian cover is understandable, the positive correlation at WIL1-FP between July MMTR and canopy closure was unexpected. The ongoing windthrow within the buffered reach resulted in highly variable shade measurements and may have caused continued warming at the F/N break even as understory regrowth in the unbuffered stream reach raised site-wide, mean canopy closure values later in the study period. This is consistent with the higher July MMTR values observed at the F/N break (T1) than at either of the upstream locations in Post 7–Post 9 (**Figure 4-27**). Another possibility is that the loss of overstory cover to windthrow and its replacement by understory regrowth allowed stream warming through increased longwave radiation. Rex and colleagues (2012) suggested that an increase in longwave radiation and sensible heat to the stream from low vegetation was responsible for continued high water temperature in spite of cover provided by the regrowth of low vegetation. Although Klos and Link (2018) did not see an increase in longwave radiation from an understory-only canopy relative to a forested canopy, they suggested that stream heating within an understory-only canopy may be due to increased sensible heat because of the lower height of radiative emittance after removal of the forest overstory. Another potential factor is the southerly aspect of WIL1-FP. South-facing sites receive greater direct exposure and warming of the hillside due to short-wave radiation (Moore *et al.* 2005b) followed by long-wave radiation to the stream. This is consistent with the observed tendency for higher July MMTR in south-facing sites (**Figure 4-20**) and may have increased the sensitivity of this site to warming.

It is not clear what role stream discharge played in temperature response. Although we detected decreased summer flows in the 100% treatment sites and an increase in summer flows in both the FP and 0% treatments, we detected no significant correlation between July MMTR and measured discharge or the estimated change in discharge. It is possible the decreased summer flows in the 100% treatment sites increased their sensitivity to canopy loss or the higher flows in the FP and 0% treatments lowered their sensitivity. However, the effect of increased discharge volume is difficult to predict in streams of this size. Higher discharge volume requires more energy input to warm (Moore *et al.* 2005b) and higher velocity will decrease the time a water parcel is exposed in the harvest unit, leading to lower stream temperatures. However, our streams included multiple, sometimes lengthy, reaches with no surface flow in the summer. Immediately below these dry reaches, stream flow is comprised of lower temperature groundwater/hyporheic flow. If higher discharge volume increased the proportion of the stream with surface flow, there could be a greater exposure to shortwave and longwave radiation and the potential for greater warming, while the greater volume could lessen the sensitivity. Janisch and colleagues (2012) noted greater temperature response after harvest as a function of the length of wetted channel upstream. However, Janisch and colleagues (2012) observed this relationship only in a subset of streams with very fine-grained substrate, unlike our streams, and, presumably, less hyporheic exchange. Given the variability in length in our streams combined with the differences among sites in riparian cover and aspect, the lack of correlation between MMTR and length of wetted channel is not surprising.

The differences in the long-term response in summer temperature among the FP treatment sites was a combination of the pattern of surface flow at CASC-FP and the condition of the buffer above the F/N junction. The CASC-FP study site was the only site not to show significant summer warming at the F/N break in spite of the warming observed within the unbuffered reach upstream (**Figure 4-28**). This was likely due to a combination of a stream reach with no surface flow that stretched from the unbuffered portion of the stream into the well-shaded, buffered reach above the F/N break. The subsurface flow mitigated some or all of the temperature increase from the unbuffered reach upstream. The dense riparian buffer downstream experienced no decrease in shade due to harvest or windthrow (**Figure 4-5**). The result was cool water delivered into a well-shaded stream reach and no detectable summer warming at the F/N junction. In contrast, the dominant factor in the trajectory of the temperature response in the OLYM-FP and WIL1-FP sites was the effect of tree mortality on shade within the buffered reach. Cumulative post-harvest tree mortality was severe, 31% in OLYM-FP and 67% in WIL1-FP, and caused ongoing losses in riparian cover within the buffer over the post-harvest period leaving canopy openings within the buffer. These openings may have allowed increased exposure to direct shortwave radiation, even as mean site-level riparian cover increased due to understory regrowth in the unbuffered reaches. Shade losses within the buffer prevented the mitigation of stream warming observed in the unbuffered portions of both sites. These sites' southerly exposure may have increased the sensitivity of the streams to warming due to greater exposure to shortwave radiation or via longwave radiation from the sun-exposed uplands. Summer low flows increased at both sites post-harvest, but, as mentioned earlier, it is difficult to know the effect on stream temperature response without more detailed information on the extent, location, and duration of subsurface-only flow in each site. Our surveys of surface flow were done on only four occasions and for descriptive purposes. The specific methods were modified over the course of the study, so a detailed analysis of temperature change as a function of surface flow patterns is not possible. However, the general location of dry reaches was consistent across surveys.

4-6.2.2.a. Effects of including WIL2-FP in the analyses

The inclusion of the WIL2-FP site in the study had no substantive impact either to the buffer treatment effects analysis or to the conclusions drawn from this study. The WIL2-FP site was harvested in 2016, approximately seven years after the other sites and, as a result, there were only two post-harvest years of data collection available. Although the Forest Practices rules were followed, this site was completely buffered due to unstable slopes and resembled the 100% treatment more so than the FP treatment. The site was included in the analyses in spite of these concerns because it fell within the range of practices allowed under the current Forest Practices rules and there was a desire for more information regarding these rules. However, from a study design perspective an argument could be made to place it into the 100% treatment or even exclude it entirely.

The effect on the buffer treatment effects analysis was negligible. The temperature response at the F/N break in WIL2-FP was lower than the average for either the 100% or the FP treatment. If WIL2-FP were excluded from the study, the estimated FP treatment effect on Δ7DTR would be 0.3°C higher (1.4°C) in Post 1 and 0.1°C higher (1.0°C) in Post 2 (McIntyre *et al.* 2018, Chapter 7 – *Stream Temperature and Cover*) compared with the values reported in **Table 4-18**. These are relatively minor differences.

4-6.2.3. Seasonal Effects

Maximum daily temperature increased over much of the year at most monitored locations in all buffer treatment streams after harvest. In contrast to the stationarity tests where only 7.3% of MMTRs exceeded 0.5°C and were equitably split between positive (14) and negative (9) results, 53% of all MMTRs in the buffer treatment sites were greater than 0.5°C ($P < 0.05$) and less than

1% were less than -0.5°C (P <0.05), indicating a widespread increase in stream temperature. This increase persisted for at least one or more seasons through nine years post-harvest, as shown by the number and pattern of elevated (>0.5°C) and significant (P <0.05) MMTRs across all sites (**Figures 4-13** through **4-15**; **Appendix Tables 4-5** through **4-16**). Gomi and colleagues (2006), who used a similar method of calculating temperature response, saw a similar pattern of increasing daily maximum temperatures early in the spring that peaked in the late summer and extended into the fall in three of their four unbuffered streams and in their only stream with a 10-m buffer. This pattern persisted through all of their four post-harvest years. MacDonald and colleagues (2003) and Rex and colleagues (2012) observed higher stream temperatures after harvest throughout the ice-free season in sub-boreal, headwater streams with a variable retention buffer, with higher temperatures persisting in both studies through the three- to five-year monitoring record. In this study, fall and spring MMTR remained elevated over most of the study period and in the later years was often higher than in the summer. This may be related to lower canopy closure during the leaf off months (**Table 4-10**) than in the summer months, especially in the FP and 0% treatments where coniferous overstory was removed and, at least initially, replaced by deciduous vegetation. All of the treatment sites are rain-dominated and persistent snow cover was rare during the study suggesting that the effects of regenerating canopy cover on snow deposition or the timing of snowmelt are not important factors in the spring and fall temperature effects noted in the study.

4-6.2.4. Longitudinal Patterns in Temperature Response

The longitudinal patterns in temperature response varied within and among treatments and reflected buffer presence, location, and density, as well as patterns of surface flow in the stream. Where we measured higher temperatures post-harvest (July MMTR >1.0°), we nearly always observed cooling (lower July MMTR) downstream after flowing through a buffered or unharvested reach (**Figures 4-23**, **4-24**, **4-27**, **4-29**, **4-30**, and **4-32)**. July MMTR tended to be higher in unbuffered reaches of the FP treatment sites (**Figures 4-26**, **4-27**, and **4-29**) and near the lower reaches of the 0% treatment sites, but not necessarily at the F/N junction, (i.e., location T2; **Figures 4-29**, **4-30**, **4-31**, and **4-32**). The lower MMTR values at the F/N junction in the 0% treatment sites may be due to shade provided by the fish-bearing stream buffer immediately below this point as illustrated in the higher CC-1m values for location 1 in **Figure 4-6**. Whether the temperature effects were transmitted downstream into fish-bearing waters depended largely upon the magnitude of the temperature increase within the harvest unit. Little change (<0.7°C) was seen below the 100% treatment sites where the upstream (within harvest unit) MMTR was less than 1.0°C (**Table 4-16**). In the FP and 0% treatment sites, where the within upstream MMTRs were greater, the differences between the upstream and downstream locations were greater (i.e., greater cooling). A number of studies have documented decreases in maximum stream temperature flowing from an open-canopy reach into a shaded reach (Johnson 2004; Malcolm *et. al.* 2004; Torgersen *et al.* 1999; Zweiniecki and Newton 1999) and cooling of water already warmed in open reaches as it flowed into forested reaches downstream (Keith *et al.* 1998; Story *et al.* 2003). On average, when July MMTR was elevated (>0.5°C) within the harvest unit, downstream temperature change was 32% less than upstream. Davis and colleagues (2016) estimated a temperature change of 56% from the harvest unit to 300 m below the harvest unit for a collection of sites in the Oregon Coast Range. In their model, temperature change was a function of stream width, depth, and gradient, and, similar to our study, the magnitude of downstream cooling was directly related to the temperature change in the upstream harvest unit.

Zwieniecki and Newton (1999) also noted a rapid decrease in stream temperature below harvested reaches as water flowed through wider, more shaded buffers and suggested that warming due to harvest rapidly dissipates downstream. Gravelle and Link (2007), Kibler and colleagues (2013), and Bladon and colleagues (2017) detected increased temperatures in small headwater streams after harvest but not at downstream locations. In all three studies, the downstream locations were hundreds to more than one thousand meters downstream and included tributary inflows, so the lack of a temperature response downstream was likely due to a combination of the dilution effect from tributary inflow, hyporheic exchange, and groundwater inputs, as well as from greater riparian cover. Although our downstream locations were approximately 100 m below the treatment unit and did not include any tributary inflow, our results are not inconsistent with a rapid decrease in temperature from the harvest unit to downstream. Likely mechanisms for a reduction in the temperature response from the harvest unit to downstream in our sites were reduced direct shortwave radiation (Burton and Likens 1973) and groundwater input and hyporheic exchange (Brown *et al.* 1971; Story *et al.* 2003; Garner *et al.* 2014).

4-6.3. SUMMARY

Overall, our results indicate that none of the buffer treatments prevented a detectable decrease in riparian cover or an increase in maximum daily stream temperature. Higher summer temperatures were observed within the harvest unit at all sites and at the F/N junction at all but one site. The primary driver of higher post-harvest temperatures was the loss of riparian cover due to harvest and post-harvest tree mortality, but other factors were also noted. Temperature tended to increase more at sites with southerly aspects and at locations with narrow or no buffers. The response was somewhat less at specific locations situated immediately below reaches of stream with no surface flow. The trajectory of temperature response (decrease) in the post-harvest period was a function of vegetation regrowth and the corresponding increase in riparian cover, but stream temperature at sites with severe, repeated, windthrow-induced tree mortality within the riparian buffer after harvest did not return toward pre-harvest temperatures as expected. The relative effectiveness, from most effective to least, of the three buffer treatments, in terms of the magnitude and longevity of temperature change is 100%, FP, and 0%. The Δ7DTR in the 100% treatment initially was approximately 1.0°C but the treatment effect was near zero by Post 3. The FP treatment initial response was similar to the 100% treatment, but remained nearly unchanged throughout the post-harvest period. The 0% treatment response was higher, over 3.0°C initially, then declined to less than 1.0° by Post 9. Although the treatment effects were small, relative to early studies, our estimates were easily detected with a high degree of confidence following our methodology and, also, were similar in magnitude to estimates in the recent literature of the effects of contemporary forest practices.

This study was unable to tease out the specific mechanims driving stream temperature at individual sites. However, more precise measures of short-wave radiation reaching the stream, discharge monitoring at all sites (rather than only eight sites), and measurements of hyporheic exchange targeted to specific monitoring locations may have been useful.

4-7. REFERENCES

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4-8. APPENDIX TABLES

Appendix Table 4-1. Pair-wise comparisons with 95% confidence intervals of canopy and topographic density (CTD) for each combination of treatments for each post-harvest year.

Comparison	CTD			95% CI	
	Estimate	t-value	P-value	Lower	Upper
REF vs. 100%-Post 1	-0.49	-2.15	0.035	-0.95	-0.04
REF vs. 100%-Post 2	-0.57	-2.46	0.016	-1.02	-0.11
REF vs. 100%-Post 3	-0.53	-2.35	0.022	-0.98	-0.08
REF vs. 100%-Post 4	-0.63	-2.84	0.006	-1.07	-0.19
REF vs. 100%-Post 5	-0.61	-2.69	0.009	-1.06	-0.16
REF vs. FP-Post 1	-1.56	-6.40	<0.0001	-2.04	-1.07
REF vs. FP-Post 2	-1.76	-7.22	<0.0001	-2.24	-1.27
REF vs. FP-Post 3	-1.69	-7.03	<0.0001	-2.17	-1.21
REF vs. FP-Post 4	-1.67	-6.98	<0.0001	-2.14	-1.19
REF vs. FP-Post 5	-1.53	-6.31	<0.0001	-2.02	-1.05
REF vs. 0%-Post 1	-2.61	-11.92	<0.0001	-3.05	-2.17
REF vs. 0%-Post 2	-2.70	-12.23	<0.0001	-3.14	-2.26
REF vs. 0%-Post 3	-2.64	-12.10	<0.0001	-3.08	-2.21
REF vs. 0%-Post 4	-2.55	-11.79	<0.0001	-2.98	-2.12
REF vs. 0%-Post 5	-2.38	-10.90	<0.0001	-2.82	-1.95
100% vs. FP-Post 1	-1.06	-4.35	<0.0001	-1.55	-0.58
100% vs. FP-Post 2	-1.19	-4.90	<0.0001	-1.67	-0.71
100% vs. FP-Post 3	-1.16	-4.81	<0.0001	-1.64	-0.68
100% vs. FP-Post 4	-1.04	-4.34	<0.0001	-1.51	-0.56
100% vs. FP-Post 5	-0.92	-3.81	0.000	-1.41	-0.44
100% vs. 0%-Post 1	-2.12	-9.60	<0.0001	-2.56	-1.68
100% vs. 0%-Post 2	-2.13	-9.68	<0.0001	-2.57	-1.69
100% vs. 0%-Post 3	-2.11	-9.62	<0.0001	-2.55	-1.67
100% vs. 0%-Post 4	-1.92	-8.88	<0.0001	-2.35	-1.49
100% vs. 0%-Post 5	-1.77	-8.14	<0.0001	-2.21	-1.34
FP vs. 0%-Post 1	-1.06	-4.48	<0.0001	-1.52	-0.59
FP vs. 0%-Post 2	-0.94	-4.02	0.000	-1.41	-0.48
FP vs. 0%-Post 3	-0.95	-4.05	0.000	-1.41	-0.48
FP vs. 0%-Post 4	-0.88	-3.79	0.000	-1.35	-0.42
FP vs. 0%-Post 5	-0.85	-3.61	0.001	-1.32	-0.38

Appendix Table 4-2. Pair-wise comparisons with 95% confidence intervals of effective shade for each combination of treatments for each post-harvest year.

Comparison	Effective Shade			95% CI	
	Estimate	t-value	P-value	Lower	Upper
REF vs. 100%-Post 1	-0.56	-2.04	0.045	-0.01	0.00
REF vs. 100%-Post 2	-0.64	-2.33	0.023	-0.09	0.00
REF vs. 100%-Post 3	-0.57	-2.05	0.044	-0.02	0.00
REF vs. 100%-Post 4	-0.71	-2.62	0.011	-0.17	0.00
REF vs. 100%-Post 5	-0.75	-2.75	0.007	-0.21	0.00
REF vs. FP-Post 1	-1.44	-5.08	<0.0001	-0.88	0.00
REF vs. FP-Post 2	-1.77	-6.21	<0.0001	-1.20	0.00
REF vs. FP-Post 3	-1.80	-6.32	<0.0001	-1.24	0.00
REF vs. FP-Post 4	-1.80	-6.36	<0.0001	-1.23	0.00
REF vs. FP-Post 5	-1.65	-5.81	<0.0001	-1.08	0.00
REF vs. 0%-Post 1	-3.49	-11.99	<0.0001	-2.91	0.00
REF vs. 0%-Post 2	-3.70	-12.44	<0.0001	-3.11	0.00
REF vs. 0%-Post 3	-3.77	-12.56	<0.0001	-3.17	0.00
REF vs. 0%-Post 4	-3.62	-12.21	<0.0001	-3.03	0.00
REF vs. 0%-Post 5	-3.14	-11.14	<0.0001	-2.58	0.00
100% vs. FP-Post 1	-0.89	-3.04	0.003	-0.31	0.00
100% vs. FP-Post 2	-1.13	-3.90	0.000	-0.55	0.00
100% vs. FP-Post 3	-1.24	-4.23	<0.0001	-0.65	0.00
100% vs. FP-Post 4	-1.09	-3.79	0.000	-0.52	0.00
100% vs. FP-Post 5	-0.90	-3.13	0.003	-0.33	0.00
100% vs. 0%-Post 1	-2.94	-9.84	<0.0001	-2.34	0.00
100% vs. 0%-Post 2	-3.06	-10.11	<0.0001	-2.46	0.00
100% vs. 0%-Post 3	-3.20	-10.45	<0.0001	-2.59	0.00
100% vs. 0%-Post 4	-2.91	-9.66	<0.0001	-2.31	0.00
100% vs. 0%-Post 5	-2.40	-8.36	<0.0001	-1.82	0.00
FP vs. 0%-Post 1	-2.05	-6.66	<0.0001	-1.44	0.00
FP vs. 0%-Post 2	-1.93	-6.18	<0.0001	-1.31	0.00
FP vs. 0%-Post 3	-1.96	-6.25	<0.0001	-1.34	0.00
FP vs. 0%-Post 4	-1.82	-5.81	<0.0001	-1.19	0.00
FP vs. 0%-Post 5	-1.49	-5.00	<0.0001	-0.90	0.00

Appendix Table 4-3. Pair-wise comparisons with 95% confidence intervals of canopy closure-1m for each combination of treatments for each post-harvest year.

Comparison	Canopy Closure–1m			95% CI	
	Estimate	t-value	P-value	Lower	Upper
REF vs. 100% Post 1	-0.55	-0.99	0.326	-1.64	0.55
REF vs. 100% Post 2	-0.55	-1.04	0.299	-1.59	0.49
REF vs. 100% Post 3	-1.01	-1.89	0.061	-2.06	0.05
REF vs. 100% Post 4	-0.60	-1.15	0.253	-1.62	0.43
REF vs. 100% Post 5	-0.38	-0.73	0.466	-1.41	0.65
REF vs. 100% Post 6	-0.31	-0.60	0.550	-1.31	0.70
REF vs. 100% Post 7	-0.11	-0.19	0.850	-1.23	1.01
REF vs. 100% Post 8	-0.57	-1.02	0.308	-1.67	0.53
REF vs. 100% Post 9	-0.32	-0.62	0.537	-1.36	0.71
REF vs. FP Post 1	-1.44	-2.74	0.007	-2.47	-0.40
REF vs. FP Post 2	-1.52	-3.00	0.003	-2.51	-0.52
REF vs. FP Post 3	-2.09	-4.07	<0.0001	-3.10	-1.08
REF vs. FP Post 4	-1.69	-3.42	0.001	-2.67	-0.71
REF vs. FP Post 5	-1.49	-2.93	0.004	-2.50	-0.48
REF vs. FP Post 6	-1.24	-2.47	0.015	-2.23	-0.25
REF vs. FP Post 7	-1.09	-1.99	0.048	-2.16	-0.01
REF vs. FP Post 8	-1.35	-2.48	0.014	-2.42	-0.27
REF vs. FP Post 9	-0.86	-1.66	0.099	-1.89	0.17
REF vs. 0% Post	-4.82	-8.12	<0.0001	-6.00	-3.65
REF vs. 0% Post 2	-5.46	-8.58	<0.0001	-6.71	-4.20
REF vs. 0% Post 3	-5.46	-8.61	<0.0001	-6.72	-4.21
REF vs. 0% Post 4	-5.21	-8.35	<0.0001	-6.44	-3.98
REF vs. 0% Post 5	-3.65	-6.70	<0.0001	-4.73	-2.57
REF vs. 0% Post 6	-3.83	-7.08	<0.0001	-4.89	-2.76
REF vs. 0% Post 7	-3.41	-6.31	<0.0001	-4.48	-2.34
REF vs. 0% Post 8	-2.02	-3.82	0.000	-3.07	-0.98
REF vs. 0% Post 9	-1.62	-3.22	0.002	-2.61	-0.63
100% vs. 0% Post 1	-4.28	-7.10	<0.0001	-5.47	-3.09
100% vs. 0% Post 2	-4.91	-7.57	<0.0001	-6.19	-3.63
100% vs. 0% Post 3	-4.46	-7.05	<0.0001	-5.70	-3.21
100% vs. 0% Post 4	-4.61	-7.28	<0.0001	-5.87	-3.36
100% vs. 0% Post 5	-3.27	-5.80	<0.0001	-4.38	-2.15
100% vs. 0% Post 6	-3.52	-6.25	<0.0001	-4.63	-2.41
100% vs. 0% Post 7	-3.30	-5.75	<0.0001	-4.43	-2.17
100% vs. 0% Post 8	-1.45	-2.69	0.008	-2.51	-0.39
100% vs. 0% Post 9	-1.30	-2.47	0.015	-2.34	-0.26
100% vs. FP Post 1	-0.89	-1.66	0.099	-1.95	0.17
100% vs. FP Post 2	-0.97	-1.85	0.066	-1.99	0.06
100% vs. FP Post 3	-1.08	-2.11	0.036	-2.09	-0.07
100%vs. FP Post 4	-1.10	-2.15	0.033	-2.10	-0.09
100%vs. FP Post 5	-1.11	-2.09	0.038	-2.16	-0.06
100%vs. FP Post 6	-0.94	-1.77	0.079	-1.98	0.11
100%vs. FP Post 7	-0.98	-1.69	0.094	-2.12	0.17
100% vs. FP Post 8	-0.78	-1.40	0.163	-1.87	0.32
100% vs. FP Post 9	-0.54	-0.99	0.323	-1.61	0.53
FP vs. 0% Post 1	-3.39	-5.89	<0.0001	-4.52	-2.25
FP vs. 0% Post 2	-3.94	-6.28	<0.0001	-5.18	-2.70
FP vs. 0% Post 3	-3.37	-5.51	<0.0001	-4.58	-2.16
FP vs. 0% Post 4	-3.52	-5.73	<0.0001	-4.73	-2.30
FP vs. 0% Post 5	-2.16	-3.92	0.000	-3.25	-1.07
FP vs. 0% Post 6	-2.59	-4.65	<0.0001	-3.68	-1.49
FP vs. 0% Post 7	-2.32	-4.20	<0.0001	-3.42	-1.23
FP vs. 0% Post 8	-0.67	-1.29	0.199	-1.71	0.36
FP vs. 0% Post 9	-0.76	-1.45	0.150	-1.79	0.28

Appendix Table 4-4. Pair-wise comparisons with 95% confidence intervals of canopy closure-0m for each combination of treatments for each post-harvest year.

Comparison	Canopy Closure–0m			95% CI	
	Estimate	t-value	P-value	Lower	Upper
REF vs. 100% Post 1	-0.65	-0.85	0.398	-2.17	0.87
REF vs. 100% Post 2	-0.29	-0.39	0.695	-1.73	1.15
REF vs. 100% Post 3	-0.59	-0.84	0.404	-1.98	0.80
REF vs. 100% Post 4	-0.15	-0.23	0.821	-1.49	1.18
REF vs. 100% Post 5	-0.13	-0.19	0.848	-1.52	1.25
REF vs. 100% Post 6	0.72	0.87	0.388	-0.92	2.36
REF vs. 100% Post 7	1.13	1.21	0.230	-0.72	2.98
REF vs. 100% Post 8	0.41	0.50	0.616	-1.21	2.04
REF vs. 100% Post 9	0.14	0.21	0.836	-1.23	1.52
REF vs. FP Post 1	-1.43	-1.82	0.072	-3.00	0.13
REF vs. FP Post 2	-1.72	-2.36	0.020	-3.16	-0.28
REF vs. FP Post 3	-2.01	-2.81	0.006	-3.43	-0.60
REF vs. FP Post 4	-1.07	-1.54	0.126	-2.45	0.31
REF vs. FP Post 5	-1.87	-2.62	0.010	-3.29	-0.46
REF vs. FP Post 6	-1.16	-1.49	0.139	-2.70	0.38
REF vs. FP Post 7	-0.90	-1.10	0.275	-2.51	0.72
REF vs. FP Post 8	-1.13	-1.40	0.163	-2.72	0.46
REF vs. FP Post 9	-0.48	-0.63	0.530	-1.98	1.03
REF vs. 0% Post 1	-3.57	-4.48	<0.0001	-5.14	-1.99
REF vs. 0% Post 2	-3.20	-4.23	<0.0001	-4.70	-1.70
REF vs. 0% Post 3	-3.00	-3.99	0.000	-4.49	-1.52
REF vs. 0% Post 4	-2.38	-3.29	0.001	-3.81	-0.95
REF vs. 0% Post 5	-2.84	-3.86	0.000	-4.30	-1.38
REF vs. 0% Post 6	-2.33	-3.04	0.003	-3.84	-0.82
REF vs. 0% Post 7	-2.78	-3.60	0.000	-4.31	-1.25
REF vs. 0% Post 8	-2.06	-2.62	0.010	-3.61	-0.51
REF vs.0% Post 9	-1.66	-2.27	0.025	-3.11	-0.21
100% vs. FP Post 1	-0.78	-1.10	0.273	-2.19	0.62
100% vs. FP Post 2	-1.44	-2.09	0.038	-2.79	-0.08
100% vs. FP Post 3	-1.42	-2.16	0.033	-2.73	-0.12
100% vs. FP Post 4	-0.92	-1.37	0.174	-2.25	0.41
100% vs. FP Post 5	-1.74	-2.52	0.013	-3.10	-0.38
100% vs. FP Post 6	-1.88	-2.24	0.027	-3.53	-0.22
100% vs. FP Post 7	-2.03	-2.12	0.036	-3.92	-0.13
100% vs. FP Post 8	-1.54	-1.86	0.064	-3.17	0.09
100% vs. FP Post 9	-0.62	-0.83	0.409	-2.11	0.87
100% vs. 0% Post 1	-2.91	-4.07	<0.0001	-4.33	-1.50
100% vs. 0% Post 2	-2.92	-4.07	<0.0001	-4.33	-1.50
100% vs. 0% Post 3	-2.41	-3.45	0.001	-3.80	-1.03
100% vs. 0% Post 4	-2.23	-3.18	0.002	-3.61	-0.84
100% vs. 0% Post 5	-2.70	-3.81	0.000	-4.11	-1.30
100% vs. 0% Post 6	-3.04	-3.70	0.000	-4.67	-1.42
100% vs. 0% Post 7	-3.91	-4.26	<0.0001	-5.73	-2.10
100% vs. 0% Post 8	-2.47	-3.06	0.003	-4.06	-0.87
100% vs. 0% Post 9	-1.81	-2.49	0.014	-3.24	-0.37
FP vs. 0% Post 1	-2.13	-2.88	0.005	-3.60	-0.67
FP vs. 0% Post 2	-1.48	-2.06	0.041	-2.90	-0.06
FP vs. 0% Post 3	-0.99	-1.40	0.165	-2.39	0.41
FP vs. 0% Post 4	-1.30	-1.80	0.073	-2.73	0.12
FP vs. 0% Post 5	-0.96	-1.33	0.187	-2.40	0.47
FP vs. 0% Post 6	-1.17	-1.52	0.132	-2.69	0.36
FP vs. 0% Post 7	-1.89	-2.37	0.019	-3.46	-0.31
FP vs. 0% Post 8	-0.93	-1.18	0.241	-2.49	0.63
FP vs. 0% Post 9	-1.18	-1.50	0.136	-2.74	0.38

Appendix Table 4-5. Mean monthly temperature response (MMTR) values for OLYM-100% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
OLYM-100%	T4	15	Pre 1	0.0	-0.1	-0.2	-0.2	-0.3	-0.1	0.2	0.3	0.3	-0.1	0.0	0.0
			Post 1	0.2	0.1	-0.1	-0.4	0.0	0.6	1.1	0.2	0.0	0.1	0.0	-0.1
			Post 2	0.5	0.1	0.2	0.2	0.2	0.1	0.2	0.2	0.3	0.0	0.0	0.1
			Post 3	0.0	0.3	0.5	0.1	-0.1	0.0	0.0	0.1	0.2	0.1	-0.1	-0.3
			Post 4	0.3	0.5	0.5	0.1	-0.1	0.0	-0.1	0.2	0.3	0.0	0.2	0.4
			Post 5	0.2	0.4	0.5	0.2	0.5	0.6	0.3	0.4	0.4	0.1	0.1	0.0
			Post 6	0.4	0.4	0.3	0.4	0.6	0.6	0.6	0.4	0.4	0.4	0.2	0.2
			Post 7			0.3	0.1	0.0	0.0	-0.2	-0.3	0.1	0.1	-0.1	0.0
			Post 8	0.1	0.3	0.6	0.2	0.1	0.2	0.1	0.1	0.0	0.1	0.3	0.3
			Post 9			-0.2	0.3	0.2	-0.3	-0.2	-0.1	0.4	-0.1	-0.2	
OLYM-100%	RB3	121	Pre 2	0.2	0.2	0.5	0.0	-0.2	-0.1	0.1	0.2	0.3	-0.1	-0.5	-0.1
			Pre 1	0.1	0.0	0.1	-0.3	-0.2	-0.2	-0.1	0.0	-0.1	0.0	-0.1	0.0
			Post 1	0.6	0.8	0.6	0.4	0.3	0.2	0.2	0.0	0.1	0.1	0.2	0.2
			Post 2	0.7	0.3	0.6	0.5	0.4	0.3	0.2	0.0	0.2	0.4	0.2	0.2
			Post 3	0.6	0.7	1.0	0.7	0.4	0.4	0.2	0.3	0.4	0.4	0.3	0.3
			Post 4	0.6	0.9	0.8	0.8	0.6	0.6	0.5	0.6	0.4	0.3	0.5	0.8
			Post 5	0.4	0.5	0.5	0.6	0.5	0.5	0.5	0.4	0.4	0.2	0.2	0.2
			Post 6	0.8	0.9	0.6	0.5	0.7	0.6	0.6	0.6	0.5	0.6	0.8	0.3
			Post 7	0.7	0.7	0.9	0.2	0.0	0.1	0.2	0.0	0.4	0.4	0.4	0.5
			Post 8	0.7	-0.2	0.0	0.0	-0.1	0.0	0.2	0.5	0.7	0.5	0.6	0.6
			Post 9			0.7	0.6	0.5	0.1	-0.4	-0.3	0.0	0.0	0.3	
OLYM-100%	RB2	153	Pre 2	0.2	-0.1	0.4	0.0	0.0	0.2	0.3	0.1	0.1	-0.2	-0.7	0.0
			Pre 1	0.3	0.5	0.0	0.1	0.3	0.0	-0.1	-0.3	-0.2	-0.2	-0.3	-0.1
			Post 1			0.2	0.1	0.3	0.0	-0.4	0.1	0.0	0.3		
			Post 2	0.6	0.2	0.6	0.5	0.4	0.3	0.1	-0.1	0.0	0.3	0.1	0.2
			Post 3	0.2	0.5	0.6	0.2	0.2	-0.1	-0.3	-0.1	0.1	0.1	0.1	-0.1
			Post 4	0.4	0.8	0.7	0.7	0.5	0.3	0.0	0.3	-0.1	0.4	0.4	0.7
			Post 5	0.5	0.5	0.6	0.6	0.8	0.9	0.7	0.5	0.6	0.5	0.5	0.2
			Post 6	1.4	1.6	1.2	1.0	0.9	0.8	0.6	0.7	0.7	0.8	1.1	0.6
			Post 7	1.1	1.0	1.0	1.1					0.4	0.3	0.0	0.3
			Post 8	1.0	0.2	0.0	-0.7	0.3	0.5	0.2	1.0	1.1	0.8	0.9	1.2
			Post 9			0.7	0.3	0.7	0.4	0.6	0.4	0.7	0.2	0.2	
OLYM-100%	RB1	221	Pre 2	0.2	0.2	0.6	0.1	0.0	-0.1	0.2	0.0	0.3	-0.1	-0.4	-0.1
			Pre 1	0.2	0.2	-0.3	-0.4	0.0	-0.3	0.1	0.1	0.1	0.0	0.0	-0.3
			Post 1	0.6	0.6	0.7	0.1	0.2	0.5	0.5	0.8	0.1	-0.3	0.0	0.0
			Post 2	0.6	0.5	0.7	0.4	0.3	0.3	0.6	0.5	0.4	0.4	0.3	0.3
			Post 3	0.7	1.0	1.1	0.5	0.2	0.0	-0.4			0.5	0.2	0.2
			Post 4	0.6	0.8	1.2	0.9	0.7	0.6	0.6	0.8	0.6	0.5	0.6	0.7
			Post 5	0.7	0.6	0.7	0.8	1.3	1.5	0.9	0.8	1.3	1.0	0.9	0.5
			Post 6	1.1	1.1	1.1	1.1	1.0	1.0	1.3	1.2	1.1	0.9	0.7	0.5
			Post 7	0.9	0.8	0.9	1.0	0.6	0.6	0.8	0.6	0.6	0.5	0.1	0.3
			Post 8	0.8	0.4	0.5	0.4	0.6	0.5	0.6	0.9	0.5	0.3	0.5	0.4
			Post 9			1.5	0.6	0.8	0.6	0.8	0.6	0.9	0.2	0.1	

Appendix Table 4-5 (continued). Mean monthly temperature response (MMTR) values for OLYM-100% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
OLYM-100%	T3	270	Pre 1	0.1	0.2	0.3	0.0	0.1	0.0	0.3	0.2	0.1	-0.2	-0.2	-0.2
			Post 1	0.4	0.3	0.4	0.2	0.5	0.8	0.8	0.4	0.4	0.2	-0.1	0.3
			Post 2			0.4	0.4	0.3	0.5	0.7	0.6	0.6			
			Post 3	0.0	0.4	0.9	0.2	0.3	0.6	0.7	0.7	0.8	-0.1	-0.2	-0.6
			Post 4			1.0	0.8	0.7	0.5	0.9	1.4	0.7	-0.4		
			Post 6				1.0	1.2	1.3	1.4	1.4	0.9	0.6	0.4	
			Post 7	0.5	0.5	0.6	1.1	0.7	0.9	1.3	1.4	0.4	0.1	-0.1	0.1
			Post 8	0.1	0.4	0.8	0.6	0.6	0.7	0.7	0.9	0.4	-0.1	0.1	0.1
OLYM-100%	LB1	293	Pre 2	0.2	0.2	0.5	0.0	-0.2	-0.1	0.1	0.2	0.3	-0.1	-0.5	-0.1
			Pre 1	0.1	0.0	0.1	-0.3	-0.2	-0.2	-0.1	0.0	-0.1	0.0	-0.1	0.0
			Post 1			-0.4	-0.1	0.1	0.6	0.8	0.4	0.2	0.1	0.3	
			Post 2	0.6	0.2	0.5	0.1	-0.1	0.1	0.5	0.3	0.6	0.1	0.0	-0.1
			Post 3	0.6	0.7	1.0	0.7	0.4	0.4	0.2	0.3	0.4	0.4	0.3	0.3
			Post 4	0.6	0.9	0.8	0.8	0.6	0.6	0.5	0.6	0.4	0.3	0.5	0.8
			Post 5	0.4	0.5	0.5	0.6	0.5	0.5	0.5	0.4	0.4	0.2	0.2	0.2
			Post 6	0.8	0.9	0.6	0.5	0.7	0.6	0.6	0.6	0.5	0.6	0.8	0.3
			Post 7	0.7	0.7	0.9	0.2	0.0	0.1	0.2	0.0	0.4	0.4	0.4	0.5
			Post 8	0.6	-0.2	0.1	0.1	-0.1	0.0	0.0	0.2	0.5	0.4	0.5	0.4
			Post 9			0.7	0.6	0.5	0.1	-0.4	-0.3	0.0	0.0	0.3	
OLYM-100%	T2	371	Pre 2	0.0	0.0	0.4	-0.2	-0.2	-0.1	0.0	-0.1	0.1	-0.2	-0.5	0.2
			Pre 1	0.0	0.4	0.1	-0.1	0.0	-0.1	0.3	0.3	0.1	-0.1	0.0	-0.1
			Post 1	0.3	0.2	0.4	0.2	0.2	0.4	0.5	0.4	0.3	0.2	0.0	-0.1
			Post 2	0.9		0.5	0.3	0.3	0.4	0.6	0.5	0.4	0.2	0.2	0.3
			Post 3	0.2	0.4	0.8	0.4	0.3	0.5	0.4	0.5	0.5	0.3	0.2	-0.1
			Post 4	0.4	0.7	0.6	0.5	0.4	0.4	0.5	0.9	0.5	0.3	0.5	0.7
			Post 5	0.2	0.3	0.5	0.4	0.6	0.8	0.8	0.9	0.7	0.3	0.3	0.0
			Post 6			0.5	0.5	0.8	0.8	0.7	0.5	0.5			
			Post 7	0.4	0.5	0.5	0.6	0.4	0.7	1.2	1.1	0.1	0.0	0.0	0.3
			Post 8	1.3	0.9	0.9	0.2	0.1	0.2	0.2	0.5	0.1	0.3	0.5	0.7
OLYM-100%	T1	452	Pre 2	0.3	0.0	0.6	0.1	-0.1	-0.1	0.2	-0.1	0.0	-0.2	-0.7	0.3
			Pre 1	0.0	0.3	0.0	-0.3	-0.1	-0.2	0.1	0.2	0.0	-0.2	-0.1	-0.3
			Post 1	0.3	0.4	0.4	0.1	0.3	0.5	0.6	0.3	0.3	0.2	0.1	0.0
			Post 2	0.5	0.1	0.6	0.4	0.2	0.4	0.5	0.5	0.5	0.2	0.1	0.0
			Post 3	0.2	0.4	0.8	0.3	0.2	0.4	0.4	0.7	0.7	0.1	0.0	-0.2
			Post 4	0.3	0.6	0.6	0.4	0.4	0.3	0.4	0.8	0.5	0.2	0.4	0.5
			Post 5	0.1	0.4	0.3	0.3	0.9	0.8	0.8	0.8	0.7	0.4	0.4	-0.1
			Post 6	1.1	1.0	0.7	0.7	0.9	1.0	1.0	1.0	0.8	0.7	0.6	0.3
			Post 7	1.2	1.6	1.4	2.1	1.6	1.8	1.3	1.9	1.4	1.2	1.1	1.1
			Post 8	0.3	-0.2	0.3	1.8	1.8	2.1	2.3	2.6	2.0	0.3	0.5	0.4
			Post 9			0.6	0.3	0.6	0.3	0.5	0.6	0.7	0.2	0.1	

Appendix Table 4-6. Mean monthly temperature response (MMTR) values for WIL1-100% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL1-100%	T4	3	Pre 2	0.2	0.2	0.3	-0.1	-0.6	-0.4	0.8	0.6	-0.1	-0.7	-0.1	0.0
			Pre 1	0.0	0.3	0.0	-0.3	0.5	0.6	1.4	1.8	1.9	0.2	-0.2	0.1
			Post 1	1.1	1.9	1.7	1.3	1.4	2.1	3.6	2.9	2.9	1.1	0.6	0.9
			Post 2	1.0	0.9	1.0	0.9	0.6	0.4	2.2	2.7	2.1	0.9	0.1	0.6
			Post 3	0.7	0.7	0.9	0.7	0.6	0.7	1.7	2.5	2.4	0.6	0.6	0.4
			Post 4	0.8	0.9	0.9	0.9	0.3	0.0	0.2			0.3	0.3	0.6
			Post 5	1.0	1.2	1.2	0.3	0.8	1.2	1.1	1.7	1.1	0.6	0.5	0.5
			Post 6	1.2	1.4	1.3	1.2	0.0					0.6	0.4	0.8
			Post 7	0.9	1.1	1.1	1.0	0.9	1.1	1.8	1.6	1.0	0.3	0.0	0.6
			Post 8			1.6	1.4	1.0	1.0	0.7	1.3	0.3	-0.1	0.3	0.5
			Post 9	0.5	0.9	1.0	0.6	0.5	0.5	1.0	1.6	1.5	0.4	0.3	0.6
WIL1-100%	LB2	9	Pre 2	-0.1	0.1	0.1	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
			Pre 1	0.0	0.0	0.0	-0.2	-0.3	-0.4	-0.3	-0.2	-0.2	0.1	0.1	0.2
			Post 1	0.1	0.3	0.5	0.6	0.7	0.7	0.9	0.8	0.8	0.3	0.0	-0.3
			Post 2	0.2	0.2	0.5	0.5	0.5	0.3	0.6	0.8	0.4	0.3	0.1	0.1
			Post 3	0.3	0.4	0.5	0.5	0.4	0.3	0.5	0.5	0.4	0.4	0.4	0.4
			Post 4	0.2	0.4	0.5	0.5	0.4	0.2	0.2	0.4	0.6	0.3	0.1	0.2
			Post 5	0.6	0.6	0.6	0.5	0.3	0.0	0.0	-0.1	-0.1	0.2	0.4	0.5
			Post 6	0.4		0.7	0.7	0.4	0.2	0.2	0.3	0.3	0.4	0.4	0.7
			Post 7	0.9	1.0	1.1	0.7	0.5	0.2	0.0	-0.1	0.2	0.2	0.7	0.8
			Post 8			1.1	0.8	0.3	0.2	0.0	-0.2	0.3	0.3	0.4	0.6
			Post 9	0.8	1.0	0.9	0.8	0.4	-0.1	-0.4	-0.5	-0.3	0.2	0.5	0.9
WIL1-100%	LB1	100	Pre 2	-0.1	0.1	0.1	0.1	0.0	-0.1	-0.2	0.1	0.1	-0.2	-0.1	-0.1
			Pre 1	0.1	0.1	0.3	0.2	-0.1	-0.4	-0.7	-0.7	-0.6	-0.1	0.1	0.0
			Post 1	0.2	0.4	0.6	0.4	0.1	-0.2	-0.4	-0.1	0.1	0.0	0.3	0.2
			Post 2			0.5	0.6	0.5	0.2	-0.1	0.0	-0.1			
			Post 3	0.4	0.5	0.5	0.7	0.5	0.1	-0.2	0.0	0.0	0.1	0.5	0.4
			Post 4	0.4	0.6	0.5	0.5	0.5	0.4	0.1	0.1	0.0	0.1	0.4	0.5
			Post 5	0.7	0.8	0.7	0.7	0.6	0.3	0.3	0.3	0.3	0.3	0.5	0.7
			Post 6	0.7	0.8	0.9	0.9	0.9					0.6	0.5	0.8
			Post 7	0.7	0.9	1.1	1.2	0.9	0.7	0.8	0.9	1.0		0.6	0.6
			Post 8			1.5	1.0	0.6	0.8	0.8	1.0	1.0	0.7	0.5	0.5
			Post 9	0.7	0.8	0.8	1.0	0.7	0.4	0.2	0.3	0.3	0.0	0.3	0.7
WIL1-100%	T3	228	Pre 2	-0.1	0.1	0.3	0.0	0.0	-0.1	0.2	0.2	0.2	0.0	-0.1	-0.1
			Pre 1	0.2	0.2	0.3	0.1	0.2	0.3	1.2	1.2	0.5	0.1	0.0	0.2
			Post 1	0.3	0.4	0.8	0.7	0.9	1.1	1.6	1.4	1.3	0.5	0.0	0.0
			Post 2	0.4	0.4	0.7	0.7	0.6	0.5	1.0	0.9	0.6	0.2	0.1	0.2
			Post 3	0.6	0.5	0.7	0.8	0.8	0.6	0.9	0.9	0.8	0.5	0.5	0.4
			Post 4	0.5	0.7	0.7	0.7	0.7	0.3	0.4	0.6	0.4	0.2	0.3	0.4
			Post 5	1.0	1.1	0.9	0.8	0.6	0.4	0.5	0.4	0.4	0.3	0.6	0.9
			Post 6	0.8	0.8	0.9	1.1	0.7	0.4	0.4	0.5	0.5	0.6	0.5	0.8
			Post 7	0.7	0.9	1.0	0.9	0.6	0.3	0.3	0.4	0.6	0.6	0.6	0.6
			Post 8			1.1	0.9	0.4	0.2	0.3	0.3	0.5	0.3	0.4	0.4
			Post 9	0.7	0.9	1.0	0.9	0.6	0.2	0.1	0.1	0.2	0.2	0.5	0.6

Appendix Table 4-6 (continued). Mean monthly temperature response (MMTR) values for WIL1-100% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL1-100%	T2	328	Pre 2	-0.1	0.0	0.0	-0.1	-0.1	-0.2	0.0	-0.1	-0.1	-0.1	-0.1	-0.1
			Pre 1	0.1	0.1	0.1	0.1	0.0	0.1	0.4	0.2	0.3	0.0	0.0	0.1
			Post 1	0.3	0.5	0.7	0.4	0.5	0.4	0.9	0.8	0.9	0.5	0.2	0.2
			Post 2	0.5	0.6	0.7	0.7	0.5	0.4	0.8	1.0	0.8	0.5	0.4	0.4
			Post 3	0.6	0.6	0.7	0.8	0.7	0.5	0.7	0.8	0.7	0.7	0.6	0.6
			Post 4			0.7	0.6	0.7	0.3	0.3	0.5	0.4	0.3		
			Post 5	0.8	0.9	0.7		0.5	0.3	0.4	0.2	0.2	0.4	0.6	0.8
			Post 6			0.8	0.9	0.4	0.2	0.1	0.1	0.0	0.0		
			Post 7	0.8	0.8	1.0	0.7	0.5	0.2	0.0	0.0	0.2	0.3	0.7	0.7
			Post 8			1.0	0.7	0.3	0.1	0.0	-0.2	0.1	0.3	0.5	0.5
			Post 9	0.8	0.8	0.9	0.8	0.4	0.1	-0.1	-0.2	-0.2	0.1	0.5	0.7
WIL1-100%	D100	668	Pre 1	0.0	0.0	0.0	0.0	-0.1	-0.1	0.1	0.0	0.0	0.0	0.0	0.0
			Post 1	0.2	0.2	0.2	0.1	0.2	0.2	0.3	0.3	0.2	0.2	0.2	0.2
			Post 2	0.5	0.5	0.5	0.3	0.2	0.1	0.3	0.3	0.3	0.4	0.4	0.4
			Post 3	0.5	0.5	0.6	0.5	0.4	0.2	0.2	0.2	0.1	0.4	0.6	0.5
			Post 4	0.3	0.5	0.5		0.2	0.0	0.0	-0.1	-0.2	0.1	0.3	0.5
			Post 5	0.6	0.7	0.5	0.3	0.2	0.0	0.0	-0.2	-0.1	0.2	0.4	0.6
			Post 6	0.5		0.5	0.5	0.2	0.3	0.5	0.6	0.6	0.4	0.5	0.7
			Post 7	0.6	0.6	0.7	0.3	0.2	-0.1	-0.4	-0.4	-0.2	0.2	0.6	0.6
			Post 8			0.5	0.4	0.0	-0.2	-0.4	-0.7	-0.2	0.3	0.5	0.5
			Post 9	0.7	0.8	0.6	0.6	0.3	-0.1	-0.3	-0.4	-0.4	0.0	0.5	0.7

Appendix Table 4-7. Mean monthly temperature response (MMTR) values for WIL2-100% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL2-100%	LB2	69	Pre 2	0.1	0.1	0.1	0.1	0.1	0.2	0.1	0.1	0.0	0.0	-0.1	0.0
			Pre 1	0.0	0.1	-0.2	-0.2	-0.1	-0.3	-0.4	-0.1	-0.1	-0.2	-0.2	-0.1
			Post 1	0.4	0.6	0.8	0.5	0.2	0.4	0.7	0.6	0.7	0.1	0.1	0.2
			Post 2	0.7	0.7	0.7	0.8	0.6	0.3	0.5	0.6	0.5	0.3	0.3	0.5
			Post 3	0.6	0.7	0.7	0.7	0.4	0.3	0.2	0.4	0.4	0.3	0.4	0.5
			Post 4	0.6	0.7	0.8	0.6	0.3	0.1	0.1	0.3	0.3	0.2	0.4	0.6
			Post 5	0.7	0.8	0.9	0.8	0.4	0.2	0.1	0.3	0.5	0.2		0.5
			Post 6	0.9	1.0	1.0	0.9	0.4	0.4	0.4	0.6	0.7	0.6	0.7	1.0
			Post 7	1.1	1.3	1.4	1.0	0.7	0.5	0.4	0.5	0.4	0.6	0.8	1.0
			Post 8	1.1	1.2	1.3	1.2	0.7	0.6	0.6	0.6	0.7	0.8	1.0	1.2
			Post 9				1.2	0.8	0.5	0.5	0.5	0.7	0.3	0.5	
WIL2-100%	LB3	168	Pre 2	0.1	-0.1	0.2	0.1	0.2	0.2	0.2	0.0	0.0	-0.1	-0.2	0.1
			Pre 1	0.0	0.1	0.0	0.0	-0.1	-0.2	-0.1	0.0	0.0	-0.2	-0.2	-0.1
			Post 1	0.3	0.4	0.6	0.5	0.3	0.5	1.0	1.0	1.1	0.2	0.2	0.1
			Post 2	0.4	0.4	0.4	0.5	0.7	0.4	0.6	0.9	0.5	0.2	0.3	0.4
			Post 3	0.4	0.4	0.4	0.5	0.4	0.4	0.5	0.7	0.6	0.3	0.4	0.2
			Post 4	0.3	0.4	0.5	0.4	0.1	0.0	0.0	0.1	0.2	-0.1	0.3	0.5
			Post 5	0.5	0.5	0.6	0.5	0.2	0.2	0.3	0.2	0.2	0.2		
			Post 6	0.6	0.6	0.5	0.6	0.2	0.1	0.2			0.2	0.4	0.8
			Post 7	0.7	0.8	0.8	0.5	0.3	0.1	0.1	0.0	0.3	0.2	0.6	0.7
			Post 8	0.7	0.8	0.8	0.6	0.0	0.0	0.2	0.0	0.2	0.3	0.6	0.8
			Post 9				0.7	0.2	-0.1	-0.1	-0.2	-0.1	-0.2	0.1	
WIL2-100%	LB1	181	Pre 2	-0.2	0.1	0.3		0.0	-0.1	0.2	0.2	0.1	0.0	-0.1	0.0
			Pre 1	-0.1	0.1	-0.1	0.0	0.0	-0.3	-0.6	0.0	0.0	-0.2	0.0	0.1
			Post 1	0.5	0.7	0.5	0.4	-0.2	-0.1	0.1	0.2	0.3	-0.1	0.2	0.0
			Post 2	0.6	0.3	0.5	0.4	0.2	0.0	0.0	0.1	0.3	0.0	0.2	0.4
			Post 3	0.4	0.5	0.5	0.5	0.4	0.3	0.2	0.5	0.5	0.2	0.5	0.3
			Post 4				0.5	0.3	0.3	0.4	0.6	0.4	0.1	0.5	
			Post 5	0.7	0.7	0.8	0.7		0.4	0.3	0.6	0.6	0.4	0.5	0.3
			Post 6				0.8	0.7	0.5	0.4	0.6	0.5	0.6		
			Post 8	0.0	0.5	0.8	0.6		0.3	0.7	0.6	0.4	0.3	0.4	0.2
			Post 9				0.6	0.6	0.5	0.6	0.7	0.6	-0.2	0.3	
WIL2-100%	T3	204	Pre 2	0.0	0.0	0.1	0.1	0.1	0.0	0.0	0.0	0.0	0.0	-0.1	0.1
			Pre 1	-0.2	0.0	0.0	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1
			Post 1	0.2	0.4	0.8	0.8	1.0	1.3	1.7	1.1	1.0	0.4	0.4	0.1
			Post 2	0.5	0.6	0.6	0.9	1.2	0.9	1.2	1.1	0.9	0.7	0.6	0.5
			Post 3	0.3	0.4	0.4	0.9	1.0	0.9	0.9	0.9	0.5	0.3	0.4	0.3
			Post 4	0.3	0.5	0.6	0.7	0.6	0.4	0.5	0.4	0.1	0.2	0.6	0.5
			Post 5	0.5	0.6	0.8	0.9	0.6	0.5	0.4	0.4	0.3	0.4	0.5	0.5
			Post 6	0.8			1.0	0.6	0.5	0.3	0.5	0.4	0.4	0.7	0.8
			Post 7	0.8	0.9	1.1	1.0	0.8	0.9	0.9	0.6	0.6	0.5	0.8	0.8
			Post 8	0.7	0.8	0.8	0.9	0.5	0.4	0.6	0.3	0.7	0.9	0.8	0.8
			Post 9				1.0	0.7	0.5	0.5	0.4	0.5		0.6	

Appendix Table 4-7 (continued). Mean monthly temperature response (MMTR) values for WIL2-100% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL2-100%	T2	637	Pre 2	0.0	-0.2	0.2	0.0	0.1	0.0	0.1	0.3	0.0	0.0	-0.3	0.0
			Pre 1	0.1	0.2	0.0	0.0	0.0	-0.4	-0.3	0.1	0.0	-0.3	-0.1	0.1
			Post 1	0.3	0.4	0.4	0.4	0.1	0.2	0.7	0.7	0.5	0.0	0.1	0.2
			Post 2	0.4	0.2	0.4	0.4	0.1	-0.1	0.0	0.4	0.5	0.0	0.1	0.3
			Post 3	0.4	0.4	0.4	0.6	0.2	0.1	0.1	0.4	0.4	0.0	0.1	0.1
			Post 4	0.0	0.3	0.4	0.5	0.0	-0.2	0.0	0.4	0.1	-0.2	0.3	0.4
			Post 5	0.5	0.7	0.6	0.5	0.1	0.1	0.2	0.5	0.6	0.1	0.2	0.3
			Post 6				0.5	0.3	0.1	0.4	0.8	0.7	0.5		
			Post 7	0.5	0.5	0.8	0.4	0.3	0.3	0.6	0.7	0.3	0.3	0.2	0.4
			Post 8	0.0	0.3	0.6	0.4	-0.1	0.0	0.5	0.6	0.4	0.2	0.3	0.2
			Post 9				0.3	0.1	-0.1	0.0	0.4	0.4	-0.6	-0.1	
WIL2-100%	T1	775	Pre 2	0.0	0.1	0.3	0.2	-0.1	-0.1	0.3	0.3	0.2	-0.1	-0.3	-0.1
			Pre 1	-0.1	0.3	0.0	-0.1	-0.1	-0.5	-0.3	0.2	0.1	-0.1		
			Post 1	0.4	0.3	0.3	0.2	0.0	0.1	0.4	0.6	0.3	0.0	0.1	0.1
			Post 2	0.6	0.4	0.5	0.3	-0.1	-0.2	0.1	0.3	0.4	0.1	0.2	0.4
			Post 3	0.5	0.6		0.0	-0.1	0.2	0.2	0.7	0.8	0.1	0.0	0.0
			Post 4						-0.1	0.0	0.5	0.6	-0.2		
			Post 5	0.7	0.6	0.7	0.8		0.3	0.2	0.4	0.4	0.1		0.8
			Post 6				0.6	0.3	0.2	0.5	0.8	0.7	0.7		
			Post 7	0.4	0.6	0.7	0.5	0.2	0.4	0.8	0.9	0.5	0.5	0.4	0.2
			Post 8	0.2	0.5	0.8	0.3	0.0	0.2	0.5	0.6	0.5	0.4	0.5	0.3

Appendix Table 4-8. Mean monthly temperature response (MMTR) values for WIL3-100% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL3-100%	T4	7	Pre 2	0.1	0.2	0.2	0.1	0.1	-0.3	-0.2		0.5	-0.3	-0.4	-0.1
			Pre 1	0.0	0.1	0.0	-0.2	-0.1	-0.6	0.7	0.6	0.2	-0.3	0.0	0.1
			Post 1	-0.1	-0.1	-0.1	-0.3					1.1	0.3	0.0	-0.4
			Post 2	0.3	0.5	0.5	0.5	0.1	0.0	1.3	1.0	0.6	-0.1	-0.2	0.0
			Post 3	0.5	0.4	0.4	0.7	0.2	0.2	1.3	2.2	1.6	-0.3	0.0	0.2
			Post 4	0.2	0.4	0.5	0.2	0.0	0.1	0.6	1.1	0.8	0.1	0.0	0.1
			Post 5	0.5	0.6	0.7	0.8	1.1	0.8	0.5	0.7	0.6	0.3	0.2	0.0
			Post 6	0.6	0.8	0.5	0.9	-0.3	-3.2			-0.4	0.5	0.3	0.5
			Post 7	0.5	0.7	0.8	1.5	1.6	1.1		0.3	0.3	-0.3	-0.1	0.3
			Post 8	0.5	1.0	1.0	0.8		0.4	0.2	-0.1	-0.2	-0.2	0.3	0.4
			Post 9				0.6	0.7	0.4	0.6	0.8	0.6	0.3	0.5	
WIL3-100%	LB1	77	Pre 2	0.1	0.4	0.0	1.0	1.0				-0.2	0.0	0.1	0.2
			Pre 1	-0.8	-0.2	-0.1	-0.4	0.1	-0.4			0.0	-0.4	0.3	-0.1
			Post 1	0.1	0.4	0.6	0.8	0.5	0.1	1.1	0.8	0.2	-0.1	0.1	-0.1
			Post 2	0.7	0.8	0.6	0.5	0.3	0.2	0.2	0.3	-0.5	0.0	0.4	0.0
			Post 3	0.7	0.6	0.5	0.2	0.0	-0.2	0.1	0.6	0.5	-0.7	0.5	0.9
			Post 4	0.3	0.5	0.1	0.3	-0.1	-0.3	0.6	1.6	1.4	0.2	0.5	0.2
			Post 5	0.4	0.7	0.6	0.3	0.6	0.4	1.0	1.2	0.6	0.3	0.7	0.9
			Post 6	1.2	1.3	0.9	0.8	1.0	1.2	1.5	0.9	0.1	0.0	0.3	1.3
			Post 7	1.1	1.2	1.2	1.3	1.2	1.5	2.1	1.6	0.5	0.2	0.9	1.5
			Post 8	1.1	1.2	1.3	1.1	1.1	1.0	1.4	0.5	0.3	0.7	0.5	1.1
			Post 9	0.5	1.0	0.2	0.2	0.6	0.8	0.7	0.8	0.9	0.5	0.6	0.5
WIL3-100%	T3	769	Pre 2	0.3	1.0	0.8	0.5	0.3	0.1	0.7		0.2	-0.4	0.1	0.3
			Pre 1	-0.9	-0.3	-0.5	-0.1	0.0	-0.3	0.1	0.5	0.3	-0.1	0.0	-0.9
			Post 1	0.4	0.6	0.7	1.2	1.2	1.1	2.5	1.7	1.2	0.1	0.2	-0.1
			Post 2	1.1	1.5	1.6	0.9	0.4	0.2	1.7	1.4	0.0	0.1	0.4	0.1
			Post 3	1.1	0.8	0.8	0.7	0.2	0.1	0.7	1.0	0.6	-0.1	0.5	0.8
			Post 4	0.8	0.9	0.6	0.8	0.5	0.0	0.4	0.1	0.1	0.3	0.7	0.6
			Post 5	1.0	1.5	1.7	1.1	1.9						0.7	0.8
			Post 7	1.8	2.1	2.1	1.8	1.5	1.3		-0.7	0.1	0.3	0.8	1.7
			Post 8	1.4	1.6	1.9	0.8	0.8	0.2	0.5	0.0	-0.1	0.5	0.5	1.0
			Post 9	0.7	0.7	0.8	0.5	0.6	0.3	0.2	0.5	0.9	0.3	0.6	0.6
WIL3-100%	T2	870	Pre 2	0.0	0.3	0.2	0.1	0.0	-0.2	-0.3		0.1	-0.4	-0.2	-0.1
			Pre 1					-0.1	-0.5	0.2	0.3	0.2			
			Post 1	0.4	0.3	0.7	0.9	0.8	0.9	2.3	1.7	1.0	0.0	0.0	0.1
			Post 2	0.6	1.0	1.1	0.6	0.2	0.0	1.2	1.4	0.2	0.0	0.2	0.1
			Post 3	0.6	0.4	0.3	0.3	-0.2	-0.2	0.6	0.9	0.9	0.0	0.3	0.5
			Post 4	0.1	0.1			0.3	-0.1	0.4	0.6	0.4	0.2	0.4	0.1
			Post 5	0.0	0.7	0.8	0.0	0.5	0.5	0.8	1.0	0.6	0.6	0.5	0.5
			Post 6	1.3	1.5	0.8	0.8	0.2	0.3	1.0	1.0	0.4	0.1	0.3	1.1
			Post 7	0.3	0.7	1.1	1.3	1.0	0.3	0.5	0.4	0.2	0.1	0.9	1.5
			Post 8	0.6	0.8	1.0	0.4	0.1	-0.1	0.2	0.0	-0.4	-0.1	1.0	0.4
			Post 9	0.5	0.5	0.3	0.0	-0.1	-0.5	-0.2	0.6	0.3	-0.6	0.2	0.2

Appendix Table 4-8 (continued). Mean monthly temperature response (MMTR) values for WIL3-100% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL3-100%	T1	971	Pre 2	0.4	0.6	0.5	0.4	0.2	0.0	-0.2		0.2	-0.1	-0.1	0.1
			Pre 1	-0.7	-0.3	-0.4	-0.6	0.1	-0.6	0.1	0.3	0.2	-0.2	0.4	-0.6
			Post 1	0.4	0.5	0.8	1.2	1.0	1.1	2.3	1.6	1.2	0.1	0.2	0.1
			Post 2	0.9	1.0	0.8	0.7	0.1	0.0	0.8	0.9	0.1	0.6	0.4	0.1
			Post 3	1.0	0.7	0.5	0.3	0.1	0.1	1.0		-0.7	-0.5	0.8	0.8
			Post 4					0.1	-0.3	0.6	1.2	1.1	0.3	0.6	
			Post 5	0.7	1.3	1.4	0.5	0.9	0.3	0.7	0.9	0.7	0.9	0.7	0.9
			Post 6	1.7	1.6	1.2	1.1	0.5	0.1	0.6	0.4	0.1	-0.1	0.6	1.1
			Post 7	0.9			0.5	1.1	0.6	0.8	0.2	-0.1	0.1	0.8	1.6
			Post 8	1.1	1.3	1.4	0.5	0.2	-0.1	0.1	-0.4	-0.7	0.1	0.4	0.9
			Post 9	0.6	0.8	0.5	0.2	0.1	-0.3	-0.1	0.9	0.8	0.0	0.6	0.4

Appendix Table 4-9. Mean monthly temperature response (MMTR) values for OLYM-FP listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
OLYM-FP	T4	5	Pre 2	0.1	0.0	0.1	0.0	0.0	-0.2	-0.7		1.7	0.3	0.1	0.1
			Pre 1	0.0	0.0	-0.1	-0.1	0.0	0.2	0.2	-0.2	-0.2	0.0	0.1	0.0
			Post 1	0.2	0.3	0.5	0.4						0.4	0.0	-0.1
			Post 2	-0.2	-0.2	0.1	-0.1	-0.1	-0.3	-0.4	0.1	0.8	0.4	-0.1	-0.1
			Post 3	0.4	0.3	0.5	0.8	0.3	0.3	0.3	0.0	0.3	0.4	0.4	0.4
			Post 4	0.3	0.4	0.4	0.5	0.5	2.2	2.3	2.1		0.3	0.3	0.3
			Post 5	-0.4		0.5	0.7	0.8	1.0	0.6	0.8	0.5	0.5	0.2	-0.6
			Post 6	-0.2	0.0		0.7	1.0	1.1	1.9	1.9	0.7	-0.1	0.3	-0.8
			Post 7	0.5	0.5	0.6	0.8	1.4	1.8				0.7	0.4	0.5
			Post 9	0.9	1.1	1.1	1.3	1.2						0.6	0.9
OLYM-FP	T3	46	Pre 2	0.1	0.0	0.1	0.1	0.0	-0.1	0.0		-0.5	0.1	0.0	0.0
			Pre 1	-0.1	0.0	-0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-0.1
			Post 1	0.6	0.5	0.7	1.1	1.0	1.6	2.8	3.1	3.7	0.6	0.0	-0.1
			Post 2	0.2	0.5	1.0	1.0	0.9	0.8	1.5	2.5	2.0	0.8	0.2	0.3
			Post 3	0.5	0.6	0.8	1.2	1.1	1.0	1.4	1.9	1.9	0.7	0.3	0.5
			Post 4	0.5	0.6	0.8	1.0	0.9	0.8	0.7			0.7	0.5	0.5
			Post 5	0.7	0.7	0.8	1.0	0.9	0.9	0.9	1.2	1.4	0.8	0.5	0.5
			Post 6	0.5	0.6	0.8	1.1	0.7	1.0	1.4	2.5	2.4	1.8	0.6	0.5
			Post 7	0.6	0.7	0.7	1.1	1.1	1.4	1.9			0.9	0.6	0.6
			Post 8	0.8	1.0	1.3	1.1	1.1	0.6				0.9	1.0	0.8
			Post 9	1.0	1.1	1.2	1.3	1.2	1.1	1.5	1.9		0.5	0.6	0.9
OLYM-FP	T2	150	Pre 2	0.0	0.0	0.2		0.0	0.0	0.0		0.3	0.4	0.2	0.1
			Pre 1	-0.1	0.0	-0.1	0.0	0.0	-0.1	0.0	-0.1	0.0	-0.1	0.0	-0.1
			Post 1	0.2	0.6	0.5	1.0	0.7	0.5	0.5	0.5	0.6	0.4	0.1	-0.1
			Post 2	0.1	0.3	0.6	0.7	0.7	0.6	1.0	1.0	1.1	0.7	0.1	0.2
			Post 3	0.5	0.6	0.9	1.2	1.1	0.7	0.7	0.8	0.7	0.6	0.3	0.4
			Post 4	0.6	0.8	1.0	1.2	1.0	0.8	0.7	0.6	0.7	0.6	0.5	0.6
			Post 5	0.7	0.7	0.9	1.0	0.7	0.6	0.6	0.6	0.5	0.7	0.5	0.5
			Post 6	0.5	0.6	0.8	0.9	0.6	0.5	0.2	0.3	0.5	0.6	0.4	0.4
			Post 7	0.6	0.7	0.8	0.9	0.7	0.4	0.2	0.4	0.7	0.6	0.4	0.6
			Post 8	0.6	0.6	0.8	1.1	0.8	0.6	0.6	0.6	0.8	0.7	0.7	0.6
			Post 9	0.8	1.0	0.9	0.9	0.9	0.9	0.7	0.7	0.4	0.8	0.8	0.8
OLYM-FP	RB2	214	Pre 2	0.0	0.1	0.2	0.1	0.1	-0.1	0.6		2.1	0.8	0.2	-0.1
			Pre 1	-0.1	0.0	0.0	0.0	0.0	-0.3	0.0	0.0	0.0	-0.2	-0.2	0.0
			Post 1	1.6	0.6	0.3	0.3	-0.1	-0.4	0.1			0.3	0.2	0.8
			Post 2	0.0	0.2	0.3	0.2	0.1	0.3	0.1	0.1	0.0	-0.1	0.0	0.0
			Post 3	0.5	0.6	0.7	0.8	0.5	-0.1	-0.2	0.4	0.6	0.2	0.1	0.3
			Post 4	0.7	0.8	0.8	0.8	0.6	0.3	0.0	0.2	0.3	0.5	0.4	0.7
			Post 5	0.4	0.4	0.6	0.4	0.5	0.5	0.5	0.6	0.7	0.4	0.4	0.5
			Post 6	0.5	0.5	0.7	0.6	0.3	0.4	0.3		1.0	0.8	0.5	0.4
			Post 8	0.6	0.6	0.7	0.6	0.5	0.5	0.2			0.8	0.6	0.6
			Post 9	0.8	0.7	0.5	0.2	0.5	0.6	0.7			0.7	0.7	0.7

Appendix Table 4-9 (continued). Mean monthly temperature response (MMTR) values for OLYM-FP listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
OLYM-FP	LB1	245	Pre 2	-0.1	-0.1	-0.1		0.1	-0.2	0.0	-0.1	-0.1	0.2	0.2	0.0
			Pre 1	0.1	0.0	0.0	0.0	0.1	0.2						
			Post 1	0.0		-0.2	0.7	0.2					0.5	0.4	-0.1
			Post 2	0.3	0.4	0.7	0.6	0.4	0.3	0.2	0.2	0.4	0.7	0.5	0.4
			Post 3	0.3	0.4	0.5	1.0	0.6	0.5	0.7	0.1	0.2	0.7	0.4	0.4
			Post 4	0.8	0.7	0.7	1.0	0.8	0.5	0.4	0.2	0.4	0.5	0.7	0.8
			Post 5	0.7	0.6	0.7	0.8	0.6	0.7	0.4	0.3	0.4	1.0	0.7	0.6
			Post 6	0.6	0.4	0.8	1.0	0.5	0.5	0.3	0.4	0.7	0.9	0.7	0.5
			Post 7	0.7	0.8	0.9	1.0	0.5	0.3	0.1	0.3	0.8	0.8	0.7	0.7
			Post 8	0.6	0.6	0.6	0.0	-0.2	-0.3	-0.5	-0.1	0.3	0.7	0.8	0.6
			Post 9	0.7	0.9	0.8	0.9	0.7	0.5	0.3	0.0	-0.1	0.8	0.7	0.7
OLYM-FP	T1	248	Pre 2	0.0	0.1	0.2	0.2	0.2	0.0	0.1		0.1	0.3	-0.1	0.0
			Pre 1	-0.2	-0.1	-0.1	-0.1	0.0	-0.2	-0.1	-0.1	0.0	0.0	0.0	-0.1
			Post 1	0.2	0.4	0.5	0.9	0.4	0.5	0.5	0.3	0.4	0.3	0.1	-0.1
			Post 2				0.7	0.5	0.3	0.5	0.5	0.4	0.2	-0.2	
			Post 3	0.3	0.4	0.6	0.8	0.6	0.4	0.5	0.6	0.6	0.3	0.1	0.3
			Post 4	0.3	0.5	0.5	0.7	0.5	0.2	0.1	0.2	0.3	0.3	0.2	0.2
			Post 5	0.4	0.5	0.8	0.8	0.7	0.5	0.4	0.5	0.5	0.4	0.3	0.4
			Post 6	0.4	0.5	0.7	0.9	0.7	0.7	0.7	0.7	0.6	0.6	0.3	0.3
			Post 7	0.7	0.9	0.9	1.2	1.0	0.9	0.9	0.8	0.7	0.6	0.4	0.5
			Post 8	0.6	0.8	0.9	1.1	1.0	0.8	0.6	0.6	0.8	0.7	0.6	0.5
			Post 9	0.7	0.7	0.6	0.6	0.7	0.9	1.0	0.7	0.6	0.7	0.7	0.6
OLYM-FP	RB1	313	Pre 2	0.4	0.5	0.6	0.1	0.0	-0.2	0.5		1.7	0.3	0.1	0.0
			Pre 1	-0.1	-0.2	-0.4	-0.4			-0.1	0.1	-0.1	-0.1	-0.2	-0.1
			Post 1				1.0	0.6		0.3	0.6	0.6	0.2		
			Post 2	0.9	0.9	1.0	0.6	0.4	1.3	1.6	1.9	1.2	0.6	0.5	0.4
			Post 3				0.3	0.1	0.0	-0.1	0.1	0.3	0.2	0.1	
			Post 4	1.0	1.0	0.9	1.0	0.8	0.7	0.9	0.5	0.4	0.3	0.1	1.0
			Post 5	0.9	1.0	0.9	0.4	0.7	0.6	0.5	0.7	0.8	0.5	0.8	0.8
			Post 6	0.9	1.0	1.1	1.1	0.9	1.0	1.2	1.2	1.1	0.7	0.6	0.8
			Post 7	1.2	1.2	1.3	1.2	0.8	0.6			1.1	0.9	1.3	0.7
			Post 8	0.9	0.8	0.9	0.9	1.1	1.1	0.7	1.1	1.0	0.8	0.6	0.9
			Post 9	0.8	0.4	0.6	0.7	1.0	0.8	0.9	1.1		0.4	0.6	0.5
OLYM-FP	D100	369	Pre 1	0.2	0.1	0.1	0.0			-0.7	-0.3	-0.1	0.0	0.0	0.3
			Post 1				1.0	0.9	1.4	1.5	2.5	1.8	0.6		
			Post 2	0.4	0.7	0.9	0.9	0.8	0.7	1.3	1.3	0.9	0.4	0.1	0.0
			Post 3	0.4	0.8	0.7	0.9	0.8	0.8	1.2	1.3	1.2	0.4	0.1	0.2
			Post 4	0.5	0.8	0.9	1.1	1.0	0.7	0.9	1.4	1.3	0.5	0.1	0.3
			Post 5	0.5	0.6	1.0	1.1	1.1	1.2	1.3	1.6	1.4	0.5	0.3	0.3

Appendix Table 4-10. Mean monthly temperature response (MMTR) values for WIL1-FP listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL1-FP	T3	157	Pre 2	-0.1	-0.1	0.1	0.1	0.1	0.0	0.0	0.0	0.0	-0.1	0.2	0.0
			Pre 1	0.0	-0.1	-0.1	-0.1	0.0	0.0	0.0	-0.1	0.0	0.0	0.0	0.1
			Post 1	0.3	0.5	0.7	0.7	0.9	1.0	1.5	1.1	1.1	0.6	0.5	0.3
			Post 2	0.4	0.5	0.8	0.8	1.0	0.8	1.1	1.0	0.7	0.8	0.6	0.5
			Post 3	0.3		0.5	1.0	1.0	0.7	0.8	0.8	0.5	0.4	0.5	0.3
			Post 4	0.3	0.4	0.5	0.8	0.8	0.5	0.4	0.5	0.4	0.4	0.5	0.5
			Post 5	0.6	0.8	0.7	0.8	1.0	0.7	0.8	0.6	0.5	0.5	0.5	0.6
			Post 6	0.8	0.9	0.9	1.3	1.0	0.9	0.9	0.9	0.7	0.8	0.8	1.0
			Post 7	0.8	0.9	1.0	1.3	1.2	1.1	1.1	0.9	0.9	0.7	1.0	0.9
			Post 8	0.9	1.0	1.0	1.0	0.9	1.0	1.0	1.0	1.2	1.0	1.0	1.0
			Post 9			0.8	1.0	1.0	0.8	0.8	0.7	0.6	0.4	0.6	
WIL1-FP	T2	257	Pre 2	-0.1	-0.1	0.1	0.0	0.1	0.1	0.2	-0.1	-0.1	-0.2	0.1	0.0
			Pre 1	0.1	-0.1	0.0	0.0	0.0	0.0	-0.1	-0.1	0.0	-0.1	0.0	0.2
			Post 1	-0.1	0.0	0.2	0.6	0.6	0.6	1.2	1.2	1.5	0.5	0.3	0.1
			Post 2	0.2	0.3	0.5	0.5	0.7	0.6	0.9	1.0	0.5	0.3	0.5	0.4
			Post 3			0.3	0.8	0.8	0.6	0.8	0.8	0.4	0.3	0.3	
			Post 4	0.1	0.2	0.3	0.5	0.6	0.3	0.3	0.4	0.3	0.1	0.3	0.4
			Post 5	0.3	0.4	0.4	0.6	0.6	0.5	0.7	0.3	0.1	0.2	0.3	0.5
			Post 6	0.3	0.3	0.5	0.6	0.5	0.4	0.3	0.3	0.1	0.1	0.3	0.6
			Post 7	0.5	0.5	0.6	0.9	0.7	0.5	0.4	0.3	0.3	0.1	0.6	0.6
			Post 8	0.5	0.6	0.7	0.9	0.4	0.3	0.4	0.3	0.4	0.4	0.5	0.7
			Post 9			0.5	0.8	0.5	0.2	0.1	-0.1	-0.3	-0.2	0.3	
WIL1-FP	T1	356	Pre 2	-0.1	0.0	0.0	0.1	0.0	-0.1	-0.3	-0.3	-0.1	0.1	0.0	-0.1
			Pre 1	0.1	0.0	-0.1	-0.1	0.1	0.1	0.2	0.2	0.1	0.0	0.0	0.0
			Post 1	-0.1	0.1	0.2	0.7	0.9	1.1	1.8	1.8	2.1	0.6	0.3	0.0
			Post 2	0.2		0.4	0.4	0.8	0.9	0.9	1.3	0.7	0.7	0.5	0.3
			Post 3	0.2	0.3	0.3	0.8	1.0	0.9	1.1	1.3	0.9	0.5	0.5	0.3
			Post 4	-0.1	0.1	0.1		0.7	0.8	0.9	0.9	0.7	0.3	0.3	0.3
			Post 5	0.2	0.4	0.5	0.5	0.8	0.8	1.0	0.9	0.8	0.3	0.2	0.2
			Post 6	0.5		0.5	0.8	0.8	1.1	1.0	1.4	1.4	0.9	0.6	0.6
			Post 7	0.5	0.6	0.5	1.2	1.4	1.9	2.1	1.7	1.5	1.0	0.8	0.5
			Post 8	0.5	0.7	0.7	1.1	1.1	1.3	1.5	1.9	1.5	0.9	0.7	0.7
			Post 9			0.7	0.9	1.0	1.1	1.4	1.5	1.3	0.4	0.4	
WIL1-FP	D100	469	Pre 2				0.1	0.0	0.0	0.0	0.0	0.0	0.1		
			Pre 1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	-0.1	0.1
			Post 1	0.1	0.1	0.3	0.3	0.4	0.3	0.4	0.5	0.4	0.2	0.2	0.0
			Post 2	0.3	0.4	0.4	0.5	0.7	0.6	0.8	0.7	0.5	0.4	0.4	0.3
			Post 3	0.3	0.3	0.4	0.6	0.6	0.5	0.6	0.5	0.3	0.3	0.4	0.1
			Post 5	0.2	0.4	0.4	0.6	0.6	0.6	0.6	0.4	0.3	0.1	0.2	0.2
			Post 6	0.4		0.5	0.8	0.6	0.6	0.5	0.5	0.4	0.4	0.4	0.6
			Post 7	0.5	0.7	0.7	0.7	0.7	0.8	0.6	0.5	0.4	0.4	0.6	0.6
			Post 8	0.7	0.8	0.9	0.9	0.7	0.7	0.5	0.5	0.5	0.7	0.7	0.8
			Post 9			0.8	1.0	0.9	0.8	0.8	0.6	0.4	0.2	0.5	

Appendix Table 4-11. Mean monthly temperature response (MMTR) values for WIL2-FP listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL2-FP	T3	76	Pre 9	0.0	0.4	0.2	0.0	0.0	0.0	0.0	0.2	0.6	0.1	-0.2	-0.1
			Pre 8	0.0	0.0	-0.1	-0.3	-0.3	-0.5	0.3	0.3	0.1	0.0	0.1	0.1
			Pre 7	-0.1	-0.2	-0.1	-0.2	-0.3	-0.2	-0.4	0.1	0.1	-0.1	-0.1	0.0
			Pre 6	0.1	0.1	0.1	0.0	-0.1	-0.3	0.4	0.4	0.4	-0.2	-0.2	-0.1
			Pre 5	0.2	0.2	0.2	0.0	-0.1	-0.2	-0.2	0.1	0.0	-0.1	0.0	0.1
			Pre 4	0.2	0.2	0.1	0.0	-0.1	-0.3	-0.3	-0.1	0.2	-0.1	0.2	0.1
			Pre 3	0.1	0.2	0.1	0.0	0.0	-0.2	-0.3	0.1	0.3	0.1	0.0	0.0
			Pre 2	0.4	0.6	0.4	0.3	0.1	0.1	0.2	0.3	0.1	0.1	0.2	0.3
			Pre 1	0.7	0.7	0.6	0.5	0.5	0.5	0.6	0.8	0.9	0.5	0.4	0.6
			Post 1	0.3	0.3	0.5	0.4	0.7	0.6	1.2	1.5	1.0	0.2	0.2	0.3
WIL2-FP	T1	602	Pre 9	0.3	0.3	0.1	0.0	0.0	0.0	0.2	0.2	0.1	0.1	-0.4	0.2
			Pre 8	-0.2	0.0	0.0	-0.1	-0.4	-0.4	0.2	0.1	0.1	-0.2	-0.2	-0.1
			Pre 7	0.0					0.2	-0.2	0.0	-0.1	-0.3	-0.3	-0.2
			Pre 6	0.3	0.1	0.2	0.1	-0.1	0.1	0.1	0.3	0.3	0.2	0.2	0.3
			Pre 5	-0.2	0.3	0.0	-0.4	-0.7	-0.4	-0.1	-0.1	0.0	-0.2	-0.2	-0.1
			Pre 4	-0.2	0.1			-0.6	-0.7	-0.2	-0.1	-0.3	-0.3	-0.4	-0.3
			Pre 3	-0.1	0.4	0.3	0.3	-0.1	-0.1	0.0	-0.1	0.0	-0.2	-0.2	-0.1
			Pre 2	0.5	0.1	0.2	0.3	0.4	0.8	0.4	0.3	0.1	-0.2	-0.1	0.5
			Pre 1				0.5	0.5	0.5	1.1		0.8	1.0	1.2	0.5
			Post 1	1.0	0.9	0.9	0.6	0.6	0.6	0.7	0.8	0.6	0.5	0.5	

Appendix Table 4-12. Mean monthly temperature response (MMTR) values for CASC-FP listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
CASC-FP	T4	0	Pre 2	-0.1	0.1	0.2	0.1	0.1	0.1	0.2	0.0	-0.1	-0.1	-0.1	-0.2
			Pre 1	0.0	0.0	0.0	-0.1	0.0	-0.3	-0.1	-0.4	-0.2		-0.1	0.0
			Post 1	-0.2	-0.1	-0.2	-0.1	-0.4	-0.1	0.1	0.0	0.2	-0.2	-0.4	-0.5
			Post 2	0.2	0.4	0.2	0.1	0.1	-0.2	0.3	0.6	0.3	-0.2	-0.1	0.1
			Post 3	0.3	0.2	0.4	0.3	0.0	-0.1	-0.1	-0.1	0.0	0.0	0.0	-0.1
			Post 4	0.0	0.2	0.2	0.3	-0.2	-0.4	-0.4	-0.2	0.1	-0.1	-0.1	-0.1
			Post 5	-0.2	-0.2	0.2	0.3	0.4	-0.1	-0.1	0.3	0.1	-0.2	-0.3	-0.2
			Post 6	0.2	0.3	0.4	0.6	0.4	0.7	0.3	0.1	0.3	0.3	0.1	0.1
			Post 7	0.5	0.7	0.7	0.6	0.6	0.7	0.9	0.8	0.8	0.8	0.4	0.6
			Post 8	0.8	0.4	0.6	0.6	0.7	0.8	0.8	1.5		1.0	0.7	0.9
CASC-FP	LB1	249	Pre 2	-0.1	-0.2	0.4	0.1	0.2					0.2	0.0	-0.3
			Pre 1	0.1	-0.2	-0.1	-0.4	-0.1	-0.4	-0.2				0.0	0.2
			Post 1	0.6	0.8	0.0	1.0	1.3	1.3				0.2	0.3	-0.1
			Post 2	1.2	1.3	1.1			-0.3	0.0	-0.7	2.0	1.5	0.7	1.0
			Post 3	0.4	0.6	0.7	1.0	1.0	0.7				1.0	0.8	0.0
			Post 4	0.2	0.7	0.9	1.7	1.5	1.3	2.5	4.0		0.5	0.7	0.4
			Post 5	0.3	0.3	1.0	1.4	2.1	2.6	3.4	4.5	4.3	1.3	0.1	0.0
			Post 6	0.6	1.0	1.3	1.5	1.2	1.6	2.4	2.7		0.6	0.5	0.5
			Post 7	1.1	1.2	1.2	1.5	1.4	1.0				-0.1	0.1	0.9
			Post 8	1.7	0.7	0.9	-1.1	-2.0					0.8	1.1	1.5
			Post 9			1.4	1.6	1.5	1.3	2.1					
CASC-FP	T3	331	Pre 2	-0.3	0.6	0.6	0.3	0.5	0.5				0.3	-0.3	-0.4
			Pre 1	0.8	0.1	0.0	0.1	0.7	-0.3	0.0				-0.9	0.4
			Post 1			0.2	1.1	0.6	0.3	1.3					
			Post 2	0.8	0.6	1.0			-0.4	0.4	1.3	0.1	0.5	0.5	0.9
			Post 3	0.5	0.5	0.9	0.9	0.2	-0.1	0.7	1.2		0.5	1.1	0.4
			Post 4	0.5	0.6	0.9	1.2	0.1	-0.2	0.1	1.3	2.1	0.3	0.7	0.8
			Post 5	0.2	0.0	0.9	0.4	0.2	0.0	0.6	2.2	1.7	0.8	0.2	0.2
			Post 6	0.3	0.2	0.3	0.3	-0.3	-0.9	-1.8			1.2	0.4	0.6
			Post 7	0.8	0.9	0.4	0.0	-0.2	-0.3				-0.9	0.2	0.7
			Post 8	0.3	0.6	0.7	0.1	-0.4	-0.6	0.1	1.4	1.0	0.1	0.4	0.5
CASC-FP	T2	431	Pre 2		-0.3	0.0	-0.2	-0.3	-0.4	-0.6	-0.4	-0.2		-0.2	-0.3
			Pre 1	-0.4			0.1	0.6	0.0	0.3	0.6	0.7	0.5	0.1	
			Post 1	-0.2	0.1	-0.3	0.3	0.1	-0.2	-0.4	-0.2	0.2	0.6	0.2	-0.5
			Post 2	-0.2	-0.2	0.1	0.4	0.3	0.2	0.7	1.0	0.9	0.5	0.1	0.1
			Post 3			0.4	0.6	0.4	0.3	0.5	0.5	0.4	0.6	0.4	-0.3
			Post 4	-0.5	-0.2	0.2		0.3	0.2	0.1	0.0	0.3	0.4	0.3	-0.1
			Post 5	-0.4	-0.5	0.3	0.2	0.3	0.1	-0.1	-0.1	0.3	0.5	0.0	-0.4
			Post 6	-0.1	0.0	0.2	0.5	0.3	0.1	0.0	-0.1	0.1	0.5	0.2	0.0
			Post 7			0.4	0.4	0.1	-0.1						
			Post 8	0.0	0.0	0.3			-0.1	-0.1	0.0	0.4	0.4	0.5	0.4
			Post 9			0.7	0.9	0.8	0.4	0.2	0.7	0.9	0.9		

Appendix Table 4-12 (continued). Mean monthly temperature response (MMTR) values for CASC-FP listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
CASC-FP	T1	528	Pre 2	-0.3	0.0	0.4	0.1	-0.2	-0.4	-0.3	-0.2	0.0	0.3	0.0	-0.1
			Pre 1	-0.2	-0.2	-0.1	0.2	0.7	-0.1	-0.1	0.0	0.3	0.2	-0.2	0.0
			Post 1	-0.1	0.3	-0.1	0.6	0.3	-0.2	-0.4	-0.1	0.7	0.5	0.2	-0.5
			Post 2	0.3		0.2	0.5		0.0	-0.1	0.2	0.4	0.5	0.1	0.2
			Post 3			0.8	0.9	0.6	0.2	0.1					
			Post 4	-0.2	0.2	0.7	0.4	0.6	0.3	0.0	0.1	0.6	0.5	0.4	0.1
			Post 5	-0.2	-0.2	0.8	0.7	0.5	0.2	-0.1	0.2	0.3	0.6	0.0	-0.4
			Post 6	0.1	0.3	0.6	0.7	0.4	0.0	-0.2	0.1	0.4	0.7	0.2	0.1
			Post 7	0.6	0.8	0.7	0.5	0.1	-0.1	0.0	0.1	0.7	0.5	0.2	0.6
			Post 8	-0.2	0.3	0.8	0.8		-0.1	-0.1	0.2	0.7	0.5	0.4	0.0
			Post 9			1.1	1.3	0.8	0.3	0.0	0.1	0.4	0.7		

Appendix Table 4-13. Mean monthly temperature response (MMTR) values for OLYM-0% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
OLYM-0%	T4	1	Pre 2	0.0	-0.1	-0.2	0.0	0.2	0.1	1.5	0.0	-0.1	-0.1	0.1	-0.1
			Pre 1	0.3	0.1	0.2	0.6	-0.2	-0.4	0.6	0.4	-0.4	-0.6	-0.5	0.3
			Post 1	0.7	1.0	1.1	1.3	1.4	1.7	2.5	2.1	0.3	0.3	0.5	0.1
			Post 2	0.5	0.5	0.6	1.1	0.7	0.8	0.9	1.0	1.0	0.8	0.6	0.6
			Post 3	0.6	0.6	0.5	0.7	0.6	0.7	0.6	0.6	1.0	0.8	0.6	0.4
			Post 4	0.7	0.7	0.7	0.8	0.7	0.8	1.2	1.0	0.7	0.8	0.9	0.8
			Post 5	0.7	0.7	0.7	0.8	0.7	0.8	1.2	1.0	0.7	0.8	0.9	0.8
			Post 6	0.8	0.7	1.0	1.1	1.2	1.7	2.8	2.2	1.7	0.8	0.8	0.7
			Post 8	1.1	0.9	0.9	0.8	0.5	0.5				1.1	1.1	1.1
OLYM-0%	RB1	31	Pre 2	-0.1	-0.1	-0.1	0.3	0.1	-0.3	-0.4	0.1	0.4	0.1	0.2	0.0
			Pre 1	0.1			0.8	0.1	-0.3	-0.3	-0.2	0.1	0.1	0.1	0.1
			Post 1	0.1	0.0	0.4	0.7	0.5	0.8	1.6	1.4	-0.4	0.4	0.3	0.1
			Post 2				1.3	0.7	0.3	-0.2	0.3	1.1			
			Post 3	0.6	0.5	0.7	0.6	0.5	0.6	0.8	1.0	0.6	0.6	0.9	0.8
			Post 4	0.7	0.5	0.7	0.6	0.4	0.5	1.1	1.8	1.8	1.2	0.8	0.8
			Post 5	0.7	0.8	1.0	1.3	0.8	0.8	1.5	2.1	1.3	0.7	0.4	0.8
			Post 6	1.3	1.3	1.9	2.5	2.6	1.8		0.7	1.1	1.3	1.1	1.2
			Post 7	0.7	0.8	0.8	0.8					0.7	0.4	0.5	0.6
			Post 8						0.7	0.8	1.1				
OLYM-0%	T3	114	Pre 2	-0.1	-0.1	-0.2	-0.1	0.0	-0.5	-0.3	-0.9	-0.6	-0.7	-0.5	-0.3
			Pre 1	0.2	0.3	0.2	0.0	-0.4	-0.7	-1.0	-0.8	-0.8	-0.6	-0.4	-0.1
			Post 1	0.2	0.6	0.8	1.0	0.8	0.5	0.7	0.4	-0.6	-0.4	-0.5	-0.1
			Post 2	0.3	0.6	0.6	1.1	0.8	-0.2	-0.5	-0.3	-0.3	-0.1	-0.2	-0.1
			Post 3	0.3	0.4	0.5	0.5	0.3	-0.2	-0.2	0.3	0.1	-0.3	-0.1	0.1
			Post 5	0.4	0.5	0.7	0.8	0.6	0.1	-0.3	-0.7	-0.6	-0.2	-0.1	0.2
			Post 6	0.7	0.9	1.1	1.1	1.4			-0.8	-0.5	-0.3	-0.1	0.2
OLYM-0%	T2	242	Pre 2	-0.2	-0.2	0.0	0.1	0.0	0.0	0.1	0.3	0.0	-0.1	-0.1	-0.2
			Pre 1	-0.3	-0.2	-0.2	-0.2	-0.1	-0.1	-0.1	0.8	0.4	-0.1	-0.3	-0.3
			Post 1	0.2	0.6	0.9	1.2	1.2	1.3	1.1	0.9	0.4	0.1	0.0	-0.1
			Post 2	0.3	0.5	0.7	1.1	1.0	0.9	1.0	0.8	0.5	0.6	0.4	0.2
			Post 3	0.4	0.5	0.6	0.9	0.9	0.8	1.1		1.0	0.7	0.4	0.3
			Post 5	0.7	0.8	1.3	1.4	1.4			0.3	0.3	0.7	0.5	
			Post 7	0.9	1.2	1.2	1.4	1.1	1.0	0.9	0.8		0.9	1.0	0.8
OLYM-0%	T1	319	Pre 2	-0.2	-0.3	-0.3	-0.2	-0.1	-0.1	0.0	0.0	0.0	0.2	0.1	0.0
			Pre 1				0.2	0.1	-0.1	-0.2	-0.3	0.0			
			Post 1	0.0	0.1	0.5	0.7	0.9	1.0	1.4	1.0	0.3	0.4	0.1	0.0
			Post 2	0.2		0.4	0.8	0.8	0.7	0.8	0.8	1.0	0.9	0.5	0.3
			Post 3	0.2	0.2	0.3	0.5	0.5	0.5	0.4	0.5	0.4	0.7	0.5	0.4
			Post 4	0.4	0.3	0.4	0.7	0.6	0.7	0.7	0.5	0.3	0.4	0.5	0.4
			Post 5	0.6	0.6	0.6	0.7	0.5	0.4	0.3	0.4	0.4	0.7	0.6	0.6
			Post 6	0.8	0.8	0.8	0.2	0.3	0.4	0.2	0.3	0.6	0.6	0.8	0.8
			Post 7	0.8	0.8	0.7	1.1	0.8	0.7	0.4	0.4	0.7	0.8	0.9	0.8
			Post 8	0.6	0.5	0.3	0.1	0.2	0.6	0.6	0.5	0.7	0.8	0.8	0.7

Appendix Table 4-13 (continued). Mean monthly temperature response (MMTR) values for OLYM-0% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
OLYM-0%	D100	423	Pre 2	-0.1	0.0	0.1	0.1	0.0	-0.2	0.0	0.0	0.1	0.0	0.0	-0.2
			Pre 1				-0.3	0.1	-0.1	-0.2	0.0	0.1			
			Post 1	0.2	0.5	0.7	0.7	0.8	0.8	0.9	0.6	0.5	0.2	0.0	0.0
			Post 2	0.2	0.5	0.6	0.7	0.6	0.4	0.2	0.0	0.6	0.4	0.2	0.1
			Post 3	0.3	0.4	0.6	0.7	0.6	0.4	0.3	0.2	0.2	0.3	0.2	0.2
			Post 4	0.3	0.5	0.6	0.7	0.6	0.6	0.5	0.5	0.5	0.2	0.2	0.2
			Post 5	0.4	0.5	0.8	1.0	0.8	0.6	0.6	0.6	0.6	0.3	0.3	0.3
			Post 6	0.7	1.0	1.0	1.1	0.9	0.7		0.8	0.9	0.6	0.4	0.4
			Post 7	0.7	0.9	1.1	1.2	0.9	0.7	0.5	0.7	0.5	0.6	0.6	0.5
			Post 8	0.5	0.7	0.9	0.9	1.0	1.0	1.1	1.3	0.6	0.6	0.6	0.6

Appendix Table 4-14. Mean monthly temperature response (MMTR) values for WIL1-0% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL1-0%	T4	2	Pre 2	0.2	0.2	0.2	0.3		0.3	0.1	0.2			0.1	0.1
			Pre 1	0.1	0.0	-0.1	-0.1	-0.3	-0.3	0.0	0.0	-0.1	-0.1	-0.2	0.0
			Post 1	0.2	0.8	1.1	1.1	1.2	1.2	1.5	1.0	0.6	-0.1	0.1	-0.1
			Post 2	0.4	0.5	0.5	0.0	-0.1	0.0	-0.1	0.1	0.2	0.2	0.3	0.4
			Post 3		0.0	0.0	-0.1	-0.1	-0.3	0.1	0.4	-0.8	-1.0	-1.7	-2.1
			Post 4	-0.2				0.6	0.5	0.4	0.3	0.1	0.0	-0.3	-0.1
			Post 5	0.2	0.0	-0.1	0.0	-0.1	-0.1	-0.1	0.2	0.1	0.0	0.2	0.1
			Post 6	0.2	0.3	0.5	0.5	0.3	1.2	1.0	1.6	1.4	1.1		0.1
			Post 7	-0.2	0.3	0.5	0.7	1.0	1.1	1.5	1.8				-0.5
			Post 8	0.3	0.0	0.2	0.1	0.6	0.7	0.7	0.8	-0.4	-0.1	0.3	0.4
			Post 9		0.0	0.1	0.1	0.3	0.2	0.2	0.4	0.6			
WIL1-0%	LB2	25	Pre 1	-0.1	-0.1	-0.1	-0.1	0.0	0.0	0.2	-0.1	0.0	0.1	0.2	-0.3
			Post 1	0.1	-0.2	0.0						0.4	0.1	0.3	-0.1
			Post 2	0.3	0.3	0.3	-0.2	-0.2	-0.5	-0.6	-0.3	-0.1	0.1	0.3	0.3
			Post 3	0.1	0.1	0.0	0.0	0.0	0.8	1.4	1.7	1.5	0.2	0.1	0.0
			Post 4	-0.1	0.2	-0.1	-0.1	-0.5	-0.6	-0.6	-0.3	-0.4	-0.1	0.3	0.3
			Post 5	0.1	-0.1	-0.1	0.0	-0.2	0.4	0.3	0.1	-0.1	0.1	0.1	0.1
			Post 6	0.7	0.3	0.3	0.2	-0.1	-0.1	0.0	0.2	0.3	0.5	0.8	0.9
			Post 7	0.5	0.4	0.4	0.3	0.3	0.5	0.4	0.6	0.5	1.0	1.0	0.5
			Post 8		0.6	0.6	0.6	0.6	1.0	0.7			2.1	1.3	
WIL1-0%	LB1	106	Pre 2	0.0	0.1	0.4	-0.2		0.0	0.1	0.0	0.3	-0.1	-0.2	-0.1
			Pre 1	0.1	0.2	-0.1	-0.2	-0.3	-0.2	0.2	0.1	0.1	-0.1	-0.3	0.1
			Post 1	0.7	0.8	1.2	1.3	1.5	1.5	1.2	0.8	1.1	0.8	0.7	0.3
			Post 2	0.8	1.0	1.4	1.1	1.4	1.1	1.7	1.7	1.1	0.3	0.5	0.6
			Post 3	0.7	0.8	1.0	1.2	1.2	1.3	1.8	1.8	1.3	0.4	0.6	0.4
			Post 4	0.6	0.8	0.8	1.1	1.0	0.8	0.8	1.0	0.6	0.1	0.5	0.8
			Post 5	0.7	1.1	1.1	1.2	0.9	0.6	0.5	0.7	0.8	0.1	0.4	0.5
			Post 6		0.9	1.2	1.2	0.7	0.6	0.7	1.0	0.9	0.8		
			Post 7	1.1	1.0		0.8	0.9	0.8	1.0	1.0	0.6	0.6	0.5	0.9
			Post 8	0.7	1.1	1.2	1.1	0.6	0.5	0.8	0.8	0.6	0.6	0.7	0.7
			Post 9		0.9	1.1	1.0	0.7	0.4	0.3	0.6	0.6	-0.4	-0.3	
WIL1-0%	T3	347	Pre 2	-0.1	0.0	0.2	0.2		-0.1	-0.1	-0.1	0.0	-0.2	0.1	0.0
			Pre 1	-0.1	0.0	0.0	0.0	-0.1	-0.1	0.1	0.1	0.1	0.0	0.0	0.1
			Post 1	0.3	-0.1	0.2	0.7	0.9	1.1	1.5	1.1	1.0	0.2	0.4	0.0
			Post 2	0.3	0.4	0.7	0.6	0.8	0.6	1.0	1.0	0.7	0.3	0.3	0.4
			Post 3	0.3	0.3	0.5	0.8	0.7	0.6	0.8	0.8	0.5	0.3	0.3	0.2
			Post 4	0.0	0.4	0.4	0.6	0.5	0.2	0.3	0.4	0.1	-0.1	0.3	0.4
			Post 5	0.3	0.4	0.7	0.8	0.7	0.4	0.4	0.4	0.5	0.1	0.2	0.3
			Post 6	0.6	0.5	0.8	1.0	0.5	0.4	0.3	0.5	0.4	0.5	0.5	0.7
			Post 7	0.6	0.7	0.9	0.8	0.8	0.5	0.4	0.4	0.4	0.5	0.5	0.6
			Post 8	0.2	0.8	1.1	1.1	0.6	0.3	0.5	0.4	0.5	0.5	0.6	0.5
			Post 9		0.4	0.6	0.9	0.5	0.2	0.2	0.1	0.1	-0.3	-0.2	

Appendix Table 4-14 (continued). Mean monthly temperature response (MMTR) values for WIL1-0% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL1-0%	T2	467	Pre 2	-0.1	-0.1	0.3	0.0		-0.1	0.0	0.1	0.1	-0.2	0.0	0.0
			Pre 1	0.0	0.1	0.0	0.0	-0.1	-0.1	0.1	0.1	0.1	-0.1	-0.1	0.1
			Post 1	0.7	0.2	0.9	1.8	1.7	2.8	4.2	3.6	3.8	1.3	0.6	0.3
			Post 2	0.6	0.9	1.6	1.7	2.2	1.7	3.2	3.7	2.6	1.0	0.6	0.7
			Post 3	0.6	0.7	1.0	1.6	1.9	2.1	2.9	2.9	1.9	0.7	0.6	0.4
			Post 4	0.3	0.7	0.9	1.4	1.6	1.2	1.4	1.8	1.2	0.2	0.5	0.7
			Post 5	0.5	0.9	1.3	1.7	1.6	1.3	1.4	1.5	1.2	0.4	0.4	0.4
			Post 6	0.7	0.8	1.3	1.8	0.9	1.0	1.1	1.4	1.1	0.8	0.6	0.9
			Post 7	0.8	0.9	1.0	1.3	1.2	0.9	0.9	0.8	0.7	0.6	0.5	0.7
			Post 8	0.4	0.9	1.2	1.3	0.8	0.6	1.0	0.9	0.8	0.6	0.7	0.7
			Post 9		0.6	0.8	1.2	0.6	0.3	0.4	0.5	0.5	-0.4	-0.3	
WIL1-0%	RB1	518	Pre 2	0.0	0.2	0.1	0.1		0.1	0.1	0.1	0.1	-0.1	0.1	0.1
			Pre 1	-0.1	0.0	-0.1	-0.1	-0.1	-0.1	0.0	0.0	-0.1	0.0	-0.1	-0.2
			Post 1	0.6			1.2	1.7	2.4	3.0	2.2	1.8	0.7	0.6	0.2
			Post 2	0.9	0.7	1.1	1.1	1.4	1.4	1.7	1.3	1.0	0.6	0.6	0.7
			Post 3		1.2	1.1	1.1	1.1	1.1	1.4	1.3	0.8	0.5	0.5	0.4
			Post 4	0.3				0.7	0.6				0.6	0.5	0.6
			Post 5	0.4	0.6	0.7	0.8	0.7	0.7	0.6	0.5	0.5	0.3	0.3	0.4
			Post 6	0.8	0.7	0.8	0.8	0.7	0.7	0.6	0.8	0.5	0.5	0.6	0.7
			Post 7	0.7	0.9	0.9	1.0	1.1	1.0	0.9	0.7	0.6	0.6	0.5	0.6
			Post 8	0.4	0.9	1.2	1.2	0.9	0.7	0.8	0.6	0.6	0.7	0.8	0.6
			Post 9		0.7	0.8	1.0	0.7	0.4	0.4	0.2	0.2	0.0	0.1	
WIL1-0%	T1	557	Pre 1	-0.1	0.0	-0.1	0.1	0.0	-0.1	0.0	0.0	0.1	0.0	-0.1	0.0
			Post 1	0.4	0.2	0.9	1.4	1.8	2.4	3.4	2.4	2.3	0.9	0.5	-0.1
			Post 2	0.5	0.6	1.1	1.3	1.7	1.5	2.5	3.1	2.2	1.0	0.4	0.5
			Post 3	0.2	0.7	1.0	1.7	1.9	1.7	1.9	1.8	1.2	0.4	0.3	0.1
			Post 4	0.3	0.3			1.3	1.0	1.4	1.6	0.8	0.0	0.5	0.6
			Post 5	0.4	0.8	1.0	1.1	1.1	1.0	1.0	1.0	0.9	0.3	0.4	0.4
			Post 6	0.9	0.8	0.9	1.3	0.9	1.0	0.9	1.1	0.8	0.8	0.6	0.8
			Post 7	0.8	0.6		1.0	1.2	1.0	0.9	0.9	0.7	0.7	0.5	0.7
			Post 8	0.5	1.1	1.3	1.4	1.0	0.6	0.7	0.5	0.6	0.8	0.9	0.7
			Post 9		0.9	1.1	1.3	0.8	0.5	0.5	0.5	0.4	0.1	0.1	
WIL1-0%	D100	671	Pre 1	-0.1	0.1	0.0	0.0	0.0	-0.1	0.0	0.1	0.1	-0.1	0.0	0.0
			Post 1	0.4	0.0	0.6	0.9	1.3	1.6	2.1	1.6	1.5	0.4	0.4	0.0
			Post 2	0.5	0.6	1.0	1.0	1.3	1.1	1.7	1.6	1.1	0.4	0.3	0.5
			Post 3	0.4	0.6	0.8	1.0	1.2	1.2	1.5	1.4	0.8	0.4	0.4	0.3
			Post 5	0.3	0.7	0.9	1.0	0.9	0.8	0.8	0.7	0.6	0.1	0.2	0.3
			Post 6	0.7	0.8	0.9	1.1	0.7	0.7	0.6	0.8	0.5	0.5	0.5	0.7
			Post 7				0.9	0.9	0.8	0.7	0.6	0.6			
			Post 8	0.4	0.8				0.5	0.7	0.5	0.4	0.5	0.6	0.6
			Post 9		0.7	0.9	1.2	0.7	0.3	0.4	0.4	0.3	-0.2	0.0	

Appendix Table 4-15. Mean monthly temperature response (MMTR) values for WIL2-0% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL2-0%	T4	4	Pre 2	0.1	0.4	0.9	0.3	-0.2	-0.2	0.0	-0.3	0.8	0.3	-0.2	-0.5
			Pre 1	0.7	-0.1	-0.6	-0.3	0.1	-0.2	0.7	0.3	0.0	0.0	-0.4	0.6
			Post 1	0.0	0.2	1.0	1.2		2.5	2.3	2.1	1.6	1.5	0.2	0.0
			Post 2	1.4	2.1	2.2	2.0	1.8	2.0	2.2	1.9	1.8	1.1	0.5	-0.8
			Post 3	0.8	0.5	1.8	2.7	2.0	1.6	1.6	1.5	1.2	0.8	0.1	-0.1
			Post 4	0.5				1.9	1.7	2.1	2.7	1.4	0.3	0.0	-0.6
			Post 5	0.8	1.1	1.1	1.1	1.7	1.4	1.8	1.7	1.4	0.6	0.6	0.7
			Post 6	1.1	1.1	1.9	3.0	2.0	2.2	2.2	2.4	1.9	1.1	0.9	0.3
			Post 7	1.6	2.0	2.1	2.1	2.5	2.1	2.2	1.6	0.8	0.9	0.9	1.0
			Post 8	1.4	1.7	1.6	1.5	1.8	1.3	1.1	1.0	0.8	0.8	0.0	0.7
			Post 9	0.2	0.2	0.6	1.0	1.2	1.1	1.1	1.3	1.7	0.4	0.7	0.3
WIL2-0%	LB2	47	Pre 2	-0.1	0.2	0.4	0.2	0.2	0.2	0.1	0.0	0.0	0.0	-0.2	-0.1
			Pre 1				-0.3	-0.5	-0.6	0.5	0.1	0.0			
			Post 1	0.6	0.6	1.1	0.6	0.6	0.5	0.5	0.4	0.7	0.8	0.5	1.1
			Post 2	0.5	1.0	1.3	1.3	0.9	0.8	0.9	0.9	1.1	2.2	0.6	0.3
			Post 3	0.7	1.1	1.1	1.2	0.7	0.7	0.6	0.4	0.1	1.0	1.0	0.5
			Post 4	0.7	0.9	0.9	1.2	0.9	0.5	0.9	1.1		0.7	1.0	0.8
			Post 5	0.3	0.8	0.8	1.0	0.5	0.1	-0.2	-0.5	-0.1	0.6	0.8	0.6
			Post 6	1.1	1.3	1.1	1.0	0.7	0.4	0.1	-0.1	0.3	0.7	1.0	1.3
			Post 7	1.1	1.4	1.6	1.3	0.8	0.2	-0.5	-0.2	0.4	0.8	1.1	1.4
			Post 8	1.0	1.4	1.6	1.4	0.6	0.1	-0.4	-1.0	-0.1	1.0	1.3	0.8
			Post 9	0.7	0.6	0.6	1.0	0.5	-0.2	-0.8	-1.0	-0.7	0.3	0.9	0.7
WIL2-0%	LB1	119	Pre 2	-0.2	0.3	0.7	0.0	-0.1	-0.1	-0.1	-0.2	0.0	-0.1	-0.2	-0.2
			Pre 1	0.4	0.3	-0.3	-0.3	-0.1	-0.3	0.3	0.1	0.2	0.2	0.1	-0.2
			Post 1	0.9	1.6	3.0	4.0	4.8	4.8	5.6	4.4	4.3	2.4	0.9	0.5
			Post 2	1.4	2.9	4.1	3.5	4.4	4.4	4.9	5.0	3.6	2.2	1.3	-0.3
			Post 3	1.2	1.5	2.2	3.8	3.9	4.3	5.0	5.6	4.5	2.1	1.1	0.5
			Post 4	0.9	1.3	1.8	3.3	4.2	3.5	4.3	4.3		1.0	1.3	0.4
			Post 5	0.9	2.0	2.0	2.3	3.2	4.2	4.1	4.3	3.2	2.0	1.4	0.9
			Post 6	1.5	2.0	2.6	4.7	3.9	4.8	5.2	5.2	4.2	2.4	1.1	0.6
			Post 7	1.7	2.7	3.2	3.5	4.3	6.2	6.9	5.0	3.6	2.0	1.2	1.2
			Post 8	1.1	2.0	2.8	3.9	4.3	3.9	3.3	3.9	3.1	1.3	0.9	0.7
			Post 9	-0.1	0.8	1.1	2.3						1.3	1.1	0.0
WIL2-0%	T3	295	Pre 2	-0.2	0.2	0.5	0.0	0.0	0.0	-0.1	-0.3	0.0	0.0	-0.2	-0.1
			Pre 1	-0.2	0.1	-0.1	-0.2			0.2	0.0	0.1	0.1	0.3	0.0
			Post 1				2.2	2.5	2.4	2.9	2.6	1.6	0.7		
			Post 2	0.7	0.9	1.8	2.0	2.3	2.3	2.2	1.9	1.4	0.7	0.5	-0.3
			Post 3	0.5	0.4	1.3	2.0	2.1	2.1	2.4	1.9	1.0	0.8	0.5	0.3
			Post 4					1.6	1.8	1.7	1.6	0.7	0.2		
			Post 5	-0.2	0.9	1.2		1.4	1.6	1.3	0.9	0.6	0.1	0.6	0.4
			Post 6	0.7	1.1	1.3	1.8	1.7	1.8	1.5	1.2	0.8	0.4	0.5	0.8
			Post 7	0.4	1.3	1.5	2.2	1.8	1.9		0.5	0.4	0.4	0.6	0.9
			Post 8	0.4	0.9	1.3	1.8	1.4	1.2	0.9	0.7	0.7	0.4	0.3	0.1
			Post 9	-0.3	0.3	1.0	1.6	2.0	1.4	1.5	0.8	0.3	-0.2	0.5	-0.2

Appendix Table 4-15 (continued). Mean monthly temperature response (MMTR) values for WIL2-0% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
WIL2-0%	T2	660	Pre 2	0.0	0.2	0.5	-0.1	-0.1	-0.1	-0.1	-0.1	-0.1	-0.3	-0.2	-0.1
			Pre 1	0.0	0.3	-0.1	-0.2	0.0	-0.2	0.3	0.1	0.2	0.0	0.2	0.1
			Post 1	0.1		1.3	1.7				2.7	1.8	0.4	0.1	0.3
			Post 2	0.7	1.0	1.5	0.7	1.1	0.8	0.1	0.6	1.3	1.7	0.5	0.0
			Post 3	0.5	0.2	0.7	1.3	1.4	2.1	2.6	2.7	1.6	0.8	0.4	0.3
			Post 4	0.5	0.5	1.0	1.6	2.1	2.4	3.1	3.5	1.7	0.2	0.3	-0.1
			Post 5	0.2	1.0	1.5	2.1	2.7	3.2	3.7	3.3	2.0	0.5	0.6	0.6
			Post 6	0.6	1.1	1.5	2.5	2.6	3.5	4.0	3.8	2.2	0.8	0.5	0.5
			Post 7	0.5	1.1	1.4	2.1	3.0	3.9	3.8	2.5	1.3	0.7	0.4	0.7
			Post 8	0.6	1.1	1.7	2.6	2.9	2.8	2.5	2.2	1.4	0.7	0.3	0.2
			Post 9	-0.2	0.4	1.0	1.7						-0.4	0.5	-0.1
WIL2-0%	T1	745	Pre 2	0.6	0.4	0.5	-0.2	-0.3	-0.2	0.0	-0.1	-0.3	-0.4	-0.3	-0.2
			Pre 1	0.1	0.4	-0.2	-0.2	0.0	-0.3	0.3	0.3	0.3	0.1	0.0	0.0
			Post 1	0.3	0.1	0.8	0.8	2.0	1.9	2.3	2.0	1.8	0.8	0.3	0.2
			Post 2	0.9	0.8	1.3	1.4	1.5	1.6	2.2	2.2	1.6	0.6	0.5	-0.4
			Post 3	1.0	0.5	1.3	1.7	1.7	2.2	2.9	2.9	1.9	1.0	0.5	0.5
			Post 4	0.9	1.0			1.4	1.6	1.9	2.2	1.0	0.4	0.4	0.2
			Post 5	0.7	0.8	1.1	1.6	1.8	1.8	2.1	1.9	1.4	0.5	0.5	0.6
			Post 6	0.6	0.7	1.0							-0.1	0.2	0.5
			Post 7	1.6			1.9	2.2	2.1	2.2	1.7	1.1	0.8	0.7	0.8
			Post 8	0.5	0.7	0.8	1.3	1.5	1.4	1.3	1.2	0.9	0.7	0.2	0.2
			Post 9	0.0	0.4	0.9	1.1	1.5	1.3	1.4	1.4	0.8	-0.3	0.7	0.0

Appendix Table 4-16. Mean monthly temperature response (MMTR) values for CASC-0% listed by location and treatment year. No shading = P >0.05 or |MMTR| <0.5°C, blue shading = MMTR <-0.5°C and P <0.05. MMTR values >0.5°C and P <0.05 are shown in a gradient of light to dark brown shading showing changes of >0.5-1.0°C, >1.0-2.0°C, >2.0-3.0°C, >3.0-4.0°C, and >4.0°C.

Site	Location	Distance	TRYR	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
CASC-0%	T4	0	Pre 2	-0.1	-0.1	0.1	0.0	0.1	0.0	-0.1	0.1	0.1	0.1	-0.1	-0.1
			Pre 1	0.0	0.0	0.0	-0.3	-0.4	-0.3	-0.2	-1.2	-0.3		0.0	0.0
			Post 1	-0.2	0.1	0.1	-0.1	-0.5	-0.4	-0.7	-0.5	-0.4	-0.3	-0.4	-0.6
			Post 2	0.0	0.2	0.2	0.1	-0.1	-0.7	-0.6	-0.4	-0.4	-0.3	-0.3	-0.2
			Post 3	-0.2	0.1	0.1	0.1	-0.2	-0.4	-0.7	-0.8	-0.5	-0.3	-0.3	-0.3
			Post 4	-0.2	0.0	0.1	0.0	-0.2	-0.4	-0.7	-0.7	-0.3	-0.2	-0.2	-0.2
			Post 5	-0.1	0.0	0.2	0.1	0.0	-0.4	-0.6	-0.5	-1.5	-0.3	-0.2	-0.1
			Post 6	0.3	0.5	0.3	0.3	0.2	0.2	-0.2	0.1	0.1	0.0	0.3	0.2
			Post 7	0.3	0.6	0.6	0.7	0.7	0.9	0.7	1.0	0.9	1.5	0.5	0.5
			Post 8	0.9	0.6	0.7	0.8	0.6	0.5	0.3	0.2		1.2	0.6	0.6
			Post 9			0.5	0.7	0.6	0.6	0.2	0.2	0.5	0.4		
CASC-0%	T3	220	Pre 2	0.1	0.3	0.6	-0.5	-0.5	-0.5	-0.2	0.2	0.1	-0.6	-0.7	-0.4
			Pre 1	0.7	0.4	0.5	0.3	1.2	-0.4	0.0	0.5	0.5	-0.3	-0.8	0.2
			Post 1	0.4	1.5	0.6	1.8	0.9				2.1	0.9	0.0	-1.0
			Post 2	0.9		1.3	1.5	1.4	1.2	3.0	3.5	2.2	0.9	0.0	0.3
			Post 3	0.3	-0.5	1.8	2.2	1.1	1.5	2.9			0.5	0.6	-0.3
			Post 4	0.4	0.9	1.5	1.0	1.6	1.4	2.8	3.6	3.2	0.4	0.1	0.3
			Post 5	0.1	0.1	1.5	1.3	1.7	1.9	3.1	2.8	1.7	1.1	-0.3	-0.2
			Post 6	0.4	0.9	1.0	1.2	0.7	0.7	1.0	2.0	1.9	1.1	-0.3	-0.1
			Post 7	0.7	1.5	1.3	1.6	1.9	1.0	1.7	1.6	1.4	0.7	0.3	0.2
			Post 8	0.3	1.1	1.6	1.2	0.9	0.0	0.8	1.3	0.9	-0.4	0.0	0.2
			Post 9			1.6	1.7	1.1	0.5	1.0	2.0	1.3	0.5		
CASC-0%	T2	340	Pre 2	0.2	0.3	0.4	-0.5	-0.7	-1.1	-0.9		-0.1	-0.1	0.0	0.1
			Post 1	0.7	1.7	0.7	1.3	0.2				2.6	1.4	0.6	-0.7
			Post 2	0.6	0.0	1.6	2.0	2.2	1.9	3.4	3.2	2.5	1.1	0.2	0.4
			Post 3			1.4	1.7	1.6	2.1	3.0	2.6		0.8	0.9	-0.5
			Post 4	0.2	0.7	1.6		2.2	1.9	3.0	2.9	1.7	0.0	0.3	0.2
			Post 5	0.0	-0.1	1.6	1.5	2.1	1.9	2.3	1.8	0.9	0.8	-0.2	-0.3
			Post 6	0.4	0.7	1.0	1.7	1.2	0.7	0.6	1.1	0.7	0.4	-0.3	0.0
			Post 7	1.3	1.5	0.8	0.6	0.5	0.3	-0.2	-0.6	-0.4	-1.0	-0.5	0.6
			Post 8	-0.2	0.7	1.3	1.3		-1.1	-1.0	-0.8	-0.5	-0.4	0.1	-0.1
			Post 9			1.6	1.7	1.3	0.0	-0.3	-0.2	-0.1	0.2		
CASC-0%	T1	415	Pre 2	-0.2	0.0	0.4	0.0	-0.2	-0.4	-0.6	-0.3	0.0	0.2	0.1	0.0
			Pre 1	0.0	-0.2	-0.3	0.3	1.1	0.2	0.3	-0.2	0.2	0.0	-0.4	0.1
			Post 1	0.1	0.8	0.2	0.9	1.5	1.0	0.2	0.0	0.3	0.6	0.2	-0.8
			Post 2	0.4	0.1	1.1	1.1		1.1	1.7	1.5	1.6	0.8	0.6	0.2
			Post 3	0.0	0.3	1.0	1.1	0.8	0.9	0.9	0.3	0.3	0.7	0.6	-0.1
			Post 4	-0.1	0.2	0.8	1.9	1.7	1.7	1.6	0.8	0.4	0.5	0.5	0.0
			Post 5	-0.1	-0.2	0.9	0.8	1.4	0.8	0.8	0.5	0.8	0.6	0.1	-0.2
			Post 6	0.3	0.6	0.7	1.5	1.3	1.3	1.4	0.9	0.5	0.6	0.3	0.1
			Post 7	0.7	1.1	0.9	0.8	0.8	0.8	0.6	0.2	0.2	0.3	0.1	0.5
			Post 8	-0.2	0.4	1.0	0.9		0.3	0.5	0.2	0.4	0.6	0.5	0.0
			Post 9			1.4	1.6	1.7	1.1	0.9	0.6	0.5	0.6		

Appendix Table 4-17. Pair-wise comparisons with 95% confidence intervals of the seven-day average temperature response (Δ7DTR) at the Buffer Treatment locations for each combination of treatments for each post-harvest year.

Comparison	Δ7DTR Buffer Treatment			95% CI	
	Estimate	t-Value	P-value	Lower	Upper
100%-Pre vs. Post 1	1.1	-3.27	0.0015	0.4	1.7
100%-Pre vs. Post 2	1.1	-3.37	0.0011	0.5	1.8
100%-Pre vs. Post 3	0.3	-0.97	0.3359	-0.3	1.0
100%-Pre vs. Post 4	0.5	-1.59	0.1147	-0.1	1.2
100%-Pre vs. Post 5	0.4	-1.28	0.2036	-0.2	1.1
100%-Pre vs. Post 6	0.2	-0.52	0.6011	-0.5	0.8
100%-Pre vs. Post 7	0.3	-0.96	0.3373	-0.3	1.0
100%-Pre vs. Post 8	0.1	-0.23	0.8159	-0.6	0.7
100%-Pre vs. Post 9	0.3	-1.00	0.3176	-0.3	1.0
FP-Pre vs. Post 1	1.1	-3.57	0.0006	0.5	1.8
FP-Pre vs. Post 2	0.9	-2.95	0.0039	0.3	1.6
FP-Pre vs. Post 3	0.8	-2.13	0.0358	0.1	1.5
FP-Pre vs. Post 4	0.5	-1.42	0.1590	-0.2	1.3
FP-Pre vs. Post 5	0.5	-1.40	0.1658	-0.2	1.2
FP-Pre vs. Post 6	0.9	-2.44	0.0163	0.2	1.6
FP-Pre vs. Post 7	1.2	-3.35	0.0012	0.5	2.0
FP-Pre vs. Post 8	1.2	-3.24	0.0016	0.5	1.9
FP-Pre vs. Post 9	0.9	-2.33	0.0221	0.1	1.6
0%-Pre vs. Post 1	3.8	-10.16	<0.0001	3.1	4.6
0%-Pre vs. Post 2	3.0	-9.04	<0.0001	2.4	3.7
0%-Pre vs. Post 3	2.4	-7.03	<0.0001	1.7	3.0
0%-Pre vs. Post 4	2.0	-6.09	<0.0001	1.4	2.7
0%-Pre vs. Post 5	1.6	-4.69	<0.0001	0.9	2.2
0%-Pre vs. Post 6	1.3	-3.92	0.0002	0.6	2.0
0%-Pre vs. Post 7	1.2	-3.64	0.0004	0.6	1.9
0%-Pre vs. Post 8	1.0	-2.91	0.0045	0.3	1.6
0%-Pre vs. Post 9	0.8	-2.19	0.0311	0.1	1.6

Appendix Table 4-17 (continued). Pair-wise comparisons with 95% confidence intervals of the seven-day average temperature response (Δ7DTR) at the Buffer Treatment locations for each combination of treatments for each post-harvest year.

Comparison	Δ7DTR Buffer Treatment			95% CI	
	Estimate	**t-Value**	**P-value**	**Lower**	**Upper**
100% vs. FP-Post 1	0.0	0.11	0.9134	-0.9	1.0
100% vs. FP-Post 2	-0.2	-0.39	0.6937	-1.1	0.7
100% vs. FP-Post 3	0.5	0.94	0.3481	-0.5	1.4
100% vs. FP-Post 4	0.0	0.00	0.9984	-1.0	1.0
100% vs. FP-Post 5	0.1	0.19	0.8509	-0.9	1.1
100% vs. FP-Post 6	0.7	1.47	0.1440	-0.3	1.7
100% vs. FP-Post 7	0.9	1.85	0.0669	-0.1	1.9
100% vs. FP-Post 8	1.1	2.26	0.0261	0.1	2.1
100% vs. FP-Post 9	0.5	1.06	0.2896	-0.5	1.5
100% vs. 0%-Post 1	2.8	5.49	<0.0001	1.8	3.7
100% vs. 0%-Post 2	1.9	4.08	<0.0001	1.0	2.9
100% vs. 0%-Post 3	2.0	4.33	<0.0001	1.1	3.0
100% vs. 0%-Post 4	1.5	3.22	0.0017	0.6	2.4
100% vs. 0%-Post 5	1.2	2.45	0.0163	0.2	2.1
100% vs. 0%-Post 6	1.1	2.42	0.0172	0.2	2.1
100% vs. 0%-Post 7	0.9	1.92	0.0584	0.0	1.8
100% vs. 0%-Post 8	0.9	1.91	0.0594	0.0	1.8
100% vs. 0%-Post 9	0.5	0.98	0.3311	-0.5	1.5
FP vs. 0%-Post 1	2.7	5.49	<0.0001	1.7	3.7
FP vs. 0%-Post 2	2.1	4.55	<0.0001	1.2	3.0
FP vs. 0%-Post 3	1.6	3.15	0.0022	0.6	2.6
FP vs. 0%-Post 4	1.5	3.04	0.0030	0.5	2.5
FP vs. 0%-Post 5	1.1	2.12	0.0367	0.1	2.0
FP vs. 0%-Post 6	0.4	0.82	0.4126	-0.6	1.4
FP vs. 0%-Post 7	0.0	-0.03	0.9734	-1.0	1.0
FP vs. 0%-Post 8	-0.2	-0.44	0.6579	-1.2	0.8
FP vs. 0%-Post 9	0.0	-0.08	0.9395	-1.1	1.0

Appendix Table 4-18. Pair-wise comparisons with 95% confidence intervals of the seven-day average temperature response (Δ7DTR) at the F/N break locations for each combination of treatments for each post-harvest year.

Comparison	Δ7DTR F/N Break			95% CI	
	Estimate	t-Value	P-value	Lower	Upper
100%-Pre vs. Post 1	0.9	-2.13	0.0359	0.1	1.6
100%-Pre vs. Post 2	0.5	-1.31	0.1947	-0.3	1.3
100%-Pre vs. Post 3	0.6	-1.46	0.1466	-0.2	1.4
100%-Pre vs. Post 4	0.6	-1.57	0.1189	-0.2	1.4
100%-Pre vs. Post 5	0.4	-0.94	0.3476	-0.4	1.2
100%-Pre vs. Post 6	0.6	-1.56	0.1225	-0.2	1.4
100%-Pre vs. Post 7	1.0	-2.54	0.0127	0.2	1.8
100%-Pre vs. Post 8	0.6	-1.39	0.1689	-0.2	1.3
100%-Pre vs. Post 9	0.3	-0.74	0.4598	-0.5	1.1
FP-Pre vs. Post 1	1.1	-2.92	0.0043	0.4	1.9
FP-Pre vs. Post 2	0.9	-2.41	0.0178	0.2	1.7
FP-Pre vs. Post 3	0.8	-1.71	0.0895	-0.1	1.6
FP-Pre vs. Post 4	0.5	-1.13	0.2623	-0.4	1.4
FP-Pre vs. Post 5	0.5	-1.11	0.2703	-0.4	1.4
FP-Pre vs. Post 6	0.9	-1.98	0.0509	0.0	1.8
FP-Pre vs. Post 7	1.2	-2.72	0.0076	0.3	2.1
FP-Pre vs. Post 8	1.2	-2.63	0.0098	0.3	2.1
FP-Pre vs. Post 9	0.8	-1.88	0.0633	0.0	1.7
0%-Pre vs. Post 1	3.2	-7.95	<0.0001	2.4	4.0
0%-Pre vs. Post 2	2.7	-6.84	<0.0001	1.9	3.5
0%-Pre vs. Post 3	1.9	-4.82	<0.0001	1.1	2.7
0%-Pre vs. Post 4	1.8	-4.51	<0.0001	1.0	2.6
0%-Pre vs. Post 5	1.6	-4.08	<0.0001	0.8	2.4
0%-Pre vs. Post 6	1.2	-2.56	0.0120	0.3	2.0
0%-Pre vs. Post 7	1.4	-3.46	0.0008	0.6	2.2
0%-Pre vs. Post 8	0.9	-2.27	0.0256	0.1	1.7
0%-Pre vs. Post 9	0.9	-1.91	0.0585	0.0	1.7

Appendix Table 4-18 (continued). Pair-wise comparisons with 95% confidence intervals of the seven-day average temperature response (Δ7DTR) at the F/N break locations for each combination of treatments for each post-harvest year.

Comparison	Δ7DTR F/N Break			95% CI	
	Estimate	t-Value	P-value	Lower	Upper
100% vs. FP-Post 1	0.3	0.48	0.6335	-0.8	1.4
100% vs. FP-Post 2	0.4	0.72	0.4729	-0.7	1.5
100% vs. FP-Post 3	0.2	0.30	0.7645	-1.0	1.4
100% vs. FP-Post 4	-0.1	-0.21	0.8339	-1.3	1.1
100% vs. FP-Post 5	0.1	0.20	0.8453	-1.1	1.3
100% vs. FP-Post 6	0.3	0.43	0.6671	-0.9	1.4
100% vs. FP-Post 7	0.2	0.33	0.7399	-1.0	1.4
100% vs. FP-Post 8	0.6	1.04	0.3029	-0.6	1.8
100% vs. FP-Post 9	0.5	0.90	0.3686	-0.6	1.7
100% vs. 0%-Post 1	2.3	4.12	<0.0001	1.2	3.5
100% vs. 0%-Post 2	2.2	3.92	0.0002	1.1	3.3
100% vs. 0%-Post 3	1.3	2.38	0.0194	0.2	2.5
100% vs. 0%-Post 4	1.2	2.08	0.0403	0.1	2.3
100% vs. 0%-Post 5	1.3	2.22	0.0287	0.1	2.4
100% vs. 0%-Post 6	0.5	0.88	0.3814	-0.7	1.7
100% vs. 0%-Post 7	0.4	0.65	0.5148	-0.8	1.5
100% vs. 0%-Post 8	0.4	0.62	0.5345	-0.8	1.5
100% vs. 0%-Post 9	0.6	0.93	0.3535	-0.6	1.7
FP vs. 0%-Post 1	2.1	3.73	0.0003	1.0	3.2
FP vs. 0%-Post 2	1.8	3.28	0.0014	0.7	2.9
FP vs. 0%-Post 3	1.2	1.94	0.0550	0.0	2.4
FP vs. 0%-Post 4	1.3	2.17	0.0322	0.1	2.5
FP vs. 0%-Post 5	1.1	1.90	0.0603	-0.1	2.3
FP vs. 0%-Post 6	0.3	0.43	0.6700	-1.0	1.5
FP vs. 0%-Post 7	0.2	0.28	0.7770	-1.0	1.4
FP vs. 0%-Post 8	-0.3	-0.45	0.6556	-1.5	0.9
FP vs. 0%-Post 9	0.0	0.03	0.9772	-1.2	1.3

CHAPTER 5 - STREAM DISCHARGE, TURBIDITY, AND SUSPENDED SEDIMENT EXPORT

Greg Stewart, William Ehinger, and Stephanie Estrella

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5-1. ABSTRACT

Stream discharge, turbidity, and suspended sediment were monitored as part of a larger study evaluating the effects of riparian buffer length associated with clearcut timber harvest on non-fish-bearing watersheds in western Washington. The study used a spatially blocked and replicated Before-After Control-Impact design at the watershed scale and was restricted to headwater (<45 ha) basins with relatively competent lithologies. Two experimental blocks, each consisting of one unharvested (Reference) basin and three treatment basins, were included in the study. Treatment basins were clearcut outside of a riparian management zone (RMZ) and received one of three riparian buffer treatments: (1) a two-sided 50-ft (15.2-m) no-harvest buffer along 100% of the perennial stream network; (2) a Washington State Type N Forest Practices (FP) buffer (minimum 50% of the perennial stream network buffered); or (3) an unbuffered RMZ (i.e., 0% buffer). In addition to RMZ buffers, all basins received unstable slope buffers and a two-sided 30-ft (9.1-m) equipment limitation zone along all stream channels as required by Washington Forest Practices rules. The data were initially analyzed after two years of post-treatment monitoring (see McIntyre *et al.* 2018), but monitoring was extended for an additional six years. This report evaluates changes for the entire eight-year post-harvest monitoring period. Consistent with the findings in the two-year analysis, we observed discharge changes in both blocks and all three buffer treatments. Treatment effects varied with buffer treatment and climate, and relative changes in discharge (ΔQ%) tended to track monthly precipitation patterns. Overall, increases in discharge following harvest were greatest in basins that received the most rainfall and may have had the greatest change in evapotranspiration, and in the FP and clearcut (0%) buffer treatments where there was less residual vegetation in riparian areas. Late summer discharge decreased in both of the watersheds where RMZ buffers were retained along 100% of the perennial channel, presumably because of increased evapotranspiration rates in the residual vegetation during periods of little rainfall. Results from the two-year and eight-year monitoring periods were similar, but the extended monitoring period was useful for illustrating seasonal treatment effects and the relationship between treatment effects and estimated monthly precipitation.

5-2. INTRODUCTION

Forest harvest effects on hydrology have been well-studied worldwide with review articles focused on the Pacific Northwest (Moore and Wondzell 2005; Grant *et al.* 2008), western United States (Bowling *et al.* 2000), United States (Stednick 1996), Europe (Robinson *et al.* 2003), and globally (Bosch and Hewlett 1982; Brown *et al.* 2005; Zhang *et al.* 2017). The primary research behind these reviews has incorporated a range of methodologies and focused on different parts of the hydrologic regime, but collectively it provides strong support for the hypothesis that discharge generally increases after harvest because of the role that forests play in partitioning precipitation into evapotranspiration (ET) and runoff (Bosch and Hewlett 1982; Moore and Wondzell 2005; Zhang *et al.* 2017; Goeking and Tarboton 2020).

If we ignore storage, runoff is the difference between precipitation and ET in a simple water balance equation (Zhang *et al.* 2004). Therefore, total catchment ET represents an upper bound on the change in discharge that could occur immediately following forest harvest. Based on a range of empirical equations and observed data from around the world, Zhang and colleagues (2001; 2004) found that ET accounts for a large percentage (>80%) of precipitation in relatively dry forests (e.g., <1000 mm of annual precipitation) and may account for as much as 50% of precipitation in wet forests (>2500 mm of annual precipitation). This explains why annual runoff is more sensitive to forest cover change in water-limited watersheds than in energy-limited watersheds (Zhang *et al.* 2017).

While discharge generally increases following harvest, the magnitude of the response varies with a range of factors including the season or storm period being examined (Buttle and Metcalfe 2000; Brown *et al.* 2005; Grant *et al.* 2008; Kuras *et al.* 2012; Winkler *et al.* 2017), the spatial pattern and amount of forest harvest (Abdelnour *et al.* 2011; Zhang *et al.* 2017), the climate and hydrologic zone (Robinson *et al.* 2003; Grant *et al.* 2008; Zhang *et al.* 2017), and soils and vegetation (Robinson *et al.* 2003; Neary 2016). Discharge changes are expected to be largest in small watersheds that are completely harvested and smallest in large watersheds with patch cuts that continue to contribute to ET (Zhang *et al.* 2017; Goeking and Tarboton 2020). Discharge changes following forest harvest are not always large however, and seasonal discharge may actually decrease following harvest (Winkler *et al.* 2017; Gronsdahl *et al.* 2019), especially if ET rates in the remaining vegetation increase during dry periods (Biederman *et al.* 2014).

Discharge changes are expected to be greatest immediately following harvest and slowly decline over subsequent decades as vigorous young vegetation with higher evapotranspiration rates becomes established (Du *et al.* 2016; Coble *et al.* 2020); however, discharge increases and decreases may still be measurable up to 40 years after harvest (Perry and Jones 2017; Safeeq *et al.* 2020). Long-term decreases in base flow are especially concerning because of the potential impact to aquatic species (Gronsdahl *et al.* 2019; Coble *et al.* 2020).

Forest practices have the potential to not only affect discharge but may also affect headwater sediment supply by affecting a range of processes including road surface erosion, windthrow, and bank erosion (Roberts and Church 1986; Grizzel and Wolff 1998; Araujo *et al.* 2013). The combination of changes in flow and/or sediment supply can affect the frequency and magnitude of sediment transporting events (Gomi *et al.* 2005; Alila *et al.* 2009; Kaufmann *et al.* 2009). Sediment routing through a basin is complex and subject to changes in bedforms, large wood, and other channel features that alter hydraulic resistance, shear stress, and in-channel sediment storage (Buffington and Montgomery 1999; Jackson *et al.* 2001; Kaufmann *et al.* 2009). A source for concern is that forest practices can increase suspended sediment loads and sediment export (MacDonald *et al.* 2003; Reiter *et al.* 2009; Klein *et al.* 2012), which can have deleterious effects on fish (Kemp *et al.* 2011) and stream-associated amphibians (Grialou *et al.* 2000; Stoddard and Hayes 2005).

One of the most common and powerful approaches for measuring runoff response in smaller watersheds (<100 km^2) is based on paired watersheds (Neary 2016; Zhang *et al.* 2017). The paired watershed approach is commonly used in evaluating the effects of forestry because it is one of the few empirical approaches that implicitly deals with climate variability by comparing two basins subject to the same climatic conditions under different forest treatments (Brown *et al.* 2005; Grant *et al.* 2008). Like all approaches, however, the paired watershed approach does have limitations including the possibility that the pre-treatment period will not include the same range of conditions as the post-treatment period, or that the treated and reference basins may change for reasons other than the treatment (e.g., fire).

5-3. STUDY DESIGN

Discharge, turbidity, and suspended sediment were measured in the eight basins of the Olympic (OLYM) and Willapa 1 (WIL1) blocks of the Washington Forest Practices Adaptive Management Program's "Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington" study (McIntyre *et al.* 2018). The 'Type N Hard Rock study' (hereafter, Hard Rock Study), as it was known, was a spatially-blocked headwater stream study that used a paired-watershed Before-After Control-Impact (BACI) study design to evaluate changes in riparian vegetation, wood recruitment and loading, stream temperature, discharge, sediment export, nutrient export, litterfall and detritus, channel characteristics, periphyton, macroinvertebrates, and stream-associated amphibians following harvest in headwater streams above the point where fish presence was considered likely. The Hard Rock Study was just one of a series of paired watershed studies intended to evaluate whether harvest under the current Forest Practices rules for western Washington were meeting functional objectives reflected in Schedule L-1 of the Forest Practices Habitat Conservation Plan (FPHCP; WADNR 2005, Appendix N).

A primary criterion for the Hard Rock Study was that each of the study watersheds had to contain a population of steam-associated amphibians. Therefore, the study was limited to headwater basins with relatively competent lithologies (i.e., hard rock) in the three westside physiographic regions where Forest Practices designated amphibians were likely to be present in large numbers (**Table 5-1**). The study also required that watersheds could be clear-cut harvested except for treatment riparian management zones (RMZ) and other leave-tree areas required by Forest Practices rules (e.g., unstable slope buffers). The OLYM and WIL1 blocks were chosen for discharge and suspended sediment monitoring because those blocks were complete (i.e., contained all three treatments) and were readily accessible.

Table 5-1. 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)
	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 (Strahler 1952)
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)
	Harvest timing	Buffer treatment sites: harvest Apr 2008–Mar 2009 References: no harvest
	Area owned	>80% owned by single participating landowner

5-3.1. EXPERIMENTAL TREATMENTS

The study used three buffer treatments associated with clearcut harvest and riparian buffers of variable length and a reference with no timber harvest as defined below:

1) **Reference (REF):** unharvested reference with no timber harvest activities within the entire study basin 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 a current Forest Practices no-harvest riparian leave-tree buffer (i.e., two-sided 50-ft [15.2-m]) along ≥50% of the RMZ, and

4) **0% treatment (0%):** clearcut harvest with no riparian leave-tree buffer retained within the RMZ.

As required, the RMZ's were given a buffer width of 50 ft (15.2 m) measured horizontally from the bankfull channel. In all treatments, a 30-ft (9.1-m) equipment limitation zone was maintained along all Type Np (non-fish) and Ns (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 basins, RMZ buffers were required for the five categories of sensitive sites (WAC 222-16-010): side-slope[1] and headwall[2] seeps, headwater springs[3], Type Np intersections[4] and alluvial fans[5]. 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.

The FP treatment buffer length followed rules which require a two-sided buffer along a minimum of 50% of the Type Np length—a minimum 1000 ft (305 m) length riparian buffer directly upstream of the F/N (fish/no-fish) break with additional riparian buffers centered on sensitive sites.

5-4. SITE DESCRIPTION

The OLYM and WIL1 blocks were both located in the western Washington coastal region. Although this was a paired watershed study, the basins were not all adjacent. In the OLYM block, the REF, FP, and 0% treatments were within 4 km of each other, but the 100% treatment was approximately 50 km away. In the WIL1 block, the REF, 100%, and FP treatment were adjacent, but the 0% treatment was about 27.5 km away (**Figure 5-1**).

The western Washington coastal region is exposed to storms coming in from the Pacific Ocean and average annual precipitation ranges from 2,240 mm/yr (88 in/yr) in the WIL1-0% to 3,750 mm/yr (147 in/yr) in the OLYM-REF, OLYM-FP, and OLYM-0% based on 1981−2010 PRISM climate estimates (PRISM Climate Group 2020). Most of the precipitation falls as rain, but snow and rain-on-snow events are possible in the winter in these basins and snow was observed for brief periods in one or more of the Olympic block basins in 2008, 2009, 2010, and 2011. Field crews noted that the snow tended to be transient and snow depths rarely exceeded 6 inches (152 mm).

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[1] 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.

[2] 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.

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

[4] The intersection of two or more Type Np Waters.

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

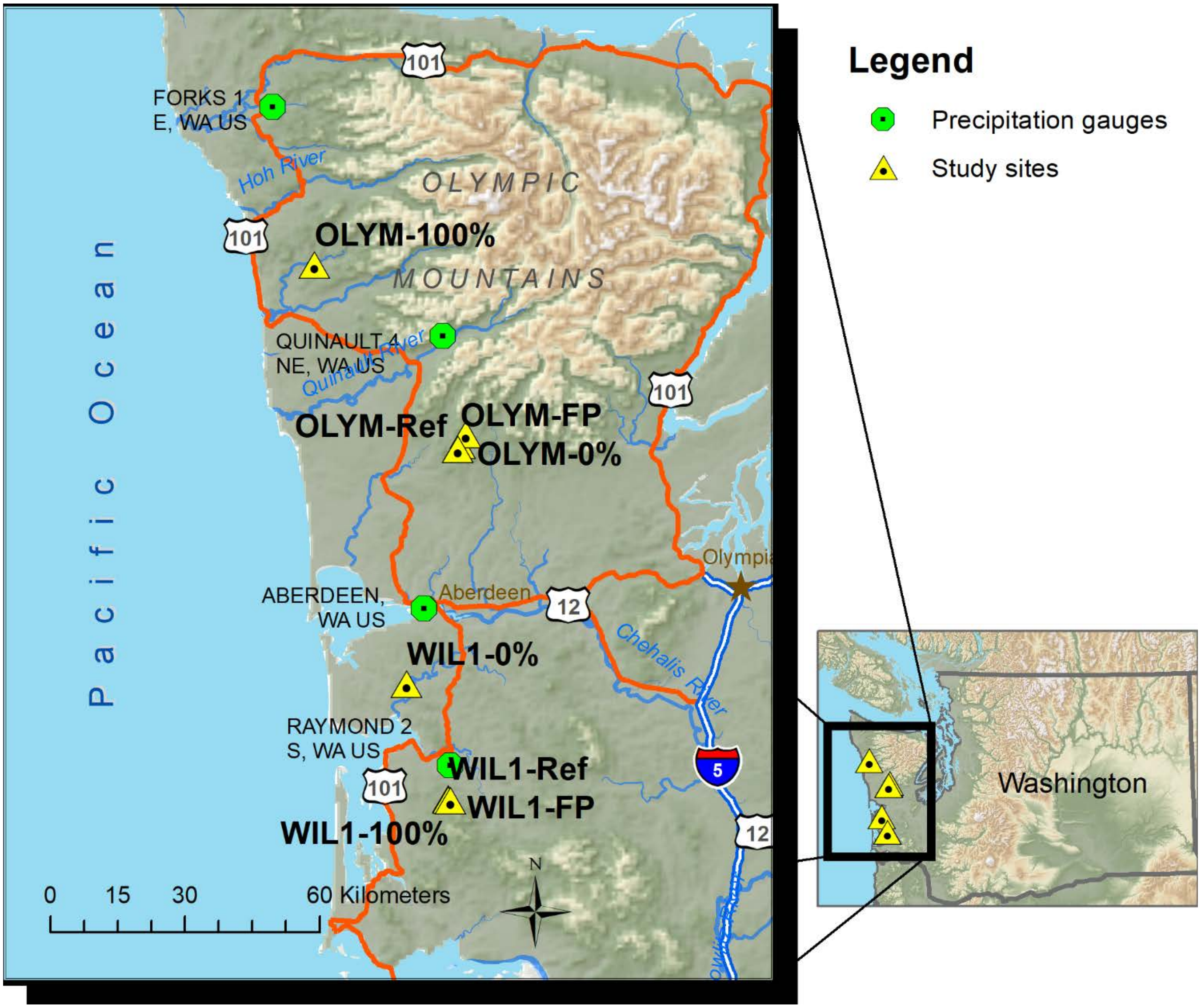

Figure 5-1. The approximate locations of the eight headwater basins included in this study (yellow triangles) and the nearest permanent precipitation gauges (marked in green; NOAA, 2020). The WIL1-REF, WIL1-100%, and WIL-FP are adjacent and shown by overlapping symbols.

Stands in the WIL1and OLYM blocks are dominated by conifers, primarily of western hemlock (*Tsuga heterophylla*) and Douglas-fir (*Pseudotsuga menziesii*). All of the basins were underlain by basalt flows and flow breccias with the exception of the OLYM-100% basin which was underlain by tectonic breccia and the WIL1-0% basin which was underlain by terraced deposits (**Table 5-2)**. In preparation for harvest, spur roads were constructed but no new road crossings were installed.

In each watershed, a flume was installed at the nearest appropriate location upstream of the designated F/N break. In two basins (OLYM-REF and OLYM-100%) there was no suitable location for a flume so stage height was measured at the upstream end of a road culvert. Basin areas above the flow measurement point varied from 11.8 to 44.3 ha (29.1 to 109 ac) and basin elevations ranged from 87 to 481 m (285 to 1578 ft; **Table 5-2**). Detailed basin maps can be found in **Appendix 2A** and more complete descriptions of the basins and buffer treatments can be found in McIntyre and colleagues (2018).

Table 5-2. Basin area and harvest above the flow gauge located near the bottom of each watershed.

Block	Treatment	Min Elev (m)	Max Elev (m)	Area (ha)	Clearcut (%)	Channel Buffered (%)	Stream Gradient (%)	Lithology
OLYM	REF	214	481	44.3	0	-	18	Basalt flows and flow breccias
	100%	103	297	22.1	45	100	27	Tectonic breccia
	FP	277	445	17.3	88	62	25	Basalt flows and flow breccias
	0%	243	481	13.1	100	0	31	Basalt flows and flow breccias
WIL1	REF	195	388	11.8	0	-	19	Basalt flows and flow breccias
	100%	217	418	26.2	89	100	18	Basalt flows and flow breccias
	FP	185	407	14.4	94	73	19	Basalt flows and flow breccias
	0%	87	225	27.7	100	0	16	Terraced deposits

5-5. METHODS

5-5.1. DATA COLLECTION

Discharge and suspended sediment data were collected between approximately 1 October 2006 and 30 September 2017 (water years 2007-2017) using a system from Forest Technology Systems (FTS; www.FTSenvironmental.com) consisting of an:

- Ott PS 1 pressure transducer,
- DTS-12 turbidity sensor,
- HDL1 datalogger,
- Teledyne ISCO 6712C portable pump sampler, and
- Forest Technology Systems StreamTrac® software.

The FTS systems were installed prior to 1 October 2006 in all the basins except for the OLYM-REF and the WIL1-100% which were not instrumented until January 2007 and August 2007, respectively.

In most basins, stage height was measured at 10-minute intervals within a stilling well in a 46-cm (18-in) or 61-cm (24-in) Montana-style Parshall flume with staff gauges used to check for drift in stage height measured by the pressure transducer. There were 21 events where the stage height recording system failed and discharge was estimated using regression against the REF basin. The events ranged in length from less than 1 day to 175 days with a median of 9 days. In total less than 1.5% of discharge values were affected. Reasons for the failures included: lightning strike, tree fall on the equipment enclosure, tree fall on the flume, transducer malfunction, an animal chewed through solar power cable, and undiagnosed power supply issues. In the culvert basins (OLYM-REF and OLYM-100%) discrete flows were measured using a Swoffer flow meter or, at very low flows, by measuring the time to fill a bucket from the culvert. These measurements were combined to build a single discharge-stage height curve for the OLYM-REF and OLYM-100% basins.

Turbidity and suspended sediment were measured near the flume or culvert. Turbidity was measured at 10-minute intervals using the DTS-12 turbidity sensor. The sensor took 100 readings over a five-second period and recorded the summary statistics of those readings.

In addition to the turbidity measurements, the FTS system was programed to conduct Turbidity Threshold Sampling (TTS; Lewis and Eads 2009). When stage height and turbidity exceeded specified thresholds for two consecutive measurements, a water sample was collected using the Teledyne ISCO 6712C portable pump sampler. TTS thresholds ranged from 10 to 1,600 Nephelometric Turbidity Units (NTUs) on both the rising and falling limbs. The ISCO sample bottles were collected within several days of each storm event and then analyzed for suspended sediment concentration (SSC). TTS sampling ensured that water samples were collected across the full range of turbidity values that occurred at each sampling location during the study period.

5-5.2. ANALYSIS

The paired watershed approach used in this study is based on a BACI study design that involves comparisons between reference and treatment basins in both the pre-treatment and post-treatment periods. In this design, the change in discharge can be analyzed using either an analysis of variance (ANOVA) or time-series frameworks. The ANOVA approach is conceptually based on the differences of differences between pre-treatment and post-treatment in the reference and treatment. The time-series approach uses regression to estimate the relationship between reference and treatment basins in the pre-treatment period. That pre-treatment relationship is then applied to the post-treatment period with the post-treatment reference basin discharge being used to estimate the natural (e.g., expected non-treatment) streamflow for the treated basins. With the time-series approach, the treatment effect is the difference between the observed discharge and the predicted natural streamflow. Unlike the ANOVA approach, the time-series approach is not greatly affected by differences in climate between the pre- and post-treatment periods.

We used the time-series approach to analyze discharge and the ANOVA framework to evaluate changes in suspended sediment export. All analyses were conducted using 64-bit R version 4.0.2 (R Core Team 2020).

5-5.2.1. Discharge

We applied the same forward regression time-series method used in the initial two-year analysis (McIntyre *et al.* 2018), a reverse regression approach as described by Safeeq and colleagues (2020), and a coarse monthly time-series analysis following the example of Watson and colleagues *(*2001). All three sets of analyses accounted for:

a) temporal autocorrelation in the measurements,
b) seasonal variation in the relationships, and
c) the non-linear nature of some relationships.

5-5.2.1.a. Temporal autocorrelation

A key assumption in regression is that the regression residuals are independent, yet sequential discharge measurements taken at a single location tend to be serially autocorrelated (**Figure 5-2**). If not accounted for the autocorrelation may result in poor estimation of the regression coefficients and will lead to an under-estimation of the error variance. One way to deal with autocorrelation is to model it. The generalized least squares (GLS) function in the Linear and Nonlinear Mixed Effects Models package (Version 3.1-148; Pinheiro *et al.* 2020) allows the specification of an autocorrelation structure. For example, if we apply ARMA(1,2) structure (described in **Section 5-5.2.2.**) to the autocorrelated data shown in **Figure 5-2**, the serial autocorrelation is accounted for and the residuals are no longer serially correlated (**Figure 5-3**). For this reason, we used GLS with an ARMA structure for all discharge analyses.

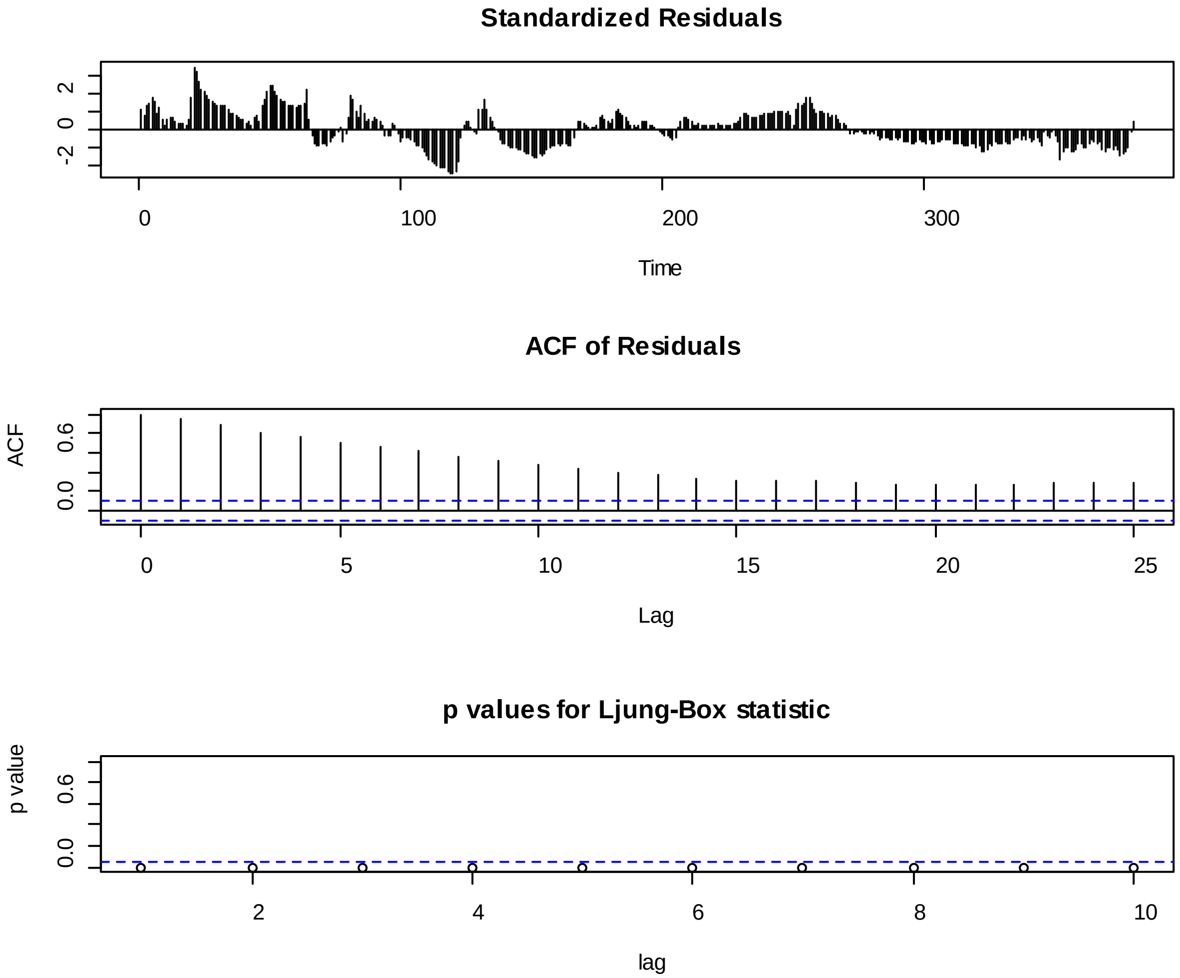

Figure 5-2. Autocorrelated residuals resulting from using Ordinary Least Squares (OLS) regression for the WIL-100% treatment. The top panel shows that positive residuals tend to be followed by positive residuals. The autocorrelation function plot, middle panel, shows the amount of correlation at different lags. The bottom panel provides Ljung-Box p-values for the statistical significance of the autocorrelation at different lags. Both the autocorrelation function (ACF) plot and Ljung Box statistics indicate that the autocorrelation is significant out through several lags. The autocorrelation affects our estimation of our model coefficients and error.

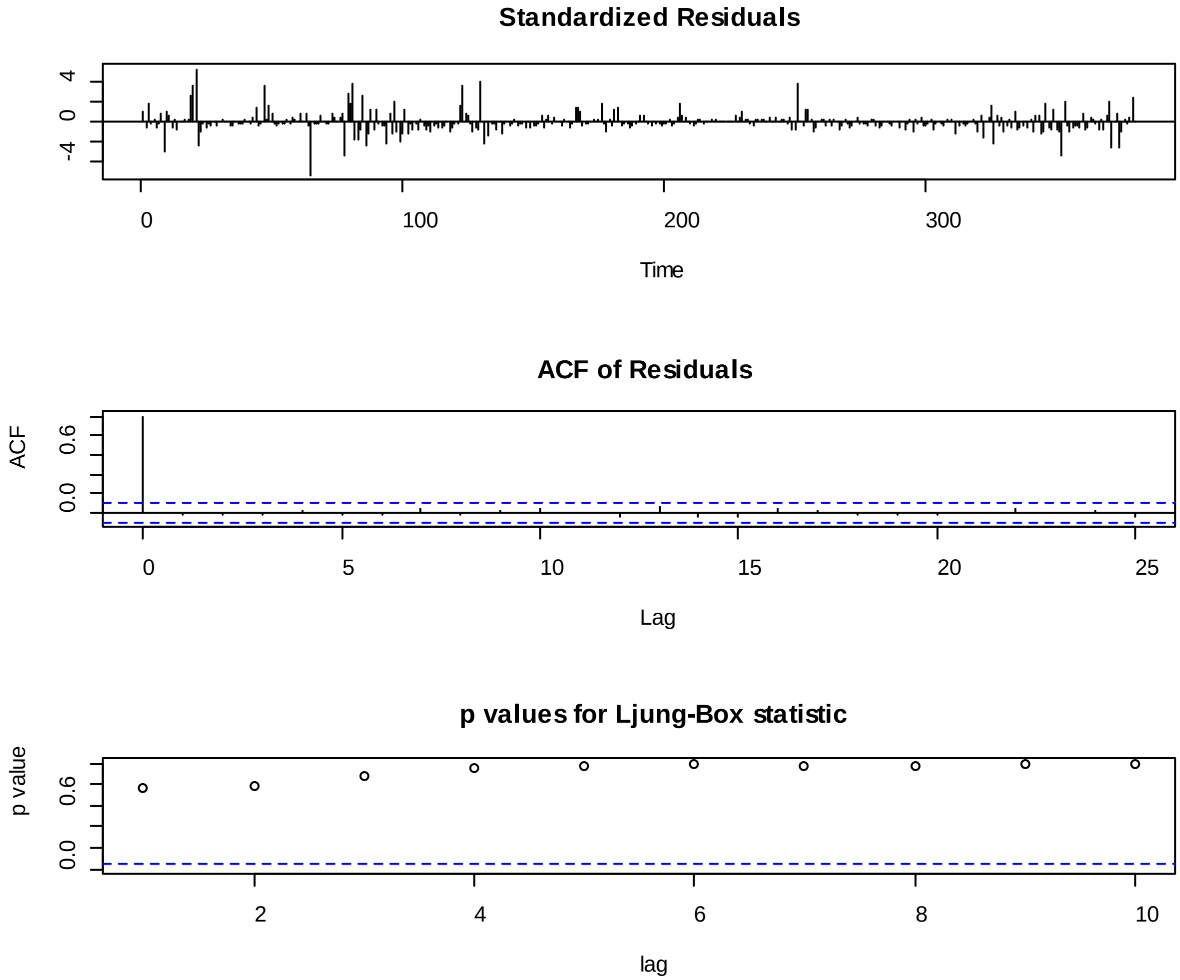

Figure 5-3. Uncorrelated time series residuals in WIL1-100% after using GLS with ARMA(1,2).

5-5.2.1.b. Seasonal variation

Another pattern we sometimes observed in our discharge data was periodic variation in the residuals (e.g., left panel, **Figure 5-4**). Periodic variation may result from differences in runoff timing associated with basin morphometry. To deal with this pattern when it was observed, we tried including a seasonal term [$Acos(\lambda t + \psi)$] in our regression models. This seasonal term explicitly assumes a sinusoidal seasonal trend in the relationship between the two catchments (Watson *et al.* 2001). Because the seasonal term is not a linear function, we used a trigonometric identity to convert it into a function that could be estimated with regression (Stolwijk *et al.* 1999):

$$Acos(\lambda t + \psi) = \beta_a \sin(\lambda t) + \beta_b \cos(\lambda t) \quad \textbf{(Eq. 5-1)}$$

where: A is the amplitude of the seasonal variation ($A = \sqrt{\beta_a + \beta_b}$),
λ is the period expressed in radians (e.g., $\frac{2\pi}{365.25}$ and/or $\frac{2\pi}{182.625}$),
t is time (e.g., Julian day), and
ψ is the phase shift in the response ($\psi = \arctan\,(\beta_a/\beta_b)/\lambda$).

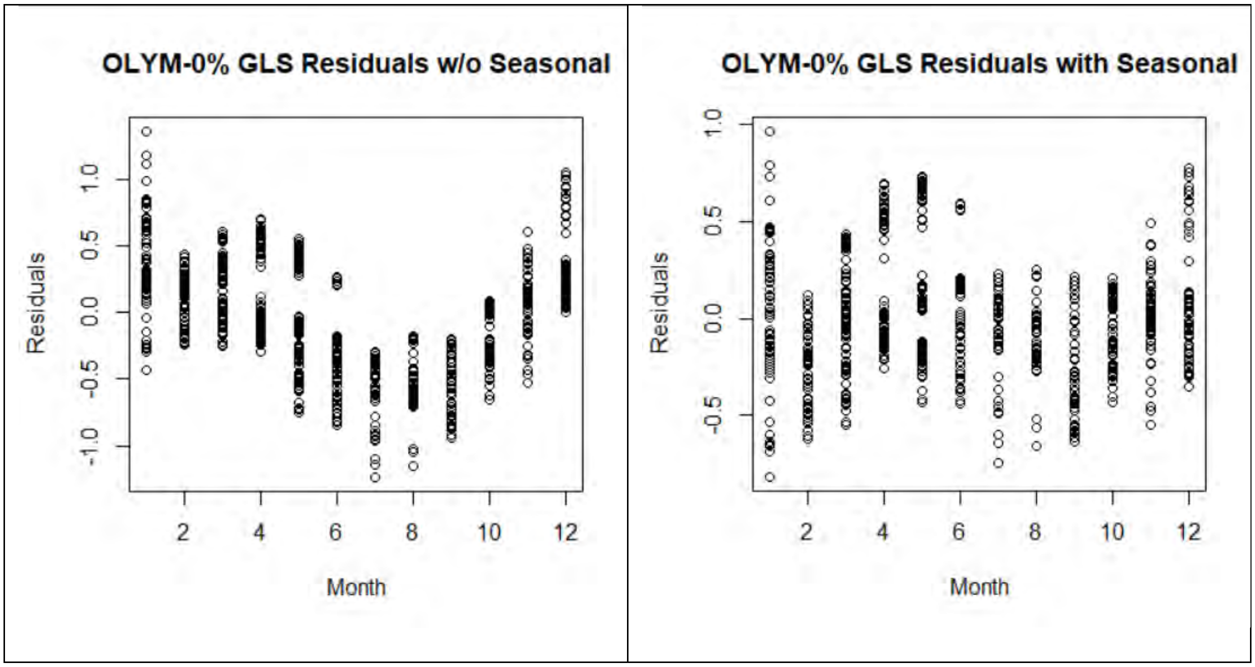

Figure 5-4. OLYM-0% GLS residuals by month with (right panel) and without (left panel) an annual seasonal term.

5-5.2.1.c. Non-linear relationships

The relationship between discharge in two different basins was not always linear through the range of measurement (e.g., **Figure 5-5**) and we found that by adding a single quadratic term [e.g., log(ref^2)] we could sometimes significantly improve the model fit. Therefore, all models were evaluated with and without a quadratic term and the most parsimonious model was chosen.

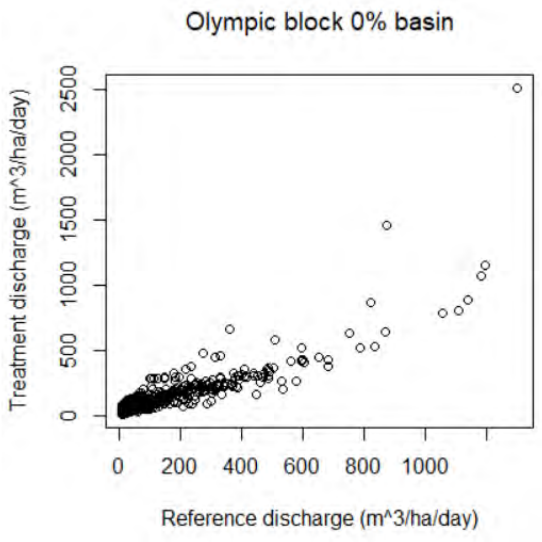

Figure 5-5. Non-linear relationship at high flows between OLYM-0% and OLYM-REF.

5-5.2.2. Estimating the Change in Daily Discharge Associated with the Treatment

To reconstruct the natural daily streamflow, or streamflow that would have been observed in the absence of the treatment effect, we:

a) estimated the pre-treatment relationship between reference and treatment basin discharge;
b) reconstructed the natural streamflow for the post-treatment period using either **forward** or **reverse** regression; and,
c) determined the treatment effect by differencing predicted natural streamflow and the observed streamflow.

5-5.2.2.a. Pre-treatment relationship between treatment and reference basins

Discharge was measured at 10-minute intervals and was summed to a total daily discharge for the daily regression analysis. Because discharge in the Olympic reference basin was truncated to a single value at low flows, the development of daily regression relationships only included those days with measured discharge to avoid introducing bias.

Regression was conducted using the GLS function in the nlme package (Version 3.1-148; Pinheiro *et al.* 2020). We tested sub-models with fewer terms while looking for the most parsimonious model, but the maximal model that we evaluated was:

$$log(Q_{trt})_t = \beta_0 + \beta_1 log(Q_{ref})_t + \beta_2 log(Q_{ref})_t^{\,2} + Acos(2\pi t/365.25 + \psi) + \varepsilon_t \quad \textbf{(Eq. 5-2)}$$

where: $log(Q_{trt})_t$is the natural logarithm of the area-weighted measured daily discharge in a buffer treatment basin on pre-harvest day *t*,
$log(Q_{ref})_t$ is the natural logarithm of the area-weighted measured daily discharge in a reference basin on pre-harvest day *t,*
β_0 is the model intercept,
β_1 and β_2 are slope coefficients for the model,
A is the amplitude of seasonal variation in the relationship between reference and treatment,
ψ is a phase shift, and
ε_t is an error term.[6]

In GLS, the regression error term (ε_t) can be modeled as an autoregressive AR(*p*) process where the current error is dependent on previous errors, or a moving average MA(q) process where the random error at time *t* is affected by both a perturbation at time *t* combined with perturbations taken before time *t* (additional correlated noise terms). As shown in Pinheiro and Bates (2000), the AR(*p*) model and the MA(*q*) models can be combined into an ARMA(*p,q*) model where the regression error term is:

$$\varepsilon_{\mathrm{t}} = \sum_{i=1}^{p} \phi_i \epsilon_{t-\mathrm{i}} + \sum_{j=1}^{q} \theta_j a_{t-j} + a_t \qquad \textbf{(Eq. 5-3)}$$

where: ϵ_{t-i} is an error term p days before,
Φ_i is the autocorrelation coefficient p days before,
a_{t-j} is the noise term at lag q,
θ_j is the correlation coefficient at lag q, and
a_t is white noise centered at 0 and assumed to be independent of previous observations.

We evaluated sub-models by dropping terms and performing model comparisons to identify the most parsimonious model. To help select our final model, we examined regression coefficients, residuals, fits against observed data, and Akaike Information Criterion (AIC). AIC is a commonly used model selection criterion that incorporates a lack-of-parsimony penalty for adding additional model parameters. After rejecting models that did not meet model assumptions (e.g., independent, and identically distributed, errors), we chose the model with the lowest AIC score.

Models were initially fit using Maximum Likelihood (ML) for model comparison purposes, but the final model was fit using Restricted Maximum Likelihood (REML). We used GLS diagnostic plots to verify that the errors in the final model were approximately independent and normally distributed ($\epsilon \sim \mathrm{N}(0, \sigma^2, \mathrm{I})$).

5-5.2.2.b. Forward and reverse regression for estimating natural streamflow

Natural streamflow is typically estimated by applying the pre-treatment regression relationship to the observed post-treatment reference basin discharge to predict what would have been observed

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[6] Note: The corresponding **Eq. 8-1** in McIntyre *et al.* 2018 fails to show the log transformation even though it was used.

in the absence of the treatment (Forward Regression). One problem with the forward regression approach is that the predicted discharges tend to follow the flow patterns (e.g., base flow characteristics, recession rate) of the reference basins. While ‘regression’ ensures that the differences (i.e., errors) average out, the forward regression approach can create short-term deviations between chronologically paired observed and predicted discharge that are clearly artifacts of the analytical technique. To remedy this issue, we also applied a reverse regression technique described by Safeeq and colleagues (2020). In reverse regression, we created a synthetic reference basin discharge using observations from the treatment basin and because this ‘synthetic’ reference is based on treatment observations, it does a better job of mimicking the flow patterns of the treatment. We then used the synthetic reference discharge in place of the measured reference discharge to estimate natural streamflow.

5-5.2.2.b.(i). Forward regression

When estimating natural streamflow using the forward regression approach, we combined the pre-treatment regression coefficients and the reference basin discharge over the period of record to estimate the streamflow that would have been observed in the treatment basin had the buffer treatment not occurred. [7]

$$\log\,(\widehat{Q_{trt}})_t = \beta_0 + \beta_1 \log\left(Q_{ref}\right)_t + \beta_2(\log(Q_{ref})_t)^2 + \mathrm{A}\cos(2\pi \mathrm{t}/365.25 + \psi) \qquad \textbf{(Eq. 5-4)}$$

where: $(\widehat{Q_{trt}})_t$ is the predicted discharge in a buffer treatment basin on day *t*,
$\left(Q_{ref}\right)_t$ is the observed discharge in a reference basin on day *t*, and
β_0, β_1, β_2, *A,* and ψ are all coefficients developed using **Eq. 5-2**.

5-5.2.2.b.(ii). Reverse regression to create a synthetic reference streamflow

As noted above, the reverse regression approach is like the forward regression approach except that we create a synthetic reference basin streamflow for use in **Eq. 5-4**. This synthetic streamflow record is created through two additional regression steps. In the first step, we modeled the post-treatment relationship between the reference and treatment basin discharges using GLS regression with the maximal form:

$$\begin{aligned}\log\left(Q_{ref}\right)_{t.post} = \beta_3 + \beta_4 \log(Q_{trt})_{t.post} + \beta_5 \mathrm{T} + \beta_6 \log(Q_{trt})_{t.post} * \mathrm{T} \; + \\ \beta_7 \log(Q_{trt})_{t.post}{}^2 + \mathrm{Asin}(2\pi * \mathrm{t}/365.25 + \psi) + \varepsilon_\mathrm{t} \qquad \textbf{(Eq. 5-5)}\end{aligned}$$

where: $(Q_{ref})_{t.post}$ is the observed discharge in the reference basin on post-harvest day *t*,
$(Q_{ref})_{t.post}$is the observed discharge in the treatment basin on post-harvest day *t*,
T is time in days, and
β_3, β_4, β_5, β_6, β_7, and ψ are all coefficients estimated by the model.

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[7] Note: Although forward regression can provide predictions over the entire period of record, the differences between observed and predicted are minimized in the pre-treatment period by the regression.

The model allows for decaying treatment effects with time and the interaction allows the decay rate to vary with discharge. For the OLYM-0% basin, we included a second treatment basin (OLYM-FP as Q_{trt2}) in the reverse regression because we obtained poor model fits against the reference without it.[8] As with all our other regression models, we evaluated sub-models and used AIC and model diagnostics to identify the most parsimonious model.

We then used the relationship developed with **Eq. 5-5** to generate a 'synthetic' reference discharge for the post-treatment period where:

$$\log\left(\widehat{Q_{ref}}\right)_{t.post} = \beta_3 + \beta_4 \log(Q_{trt})_{t.post} + \beta_5 T + \beta_6 \log(Q_{trt})_{t.post} * T + \beta_7 \log(Q_{trt})_{t.post}{}^2 + A\sin(2\pi t/365.25 + \psi) \qquad \textbf{(Eq. 5-6)}$$

The synthetic reference basin discharge ($\widehat{Q_{ref}}$) calculated by **Eq. 5-6** then replaces the observed reference discharge (Q_{ref}) in **Eq. 5-4** to get the predicted reverse regression discharge on day *t*.

5-5.2.2.c. Calculating a treatment effect

To account for the bias introduced by performing regression on the log-scale and re-transforming the predictions to the original scale, we applied the Duan non-parametric smearing estimator (Duan 1983) to obtain our expected discharge on the original scale.

$$\hat{Q}_{trt.unbiased} = \frac{1}{n}\sum_1^n e^{\varepsilon_n} * e^{\log\left(\widehat{Q_{ref}}\right)_{t.post}} \qquad \textbf{(Eq. 5-7)}$$

where: $\frac{1}{n}\sum_1^n e^{\varepsilon_n}$ is the Duan smearing estimator, and

$e^{\log\left(\widehat{Q_{ref}}\right)_{t.post}}$ is the forward or reverse regression prediction on the original scale.[9]

Regardless of whether the expected treatment discharge was estimated using forward or reverse regression, absolute treatment effects (TE; mm/day) were calculated as the difference between what was observed in the treatment basin and the natural streamflow that was predicted by the reference basin.

$$TE = Q_{trt} - \hat{Q}_{trt.unbiased} \qquad \textbf{(Eq. 5-8)}$$

where: Q_{trt} is the observed treatment basin discharge on day *t*, and

$\hat{Q}_{trt.unbiased}$ is the predicted (i.e., expected) treatment basin discharge on day *t*.

The relative change in discharge (TE%) was calculated by dividing TE by the expected natural streamflow:

$$TE\% = TE/\hat{Q}_{trt.unbiased} \qquad \textbf{(Eq. 5-9)}$$

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[8] This is an issue with having truncated low flows in the Reference and reporting NSE based on the entire dataset.

[9] The bias correction was not performed in the original two-year analysis, and when the estimator is greater than 1, the failure to include the estimator will decrease the predicted discharge which increases the treatment effect.

5-5.2.2.d. Frequency pairing of daily discharge

We used the Frequency Pairing approach as described by Alila and colleagues (2009) to look for changes in the frequency at which daily discharges are observed. In frequency pairing, the observed and expected treatment basin daily discharges are paired based on their historic return period as opposed to being compared chronologically.

The historic return period for expected and observed time series is determined by independently ranking observed and expected values to create a ranked dataset (m), in which the greatest observed and expected values (y) are rank 1 (m_1), the second greatest values are rank 2 (m_2), and the ranked series is monotonically decreasing over its length ($y_{m_1} > y_{m_2} > \cdots > y_{m_n}$). A cumulative frequency analysis is then used to determine the return period (T_{days}) for any given rank (m_i):

$$T_{days} = \frac{n+0.2}{m_i-0.4} \qquad \textbf{(Eq. 5-10)}$$

where: n is the total number of days in the period of observation, and
m_i is the rank for a set of frequency paired observations.

Eq. 5-10 incorporates Cunnane 1978 plotting positions that are appropriate for Q-Q plots, flood frequency curves, and the calculation of exceedance probabilities (Helsel and Hirsch 1992).

The predicted values ($\hat{y}_t$) are corrected for loss of variance resulting from the regression analysis (Green and Alila 2012). If unaccounted for, the loss of variance could inflate estimates of treatment effects in the upper tail of the frequency distribution. The correction is performed in a three-step Monte Carlo (MC) simulation. The first step is the addition of a random error sampled from a t-distribution (e_t) to the predicted values ($\tilde{y}_t = \hat{y}_t + e_t$), the second step involves ranking the updated estimates ($\tilde{y}_m$), and the third step involves repeating the first two steps over 10,000 iterations and calculating the mean ($\bar{\tilde{y}}_{m_i}$) and variance ($var(\tilde{y}_{m_i})$) for each rank. The random errors (e_t) are scaled to the standard error of each predictor variable (x_t) in the original regression using a t-distribution such that:

$$e_t = \sigma_{pred_t} * rt(\mathrm{df}_{resid} - 1) \qquad \textbf{(Eq. 5-11)}$$

where: e_t is the random error,
σ_{pred_t} is the standard error for the predictor on day t, and
$rt(\mathrm{df}_{resid} - 1)$ is a function for extracting random variates from a t-distribution with one less than the residual degrees of freedom from the initial regression model.

The standard errors for each day (σ_{pred_t}) are calculated as:

$$\sigma_{pred_t} = rse\sqrt{1 + \frac{1}{n_{pre}} + \frac{(x_t-\bar{x})^2}{\sum(x_t-\bar{x})^2}} \qquad \textbf{(Eq. 5-12)}$$

where: rse is the residual standard error from the initial regression,
n_{pre} is pre-harvest sample size,
x_t is the reference basin value on post-harvest day t, and
$\bar{x}$ is the mean of the post-harvest reference basin observations (K.C. Green, personal communication).

Confidence intervals for the variance corrected means ($\bar{\tilde{y}}$) are calculated as a combination of predictive uncertainty ($var(\tilde{y})$) and quantile uncertainty. Quantile uncertainty was estimated with a Monte Carlo simulation in which (1) a generalized extreme value (GEV) distribution was fit to the mean updated estimates ($\bar{\tilde{y}}$) using the evd package in R (Stephenson 2002); (2) samples were randomly drawn from an extreme value distribution with the fitted parameters and the samples were ranked in descending order; and (3) the process was repeated over 10,000 iterations with variance ($var(GEV)$) calculated for each rank. The variance corrected means were found to be approximately normally distributed so the 95% confidence limits for each rank were calculated using the *qnorm* function in R, such that:

$$95\%CI = qnorm((0.025{,}0.975), \bar{\tilde{y}}_i, \sqrt{var(\tilde{y})_i + var(GEV)_i}) \qquad \textbf{(Eq. 5-13)}$$

where: *qnorm* is a function in R that calculates the 0.025 and 0.975 quantiles for a normal distribution with a given mean and standard deviation.

5-5.2.3. Monthly Discharge Analysis

We also included a monthly analysis like that performed by Watson and colleagues (2001). Monthly summaries are aggregated over longer periods and are therefore less sensitive to deviations in storm timing between basins, but a disadvantage of aggregating over these longer timespans is that the pre-treatment period was relatively short (n=13-28 months) and the OLYM-REF and OLYM-100% basins had their discharges truncated at low flows. Because months vary in their length, we analyzed mean monthly discharge rather than total discharge for each month. Unlike the daily analysis, we included days with fixed (truncated) discharges to avoid the bias that would be introduced by varying the number of days used to calculate the monthly mean.

5-5.2.4. Discharge Goodness of Fit Statistics

We reported two goodness of fit statistics for each discharge regression analysis. The first is a coefficient of determination based on the likelihood-ratio test (R^2). This statistic represents the improvement of the fitted model over a null (intercept only) model and is calculated as:

$$R^2{}_{LR} = 1 - \exp\left({}^{-2}/_{n} * \left(\log Lik(x) - \log Lik(0)\right)\right) \qquad \textbf{(Eq. 5-14)}$$

where: log*Lik(x)* is the log-likelihood from the fitted model, and
log*Lik(0)* is the log-likelihood from the null model (i.e., intercept only).

R^2 is interpreted in the same manner as a coefficient of determination (r^2), with $R^2 = 0$ indicating that the model explains no variation and $R^2 = 1$ indicating that the model perfectly explains all the observed variation. The extraction of log-likelihoods and calculation of R^2 was performed using routines in the R MuMIn package (Barton 2012), and the ARMA correlation structure was incorporated into the null model so that R^2 properly reflects the adequacy of the prediction model.[10]

We also reported the Nash-Sutcliffe Efficiency (NSE) between the observed and predicted discharge on the original scale after applying the smearing estimator. NSE is a normalized statistic that determines the relative magnitude of the residual variance ("noise") compared to the measured data variance ("information") (Nash and Sutcliffe 1970). Nash-Sutcliffe efficiencies range from -Inf to 1 where NSE = 1 corresponds to a perfect match between the modelled and observed data; NSE = 0 indicates that the model predictions are as accurate as the mean of the observed data; and Inf < NSE < 0 indicates that the observed mean is a better predictor than the model. NSE was calculated using the hydroGOF package (Zambrano-Bigiarini, 2020).

5-5.2.5. Suspended Sediment

Lab-measured SSC from the ISCO bottles and the corresponding turbidity data were used to build a simple linear regression model for each site. Those regression models were then used to estimate SSC from turbidity for the entire data record. Suspended sediment export (SSE) was then calculated as the product of SSC and discharge.

The SSE data did not conform to BACI assumptions, so we did not perform statistical tests. Instead, we illustrated how SSE rates vary with discharge before and after harvest in the same ANOVA-like framework that was used in the two-year report.

In addition, we tried to examine whether the relationship between suspended sediment yield (SSY) and discharge (Q) changed between the pre-harvest and post-harvest periods. That relationship can be described by the power function:

$$Q_s = \alpha Q^B \qquad \textbf{(Eq. 5-15)}$$

where: Q_s is SSY (kg/ha/h),

Q is unit discharge (m^3/ha/h),

α is the intercept and a metric of erosion severity, and

β is the slope and an indicator of the erosive power of a stream.

If we linearized it by taking the log of both sides, we can interpret the fitted regression intercept as an indicator of erosion severity and the fitted regression slope as an indicator of erosive power (Bywater-Reyes *et al.* 2017). We fit a series of linear mixed-effects models using this approach, but eventually realized that the analyses were predicated on assumptions that could not be supported with these data for our questions. We discuss these assumptions and why we did not follow-through with a formal statistical analysis in the results.

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[10] The ARMA structure is not incorporated in predictions because the errors ($\varepsilon = y - \hat{y}$) are not known, although under the assumption that the null hypothesis is true, one would expect the errors in the post-harvest period to have the same error structure with identically distributed random residuals.

5-6. RESULTS

The discharge versus stage curves for the culvert basins (OLYM-REF and OLYM-100%) spanned up to the 99th percentile of estimated flows and the adjusted R-squared values were 0.886 and 0.904, respectively. Low flows were difficult to measure accurately and were therefore truncated at 4 and 3 L s^{-1} (7.9 and 11.7 m^3/ha/day), respectively (**Table 5-3**). Because of a poor flow versus stage height relationship in OLYM-100% for the first year of the study, only those OLYM-100% flow data collected after 1 January 2008 were analyzed.

Table 5-3. Flow rating at specific percentiles for basins where discharge was measured upstream of a culvert instead of in a flume (OLYM-REF and OLYM-100%).

	OLYM-REF		**OLYM-100%**	
Percentile	**Measured Q**	**Estimated Q**	**Measured Q**	**Estimated Q**
10	4.3	<4	0.9	<3
25	5.8	<4	1.7	3.2
50	28.5	22.5	7.0	11.5
75	120.2	52.7	25.4	24.8
90	280.4	111.2	94.0	48.5
99	392.8	326.7	220.1	136.1
100	392.8	856.7	220.1	493.6

5-6.1. REFERENCE BASIN DISCHARGE AND RELATIONSHIP TO PRECIPITATION

Precipitation was not monitored as part of this study and the nearest permanent precipitation gauging stations are Forks 1 E, Quinault 4 NE, Raymond 2 S, and Aberdeen (NOAA 2020). The local precipitation gauging stations were at lower elevations than the study basins (**Table 5-4**), and while monthly and annual precipitation is highly correlated among precipitation gages, correlations between precipitation and runoff are relatively poor even at a water year scale (**Figures 5-6** and **5-7**). To evaluate whether the poor relationships were the result of lags, we applied cross-correlation functions at both the monthly and annual scale and found that the strongest correlations were at LAG=0 indicating that the relationships were poor but not necessarily lagged. However, poor relationships between precipitation and runoff were expected given the complexities associated with ET, water storage dynamics within the basin, and runoff.

Our best estimates of precipitation in the study sites are likely to come from the gridded estimates of monthly precipitation created by the Parameter-elevation Regressions on Independent Slopes Model (PRISM; PRISM Climate Group 2020). PRISM is a regression model that uses correlations among neighboring precipitation stations to account for effects like orographic lift. Because PRISM is a regression model, the PRISM estimates are still highly correlated with the measured precipitation rather than runoff. However, we used them to examine how treatment effects varied seasonally and in response to estimated precipitation patterns as shown in **Section 5-6.5.**

Table 5-4. Precipitation gauging stations (bold) and study basins ordered from north (top) to south (bottom) with elevation in meters. Precipitation gauges are all located at lower elevations than the study basins.

Site	Elevation (m)
FORKS 1 E, WA US	**107**
OLYM-100%	200
QUINAULT 4 NE, WA US	**87**
OLYM-REF	348
OLYM-FP	361
OLYM-0%	362
ABERDEEN, WA US	**3**
WIL1-0%	156
RAYMOND 2 S, WA US	**9.1**
WIL1-REF	292
WIL1-FP	296
WIL1-100%	318

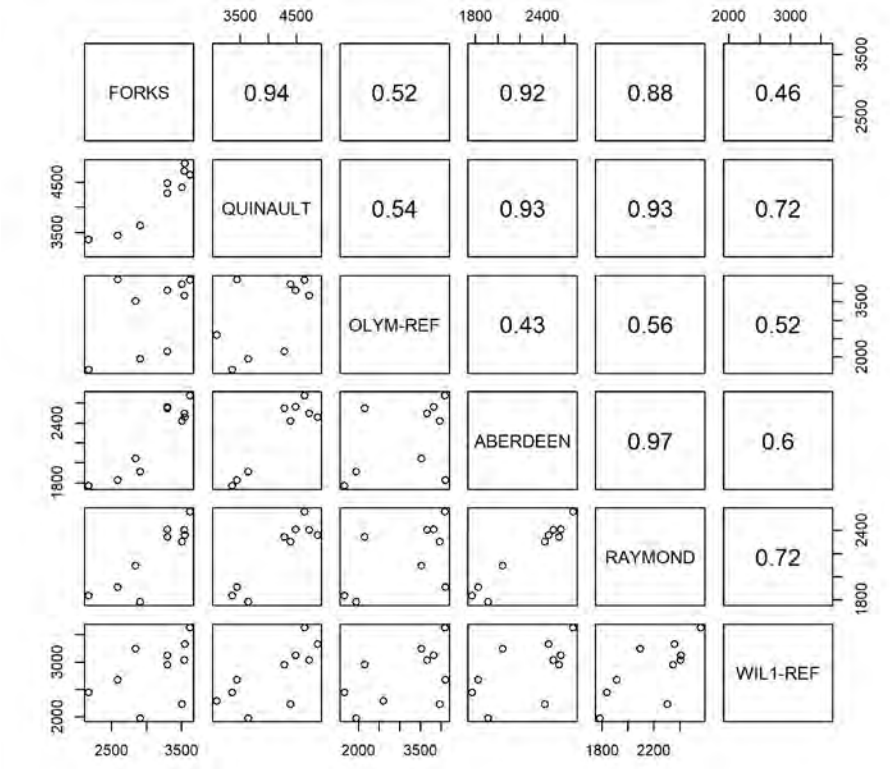


Figure 5-6. Correlation matrix between annual water year precipitation at nearby precipitation gauging stations and reference basin discharge. Basins are ordered from north to south. Correlations among precipitation gages are good, but correlations between precipitation and discharge are relatively poor.

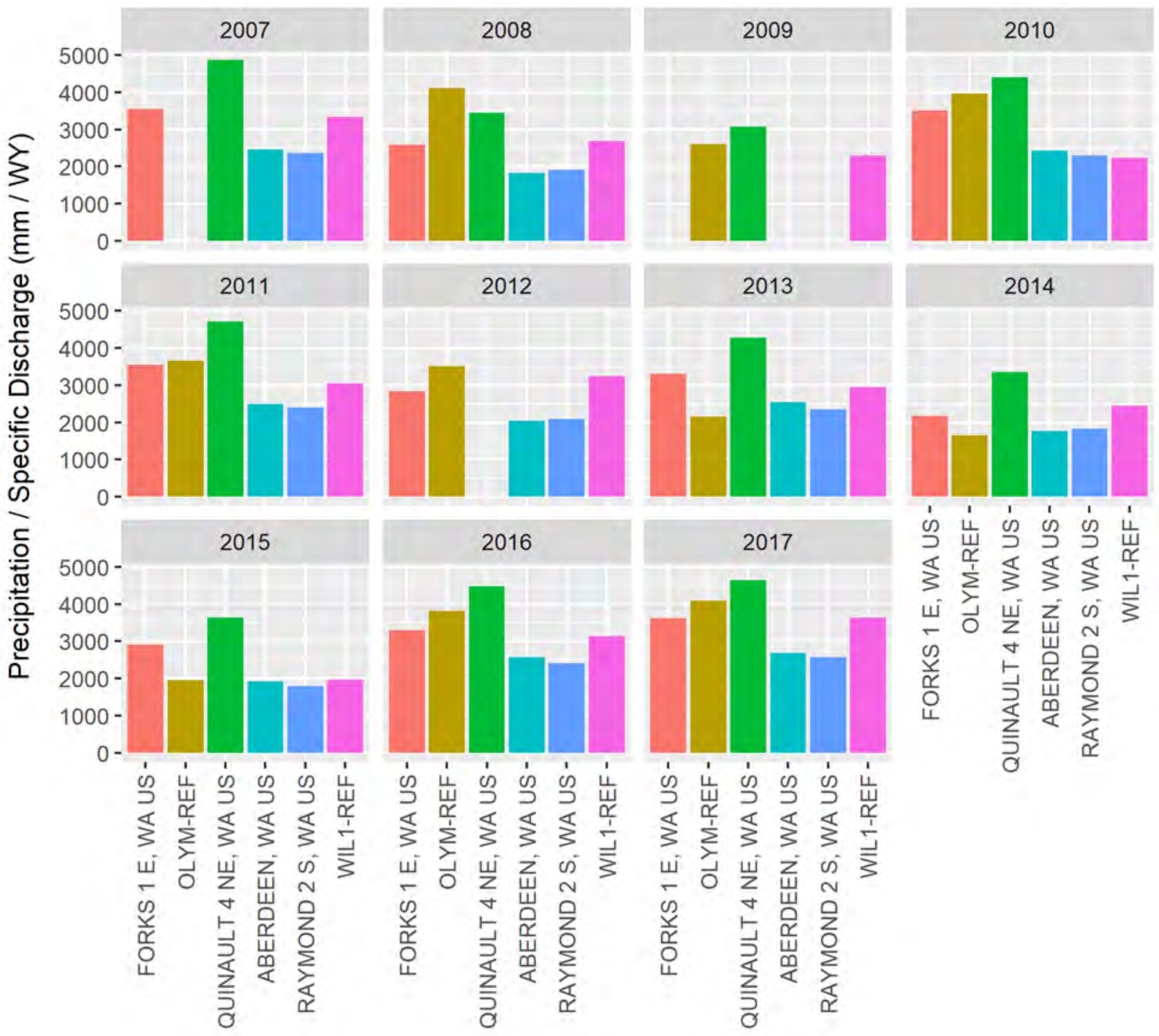


Figure 5-7. Total water year (WY) precipitation and runoff for the local low elevation precipitation gauging stations and the study reference basins. The Quinault precipitation station, which is located between the Olympic block basins, averaged over 4000 mm of precipitation per year during the study.

5-6.2. OBSERVED DISCHARGE AND CHOICE OF REFERENCE BASINS

During the 10 water years (2007-2017) in the study, daily water yield ranged from 7.8 to 1,304 m^3/ha/day in the OLYM-REF to 3.9 to 980 m^3/ha/day in the WIL1-REF (**Table 5-5**), and discharge was more highly correlated within blocks than between them (**Figure 5-8**).

We expected higher correlations among basins within blocks, largely as a function of differences in climate and the variability associated with individual storm trajectories. The Olympic basins are further north, receive more precipitation, and are at slightly higher elevations than the Willapa basins.

Plots of raw specific discharge by basin (**Figures 5-9** and **5-10**) illustrate truncated discharges at the culvert basins, differences in pre-treatment record length, and harvest timing.

Table 5-5. Statistics for daily specific discharge (m^3/ha/day).

		Daily Discharge (m^3/ha/day)			
Basin	**n (days)**	**Min**	**Mean**	**Median**	**Max**
OLYM-REF	3887	7.81	88.3	44.8	1304
OLYM-100%	3561	11.7	79.7	45.7	1673
OLYM-FP	4018	1.14	96.8	66.2	1860
OLYM-0%	4018	10.3	142	102	2512
WIL1-REF	4018	3.86	77.1	58.7	980
WIL1-100%	3653	1.18	55.1	38.9	1023
WIL1-FP	4018	5.42	66.8	53	864
WIL1-0%	4018	8.4	67.3	50.8	817

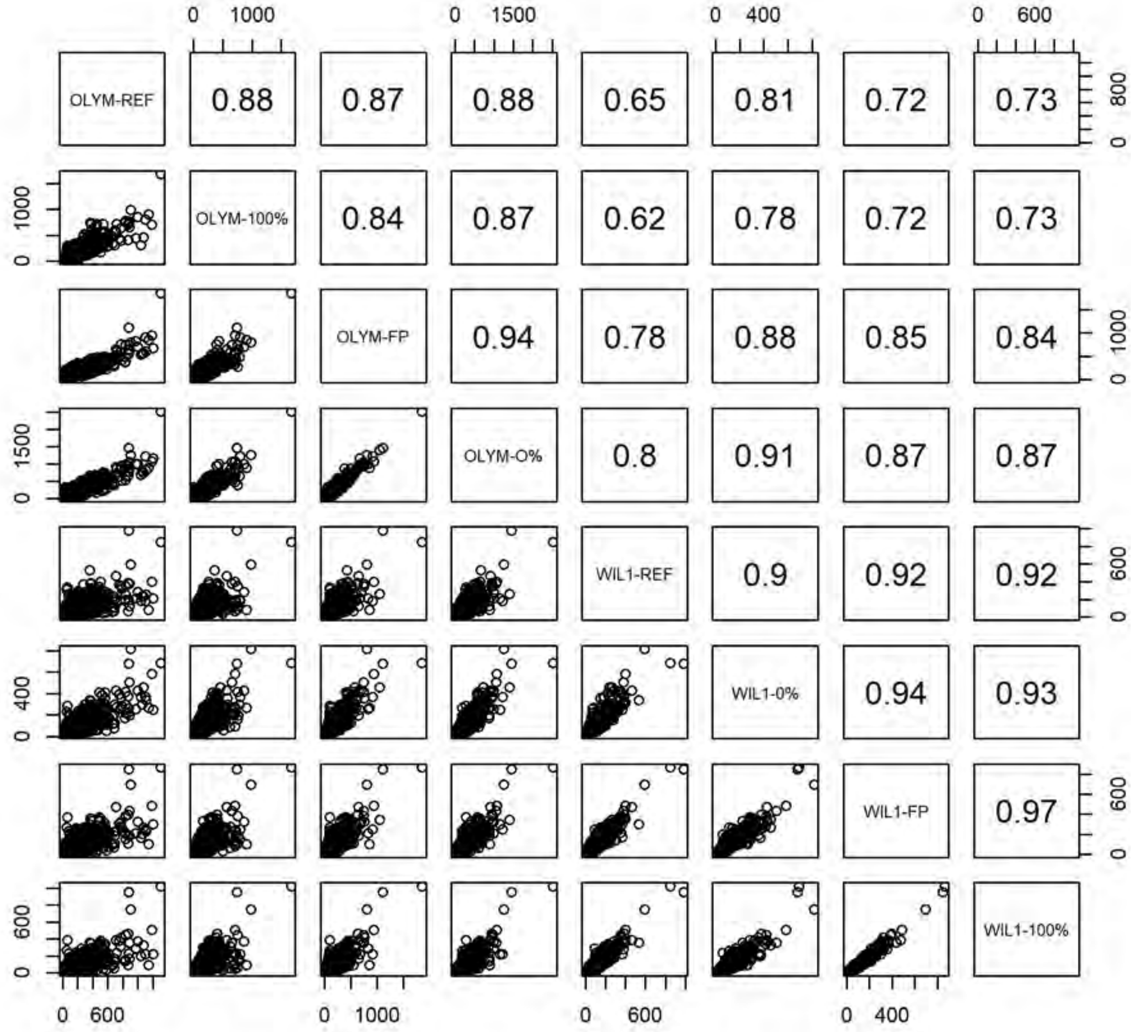


Figure 5-8. Simple correlations in daily discharge among basins excluding the truncated discharges in OLYM-REF and OLYM-100%. Correlation coefficients are greater within blocks than between blocks supporting the use of within-block references.

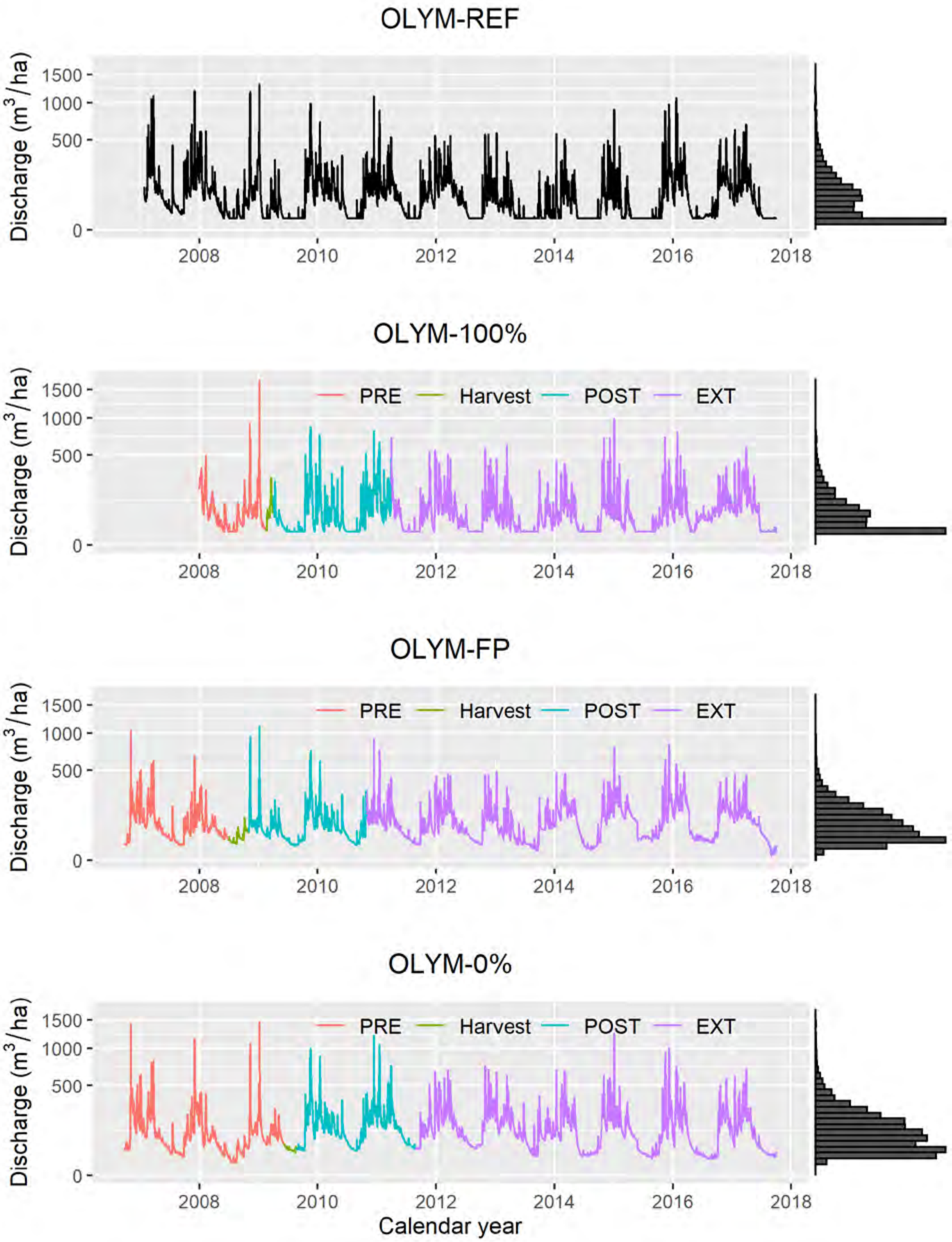


Figure 5-9. Observed daily specific discharge in the Olympic block basins. Histograms give discharge frequency and color indicates treatment period.

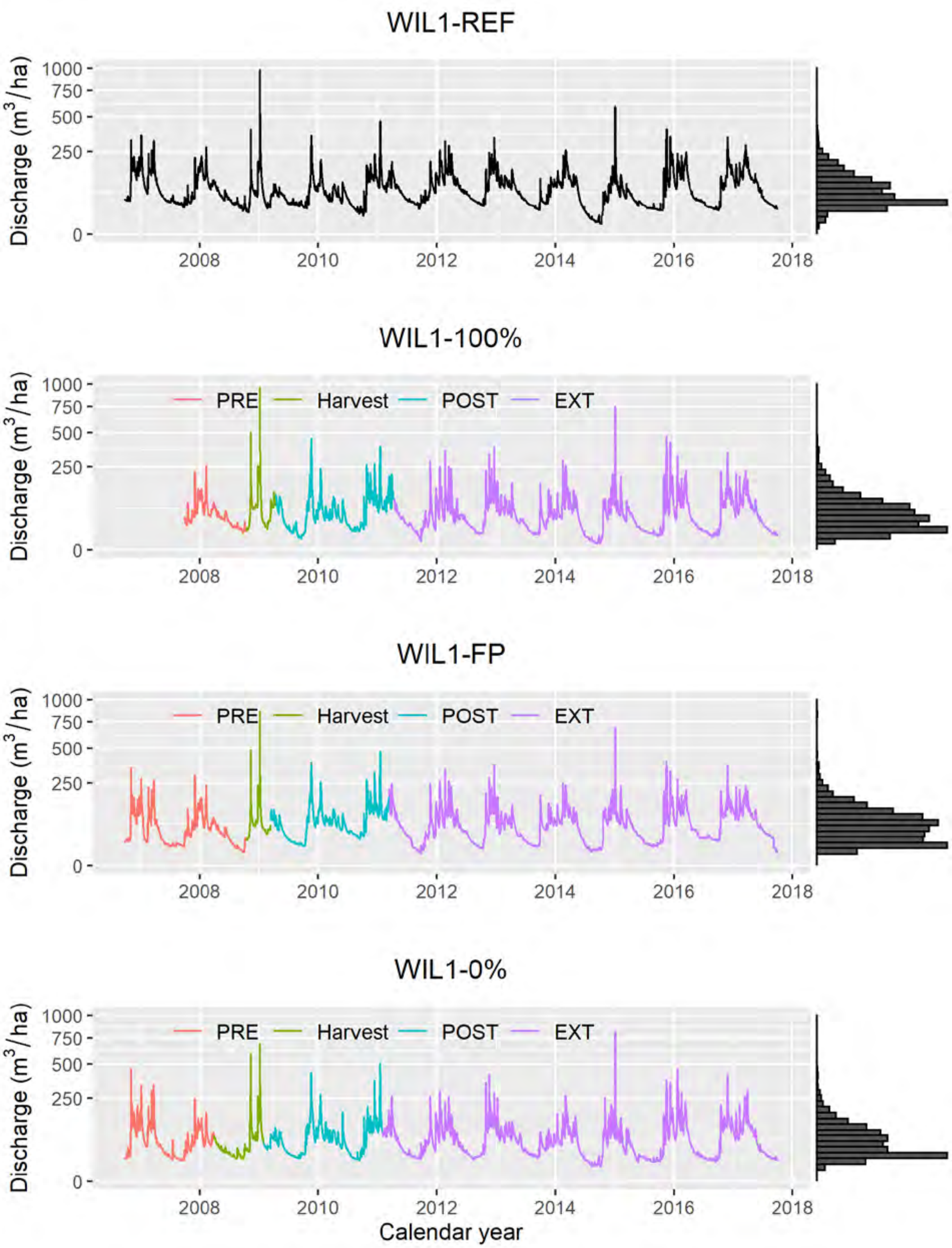


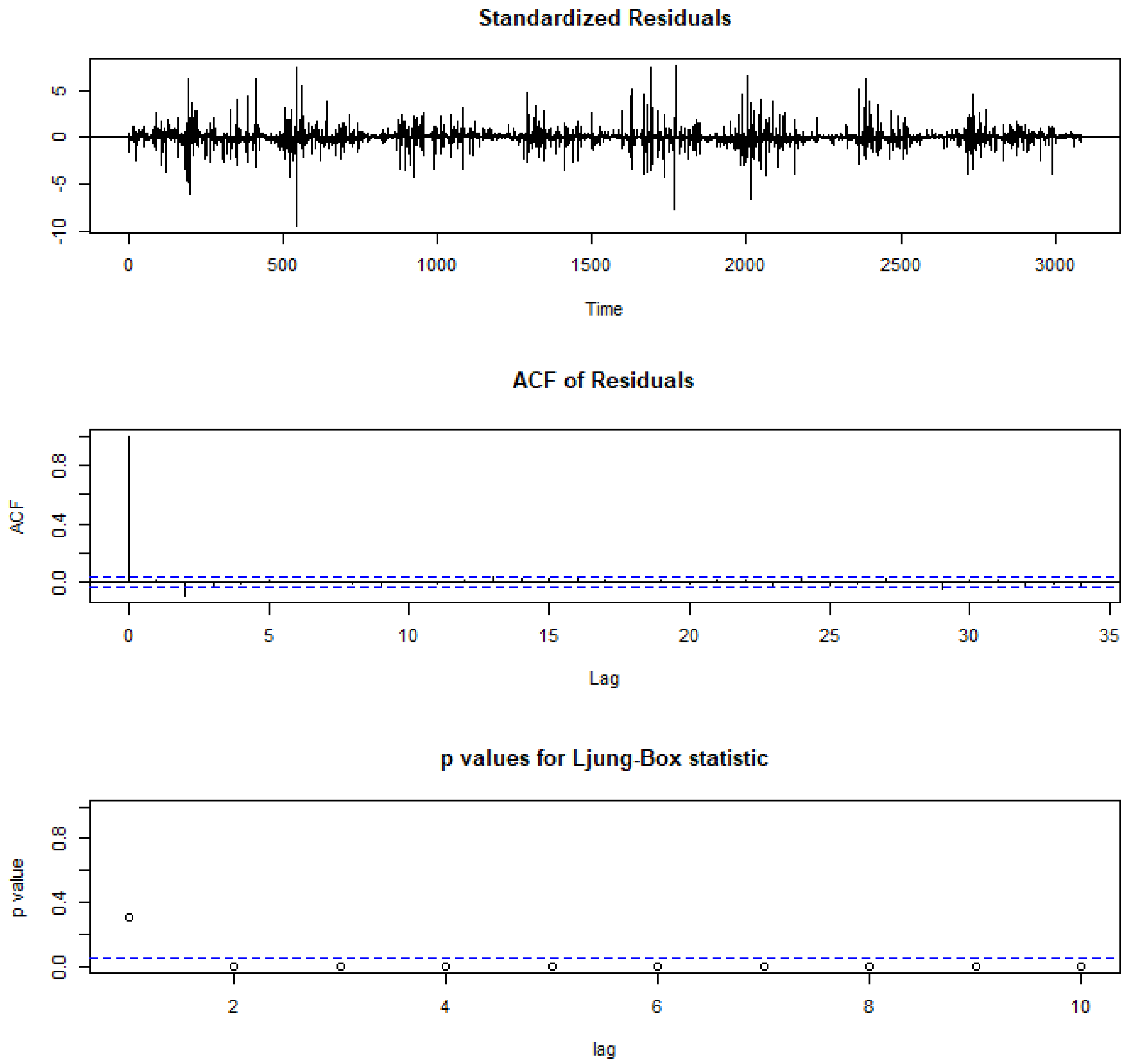
Figure 5-10. Observed daily specific discharge in the Willapa 1 block basins. Histograms give discharge frequency and color indicates treatment period.

5-6.3. CUMULATIVE AND ANNUAL CHANGES IN DAILY DISCHARGE

One of the primary questions of this study was whether discharge changed following harvest. To evaluate this, the change in daily discharge associated with each treatment was calculated using both forward and reverse regression techniques. The forward regression generally resulted in good fits with R^2 above 0.73 and Nash-Sutcliffe efficiencies above 0.75 on record lengths ranging from 342 to 784 days (**Table 5-6**). Because of the much longer post-treatment record lengths, we were only able to consistently fit ARMA(1,0) models (e.g., AR1) for reverse regression, because models with more complicated ARMA structures failed to converge. While AR1 did account for most of the autocorrelation, the residuals were still partially autocorrelated at lags greater than one (**Figure 5-11**). Because the reverse regression was only used to create the synthetic basin discharge, and we never used reverse regression error estimates, the residual autocorrelation in the reverse regression was not expected to significantly affect our interpretation. Pseudo-R-squared and NSE for the reverse regression were slightly worse than for the forward regression but over longer records (>2000 days) (**Table 5-7**).

Table 5-6. Coefficients and goodness of fit statistics for the forward regression on pre-treatment daily discharge.

Site	N (days)	ARMA	β_0	β_1	β_2	Seasonal	Smearing Factor	R^2	NSE
OLYM-100%	342	1,1	1.6	0.49	0.03	365	1.042	0.77	0.82
OLYM-FP	522	1,2	3.3	-0.23	0.09	365	1.001	0.92	0.95
OLYM-0%	784	1,2	4.2	-0.36	0.09	365	1.031	0.83	0.75
WIL1-100%	380	1,2	2.4	-0.38	0.16	182	1.016	0.83	0.92
WIL1-FP	745	1,1	3.1	-0.77	0.21	365/182	1.030	0.78	0.89
WIL1-0%	548	1,2	-0.06	0.72	0.05	182	1.032	0.73	0.90



Standardized Residuals

ACF of Residuals

p values for Ljung-Box statistic

Figure 5-11. GLS residuals for reverse regression with AR1 in the WIL1-100%. The ACF plot shows a small but statistically significant (α <0.05) negative correlation at lag 2. Models with more sophisticated ARMA structures often failed to converge. While the reverse regression does not meet the independence assumption, the residual correlation is much reduced and is likely to have little effect on the overall model fit and/or our interpretations based on the results.

Table 5-7. Coefficients and goodness of fit statistics for the reverse regression on post-treatment daily discharge.

Site	N* (days)	β_3	β_4	β_5	β_6	β_7	Season	Smear Factor	R^2	NSE
OLYM-100%	2177	1.0	0.9	-5.0E-4	7.9E-5	-1.7E-2	365	1.16	0.68	0.79
OLYM-FP	2299	-3.4	2.1	-4.2E-4	7.0E-5	-	-	1.19	0.73	0.71
OLYM-0%**	2943	-3.4	0.8	-2.6E-4	4.9E-5	-3.8E-2	365	1.22	0.73	0.73
WIL1-100%	3091	2.4	0.38	-3.0E-5	2.1E-5	1.3E-2	365	1.04	0.68	0.85
WIL1-FP	3091	1.1	0.75	-	-	-	-	1.05	0.66	0.82
WIL1-0%	3165	1.5	0.54	1.7E-4	-2.6E-5	1.7E-2	365	1.05	0.63	0.80

Note: *N refers to the number of points used in the regression analysis. For the Olympic basins, N is reduced by the number of days with truncated low flows. We did not use OLYM-REF daily discharge values that were truncated while performing the regression fit, but we did include those data when calculating R^2 and NSE. **The reverse regression to create a synthetic OLYM-0% reference also includes $\log(Q_{OLYM\text{-}FP})$ as a predictor, and the fitted parameter value is 2.2.

The forward and reverse regression approaches each have advantages and limitations, but both provide very similar answers to our primary question about whether discharge changed following harvest and whether the change differed by buffer treatment. Both approaches indicated that discharge did increase following harvest (**Figures 5-12** and **5-13**), with the forward regression showing that the onset of change coincided with the harvest activity as opposed to happening before or after harvest because of some other changing condition. Treatment effects continued through the entire monitoring period consistent with our understanding of hydrologic recovery.

Both methods indicate that the 100% treatments yielded smaller annual changes in water yield than the FP or 0% treatments. In relative terms, discharge increased by 5-7% on average in the 100% treatments while increasing between 26-66% in the FP and 0% treatments. In the Olympic block, which received more precipitation, the FP and 0% basins had absolute increases in runoff that were 1.5-3 times larger than in the WIL1-FP and 0%, with the biggest absolute increases observed in the smallest basin (OLYM-0%; **Table 5-8**).

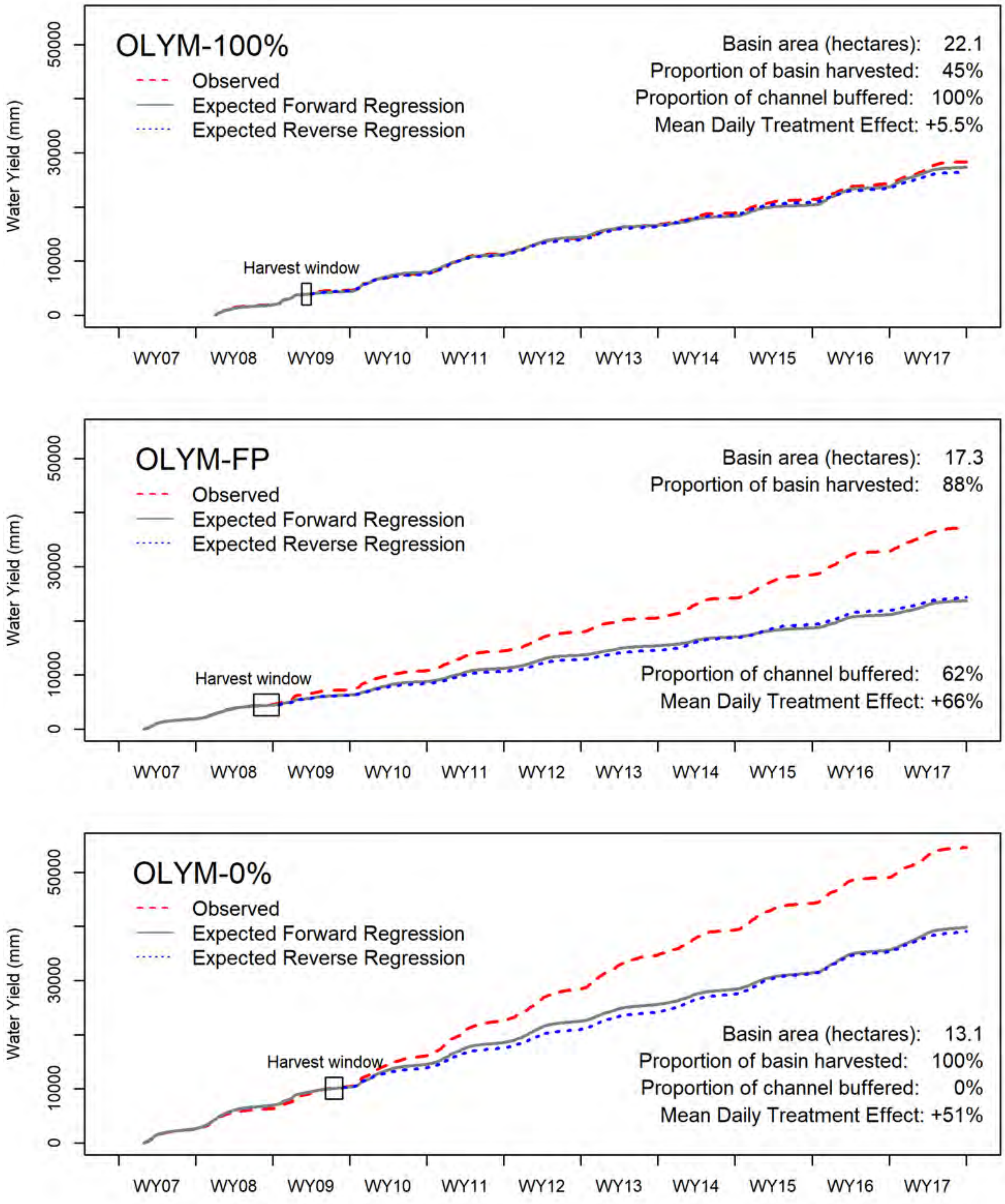


Figure 5-12. Cumulative water yield by forward and reverse regression in the Olympic block. The period of harvest is denoted by the small square box labeled harvest window. Both forward and reverse regression approaches yielded similar annual estimates for expected discharge with treatment effects continuing to accumulate through the post-harvest monitoring period.

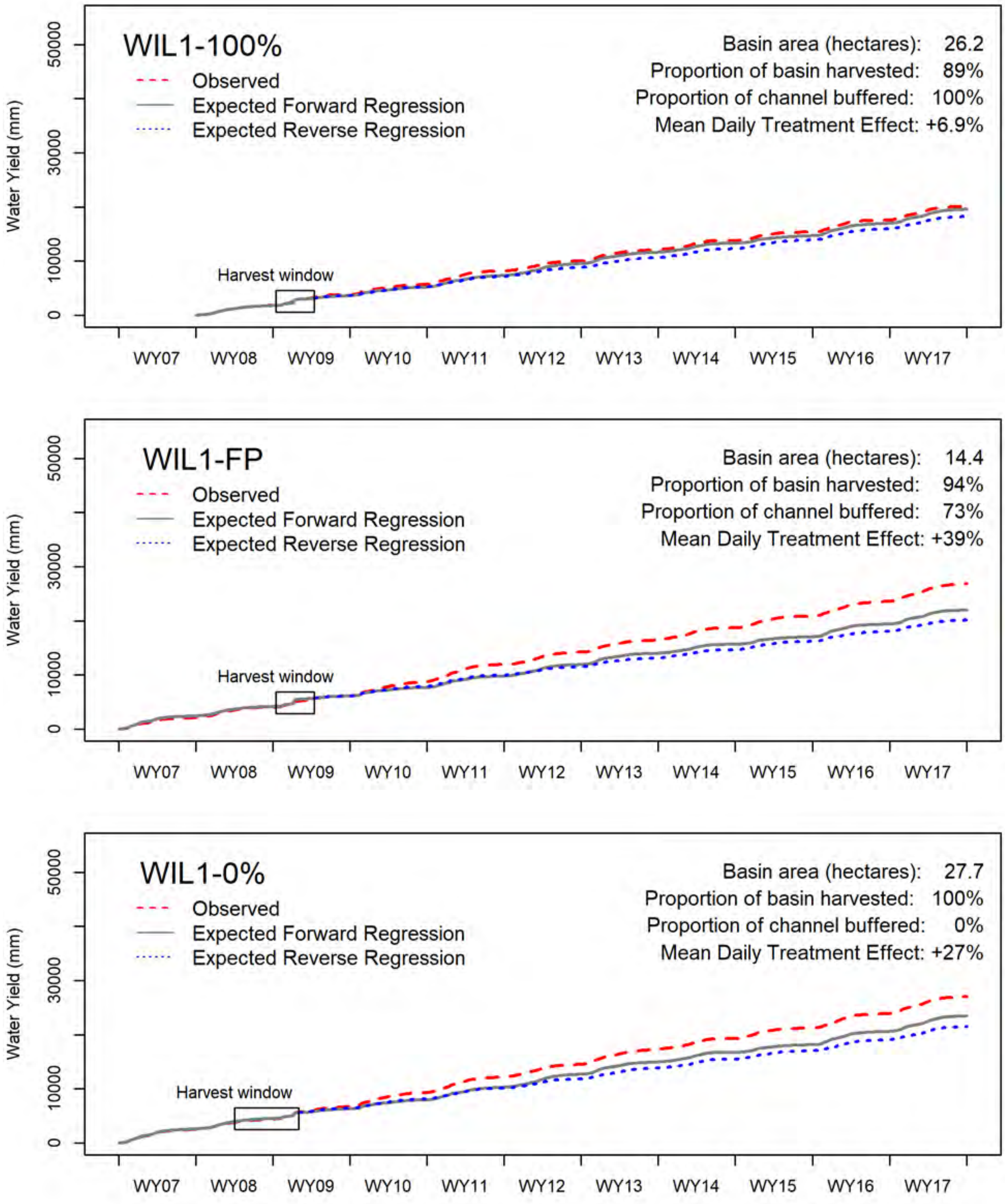


Figure 5-13. Cumulative water yield by forward and reverse regression in the Willapa 1 block. The period of harvest is denoted by the small square box labeled harvest window. Both forward and reverse regression approaches yielded similar annual estimates for expected discharge with treatment effects continuing to accumulate through the post-harvest monitoring period.

Table 5-8. Mean annual change in cumulative daily discharge for the first eight years following harvest.

		Forward Regression		Reverse Regression			
Treatment	**Block**	**Mean Annual Treatment Effect (mm/yr)**	**Mean Annual Change (%)**	**Mean Annual Treatment Effect (mm/yr)**	**Mean Annual Change (%)**	**Basin Area (ha)**	**Proportion harvested (%)**
100%	OLYM	+63	+4.6%	+158	+5.9%	22.1	43
	WIL1	+72	+4.7%	+225	+12%	26.2	89
FP	OLYM	+1480	+75%	+1387	+62%	17.3	88
	WIL1	+594	+32%	+817	+47%	14.4	94
0%	OLYM	+1834	+53%	+1931	+55%	13.1	100
	WIL1	+431	+23%	+650	+36%	27.7	100

One other difference that is obvious from the cumulative response is that the reverse regression technique tended to estimate slightly larger treatment effects in all basins except for the OLYM-FP. The reverse regression approach was expected to provide slightly better estimates for low flows in the OLYM-FP and OLYM-0% basins because the observed OLYM-REF had the baseflows truncated to a single value while the synthetic OLYM-REF for those basins did not. As discussed in **Section 5-6.6.1.**, the reverse regression suffers from regression to the mean (especially in the WIL1 basins) which would tend to inflate treatment effects if the effect was dominated by increases in discharge above the median which is what we observed (see **Section 5-6.6.2.**).

One question that has been raised is whether our estimates of 1480 mm and 1834 mm of increased water yield in the OLYM-FP and OLYM-0% are reasonable. If we look at PRISM estimates of normal precipitation for the Olympic basins, we see that those basins typically receive approximately 3700 millimeters of rainfall per year (PRISM Climate Group 2020). We do not have estimates of evapotranspiration, but if we assume that evapotranspiration might be 50% of precipitation in this wet region (Zhang *et al.* 2004), we could reasonably predict at least 1850 mm/yr of additional runoff following 100% vegetation removal.

The surprising result in these estimates is the relatively subdued response in the WIL1-0% basin. However, the WIL1-0% was one of the largest treatment basins and it received the least precipitation in the post-harvest period.

5-6.4. ANNUAL CHANGES BASED ON MONTHLY DISCHARGE

Although the monthly discharge analysis suffers from a relatively short pre-treatment record and the need to include days with fixed minimum discharges in the regression analysis, the forward regression fits using monthly discharge were good with R^2 and NSE above 0.9 for all basins except for the OLYM-0% (**Table 5-9**). It is not surprising that the OLYM-0% would have the worst fit given the large basin size difference between the OLYM-REF and OLYM-0%. The OLYM-REF is the largest basin in the study and is three times larger than the OLYM-0%, which is the smallest basin.

As might be expected, the average annual change estimated from the monthly discharge analysis was very similar to the magnitudes estimated in the daily analyses (**Table 5-10**). All three sets of analyses (forward daily, reverse daily, forward monthly) provided similar estimates for the magnitude of annual change in the different treatments, with larger effects in the Olympic basins where precipitation was greater and much larger increases in discharge (both relative and absolute) in the FP and 0% treatments (**Table 5-11**).

Table 5-9. Coefficients and goodness of fit statistics for the forward regression on monthly discharge. N is the number of months in the pre-treatment period.

Site	N (mo.)	ARMA	β_0	β_1	β_2	Seasonal	Smearing Factor	r^2	NSE
OLYM-100%	13	2,2	1.22	0.72		12/6	1.022	0.98	0.96
OLYM-FP	18	1,1	3.12	-0.23	0.09	12/6	1.003	0.96	0.96
OLYM-0%	28	1,0	3.09	0.89	0.51	12	1.010	0.85	0.81
WIL1-100%	13	1,0	-9.2	5.2	-0.48	12	1.004	0.96	0.98
WIL1-FP	24	1,1	-5.3	3.0	-0.2	6	1.041	0.91	0.94
WIL1-0%	18	2,2	-4.7	2.6	-0.14	12	1.016	0.94	0.89

Table 5-10. Cumulative treatment effects from forward regression on monthly discharge.

Treatment	Block	Mean Annual Treatment Effect (mm/yr)	Percent change (%)	Basin Area (ha)	Proportion harvested (%)
100%	OLYM	+66	+2.3%	22.1	43
	WIL1	+133	+7.2%	26.2	89
FP	OLYM	+1425	+64%	17.3	88
	WIL1	+586	+31%	14.4	94
0%	OLYM	+1957	+55%	13.1	100
	WIL1	+411	+20%	27.7	100

Table 5-11. Comparison of cumulative annual treatment effect derived from daily forward regression, daily reverse regression, and monthly forward regression organized by method.

Treatment	Block	Mean Daily Forward (mm/yr)	Mean Daily Reverse (mm/yr)	Mean Monthly Forward (mm/yr)	Mean Daily Forward (%)	Mean Daily Reverse (%)	Mean Monthly Forward (%)
100%	OLYM	+63	+158	+66	+4.6%	+5.9%	+2.3%
	WIL1	+72	+225	+133	+4.7%	+12%	+7.2%
FP	OLYM	+1480	+1387	+1425	+75%	+62%	+64%
	WIL1	+594	+817	+586	+32%	+47%	+31%
0%	OLYM	+1834	+1931	+1957	+53%	+55%	+55%
	WIL1	+431	+650	+411	+23%	+36%	+20%

5-6.5. SEASONAL CHANGES IN DISCHARGE

We used the results from the reverse regression on daily discharge to examine the seasonal changes in discharge. Seasonal patterns were identifiable using all three approaches (forward daily, reverse daily, forward monthly) but we relied upon the reverse daily regression approach because it minimizes variability by forcing the post-treatment reference basin discharges to follow the flow characteristics of the treatment basins (Safeeq *et al.* 2020). The reverse regression approach also exhibited fewer outliers than the forward analysis of monthly discharge, which was based on a relatively sparse and coarse dataset.

The change in discharge that we observed after harvest was conditioned on many factors including climate, weather, buffer treatment, and the physical hydrology of the watershed. When relative change in discharge (ΔQ%) is plotted against independent estimates of monthly precipitation obtained from gridded PRISM model output (PRISM Climate Group 2020), we see all four factors in play. In all the basins, we saw that the relative change in discharge varies with precipitation (**Figures 5-14**, **5-15**, and **5-16**). The largest absolute and relative changes occurred in the Olympic block FP and 0% basins, which are adjacent to each other and have a wetter climate (and most likely greater ET) than the other basins. In all the analyses, including this one, we also observed differences by treatment, especially the 100% treatments where we saw decreased summer discharge (e.g., ΔQ% <0) during periods of little rainfall. Finally, we observed lagged responses to changes in precipitation in basins like the OLYM-100%, WIL1-FP, and OLYM-0% that may be related to the physical hydrology of those watersheds.

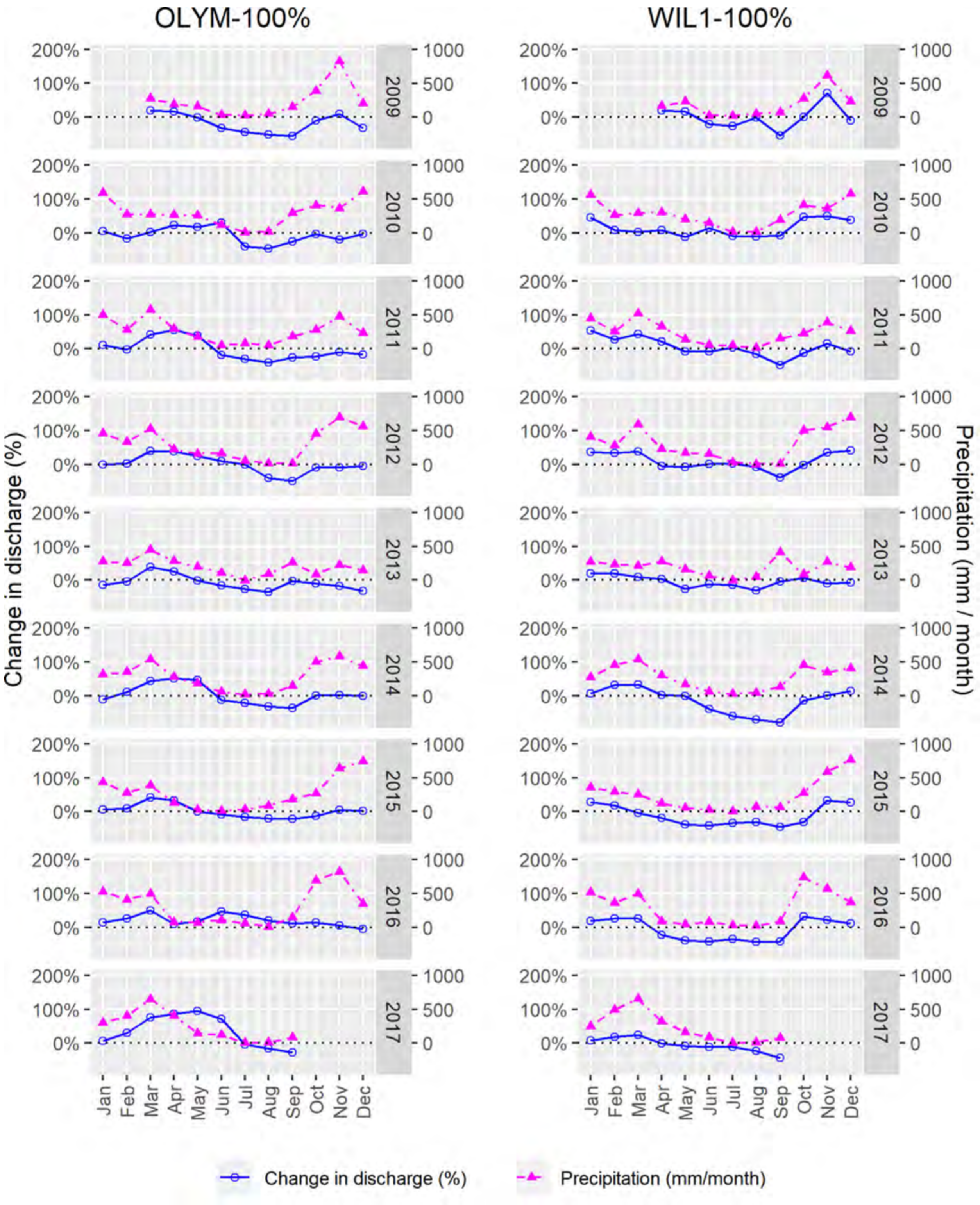


Figure 5-14. Monthly relative change in discharge (ΔQ%) following harvest and monthly PRISM precipitation estimates in the OLYM-100% and WIL1-100% treatment basins. The y-axes are plotted using a fixed ratio so relative effects as a function of precipitation can be compared across basins.

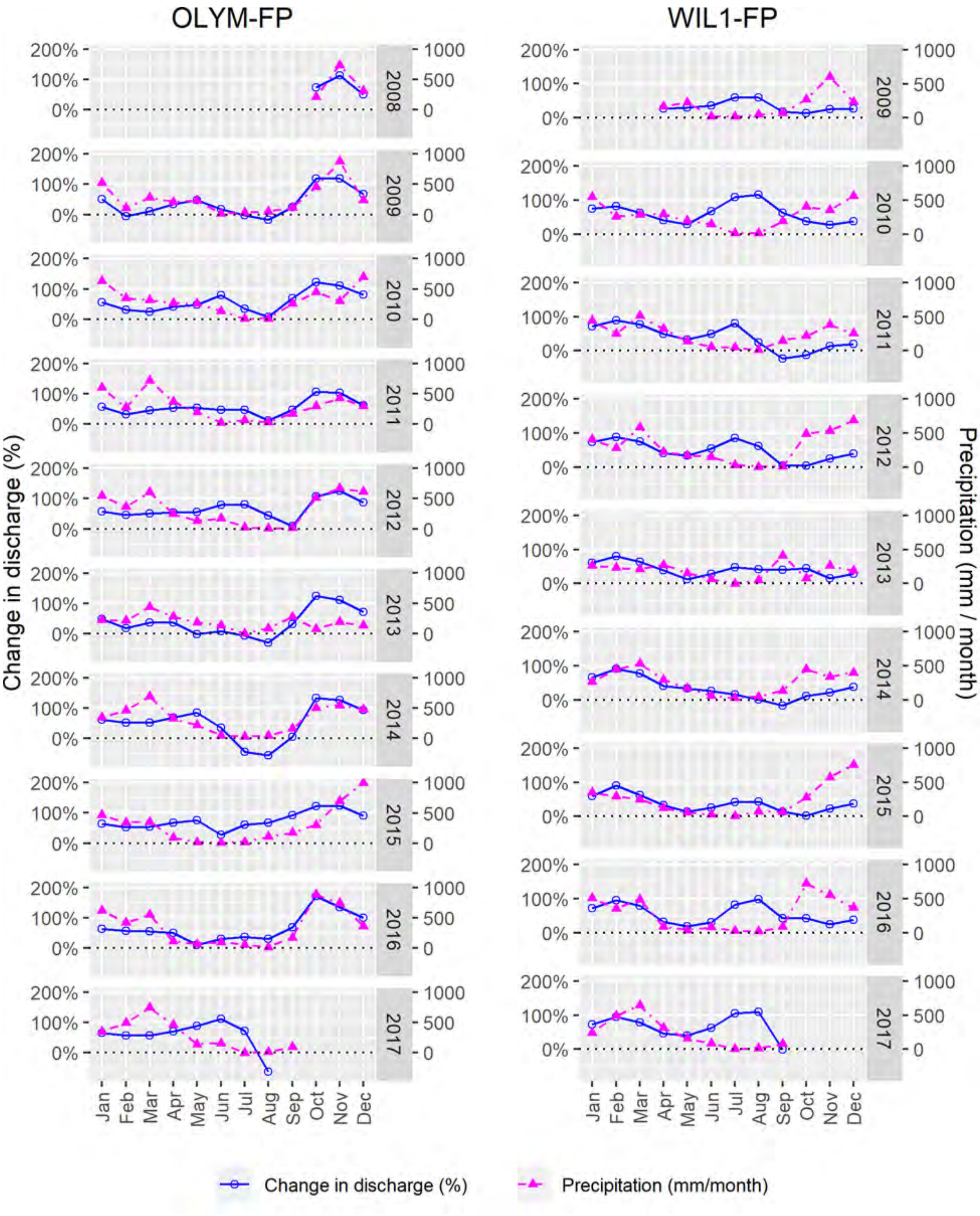


Figure 5-15. Monthly relative change in discharge (ΔQ%) following harvest and monthly PRISM precipitation estimates in the OLYM-FP and WIL1-FP treatment basins. The y-axes are plotted using a fixed ratio so relative effects as a function of precipitation can be compared across basins.

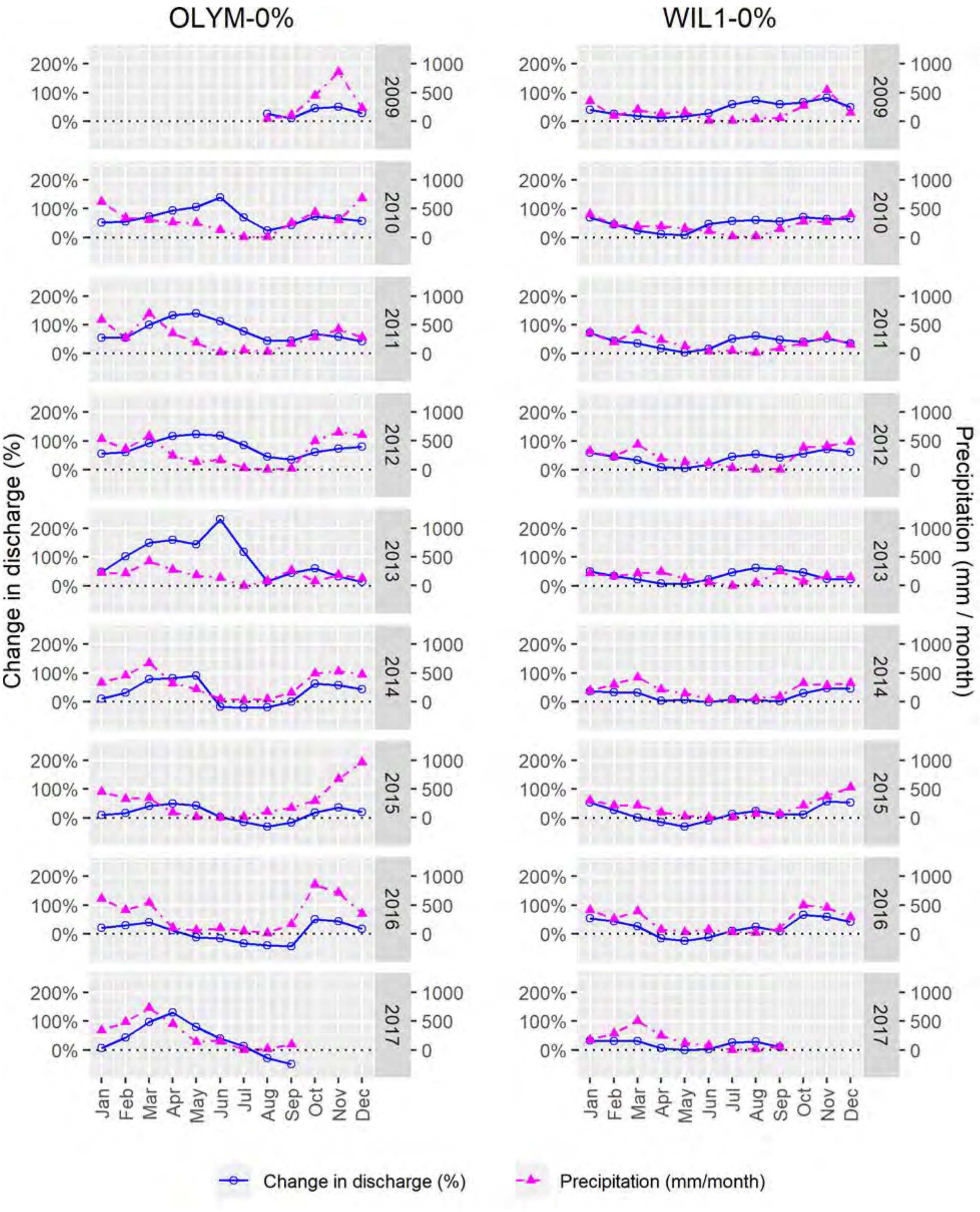


Figure 5-16. Monthly relative change in discharge (ΔQ%) following harvest and monthly PRISM precipitation estimates in the OLYM-0% and WIL1-0% treatment basins. The y-axes are plotted using a fixed ratio so relative effects as a function of precipitation can be compared across basins.

In the OLYM-100% basin, average post-treatment discharge increased from January through June with the largest changes occurring in March and April (+4.6 mm/day (+42%) and +2.5 mm/day (+38%), respectively). While spring discharge increased, summer discharges decreased with the largest decreases occurring in July and August (-0.56 mm/day (-29%) and -0.72 mm/day (-26%)). Relative changes in discharge appeared to correlate with precipitation, but the relationship is complex with treatment effects often lagging precipitation (left panel, **Figure 5-14**).

In the WIL1-100% basin, changes in average post-treatment discharge were slightly different. The maximum changes were observed in November through March with the largest increases occurring in November and January (+2 mm/day (+27%) and +2.4 mm/day (+27%)) and the largest relative decreases occurring in August and September (-3.6 mm/day (-26%) and -6 mm/day (-40%)). The WIL1-100% treatment exhibited less of a lagged relative response to precipitation despite being one of the larger basins in the study (right panel, **Figure 5-14**). Both 100% treatment basins exhibited more variability in the relationship between absolute change in runoff (mm/day) and precipitation than the FP or 0% treatments, possibly reflecting the greater role of vegetation in hydrologic partitioning in those basins (**Figures 5-17**, **5-18**, and **5-19**).

Discharge in the FP basins generally increased following harvest with the wetter OLYM-FP basin exhibiting larger absolute gains (left panel, **Figure 5-18**). In the OLYM-FP basin, increases were greatest in the October through March period with the largest increases in November and January (+9.5 mm/day (120%) and +7 mm/day (+58%)). The only month that exhibited an average decrease was August (-0.2 mm/day, -0.6%) and monthly discharge only decreased sporadically (left panel, **Figure 5-15**).

In the WIL1-FP, discharge increased in all months with the November through March period exhibiting large absolute gains. On average, the January through March increases were +4.6 mm/day (+69%), +4.6 mm/day (+89%), and +4.9 mm/day (+72%), respectively. The most obvious difference in the WIL1-FP basin response to harvest were the large relative increases in discharge during July and August (+1.1 mm/day (+70%) and +0.8 mm/day (+62%)) (right panel, **Figure 5-15**). The dryer WIL1-FP treatment exhibited more variability in runoff response to precipitation changes than the wetter OLYM-FP, but changes in absolute runoff generally followed precipitation (right panel, **Figure 5-18**).

The 0% buffer basins were included as endmembers and were expected to exhibit larger changes in discharge than the FP or 100% treatments. The OLYM-0% basin, which was the smallest basin, did exhibit the largest absolute increase in specific discharge (mm/day) as a function of precipitation (left panel, **Figure 5-19**), but exhibited smaller relative changes than the slightly larger OLYM-FP basin (**Table 5-11**). As with the OLYM-FP, the largest absolute changes occurred from October to March on average, with November and March exhibiting the largest average absolute increases (+8.9 mm/day (+52%) and +12 mm/day (+83%)). While absolute treatment effects generally tracked precipitation, relative treatment effects appeared to exhibit a lagged response with peak relative effects occurring from April through June (+8.6 mm/day (+98%), +5 mm/day (+89%), +3.3 mm/day (+76%)) (left panel, **Figure 5-16**).

The WIL1-0%, which was the largest treatment basin at the lowest elevation and with the least precipitation (~2260 mm/yr), exhibited the smallest absolute and relative response among the four 0% and FP basins (**Table 5-11**). As with the WIL1-FP basin, absolute responses were greatest in November through March, but with November through January exhibiting the largest absolute increases (+4.6 mm/day (+58%), +3.6 mm/day (+47%), +4.2 mm/day (+53%)). July and August also increased (+0.6 mm/day (+35%) and +0.6 mm/day (+44%)) on average, but summer absolute and relative increases were smaller than observed in the slightly wetter (~2900 mm/yr) WIL-FP basin (right panel, **Figure 5-16**). Absolute changes in discharge generally tracked precipitation (right panel, **Figure 5-19**).

5-6.6. CHANGES IN FLOW MAGNITUDE AND FREQUENCY

Flow duration curves have long been used to examine the effects of harvest on streamflow characteristics because changes in magnitude also affect the frequency that a given discharge is exceeded. Plotting discharge by flow duration gives us the ability to examine both changes at the same time.

5-6.6.1. Reverse Regression – The Issue of Regression toward the Mean

One problem with the reverse regression approach becomes clear when plotting flow-duration curves for expected discharge using both the forward and reverse regression approaches. The use of regression to create the synthetic reference basin discharge in the reverse regression approach reduces the variance so that predicted discharges fall closer to the mean. When expected discharge from the forward and reverse regression approaches are plotted together, we see that the reverse regression systematically underestimates the expected peak flows in four of the six basins (**Figure 5-20**). At the same time, the reverse regression approach overestimates expected base flows. If left unchecked, the reverse regression approach would overestimate treatment effects associated with peak flows and underestimate treatment effects for base flows.

The Willapa basins exhibited a larger bias than the Olympic basins. While it is not clear why, one reason might be the better reverse regression fits to WIL1-REF.[11] Only two basins, the OLYM-FP and OLYM-0%, did not exhibit bias. Those were the two basins where we expected the reverse regression approach to offer advantages over forward regression because the synthetic reference created through reverse regression did not suffer from the truncated baseflows present in the actual OLYM-REF. The fact that the forward and reverse regression approaches produced similar predictions in those basins improved our confidence in their results.

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[11] As reflected by NSE.

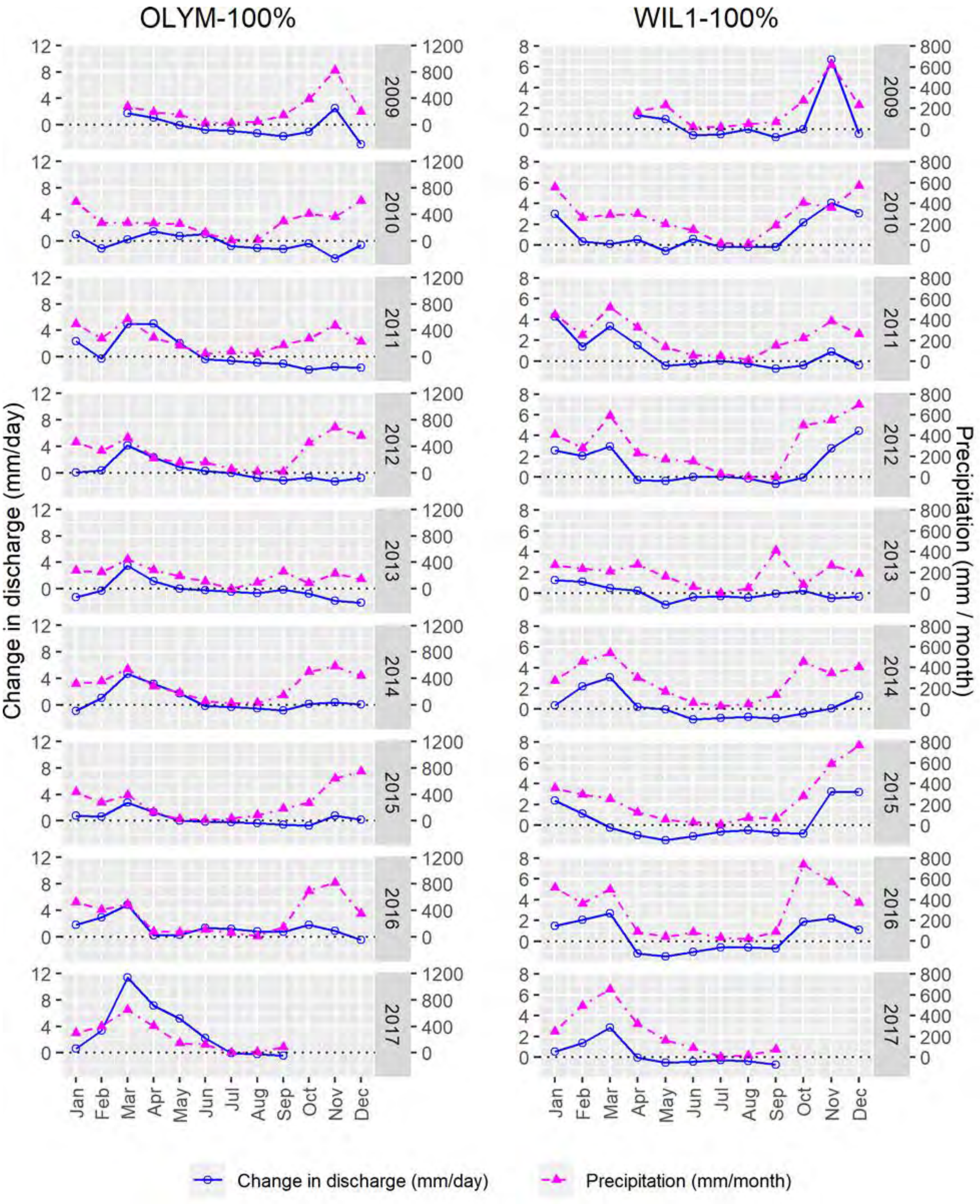

Figure 5-17. Monthly absolute change in discharge (mm/day) following harvest and monthly PRISM precipitation estimates in the OLYM-100% and WIL1-100% treatment basins. Precipitation and runoff are plotted in a fixed ratio, but y-limits vary by basin reflecting differences in climate.

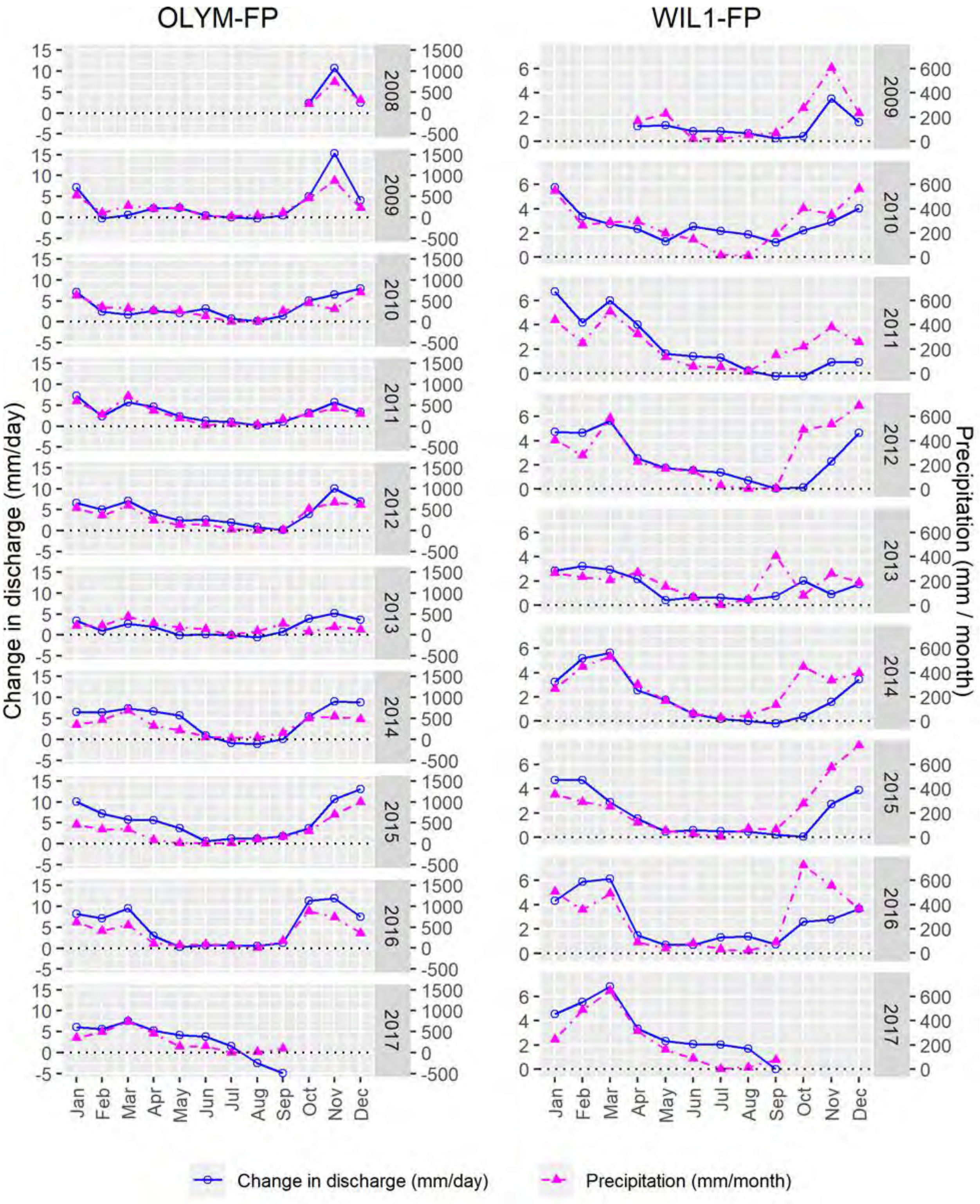

Figure 5-18. Monthly absolute change in discharge (mm/day) following harvest and monthly PRISM precipitation estimates in the OLYM-FP and WIL1-FP treatment basins. Precipitation and runoff are plotted in a fixed ratio, but y-limits vary by basin reflecting differences in climate.

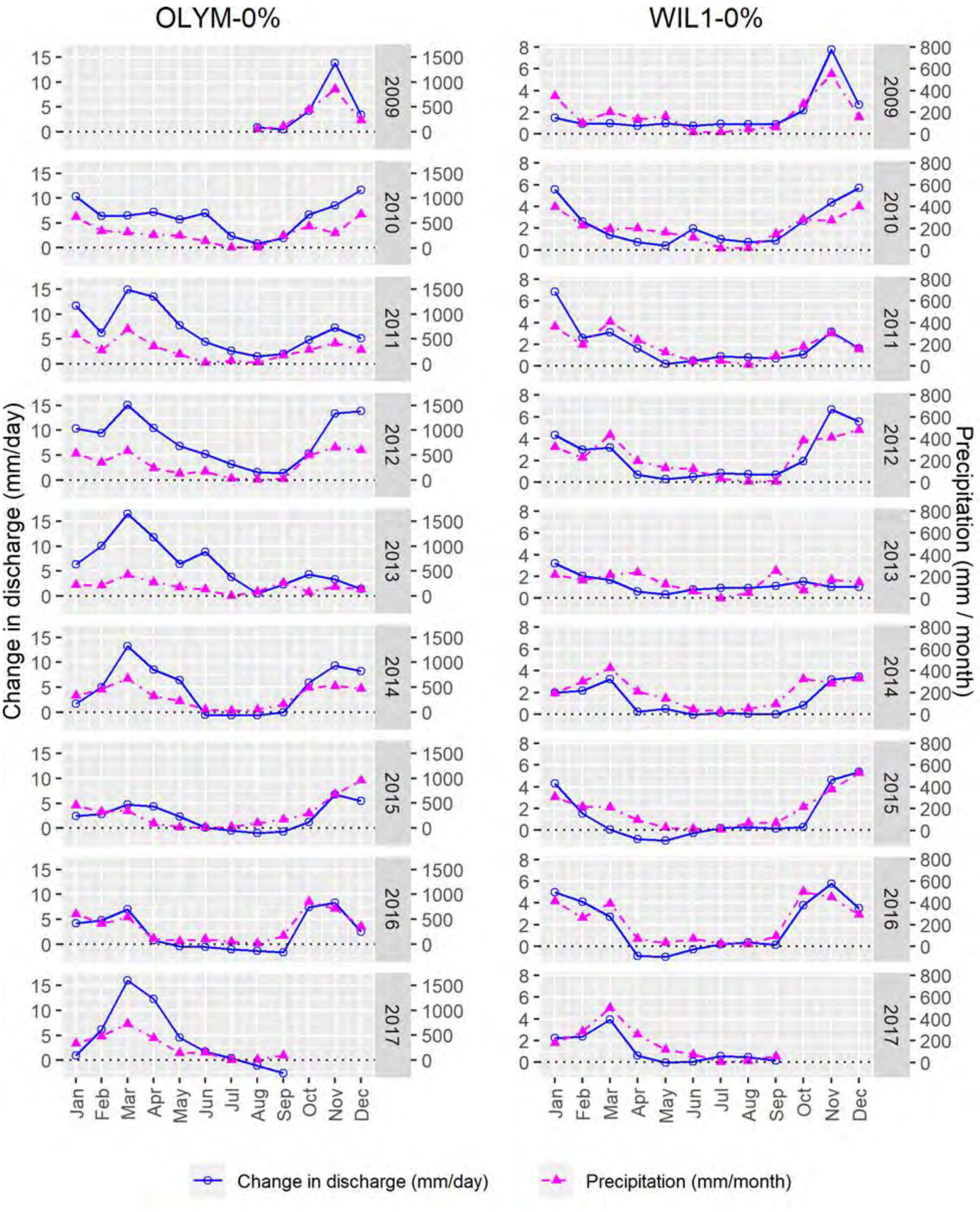

Figure 5-19. Monthly absolute change in discharge (mm/day) following harvest and monthly PRISM precipitation estimates in the OLYM-0% and WIL1-0% treatment basins. Precipitation and runoff are plotted in a fixed ratio, but y-limits vary by basin reflecting differences in climate.

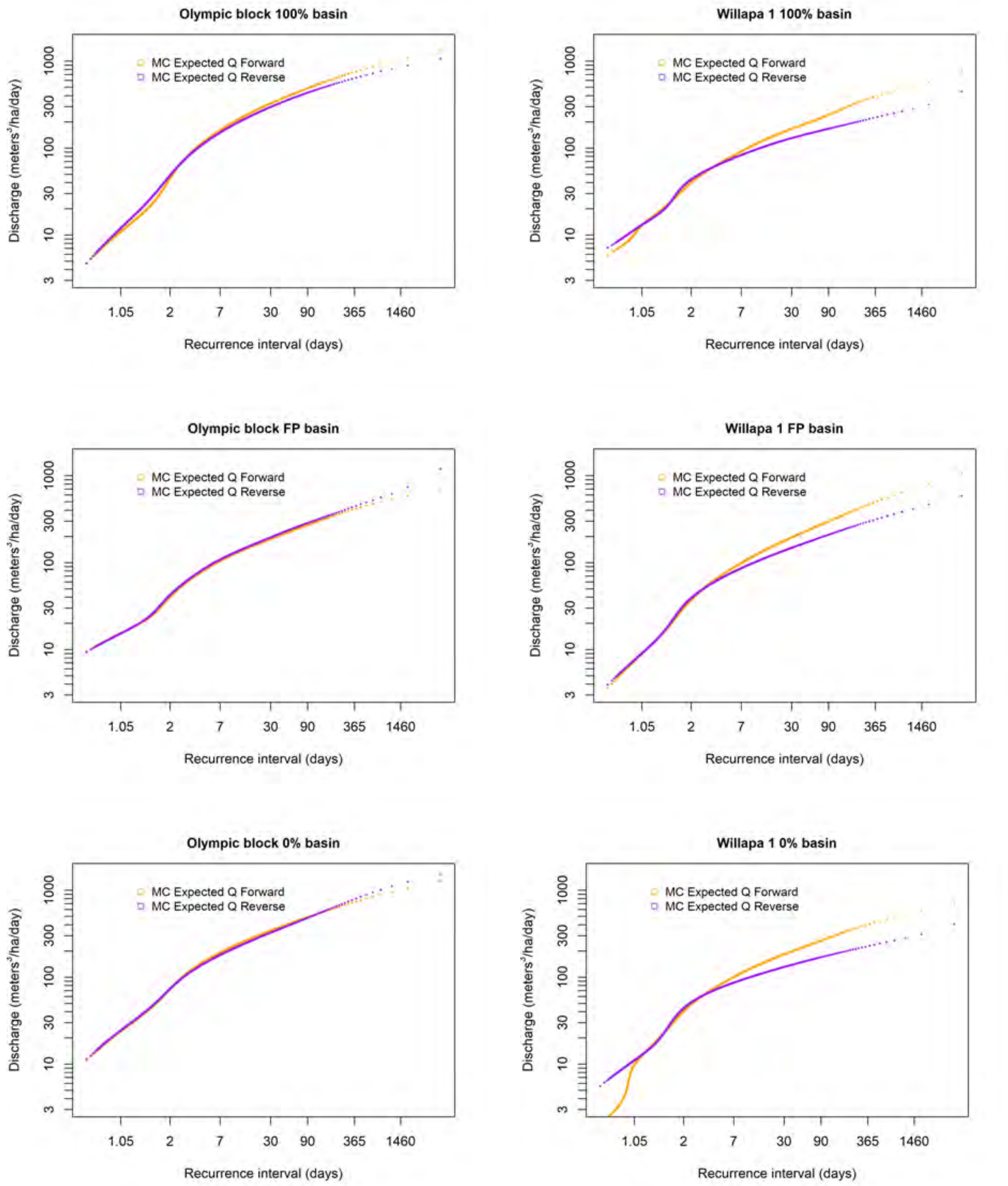

Figure 5-20. Comparison of forward and reverse regression. The additional regression involved in the reverse approach appears to reduce the variance (see regression toward the mean). In this figure, the two-day flow is the median and the x-axis is adjusted to highlight the reverse bias in flows greater than the median.

5-6.6.2. Change in Discharge Magnitude and Frequency

We plotted flow duration curves based on the forward regression approach with daily discharge to evaluate changes in daily discharge over the first eight years following harvest. The partial post-treatment year nine was eliminated to avoid introducing bias.[12] The expected discharges are corrected for the loss of variance in the forward regression to prevent overestimating treatment effects in the tails of the distributions.

The flow duration curves show that in the 100% treatments, there was a reduction in flows less than the median (2-day) recurrence interval (RI) (**Figure 5-21**). The measurements in the OLYM-100% are truncated for low flows which makes it difficult to quantify low-flow changes, but baseflows (defined here as <2-day RI) in the WIL1-100% basin decreased by an average of -0.48 mm/day (-26%), with a maximum decrease of -0.68 mm/day (-80%) at the lowest flows in the post-treatment period. Reductions in baseflow in those basins were offset by increases in stormflows (defined here as ~7-30 day RI) resulting in slight increases in post-harvest discharge. Stormflow increased by an average of +1.4 mm/day (+12%). Stormflow also increased slightly in the OLYM-100% basin, but the magnitude of change was within 95% confidence intervals around expected discharge.

Response in the FP and 0% buffer treatments varied more by block than by treatment. In the OLYM-FP, just under 10% of the days exhibited decreased discharge following harvest, while the remaining days all had increased discharge. The median (2-day RI) increase was +2.6 mm/day (+63%) and the effect size increased with increasing runoff magnitude from there (**Figure 5-22**). In the WIL1-FP, all flows up to the 30-day RI had higher runoff. The median increase was +2.2 mm/day (+58%), but the effect size decreased with increasing runoff magnitude above the median (**Figure 5-21**).

The OLYM-0% basin displayed a similar pattern to the OLYM-FP basin with a positive relationship between runoff and change, except that discharge increased over almost the entire post-treatment period (**Figure 5-22**). The median increase in the OLYM-0% was +4.6 mm/day (+62%). The WIL1-0% basin displayed a similar pattern to the WIL1-FP and had a median increase of +1.3 mm/day (+32%). Peak flows (RI >1.5 years) appeared to have increased in the Olympic block basins but not the Willapa 1 block basins, but all changes in peak flow were within the 95% confidence intervals (**Figure 5-22**). The most consistent changes in the four FP and 0% treatment basins were increases in discharge for events with recurrence intervals between 1.5 and 7 days.

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[12] Harvest could not be synchronized so the "post-harvest" period started on different days in different sites. In most analyses, we use the full post-treatment period through the end of WY2017 but not with flow-duration. Flow durations based on partial years would be biased so we used the eight years following harvest starting from the day that harvest was completed in the site.

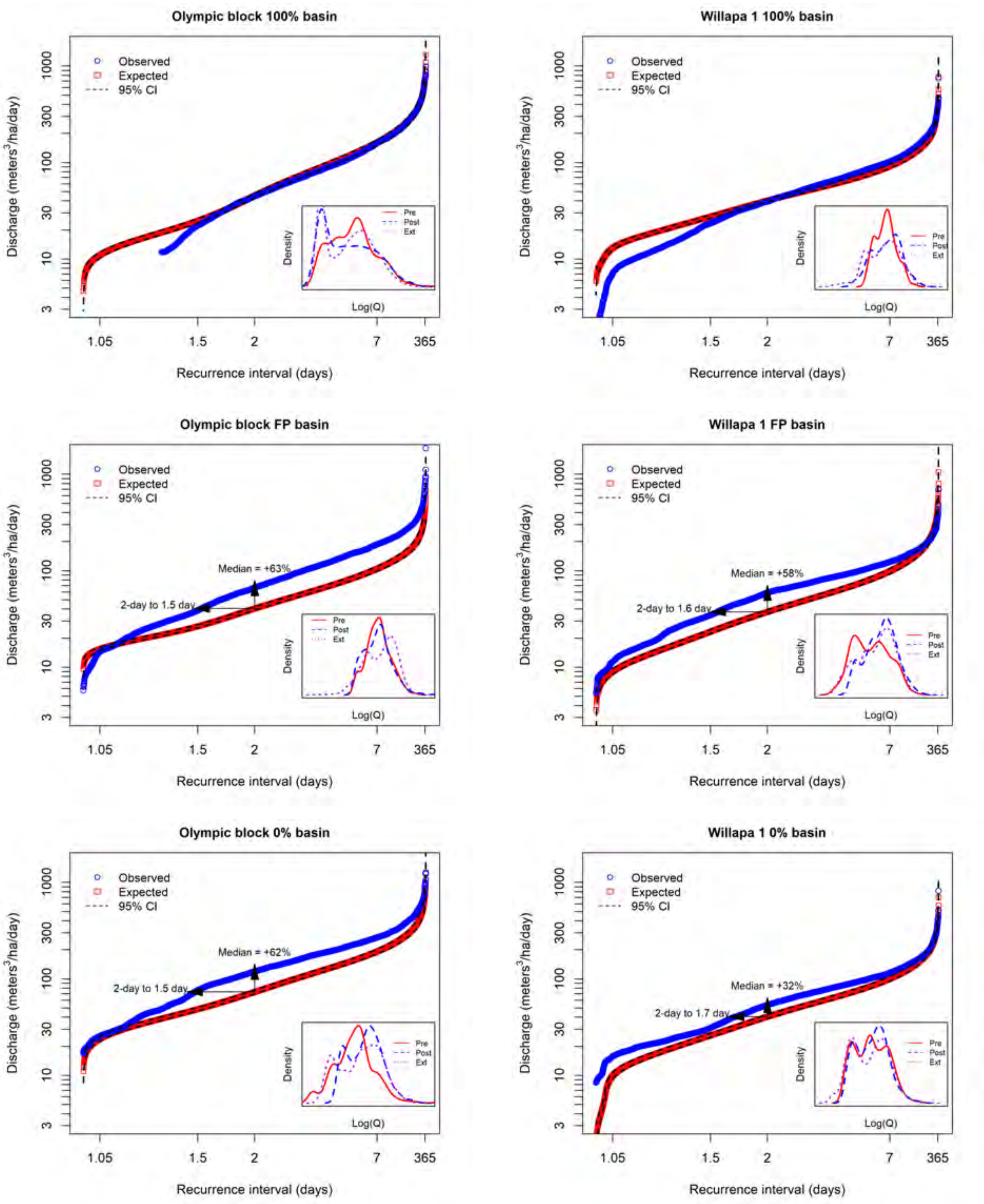

Figure 5-21. Flow duration curves with arrows showing changes in the median (2-day RI) flow magnitude and frequency. The log y-axis preserves ratios and illustrates where relative change (ΔQ%) in magnitude is greatest. The inset shows the density of raw $\log_{10}$ discharge in the first 2-year post-treatment period (Post) and the 6-year extended period (Ext) and contains both weather and treatment effects.

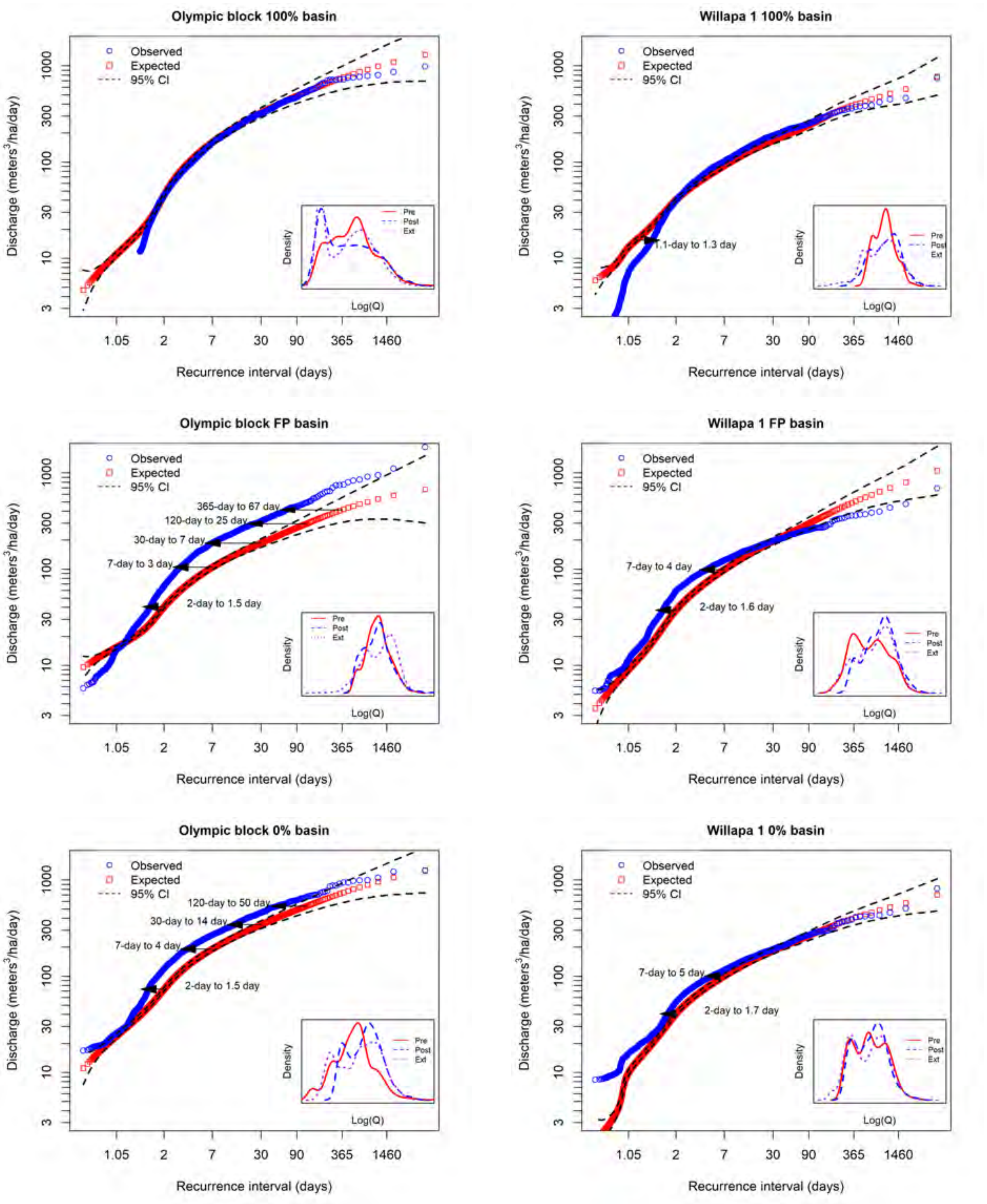

Figure 5-22. Change in discharge Recurrence Interval (RI or flow exceedance) for the 2, 7, 30, 120, and 365-day RI events in the first eight water years following harvest with the x-axis scaled to highlight changes above the median (2-day RI). Only those changes outside of the 95% CI are plotted with arrows. The log y-axis preserves ratios and therefore illustrates relative (%) change.

In the 100% basins, where the primary effect was a decrease in base flows, the flow magnitude in the WIL1-100% with a 1.1 day recurrence interval shifted to a recurrence interval of 1.3 days (**Figure 5-22**).[13] Thus, for the WIL1-100% basin, a discharge of 15 m^3/ha/day, which would have been exceeded 91% of the time prior to harvest was only exceeded 77% of the time following harvest. In the FP and 0% treatment basins, discharge magnitude generally increased and so the shifts resulted in reduced recurrence intervals for a given magnitude of discharge. In the OLYM-FP, the median (2-day RI) pre-harvest discharge was 34 m^3/ha/day and would only have been exceeded 50% of the time prior to harvest but was exceeded 65% of the time following harvest. In the FP and 0% treatment basins, all but the top 7% of flows (and lowest 15% of flows for the Olympic basins) exhibited shifts in flow frequency following harvest that fell outside of the 95% CI for expected discharge.

5-6.7. TURBIDITY AND SUSPENDED SEDIMENT EXPORT

Over 11 years of the study, 10-minute turbidity readings were less than 3 NTU for most of the time (>95%) and only one basin had median (50th percentile) turbidity greater than 0.2 (**Table 5-12**). Both turbidity and SSC increased with increasing discharge during storm events but then rapidly declined with all basins intermittently exhibiting suspended sediment hysteresis loops, with greater turbidity/SSC on the rising limb of the hydrograph compared with the descending limb for a given discharge.

Graphs of cumulative suspended sediment export (SSE) and discharge show that the suspended sediment budgets were dominated by a relatively small number of sediment exporting events in both the reference and treatment basins (**Figures 5-23** and **5-24**). The figures show rates of SSE and discharge accumulation before and after harvest, and while discharge accumulates steadily though time, SSE is episodic and not synchronized across all basins or strongly correlated with discharge (**Table 5-13**). The lack of SSE in some high discharge events, and poor correlations between SSC and Q, suggests that the basins are likely to be supply limited.

Table 5-12. Median, 90th, 95th, and 99th percentiles and maximum recorded 10-minute turbidity by basin.

Block	Treatment	50%	90%	95%	99.5%	Maximum
		Turbidity (NTU)				
OLYM	REF	0.1	1.7	3.0	226	2,193
	100%	0.8	3.9	6.7	55	2,056
	FP	0.0	0.6	1.1	12	2,075
	0%	0.2	1.4	2.4	22	1,986
WIL1	REF	0.0	2.1	3.2	16	2,001
	100%	0.0	0.9	2.0	16	636
	FP	0.1	0.8	1.4	17	1,132
	0%	0.2	2.5	4.1	29	2,438

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[13] The relationship between recurrence interval and probability of exceedance is p=1/RI, so a 1.1 day RI has an exceedance probability 0.91, meaning that flows exceed that value 91% of the time.

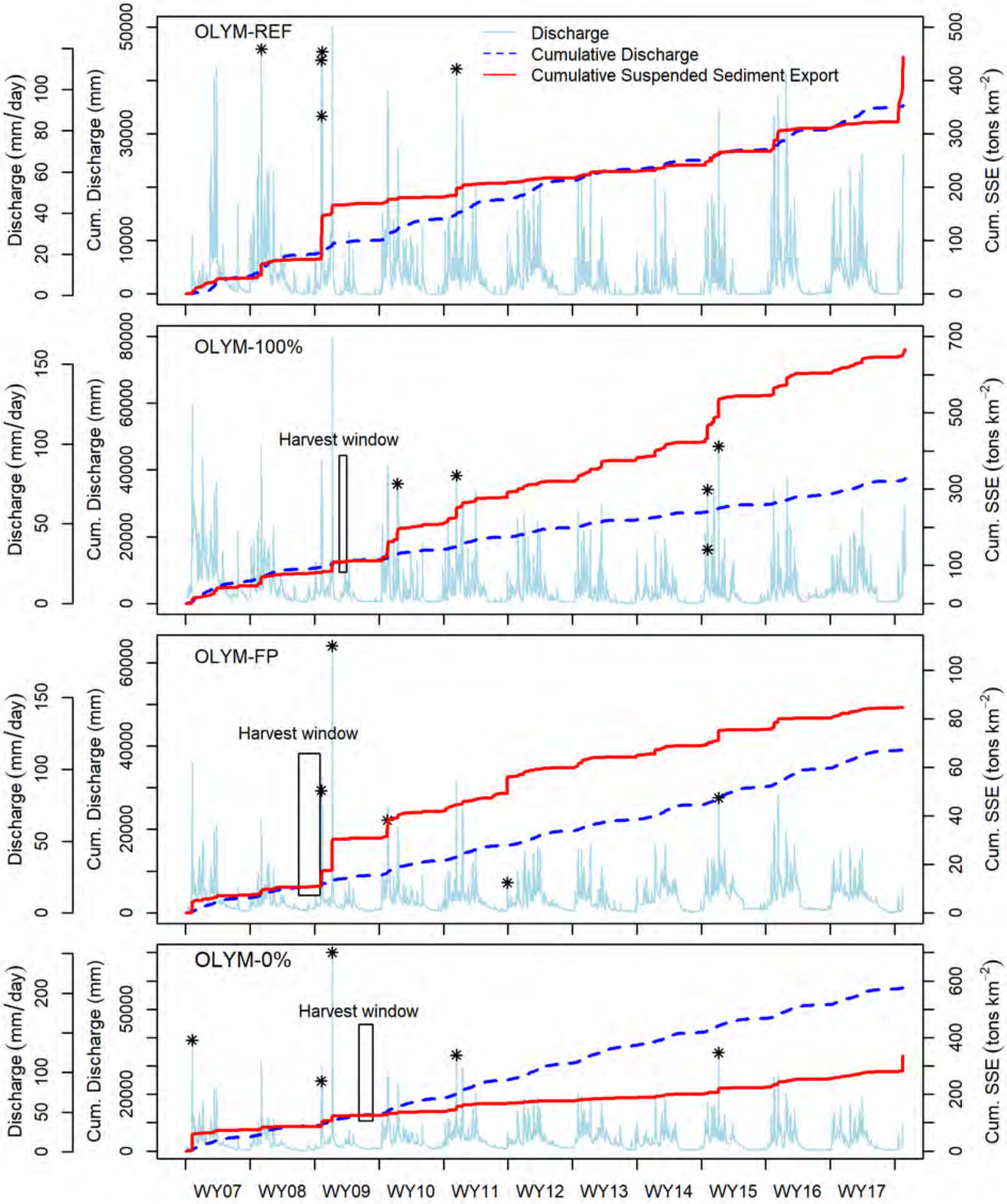

Figure 5-23. Cumulative discharge and suspended sediment export in the Olympic block. The harvest window is denoted with a box and the five highest days of suspended sediment export are denoted with asterisks on the flow magnitude for that day.

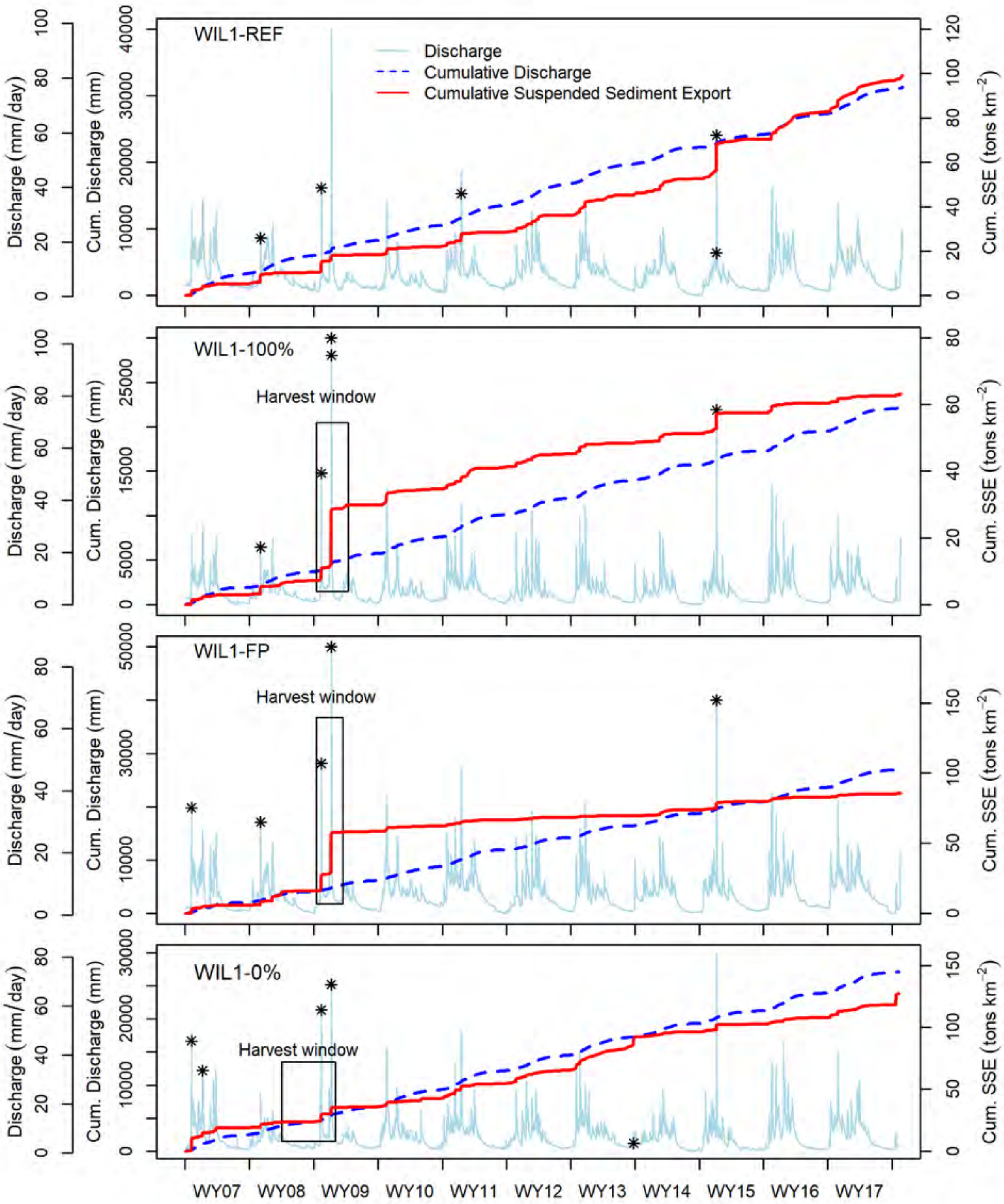

Figure 5-24. Cumulative discharge and suspended sediment export in the Willapa 1 block. The harvest window is denoted with a box and the five highest days of suspended sediment export are denoted with asterisks on the flow magnitude for that day.

Table 5-13. Top five sediment producing days in each basin with suspended sediment export (SSE) and discharge (Q).

OLYM -REF	**Date**	**SSE (tons /km^2/ day)**	**Q (mm/day)**	**WIL1-REF**	**Date**	**SSE (tons /km^2/ day)**	**Q (mm/day)**
1	11/12/2008	30	118	1	1/5/2015	10	59
2	11/7/2008	27	114	2	11/12/2008	4	40
3	12/3/2007	19	120	3	12/3/2007	3	21
4	11/8/2008	14	87	4	1/4/2015	2	16
5	12/12/2010	11	110	5	1/16/2011	2	38
OLYM -100%	**Date**	**SSE (tons /km^2/ day)**	**Q (mm/day)**	**WIL1-100%**	**Date**	**SSE (tons /km^2/ day)**	**Q (mm/day)**
1	1/5/2015	37	98	1	1/7/2009	15	102
2	12/12/2010	23	80	2	1/5/2015	4	75
3	11/4/2014	20	71	3	11/12/2008	3	50
4	1/11/2010	15	75	4	12/3/2007	2	22
5	11/3/2014	15	34	5	1/8/2009	2	96
OLYM -FP	**Date**	**SSE (tons /km^2/ day))**	**Q (mm/day)**	**WIL1-FP**	**Date**	**SSE (tons /km^2/ day)**	**Q (mm/day)**
1	1/7/2009	11	186	1	1/7/2009	25	86
2	9/26/2011	7	21	2	11/12/2008	8	49
3	11/7/2008	4	85	3	1/5/2015	3	69
4	1/5/2015	3	80	4	11/6/2006	2	35
5	11/16/2009	3	65	5	12/3/2007	2	30
OLYM -0%	**Date**	**SSE (tons /km^2/ day)**	**Q (mm/day)**	**WIL1-0%**	**Date**	**SSE (tons /km^2/ day)**	**Q (mm/day)**
1	11/6/2006	51	141	1	11/6/2006	8	46
2	1/7/2009	12	251	2	1/7/2009	4	69
3	1/5/2015	11	125	3	11/12/2008	4	58
4	11/7/2008	10	89	4	9/22/2013	3	4
5	12/12/2010	8	122	5	1/7/2007	2	34

In three of the basins, OLYM-0%, WIL1-FP, and WIL1-0%, the five largest sediment producing events included the storm on 6 November 2006. In the OLYM-0%, this storm produced an estimated 51 metric tons/km^2, which was 18% of the total sediment budget for that basin over the entire period of record. The November 2006 storm also exported a large proportion of the suspended sediment budget of the WIL1-0% treatment. Because so much sediment was exported in that single pre-treatment storm, SSE appeared to decrease in the post-treatment period relative to the pre-treatment period of both 0% buffer basins (**Figure 5-24**).

In all but the OLYM-100%, a large proportion of SSE was exported during two storm events centered on 9 November 2008 and 7 January 2009 (**Figures 5-23** and **5-24**). Because those storms occurred during the harvest period of WIL1-100% and WIL1-FP and post-harvest period in OLYM-FP, SSE appeared to increase in the harvest/post-treatment period relative to what was observed in the pre-treatment period. Those storms were in the pre-treatment period of the OLYM-0%, which contributed to a lack of a treatment effect there, and in the harvest period of the WIL1-0%. Those two storms also produced 26% of the sediment exported in the OLYM-REF, which received no treatment at all.

Over the course of the study, we observed greater SSE in the post-treatment period relative to the pre-treatment period in four of our treatment basins (both 100% and FP treatments), and one of our two references. We have no physical explanation as to why the 100% and FP treatments would exhibit increased sediment export while basins without RMZ buffers would yield decreased SSE. We could perform mixed model statistical analysis to evaluate the probability of this outcome, but just using binomial probability theory and asking what the odds of observing four (or more) increases out of six treatments with an underlying probability of observing an increase being 0.5 (based on the references), we would estimate p=0.344 and would not reject the null hypothesis. Given the lack of any coherent pattern and process, we conclude that observed SSE was probably driven by a random stochastic process like small-scale mass wasting that was not observed by field crews.

Another method for looking at treatment effects is to examine the relationship between suspended sediment yield (Qs) and discharge (Q) following harvest (Bywater-Reyes *et al.* 2017). When we plot drainage area-normalized Qs and Q in log-log space, we can interpret the fitted intercept as an indicator of erosion severity and the slope as an indicator of erosive power (e.g., the ability to tap into new sources of sediment as discharge increases). Timber harvest can affect either parameter by changing discharge and/or sediment supply (Bywater-Reyes *et al.* 2017).

In a naive analysis focused on the treatment basins only, we observed differences in erosive power between watersheds (i.e., different slopes) and increases in erosive severity (i.e., intercepts) after harvest (**Figure 5-25**). The reference basins do not have treatment periods, but if we artificially apply the treatment basin harvest periods to the references, we see that erosion severity also appears to increase in the reference basins in the fake ‘post-harvest’ period, and in some cases by amounts larger than the increases observed in the treatments (**Figure 5-26**).[14] We could analyze these relationships in a statistical analysis to see how the treatments changed relative to the references, but by looking closely at the graphics we would likely determine that SSE decreased relative to the references in some sites (e.g., OLYM-100% and WIL1-100%).

Given that result, we might try to explain that the decrease resulted from dilution (i.e., increased discharge without a corresponding increase in sediment supply). In fact, dilution would be our null hypothesis in a naive analysis that tried to use annual increases in water yield as the change in discharge rather than the subset of flows that typically transport sediment (which only changed significantly in the OLYM-FP). We did not attempt that analysis, however, because it would be based on assumptions that we find unreasonable, including: 1) the assumption that SSE or SSY

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[14] You cannot directly compare intercepts without holding the slope constant. The observation of a larger change in the references as a function of ‘harvest timing’ is a qualitative assessment based on observation alone.

in the reference and treatments will track each other in the absence of a treatment effect, and 2) that it would be appropriate to fit a common slope between reference and treatment basins to isolate differences in erosive severity (e.g., intercepts). We can see from **Figures 5-23** and **5-24** that there is not a strong temporal correlation in SSE across sites in the pre-treatment period, and in **Figure 5-26** that our "treatment effect" would largely be a function of unexplained changes in the reference basins. Further, **Figures 5-25** and **5-26** show that fitting a common slope between reference and treatment basins would not be appropriate in four of our six sites.

Given the limited number of sediment-producing storms, and the stochastic nature of sediment export, we conclude that it is not appropriate to try to draw strong conclusions about harvest effects on suspended sediment transport from this study. This finding is consistent with the two-year post-harvest report (McIntyre *et al.* 2018).

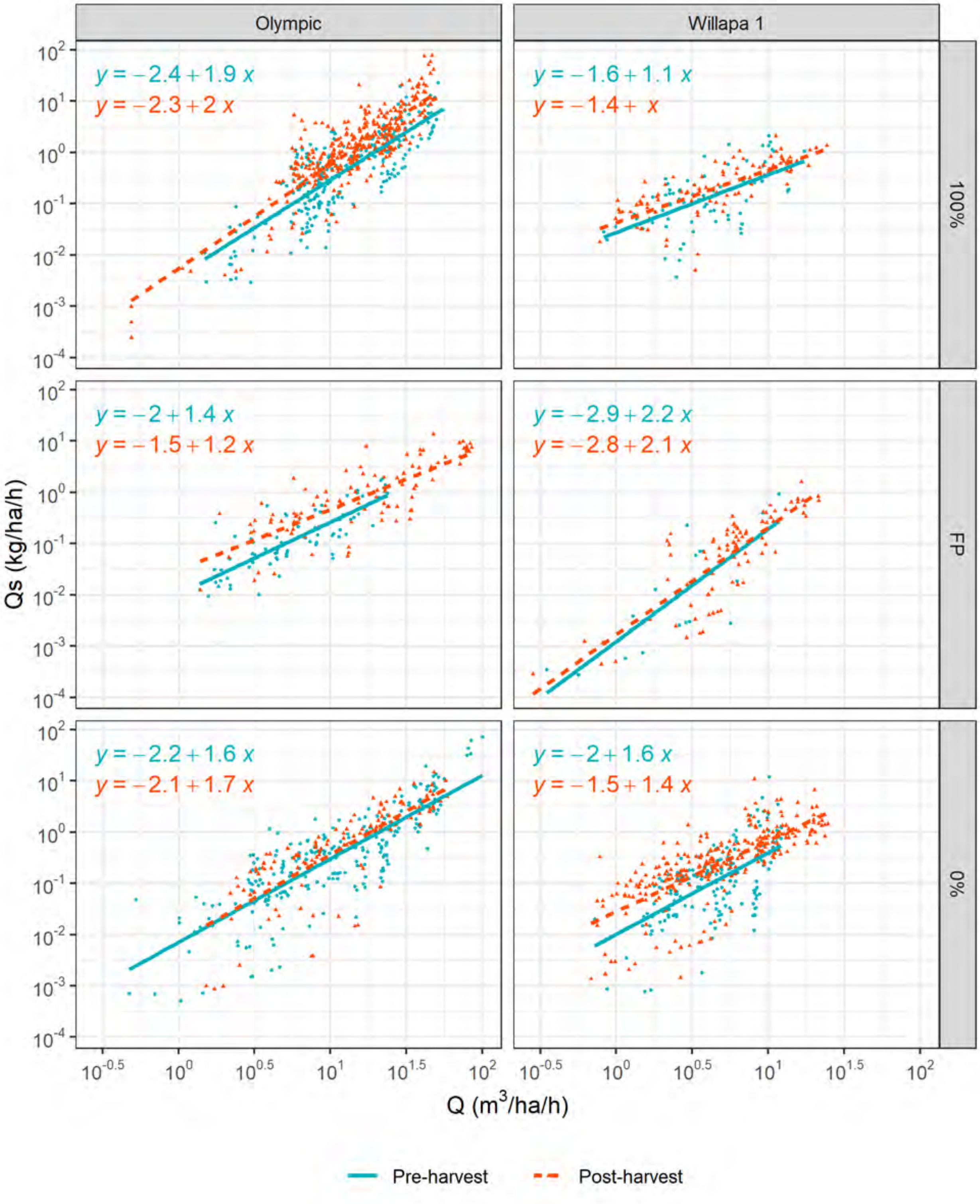

Figure 5-25. Treatment basin suspended sediment yield (Qs) as a function of discharge (Q) in the pre-harvest (top equation) and post-harvest (bottom equation) periods.

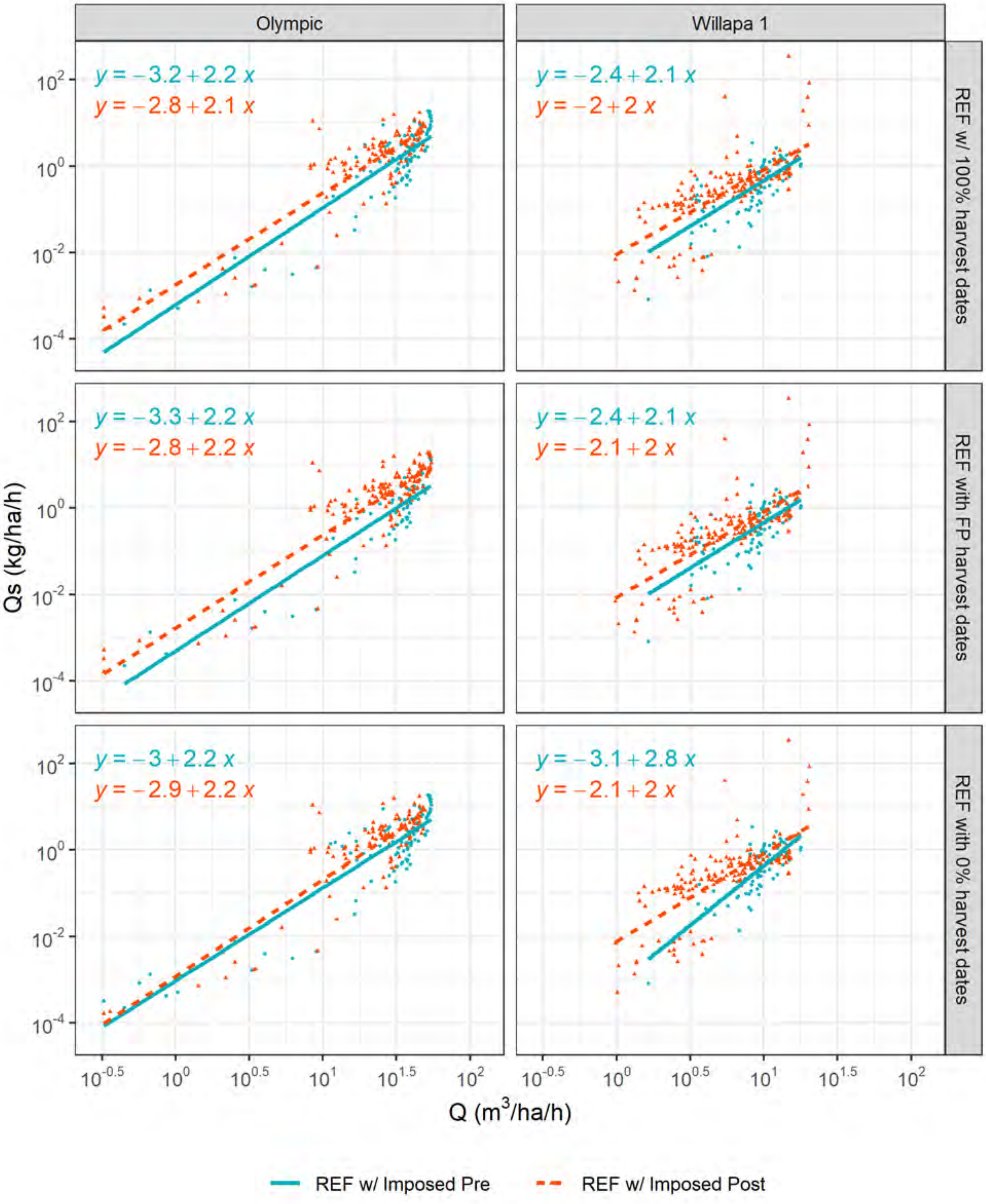

Figure 5-26. Reference basin suspended sediment yield (Qs) as a function of discharge (Q) and treatment basin harvest timing in the pre-harvest (top equation) and post-harvest (bottom equation) periods.

5-7. DISCUSSION

Research has shown that changes in evapotranspiration associated with harvest lead to increases in annual water yield, though the magnitude and timing of change is affected by many factors (Bosch and Hewlett 1982; Stednick 1996; Jones and Post 2004; Brown *et al.* 2005; Moore and Wondzell 2005). In the Pacific Northwest, basins with 80% clearcut harvest have been shown to yield 483 to 615 mm more water per year in the Oregon Coast Range (Harr *et al.* 1975; Harris 1977; Harr 1983), 290 to 410 mm in the Oregon Cascades (Harr and McCorison 1979; Harr *et al.* 1982; Harr 1983, 1986), and 360 mm on Vancouver Island (Hetherington 1982). In rain-dominated areas, measurable annual runoff is thought to generally increase by as much as 6 mm per year for each percent of the basin harvested above some threshold, or -2 to 8 mm/day following 100% forest removal with strong seasonal variations in the response (Hicks *et al.* 1991; Jones and Post 2004; Moore and Wondzell 2005). Over time, discharge decreases from its post-harvest peak as young vegetation with higher evapotranspiration rates becomes established (Du *et al.* 2016; Perry and Jones 2016), though complete recovery to pre-harvest conditions may take 25-40 years (Hicks *et al.* 1991; Du *et al.* 2016). Even in cases where harvest has only a small effect on annual yield (e.g., 5% increase) there may be dramatic changes in the timing and magnitude of summer discharge including decreases (Winkler *et al.* 2017; Gronsdahl *et al.* 2019).

Results from this study are largely consistent with previous research. In the rainforests of the Olympic mountains which receive more than 3700 millimeters of rainfall annually, water yields increased by an average of 60 to 1830 mm/yr (+5 to 75%) in basins that were 43% to 100% clearcut. In the dryer Willapa Hills, water yields increased by 70 to 600 mm/yr in basins that were 89% to 100% clearcut. On average, discharge increased 1 to 18 mm/yr for each percent of the watershed that was harvested, although strong variation existed as a function of buffer treatment, climate, and precipitation.

All treatments exhibited statistically significant changes in magnitude and frequency over the period of study and in every single post-treatment year. In the 100% treatment basins, the biggest change was a reduction in baseflow (RI <2). These decreases were offset by small increases in stormflow. In the FP and 0% treatment basins, the most consistent change was an increase (1.3 to 4.6 mm/day) in the median water yield, though large differences were evident among blocks. In the Olympic block, absolute change in water yield increased with storm magnitude, while it reached its maximum at some point between the 2-day and 7-day RI in the Willapa 1 block.

While most changes were consistent with our expectations, the observed base flow decreases in the 100% treatment basins were not expected despite similar observations in other studies. In the two 100% treatment basins with complete buffering of the perennial channel, base flows (RI <2 days) decreased following harvest even though 45% and 89% of the basin area was clearcut harvested. While we do not have the information needed to determine the exact cause, decreased base flows in the 100% basins may reflect increased evapotranspiration in the riparian zone during times when rain is absent and soil moisture is low. A recent study showed that groundwater evapotranspiration can be spatially restricted to riparian areas accounting for 6% to 18% of the total evapotranspiration in a headwater basin (Tsang *et al.* 2014), and hydraulic simulations indicate that streamflow is very sensitive to where harvest occurs relative to the stream channel (Abdelnour *et al.* 2011). Thus, riparian plants may have been light-limited prior to harvest and increased light availability associated with adjacent harvest increased evapotranspiration enough to decrease streamflow during relatively dry periods when soil water is depleted and stream discharge is low.

Although stormflow increased in all FP and 0% treatment basins, it did not translate to a commensurate increase in suspended sediment export. The basins appeared to be supply limited both before and after harvest based on the lack of sediment transport across a range of large storm events.

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CHAPTER 6 - NITROGEN EXPORT

William Ehinger and Stephanie Estrella

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6-1. ABSTRACT

The purpose of this study was to estimate the effect of timber harvest on the quantity of instream nitrogen exported from headwater streams. We sampled eight non-fish-bearing stream (Type Np) catchments, ranging in size from 11.8-44.3 ha, distributed in two blocks of four streams each, with one block in the Willapa Hills of southwestern Washington State and one on the west side of the Olympic Peninsula. Each block included one unharvested reference site and three clearcut harvest sites with one of three different riparian buffer treatments: 100% treatment (two-sided 50-ft width riparian buffer along the entire length of the Type Np stream network), FP treatment (two-sided 50-ft width riparian buffer along at least 50% of the Type Np stream network, according to current Forest Practices Rules), and 0% treatment (harvested to the stream edge with no riparian buffer). We measured stream discharge and collected water samples for the analysis of nitrogen concentration from October 2006 through September 2011 (pre-harvest and post-harvest periods) and again from July 2015 through June 2017 (extended sampling period). Nitrogen concentrations were determined from unfiltered water samples so that our nitrogen export estimates included both particulate and dissolved fractions.

We found greater variability in pre-harvest total nitrogen (total-N) and nitrate nitrogen (nitrate-N) concentration among the study sites than expected. Pre-harvest N exports ranged from 1.8 to 14.4 kg/ha/yr for total-N and 1.7 to 13.3 kg/ha/yr for nitrate-N. Post-harvest, mean total-N and nitrate-N concentrations increased at all treatment sites, but the magnitude of the increase varied greatly among sites. Post-harvest N export ranged from 8.2 to 32.9 kg/ha/yr (7 to 358% increase) for total-N and 7.3 to 30 kg/ha/yr (13 to 327% increase) for nitrate-N. The estimated change, relative to the reference sites, was greatest in the 0% treatment, intermediate in the FP treatment, and lowest in the 100% treatment.

In the extended sampling period, total-N export declined from post-harvest levels at three sites and increased slightly at three sites while nitrate-N export declined from post-harvest levels at four sites and increased slightly at two sites. Only one of our sites had recovered to pre-harvest export rates in the extended period. There was no consistent response in nitrogen concentration and export to buffer treatment.

6-2. INTRODUCTION

Forest practices are one of the factors influencing nutrient loads exported from streams. An increase in nutrient loads may increase primary productivity in downstream receiving waters, leading to a decrease in dissolved oxygen with decomposition (Roberts *et al.* 2008). Because much of the land managed for timber production in western Washington drains to Puget Sound, Grays Harbor, and Willapa Bay, the effect of forest practices on nutrient loads is a concern for state environmental regulators.

Forest practices may influence stream chemistry through changes in (1) geological weathering, (2) precipitation chemistry, hydrology, and temperature, (3) chemical uptake and transformation through terrestrial biological processes, (4) physical and chemical reactions in soils, and (5) processes within aquatic ecosystems (Feller 2005). Timber harvest and subsequent control of vegetation regrowth reduces canopy interception of rainfall and evapotranspiration of soil water leading to an increase in runoff (Likens *et al.* 1970; Bosch and Hewlett 1982; Harr 1983; Stednick 1996; Feller 2005; Moore and Wondzell 2005). Nitrate concentrations in soil water and streams may increase with a decrease in uptake resulting from vegetation removal (Dahlgren 1998; Feller *et al.* 2000), an increase in microbial nitrification from warmer soil temperatures (Feller 2005; Boczulak *et al.* 2015), slash burning (Fredriksen 1971; Stark 1979; Feller and Kimmins 1984; Gravelle *et al.* 2009), and growth of nitrogen-fixing alder (Feller 2005). Forest practices may also adversely affect soil mycorrhizae, at least temporarily (Harvey *et al.* 1980; Hagerman *et al.* 1999), which may further decrease uptake. An increase in nitrate concentration combined with an increase in runoff may intensify leaching of nutrients from the soil and export of nutrients downstream.

Numerous studies have measured increases in stream concentration of nitrate, especially during the first fall freshets, and increases in nitrate export post-harvest (Likens *et al.* 1970; Brown *et al.* 1973; Feller and Kimmins 1984; Harr and Fredriksen 1988; Dahlgren 1998; Gravelle *et al.* 2009; Schelker *et al.* 2016). Generally, the higher the proportion of a watershed harvested, the greater the increase in concentrations of soil and soil water nitrate (Feller *et al.* 2000) and in concentrations of stream nitrate (Stark 1979; Martin *et al.* 1984; Fowler *et al.* 1988; Tiedemann *et al.* 1988). In the Hard Rock Study Phase 1 report (*McIntyre et al.* 2018, Chapter 9 – *Nutrient Export*), we reported an increase in total nitrogen (total-N) and nitrate-nitrogen (nitrate-N) export from all treatments in the two-year post-harvest period. The estimated change, relative to the references, was greatest in the 0% treatment, intermediate in the Forest Practices treatment, and lowest in the 100% treatment, which was consistent with our expectations of increased nitrogen export with an increase in the proportion of the watershed harvested and an increase in annual runoff.

Although long-term studies are few, researchers have reported recovery of post-harvest nitrate concentrations to pre-harvest levels in two to three years (Feller and Kimmins 1984) and in six years (Brown *et al.* 1973) or the beginning of recovery toward pre-harvest concentrations in three years (Gravelle *et al.* 2009). Another study found that nitrate loads were still elevated ten years after logging (Harr and Fredriksen 1988). The objective of the Hard Rock Study extended monitoring is to measure the response of nutrient concentration and export from the different buffer treatments seven and eight years after timber harvest.

6-3. METHODS

6-3.1. STUDY SITES

Cost and logistical constraints restricted nutrient sampling and flow monitoring to only the Olympic Peninsula (OLYM) and one of the Willapa Hills (WIL1) blocks (eight sites total, two replicates of each experimental treatment). These eight study sites were non-fish-bearing, perennial (Type Np), first-, second-, and third-order stream catchments draining into the Clearwater River, Humptulips River, and Wishkah River in the Olympic physiographic region, and the North River and Willapa River in the Willapa Hills region of southwest Washington (**Table 6-1**). Catchment area above the stream discharge monitoring locations ranged from 11.8 to 44.3 ha (**Table 6-2**). Areas of some sites differ from those presented in **Table 2-2** (see Chapter

2 – *Study Design* in this report) because we could not always measure discharge at the regulatory break between fish-bearing and non-fish-bearing stream segments (F/N break).

The study sites were located in managed Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*)-dominated second-growth forests on private, state, and federal forestlands. Site-wide estimates of vegetation type were not available, but the overstory ranged from 78 to 100% conifer based on basal area within a 50-ft (15.2-m) wide riparian buffer. Stand age ranged from 30 to 80 years. Sites were located in areas dominated by competent lithology types, with average Type Np channel gradients ranging from 16 to 31%. Site selection criteria are presented in McIntyre and colleagues (2018, Chapter 2 – *Study Design*).

The climate in western Washington, as described by the Western Regional Climate Center (wrcc.dri.edu), is cool and comparatively dry in summer, and mild, wet, and cloudy in winter. In the interior valleys, measurable rainfall is recorded on 150 days each year and on 190 days in the mountains and along the coast. Annual precipitation ranges from 1,778 to 2,540 mm (70 to 100 in) over the Coastal Plains to 3,810 mm (150 in) or more along the windward slopes of the mountains. Average estimated 30-year (1981-2010) minimum and maximum monthly temperatures were -2.4 to 1.2°C (27.7 to 34.2°F) and 22.2 to 25.0°C (72.0 to 77°F) across our sites in December and August, respectively (PRISM Climate Group 2013). The average estimated annual precipitation over that same 30-year period was 2,242 to 3,855 mm (88 to 152 in).

Table 6-1. Elevation, lithology, stream gradient, and stream order (Strahler 1952) for the Olympic (OLYM) and Willapa 1 (WIL1) blocks where discharge and nutrient concentrations were determined. Treatments included unharvested reference sites (REF) and sites that received a clearcut harvest with one of three riparian buffer treatments along the Type Np Water riparian management zone: two-sided 50-ft (15.2-m) riparian buffers of 100%, Forest Practices (FP), and 0%. Elevation was at the field-verified F/N break. We calculated stream gradient as the average stream gradient for the entire Type Np stream network using a 10-m digital elevation model in ArcMap (ESRI 2004).

Block	Treatment	Elevation (m)	Lithology	Stream Gradient (%)	Stream Order
OLYM	REF	163	Basalt flows and flow breccias	18	3
	100%	72	Tectonic breccia	27	3
	FP	277	Basalt flows and flow breccias	25	3
	0%	233	Basalt flows and flow breccias	31	2
WIL1	REF	200	Basalt flows and flow breccias	19	2
	100%	198	Basalt flows and flow breccias	18	2
	FP	197	Basalt flows and flow breccias	19	1
	0%	87	Terraced deposits	16	3

Table 6-2. Catchment area above the flow gauge, percent of catchment harvested, and percent hardwood trees in riparian stand.

Block	Treatment	Area (ha)	% Harvested	% Hardwood in Riparian Stand
OLYM	REF	44.3	0	1
	100%	22.1	43	16
	FP	17.3	88	12
	0%	13.1	100	22
WIL1	REF	11.8	0	4
	100%	26.2	89	1
	FP	14.4	94	0
	0%	27.7	100	9

6-3.2. EXPERIMENTAL TREATMENTS

The four experimental treatments included in the study are (**Figure 6-1**):

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 two-sided 50-ft (15.2-m) riparian buffer along the entire perennial stream length,

3) **Forest Practices treatment (FP):** clearcut harvest with a current Forest Practices' two-sided 50-ft (15.2-m) riparian buffer along at least 50% of the perennial stream length, and

4) **0% treatment (0%):** clearcut harvest with no riparian buffer.

The riparian management zone for Type Np and non-fish-bearing seasonal (Type Ns) waters in western Washington also includes a two-sided, 30-ft (9.1-m) wide equipment limitation zone (WAC 222-30-021(2)) to limit the amount of ground disturbance near the stream. Timber harvest on potentially unstable slopes or landforms that have the potential to deliver sediment or debris to a public resource, or that have the potential to threaten public safely, require an environmental checklist in compliance with the State Environmental Policy Act (SEPA; WAC 222-16-050(1)(d)), so harvest in these areas is generally avoided. In this study, no harvest activities were conducted on any potentially unstable slopes, regardless of buffer treatment, and all treatments included the equipment limitation zone.

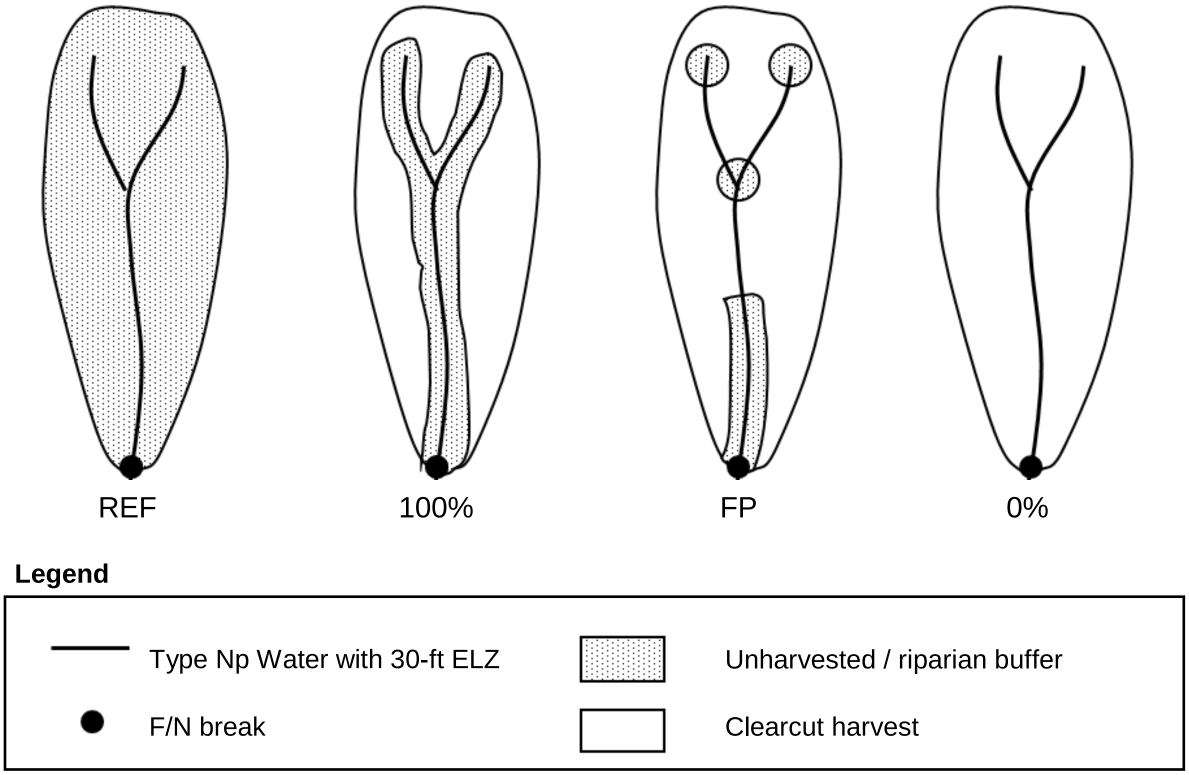

Figure 6-1. Schematic of the four experimental treatments included in the Hard Rock Study. Treatments include unharvested reference sites (REF) and sites receiving a clearcut harvest with one of three riparian buffer treatments along the Type Np Water riparian management zone: two-sided 50 ft (15.2 m) riparian buffers of 100%, Forest Practices (FP), and 0%. FP and 100% treatments include 56-ft (17.1-m) radius buffers around Type Np intersections and headwater springs. All streams are protected by a two-sided 30 ft (9.1 m) equipment limitation zone (ELZ).

6-3.3. DATA COLLECTION

6-3.3.1. Streamflow

We monitored streamflow in the eight sites of the OLYM and WIL1 blocks from September 2006 to September 2017 (see Chapter 5 – *Stream Discharge, Turbidity, and Suspended Sediment Export* in this report). Our intent was to collect two complete years of flow data for the pre-harvest, post-harvest, and extended monitoring periods; however, we collected only one year of pre-harvest data at three sites. The WIL1-0% and OLYM-FP were harvested earlier than expected (less than two years after study initiation) and the stage height measurements used to predict discharge at the OLYM-100% were compromised in the first months of the study. As a result, these three sites have only one complete year of pre-harvest flow data and nitrogen (N) export estimates.

6-3.3.2. Water Sample Collection and Chemical Analysis

We manually collected water samples at six to eight week intervals from October 2006 through September 2011 and again from July 2015 through June 2017, unless the site was inaccessible due to weather, road maintenance, or harvest activities. Water was collected at the flow gauging location into acid-washed Nalgene bottles containing concentrated sulfuric acid as a preservative. Sample bottles were cooled to <6°C and transported to the lab within 24 hours.

We were unable to manually sample high flow events regularly because of the long distances to and between sites. Instead, we implemented turbidity threshold sampling (TTS; Lewis and Eads 2009) to collect water samples during high flow events across the range of turbidity and flow values. TTS was designed to collect samples for analysis of suspended sediment concentration (SSC). The SSC and turbidity values are used to build a regression model predicting SSC from the continuous turbidity data. We analyzed these samples for nutrients and used a similar approach to predict total-N and nitrate-N concentrations using the continuous flow and, in some cases, turbidity data. Twelve turbidity thresholds, ranging from 10 to 1,600 nephelometric turbidity units, were set for both the rising and falling limbs of the turbidity graph. Samples were pumped into acid-washed Nalgene bottles by an ISCO TM pump sampler when the turbidity value crossed a (rising or falling) threshold and flow exceeded approximately ~10 to 20 L/s. We set the discharge threshold to avoid triggering the pump sampler during turbidity events unrelated to discharge events (e.g., wildlife crossing upstream). Sample bottles were collected as soon as practicable after collection (within 7 days), transferred into acid-washed Nalgene containers, preserved, chilled, and transported to the lab, as described above.

We were concerned about the effect of biological activity (uptake and transformation of N) while bottles were left in the pump sampler. We independently tested the effect of storing samples for one to four weeks at ambient air temperatures (daily mean 9.9 to 15.1°C) prior to adding preservative and cooling to <6°C and found no measureable effect on either total-N or nitrate-N concentration (McIntyre *et al.* 2018, Chapter 9 – *Nutrient Export*, Appendix 9-A). All samples were analyzed for total-N and nitrate-N (**Table 6-3**). Total-N and nitrate-N were determined from unfiltered water samples and represent the sum of particulate and dissolved forms. We used totals because these were used in other studies in forested streams and are used in models for estimating dissolved oxygen in marine systems (Mohamedali *et al.* 2011). All chemical analyses were done by the Washington State Department of Ecology’s Manchester Environmental Laboratory in Port Orchard, Washington.

Atmospheric deposition data were obtained from National Atmospheric Deposition Program site WA14 at the Hoh River Ranger station in Olympic National Park (47.8597, -123.9325, elevation 182 m), and WA21 in La Grande, WA (46.8353, -122.2867, elevation 617 m) (NADP 2018).

Table 6-3. Nitrogen analytical methods.

Analyte	Method	Reporting Limit (µg/L)
Nitrate-N[1]	4500-NO3-I	10
Total-N[1]	4500-N B	25

[1] APHA (2016) *Standard Methods for the Examination of Water and Waste*water, 22nd edition.

6-3.4. NUTRIENT LOAD CALCULATIONS

We used a regression model to empirically predict nutrient concentration as a function of discharge and turbidity. We calculated loads (product of estimated concentration and discharge) of total-N and nitrate-N following the methods of Helsel and Hirsch (2002) except we used discharge data collected at 10-minute intervals, rather than the more commonly used mean daily discharge. We based our calculations on the shorter time interval because storm events were often short-lived (less than one day) and both discharge and nutrient concentration changed rapidly over a given event. The predictive equations using the 10-minute data provided better temporal resolution and fit the data better than when using daily mean values. We used **Equation 6-1** to calculate total-N and nitrate-N:

$$\mathrm{Log}[N]_\mathrm{i} = \beta_0 + \beta_1 LogQ_i + \beta_2 (LogQ)_i{}^2 + \beta_3 \sin\frac{2\pi t}{365.25} + \beta_4 \cos\frac{2\pi t}{365.25} + \beta_5 Log(T_i) + \varepsilon_i \qquad \textbf{(Eq. 6-1)}$$

where: $\mathrm{Log}[N]_\mathrm{i}$ is base 10 logarithm of total-N or nitrate-N concentration of the ith sample,
$\beta_0 - \beta_5$ are regression coefficients,
$\mathrm{Log}Q$ is base 10 logarithm of flow,
sin and cos functions are seasonal terms,
t is time (years),
$\mathrm{Log}(T)$ is the base 10 logarithm of (turbidity), and
ε_i is an error term.

We developed separate regression models for the pre-harvest, post-harvest, and extended periods at all six treatment sites (**Table 6-4**) because there was a substantial and significant difference in the regression relationship among the periods. One model was used for the entire 2006 to 2011 period at each of the reference sites but new models were developed for the extended period because the predictive relationship differed. Flow and the seasonal terms were used in the total-N and nitrate-N models for all sites and all time periods. The turbidity term was included in the model where it substantially improved the model's predictive capability (higher R^2 and lower standard error) or improved the distribution (normality or homogeneity) of the residuals. We examined the residuals of each regression to ensure that they were homoscedastic and approximately normally distributed.

Concentration estimates were adjusted using a smearing correction (Duan 1983) to correct for bias introduced when transforming from log-scale to untransformed scale. Instantaneous N loads were calculated as the product of predicted nutrient concentration and flow for each 10-minute record. We assumed that each instantaneous load value applied to the entire preceding 10-minute interval so that the cumulative 10-minute load equaled 600 (seconds) times the instantaneous load (kg/sec). Annual export values are the sum of these cumulative 10-minute loads for each complete year immediately before the start of timber harvest and each complete year immediately after the end of harvest activities divided by the area of the drainage basin above the flume (units= kg/ha/yr).

All regressions and load calculations were done using SYSTAT 13 statistical software (Systat Software Inc 2009).

Table 6-4. Regression models for estimating nutrient concentration. Separate models were developed for each monitoring period except for the reference sites where one model was used for both the pre- and post-harvest periods. Adequate regression models could not be developed for nitrate-N in the extended period in WIL1-REF and WIL1-FP, so a flow-weighted mean concentration was used. SE = standard error, Var = variables used in regression (Q = flow, T = turbidity).

Block	Treatment	Period	Total-N				Nitrate-N			
			N	r^2	SE	Var	N	r^2	SE	Var
WIL1	REF	Pre- & Post-	97	0.489	0.151	Q	98	0.435	0.124	Q
		Extended	104	0.223	0.078	Q	104	flow-weighted average		
	100%	Pre-	49	0.442	0.137	Q. T	47	0.593	0.097	Q
		Post-	47	0.403	0.263	Q, T	47	0.496	0.238	Q, T
		Extended	59	0.243	0.155	Q	59	0.420	0.197	Q
	FP	Pre-	31	0.824	0.129	Q, T	32	0.678	0.193	Q,T
		Post-	21	0.705	0.242	Q, T	21	0.585	0.232	Q
		Extended	56	0.181	0.075	Q	56	flow-weighted average		
	0%	Pre-	21	0.698	0.041	Q, T	22	0.746	0.041	Q
		Post-	66	0.444	0.152	Q,T	66	0.564	0.121	Q, T
		Extended	83	0.625	0.056	Q,T	83	0.562	0.078	Q,T
OLYM	REF	Pre-&Post-	125	0.647	0.156	Q, T	114	0.600	0.120	Q
		Extended	56	0.436	0.148	Q, T	54	0.693	0.098	Q
	100%	Pre-	94	0.696	0.126	Q	94	0.722	0.123	Q
		Post-	164	0.519	0.123	Q, T	164	0.511	0.120	Q
		Extended	97	0.410	0.180	Q,T	97	0.220	0.291	Q
	FP	Pre-	23	0.737	0.120	Q, T	23	0.770	0.107	Q, T
		Post-	40	0.521	0.109	Q,	59	0.599	0.118	Q
		Extended	55	0.448	0.145	Q	55	0.518	0.145	Q
	0%	Pre-	74	0.719	0.110	Q, T	74	0.751	0.094	Q,T
		Post-	66	0.742	0.103	Q, T	66	0.776	0.091	Q, T
		Extended	100	0.712	0.167	Q, T	100	0.663	0.203	Q, T

6-3.5. STATISTICAL ANALYSIS

The timing of the timber harvest could not be synchronized across all buffer treatment sites. The start date, end date, and duration of harvest were determined by the landowner and varied among sites (**Table 6-5**). Export from each watershed was highly dependent upon flow, which varied across years. To compare treatment (TRT) and reference (REF) exports across similar time and discharge conditions, we analyzed N export as the difference (TRT minus REF) in annual export or mean flow-weighted concentration between each buffer treatment site and its reference site over the same period (e.g., pre-harvest, post-harvest, or extended period). There were two pre-harvest years for each site except WIL1-0%, OLYM-100%, and OLYM-FP, which each had only one pre-harvest year. All sites had two years in the post-harvest period and two years in the extended period.

We used a generalized linear mixed effects model (GLMM) with site as a random effect and buffer treatment (100%, FP, and 0%), period (pre-harvest, post-harvest, or extended period), and the treatment × period interaction as fixed effects. We initially included block as a random effect but dropped it because the variance estimate associated with block was zero (i.e., block did not explain any additional variation in any dependent variables). We used the Kenward-Roger method (Kenward and Roger 1997) for estimating the denominator degrees of freedom for fixed effects because of the unbalanced design (unequal number of pre-harvest years among the sites). We used SAS software version 9.4 for GLMM analyses (SAS Institute Inc. 2013).

We evaluated two basic hypotheses:

1) What is the magnitude of change in nutrient (total-N and nitrate-N) concentration and annual export relative to a reference site following timber harvest? This was addressed with three post-hoc comparisons testing the following hypothesis for each buffer treatment:

    a) H_0: $(TRT_{100\%,Pre}-REF_{Pre}) = (TRT_{100\%,Post}-REF_{Post}) = (TRT_{100\%,Ext}-REF_{Ext})$ (**Eq. 6-2**)

    b) H_0: $(TRT_{FP,Pre}-REF_{Pre}) = (TRT_{FP,Post}-REF_{Post}) = (TRT_{FP,Ext}-REF_{Ext})$ (**Eq. 6-3**)

    c) H_0: $(TRT_{0\%,Pre}-REF_{Pre}) = (TRT_{0\%,Post}-REF_{Post}) = (TRT_{0\%,Ext}-REF_{Ext})$ (**Eq. 6-4**)

where: TRT is export from the treatment site,
REF is export from the reference site over the same time period, and
Pre, Post, Ext denote pre-harvest, post-harvest, and extended monitoring periods.

2) What are the differences in the magnitude of the change in concentration and export among the three buffer treatments? This was addressed with three *post hoc* comparisons comparing the changes across the three buffer treatments.

Estimates of the effects and the associated 95% confidence intervals are presented. The P-values were not adjusted for multiple comparisons because the large number of comparisons relative to the limited replication of each treatment (two) increases the chance of Type II error and can mask subtle treatment effects. Instead, we consider the P-value, effect size, patterns of the effect size across the buffer treatments, and sample size when interpreting the results.

Table 6-5. Pre-harvest, post-harvest, and extended periods for the six treatment sites included in nutrient export sampling (modified from McIntyre *et al.* 2018, Chapter 3 – *Management Prescriptions*).

Block	Treatment	Period			No. of years Pre-/Post-/Extended
		Pre-harvest	Post-harvest	Extended	
OLYM	100%	02/2008 to 02/2009	03/2009 to 03/2011		1/2/2
	FP	07/2007 to 07/2008	10/2008 to 10/2010	08/2015 to 08/2017	1/2/2
	0%	06/2007 to 06/2009	08/2009 to 08/2011		2/2/2
WIL1	100%	10/2006 to 10/2008	04/2009 to 04/2011		2/2/2
	FP	10/2006 to 10/2008	03/2009 to 03/2011	08/2015 to 08/2017	2/2/2
	0%	04/2007 to 04/2008	01/2009 to 01/2011		1/2/2

6-4. RESULTS

6-4.1. NITROGEN CONCENTRATION

Nitrogen concentrations, both total-N and nitrate-N, varied widely among sites, especially in the pre-harvest period. Mean, flow-weighted, pre-harvest total-N concentration ranged from 120 to 841 µg/L, with the lowest and highest concentrations in adjacent sites, WIL1-100% and WIL-FP, respectively (**Table 6-6**). Post-harvest mean concentration increased at all buffer treatment sites and ranged from 20 to 398 µg/L higher with no obvious relationship with either treatment or block. Concentrations in the extended period were lower relative to post-harvest at all sites except OLYM-100%. However, extended period concentrations were elevated, compared to pre-harvest, at all buffer treatment sites except WIL1-FP and OLYM-0%. Total-N concentration responded differently over time at the two reference sites. OLYM-REF was relatively stable with mean concentrations of 275 and 243 µg/L in the pre-post and extended periods, respectively. In contrast, concentration at WIL1-REF was 621 µg/L in the pre-post but only 328 µg/L in the extended period.

Nitrate-N comprised 85%, on average, of total-N across all samples collected and so the results of nitrate-N concentration were very similar in magnitude and direction to total-N. Nitrate-N concentration increased post-harvest at all sites, then were lower than post-harvest in the extended period at five of six sites, but still higher than pre-harvest levels at all sites but WIL1-FP and OLYM-0%.

6-4.1.1. Seasonal N Concentrations

Total-N and nitrate-N concentrations were seasonally variable with higher concentrations during the first fall flow events and low concentrations during summer low flows. This pattern was most visible in the pre- and post-harvest periods and tended to be weaker or non-existent in the extended period (**Figures 6-2** and **6-3**).

Table 6-6. Mean concentrations (µg/L) of total-N and nitrate-N for the pre-harvest, post-harvest, and extended periods.

Block	Treatment	Period	Total-N	NO_3-N
WIL1	REF	Pre-Post	621	466
		Extended	328	280
		Ext-Pre/Post Difference	-293	-186
	100%	Pre-	120	98
		Post-	437	263
		Extended	254	190
		Pre-Post Difference	317	165
		Ext-Pre Difference	134	92
		Ext-Post Difference	-183	-73
	FP	Pre-	841	800
		Post-	887	906
		Extended	362	339
		Pre-Post Difference	46	106
		Ext-Pre Difference	-479	-461
		Ext-Post Difference	-525	-567
	0%	Pre-	463	450
		Post-	861	727
		Extended	623	552
		Pre-Post Difference	398	277
		Ext-Pre Difference	160	102
		Ext-Post Difference	-238	-175
OLYM	REF	Pre-Post	275	167
		Extended	243	205
		Ext-Pre/Post Difference	-32	38
	100%	Pre-	396	319
		Post-	416	340
		Extended	439	388
		Pre-Post Difference	20	21
		Ext-Pre Difference	43	69
		Ext-Post Difference	23	48
	FP	Pre-	235	225
		Post-	546	473
		Extended	394	307
		Pre-Post Difference	311	248
		Ext-Pre Difference	159	82
		Ext-Post Difference	-152	-166
	0%	Pre-	570	460
		Post-	668	604
		Extended	393	334
		Pre-Post Difference	98	144
		Ext-Pre Difference	-177	-126
		Ext-Post Difference	-275	-270

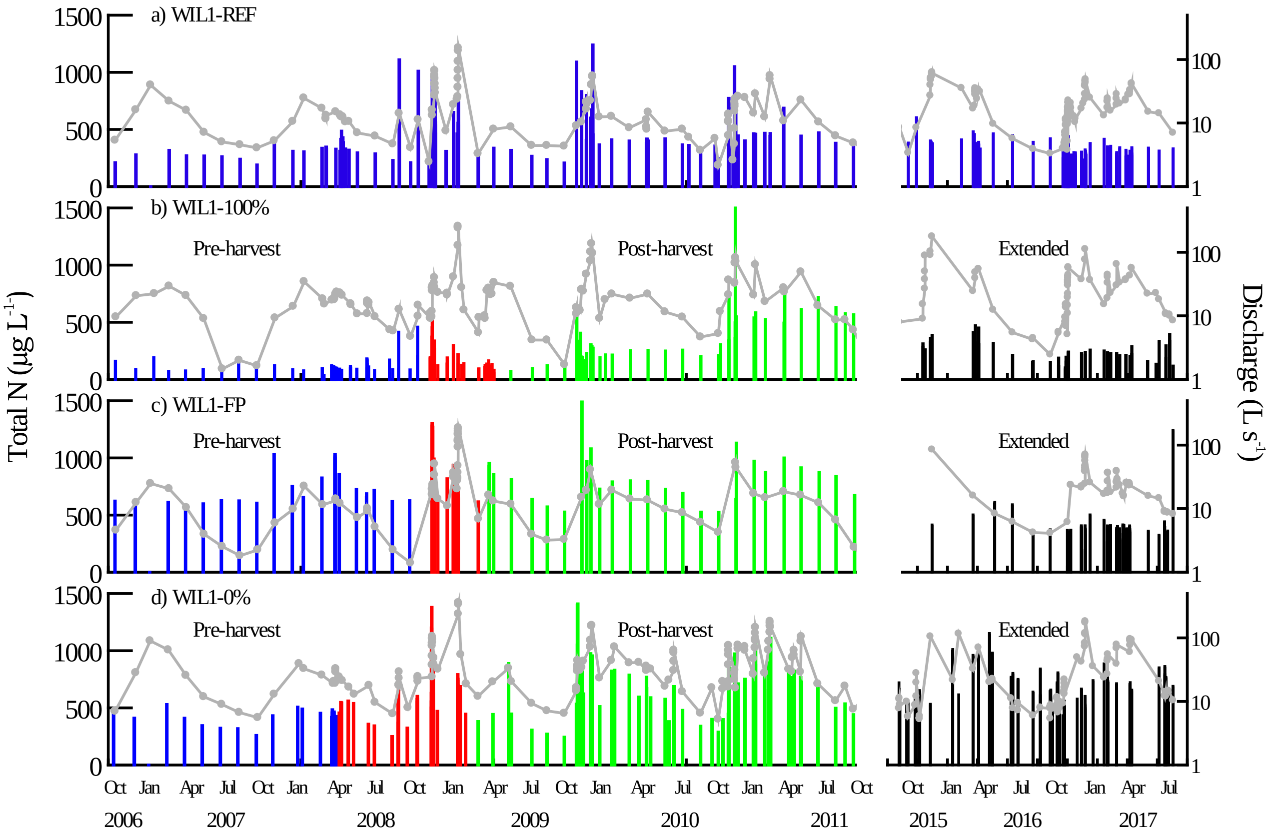

Figure 6-2a. Total-N concentration (bars) and discharge (line) over time in the WIL1 block. Blue = pre-harvest or no harvest; red = during harvest; green = post-harvest; black = extended.

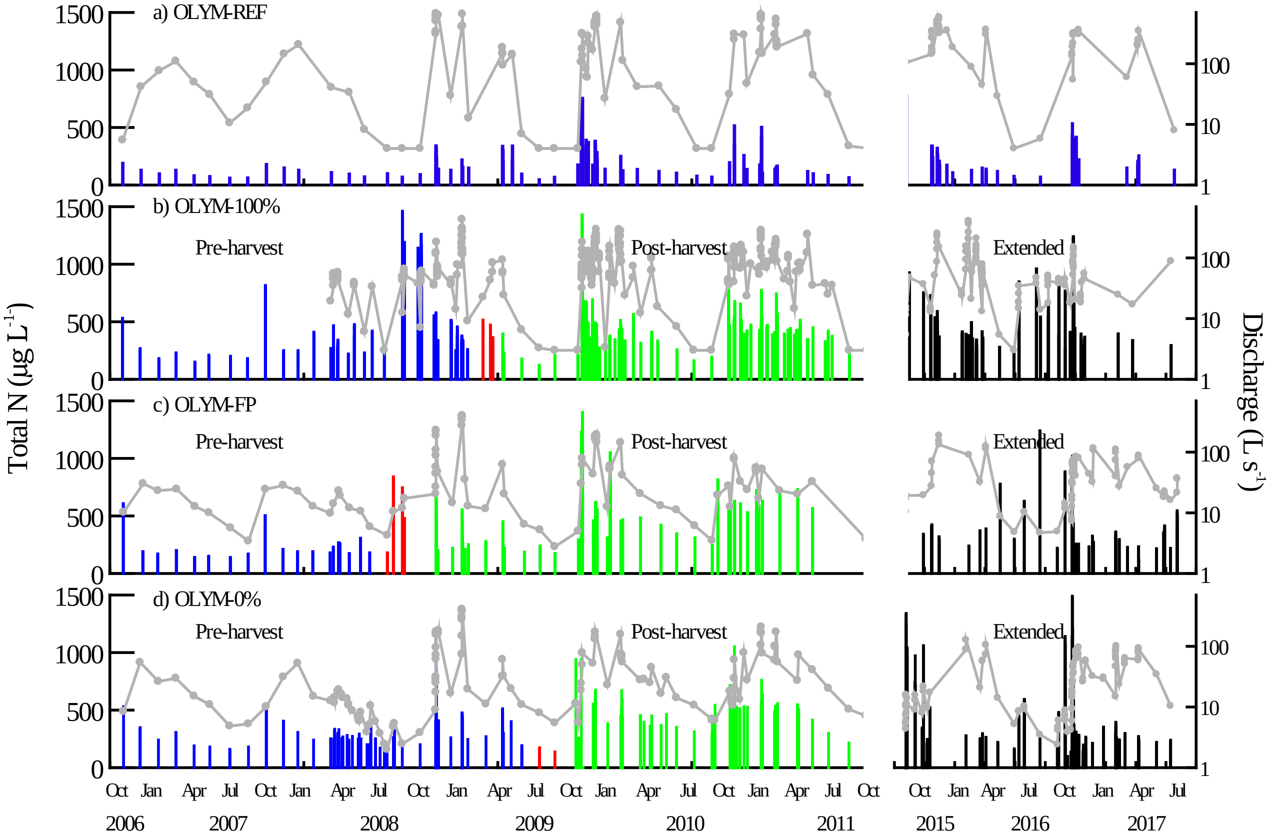

Figure 6-2b. Total-N concentration (bars) and discharge (line) over time in the OLYM block. Blue = pre-harvest or no harvest; red = during harvest; green = post-harvest; black = extended.

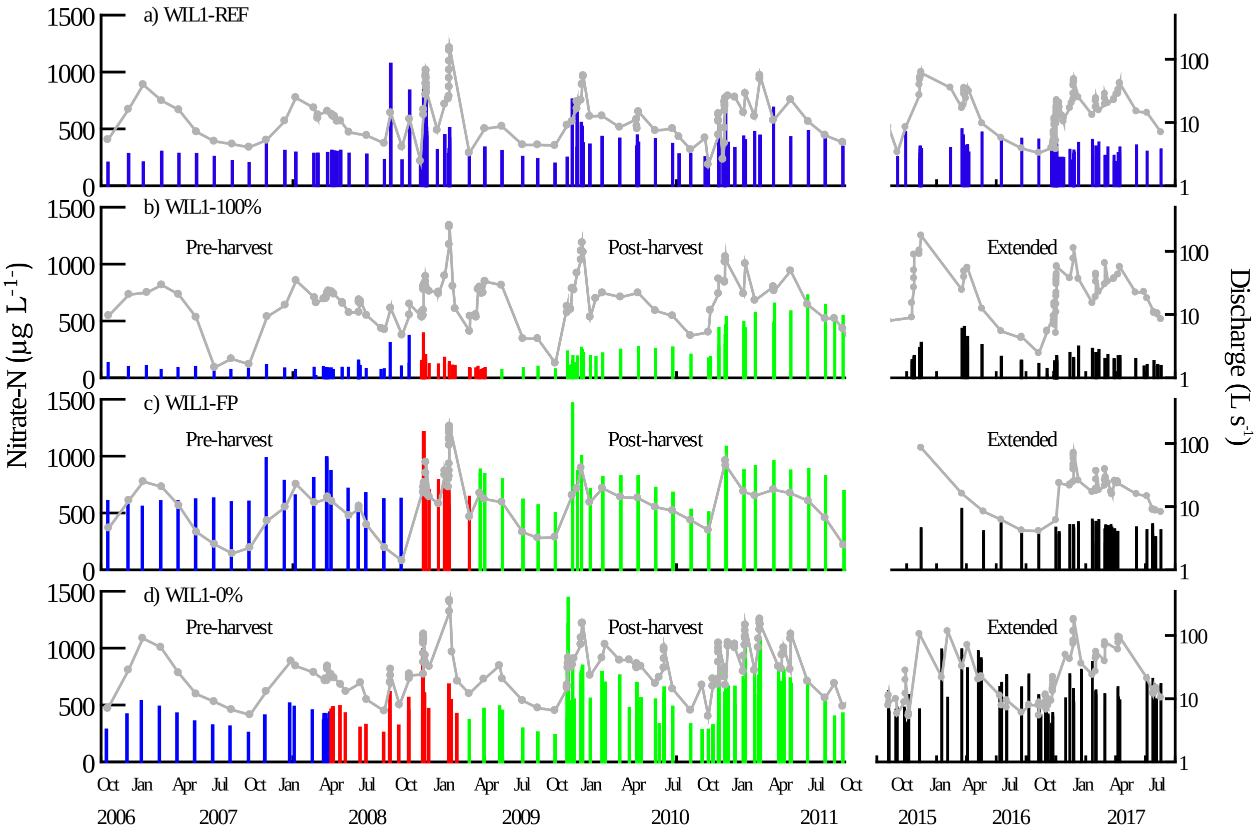

Figure 6-3a. Nitrate-N concentration (bars) and discharge (line) over time in the WIL1 block. Blue = pre-harvest or no harvest; red = during harvest; green = post-harvest; black = extended.

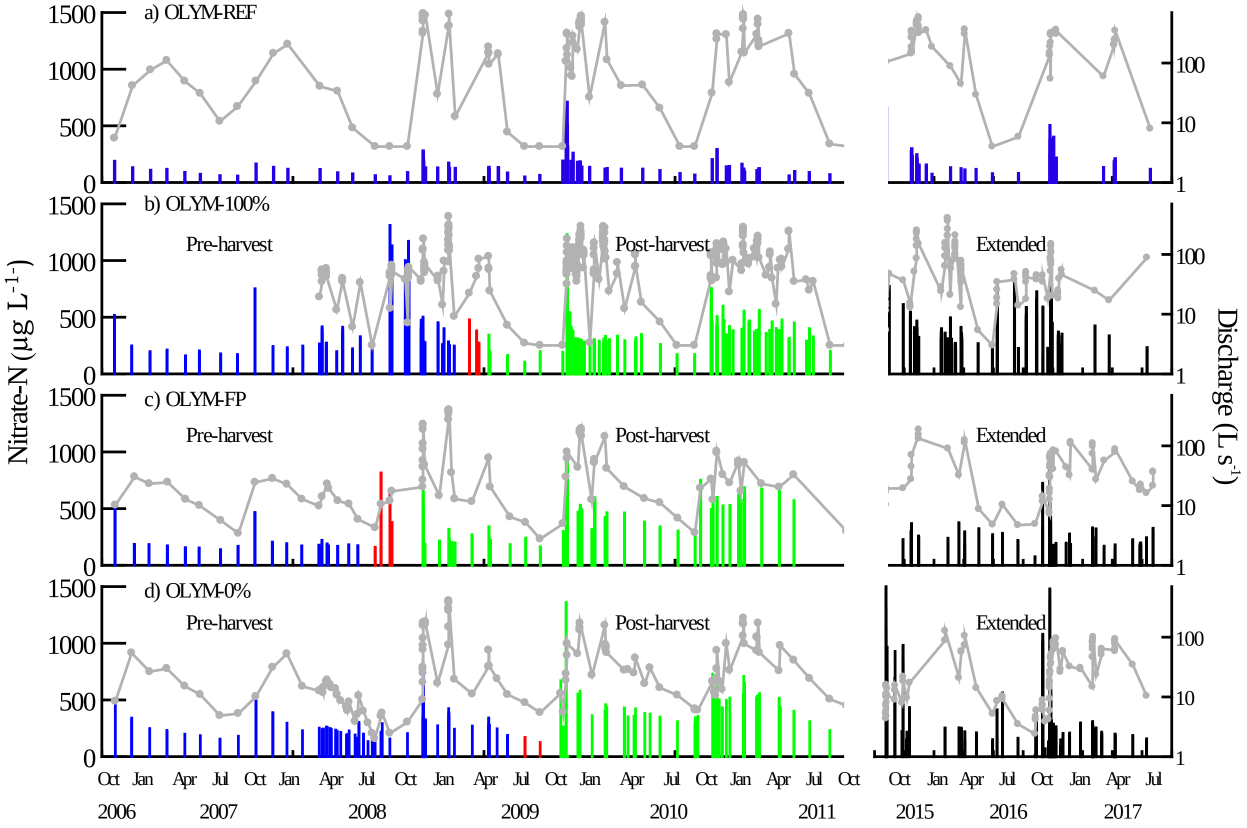

Figure 6-3b. Nitrate-N concentration (bars) and discharge (line) over time in the OLYM block. Blue = pre-harvest or no harvest; red = during harvest; green = post-harvest; black = extended.

6-4.1.2. Buffer Treatment Effects

There was no treatment × period effect for total-N concentration (**Table 6-7**) and the pairwise contrasts (**Table 6-8**) are inconsistent showing slight increases over time in the 100% treatment, decreases in the FP treatment, and larger increases in the 0% treatment ($P > 0.05$).

There was a treatment × period interaction effect ($P < 0.05$) for nitrate-N concentration (**Table 6-7**), however, the results were quite similar to total-N. Relative to the reference treatment, nitrate-N concentration was 177 µg/L lower in the extended period than in the pre-harvest period in the FP treatment, and 147 µg/L higher in the post-harvest period than in the pre-harvest period in the 0% treatment (**Table 6-8**). Differences were also noted between the FP treatment and both the 100% and 0% treatments but only in the extended period.

Table 6-7. Tests of fixed effects for nitrogen concentration and export.

Effect	Num DF	Den DF	F Value	P value
Total-N Concentration				
Treatment	2	3	1.09	0.441
Period	2	21	0.51	0.609
Treatment × Period	4	21	1.67	0.194
Nitrate-N Concentration				
Treatment	2	3	0.96	0.477
Period	2	21	2.43	0.112
Treatment × Period	**4**	**21**	**3.80**	**0.018**
Total-N Export				
Treatment	2	3	2.09	0.271
Period	**2**	**21.1**	**14.04**	**0.000**
Treatment × Period	4	21.1	1.31	0.297
Nitrate-N export				
Treatment	2	3	1.76	0.313
Period	**2**	**21.1**	**16.86**	**<0.0001**
Treatment × Period	4	21.1	1.81	0.165

Table 6-8. Results of pairwise comparisons for nutrient concentration are shown below. Comparisons were made of pre-harvest vs. post-harvest and extended periods and between buffer treatments for post-harvest and extended periods. Comparisons in **bold** indicate the null hypothesis was rejected at P <0.05. P-values were not adjusted for multiple comparisons.

Hypothesis	Change	P-value	Lower	Upper
Total-N				
100%-Pre- vs. Post-	26	0.761	-147	199
100%-Pre- vs. Extended	75	0.378	-98	248
FP-Pre- vs. Post-	-15	0.861	-188	158
FP-Pre- vs. Extended	-130	0.133	-303	43
0%-Pre- vs. Post-	134	0.121	-39	308
0%-Pre- vs. Extended	145	0.096	-28	318
100% vs. FP Post-	-40	0.735	-285	204
100% vs. FP Extended	-205	0.096	-450	40
100% vs. 0% Post-	109	0.366	-136	354
100% vs. 0% Extended	70	0.557	-175	315
FP vs. 0% Post-	149	0.219	-96	394
FP vs. 0% Extended	**275**	**0.029**	**31**	**520**
Nitrate-N				
100%-Pre- vs. Post-	22	0.752	-121	165
100%-Pre- vs. Extended	57	0.414	-86	200
FP-Pre- vs. Post-	51	0.463	-92	194
FP-Pre- vs. Extended	**-177**	**0.018**	**-320**	**-34**
0%-Pre- vs. Post-	**147**	**0.044**	**4**	**290**
0%-Pre- vs. Extended	131	0.071	-12	274
100% vs. FP Post-	29	0.765	-173	232
100% vs. FP Extended	**-234**	**0.026**	**-436**	**-32**
100% vs. 0% Post-	125	0.212	-77	328
100% vs. 0% Extended	73	0.459	-129	276
FP vs. 0% Post-	96	0.336	-106	298
FP vs. 0% Extended	**307**	**0.005**	**105**	**509**

6-4.2. NITROGEN EXPORT

6-4.2.1. Regression Models

The regression models tended to perform better in the pre- and post-harvest periods than in the extended period (**Table 6-4**). R^2 values ranged from 0.403 to 0.824 for total-N and 0.435 to 0.776 for nitrate-N in the pre- and post-harvest periods. In contrast, the extended period total-N R^2 values ranged from 0.181 to 0.712 and nitrate-N from 0.220 to 0.693 with non-significant ($P > 0.05$) regressions at two sites. In these two cases, WIL1-REF and WIL1-FP, we calculated nitrate-N load in the extended period as the product of flow-weighted mean concentration.

We relied on automatic pump samplers to collect samples at high discharges. Differences in the number, timing, and magnitude of flow events as well as harvest timing, site access, and occasional equipment malfunction, affected the range of discharges actually sampled across each period (**Table 6-9**). The Q-s, discharge at the time of sampling, exceeded the 99th percentile of recorded discharges in all periods at all sites except three. During the pre-harvest period at WIL1-100%, WIL1-FP, and OLYM-FP, the highest discharge sampled was approximately equal to the 91st, 94th, and 84th percentile of discharge for that period.

6-4.2.2. Seasonal N Loads

Post-harvest total-N and nitrate-N export were higher at all buffer treatment sites both in absolute terms and relative to the REF site (**Figure 6-4** and **Figure 6-5**). This increase in total-N and nitrate-N export tended to be highest during the high flow months in the fall and early winter.

6-4.2.3. Annual N Loads

Nitrogen export varied widely across sites even prior to harvest. Mean, pre-harvest total-N export ranged from 1.8 kg/ha/yr in WIL1-100% to 14.4 kg/ha/yr in WIL1-FP (**Table 6-10**). That these sites are adjacent to each other (and to WIL1-REF) shows the variability in concentration and export over small spatial scales even in the pre-treatment period. Mean pre-harvest total-N export from the OLYM buffer treatment sites ranged from 5.2 to 12.0 kg/ha/yr.

Total-N export increased in the post-harvest period at all treatment sites, with the smallest increase in the OLYM-100% site and the largest in the OLYM-0% treatment (**Table 6-10**; **Figure 6-6**). On a percentage basis, increases ranged from 7% in OLYM-100% to 358% in WIL1-100%.

In the extended period, changes in total-N export varied widely among the treatment sites with three sites remaining elevated at or slightly above post-harvest levels (OLYM-100%, OLYM-FP, and WIL1-0%) and three sites decreasing from post-harvest levels (OLYM-0%, WIL1-100%, and WIL1-FP; **Table 6-10**). Mean total-N export in the extended period ranged from 5.9 kg/ha/yr in WIL1-100% to 21.6 kg/ha/yr in WIL1-0%. Total-N export was still elevated compared to pre-harvest at all treatment sites except WIL1-FP.

Annual total-N export was relatively stable over the course of the study in the OLYM-REF site, averaging 6.3 kg/ha/yr and varying over a range of less than 2.6 kg/ha/yr (**Table 6-10**). In contrast, the total-N export at the WIL1-REF was more variable and tended to decrease over the course of the study. Average export was 9.9 kg/ha/yr with a range of 6.3 kg/ha/yr.

The overall pattern of nitrate-N export was very similar to total-N export because nitrate-N comprised 70 to more than 99% of the estimated total-N load across all sites and years. Mean pre-harvest nitrate-N export ranged from 1.7 kg/ha/yr in WIL1-100% to 13.3 kg/ha/yr in WIL1-FP (**Table 6-10**; **Figure 6-6**). Mean export from the OLYM block ranged from 4.9 to 11.2 kg/ha/yr. Post-harvest, nitrate-N export increased by 1.2 to 18.8 kg/ha/yr. On a percentage basis, increases ranged from 13% in OLYM-100% to 327% in WIL1-100%. Export in the extended period decreased relative to post-harvest in four of the six treatment sites (WIL1-100%, WIL1-FP, OLYM-FP, and OLYM-0%) but changed little in the other two sites where extended export values overlapped with post-harvest values. Nitrate-N export remained higher than pre-harvest levels at all but the WIL1-FP site.

6-4.2.4. Buffer Treatment Effects

The GLMM analysis showed an increase in total-N export of 5.73 ($P = 0.121$), 10.85 ($P = 0.006$), and 15.94 ($P = 0.000$) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments, respectively, and of 6.20 ($P = 0.095$), 5.34 ($P = 0.147$), and 8.49 ($P = 0.026$) kg/ha/yr in the extended period (**Table 6-11**). Significant ($P < 0.05$) treatment effects were present in the FP treatment post-harvest and in the 0% treatment in the post-harvest and extended periods relative to the reference sites, but there were no significant differences in total-N export between the treatments.

The GLMM analysis for nitrate-N export showed changes similar to but slightly less than those seen in the total-N analysis with a relative increase in nitrate-N export of 4.79 ($P = 0.123$), 9.63 ($P = 0.004$), and 14.41 ($P < 0.001$) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments, respectively (**Table 6-11**). In the extended period, the change in export relative to the pre-harvest period was 3.31, 1.54, and 5.11 kg/ha/yr in the 100%, FP, and 0% treatments, respectively. None of the changes in the extended period were significant ($P < 0.05$).

Table 6-9. Comparison of the percentiles of recorded discharges (Q-m) and discharge when nutrient samples were collected (Q-s) across each study site and period.

	WIL1-REF						OLYM-REF					
	Pre-Post				Extended		Pre-Post				Extended	
P-tile	Q-m	Q-s			Q-m	Q-s	Q-m	Q-s			Q-m	Q-s
1	2	2			3	3	<4	<4			<4	4
5	3	4			3	4	<4	<4			<4	22
10	4	4			3	6	<4	8			<4	59
25	5	8			4	12	6	59			6	157
50	7	13			11	19	27	290			32	294
75	12	25			18	32	58	510			78	361
90	20	52			25	48	117	616			135	483
95	25	66			30	51	177	642			195	541
99	39	144			41	62	396	666			359	582

	WIL1-100%						OLYM-100%					
	Pre		Post		Extended		Pre		Post		Extended	
P-tile	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s
1	1	1	1	1	2	3	3	4	3	3	1	4
5	2	2	3	4	2	4	3	11	3	11	1	13
10	2	6	3	5	3	5	5	31	3	35	3	14
25	6	11	6	11	4	11	10	42	3	59	9	25
50	11	16	12	21	14	27	20	52	8	96	17	51
75	21	21	22	33	28	44	48	76	24	152	31	93
90	34	23	32	73	45	62	82	177	52	230	55	174
95	41	24	47	97	56	91	118	273	75	258	73	239
99	67	34	83	137	92	165	186	391	156	289	122	406

	WIL1-FP						OLYM-FP					
	Pre		Post		Extended		Pre		Post		Extended	
P-tile	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s
1	1	1	1	1	2	4	3	3	3	3	4	5
5	2	2	2	2	3	6	3	4	3	5	5	7
10	2	2	3	3	3	9	4	6	4	9	5	10
25	3	4	5	6	5	16	7	10	6	20	7	19
50	6	10	10	13	12	24	13	13	13	52	20	33
75	12	13	15	17	19	28	24	23	21	102	34	69
90	21	21	22	34	26	45	39	27	36	228	50	110
95	26	23	30	45	30	66	54	29	54	372	62	125
99	38	25	47	53	46	82	100	30	118	397	99	181

	WIL1-0%						OLYM-0%					
	Pre		Post		Extended		Pre		Post		Extended	
P-tile	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s	Q-m	Q-s
1	5	6	6	6	4	5	2	2	6	6	3	3
5	6	6	6	7	5	5	3	4	6	6	3	4
10	6	7	7	10	5	6	4	5	7	6	3	6
25	8	13	9	17	6	9	7	10	9	11	4	9
50	16	22	17	32	20	13	13	19	20	28	14	24
75	31	32	28	70	37	43	21	148	31	72	32	62
90	44	47	40	119	52	85	40	314	46	141	48	85
95	55	75	52	154	64	113	55	390	62	168	64	93
99	88	90	87	177	100	178	113	406	118	198	103	114

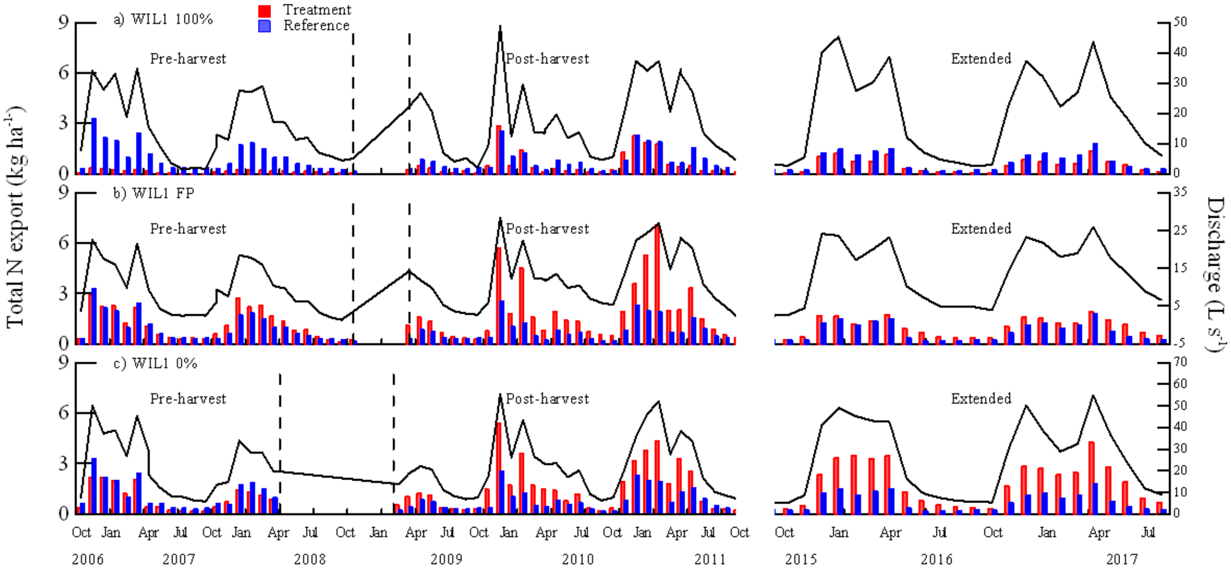

Figure 6-4a. Monthly total-N load in WIL1 treatment (red bars) and reference (blue bars) sites over time and mean monthly discharge (line and right axis). The vertical dashed lines bracket the active harvest period.

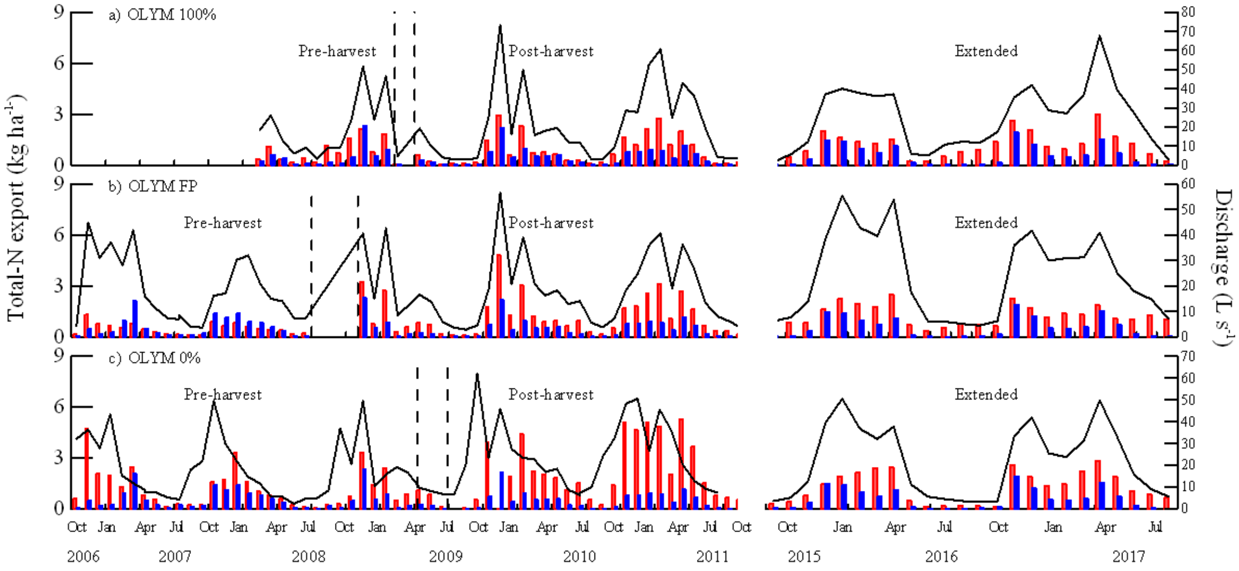

Figure 6-4b. Monthly total-N load in OLYM treatment (red bars) and reference (blue bars) sites over time and mean monthly discharge (line and right axis). The vertical dashed lines bracket the active harvest period.

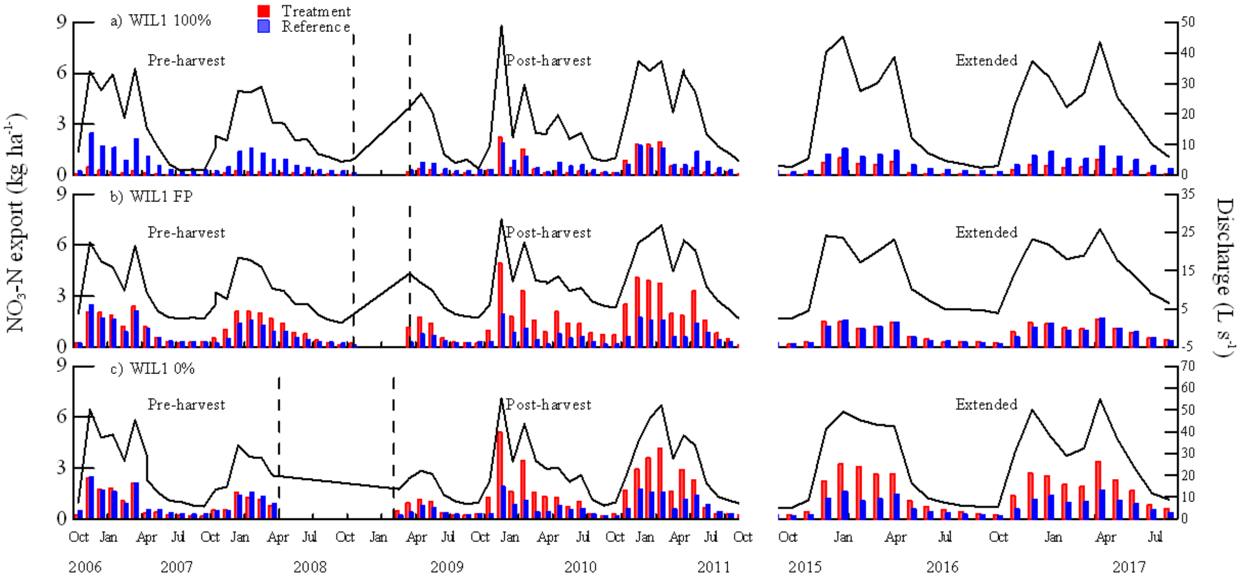

Figure 6-5a. Monthly nitrate-N load in WIL1 treatment (red bars) and reference (blue bars) sites over time and mean monthly discharge (line and right axis). The vertical dashed lines bracket the active harvest period.

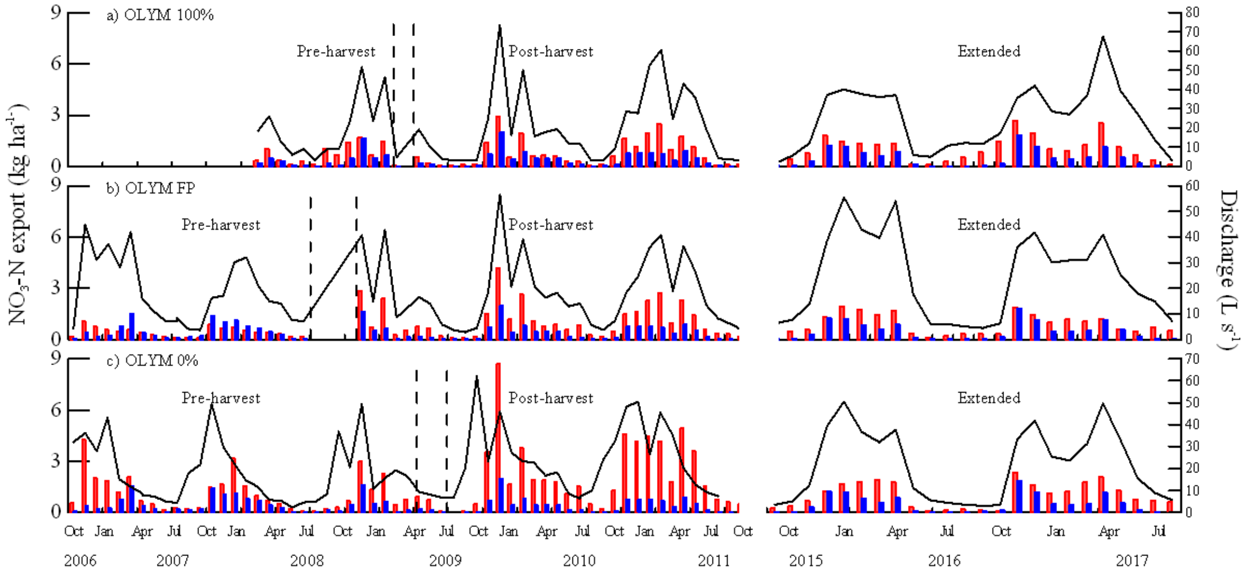

Figure 6-5b. Monthly nitrate-N load in OLYM treatment (red bars) and reference (blue bars) sites over time and mean monthly discharge (line and right axis). The vertical dashed lines bracket the active harvest period.

Table 6-10. Mean annual nutrient export (kg/ha/yr) and annual mean discharge (L/s) for pre-harvest, post-harvest, and extended monitoring periods for each treatment site and the corresponding period in the reference site.

Block	Treatment	Period	Treatment Total-N (kg/ha/yr)	Treatment Nitrate-N (kg/ha/yr)	Treatment Discharge (L/s)	Reference Total-N (kg/ha/yr)	Reference Nitrate-N (kg/ha/yr)	Reference Discharge (L/s)
WIL1	100%	Pre 2	1.8	1.9	15	14.4	12.1	10
		Pre 1	1.7	1.5	16	9.9	8.7	12
		Post 1	6.9	6.1	16	8.9	7.6	8
		Post 2	9.4	8.4	20	11.0	9.3	10
		Extended 1	5.6	4.2	18	8.1	6.6	12
		Extended 2	6.3	4.2	21	8.8	7.5	13
	FP	Pre 2	14.3	13.2	9	14.4	12.1	10
		Pre 1	14.5	13.5	10	9.9	8.7	12
		Post 1	20.8	19.1	11	8.9	7.6	8
		Post 2	28.7	26.0	14	11.0	9.3	10
		Extended 1	10.8	9.0	12	8.1	6.6	12
		Extended 2	12.9	11.0	15	8.8	7.5	13
	0%	Pre 1	7.7	7.7	16	10.1	8.9	10
		Post 1	17.2	16.1	21	8.6	7.3	8
		Post 2	20.6	19.0	24	10.4	8.8	10
		Extended 1	20.0	17.5	23	8.1	6.6	12
		Extended 2	23.2	19.8	27	8.8	7.5	13
OLYM	100%	Pre 1	10.6	9.2	19	5.8	4.6	44
		Post 1	10.0	9.2	20	5.9	5.3	50
		Post 2	12.9	11.9	25	6.0	5.2	52
		Extended 1	11.0	9.6	20	6.3	5.3	55
		Extended 2	17.0	14.9	29	7.5	6.6	58
	FP	Pre 2	5.2	4.5	15	4.9	3.9	61
		Pre 1	5.2	4.9	24	7.3	6.5	55
		Post 1	11.5	10.3	15	5.2	4.1	38
		Post 2	16.5	14.6	19	6.5	5.8	56
		Extended 1	15.2	10.7	24	6.3	5.3	55
		Extended 2	15.5	10.7	24	7.5	6.6	58
	0%	Pre 2	11.9	11.3	16	7.3	6.5	38
		Pre 1	12.0	11.0	16	5.1	4.1	61
		Post 1	30.1	27.4	23	6.4	5.7	55
		Post 2	35.6	32.5	27	5.9	5.1	51
		Extended 1	12.7	10.4	20	6.3	5.3	55
		Extended 2	17.0	13.8	23	7.5	6.6	58

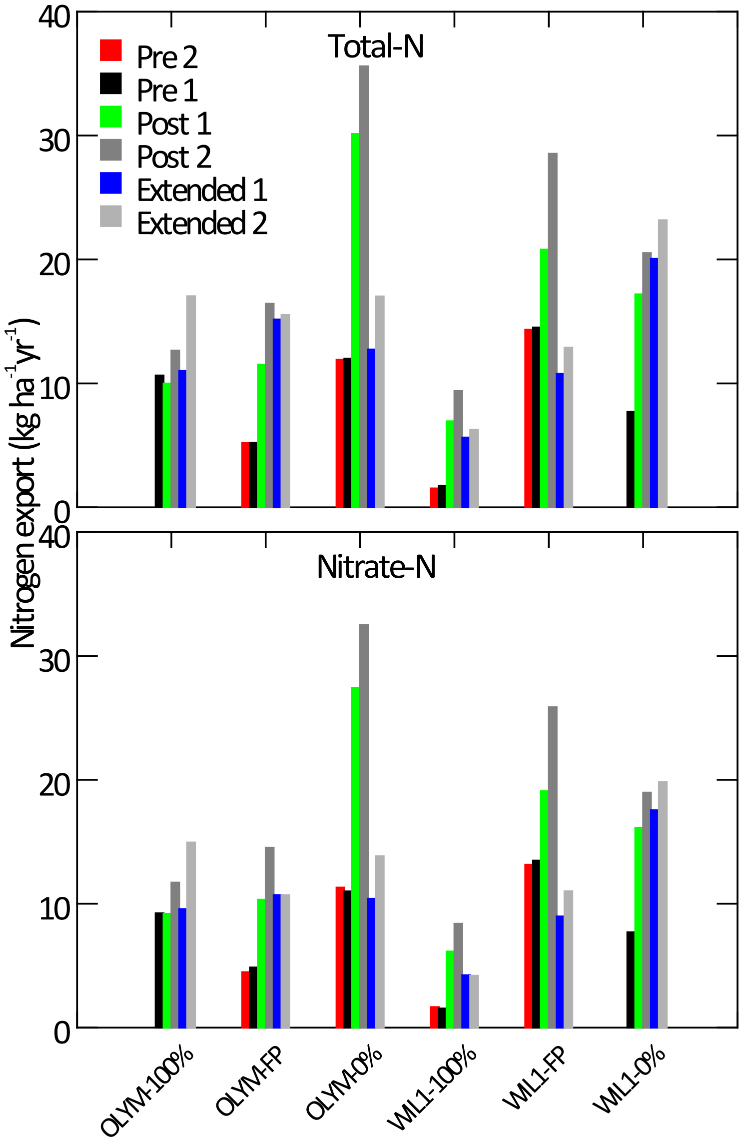

Figure 6-6. Nitrogen export (kg/ha/yr) from buffer treatment sites by harvest year. See **Table 6-5** for harvest year dates for each site.

Table 6-11. Results of pairwise comparisons for nutrient export are shown below. Comparisons were made of pre-harvest vs. post-harvest and extended periods and between buffer treatments for post-harvest and extended periods. Comparisons in **bold** indicate the null hypothesis was rejected at P <0.05. P-values were not adjusted for multiple comparisons.

Comparison	Change	P-value	Lower	Upper
		Total-N		
100%-Pre- vs. Post-	5.73	0.121	-1.6	13.1
100%-Pre- vs. Extended	6.20	0.095	-1.2	13.6
FP-Pre- vs. Post-	**10.85**	**0.006**	**3.5**	**18.2**
FP-Pre- vs. Extended	5.34	0.147	-2.0	12.7
0%-Pre- vs. Post-	**15.94**	**0.000**	**8.6**	**23.3**
0%-Pre- vs. Extended	**8.49**	**0.026**	**1.1**	**15.9**
100% vs. FP Post-	5.12	0.319	-5.3	15.5
100% vs. FP Extended	-0.86	0.866	-11.3	9.6
100% vs. 0% Post-	10.21	0.055	-0.2	20.6
100% vs. 0% Extended	2.29	0.653	-8.1	12.7
FP vs. 0% Post-	5.09	0.321	-5.3	15.5
FP vs. 0% Extended	3.15	0.537	-7.3	13.6
		Nitrate-N		
100%-Pre- vs. Post-	4.79	0.123	-1.4	11.0
100%-Pre- vs. Extended	3.31	0.279	-2.9	9.5
FP-Pre- vs. Post-	**9.63**	**0.004**	**3.4**	**15.8**
FP-Pre- vs. Extended	1.54	0.611	-4.7	7.7
0%-Pre- vs. Post-	**14.41**	**<0.001**	**8.2**	**20.6**
0%-Pre- vs. Extended	5.11	0.101	-1.1	11.3
100% vs. FP Post-	4.84	0.264	-3.9	13.6
100% vs. FP Extended	-1.77	0.678	-10.5	7.0
100% vs. 0% Post-	**9.26**	**0.033**	**0.9**	**18.4**
100% vs. 0% Extended	1.80	0.674	-7.0	10.6
FP vs. 0% Post-	4.77	0.270	-4.0	13.5
FP vs. 0% Extended	3.57	0.407	-5.2	12.3

6-5. DISCUSSION

We expected that nitrogen concentration and export would increase in the immediate post-harvest period, then decrease as vegetative uptake of nitrogen increased with new vegetation growth and as water yield declined toward pre-harvest levels. Pre-harvest total-N and nitrate-N concentrations were quite variable but well within the range of values reported elsewhere in managed watersheds west of the Cascades in Washington (Edmonds *et al.* 1995; Murray *et al.* 2000; Liles 2005; Taylor 2008), Oregon (Brown *et al.* 1973; Harr and Fredriksen 1988; Cairns and Lajtha 2005; Meininger 2011), and British Columbia (Feller and Kimmins 1984).

Nitrate-N concentration and export can be influenced by atmospheric deposition (Feller 2005) and the proportion of the watershed in red alder (*Alnus rubra*) or mixed hardwood-conifer forests (Wigington Jr *et al.* 1998; Compton *et al.* 2003). However, it is unlikely that atmospheric deposition was a factor here. Mean annual atmospheric nitrogen deposition measured at the Hoh River Ranger Station (WA14) and near La Grande, Washington (WA21) was uniformly low, averaging 1.19 and 0.96 kg/ha/yr, respectively, from 2006 to 2017 (NADP 2018). This is much less than is typical for eastern states or downwind of urban or industrial centers and lower than that measured at any of our eight sites. N export varied enough among our sites even pre-harvest that atmospheric deposition is unlikely to be a major factor in explaining the variability.

We did not have basin-wide estimates of vegetation composition; however, riparian vegetation in our study sites was dominated by conifers before harvest, ranging from 78 to 100% of total basal area within the 50-ft (15.2-m) riparian zone (McIntyre *et al.* 2018, Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers*). Although nitrate-N concentration (Wigington Jr *et al.* 1998; Compton *et al.* 2003) and nitrate-N export (Compton *et al.* 2003) from western Oregon forests were observed to increase with the proportion of the watershed or riparian stand in alder or mixed hardwood-conifer stands, we found no correlation between pre-harvest nitrate-N export or the relative increase in nitrate-N export and the proportion of the riparian stand in hardwoods ($P > 0.05$). In fact, our lowest and highest N concentration and export came from adjacent sites both with less than 1% red alder cover (by basal area) in the riparian stands. The lack of a significant relationship may reflect that our sites were predominantly conifer (our sites would rank in the bottom quartile of broadleaf forest cover in Compton and colleagues (2003)), our relatively small number of sites, or the lack of watershed-wide forest cover data.

Post-harvest values of nitrate-N concentration and export increased in all of our sites, as expected, but remained as variable as the pre-harvest values. Other studies have also reported increases in nitrate-N concentration and export following timber harvest (e.g., Brown *et al.* 1973; Feller and Kimmins 1984; Harr and Fredriksen 1988; Dahlgren 1998; Gravelle *et al.* 2009; Clinton 2011; Boggs *et al.* 2016; Schelker *et al.* 2016). We measured an increase in nitrate-N export of 1.2 to 18.8 kg/ha/yr, which is comparable to the 4.9 to more than 15 kg/ha/yr increase that Brown and colleagues (1973) measured from their conifer-dominated Needle Branch site in the Oregon Coast range after harvest. The increase in nitrate-N concentration and export from our sites likely resulted from a decrease in nutrient uptake following tree removal and an increase in discharge after harvest, particularly in the FP and 0% treatments (McIntyre *et al.* 2018, Chapter 8 – *Discharge*). Dahlgren (1998) also attributed an increase in nitrate-N export to an increase in higher flows and leaching following a clearcut harvest of a Douglas fir/redwood (*Pseudotsuga menziesii*/*Sequoia sempervirens*) forest in northern California.

By the extended period, approximately 7 and 8 years after harvest, we observed a divergence in the direction of nitrate-N export, with four sites decreasing substantially and two remaining near post-harvest values. Only one of our sites, however, had recovered to pre-harvest nitrate-N export rates. Previous studies show that the duration of recovery to pre-harvest nitrate concentrations and export rates is variable, with recovery occurring two to three years after harvest (Feller and Kimmins 1984) or in five or more years (Brown *et al.* 1973; Fredriksen *et al.* 1975; Martin 1986; Hornbeck 1987; Dahlgren 1998). Harr and Fredriksen (1988) measured elevated nitrate concentrations during high flow events 10 years after harvest. In a revisit of the

Alsea Watershed Study, Hale (2007) found that nitrate export increased over rates previously measured at their Needle Branch site that was clearcut 40 years earlier.

The lag in recovery of nitrate-N exports to pre-harvest rates was likely due to changes in discharge that persisted in the Hard Rock Study sites through the extended period. We measured an increase in annual runoff from all of the treatment sites over the 8-year period, with the largest increases in annual water yield from the FP and 0% treatments (see Chapter 5 – *Stream Discharge, Turbidity, and Suspended Sediment Export* in this report). In addition, the Olympic block yielded more water for a given treatment than the Willapa 1 block sites. While the effects of harvest on discharge are diminishing, flows have yet to recover to pre-harvest values and so may explain the slow recovery in nitrate-N export from most of our sites.

Differences in vegetation regrowth may also account for delays in recovery of nitrate-N export to pre-harvest rates. We measured low rates of tree regrowth five and eight years after harvest in all of our Hard Rock Study sites (see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report). Fredriksen and colleagues (1975) found that nitrate export recovered to pre-harvest rates more slowly in their H.J. Andrews Experimental Forest sites where vegetation regrowth was slower relative to their sites on the Oregon coast. While rapid regrowth of vegetation has the potential to increase uptake, colonization with fast-growing red alder may result in higher rates of nitrate export, which is what Hale (2007) found at the Alsea Watershed Study Needle Branch site decades after harvest. In the Hard Rock Study, conifer species comprised the majority of plots surveyed for regeneration of seedlings and saplings (see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report). However, we surveyed tree regeneration 12.5 and 37.5 ft from the stream and did not document regeneration immediately adjacent to the stream where red alder saplings most often established.

Logging slash resulting from harvest and subsequent windthrow of riparian trees may affect recovery of nitrate concentration or export to pre-harvest values. Other studies have attributed an increase in nutrient concentration to slash decomposition (Fredriksen *et al.* 1975; Dahlgren 1998). Harr and Fredriksen (1988) reported a greater increase in nitrate concentrations and a longer duration of recovery in sites where logging slash was left to decompose naturally compared with sites where slash was burned. Slash was left to decompose naturally in the Hard Rock Study sites. While we measured an increase in logging slash and windthrow with treatment severity (see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report), we found no consistent response in the recovery of nitrate concentration and export between those treatments.

Sources of site-scale variability may include the laboratory analyses, discharge estimates, and models used to estimate N exports. It is unlikely that the laboratory analyses, conducted at an accredited lab with stringent quality assurance measures, played a role. Discharge estimates were somewhat higher than expected for all sites compared with PRISM precipitation projections and sources of error may include improper installation of flumes, shifting of flumes over time, malfunctioning pressure transducers, incorrect stage-discharge equations, or incorrect watershed area estimate. However, we adequately sited and checked the flumes annually to ensure they remained level, we compared pressure transducer readings with a staff gauge to track any deviation over time, and we used standard stage-discharge equations for the flumes and developed equations with high r^2 values for the OLYM-REF and OLYM-100% sites based on a broad range of estimated discharges. We used GIS-based digital elevation models to estimate watershed area so it is possible that the areas were consistently less than the actual area, which would bias estimates of export per hectare high, but we have no means of checking this at this time.

The models used to estimate N export were less robust in the extended period. It is plausible that after harvest, any two-year period represents a rapidly changing relationship between concentration and discharge and that by the extended period the relationship was noticeably weaker. In addition, the WIL-REF site was not stationary over time with respect to N concentration or export, which affected the analysis of buffer treatment effect. While this may have been less consequential had we sampled more than two blocks, it is clear that by seven years after harvest there were diverging effects of N concentration and export not related to the buffer treatments.

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CHAPTER 7 - STREAM CHANNEL CHARACTERISTICS

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7-1. ABSTRACT

Timber harvest along and near streams can alter wood recruitment and loading, stream flow and sediment transport, which can result in changes to stream channel characteristics. We compared the response of headwater stream channel characteristics to clearcut timber harvest in a long-term (2006–2016), spatially blocked and replicated Before-After Control-Impact (BACI) study in western Washington. We included three alternative riparian buffer treatments and reference sites that were not harvested. Buffer treatment basins were clearcut outside of a riparian management zone (RMZ) and received one of three riparian buffer treatments: 100% treatment (two-sided 50-ft width riparian buffer along the entire length of the Type Np stream network), Forest Practices (FP) treatment (two-sided 50-ft width riparian buffer along at least 50% of the Type Np stream network, according to current Forest Practices Rules), and 0% treatment (harvested to the stream edge with no riparian buffer). We evaluated the response of headwater streams to timber harvest in 17 study sites using common stream channel metrics, including stream wetted width, stream substrate, and channel unit composition and characteristics. Sampling was conducted during the summer low flow period in two years preceding harvest (Pre 3 & 2), the two years immediately following harvest (Post 1 and Post 2), and five (Post 5) and eight (Post 8) years after harvest.

Compared to the pre-harvest period after controlling for temporal changes in the reference, the change in stream wetted width averaged 0.3 m less in the 0% treatment in Post 1 and Post 2, a difference that was sustained in Post 5, and was 0.4 m less in Post 8. The change in bankfull width was also less in the 0% treatment compared with the reference, averaging 0.4 m less in Post 1 and Post 2, 0.6 m less in Post 5, and 0.7 m less in Post 8. We found evidence of a post-harvest increase in the dominance of fine sediment in the stream substrate of some buffer treatments. In Post 1 and Post 2, we estimated a 143% increase in the mean odds for the proportion of stream substrate dominated by fines and sand in the 0% treatment compared to the reference, a difference that increased to 154% in Post 8. An increase of 371% in the mean odds was evident in the FP treatment in Post 8, but not other sample years. We also noted a 49%, 34%, and 39% decrease in the mean odds for the proportion of channel rise attributed to steps in the 0% treatment in Post 1 and Post 2, Post 5 and Post 8, respectively. Changes in wetted and bankfull widths, the proportion of sand and fines-dominated stream, and proportion of channel rise attributed to steps may be attributable, at least in part, to the post-harvest increase in in-channel wood recruitment and loading that we observed in buffer treatments, which was especially evident along clearcut stream reaches without a riparian buffer. However, these changes did not result in any significant differences among treatments for the densities or characteristics of stream channel units, including cascades, riffles, pools or steps. Changes to the quantity and quality of certain instream characteristics may have consequences for instream biota, including macroinvertebrates and amphibians.

7-2. INTRODUCTION

Timber harvest activities have been shown to impact wood regimes (Benda *et al.* 2003; Boyer *et al.* 2003; Hassan *et al.* 2005b), stream flow (Lewis *et al.* 2001; Kuraś *et al.* 2012), and sediment transport (Gomi *et al.* 2005; Kaufmann *et al.* 2009), all of which have the potential to impact stream channel characteristics. Many studies have recognized in-channel wood as a primary determinant of channel structure (Harmon *et al.* 1986; Montgomery and Buffington 1998; Brookshire and Dwire 2003; Hoover *et al.* 2006). Wood pieces trap and store sediment and organic material (Bilby and Ward 1991; Hassan *et al.* 2005a), create steps (Gomi *et al.* 2002) and pool habitat (Andrus *et al.* 1988; Beechie and Sibley 1997), stabilize streambeds and banks, and dissipate energy (Curran and Wohl 2003). Timber management activities that alter riparian stand conditions affect short-term wood loading and longer-term recruitment potential (Bilby and Ward 1991; Ralph *et al.* 1994; Grizzel and Wolff 1998; Jackson and Sturm 2002). Loss of wood input potential as a result of timber harvest can lead to channel incision in headwater streams and mobilization of large amounts of sediment (Montgomery *et al.* 2003). The changes in wood loading associated with timber harvest have been linked to changes in pool characteristics (Lisle 1986; Montgomery *et al.* 1995; Jackson *et al.* 2001) and substrate composition (Dupuis and Steventon 1999; Jackson *et al.* 2001).

Forest management can also affect headwater stream hydrology (Moore and Wondzell 2005). Logging has been shown to effect annual water yield and the timing and magnitude of flow (Jones and Post 2004; Brown *et al.* 2005; Winkler *et al.* 2017). The removal of forest canopy reduces interception and evapotranspiration, altering the magnitude and timing of water delivery to the soil (Lewis *et al.* 2001; Keim and Skaugset 2003; Johnson *et al.* 2007). Forest roads have the potential to extend the surface channel network and intercept subsurface flow, increasing the speed and volume of water entering the channel (Wemple *et al.* 1996; Wemple and Jones 2003). While small drainages typically lack the flow capacity to move large wood (Gurnell 2003), increased debris flows as a result of timber harvest can increase wood transport out of headwater channels (Nakamura *et al.* 2000; May 2001).

Timber harvest also has the potential to influence sediment regimes (Bathurst and Iroumé 2014). Sediment transport is a function of sediment supply and transport capacity (Schumm 1971), both of which can be affected by forest practices (Gomi *et al.* 2005). Forest practices have been shown to increase suspended sediment loads and export (Reiter *et al.* 2009; Klein *et al.* 2012), which can have deleterious effects on stream-associated amphibians (Wilkins and Peterson 2000; Stoddard and Hayes 2005).

The effect of forest practices on the physical characteristics of headwater streams has been well studied. Like our study, many of these have focused on the response of aquatic or stream-associated taxa, including macroinvertebrates and amphibians (e.g., Murphy and Hall 1981; Carlson *et al.* 1990; Bull and Carter 1996; O'Connell *et al.* 2000; Jackson *et al.* 2007). Many of the channel characteristics we measured were intended to help us refine our description of harvest effects on habitat availability and quality for stream-associated amphibians. While most studies have focused on large wood (≥10 cm [4 in] diameter), we included the response of small wood (<10 cm [4 in] diameter), which is frequently abundant in smaller channels (Bilby and Ward 1991; Maxa 2009) and influences channel morphology (Gomi *et al.* 2001; Maxa 2009). A CMER Work Plan Resource Objective, derived from the Schedule L-1, is to provide complex in- and near-stream habitat by recruiting wood and litter.

There is scientific uncertainty surrounding the potential for the Forest Practices Habitat Conservation Plan (FPHCP) buffer strategy for Type N (non-fish-bearing) streams to provide complex instream habitat, especially for Forest Practices (FP)-designated amphibians. We used a basin-scale approach to compare changes in the response of stream channel characteristics to a range of Type Np buffering strategies that varied in the proportion of stream length buffered.

7-3. METHODS

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 stream channel characteristics (e.g., wetted and bankfull widths, stream depth, dry length, channel units (i.e., pool, riffle, cascade and step density and characteristics), and substrate (i.e., proportion dominated by fines and sand)) among reference and treatment sites in a BACI-designed study (see Chapter 2–*Study Design* in this report). We compared conditions in Type Np reference basins (n = 6) to the response in basins with clearcut harvest and one of three riparian buffer treatments in the riparian management zone (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). Our objective was to determine the magnitude and duration of change in stream channel characteristics following harvest.

Study sites comprised first-, second- or third-order (12 to 54 ha (29 to 133 ac)) drainage basins with bankfull widths averaging less than 1 to approximately 3 m and channel gradients averaging 12% to 32% (7 to 18 degrees). According to Montgomery and Buffington (1998), streams in Pacific Northwest headwater basins are typically colluvial channel reaches exhibiting weak or intermittent sediment transport near the channel head (**Figure 7-1a**) and leading to one or more of the following channel-reach morphologies downstream: bedrock, cascade, or forced and free-formed step-pool. Bedrock reaches generally occur on steeper slopes than alluvial cascade and step-pool reaches (Montgomery *et al.* 1996) and, due to high transport capacity, lack an alluvial bed except when alluvial material is temporarily stored in scour holes or behind flow obstructions (Montgomery and Buffington 1997; **Figure 7-1b**). Cascade reaches occur on steep slopes, have high rates of energy dissipation, are characterized by longitudinally and laterally disorganized bed material typically consisting of cobbles and boulders, are confined by valley walls, and have a large particle size relative to flow depth (Montgomery and Buffington 1998; **Figure 7-1c**). Step-pool reaches are created by discrete channel-spanning accumulations of substrates (typically boulders and cobbles) that separate pools containing finer material, have steep gradients, small width-to-depth ratios, and pronounced confinement by valley walls (Montgomery and Buffington 1997; **Figure 7-1d**). Forced step-pool reaches are created when wood forms most of the channel-spanning steps (**Figure 7-1e, f**), altering bed morphology and creating a step-pool reach that extends beyond the range of conditions characteristic of non-wood-formed (i.e., free-formed) step-pool channel-reach morphology (Montgomery and Buffington 1997). Jackson and colleagues (2001) defined a step-riffle morphology as a sub-type of colluvial channel within first- and second-order streams located in the Coast Ranges of western Washington, noting that riffles and steps accounted for 64% and 15% of channel length, respectively. The authors noted in their study that streams featured a relatively high frequency of steps and that pools were rare because the streams lacked the fluvial power to form plunge pools.

We expected that, given the overlap in study areas, our headwater study streams would be morphologically similar to those described by both Jackson and colleagues (2001) and Jackson and Sturm (2002), and include the colluvial, step-pool and forced step-pool morphologies described by Montgomery and Buffington (1997).

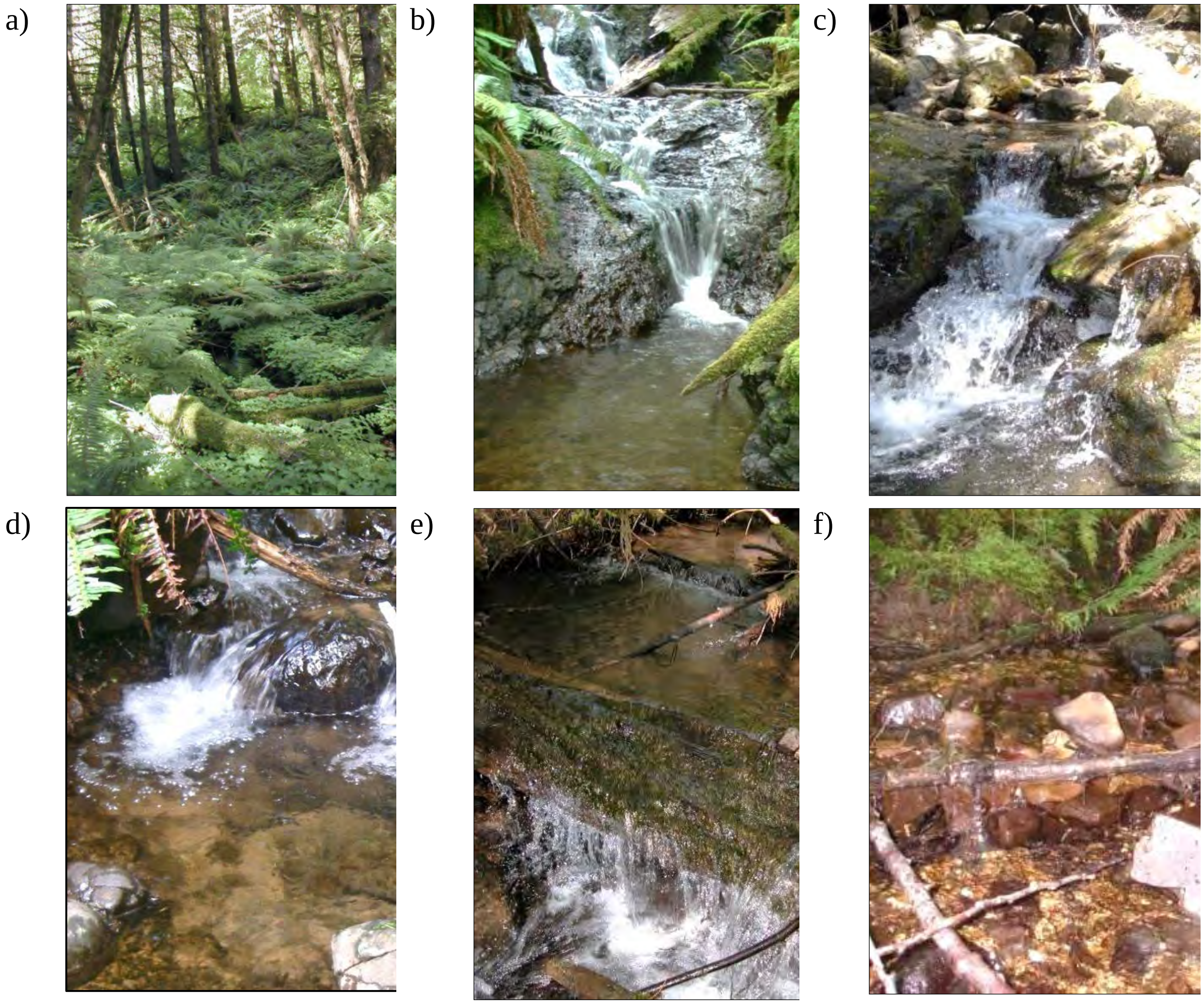

Figure 7-1. Study stream reaches in the Hard Rock Study exhibiting (a) colluvial, (b) bedrock, (c) cascade, and (d) step-pool channel-reach morphologies; and (e) forced step-pool channel-reach morphology composed of large wood (LW; i.e., >10 cm diameter) and (f) a step composed of small wood (SW; i.e., ≤10 cm diameter).

7-3.1. DATA COLLECTION

We conducted data collection for stream channel characteristics in two-person teams in two pre-treatment years (Pre 3 and Pre 2), the two years immediately following harvest (Post 1 and Post 2), and five and eight years following harvest (Post 5 and Post 8, respectively). Crew members were extensively trained and calibrated, and teams were reassigned daily to reduce bias. For most metrics, sampling occurred from April through June of each year, with some exceptions[1]. We delineated 10-m sample intervals along the mainstem channel starting at the fish end point (i.e., where the Type N joins the Type F (fish-bearing) Waters, hereafter F/N break) and upstream to but not including the uppermost points of perennial flow (i.e., perennial initiation point; PIP). The PIP was sampled as a single 17.1-m (56-ft) long interval, consistent with the riparian buffer required for PIPs under Forest Practices Rules (WFPB 2001). We sampled 20 contiguous 10-m (33-ft) intervals (i.e., 200 m (656 ft)) of stream immediately upstream of the F/N break. We sampled additional 10-m intervals based on the total length of the mainstem channel, according to the following criteria:

1) The entire contiguous length of the remaining mainstem channel for the smallest sites (<300 m mainstem length; n = 1),
2) A minimum of 50% of the remaining mainstem channel length for moderately sized sites (300–800 m mainstem length; n = 10), or
3) A minimum of 25% of the remaining mainstem channel length for the largest sites (>800 m mainstem length; n = 6).

The systematic sample for (2) and (3) occurred above the 200-m contiguously sampled reach and were evenly distributed throughout the remainder of the mainstem channel. Besides the 10-m intervals sampled, we characterized two intervals centered on the PIP. The interval located downstream of the PIP was 17.1 m (56 ft) in length. The length of the interval located upstream of the PIP was either 17.1 m or equal to the length of the non-perennial Type Ns Water[2] located between the PIP and the channel head, whichever was shorter. The total number of intervals sampled at a site in a year ranged from 21 to 49.

We measured stream wetted width, stream depth, bankfull width (**Table 7-1**) and dominant stream substrate (**Table 7-2**) as a point measure located at the beginning of each sample interval. If the stream was not accessible at the beginning of the sample interval (e.g., due to instream wood), measurements were taken at the nearest location where the stream could be accessed. We defined inorganic substrates according to a modified Wentworth classification (Wentworth 1922) and visually evaluated the dominant substrate as that most commonly encountered along a line perpendicular to the stream axis and within the wetted stream channel. If the stream was dry, we

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[1] In Post 5, two 10-m intervals each in the WIL3-100% and CASC-REF sites were sampled on 10 July 2013 and 6 August 2016, respectively, and one interval in the CASC-0% site was sampled on 5 August 2013. In Post 8, one interval in the OLYM-FP site was sampled on 22 August 2016.

[2] Seasonal, non-fish habitat streams in which surface flow is not present for at least some portion of a year of normal rainfall and that is not located downstream from any perennial stream reach. Ns Waters must be physically connected to a downstream perennial channel by an above-ground channel system (Washington Administrative Code [WAC] 222-16-030).

evaluated dominant substrate within the bankfull width. We measured the stream channel gradient between the start and end of each sample interval for calculations of the stream channel rise that could be attributed to steps.

We also recorded the occurrence of, and data associated with, stream channel units, or morphologically distinct areas that extend up to several channel widths in length (Montgomery and Buffington 1998). We used physical parameters such as channel slope, depth, bed material, roughness and flow velocity to distinguish channel units (MacDonald *et al.* 1991). Classification and size of channel units are flow dependent, changing in area and volume with fluctuating discharge (MacDonald *et al.* 1991). To make our measurements comparable across sites and years, and to reduce variability, we carried out our unit surveys during a similar time each year and categorized units into only four types: pool, riffle, cascade and step (**Table 7-3**). In the post-harvest period, we also recorded the lengths of stream obstructed by wood where stream channel unit type could not be determined (see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report).

Pool length (0.1 m) was measured along the longest linear length. Pool maximum depth was recorded at the deepest location (cm) as the distance from the existing water surface to the channel bottom (modified from Platts *et al.* 1983). We recorded the step key piece (i.e., the piece responsible for the formation of the step) by type (**Table 7-4**), the diameter (cm) of the key piece for wood pieces, and the step height (cm) from the channel bed to the top of the step.

In Pre 3, Post 2, Post 7 and Post 8, we censused the stream length that was dry during the summer low flow period, concurrent with amphibian sampling and including the mainstem channel and all tributaries. We used these data to look at changes in the dry stream length between periods.

Table 7-1. Stream channel metrics measured as a part of the Hard Rock Study (modified from Platts *et al.* 1983). Unit specifies the unit of measure. Precision specifies the resolution to which a value was measured.

Variable	Definition	Unit	Precision
Wetted width	Width of stream from wetted edge to wetted edge, measured perpendicular to the stream axis and along the existing water surface. Where the channel is braided, wetted width is the sum of the individual widths for each braid. [modified from Platts *et al.* 1983]	m	0.1
Stream depth	Vertical height of the water column from the existing water surface level to the channel bottom, measured at the center of the wetted channel. Where the channel is braided, depth is measured for the dominant thread. [modified from Platts *et al.* 1983]	cm	1
Bankfull width	The measurement of the lateral extent of the water surface elevation perpendicular to the channel at the bankfull depth (i.e., the vertical distance between the channel bed and the estimated water surface elevation required to completely fill the channel to a point above which water would enter the floodplain or intersect a terrace or hillslope). Where the channel is braided, bankfull width is the width as measured across all braids. [WAC 222-16-010]	m	0.1

Table 7-2. Substrate types and definitions for evaluation of dominant inorganic substrate in the Hard Rock Study, based on a modified Wentworth classification (Wentworth 1922).

Substrate Type	Definition
Boulder and bedrock	>256 mm (10 in) in diameter
Cobble and gravel	2 - 256 mm (2.5 - 10 in) in diameter
Fines and sand	<2 mm (0.1 in) in diameter

Table 7-3. Stream channel unit types and definitions used in the Hard Rock Study. Channel unit types were classified according to the current hydrology of the stream at the time of sampling.

Channel Unit	Definition
Pool	Slow water unit: deeper with slow, non-turbulent flow (except in the case of plunge pool where water flow is often turbulent). Pool depth is typically greater than 10% of the bankfull channel width.
Riffle	Fast water unit: shallow with rapid and turbulent water, includes high and low gradient riffles and runs with a gradient <6°. Dominant substrates include cobbles and gravels, and sometimes bedrock.
Cascade	Fast water unit: shallow with rapid and turbulent water, includes chutes and some waterfalls with a gradient >6°. Dominant substrates include bedrock, boulders and larger cobbles.
Step	Formed by an obstruction (key piece, e.g., boulder, bedrock, log, culvert, or other structure) in the stream channel and includes an accumulation of organic debris or sediment; step height is typically greater than 10 cm.

Table 7-4. Step key piece types and definitions used in the Hard Rock Study.

Step Key Piece	Definition
Boulder or Bedrock	Rock >256 mm (10 in) in diameter
Cobble	Rock >64 - 256 mm (2.5 - 10 in) in diameter
Root	Root from a living shrub or tree
Wood	Any dead wood
Other	Other; may include clay or other underlying geology

Our objective was to evaluate the response to treatment at the level of the entire Type N basin, so we summarized data for individual response variables into one value for each study site and year. For wetted width, stream depth and bankfull width, pool length and depth, and step height, we calculated means for each study site and year. For substrate, we calculated the proportion of sample intervals dominated by fines and sand by site and year. We calculated the proportion of steps keyed by wood as the number of instances that the step key piece was wood divided by the total number of steps by site and year. We averaged the diameters of wood key pieces by site and year. We calculated the channel rise attributed to steps from the interval slope length (typically 10 m, with some exceptions) and the channel gradient (McIntyre *et al.* 2018, Chapter 11 – *Stream Channel Characteristics*), again taking the mean by site and year. To calculate the proportion of the stream that was dry at the time of sampling, we divided the dry stream length by the stream total length.

To compare the frequency of each channel unit type, we calculated an adjusted sample length by subtracting the length obstructed by wood from the target sample length for each site and year.

We calculated unit density as the channel unit count by site and year, with sample length as an offset term in the analysis to account for the fact that survey lengths were not constant.

Some research has focused on static measures, such as residual water depth (Bathurst 1981), that are independent of current discharge rates and remain constant regardless of the timing of measurement. We included some static measures in our evaluation of stream conditions; however, we also included non-static measures, such as wetted water width, which are flow dependent. Including non-static measures allowed us to evaluate current conditions during a time identified as biologically important for stream-associated amphibians. We relied on our statistical models to account for environmental variability, with a year term that accounts for inter-annual variability and a blocking term that grouped sites geographically to increase precision. Consistent changes observed across all treatments, including the reference, would be an indication of annual variability (e.g., changes in annual precipitation) that is independent of a treatment effect.

7-3.2. STATISTICAL ANALYSIS

Analyses evaluated the generalized null hypothesis:

$$\Delta T_{REF} = \Delta T_{100\%} = \Delta T_{FP} = \Delta T_{0\%} \qquad \textbf{(Eq.7-1)}$$

where: ΔT_{REF} is the change (post-harvest − pre-harvest) in the reference, and $\Delta T_{100\%}$, $\Delta\ T_{FP}$, and $\Delta T_{0\%}$ are the changes in the 100%, FP and 0% treatments, respectively. We present the between-treatment comparison of change for the pre-treatment mean (Pre 3 and Pre 2) and Post 1 and Post 2 mean, pre-treatment and Post 5, and pre-treatment and Post 8.

We used a generalized linear mixed effects model to evaluate this pre- versus post-harvest hypothesis. We utilized the Mixed Procedure in SAS (SAS Institute Inc. 2013) for all analyses with the exception of the analysis of count data (i.e., channel unit density [#/m]) for which we utilized the GLIMMIX Procedure. In both models, block and site were random effects and the fixed effects were year, treatment, and the treatment × year interaction. For count data, we used the natural logarithm of the survey length for each unit as an offset term in the model to account for the fact that survey lengths were not constant. This allowed for estimation of treatment effects on channel unit density, rather than count per se. The blocking term groups sites geographically to increase precision. The year term accounts 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 × time interaction, and site × treatment interaction. The latter two terms correspond to the variation in the year effect by site, and the variation in treatment effect by site. Other sources of variation are also included in the experimental error.

We evaluated the null hypotheses with a Wald-type test using linear contrasts of the model fixed effects. We constructed the contrasts to test the difference in mean response for pre- and post-buffer treatment periods, where period corresponded to all years in either the pre- or post-harvest condition. If the period × treatment contrast had a P-value ≤ 0.1, we examined pairwise contrasts to test for differences among the six combinations of references and treatments: REF vs. 0%, REF vs. FP, REF vs. 100%, 0% vs. FP, 0% vs. 100%, and FP vs. 100%. If the period × treatment contrast was greater than 0.1, we provided estimates of pairwise contrasts, but did not report test results for these terms. We present means for the pairwise contrasts and 95% confidence intervals from the output for the Mixed Procedure or GLIMMIX Procedure for count data. The uneven distribution of treatments among blocks required utilizing the Kenward-Roger method (Kenward and Roger 1997) for estimating the denominator degrees of freedom in the channel unit density analyses (GLIMMIX Procedure).

A logit transformation was taken on all proportion response data (i.e., fines and sand, steps keyed by wood, dry channel length, and channel rise attributed to steps) prior to analysis, to constrain estimates to the (0,1) interval. Due to the presence of zeroes and ones in the response data, a constant value of 0.02 was added to the numerator and denominator prior to performing the logit transformation (Warton and Hui 2010). The value of 0.02 was used for all transformations and represents the smallest non-zero value among all ratio or proportion response data.

For results reported on the natural log (ln) scale (from GLIMMIX Procedure in SAS) exponentiating the difference in the natural logs of post- and pre-harvest values gives an estimate of the relative change in the variable on its original scale. Therefore, a back-transformed result equal to 1 equates to no change in the mean pre- and post-harvest estimates, a value between 0 and 1 equates to a result post-harvest that is less than the mean in the pre-harvest period, and a value greater than 1 equates to a result post-harvest that is more than the mean 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 report back-transformed relative change on the original scale in the results.

For results reported on the logit scale, exponentiating the difference of post-and pre-harvest values gives an estimate of the odds-ratio, a relative comparison of change. For example, if the proportion of fines and sand was 0.1 in the pre-harvest period and 0.2 post-harvest, the odds-ratio would be (0.2/0.8) / (0.1/0.9) = 2.25, or the odds of fines during post-harvest is 2.25 times that of the pre-harvest period. Tabular summaries of contrasts for analyses performed on the logit scale are given in terms of odds-ratios, while in text we typically report these as relative percentage changes. In the above example, an estimated odds ratio of 2.25 would be reported as a 125% increase in post-harvest odds relative to the pre-harvest odds. A back-transformed result greater than 1 indicates an increase in the proportion, while a value less than 1 indicates a decrease in the proportion. Tables and figures report the back-transformed estimates of odds-ratios for all proportion response data.

We note that all statistical inference for transformed response variables in this chapter takes place on the transformed scale. We provided back-transformations to aid with understanding the magnitude of estimates on the response scale. Further, we recognize that with our number of statistical comparisons (n = 15), and especially with an alpha of 0.1, we could expect to have several “statistically significant” results based on chance alone. The mixed-model procedure does not provide an estimate of the power of the tests or of the likelihood of a Type II error (failing to reject the null hypothesis when it is not true), but autocorrelation within sites, limited replication of treatments, and missing treatments within blocks all contributed to a small effective sample size. We caution the reader to interpret our results with this in mind.

7-4. RESULTS

7-4.1. WETTED WIDTH

Mean annual wetted width ranged from 0.4 to 2.1 m in the pre-harvest period, 0.4 to 2.1 m in Post 1 and Post 2, 1.0 to 2.7 m in Post 5, and 0.6 to 1.8 m Post 8 (**Figure 7-2**). We found evidence that treatments differed in the magnitude of change over time (P = 0.02; **Table 7-5**; **Figure 7-3**; **Table 7-6**). In Post 1 and Post 2, we estimated the between-treatment comparison of wetted width for the 0% treatment and reference to be a 0.3 m (P = 0.01) decrease compared to the pre-harvest period, after controlling for temporal changes in the references. Note that mean wetted width in all other treatments, including the reference, increased by 0.4 m in this same period (**Table 7-5**). We also estimated that mean width was 0.3 m less in the 0% treatment than in the 100% (P = 0.01) and FP (P = 0.03) treatments, after adjusting for pre-harvest differences among the treatment sites. In Post 5, we estimated a 0.3 m decrease in mean width in the 100% (P = 0.05) and 0% (P = 0.06) treatments, compared to the pre-harvest period, after controlling for temporal changes in the references. We also estimated a 0.2 m decrease in the FP treatment relative to the reference (P = 0.11), however, the 95% confidence interval for this comparison included 0, indicating uncertainty in the direction of the treatment effect. In Post 8, we estimated a 0.4 m (P = 0.01) decrease in mean width in the 0% treatment compared to the pre-harvest period. The estimated mean width in the 0% treatment was also 0.4 m (P = 0.01) less than the estimate for the FP treatment, after adjusting for pre-harvest differences among the treatment sites.

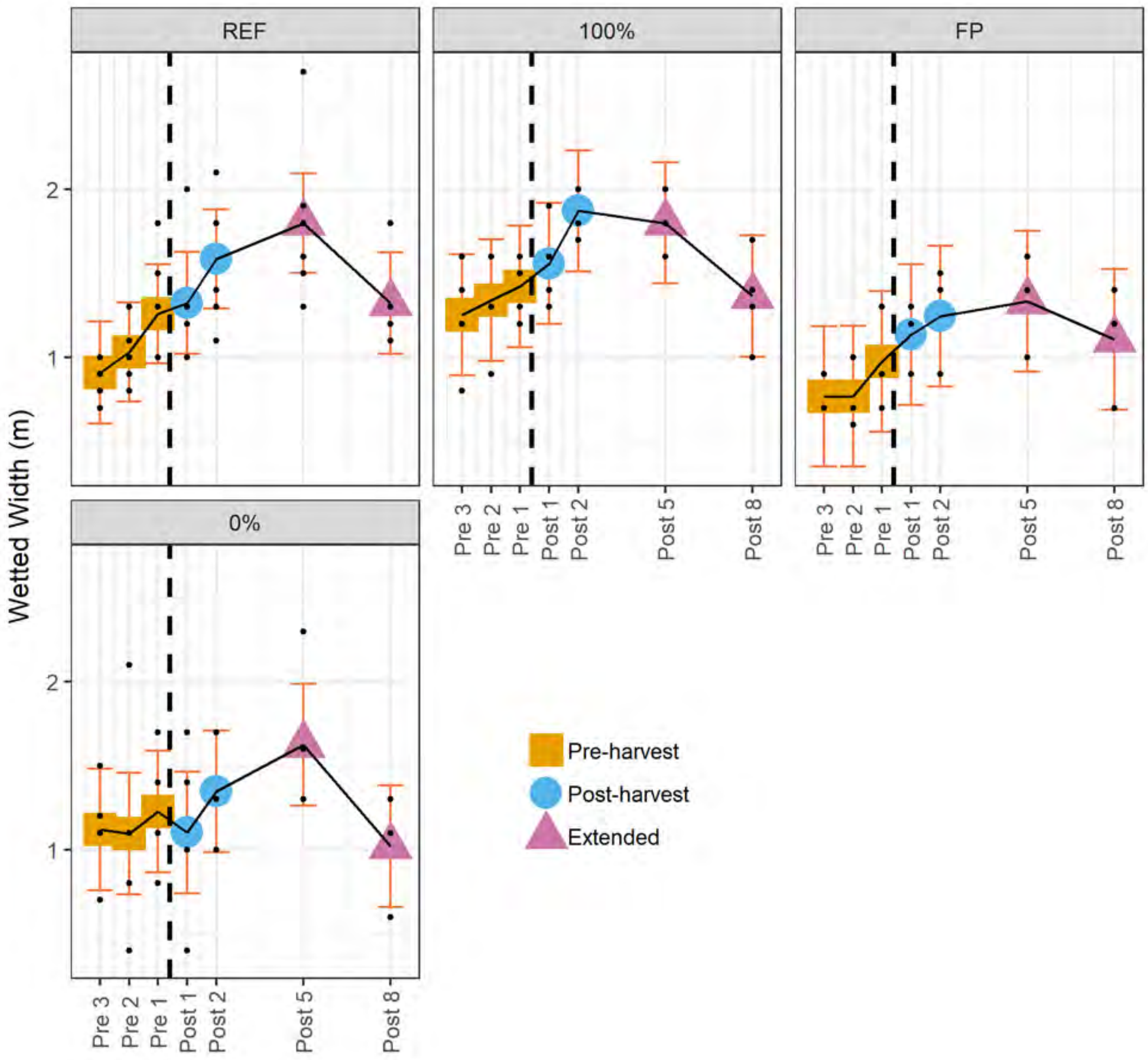

Figure 7-2. Mean wetted width (m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-5. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean annual wetted width (m) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	Post 5	Post 8
Treatment	Estimate (CI)	Estimate (CI)	Estimate (CI)
REF (n = 6)	0.39 (0.25, 0.53)	0.73 (0.56, 0.9)	0.26 (0.07, 0.44)
100% (n = 4)	0.38 (0.21, 0.54)	0.46 (0.25, 0.67)	0.03 (-0.18, 0.23)
FP (n = 3)	0.35 (0.16, 0.54)	0.50 (0.26, 0.74)	0.27 (0.03, 0.51)
0% (n = 4)	0.08 (-0.09, 0.24)	0.48 (0.27, 0.69)	-0.13 (-0.34, 0.08)

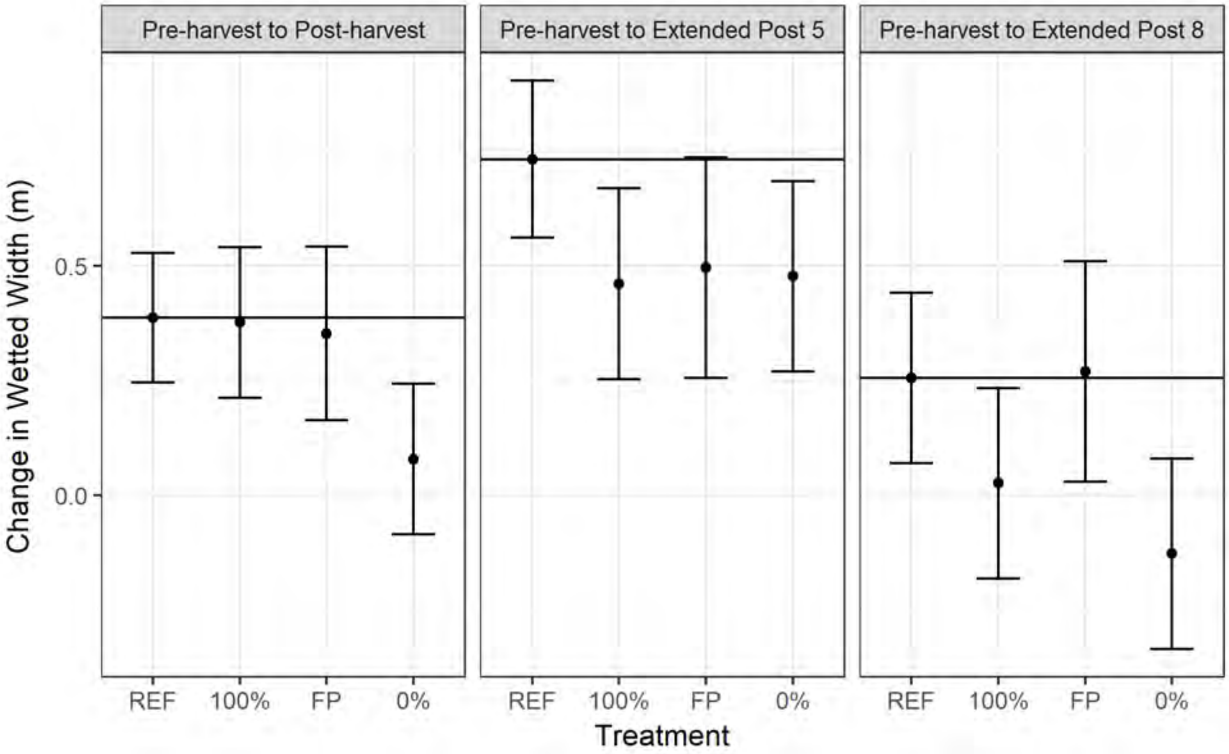

Figure 7-3. The within-treatment estimate of the change and 95% confidence intervals for mean annual wetted width (m) between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions

Table 7-6. The between-treatment comparison of the change, 95% confidence intervals (CI) and P-values (P) for mean annual wetted width (m) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. Estimates with P-values ≤0.10 are bolded. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2		**Post 5**		**Post 8**	
Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
100% vs. REF	-0.01 (-0.23, 0.21)	0.92	**-0.27 (-0.54, 0)**	**0.05**	-0.23 (-0.51, 0.05)	0.11
FP vs. REF	-0.03 (-0.27, 0.20)	0.77	-0.24 (-0.53, 0.06)	0.11	0.01 (-0.29, 0.32)	0.93
0% vs. REF	**-0.31 (-0.53, -0.09)**	**0.01**	**-0.26 (-0.52, 0.01)**	**0.06**	**-0.38 (-0.66, -0.10)**	**0.01**
0% vs. FP	**-0.27 (-0.53, -0.02)**	**0.03**	-0.02 (-0.34, 0.3)	0.91	**-0.4 (-0.72, -0.08)**	**0.01**
0% vs. 100%	**-0.30 (-0.53, -0.07)**	**0.01**	0.02 (-0.28, 0.31)	0.91	-0.15 (-0.45, 0.14)	0.30
FP vs. 100%	-0.02 (-0.28, 0.23)	0.85	0.03 (-0.28, 0.35)	0.83	0.24 (-0.07, 0.56)	0.13

7-4.2. STREAM DEPTH

Mean annual depth ranged from 3 to 14 cm in the pre-harvest period, 3 to 13 cm in Post 1 and Post 2, 4 to 11 cm in Post 5, and 3 to 11 cm in Post 8 (**Figure 7-4**). We did not find clear evidence that depth varied among treatments over time (P = 0.22; **Table 7-7**; **Figure 7-5**; **Table 7-8**).

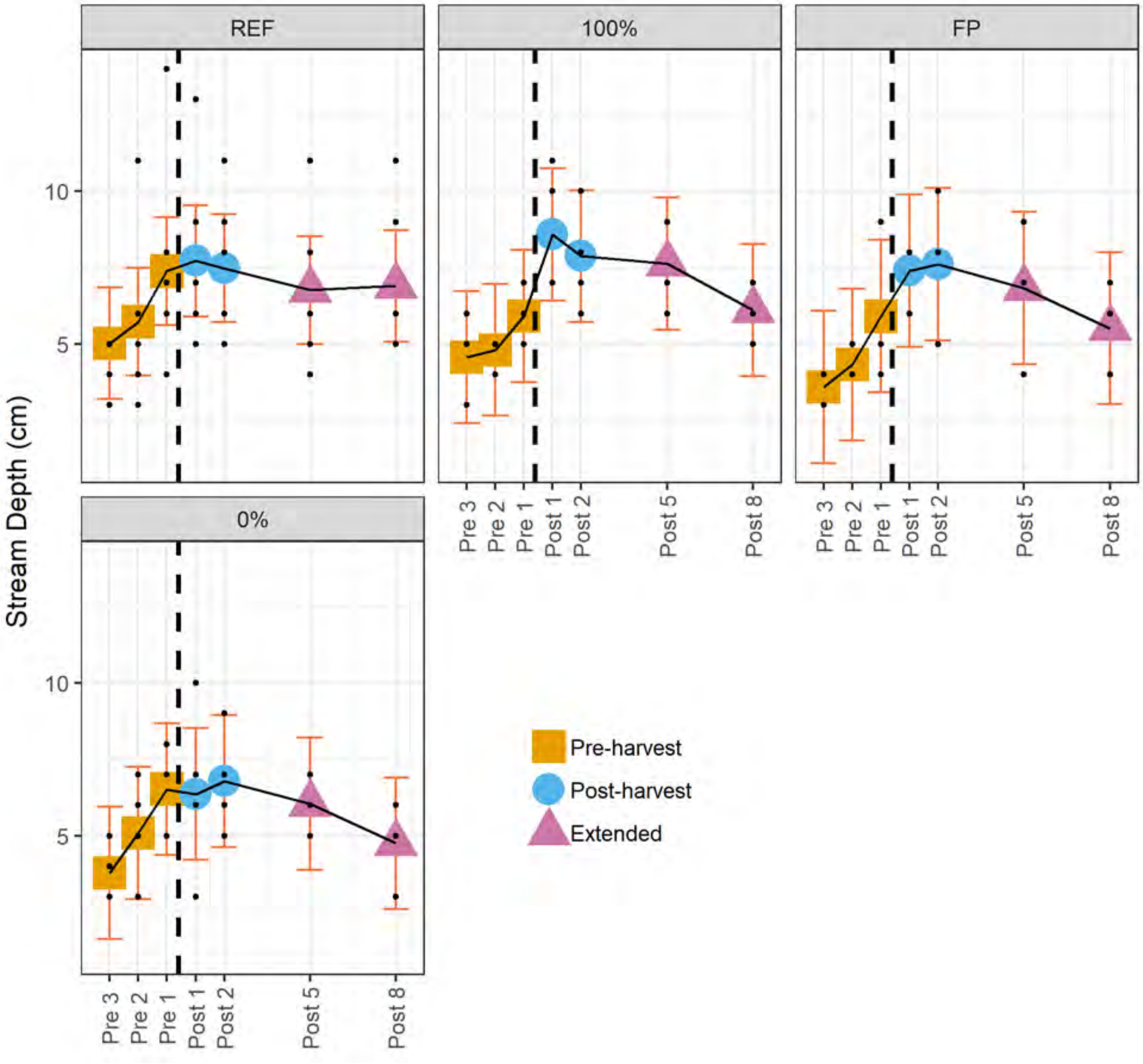

Figure 7-4. Mean depth (cm) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-7. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean annual depth (cm) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	**Post 5**	**Post 8**
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.6 (0.6, 2.5)	0.7 (-0.4, 1.9)	0.9 (-0.4, 2.1)
100% (n = 4)	3.1 (2.0, 4.2)	2.5 (1.1, 3.9)	1.0 (-0.4, 2.4)
FP (n = 3)	2.9 (1.6, 4.2)	2.2 (0.6, 3.8)	0.9 (-0.7, 2.5)
0% (n = 4)	1.4 (0.3, 2.5)	0.9 (-0.5, 2.3)	-0.4 (-1.8, 1.0)

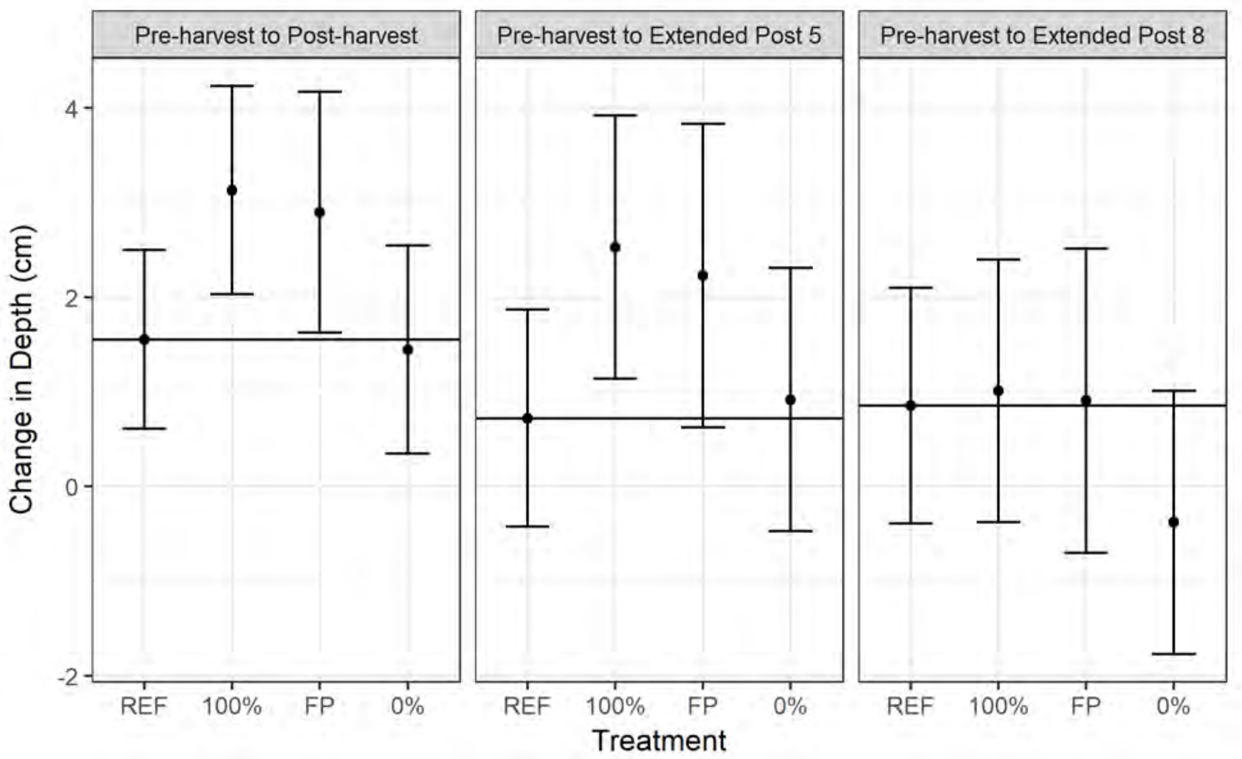

Figure 7-5. The within-treatment estimate of the change and 95% confidence intervals for mean annual depth (cm) between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-8. The between-treatment comparison of the change and 95% confidence intervals (CI) for mean annual depth (cm) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison and is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	1.6 (0.1, 3.0)	1.8 (0.0, 3.6)	0.2 (-1.7, 2.0)
FP vs. REF	1.3 (-0.2, 2.9)	1.5 (-0.5, 3.5)	0.1 (-2.0, 2.1)
0% vs. REF	-0.1 (-1.6, 1.3)	0.2 (-1.6, 2.0)	-1.2 (-3.1, 0.6)
0% vs. FP	-1.5 (-3.1, 0.2)	-1.3 (-3.4, 0.8)	-1.3 (-3.4, 0.8)
0% vs. 100%	-1.7 (-3.2, -0.1)	-1.6 (-3.6, 0.4)	-1.4 (-3.4, 0.6)
FP vs. 100%	-0.2 (-1.9, 1.4)	-0.3 (-2.4, 1.8)	-0.1 (-2.2, 2.0)

7-4.3. PROPORTION OF DRY LENGTH

Mean annual proportion of dry channel length ranged from 0.00 to 0.51 in the pre-harvest period, 0.00 to 0.46 in Post 1 and Post 2, 0.00 to 0.63 in Post 7, and 0.00 to 0.42 in Post 8 (**Figure 7-6**). We did not find clear evidence that dry channel length varied among treatments (P = 0.25; **Table 7-9**; **Figure 7-7**; **Table 7-10**).

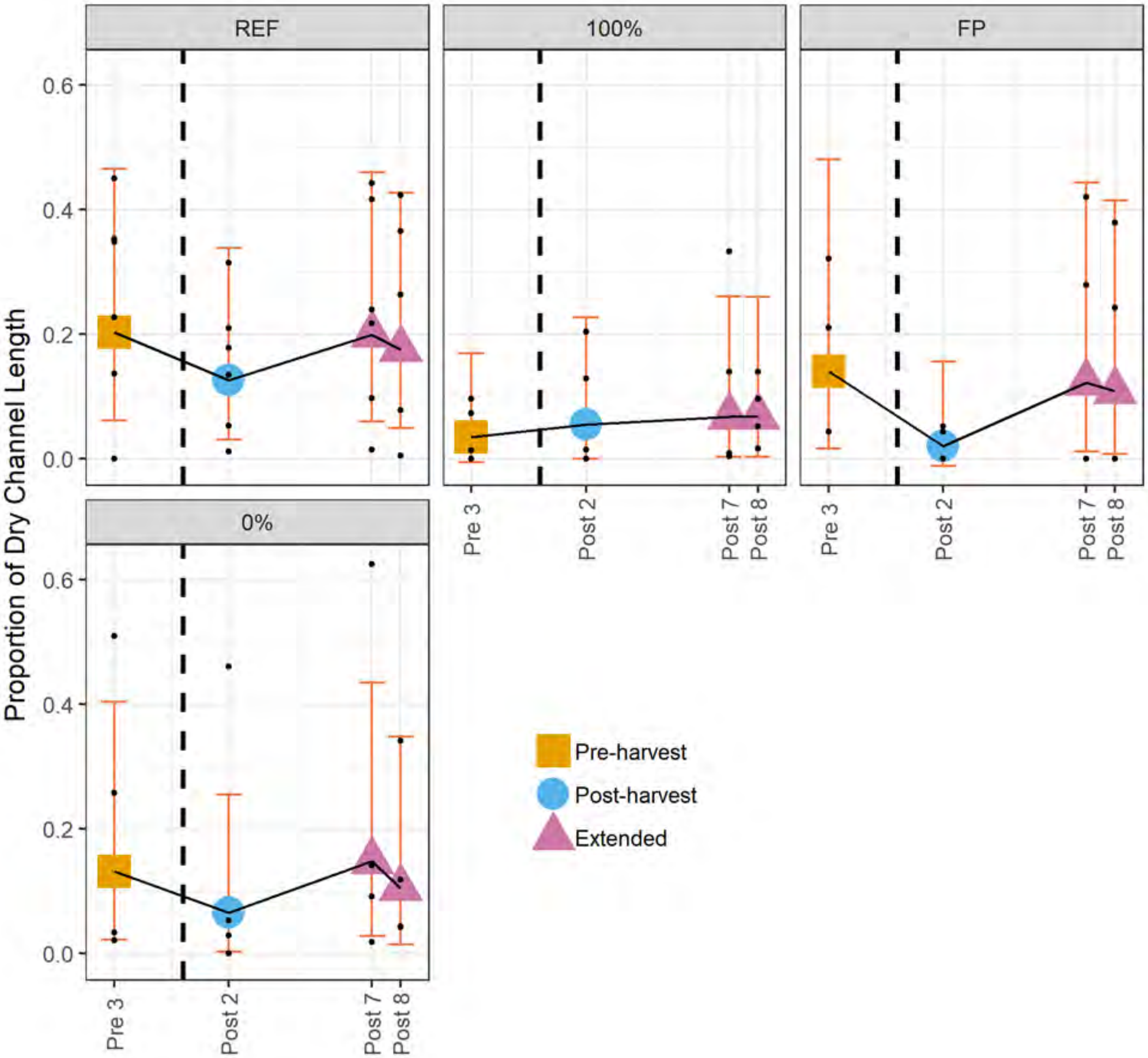

Figure 7-6. Mean proportion of dry channel length by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 2, and extended includes Post 7 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-9. The within-treatment estimate of odds ratios and 95% confidence intervals (CI) for the change in the mean annual proportion of dry channel between the pre-harvest period and Post 2, Post 7 and Post 8.

	Post 2	Post 7	Post 8
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	0.60 (0.32, 1.12)	0.98 (0.52, 1.82)	0.85 (0.44, 1.65)
100% (n = 4)	1.39 (0.65, 2.99)	1.66 (0.77, 3.55)	1.65 (0.77, 3.54)
FP (n = 3)	0.22 (0.09, 0.53)	0.86 (0.36, 2.09)	0.77 (0.32, 1.86)
0% (n = 4)	0.52 (0.24, 1.12)	1.13 (0.53, 2.42)	0.80 (0.37, 1.71)

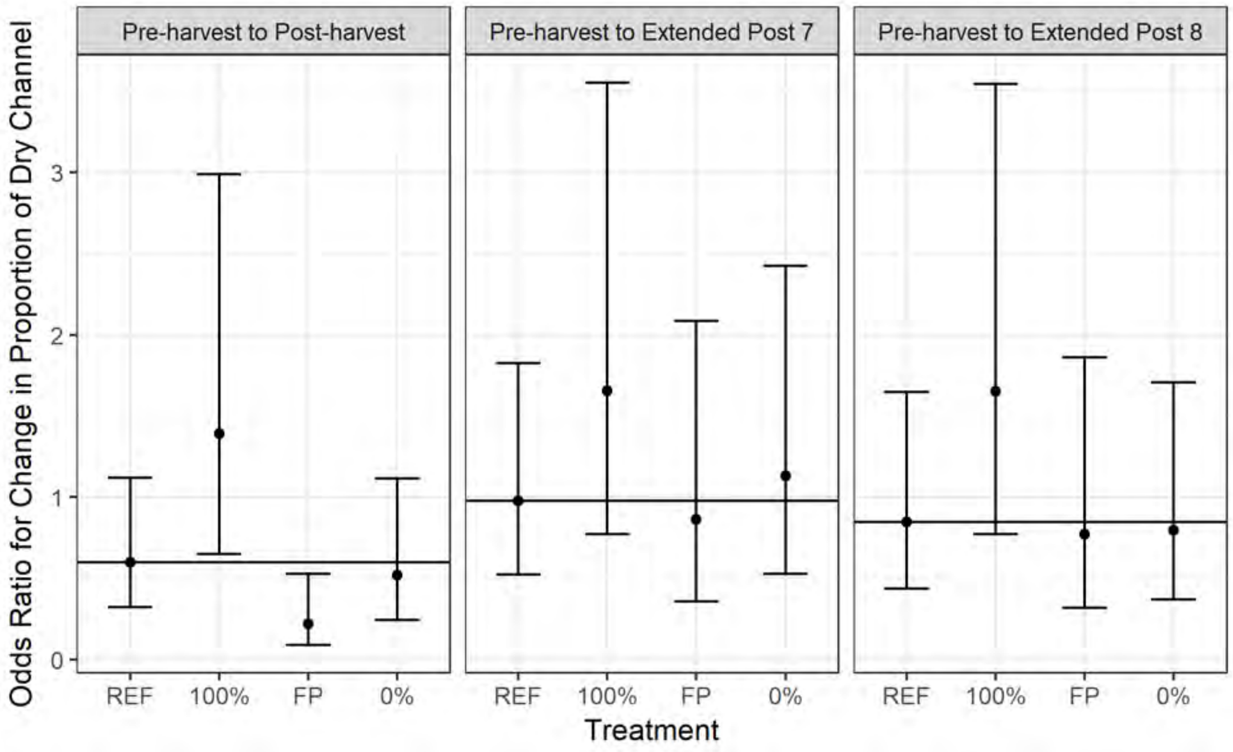

Figure 7-7. The within-treatment estimate of odds ratios and 95% confidence intervals for the change in the mean annual proportion of dry channel between the pre-harvest and post-harvest (Post 2), and extended (Post 7 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-10. The between-treatment comparison of odds ratios and 95% confidence intervals (CI) for the relative change in the mean annual proportion of the channel that was dry between the pre-harvest period and Post 2, Post 7 and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 2	**Post 7**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	2.32 (0.87, 6.21)	1.69 (0.63, 4.53)	1.95 (0.71, 5.35)
FP vs. REF	0.36 (0.12, 1.07)	0.88 (0.30, 2.60)	0.91 (0.30, 2.74)
0% vs. REF	0.87 (0.32, 2.32)	1.15 (0.43, 3.09)	0.94 (0.34, 2.58)
0% vs. FP	2.38 (0.74, 7.63)	1.31 (0.41, 4.19)	1.03 (0.32, 3.31)
0% vs. 100%	0.37 (0.13, 1.10)	0.68 (0.23, 2.01)	0.48 (0.16, 1.42)
FP vs. 100%	0.16 (0.05, 0.50)	0.52 (0.16, 1.67)	0.47 (0.15, 1.50)

7-4.4. BANKFULL WIDTH

Mean annual bankfull width ranged from 0.8 to 3.1 m in the pre-harvest period, 1.0 to 2.6 m in Post 1 and Post 2, 1.4 to 3.5 m in Post 5, and 1.0 to 2.8 m in Post 8 (**Figure 7-8**). We found evidence that treatments differed in the magnitude of change over time (P <0.001; **Table 7-11**; **Figure 7-9**; **Table 7-12**). In Post 1 and Post 2, we estimated the between-treatment comparison of bankfull width for the 0% treatment and reference to be a 0.4 m (P = 0.01) decrease compared to the pre-harvest period, after controlling for temporal changes in the references. The estimated mean width in the 0% treatment was also 0.5 m (P <0.01) and 0.5 m (P <0.001) less than estimates for the 100% and FP treatments, after adjusting for pre-harvest differences among the treatment sites. In Post 5, we estimated a 0.4 m (P = 0.04), 0.4 m (P = 0.07) and 0.6 m (P <0.01) decrease in mean width in the 100%, FP and 0% treatments, compared to the pre-harvest period, after controlling for temporal changes in the references (for which we estimated a 0.7 m increase in the same period; **Table 7-11**). In Post 8, we estimated a 0.7 m (P <0.001) decrease in the 0% treatment, compared to the pre-harvest period, after controlling for temporal changes in the references. The estimated mean in the 0% treatment was also 0.5 m (P = 0.01) and 0.6 m (P <0.01) less, respectively, than the estimates for the 100% and FP treatments, after adjusting for pre-harvest differences among the treatment sites.

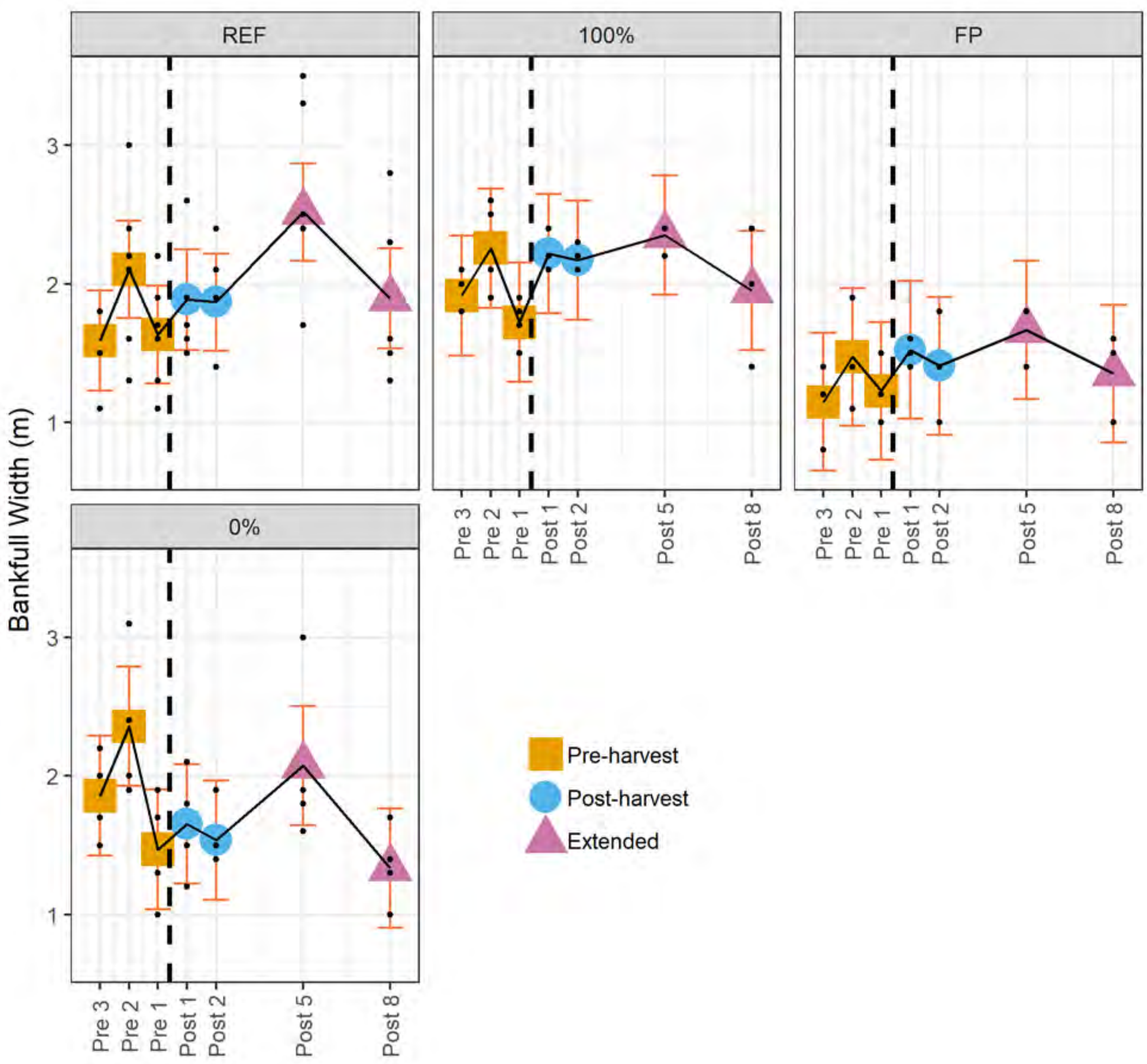

Figure 7-8. Mean bankfull width (m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-11. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean annual bankfull width (m) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	Post 5	Post 8
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	0.10 (-0.08, 0.28)	0.74 (0.52, 0.96)	0.12 (-0.12, 0.36)
100% (n = 4)	0.23 (0.02, 0.44)	0.39 (0.12, 0.65)	-0.01 (-0.28, 0.26)
FP (n = 3)	0.18 (-0.06, 0.43)	0.38 (0.08, 0.69)	0.07 (-0.24, 0.38)
0% (n = 4)	-0.30 (-0.51, -0.09)	0.18 (-0.09, 0.45)	-0.56 (-0.83, -0.29)

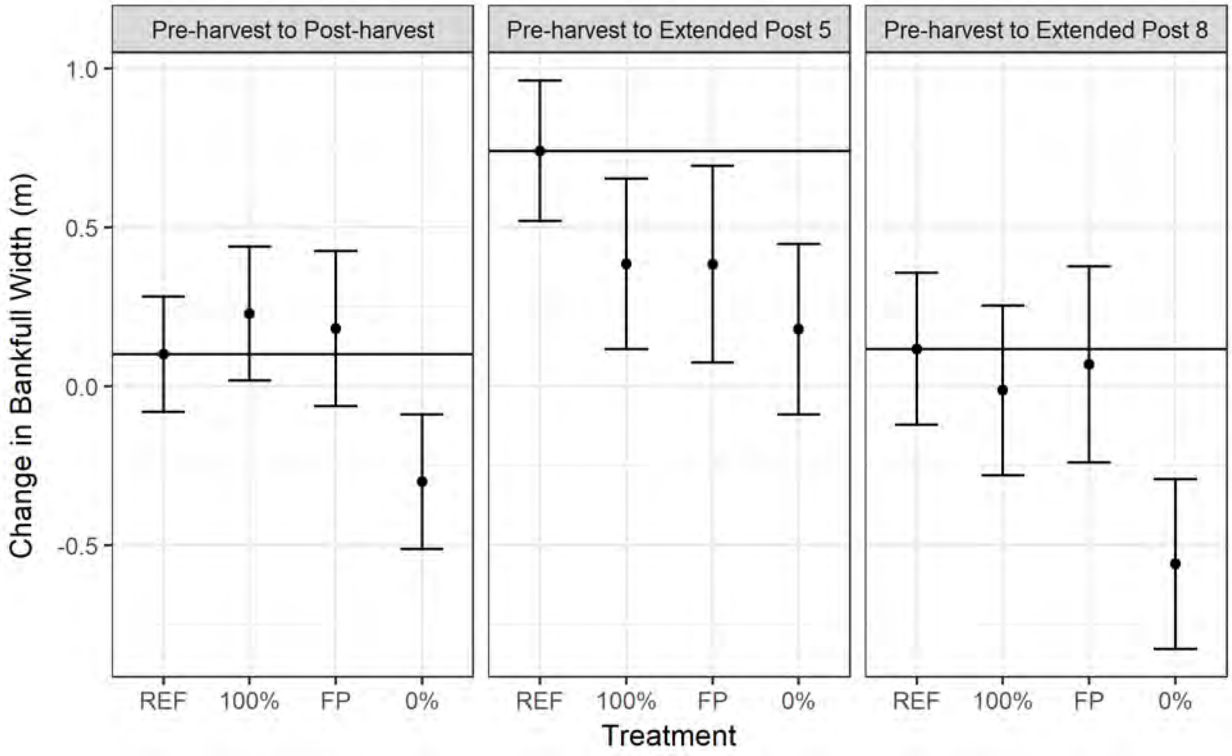

Figure 7-9. The within-treatment estimate of the change and 95% confidence intervals for mean annual bankfull width (m) between the pre-harvest and post-harvest (Post 1 and Post 2), extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-12. The between-treatment comparison of the change, 95% confidence intervals (CI) and P-values (P) for mean annual bankfull width (m) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. Estimates with P-values ≤0.10 are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2		**Post 5**		**Post 8**	
Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
100% vs. REF	0.13 (-0.15, 0.41)	0.36	**-0.36 (-0.70, -0.01)**	**0.04**	-0.13 (-0.49, 0.23)	0.47
FP vs. REF	0.08 (-0.22, 0.39)	0.59	**-0.36 (-0.74, 0.02)**	**0.07**	-0.05 (-0.44, 0.34)	0.80
0% vs. REF	**-0.40 (-0.68, -0.12)**	**0.01**	**-0.56 (-0.91, -0.21)**	**<0.01**	**-0.68 (-1.04, -0.32)**	**<0.001**
0% vs. FP	**-0.48 (-0.81, -0.16)**	**<0.01**	-0.20 (-0.61, 0.2)	0.32	**-0.63 (-1.04, -0.22)**	**<0.01**
0% vs. 100%	**-0.53 (-0.83, -0.23)**	**<0.001**	-0.21 (-0.58, 0.17)	0.28	**-0.55 (-0.92, -0.17)**	**0.01**
FP vs. 100%	-0.05 (-0.37, 0.28)	0.78	0.00 (-0.41, 0.41)	1.00	0.08 (-0.33, 0.49)	0.69

7-4.5. FINES AND SAND SUBSTRATES

Mean annual proportion of stream bed dominated by fines and sand ranged from 0.00 to 0.86 in the pre-harvest period, 0.02 to 0.70 in Post 1 and Post 2, 0.02 to 0.48 in Post 5, and 0.03 to 0.33 and Post 8 (**Figure 7-10**). We found evidence that treatments differed significantly in the magnitude of change over time (P = 0.10; **Table 7-13**; **Figure 7-11**; **Table 7-14**). In Post 1 and Post 2, we estimated the between-treatment comparison for the 0% treatment and reference to be a 143% increase (P = 0.03) in the mean odds compared to the pre-harvest period, after controlling for temporal changes in the references. We also estimated a 100% increase (P = 0.11) in the mean odds for the proportion of stream dominated by fines and sand in the 0% treatment compared with the 100% treatment, after adjusting for pre-harvest differences among the treatment sites. In Post 5, we estimated a 128% increase (P = 0.11) in the odds for the proportion of stream dominated by fines and sand in the 0% treatment compared to the pre-harvest period. We also estimated 182% (P = 0.06) greater odds of fines and sand in the 0% treatment relative to the 100% treatment, after adjusting for pre-harvest differences among the treatment sites. In Post 8, we estimated a 371% (P <0.01) and 154% (P = 0.08) increase in the odds of fines and sand for the FP and 0% treatments, respectively, compared with the pre-harvest period; however, we observed a 58% decline in the odds of fines and sand in the reference in the same period (**Table 7-13**).

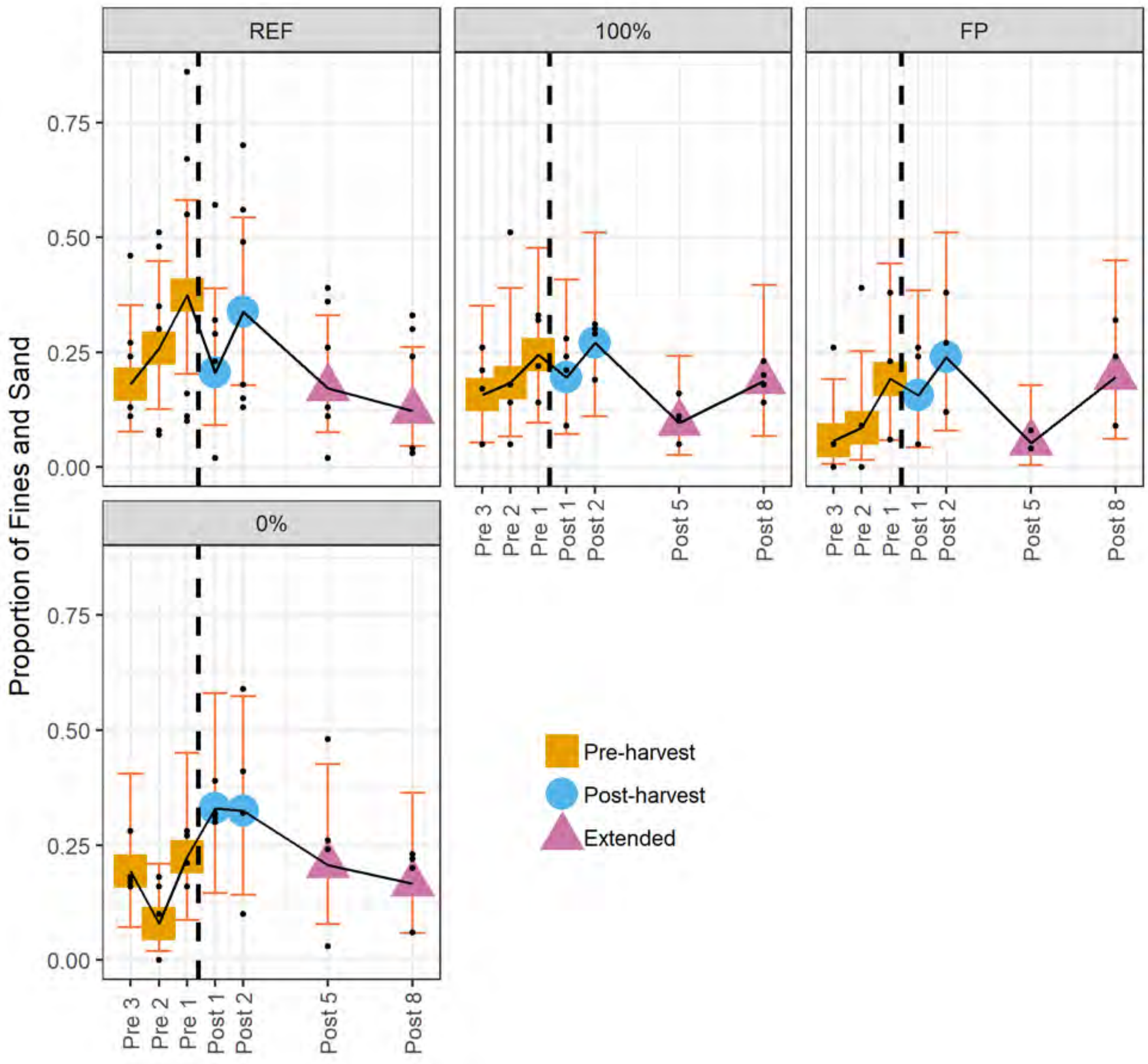

Figure 7-10. Mean proportion of the streambed dominated by fines and sand by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-13. The within-treatment estimate of odds ratios and 95% confidence intervals (CI) for the change in the mean proportion of stream dominated by fines and sand between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	Post 5	Post 8
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.02 (0.60, 1.72)	0.60 (0.32, 1.14)	0.42 (0.21, 0.84)
100% (n = 4)	1.23 (0.67, 2.28)	0.49 (0.22, 1.06)	0.97 (0.44, 2.10)
FP (n = 3)	1.96 (0.97, 3.98)	0.57 (0.23, 1.39)	1.98 (0.81, 4.85)
0% (n = 4)	2.47 (1.34, 4.56)	1.37 (0.63, 2.99)	1.07 (0.49, 2.33)

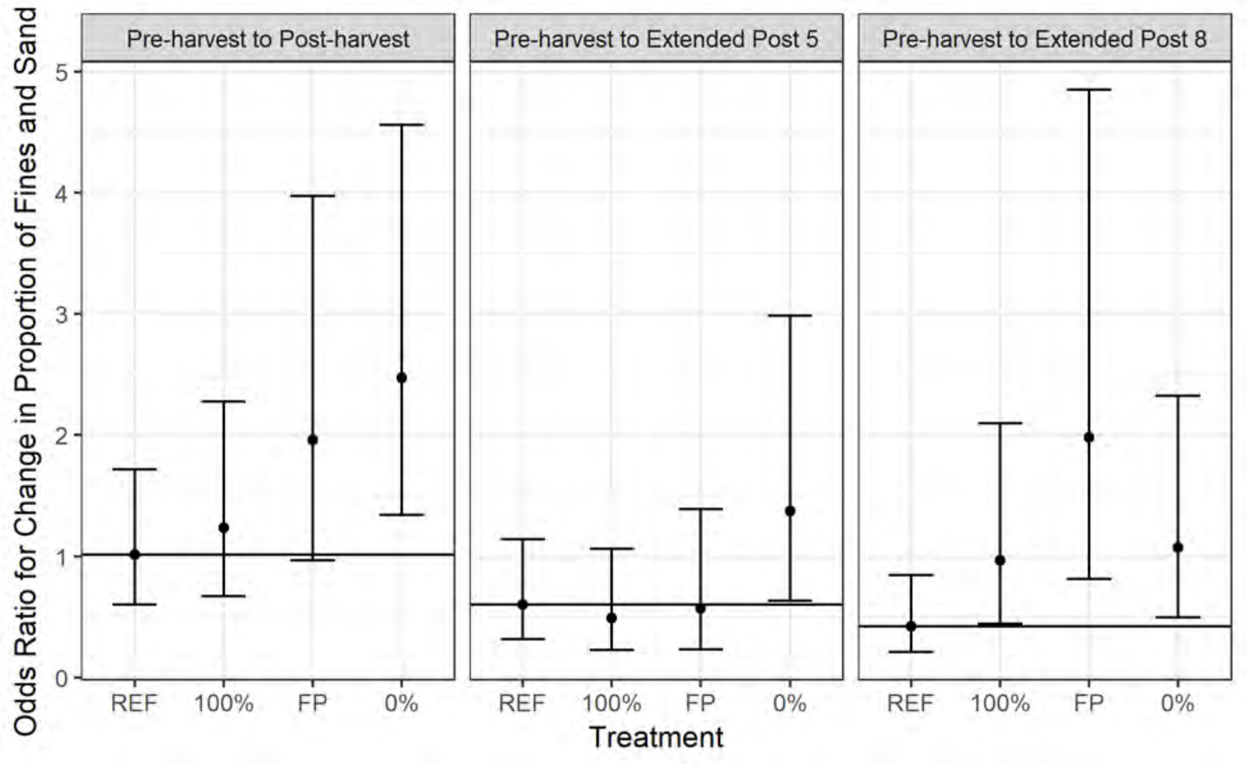

Figure 7-11. The within-treatment estimate of odds ratios and 95% confidence intervals for the change in the mean proportion of stream dominated by fines and sand between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-14. The between-treatment comparison of odds ratios, 95% confidence intervals (CI), and P-values (P) for the relative change in the mean annual proportion of the stream dominated by fines and sand between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. Estimates with P-values ≤0.10 are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2		**Post 5**		**Post 8**	
Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
100% vs. REF	1.22 (0.54, 2.73)	0.63	0.81 (0.30, 2.21)	0.68	2.29 (0.81, 6.48)	0.12
FP vs. REF	1.93 (0.80, 4.66)	0.14	0.94 (0.31, 2.83)	0.91	**4.71 (1.52, 14.6)**	**<0.01**
0% vs. REF	**2.43 (1.09, 5.46)**	**0.03**	2.28 (0.84, 6.24)	0.11	**2.54 (0.90, 7.19)**	**0.08**
0% vs. FP	1.26 (0.49, 3.22)	0.62	2.42 (0.74, 7.92)	0.14	0.54 (0.17, 1.77)	0.30
0% vs. 100%	2.00 (0.84, 4.76)	0.11	**2.82 (0.94, 8.44)**	**0.06**	1.11 (0.37, 3.32)	0.85
FP vs. 100%	1.59 (0.62, 4.05)	0.33	1.16 (0.36, 3.80)	0.80	2.05 (0.63, 6.71)	0.23

7-4.6. Pools

7-4.6.1. Pool Length

Mean annual pool length ranged from 0.7 to 1.7 m in the pre-harvest period, 0.7 to 1.5 m in Post 1 and Post 2, 0.8 to 1.5 m in Post 5, and 0.7 to 1.2 m in Post 8 (**Figure 7-12**). We did not find clear evidence that pool length varied among treatments over time (P = 0.18; **Table 7-15**; **Figure 7-13**; **Table 7-16**).

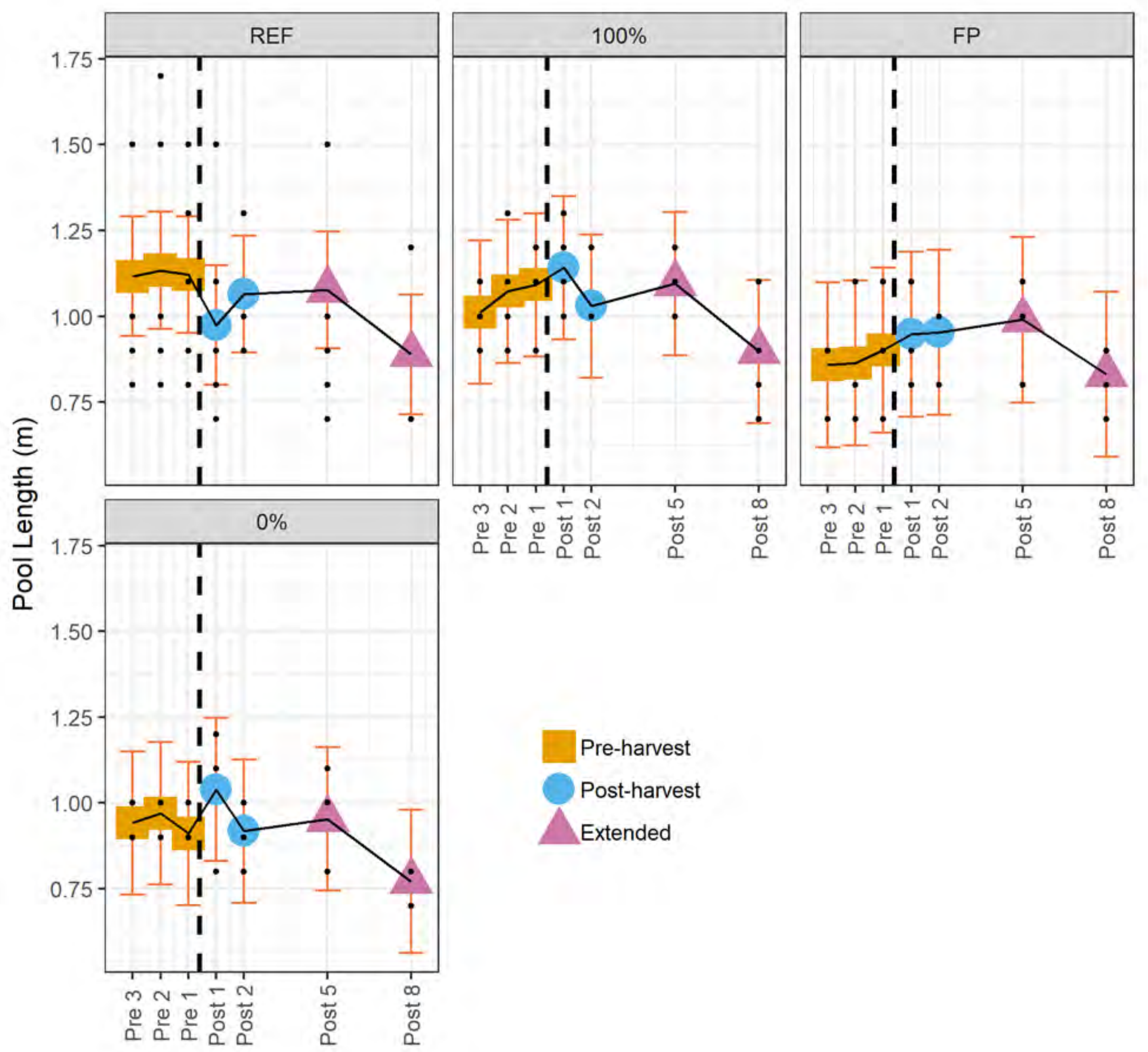

Figure 7-12. Mean pool length (m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-15. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean pool length (m) between the pre-harvest period and Post 1 and Post 2, Post 5 and Post 8.

	Post 1 & 2	**Post 5**	**Post 8**
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	-0.10 (-0.18, -0.02)	-0.05 (-0.14, 0.05)	-0.24 (-0.34, -0.13)
100% (n = 4)	0.03 (-0.07, 0.12)	0.04 (-0.08, 0.15)	-0.16 (-0.28, -0.04)
FP (n = 3)	0.08 (-0.03, 0.18)	0.12 (-0.02, 0.25)	-0.04 (-0.18, 0.09)
0% (n = 4)	0.04 (-0.06, 0.13)	0.01 (-0.11, 0.13)	-0.17 (-0.29, -0.05)

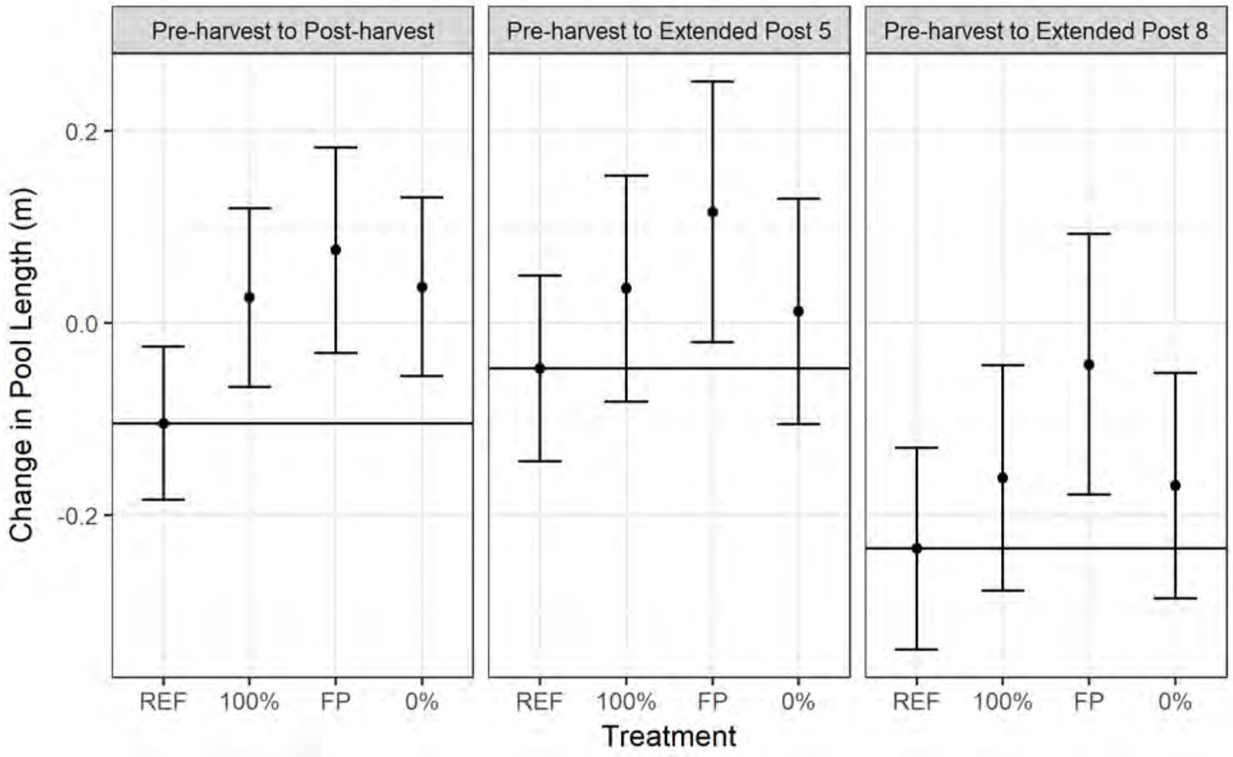

Figure 7-13. The within-treatment estimate of the change and 95% confidence intervals for mean pool length (m) between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-16. The between-treatment comparison of the change, 95% confidence intervals (CI) for mean pool length (m) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison is the treatment with the fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	0.13 (0.01, 0.25)	0.08 (-0.07, 0.24)	0.07 (-0.08, 0.23)
FP vs. REF	0.18 (0.05, 0.31)	0.16 (0.00, 0.33)	0.19 (0.02, 0.36)
0% vs. REF	0.14 (0.02, 0.26)	0.06 (-0.09, 0.21)	0.07 (-0.09, 0.22)
0% vs. FP	-0.04 (-0.18, 0.10)	-0.10 (-0.28, 0.08)	-0.13 (-0.31, 0.05)
0% vs. 100%	0.01 (-0.12, 0.14)	-0.02 (-0.19, 0.14)	-0.01 (-0.17, 0.16)
FP vs. 100%	0.05 (-0.09, 0.19)	0.08 (-0.10, 0.26)	0.12 (-0.06, 0.3)

7-4.6.2. Pool Maximum Depth

Mean annual pool maximum depth ranged from 11 to 28 cm in the pre-harvest period, 11 to 26 cm in Post 1 and Post 2, 14 to 26 cm in Post 5, and 9 to 17 cm in Post 8 (**Figure 7-14**). We did not find clear evidence that pool depth varied among treatments over time (P = 0.99; **Table 7-17**; **Figure 7-15**; **Table 7-18**).

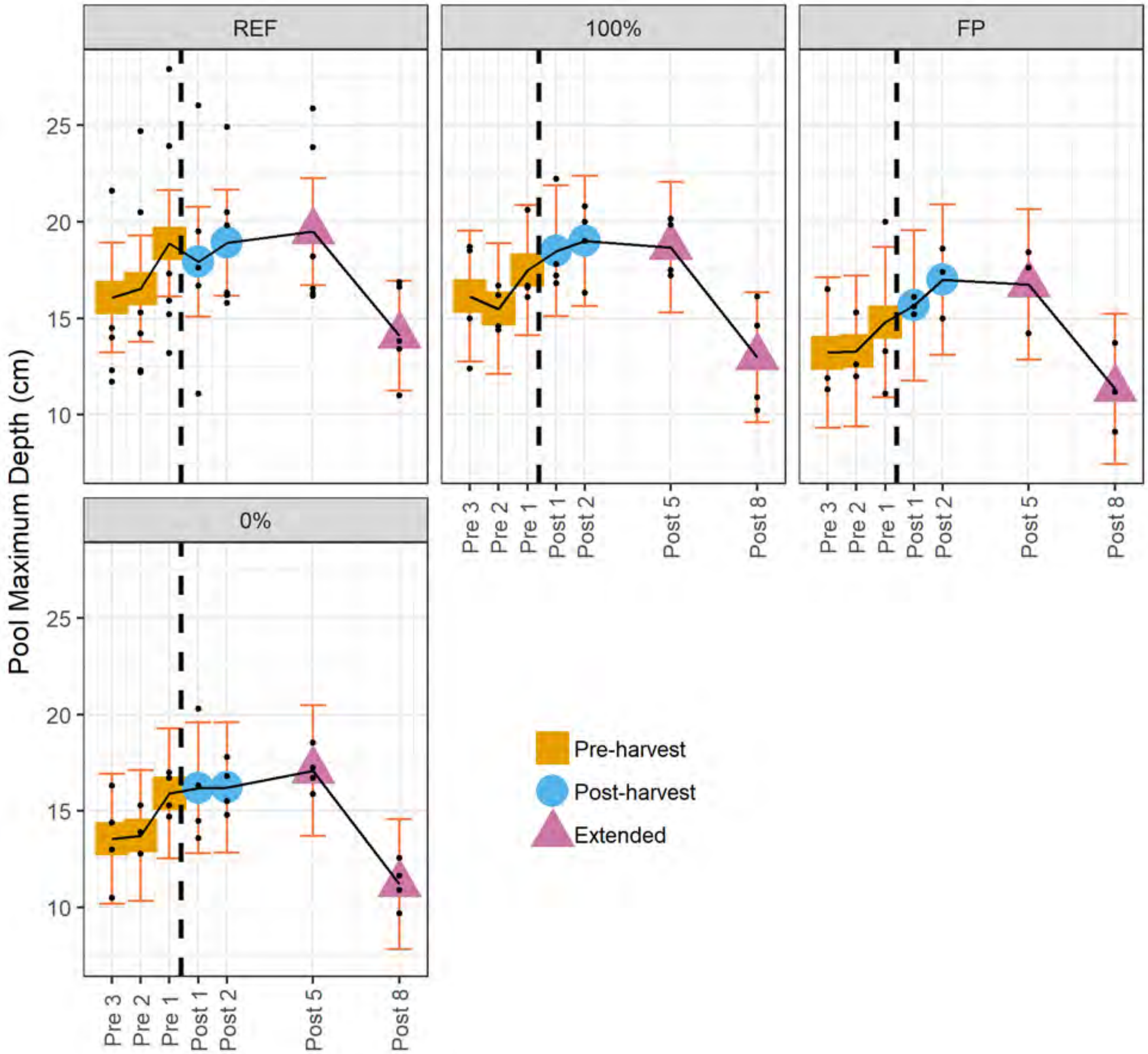

Figure 7-14. Mean pool maximum depth (cm) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-17. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean pool maximum depth (cm) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	Post 5	Post 8
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.3 (-0.2, 2.7)	2.3 (0.5, 4.1)	-3.1 (-5.0, -1.1)
100% (n = 4)	2.4 (0.6, 4.1)	2.3 (0.1, 4.5)	-3.4 (-5.6, -1.2)
FP (n = 3)	2.6 (0.6, 4.6)	3.0 (0.4, 5.5)	-2.4 (-5.0, 0.1)
0% (n = 4)	1.8 (0.1, 3.5)	2.7 (0.5, 4.9)	-3.2 (-5.4, -1.0)

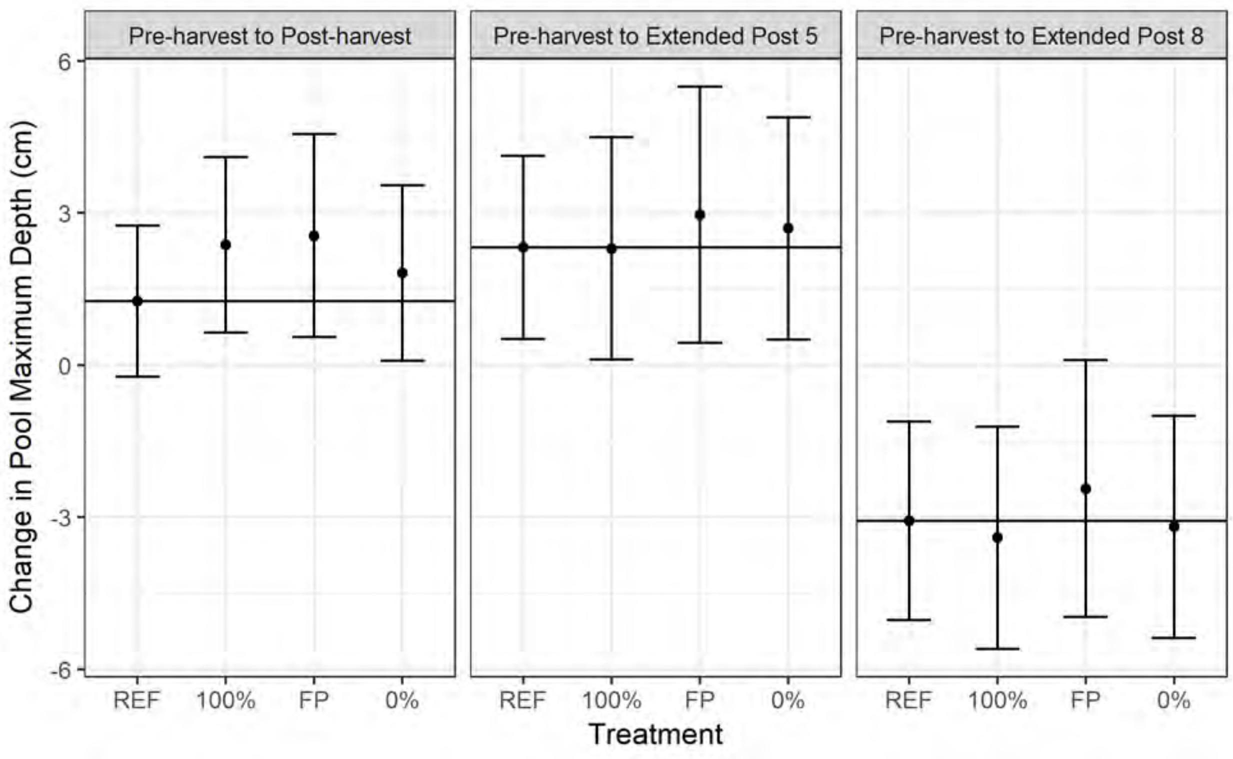

Figure 7-15. The within-treatment estimate of the change and 95% confidence intervals for mean annual pool maximum depth (cm) between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-18. The between-treatment comparison of the change and 95% confidence intervals (CI) for mean annual pool maximum depth (cm) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	1.1 (-1.2, 3.4)	0.0 (-2.9, 2.8)	-0.3 (-3.3, 2.6)
FP vs. REF	1.3 (-1.2, 3.8)	0.6 (-2.5, 3.8)	0.6 (-2.6, 3.8)
0% vs. REF	0.6 (-1.7, 2.8)	0.4 (-2.5, 3.2)	-0.1 (-3.1, 2.8)
0% vs. FP	-0.7 (-3.4, 1.9)	-0.3 (-3.6, 3.1)	-0.8 (-4.1, 2.6)
0% vs. 100%	-0.6 (-3.0, 1.9)	0.4 (-2.7, 3.5)	0.2 (-2.9, 3.3)
FP vs. 100%	0.2 (-2.5, 2.8)	0.7 (-2.7, 4.0)	1.0 (-2.4, 4.3)

7-4.6.3. Pool Density

Mean annual pool density ranged from 0.10 to 0.30 units/m in the pre-harvest period, 0.08 to 0.40 units/m Post 1 and Post 2, 0.07 to 0.29 units/m in Post 5, and 0.23 to 0.69 units/m in Post 8 (**Figure 7-16**). We did not find clear evidence that pool density varied among treatments over time (P = 0.59; **Table 7-19**; **Figure 7-17**; **Table 7-20**).

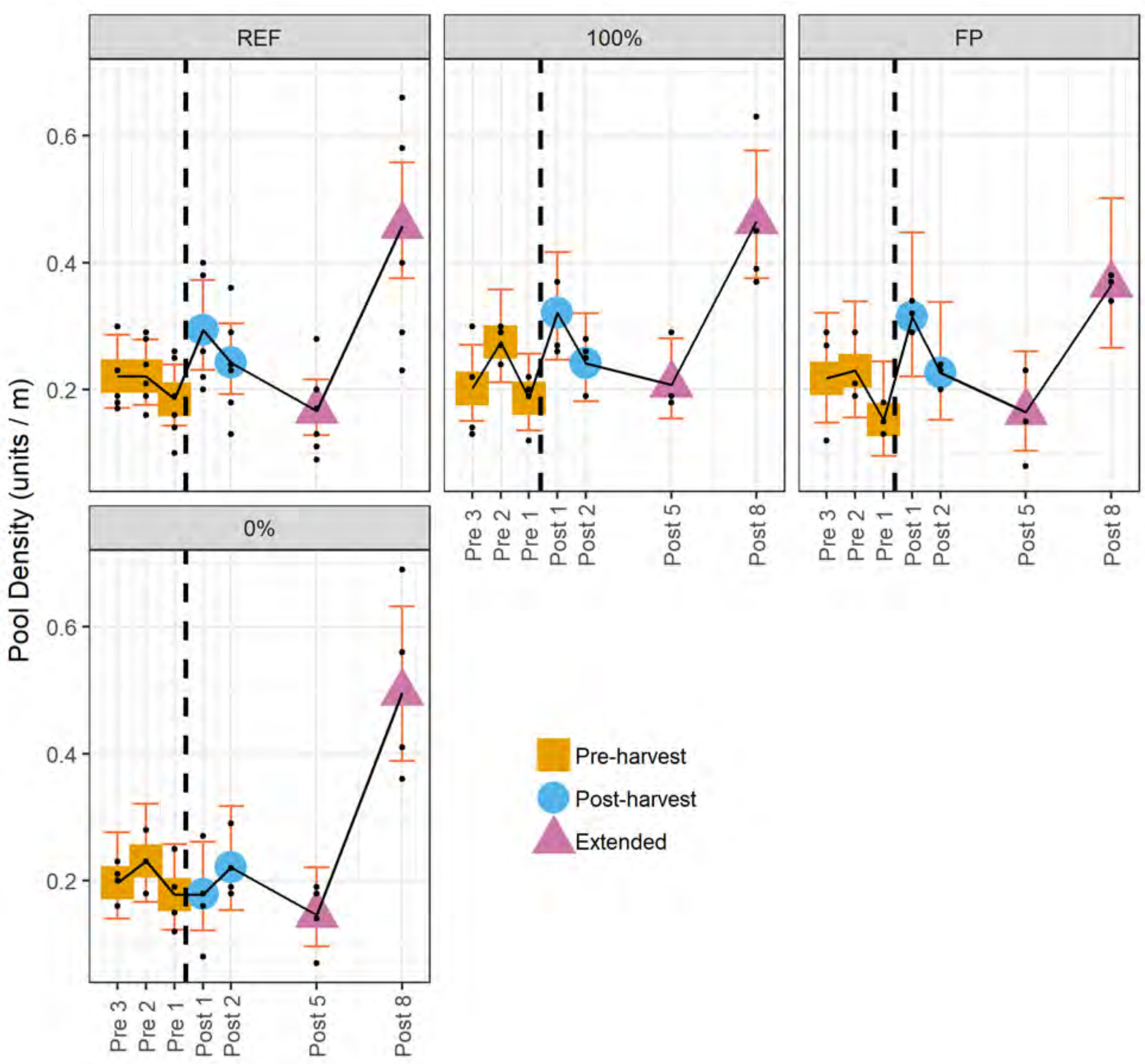

Figure 7-16. Mean pool density (units/m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-19. The within-treatment estimate of the relative change and 95% confidence intervals (CI) for mean annual pool density between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	Post 5	Post 8
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.28 (1.04, 1.57)	0.80 (0.60, 1.06)	2.20 (1.75, 2.76)
100% (n = 4)	1.28 (1.00, 1.62)	0.96 (0.69, 1.32)	2.14 (1.67, 2.74)
FP (n = 3)	1.36 (0.96, 1.92)	0.84 (0.51, 1.38)	1.86 (1.27, 2.71)
0% (n = 4)	0.99 (0.72, 1.36)	0.73 (0.46, 1.14)	2.47 (1.84, 3.32)

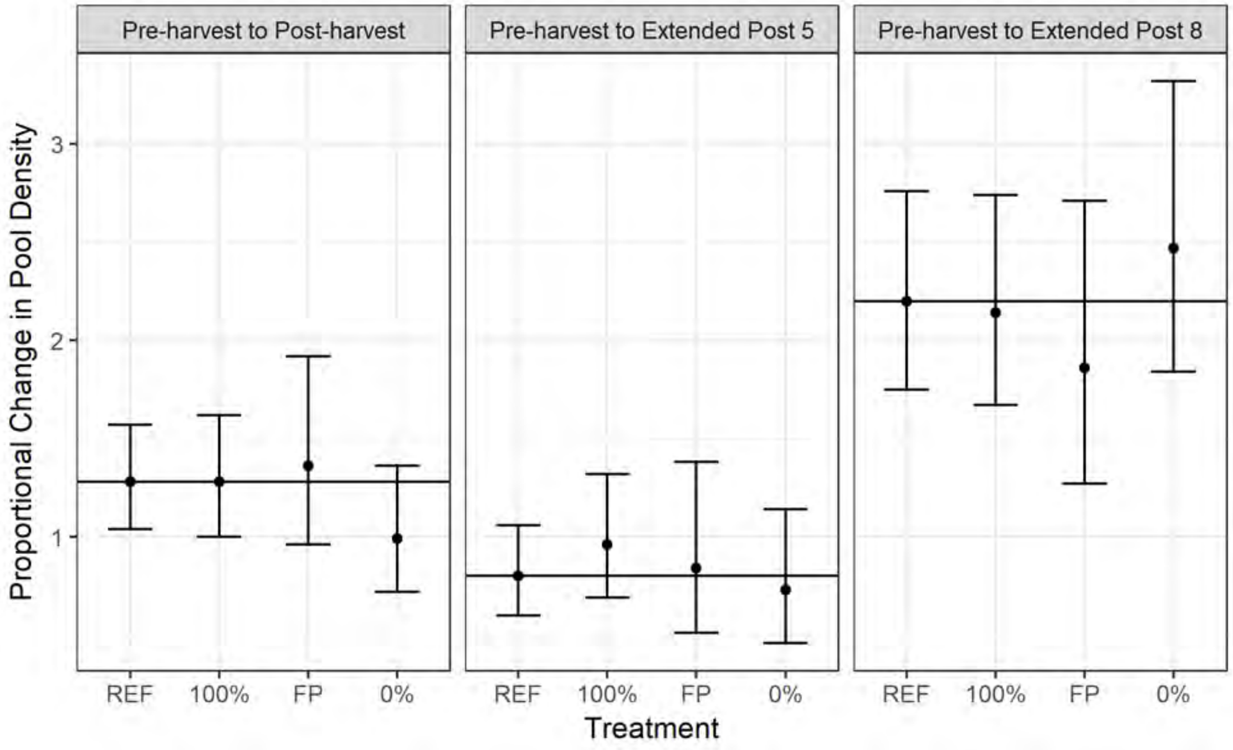

Figure 7-17. The within-treatment estimate of relative change and 95% confidence intervals for mean annual pool density between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-20. The between-treatment comparison of the proportional change and 95% confidence intervals (CI) for mean annual pool density between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	1.00 (0.73, 1.37)	1.20 (0.78, 1.84)	0.97 (0.69, 1.36)
FP vs. REF	1.06 (0.71, 1.59)	1.05 (0.59, 1.86)	0.84 (0.54, 1.31)
0% vs. REF	0.77 (0.53, 1.13)	0.91 (0.53, 1.55)	1.13 (0.78, 1.63)
0% vs. FP	0.73 (0.46, 1.17)	0.87 (0.44, 1.71)	1.33 (0.82, 2.15)
0% vs. 100%	0.77 (0.52, 1.16)	0.76 (0.44, 1.32)	1.16 (0.79, 1.70)
FP vs. 100%	1.06 (0.70, 1.62)	0.87 (0.48, 1.59)	0.87 (0.55, 1.36)

7-4.7. RIFFLES

7-4.7.1. Riffle Density

Mean annual riffle density ranged from 0.21 to 0.61 units/m in the pre-harvest period, 0.10 to 0.66 units/m in Post 1 and Post 2, 0.18 to 0.64 units/m in Post 5, and 0.27 to 0.58 units/m in Post 8 (**Figure 7-18**). We did not find clear evidence that riffle density varied among treatments over time (P = 0.55; **Table 7-21**; **Figure 7-19**; **Table 7-22**).

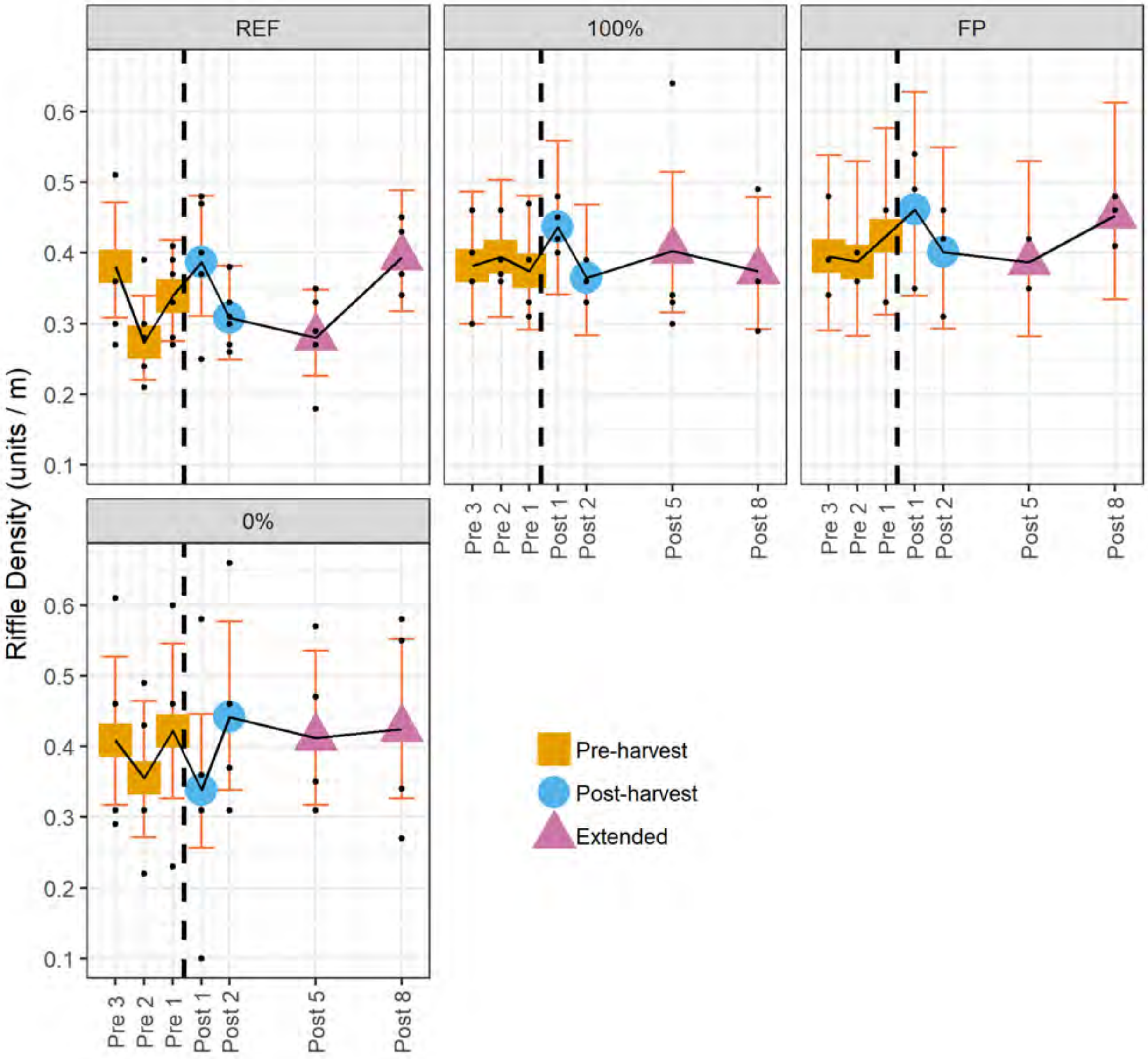

Figure 7-18. Mean riffle density (units/m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-21. The within-treatment estimate of the relative change and 95% confidence intervals (CI) for mean annual riffle density between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	**Post 5**	**Post 8**
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.05 (0.91, 1.21)	0.85 (0.71, 1.02)	1.20 (1.00, 1.44)
100% (n = 4)	1.04 (0.89, 1.22)	1.05 (0.87, 1.28)	0.98 (0.80, 1.19)
FP (n = 3)	1.07 (0.87, 1.32)	0.96 (0.73, 1.26)	1.13 (0.87, 1.46)
0% (n = 4)	0.98 (0.82, 1.18)	1.05 (0.84, 1.31)	1.08 (0.86, 1.35)

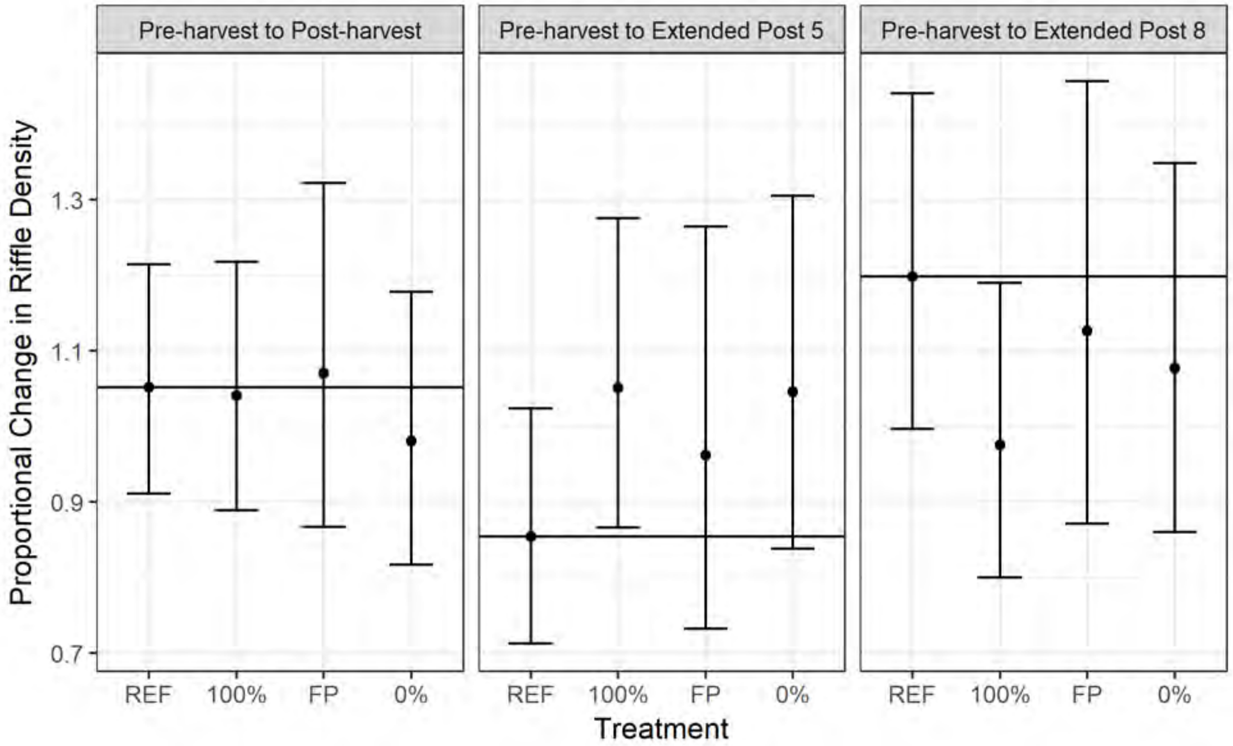

Figure 7-19. The within-treatment estimate of relative change and 95% confidence intervals for mean annual riffle denisty between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-22. The between-treatment comparison of the proportional change and 95% confidence intervals (CI) for mean annual riffle density between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	0.99 (0.80, 1.22)	1.23 (0.94, 1.60)	0.81 (0.62, 1.07)
FP vs. REF	1.02 (0.79, 1.31)	1.13 (0.81, 1.56)	0.94 (0.68, 1.29)
0% vs. REF	0.93 (0.74, 1.18)	1.22 (0.92, 1.63)	0.90 (0.67, 1.20)
0% vs. FP	0.92 (0.69, 1.21)	1.09 (0.76, 1.55)	0.96 (0.68, 1.35)
0% vs. 100%	0.94 (0.74, 1.20)	0.99 (0.74, 1.34)	1.10 (0.82, 1.49)
FP vs. 100%	1.03 (0.79, 1.34)	0.92 (0.65, 1.28)	1.15 (0.83, 1.60)

7-4.8. CASCADES

7-4.8.1. Cascade Density

Mean annual cascade density ranged from 0.01 to 0.31 units/m in the pre-harvest period, 0.02 to 0.60 units/m in Post 1 and Post 2, 0.05 to 0.24 units/m in Post 5, and 0.05 to 0.43 units/m in Post 8 (**Figure 7-20**). We did not find clear evidence that cascade density varied among treatments over time (P = 0.81; **Table 7-23**; **Figure 7-21**; **Table 7-24**).

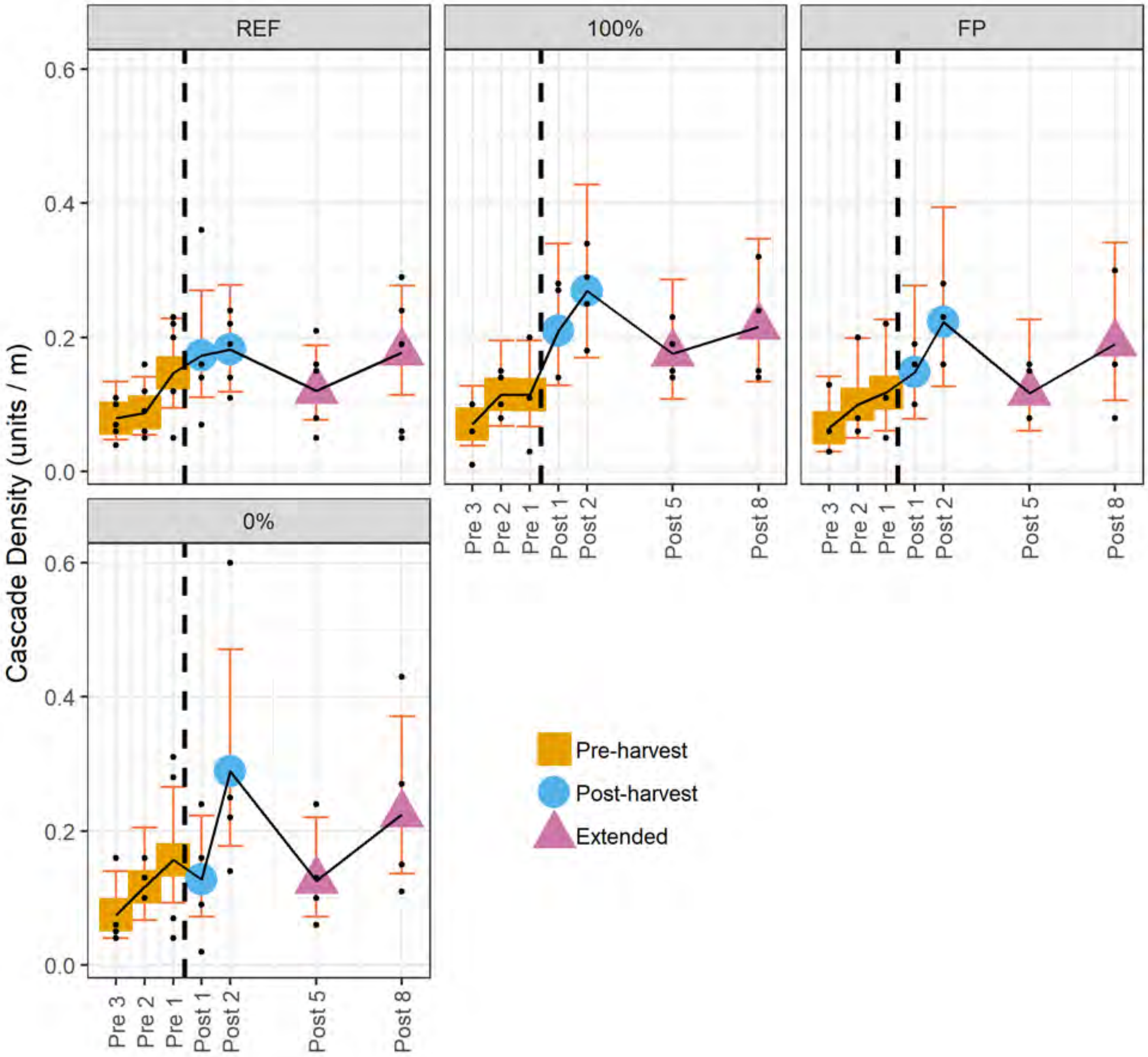

Figure 7-20. Mean cascade density (units/m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-23. The within-treatment estimate of the relative change and 95% confidence intervals (CI) for mean annual cascade density between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	**Post 5**	**Post 8**
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.75 (1.34, 2.29)	1.19 (0.85, 1.67)	1.76 (1.25, 2.47)
100% (n = 4)	2.43 (1.80, 3.29)	1.80 (1.23, 2.64)	2.21 (1.54, 3.16)
FP (n = 3)	1.98 (1.27, 3.10)	1.28 (0.70, 2.35)	2.08 (1.24, 3.49)
0% (n = 4)	1.72 (1.21, 2.45)	1.13 (0.70, 1.83)	2.02 (1.36, 3.01)

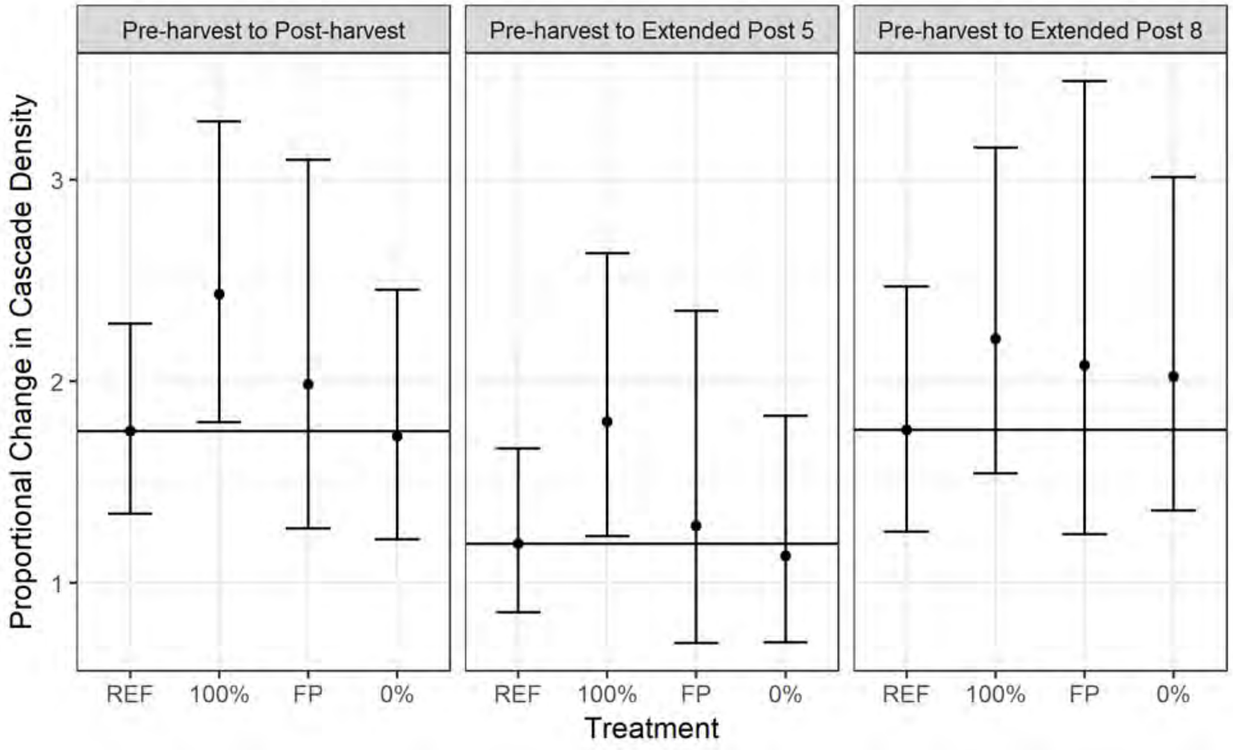

Figure 7-21. The within-treatment estimate of relative change and 95% confidence intervals for mean annual cascade denisty between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-24. The between-treatment comparison of the proportional change and 95% confidence intervals (CI) for mean annual cascade density between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	1.39 (0.93, 2.08)	1.51 (0.91, 2.51)	1.26 (0.77, 2.06)
FP vs. REF	1.13 (0.67, 1.90)	1.08 (0.54, 2.15)	1.18 (0.64, 2.19)
0% vs. REF	0.98 (0.63, 1.53)	0.95 (0.53, 1.71)	1.15 (0.68, 1.94)
0% vs. FP	0.87 (0.49, 1.54)	0.88 (0.41, 1.91)	0.97 (0.51, 1.87)
0% vs. 100%	0.71 (0.45, 1.13)	0.63 (0.34, 1.16)	0.92 (0.54, 1.57)
FP vs. 100%	0.82 (0.48, 1.40)	0.71 (0.35, 1.46)	0.94 (0.50, 1.77)

7-4.9. STEPS

7-4.9.1. Step Height

Mean annual step height ranged from 22 to 46 cm in the pre-harvest period, 26 to 53 cm in Post 1 and Post 2, 22 to 48 cm in Post 5, and 24 to 45 cm in Post 8 (**Figure 7-22**). We did not find clear evidence that step height varied among treatments over time (P = 0.53; **Table 7-25**; **Figure 7-23**; **Table 7-26**).

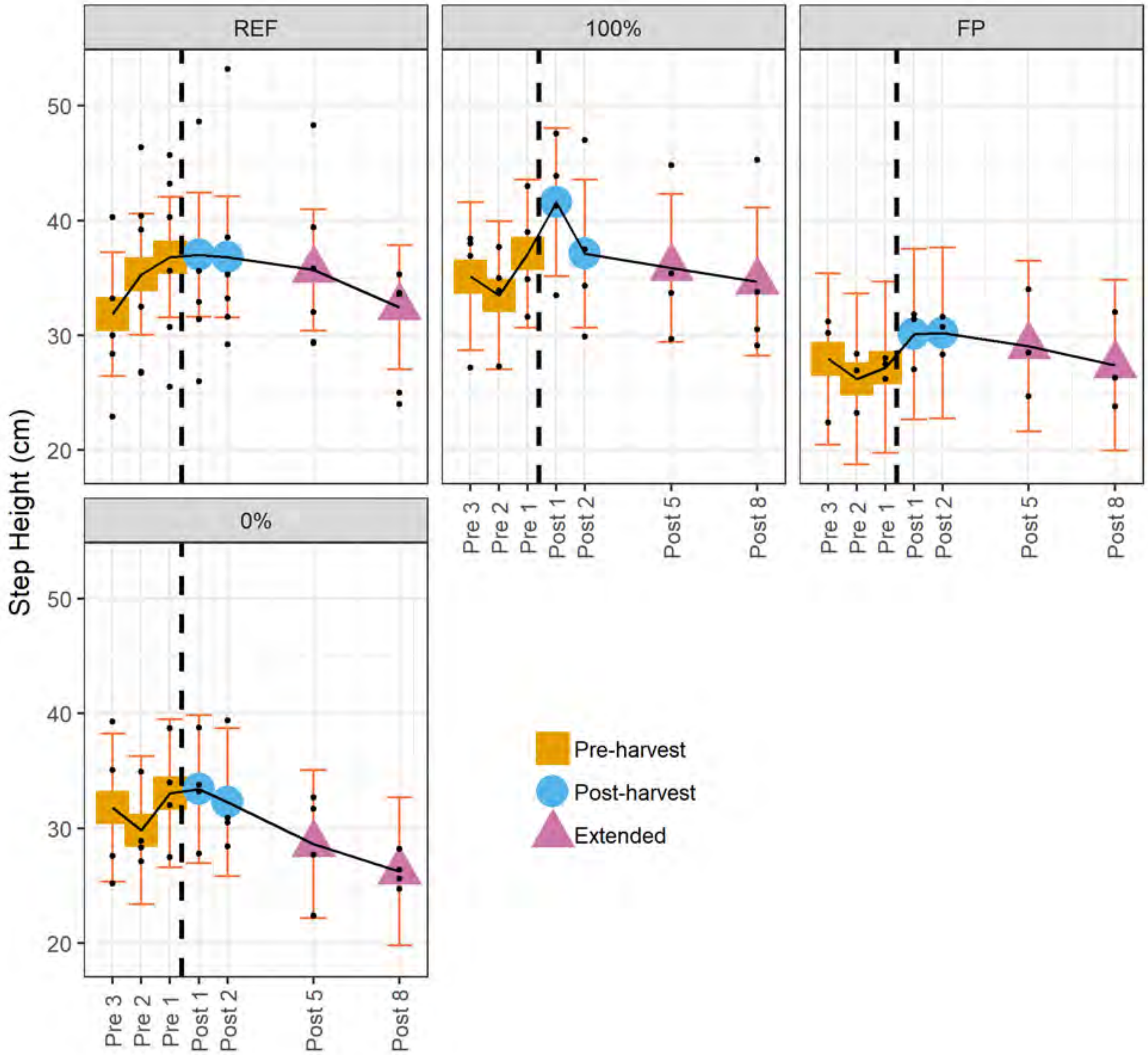

Figure 7-22. Mean step height (cm) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-25. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean annual step height (cm) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	**Post 5**	**Post 8**
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	2.3 (-0.3, 4.8)	1.0 (-2.0, 4.1)	-2.2 (-5.5, 1.1)
100% (n = 4)	4.1 (1.2, 7.1)	0.6 (-3.1, 4.4)	-0.6 (-4.3, 3.1)
FP (n = 3)	3.0 (-0.4, 6.4)	1.9 (-2.3, 6.2)	0.3 (-4.0, 4.6)
0% (n = 4)	1.3 (-1.6, 4.2)	-2.9 (-6.7, 0.8)	-5.3 (-9.0, -1.6)

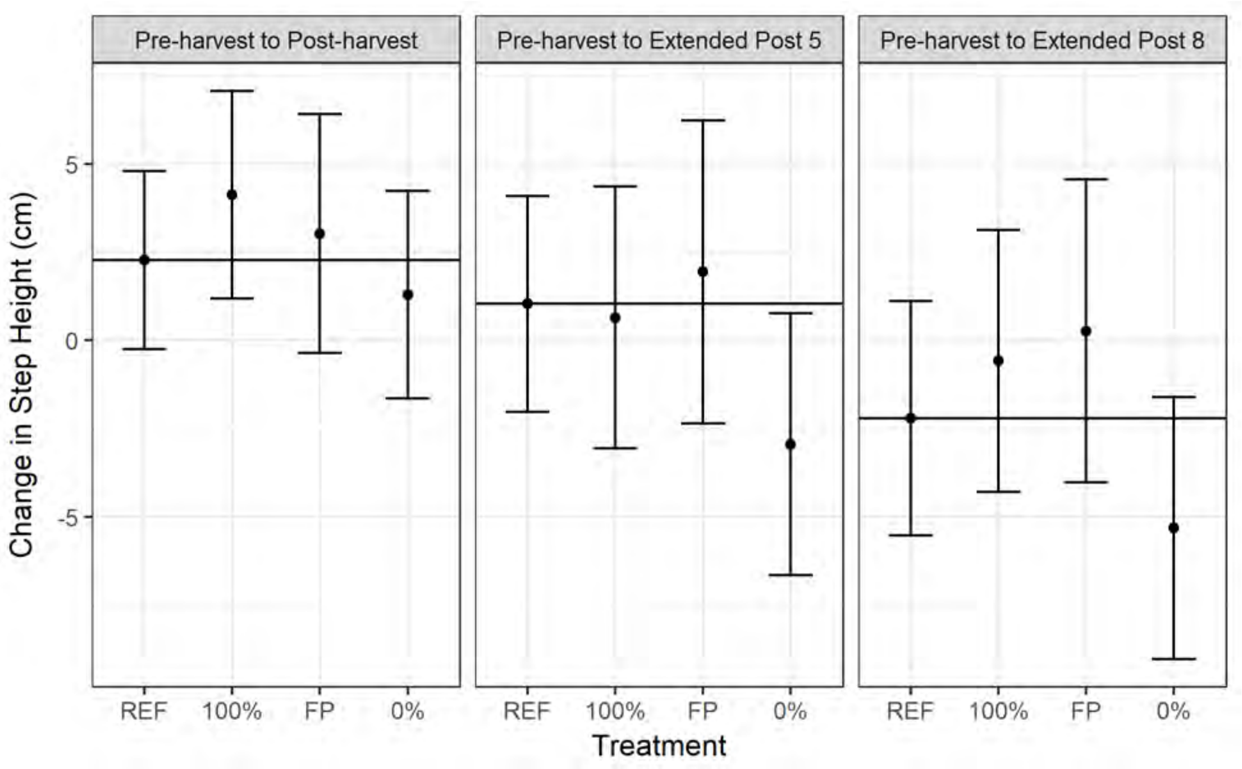

Figure 7-23. The within-treatment estimate of the change and 95% confidence intervals for mean annual step height (cm) between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-26. The between-treatment comparison of the change and 95% confidence intervals (CI) for mean annual step height (cm) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	1.9 (-2, 5.7)	-0.4 (-5.2, 4.4)	1.6 (-3.4, 6.6)
FP vs. REF	0.8 (-3.5, 5)	0.9 (-4.4, 6.2)	2.5 (-2.9, 7.9)
0% vs. REF	-1 (-4.8, 2.9)	-4 (-8.8, 0.8)	-3.1 (-8.1, 1.9)
0% vs. FP	-1.7 (-6.2, 2.8)	-4.9 (-10.6, 0.8)	-5.6 (-11.3, 0.1)
0% vs. 100%	-2.8 (-7, 1.3)	-3.6 (-8.8, 1.7)	-4.7 (-10, 0.5)
FP vs. 100%	-1.1 (-5.6, 3.4)	1.3 (-4.4, 7)	0.8 (-4.8, 6.5)

7-4.9.2. Step Density

Mean annual step density ranged from 0.10 to 0.54 units/m in the pre-harvest period, 0.15 to 0.54 units/m in Post 1 and Post 2, 0.19 to 0.63 units/m in Post 5, and 0.32 to 0.78 units/m in Post 8 (**Figure 7-24**). We did not find clear evidence that step density varied among treatments over time (P = 0.17; **Table 7-27**; **Figure 7-25**; **Table 7-28**).

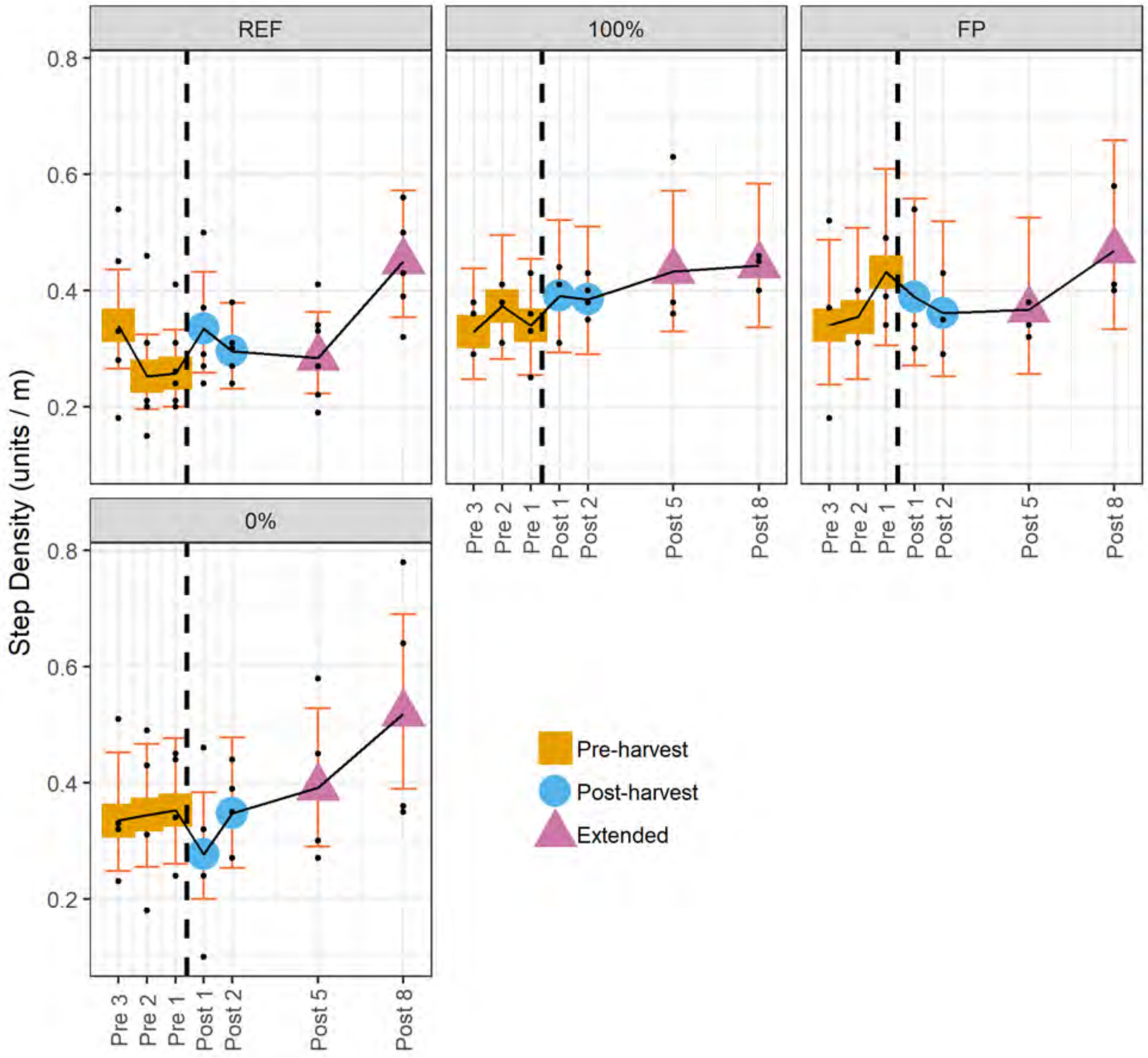

Figure 7-24. Mean step density (units/m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-27. The within-treatment estimate of the relative change and 95% confidence intervals (CI) for mean annual step density between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	**Post 5**	**Post 8**
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.12 (0.96, 1.31)	1.01 (0.84, 1.22)	1.60 (1.33, 1.93)
100% (n = 4)	1.12 (0.95, 1.32)	1.25 (1.03, 1.52)	1.28 (1.05, 1.55)
FP (n = 3)	1.00 (0.80, 1.27)	0.98 (0.73, 1.32)	1.25 (0.96, 1.64)
0% (n = 4)	0.90 (0.74, 1.11)	1.14 (0.90, 1.44)	1.51 (1.21, 1.87)

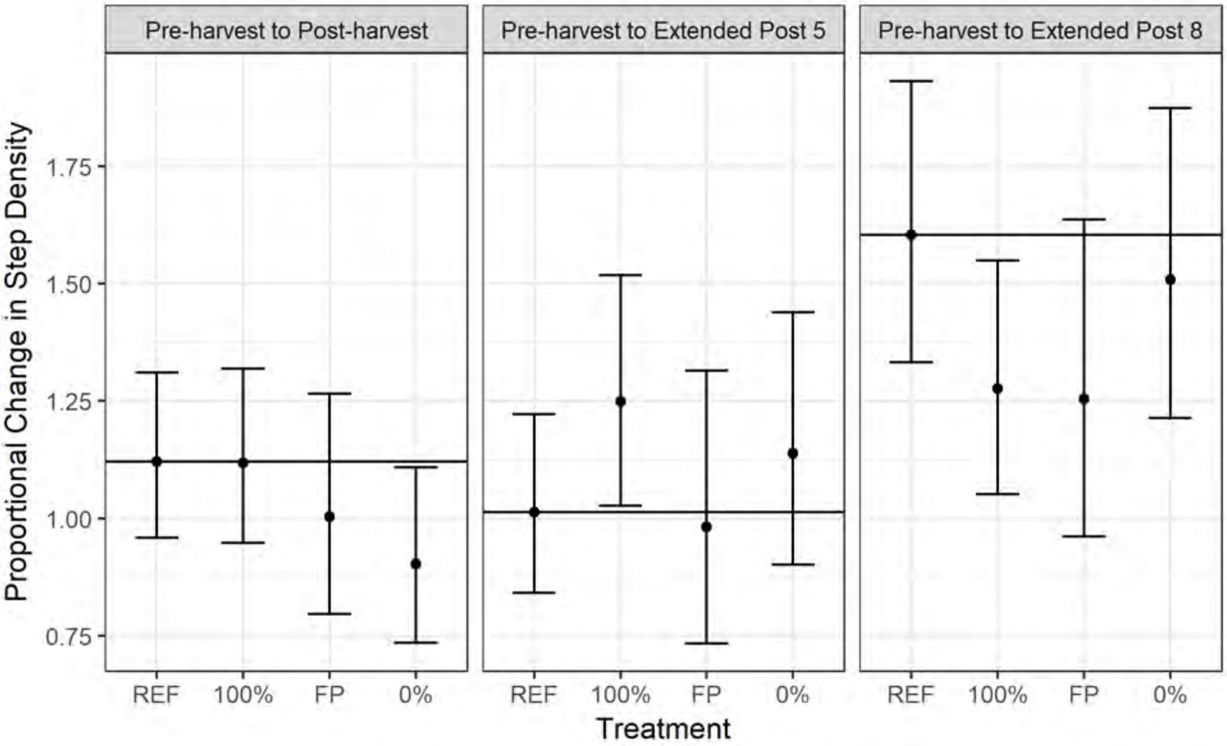

Figure 7-25. The within-treatment estimate of relative change and 95% confidence intervals for mean annual step density between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-28. The between-treatment comparison of the proportional change and 95% confidence intervals (CI) for mean annual step density between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	1.00 (0.79, 1.25)	1.23 (0.94, 1.61)	0.80 (0.61, 1.04)
FP vs. REF	0.90 (0.68, 1.18)	0.97 (0.69, 1.37)	0.78 (0.57, 1.08)
0% vs. REF	0.81 (0.62, 1.04)	1.12 (0.83, 1.52)	0.94 (0.71, 1.25)
0% vs. FP	0.90 (0.66, 1.22)	1.16 (0.80, 1.68)	1.20 (0.85, 1.70)
0% vs. 100%	0.81 (0.62, 1.05)	0.91 (0.67, 1.24)	1.18 (0.88, 1.58)
FP vs. 100%	0.90 (0.68, 1.19)	0.79 (0.55, 1.12)	0.98 (0.71, 1.37)

7-4.9.3. Steps Keyed by Wood

Mean annual proportion of steps keyed by wood ranged from 0.25 to 0.91 in the pre-harvest period, 0.32 to 0.92 in Post 1 and Post 2, 0.36 to 0.81 in Post 5, and 0.36 to 0.91 in Post 8 (**Figure 7-26**). We did not find clear evidence that the proportion of steps keyed by wood varied among treatments over time (P = 0.53; **Table 7-29**; **Figure 7-27**; **Table 7-30**).

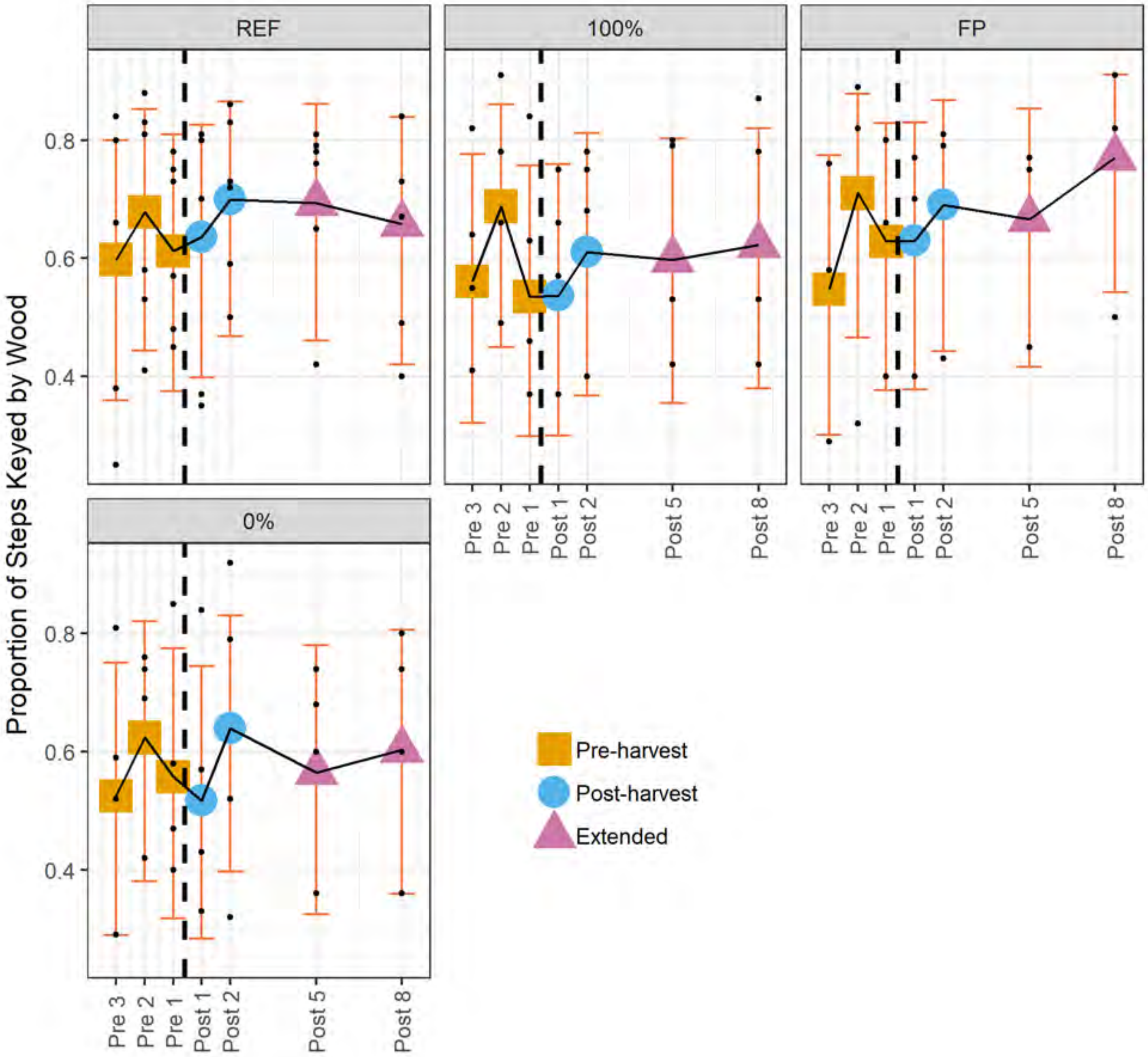

Figure 7-26. Mean proportion of steps keyed by wood by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-29. The within-treatment estimate of odds ratios and 95% confidence intervals (CI) for the change in the mean annual proportion of steps keyed by wood between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	Post 5	Post 8
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.17 (0.91, 1.51)	1.31 (0.96, 1.78)	1.12 (0.80, 1.56)
100% (n = 4)	0.92 (0.68, 1.23)	1.00 (0.69, 1.46)	1.11 (0.77, 1.61)
FP (n = 3)	1.13 (0.81, 1.59)	1.16 (0.75, 1.78)	1.89 (1.23, 2.91)
0% (n = 4)	1.04 (0.78, 1.40)	0.98 (0.68, 1.43)	1.14 (0.79, 1.66)

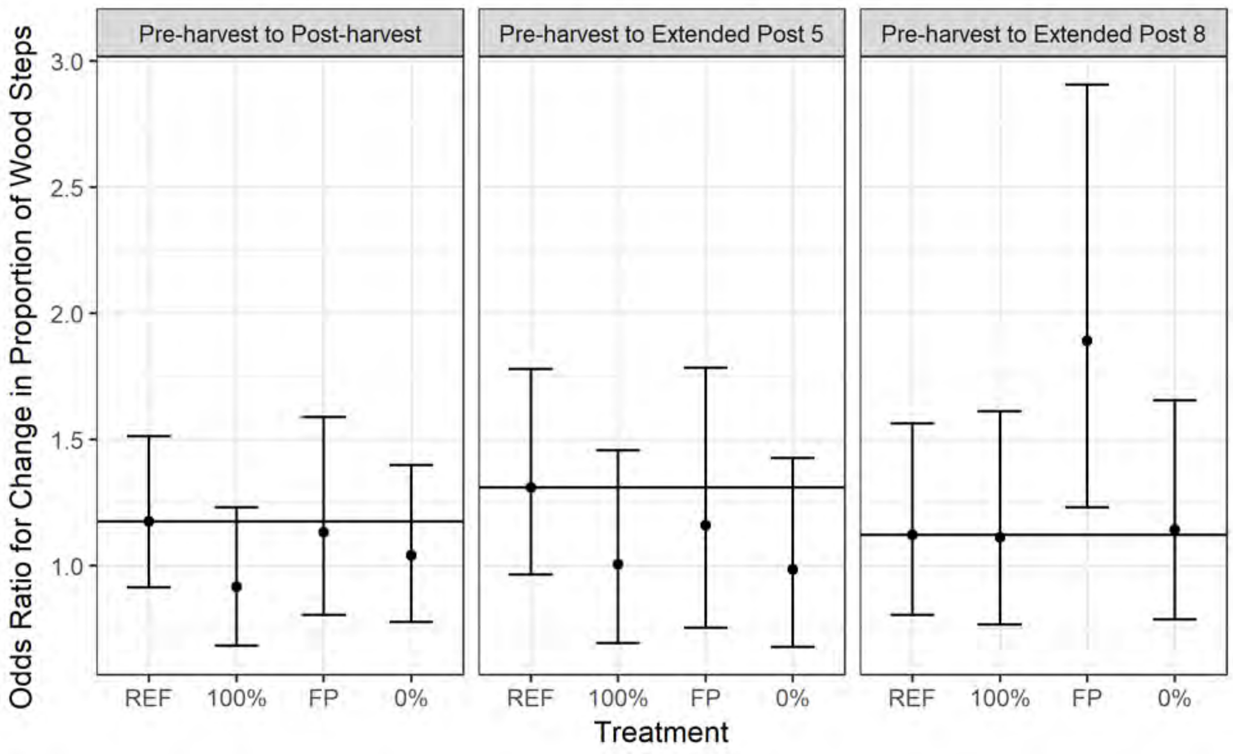

Figure 7-27. The within-treatment estimate of odds ratios and 95% confidence intervals for the change in the mean proportion of steps keyed by wood between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-30. The between-treatment comparison of odds ratios and 95% confidence intervals (CI) for the relative change in the mean annual proportion of steps keyed by wood between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	0.78 (0.53, 1.15)	0.77 (0.47, 1.24)	0.99 (0.60, 1.64)
FP vs. REF	0.96 (0.63, 1.47)	0.89 (0.52, 1.50)	1.69 (0.98, 2.91)
0% vs. REF	0.89 (0.60, 1.31)	0.75 (0.46, 1.22)	1.02 (0.62, 1.68)
0% vs. FP	0.92 (0.59, 1.44)	0.85 (0.48, 1.50)	0.60 (0.34, 1.07)
0% vs. 100%	1.14 (0.75, 1.72)	0.98 (0.58, 1.66)	1.03 (0.61, 1.74)
FP vs. 100%	1.24 (0.79, 1.94)	1.16 (0.65, 2.04)	1.70 (0.96, 3.01)

7-4.9.4. Diameter of Key Pieces in Steps Formed by Wood

Mean annual step key piece diameter ranged from 10 to 26 cm in the pre-harvest period, 10 to 38 cm in Post 1 and Post 2, 13 to 30 cm in Post 5, and 9 to 21 cm in Post 8 (**Figure 7-28**). We did not find clear evidence that step key piece diameter varied among treatments over time (P = 0.55; **Table 7-31**; **Figure 7-29**; **Table 7-32**).

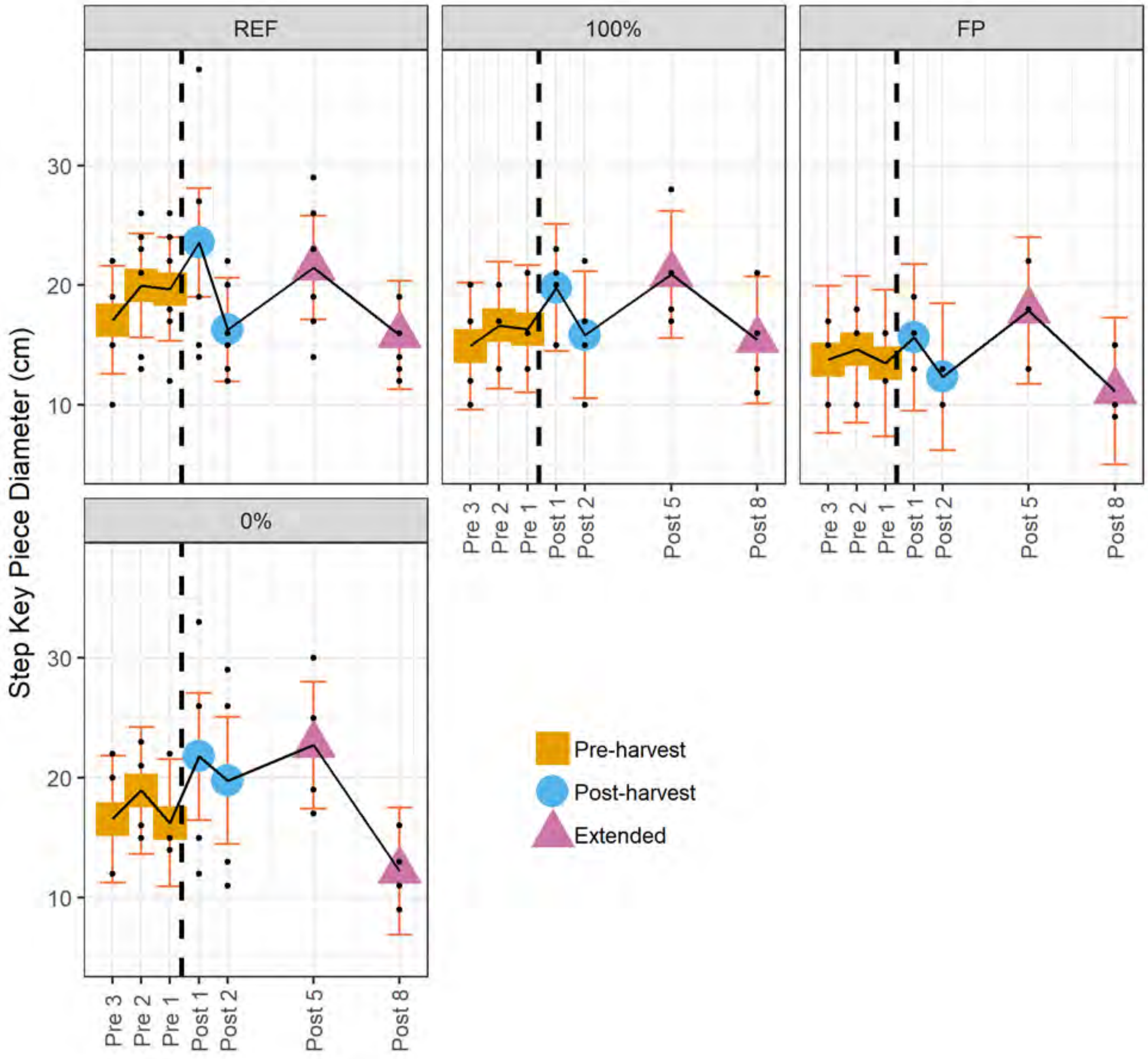

Figure 7-28. Mean step key piece diameter (cm) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-31. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean annual step key piece diameter (cm) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8.

	Post 1 & 2	Post 5	Post 8
Treatment	Estimate (CI)	Estimate (CI)	Estimate (CI)
REF (n = 6)	1.0 (-1.6, 3.6)	2.6 (-0.6, 5.7)	-3.1 (-6.5, 0.4)
100% (n = 4)	1.8 (-1.2, 4.9)	4.9 (1.1, 8.8)	-0.6 (-4.4, 3.3)
FP (n = 3)	0.0 (-3.5, 3.6)	3.9 (-0.5, 8.4)	-2.8 (-7.3, 1.6)
0% (n = 4)	3.5 (0.5, 6.6)	5.5 (1.6, 9.4)	-5.0 (-8.9, -1.2)

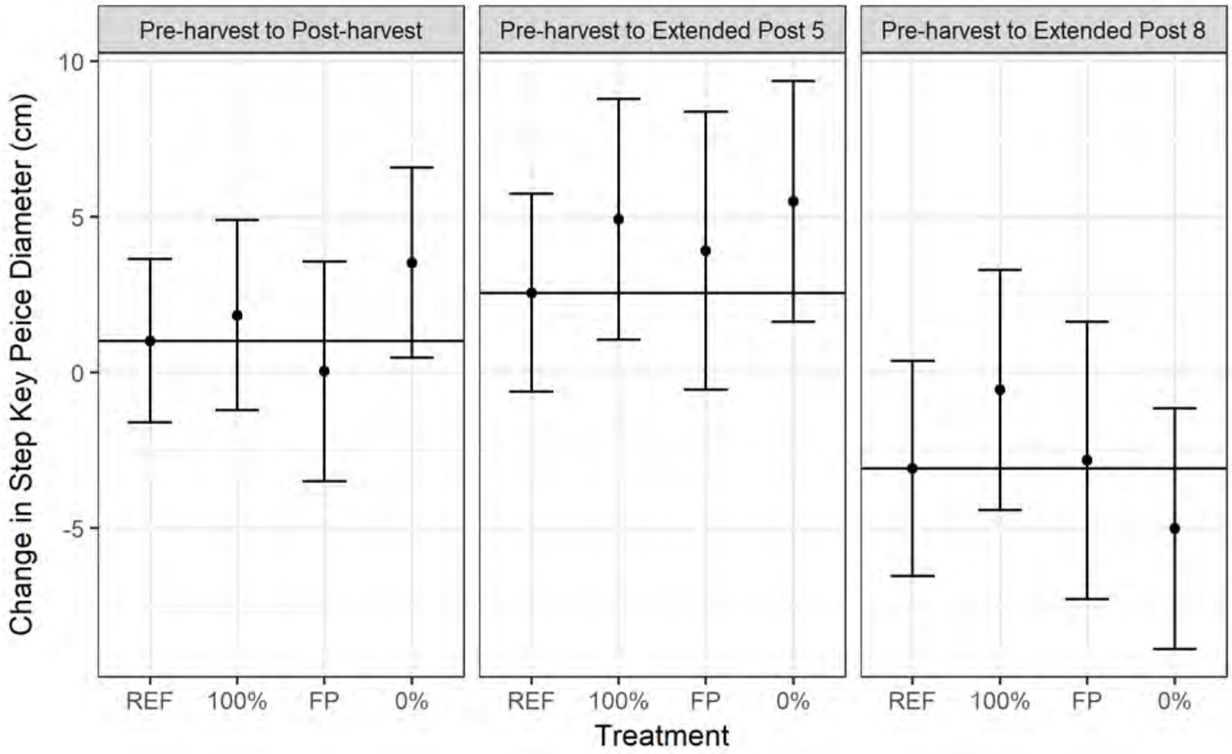

Figure 7-29. The within-treatment estimate of the change and 95% confidence intervals for mean annual step key piece diameter (cm) between the pre-harvest and post-harvest (Post 1 and Post 2), and extended (Post 5 and Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-32. The between-treatment comparison of the change and 95% confidence intervals (CI) for mean annual step key piece diameter (cm) between the pre-harvest period and Post 1 and Post 2, Post 5, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2	**Post 5**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	0.8 (-3.2, 4.8)	2.4 (-2.6, 7.4)	2.5 (-2.7, 7.7)
FP vs. REF	-1.0 (-5.4, 3.4)	1.4 (-4.1, 6.8)	0.3 (-5.4, 5.9)
0% vs. REF	2.5 (-1.5, 6.5)	2.9 (-2.1, 7.9)	-1.9 (-7.1, 3.2)
0% vs. FP	3.5 (-1.2, 8.2)	1.6 (-4.3, 7.5)	-2.2 (-8.1, 3.7)
0% vs. 100%	1.7 (-2.6, 6.0)	0.6 (-4.9, 6)	-4.5 (-9.9, 1.0)
FP vs. 100%	-1.8 (-6.5, 2.9)	-1.0 (-6.9, 4.9)	-2.3 (-8.2, 3.6)

7-4.9.5. Channel Rise Attributed to Steps

Mean annual proportion of channel rise attributed to steps ranged from 0.31 to 0.79 in the pre-harvest period, 0.15 to 0.74 in Post 1 and Post 2, 0.25 to 0.70 in Post 5, and 0.38 to 0.71 in Post 8 (**Figure 7-30**). We found evidence that treatments differed in the magnitude of change over time (P = 0.07; **Table 7-33**; **Figure 7-31**; **Table 7-34**). In the Post 1 and Post 2, we estimated the between-treatment comparison for the 0% treatment and reference to be a 49% (P <0.01) decrease in the mean odds for the proportion of channel rise attributed to steps compared to the pre-harvest period, after controlling for temporal changes in the references. The estimated odds for the 0% treatment was also 48% (P <0.01) and 47% (P <0.01) lower relative to the estimated odds for the 100% and FP treatments, respectively, after adjusting for pre-harvest differences among the treatment sites. In Post 5, we estimated a 34% (P = 0.10) relative decrease in the odds of the proportion of channel rise attributed to steps in the 0% treatment, compared to the pre-harvest period, after controlling for temporal changes in the references. The estimated mean odds in the 0% treatment was also 38% (P = 0.09) less than the estimated odds for the 100% treatment, after adjusting for pre-harvest differences among the treatment sites. In Post 8, we estimated a 39% (P = 0.06) relative decrease in the estimated odds of the proportion of channel rise attributed to steps for the 0% treatment, compared to the pre-harvest period, after controlling for temporal changes in the references. We noted a 66% relative increase in the odds of the proportion of channel rise attributed to steps for reference sites in this same period (**Table 7-33**). The estimated mean odds in the 0% treatment was also 43% (P = 0.06) lower relative to the FP treatment, after adjusting for pre-harvest differences among the treatment sites.

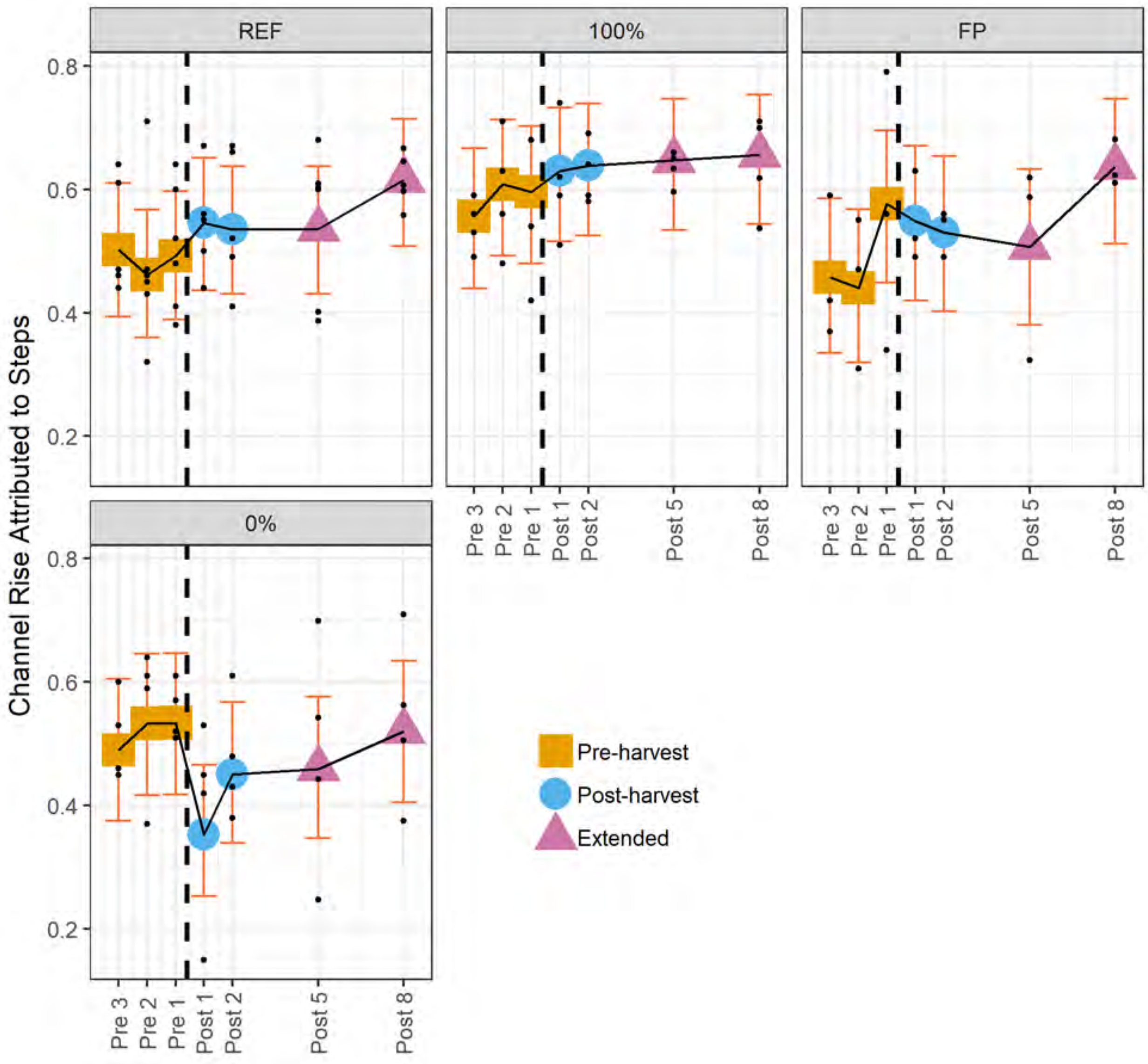

Figure 7-30. Mean proportion of channel rise attributed to steps by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 and Post 2, and extended includes Post 5 and Post 8). Vertical colored lines show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 7-33. The within-treatment estimate of odds ratios and 95% confidence intervals (CI) for the change in the mean annual proportion of channel rise attributed to steps between the pre-harvest period and Post 1 and Post 2, Post 5 and Post 8.

	Post 1 & 2	Post 5	Post 8
Treatment	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**
REF (n = 6)	1.24 (0.95, 1.61)	1.21 (0.88, 1.67)	1.66 (1.18, 2.34)
100% (n = 4)	1.21 (0.89, 1.64)	1.28 (0.87, 1.88)	1.32 (0.90, 1.95)
FP (n = 3)	1.20 (0.84, 1.71)	1.06 (0.68, 1.65)	1.77 (1.13, 2.76)
0% (n = 4)	0.63 (0.47, 0.86)	0.80 (0.54, 1.17)	1.01 (0.68, 1.48)

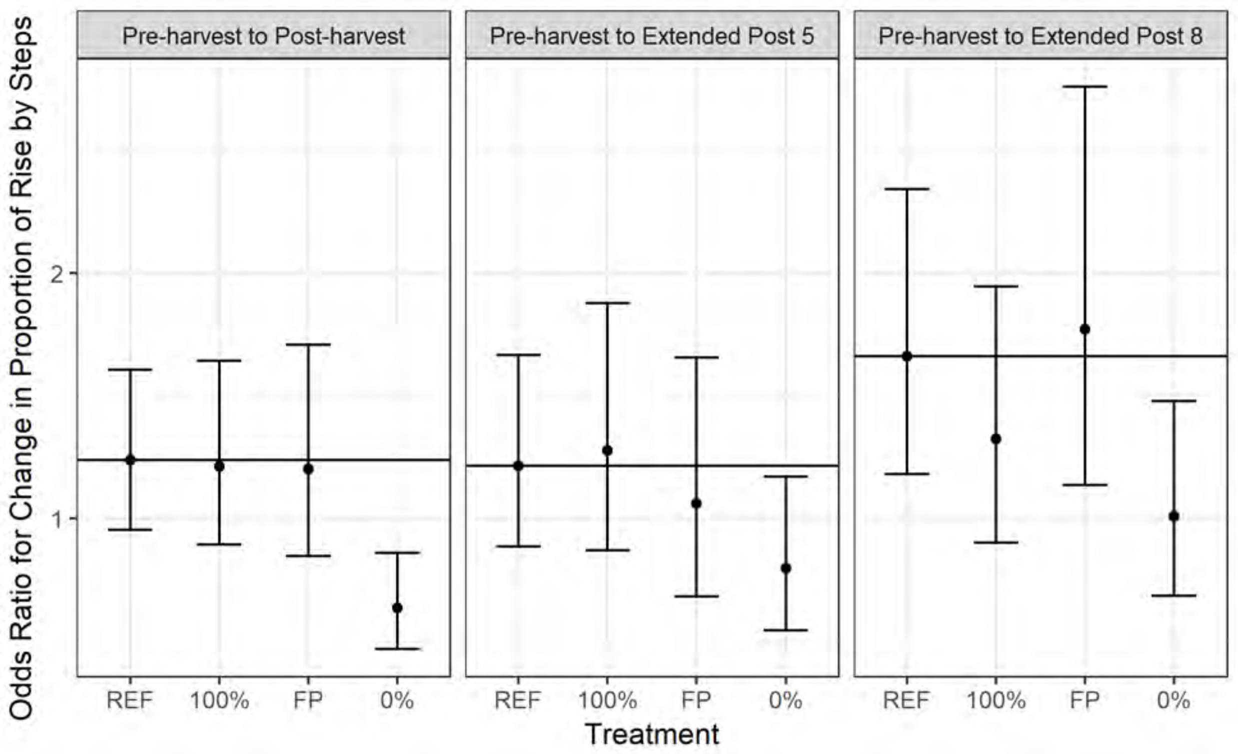

Figure 7-31. The within-treatment estimate of odds ratios and 95% confidence intervals for the change in the mean annual proportion of channel rise attributed to steps between the pre-harvest and post-harvest periods (Post 1 and Post 2), Post 5 and Post 8. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 7-34. The between-treatment comparison of odds ratios, 95% confidence intervals (CI), and P-values (P) for the relative change in the mean annual proportion of channel rise attributed to steps between the pre-harvest period and Post 1 and Post 2, Post 5 and Post 8. Estimates with P-values ≤0.10 are bolded. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	Post 1 & 2		**Post 5**		**Post 8**	
Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
100% vs. REF	0.98 (0.66, 1.46)	0.92	1.05 (0.64, 1.73)	0.84	0.80 (0.48, 1.33)	0.38
FP vs. REF	0.97 (0.63, 1.50)	0.89	0.87 (0.51, 1.51)	0.63	1.07 (0.61, 1.87)	0.82
0% vs. REF	**0.51 (0.34, 0.76)**	**<0.01**	**0.66 (0.40, 1.08)**	**0.10**	**0.61 (0.36, 1.01)**	**0.06**
0% vs. FP	**0.53 (0.33, 0.84)**	**<0.01**	0.75 (0.42, 1.35)	0.33	**0.57 (0.32, 1.02)**	**0.06**
0% vs. 100%	**0.52 (0.34, 0.80)**	**<0.01**	**0.62 (0.36, 1.07)**	**0.09**	0.76 (0.44, 1.31)	0.32
FP vs. 100%	0.99 (0.62, 1.58)	0.97	0.83 (0.46, 1.50)	0.53	1.34 (0.74, 2.41)	0.33

7-5. DISCUSSION

Headwaters in the Pacific Northwest have a high edge-to-area ratio (Gomi *et al.* 2002), are extremely responsive to fluctuations in discharge (Gomi *et al.* 2002; Moore and Wondzell 2005) and wood inputs (Bilby and Bisson 1998), and often do not support continuous annual surface flows (Jackson *et al.* 2001; Olson and Weaver 2007). These characteristics can make headwater streams and associated headwater biota sensitive to the impacts of upland and riparian forest harvest (Richardson and Danehy 2007). Changes to headwater streams may directly impact resident biota or indirectly impact downstream reaches by altering or disrupting the services they provide, such as energy inputs, clean water and habitat connectivity (Wipfli and Gregovich 2002; Freeman *et al.* 2007; Richardson and Danehy 2007; Wipfli *et al.* 2007).

Riparian timber management activities affect structural characteristics of streams by altering short-term wood loading and recruitment potential (Bilby and Ward 1991; Ralph *et al.* 1994; Jackson and Sturm 2002; Hassan *et al.* 2005b), stream flow (Lewis *et al.* 2001; Kuraś *et al.* 2012), and sediment supply and transport capacity (Gomi *et al.* 2005; Kaufmann *et al.* 2009; Bathurst and Iroumé 2014). Our objective was to evaluate the magnitude and duration of change in stream channel characteristics following harvest as they related to wood loading, stream flow and stored sediment.

Wood is a primary determinant of channel form (Swanson and Lienkaemper 1978; Harmon *et al.* 1986; Montgomery and Buffington 1998; Gomi *et al.* 2002), especially in small headwater streams (Bilby and Bisson 1998), which typically lack discharge rates capable of transporting wood out of the system (Keller and Swanson 1979; Gurnell *et al.* 2002). The mechanical role of wood in streams can be broadly categorized as hydraulic alteration, affecting both flow and sediment routing as the result of scour and sediment deposition (Jackson and Sturm 2002). Large wood has been the focus of most studies, but small wood is more abundant than large wood in headwater streams, playing an increasingly important role as channel size decreases (Bilby and Ward 1989; Gurnell *et al.* 2002; Maxa 2009). Clearcut harvest of the streamside forest typically results in large inputs of mostly small wood in the form of logging slash (Jackson *et al.* 2001) and reduces future wood recruitment potential from tree fall (Beechie *et al.* 2000).

Stream flow and hydrology play important roles in creating and maintaining habitats in small headwater streams. Small streams typically lack the fluvial power to export large, inorganic and organic materials (Moore and Wondzell 2005). Previous research on headwater streams in the Pacific Northwest has shown that water yield increases as a result of clearcut harvest (Harr 1986; Hicks *et al.* 1991; Jones and Post 2004; Moore and Wondzell 2005), which has been linked with reductions in evapotranspiration associated with the removal of forested stands (Jones and Post 2004; Brown *et al.* 2005; Moore and Wondzell 2005). Consistent with these studies, we observed changes in the magnitude and frequency of discharge in all treatments, with an increase in water yield in the FP and 0% treatments that continued eight years following harvest (see Chapter 5 – *Stream Discharge, Turbidity, and Suspended Sediment Export* in this report). Increased water yield can impact the transport and storage of sediment and larger substrates from headwater systems, ultimately impacting channel characteristics.

Forest practices can result in increased sediment supply in headwater streams by altering road surface erosion, windthrow, and bank erosion (Roberts and Church 1986; Grizzel and Wolff 1998; Araujo *et al.* 2013). Changes in sediment supply, in combination with changes in the magnitude and timing of water delivery as a result of timber harvest, can affect the frequency and magnitude of sediment transporting events (Gomi *et al.* 2005; Alila *et al.* 2009; Kaufmann *et al.* 2009). We did have some evidence of an increase in sediment storage as reflected by our results for the proportion of the stream length dominated by fines and sand substrates. However, we did not note an increase in sediment transport from study sites (see Chapter 5 – *Stream Discharge, Turbidity, and Suspended Sediment Export* in this report). The lack of a difference in sediment export may be related to the extremely limited sediment supply we observed across all study sites both before and after harvest. Though sediment export increased during large storm events, the timing and magnitude appeared stochastic with respect to harvest timing.

In our evaluation of the response of stream channel characteristics to clearcut harvest with variable length riparian buffers, we observed post-harvest differences in stream wetted and bankfull widths among treatments and references. In the two years post-harvest, the pre- to post-harvest change in wetted width in the 0% treatment averaged 0.3 m less than in the other treatments. Five years post-harvest, the 0% treatment continued to be 0.3 m less than the reference, as it also was in the 100% treatment. Eight years post-harvest, the change in wetted width was 0.4 m less in 0% treatment, but the other buffer treatments did not differ from the reference. We observed significant increases in wood densities in all buffer treatments in the two years post-harvest as a result of inputs from logging slash and windthrow from riparian buffers, with the greatest increase in the 0% treatment (see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report). We continued to observe increased wood loads in the FP and 0% treatments through Post 5. It is possible that increased wood loading affected stream width, with large wood concentrating towards the channel margins. Ralph and colleagues (1994) observed an increase in wood concentrations towards the stream edge in intensively and moderately logged basins compared with unharvested old-growth forests in their study of streams located throughout western Washington. Carlson and colleagues (1990) found that a large proportion of in-channel wood pieces restricted stream wetted width in northeastern

Oregon. Patterns in the change in bankfull width were largely consistent with those observed for wetted width. It is possible that bankfull width was similarly impacted, especially for the 0% treatment, where we observed a decline in bankfull width starting in the two years post-harvest and persisting throughout the study period.

Other research investigating the impacts of timber harvest on stream width has produced mixed results. One study found no changes in wetted stream widths or bankfull widths between logged and uncut stands (O'Connell *et al.* 2000). In contrast, Jackson and Sturm (2002) found that bankfull width increased with an increase in functional wood inputs as a result of harvest, concluding that channels widened as the frequency of wood obstructions increased. While this conclusion is consistent with the idea that wood additions in headwater streams decreases stream velocity and increases stream wetted width (Trotter 1990), our results to not support this finding.

We observed a difference in the pre- to post-harvest change in wetted stream width in the 0% treatment compared to that in the reference, 100% and FP treatments. Interpretation of this result is complicated since the difference in the change was due to a consistent post-harvest increase in the latter treatments, that we did not observe in the 0% treatment. We believe the increase in the reference, 100% and FP treatments was reflective of annual variation that was not realized in the 0% treatment in the post-harvest state. Given the relatively small storage capacity and short flow paths of headwater streams, it is not surprising that wetted stream width would be subject to annual variation. For example, stream flows in headwater streams respond more rapidly to rainfall than streams flows in larger basins (Gomi *et al.* 2002). In a thinning study, Olson and Rugger (2007) also observed changes in instream conditions that they attributed to annual effects rather than treatment effects (i.e., thinning).

We had evidence of an increase in stored fines and sand in streams. We observed increases in the proportion of the stream channel length dominated by fines and sand in the 0% treatment in the two years post-harvest. Five years post-harvest, an increase in the 0% treatment was greater than in the 100% treatment, but did not differ statistically from the reference. Eight years post-harvest, we again observed the increase in the 0% relative to the reference. We also detected a delayed response in the FP treatment, with an increase relative to the reference that was even greater than that for the 0% treatment. Numerous studies have noted an increase in fine sediment in headwater streams following harvest (Bilby and Ward 1989; Corn and Bury 1989; Dupuis and Steventon 1999). The patterns observed by Jackson and colleagues (2001) are similar to our own, with a 32% increase in fine sediment observed for recently clearcut streams without a riparian buffer in the RMZ (similar to our 0% treatment). Jackson and colleagues (2001) attributed their finding to the trapping of fine sediment in wood and logging slash (Lisle 1986; Montgomery *et al.* 1996; Gomi *et al.* 2001). Large wood facilitates sediment deposition (Bilby and Bisson 1998) and creates areas of low energy that slows the transport of sediment and organic material (Heede 1972; Bilby and Ward 1989).

It is possible that the increase we observed in stored fine sediment in some treatments was the result not only of reduced transport capacity as a result of increased wood loading from the adjacent timber harvest, but as a result of increased sediment input related to timber harvest and other management activities. Forest practices have the potential to alter headwater sediment supply by affecting a range of processes including road surface erosion, windthrow, and bank erosion (Roberts and Church 1986; Grizzel and Wolff 1998; Araujo *et al.* 2013). We did not include an evaluation of sediment supply in the current study period, but did evaluate road surface erosion, windthrow, and bank erosion in the pre- and two-year post-harvest periods

(McIntyre *et al.* 2018, Chapter 10 – *Sediment Processes*). We did not find evidence of increased sediment inputs to sites as a result of road surface erosion, windthrow or bank erosion in our evaluation through two years post-harvest.

As a part of our evaluation, we investigated the stream channel rise that could be attributed to steps. Stream adjacent timber harvest has been shown to increase both the amount of wood and sediment stored in small headwater streams. We were interested in whether these increases might fill or bury the stream channel with wood, debris, and organic and inorganic accumulations in the post-harvest period that might obscure small, morphologically distinct features such as steps and pools. In the two years post-harvest, we estimated a decrease in the stream channel rise that could be attributed to steps in the 0% treatment – a decline that persisted eight years post-harvest. We believe this decline was related to the lack of a full riparian buffer in the 0% treatment and unbuffered reaches of the FP treatment. Riparian buffers have been shown to prevent the input of logging debris into the stream channel (Jackson *et al.* 2001; Maxa 2009; Schuett-Hames *et al.* 2012). In the absence of buffers, wood slash from timber harvest accumulated in dense matrices of branches, twigs, and tree tops, essentially filling the stream channel with logging-related slash and burying steps in some reaches. In fact, investigation of wood function in the two years following harvest indicated that wood pieces in slash-filled stream reaches contributed proportionally less to step formation than wood pieces located outside of slash-filled reaches (McIntyre *et al.* 2018, Chapter 6 – *Wood Recruitment and Loading*, Figure 6-17). The effects of riparian harvest in our study conflict with the findings of Jackson and Sturm (2002), who found that channel width decreased as the percent drop in steps increased, whereas in our study, we found that the channel width and percent drop of steps decreased in the 0% treatment following harvest.

Though we measured 15 stream channel characteristics, we detected significant treatment effects for only four characteristics. We recognize that with our number of statistical comparisons (n = 15), and especially with an alpha of 0.1, we could expect to have several “statistically significant” results based on chance alone. However, there was a consistency in our results through time and among treatments, giving us confidence that significant differences among treatments in the post-harvest and extended periods were caused by activities associated with forest harvest. One caveat is for the response of stored fine sediments, for which results were marginal. However, trends in results were somewhat consistent for all post-harvest periods, especially for the 0% treatment, which provides some confidence for this response. Furthermore, our intensive sampling effort also gives us confidence in our results, with ≥50% of the mainstream channel represented at each study site. We are least confident in our results for stream bankfull width, which is innately difficult to accurately measure (Platts *et al.* 1983), a fact evidenced by the variability even in the references, and made even more difficult when attempting to locate streambanks that were frequently buried under logging slash and windthrow in buffer treatment sites.

Study sites were impacted variably by the region-wide windthrow event that occurred in December of 2007, prior to treatment implementation (see Chapter 2 – *Study Design* in this report). Recruitment of wood to the stream channel in the affected sites, which included both references and future buffer treatment sites, resulted in a large increase in wood loading immediately prior to harvest, likely effecting future wood recruitment and loading. Fortunately, the windthrow event reflects the natural variability that occurs throughout western Washington. Furthermore, we detected treatment effects despite the pre-harvest windthrow event, in part because all sites in the windthrow-affected areas were affected, including sites in all treatments. Additionally, the most affected sites were grouped geographically into blocks for analysis. However, the timing of the windthrow event should be considered when interpreting the magnitude of differences and may have affected our ability to distinguish differences among buffer treatments.

The inclusion of channel responses in this study was intended to help us refine our description of harvest effects on the availability and quality of habitats for stream-associated amphibians. Characteristics of headwater streams, such as the high edge-to-area ratio (Gomi *et al.* 2002), responsiveness to changes in discharge and wood input (Bilby and Bisson 1998; Moore and Wondzell 2005), and discontinuous annual surface flows (Jackson *et al.* 2001; Jaeger *et al.* 2007; Olson and Weaver 2007), may make small streams especially sensitive to the impacts of harvest of the riparian forest (see Richardson and Danehy 2007). Though we saw harvest changes to stream wetted and bankfull widths, proportion of fines and sand, and the channel rise attributed to steps, especially in the 0% treatment, these changes did not result in measurable changes in any of our stream channel unit metrics, including density of cascades, riffles, pool or steps, or measures associated with individual units, such as length or depth. Declines in bankfull and wetted widths in the 0% treatment may translate into a decline in available surface amphibian habitat. Furthermore, the increased incidence of sand and fines in the FP and 0% treatments has the potential to eliminate critical microhabitats through the filling of interstitial spaces (Murphy and Hall 1981; Corn and Bury 1989; Welsh and Ollivier 1998). For Coastal Tailed Frog larvae which graze mainly on diatoms growing on rock surfaces (Altig and Brodie 1972; Nussbaum *et al.* 1983), sedimentation may impact grazing, negatively effecting the quality of amphibian habitat.

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CHAPTER 8 - STABLE ISOTOPES

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8-1. ABSTRACT

Canopy modification along forested streams has been associated with an increase in the contribution of algae to the trophic support of the system. However, the effect of canopy modification on food webs of very small, fishless streams has not been thoroughly evaluated. Stable isotope ratios are especially useful for identifying shifts in trophic system organization due to canopy modification. The carbon (^{13}C) and nitrogen (^{15}N) isotopic signatures of algae can differ from those of terrestrially-derived organic matter, enabling determination of the relative contributions of each to the diet of primary consumers in aquatic systems. We compared the response of stable isotopes to clearcut timber harvest in a long-term (2006–2016), spatially blocked and replicated Before-After Control-Impact (BACI) study in western Washington. We included three alternative riparian buffer treatments and reference sites that were not harvested. Buffer treatment basins were clearcut outside of a riparian management zone (RMZ) and received one of three riparian buffer treatments: 100% treatment (two-sided 50-ft width riparian buffer along the entire length of the Type Np stream network), Forest Practices (FP) treatment (two-sided 50-ft width riparian buffer along at least 50% of the Type Np stream network, according to current Forest Practices Rules), and 0% treatment (harvested to the stream edge with no riparian buffer). We collected samples of organic matter sources (biofilm, and coniferous, deciduous, and wood litterfall and instream detritus), macroinvertebrates (aquatic gatherers and shredders), and stream-associated amphibians (Coastal Tailed Frog, and giant and torrent salamanders) for stable isotope analysis of C and N isotope ratios in up to 17 study sites in up to three years preceding harvest (Pre 3, 2 & 1), two years immediately following harvest (Post 1 & 2), and eight years after harvest (Post 8). In our BACI analysis of stable isotope response we evaluated whether variable length riparian buffer treatments caused changes in the primary energy source supporting food webs in small streams. A secondary focus was to identify important food items for the amphibians in these systems, for which a subset of metrics was evaluated only in Post 8, including two types of terrestrial invertebrates that were not previously sampled (springtails and spiders).

We had no evidence of a change for C or N isotope ratios for many responses and periods. However, we did observe a response for $\delta^{13}C$ isotope ratios for biofilm in Post 8, with an estimated $4.23^{o}/_{oo}$ and $5.20^{o}/_{oo}$ mean increase in the 100% and FP treatments, respectively, compared to the pre-harvest period after controlling for temporal changes in the reference. However, we detected no change in the 0% treatment and variability in biofilm isotope values likely reflected spatial and temporal differences in biofilm composition. Furthermore, the similarity in $\delta^{13}C$ values for biofilm and the three types of leaf litter and instream detritus sampled suggests that the biofilm derives primarily from terrestrial material.

We estimated a $1.84^{o}/_{oo}$ decrease in $\delta^{15}N$ for aquatic invertebrate gatherers in the FP treatment in Post 1 & 2, and a $1.63^{o}/_{oo}$ increase in the 0% treatment in Post 8, compared to the pre-harvest period after controlling for temporal changes in the reference. However, in Post 1 & 2 there was no relationship between the length of the riparian buffer retained in the RMZ and isotope values and the Post 8 increase in the 0% treatment is unlikely to be due to an increase in autotrophic production since $\delta^{15}N$ values for algae are typically higher than those for terrestrial organic matter.

Due to small sample sizes for Coastal Tailed Frog pre-harvest and in Post 1 & 2, we could only evaluate between-treatment differences in our Post 8 comparison for this taxon. For our remaining salamander taxa, we had clear evidence of a treatment response for giant salamanders only. We estimated a 1.14‰ increase and a 1.46‰ decrease in mean $\delta^{13}C$ for giant salamander larvae in Post 1 & 2 in the 100% and FP treatments, respectively. For neotenes, we estimated a 1.31‰ and 1.44‰ increase in mean $\delta^{13}C$ and $\delta^{15}N$, respectively, in the FP treatment in Post 1 & 2. However, by Post 8 we no longer had clear evidence of variation among treatments for any comparisons. In our Post 8 comparison of Coastal Tailed Frog, we estimated that $\delta^{15}N$ for larvae was 2.56 ‰ and 2.20 ‰ higher in the 100% and 0% treatments, respectively, relative to the reference. We also estimated that mean $\delta^{13}C$ for larvae was 5.20 ‰ and 5.63 ‰ higher in the 100% and 0% treatments, respectively, than in the reference, and mean $\delta^{13}C$ for post-metamorphic frogs was 2.84‰ lower in the FP treatment than in the reference.

We found limited and inconsistent differences in treatments; however, the stable isotope signals suggested that the organic matter sources supporting the biofilm were not appreciably changed because of buffer treatments. Though we did not detect a notable difference in the biofilm isotopic values between the pre- and post-harvest period, we did observe a decrease in mean $\delta^{13}C$ for giant salamander larvae in the FP treatment and an increase in the 100% treatment in Post 1 & 2. Over this same period, we observed a decrease in mean $\delta^{15}N$ for gatherer invertebrates in the FP and 0% treatments. However, because we found no evidence of an increase in algal content in the biofilm, these changes did not support our hypothesis that canopy modification resulted in increased trophic support from autotrophic sources. The $\delta^{13}C$ versus $\delta^{15}N$ comparison of stable isotope data for Post 8 provided us with some indication of the stream-associated amphibian diet. Results indicated that Coastal Tailed Frog larvae were ingesting primarily biofilm. The post-metamorphic Coastal Tailed Frogs, torrent salamanders and giant salamanders all exhibited stable isotope values that suggested a diet of aquatic predators and shredders, and terrestrial spiders. Coupling stable isotope analysis with a direct evaluation of amphibian diets in future studies would provide a more definitive characterization of trophic relationships of these animals.

8-2. INTRODUCTION

Timber harvest has the potential to influence food web organization. An increase in light from the removal or thinning of riparian vegetation often results in an increase in instream algal production (Murphy *et al.* 1981), which is associated with an increase in scraper invertebrates that ingest algae (Newbold *et al.* 1980; Hawkins *et al.* 1982). Reduction in riparian vegetation also decreases input of leaves, needles, and other terrestrial organic matter (Bilby and Bisson 1992). Vertebrate animals such as fish and stream-associated amphibians may also respond to reductions in riparian vegetation with an increase in growth rate, production, and/or density (Murphy and Hall 1981; Bilby and Bisson 1992; Wilzbach *et al.* 2005; Kiffney *et al.* 2014; Kaylor and Warren 2017).

The use of stable isotopes, especially of carbon (^{13}C) and nitrogen (^{15}N), has enhanced the understanding of food web organization due to the difference in isotopic signature among energy sources (Peterson and Fry 1987; see McIntyre et al. 2018, Supplement 2 – Stable Isotopes Analysis). For example, terrestrial organic matter typically contains lower proportions of the heavier ^{13}C than aquatic algae. This difference in ^{13}C isotope values enables the determination of the relative contribution of each energy source to the diet of primary consumers in aquatic ecosystems (Keough *et al.* 1996). However, because streambed biofilm is formed from a combination of terrestrial and algal organic matter, there can be large variations in the ^{13}C content of biofilm among streams (France 1995). In some systems, there is considerable overlap between terrestrial and biofilm ^{13}C levels, making determination of the relative role each carbon source makes to the support of higher trophic levels impossible. ^{15}N stable isotopes can be used in conjunction with ^{13}C to partially address this problem (Mulholland *et al.* 2000), with ^{15}N levels typically higher in algae than terrestrial organic matter. The fact that the heavier isotope of ^{15}N increases in concentration with each trophic exchange also makes ^{15}N isotopes useful for determining the trophic level of an organism (DeNiro and Epstein 1978). In larger, fish-bearing streams, stable isotope analysis has demonstrated that an increase in light can increase the proportion of algae in streambed biofilm sufficiently to affect isotope values (Mulholland *et al.* 2000).

Stable isotope analysis was included in this study as a method of evaluating whether variable length riparian buffer treatments caused changes in the primary energy source supporting food webs in small streams. A secondary focus was to identify important food items of the amphibians in these systems. The controlling influence of light on in-channel primary production (Kiffney *et al.* 2004; Julian *et al.* 2011; Warren *et al.* 2017) and the effect that increased primary production can have on higher trophic levels (Hill *et al.* 1995; Quinn *et al.* 1997; Kaylor *et al.* 2017) has been the focus of numerous studies. A consistent increase in primary and secondary production has been reported with increased light in stream channels large enough to support fish (Murphy and Hall 1981; Bilby and Bisson 1992; Kaylor and Warren 2017). Though most research on trophic response to canopy modification has focused on fish-bearing streams and not smaller, headwater systems, we hypothesized that small streams are also likely to experience changes in trophic organization related to canopy modification following timber harvest (Kaylor *et al.* 2017). We used stable isotope analysis to evaluate whether different riparian buffer treatments caused changes in the stable isotope values of the primary energy sources supporting food webs of the small streams. We also used the stable isotope values obtained from invertebrates and amphibians to identify potential food sources of Coastal Tailed Frogs, torrent salamanders and giant salamanders.

8-3. METHODS

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 stream food webs to riparian canopy modification among reference and treatment sites in a BACI-designed study (see Chapter 2–*Study Design* in this report). We compared conditions 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), Forest Practices (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 to the stream edge with no riparian buffer; n = 4).

A focus of our effort in Post 8 included the desire to generate a better understanding of the dietary habits of amphibians in small, headwater streams. To do this, we focused on increasing our sample sizes for Coastal Tailed Frog, which was under-represented in previous sample years.

In addition, we observed a difference in isotope values between the invertebrate groups and stream-associated amphibians sampled in previous years, indicating that we did not sample the full range of food resources consumed by the amphibians. Therefore, we hoped to better represent the full range of possible diet items for amphibians by modifying our sampling effort to include additional aquatic and terrestrial invertebrates.

8-3.1. SAMPLE COLLECTION

Sample collection for biofilm followed the methods of McIntyre and colleagues (2018, Supplement 2 – *Stable Isotopes Analysis*) that we used in three pre-harvest and two post-harvest (Post 1 and Post 2) years. We used a wire brush to scrub biofilm off cobbles (64–256 mm) collected from the streambed within the reach where we sampled drift near the downstream end of our study sites. Generally, sufficient organic matter was produced by scrubbing fewer than five cobbles. We rinsed the scrub brush and cobbles with stream water into a wide-mouth plastic jar. We then poured the water and biofilm mixture from the jar into a labeled sampling container and stored the container on ice during transport to the lab, where we froze the sample pending processing.

We collected samples for litterfall, instream detritus, and macroinvertebrates at locations downstream of the treated area in riparian buffer treatments sites and near the fish end point (i.e., Type F/N break) in the references. Conifer, deciduous, and wood litterfall samples were collected directly from the streambank. We used a D-frame kick net to obtain benthic detritus and macroinvertebrates samples from riffles. This sampling method enabled us to collect samples for aquatic macroinvertebrate feeding groups that we either did not detect with the drift net collection we used previously (reported in McIntyre *et al.* 2018, Chapter 14 – *Macroinvertebrate Export*) or did not obtain in sufficient numbers to allow for analysis. We collected instream conifer, deciduous, and wood detritus from the benthic samples, and selected one or two aquatic macroinvertebrate taxa representing each of the main feeding groups. The feeding groups used consisted of gatherers (mayflies; *Baetis* [Ephemeroptera: Baetidae] and *Paraleptophlebia* [Ephemeroptera: Leptophlebiidae]) and shredders (Capniidae/Leuctridae [Plecoptera]), which were included in the pre- and post-harvest analysis through Post 2, and predators (stoneflies; Chloroperlinae [Plecoptera: Chloroperlidae]) and scrapers (Heptageniidae [Ephemeroptera] and *Ironodes* [Ephemeroptera: Heptageniidae]), which were added in Post 8. In addition, we collected forest duff and soil samples for terrestrial gatherers (springtails; Collembola) and predators (spiders; Araneae). We did not separate spiders into taxonomic categories, limiting our ability to conclude anything about their diet. Aquatic and terrestrial invertebrates from each study site were composited by taxa to ensure sufficient weight for stable isotope analysis. While sampling in the through Post 2 was restricted to the Olympic and Willapa 1 blocks (i.e., those in which we conducted drift sampling), the kick net sampling we used in Post 8 allowed us to expand our litterfall, instream detritus and macroinvertebrate sampling efforts to include the Willapa 2 block.

We collected tissue samples from three stream-associated amphibian genera: Coastal Tailed Frog (*Ascaphus truei*), torrent salamanders (Olympic *R. olympicus,* Columbia *R. kezeri* and Cascade *R. cascadae*) and giant salamanders (Cope's *Dicamptodon copei* and Coastal *D. tenebrosus*). We sampled for amphibians as a part of our amphibian response evaluation using standard light-touch (Lowe and Bolger 2002) and rubble-rouse (Bury and Corn 1991) sampling methods (see Chapter 9 – *Stream-Associated Amphibians* in this report). During light-touch sampling samplers moved upstream, sampling within the ordinary high-water mark (WFPB 2001), turning all moveable surface substrates small cobble-sized or larger (≥64 mm) and visually searching for amphibians. During rubble-rouse sampling, we blocked off a section of the stream with downstream and upstream nets, removed all coarse substrate large gravel-sized or larger (≥32 mm diameter) from within the wetted channel to a depth of 30 cm or until bedrock was reached, and sifted the remaining unconsolidated fine substrates in search of animals.

For each stream-associated amphibian taxa, we collected tissue from two developmental stages, or size classes: larvae or post-metamorphs for each of Coastal Tailed Frog and torrent salamander; and larvae (i.e., ≤50 mm snout-vent length) and neotene (i.e., >50 mm snout-vent length) for giant salamanders. Our goal was to collect samples from 10 individuals for each taxon, life stage, study site, and year. We collected tail tissue from all salamanders and Coastal Tailed Frog larvae and toe clips from post-metamorphic Coastal Tailed Frogs. We attempted to collect samples that were approximately 0.5 cm^2 in size; however, this was not possible for smaller animals or metamorphosed frogs. We used sterilized dissecting scissors to remove tissue and placed samples in 1.5-ml sample vials on ice for transport from the field to the lab, where they were immediately placed in a freezer. Unlike sampling for litterfall, instream detritus and macroinvertebrates, sampling for amphibians was conducted at all study sites and blocks. Unfortunately, limited samples for Coastal Tailed Frog through Post 2 prevented inclusion of this species in the analysis presented in McIntyre and colleagues (2018; Supplement 2 – *Stable Isotopes Analysis*). In Post 8, we made an effort to increase sample sizes for Coastal Tailed Frog. For study sites where our minimum sample size was not obtained with our standard amphibian sampling (see Chapter 9 – *Stream-Associated Amphibians* in this report), we conducted additional sampling that included nocturnal light-touch, diurnal light-touch, rubble-rouse and kick sampling (Arkle and Pilliod 2010). Unfortunately, even with our additional effort, we did not meet our goal of 10 samples for each taxa and life stage at all sites in Post 8. Through Post 2, amphibian samples from each site were composited by taxa to ensure sufficient weight for stable isotope analysis. In Post 8, we did not composite samples, but determined the stable isotope value for each individual sampled. Nonetheless, it was the mean basin level value that we used in the BACI test for a difference among treatments.

Since not all groups were sampled in all years, not all were available for inclusion in the BACI analysis of stable isotopes. See **Table 8-1** for the groups that were included in the BACI analysis. See **Table 8-2** for the groups that were included only in the Post 8 comparison among treatments.

Table 8-1. The sample size for each group included in the BACI stable isotopes analysis for each year sampled.

Sample Year	REF	100%	FP	0%
Biofilm				
Pre 3	2	2	2	2
Pre 2	1	1	2	2
Pre 1	1	1	2	2
Post 1	2	2	2	2
Post 2	2	2	2	2
Post 8	3	3	2	3
Macroinvertebrate Gatherers				
Pre 3	1	2	2	2
Pre 2	2	2	2	1
Pre 1	2	2	2	2
Post 1	2	2	2	2
Post 2	2	2	2	2
Post 8	3	3	2	3
Macroinvertebrate Shredders				
Pre 3	2	1	2	2
Pre 2	2	0	2	2
Pre 1	1	1	1	2
Post 1	1	1	2	2
Post 2	2	1	2	1
Post 8	3	3	2	3

Sample Year	REF	100%	FP	0%
Torrent Salamander Larvae				
Pre 2	6	4	3	4
Post 1	5	4	2	3
Post 2	6	4	3	4
Post 8	5	4	3	4
Torrent Salamander Post-metamorphs				
Pre 2	5	4	3	3
Post 1	5	4	2	2
Post 2	6	4	3	3
Post 8	5	4	3	3
Giant Salamander Larvae				
Pre 2	6	3	2	4
Post 1	5	4	3	4
Post 2	6	4	3	4
Post 8	5	4	3	4
Giant Salamander Neotenes				
Pre 2	6	3	2	4
Post 1	5	4	3	4
Post 2	6	4	3	4
Post 8	5	4	3	4

Table 8-2. The sample size for each group for which only a Post 8 comparison among treatments was possible due to a lack of samples in other sample years.

Group	Treatment	Ref	100%	FP	0%	Total
Coastal Tailed Frog	Larvae	5	3	1	3	12
	Post-metamorphs	5	4	1	2	12
Aquatic invertebrates	Predator (stoneflies)	3	3	2	3	11
	Gatherer (mayflies)	3	3	2	3	11
	Scraper	3	3	2	3	11
	Shredder	3	3	2	3	11
Terrestrial invertebrates	Predator (spiders)	3	3	2	3	11
	Gatherer (springtails)	3	3	2	3	11
Instream detritus	Coniferous	3	3	2	3	11
	Deciduous	1	3	1	3	8
	Wood	3	3	2	3	11
Litterfall	Coniferous	3	3	2	3	11
	Deciduous	3	3	2	3	11
	Wood	3	3	2	3	11
Biofilm		3	3	2	3	11

8-3.2. LABORATORY ANALYSIS

Sample processing for stable isotope analysis consisted of thawing samples and rinsing them with 10% hydrochloric acid solution on a glass fiber filter with a nominal pore size of 1.2µm over a vacuum aspirator. We then wrapped the sample in the filter and placed the sample and filter in a sterilized shell vial, and the shell vial into a sterilized glass scintillation vial containing desiccant. The vials were oven dried at 60° C for at least two weeks and stored in a desiccation chamber until shipped for analysis.

We contracted with the Cornell University Stable Isotope Laboratory for analysis of ^{13}C and ^{15}N isotopes. Lab personnel removed the samples from the filters, ground the samples with mortar and pestle, and weighed the samples into 4 × 6 mm tin capsules using a Sartorius MC5 scale (K. Sparks, personal communication). The lab's target mass is 3 mg ± 0.2 for leaf and litter material and 1 mg ± 0.1 for all other material. The samples were analyzed on a CarloErba 2500 elemental analyzer coupled to a ThermoScientific Delta V isotope ratio mass spectrometer.

8-3.3. STATISTICAL ANALYSIS

The BACI analyses evaluated the generalized null hypothesis:

$$\Delta T_{\text{REF}} = \Delta T_{100\%} = \Delta T_{\text{FP}} = \Delta T_{0\%} \qquad \textbf{(Eq. 8-1)}$$

where ΔT_{REF} is the change (post-harvest - pre-harvest) in the reference, and $\Delta T_{100\%}$, ΔT_{FP}, and $\Delta T_{0\%}$ are the changes in the 100%, FP and 0% buffer treatments, respectively.

We used a generalized linear mixed effects model to evaluate this pre- versus post-harvest hypothesis. In this model, block and site were random effects and the fixed effects were period (pre- and post-harvest), treatment, and the treatment × period interaction. We evaluated the null hypothesis with a Wald-type test using linear contrasts of the model fixed effects. If the treatment × period contrast had a P-value ≤0.010 we examined pairwise contrasts to determine whether a difference existed in this term for all combinations among references and treatments: REF vs. 0%, REF vs. FP, REF vs. 100%, 0% vs. FP, 0% vs. 100%, FP vs. 100% and pre-post treatment comparison of each treatment type. The uneven distribution of treatments among blocks required utilizing the Kenward-Roger method (Kenward and Roger 1997) for estimating the denominator degrees of freedom. We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals and found no evidence of heteroscedasticity.

For those groups sampled only in Post 8 (Post-harvest Post 8 Treatment Comparison), we present a statistical comparison of δ^{13}C and δ^{15}N between treatments using a linear mixed effects model. We modeled basin-average isotope Post 8 values with fixed-effects terms for treatment and a random block effect.

$$y_{ijk} = \mu + \beta_1 Trt_{j\in 0\%} + \beta_2 Trt_{j\in 100\%} + \beta_3 Trt_{j\in FP} + \gamma_k + \varepsilon_{ijk} \qquad \textbf{(Eq. 8-2)}$$

Where y_{ijk} is the isotope measurement from basin j of block k, the β parameters represent the contrasts between the individual buffered and reference treatments, and γ_k is a random intercept adjustment for block k where we assume $\gamma_k \sim N(0, \sigma_\gamma^2)$ and $\varepsilon_{ijk} \sim N(0, \sigma_e^2)$. An overall F-test was used to assess if there was evidence of a mean difference among any of the treatments. If there was evidence of a difference among treatments, pairwise comparisons were evaluated for evidence of differences between specific pairs of treatments.

8-4. RESULTS

8-4.1. BACI ANALYSIS

8-4.1.1. *Biofilm*

We did not find clear evidence that mean $\delta^{15}N$ varied among treatments for biofilm (pre-harvest to Post 1 and Post 2, P = 0.30; pre-harvest to Post 8, P = 0.59). While we did not find clear evidence that mean $\delta^{13}C$ varied among treatments in Post 1 and Post 2 (P = 0.11), we did find evidence that $\delta^{13}C$ varied among treatments in Post 8 (P = 0.02). In Post 8, we estimated a 4.23$^{o}/_{oo}$ (P = 0.04) and 5.20$^{o}/_{oo}$ (P = 0.01) mean increase in $\delta^{13}C$ in the 100% and FP treatments, respectively, compared with pre-harvest, after controlling for temporal changes in the references. We also estimated a 3.57 $^{o}/_{oo}$ (P = 0.06) and 4.54 $^{o}/_{oo}$ (P = 0.02) decrease in the 0% treatment compared with the 100% and FP treatments, respectively, after adjusting for pre-harvest differences among the treatment sites. See **Table 8-3**, **Table 8-4, Figure 8-1** and **Figure 8-2**.

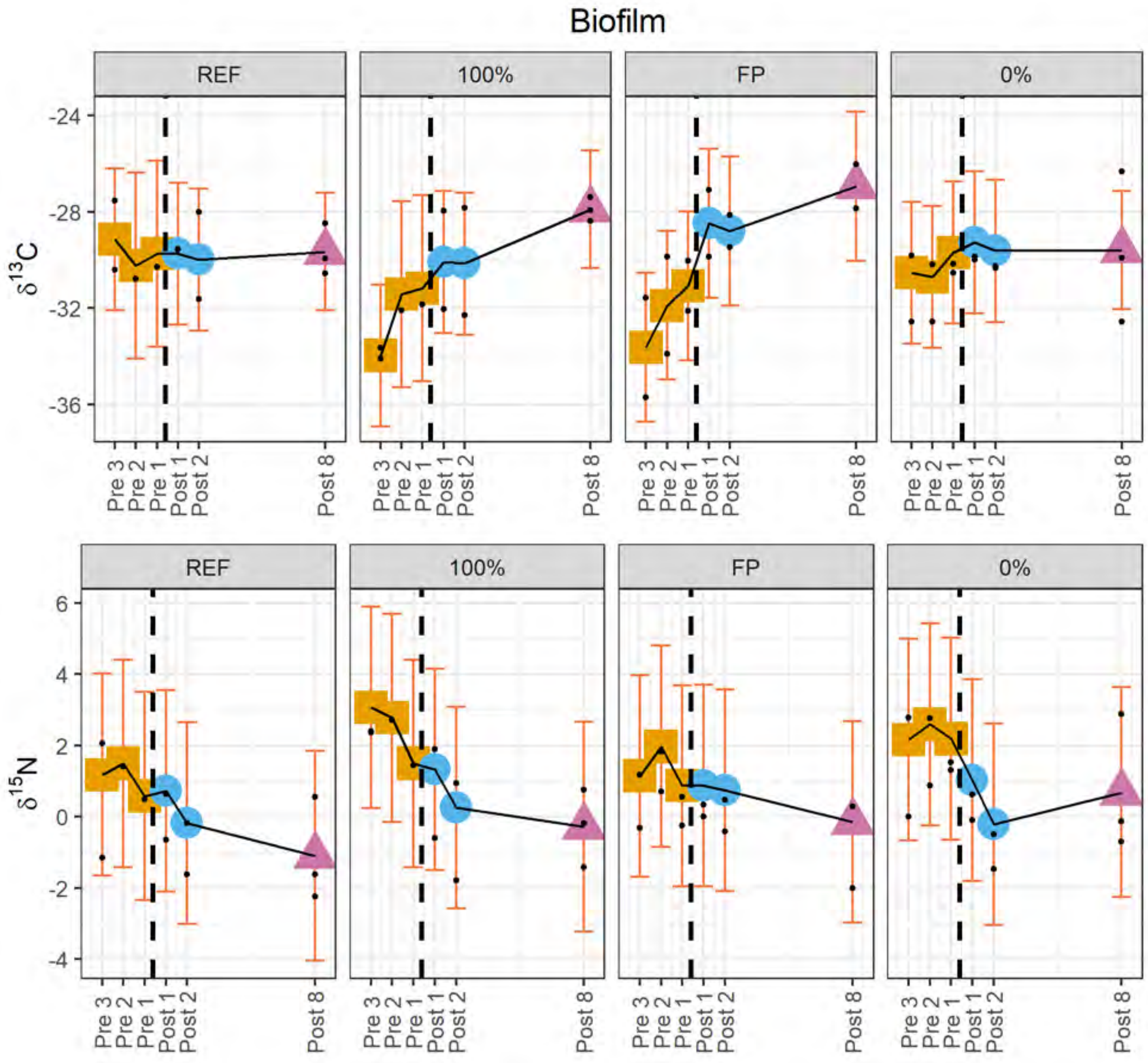

Figure 8-1. Mean $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for biofilm by sample year (where pre-harvest includes Pre 3, Pre 2, and Pre 1 and post-harvest includes Post 1, Post 2, and Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at riparian buffer treatment sites. Pre-harvest, Post 1 and Post 2, and Post 8 estimates are orange squares, blue circles, and pink triangles, respectively. Site means are dots.

Table 8-3. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for biofilm between pre-harvest and Post 1 and Post 2, and pre-harvest and Post 8.

	$\delta^{13}C$		$\delta^{15}N$	
Treatment	**Pre-Post 1 & 2**	**Pre-Post 8**	**Pre-Post 1 & 2**	**Pre-Post 8**
REF (n = 6)	-0.16 (-2.64, 2.33)	0.05 (-2.80, 2.90)	-0.81 (-2.19, 0.56)	-2.19 (-3.75, -0.63)
100% (n = 4)	2.07 (-0.42, 4.56)	4.28 (1.43, 7.14)	-1.65 (-3.03, -0.27)	-2.73 (-4.29, -1.17)
FP (n = 3)	3.57 (1.48, 5.65)	5.25 (2.62, 7.89)	-0.52 (-1.69, 0.65)	-1.48 (-2.95, 0.00)
0% (n = 4)	0.86 (-1.22, 2.94)	0.71 (-1.79, 3.22)	-1.9 (-3.07, -0.73)	-1.63 (-3.01, -0.25)

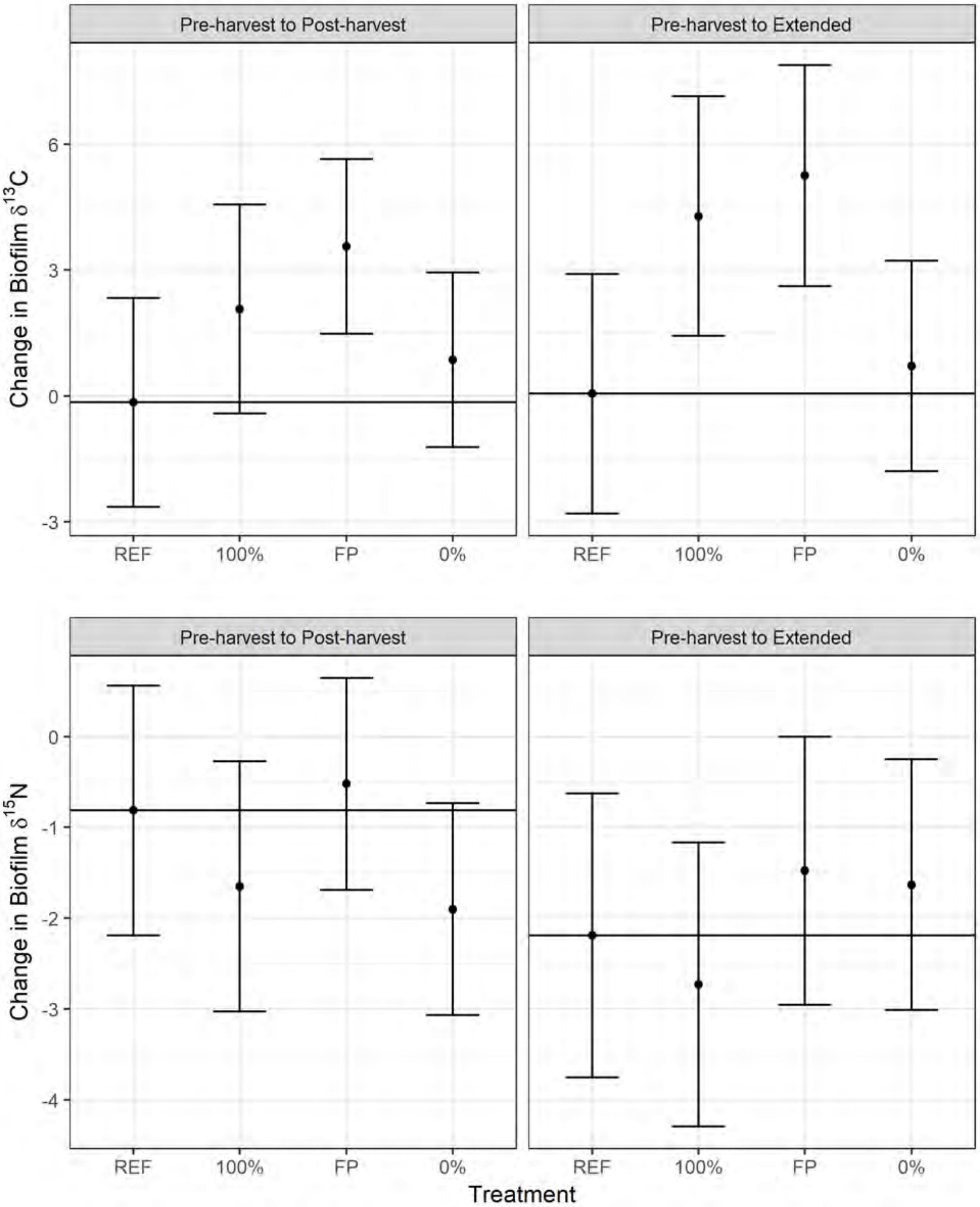

Figure 8-2. The within-treatment estimate of the change and 95% confidence intervals for mean $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for biofilm between pre-harvest and post-harvest (Post 1 and Post 2), and pre-harvest and extended (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 8-4. The between-treatment comparison of the change and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for biofilm between pre-harvest and Post 1 and Post 2, and Post 8. P-values for estimates are only shown when the treatment × period interaction was ≤ 0.10. Estimates for contrasts with a significant P-value are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	$\delta^{13}C$			$\delta^{15}N$	
	Post 1 & 2	**Post 8**		**Post 1 & 2**	**Post 8**
Contrast	**Estimate (CI)**	**Estimate (CI)**	**P**	**Estimate (CI)**	**Estimate (CI)**
100% vs. REF	2.22	**4.23**	**0.04**	-0.84	-0.54
	(-1.28, 5.73)	**(0.21, 8.25)**		(-2.78, 1.10)	(-2.67, 1.59)
FP vs. REF	3.72	**5.20**	**0.01**	0.29	0.72
	(0.48, 6.97)	**(1.32, 9.08)**		(-1.51, 2.10)	(-1.43, 2.86)
0% vs. REF	1.01	0.66	0.72	-1.09	0.56
	(-2.23, 4.26)	(-3.13, 4.46)		(-2.89, 0.72)	(-1.45, 2.57)
0% vs. FP	-2.71	**-4.54**	**0.02**	-1.38	-0.15
	(-5.65, 0.24)	**(-8.17, -0.9)**		(-3.03, 0.27)	(-2.18, 1.87)
0% vs. 100%	-1.21	**-3.57**	**0.06**	-0.25	1.10
	(-4.45, 2.04)	**(-7.36, 0.22)**		(-2.05, 1.56)	(-0.91, 3.11)
FP vs. 100%	1.50	0.97	0.61	1.13	1.25
	(-1.75, 4.74)	(-2.91, 4.85)		(-0.67, 2.94)	(-0.90, 3.40)

8-4.1.2. *Aquatic Macroinvertebrate Gatherers*

We did not find clear evidence that mean $\delta^{13}C$ varied among treatments for aquatic macroinvertebrate gatherers (pre-harvest to Post 1 and Post 2, P = 0.22; pre-harvest to Post 8, P = 0.96). We did find evidence that mean $\delta^{15}N$ varied among treatments for both comparisons (pre-harvest to Post 1 and Post 2, P = 0.03; pre-harvest to Post 8, P = 0.02). In Post 1 and 2, we estimated a 1.84$^{o}/_{oo}$ decrease (P = 0.016) in $\delta^{15}N$ in the FP treatment compared with pre-harvest, after controlling for temporal changes in the references. We also estimated that mean $\delta^{15}N$ was 1.27$^{o}/_{oo}$ higher (P = 0.086) in the 0% treatment than in the FP treatment, and 2.08$^{o}/_{oo}$ lower (P = 0.006) in the FP treatment than the 100% treatment, after adjusting for pre-harvest differences among the treatment sites. In Post 8, we estimated a 1.63$^{o}/_{oo}$ increase (P = 0.05) in the 0% treatment compared with pre-harvest, after controlling for temporal changes in the references. We also estimated that mean $\delta^{15}N$ was 2.88$^{o}/_{oo}$ (P <0.01) and 1.66$^{o}/_{oo}$ (P = 0.04) higher in the 0% treatment than in the FP and 100% treatments, respectively, after adjusting for pre-harvest differences among the treatment sites. See **Table 8-5**, **Table 8-6**, **Figure 8-3** and **Figure 8-4**.

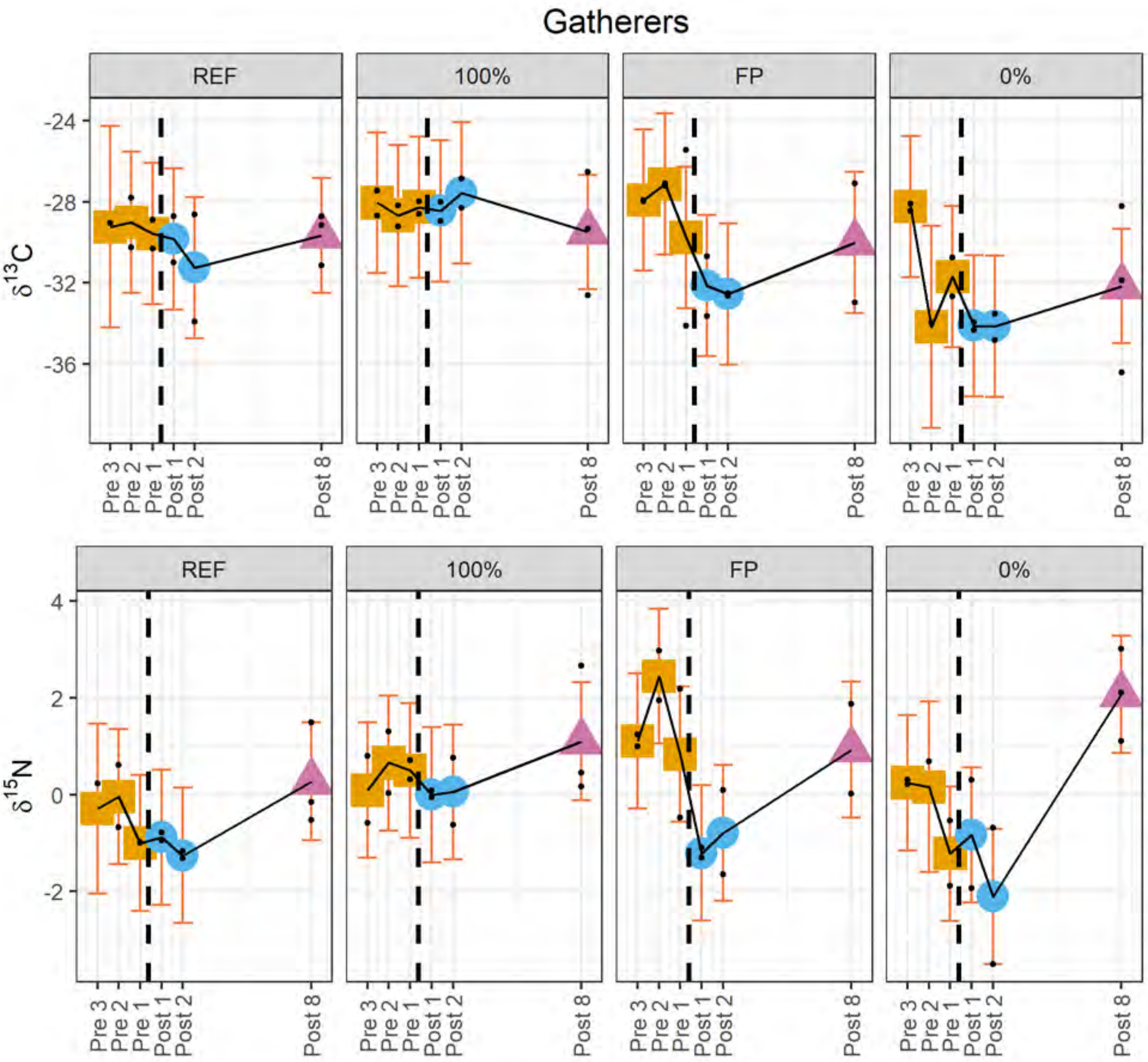

Figure 8-3. Means for $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for aquatic macroinvertebrate gatherers by sample year (where pre-harvest includes Pre 3, Pre 2, and Pre 1 and post-harvest includes Post 1, Post 2, and Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at riparian buffer treatment sites. Pre-harvest, Post 1 and Post 2, and Post 8 estimates are orange squares, blue circles, and pink triangles, respectively. Site means are dots.

Table 8-5. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for aquatic macroinvertebrate gatherers between the pre-harvest and Post 1 and Post 2, and pre-harvest and Post 8.

	$\delta^{13}C$		**$\delta^{15}N$**	
Treatment	**Pre-Post 1 & 2**	**Pre-Post 8**	**Pre-Post 1 & 2**	**Pre-Post 8**
REF (n = 6)	-1.27 (-4.59, 2.05)	-0.39 (-4.03, 3.24)	-0.62 (-1.69, 0.44)	0.72 (-0.5, 1.93)
100% (n = 4)	0.32 (-2.79, 3.42)	-1.17 (-4.61, 2.27)	-0.39 (-1.39, 0.61)	0.68 (-0.47, 1.84)
FP (n = 3)	-4.08 (-7.18, -0.98)	-1.74 (-5.66, 2.19)	-2.47 (-3.47, -1.47)	-0.54 (-1.8, 0.73)
0% (n = 4)	-2.76 (-6.08, 0.56)	-0.79 (-4.43, 2.85)	-1.2 (-2.27, -0.13)	2.35 (1.13, 3.56)

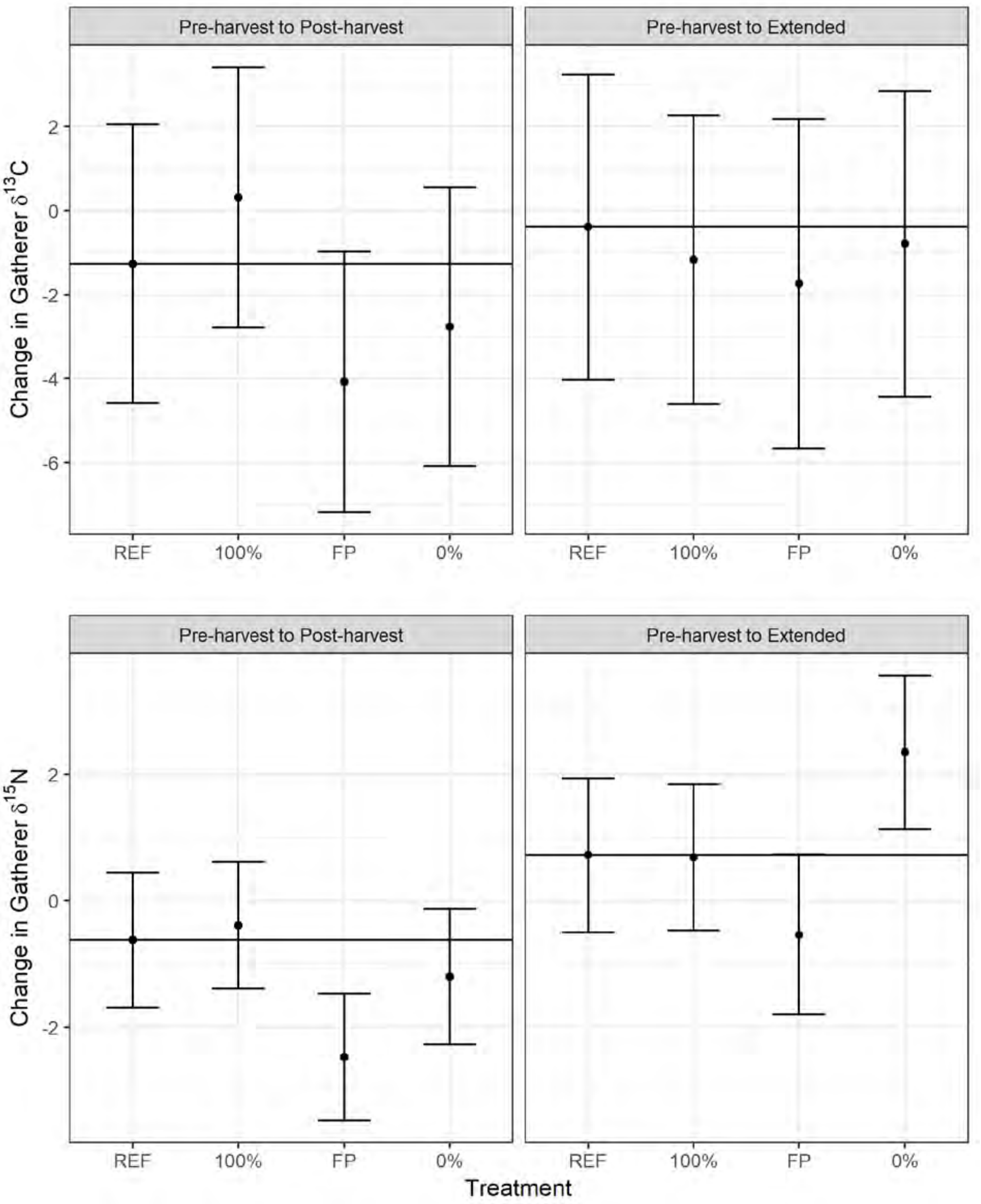

Figure 8-4. The within-treatment estimate of the change and 95% confidence intervals for mean $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for aquatic macroinvertebrate gatherers between pre-harvest and post-harvest (Post 1 and Post 2), and pre-harvest and extended (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 8-6. The between-treatment comparison of the change and 95% confidence intervals (CI) for mean δ^{13}C and δ^{15}N for aquatic macroinvertebrate gatherers between pre-harvest and Post 1 and Post 2, and pre-harvest and Post 8. P-values for estimates are only shown when the treatment × period interaction was ≤ 0.10. Estimates for contrasts with a significant P-value are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	δ^{13}C		**δ^{15}N**			
	Post 1 & 2	**Post 8**	**Post 1 & 2**		**Post 8**	
Contrast	**Estimate (CI)**	**Estimate (CI)**	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
100% vs. REF	1.59 (-2.96, 6.13)	-0.77 (-5.73, 4.18)	0.24 (-1.23, 1.70)	0.74	-0.03 (-1.63, 1.56)	0.97
FP vs. REF	-2.81 (-7.36, 1.74)	-1.34 (-6.7, 4.01)	**-1.84** **(-3.31, -0.38)**	**0.02**	-1.25 (-3.01, 0.50)	0.15
0% vs. REF	-1.49 (-6.16, 3.19)	-0.40 (-5.46, 4.67)	-0.57 (-2.08, 0.93)	0.44	**1.63** **(0.00, 3.27)**	**0.05**
0% vs. FP	1.32 (-3.22, 5.87)	0.95 (-4.40, 6.30)	**1.27** **(-0.20, 2.73)**	**0.09**	**2.88** **(1.13, 4.64)**	**<0.01**
0% vs. 100%	-3.07 (-7.62, 1.47)	0.38 (-4.57, 5.33)	-0.81 (-2.27, 0.65)	0.26	**1.66** **(0.07, 3.26)**	**0.04**
FP vs. 100%	-4.40 (-8.79, -0.01)	-0.57 (-5.79, 4.65)	**-2.08** **(-3.50, -0.66)**	**<0.01**	-1.22 (-2.94, 0.49)	0.15

8-4.1.3. *Aquatic Macroinvertebrate Shredders*

We did not find clear evidence that mean δ^{13}C varied among treatments (pre-harvest to Post 1 and Post 2, P = 0.50; pre-harvest to Post 8, P = 0.96) for aquatic macroinvertebrate shredders. We also did not find clear evidence that mean δ^{15}N varied among treatments (pre-harvest to Post 1 and Post 2, P = 0.89; pre-harvest to Post 8, P = 0.56). See **Table 8-7**, **Table 8-8**, **Figure 8-5** and **Figure 8-6**.

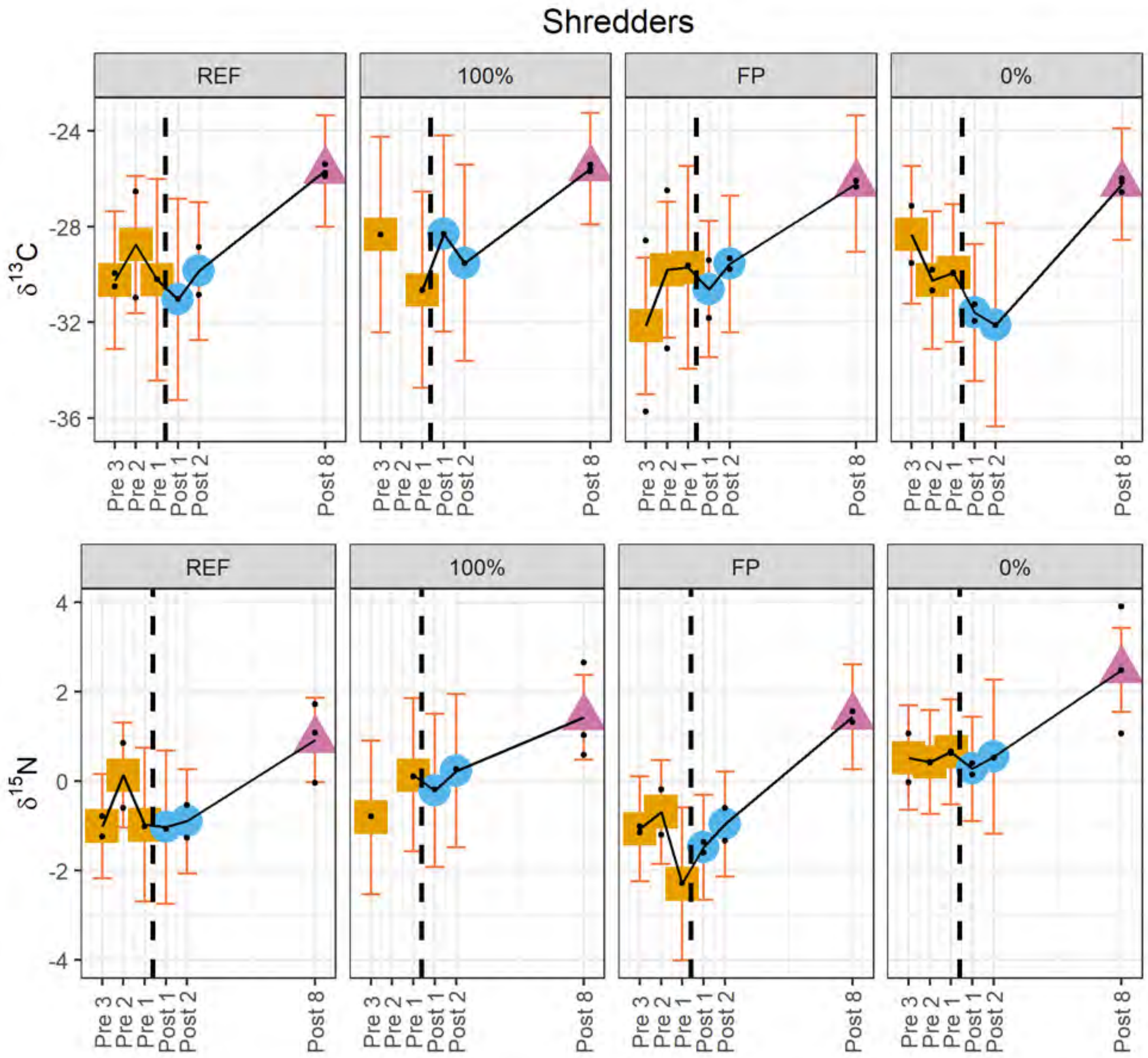

Figure 8-5. Means for δ^{13}C (top panel) and δ^{15}N (bottom panel) for aquatic macroinvertebrate shredders by sample year (where pre-harvest includes Pre 3, Pre 2, and Pre 1 and post-harvest includes Post 1, Post 2, and Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at riparian buffer treatment sites. Pre-harvest, Post 1 and Post 2, and Post 8 estimates are orange squares, blue circles, and pink triangles, respectively. Site means are dots.

Table 8-7. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for aquatic macroinvertebrate shredders between pre-harvest and Post 1 and Post 2, and pre-harvest and Post 8.

	$\delta^{13}C$		**$\delta^{15}N$**	
Treatment	**Pre-Post 1 & 2**	**Pre-Post 8**	**Pre-Post 1 & 2**	**Pre-Post 8**
REF (n = 6)	-0.71 (-3.86, 2.43)	4.07 (1.01, 7.12)	-0.35 (-1.61, 0.92)	1.54 (0.30, 2.77)
100% (n = 4)	0.58 (-3.50, 4.66)	3.91 (0.20, 7.62)	0.35 (-1.36, 2.06)	1.76 (0.26, 3.26)
FP (n = 3)	0.46 (-2.36, 3.28)	4.34 (0.87, 7.82)	0.14 (-1.00, 1.27)	2.80 (1.40, 4.19)
0% (n = 4)	-2.35 (-5.39, 0.70)	3.27 (0.39, 6.15)	-0.12 (-1.35, 1.11)	1.95 (0.79, 3.12)

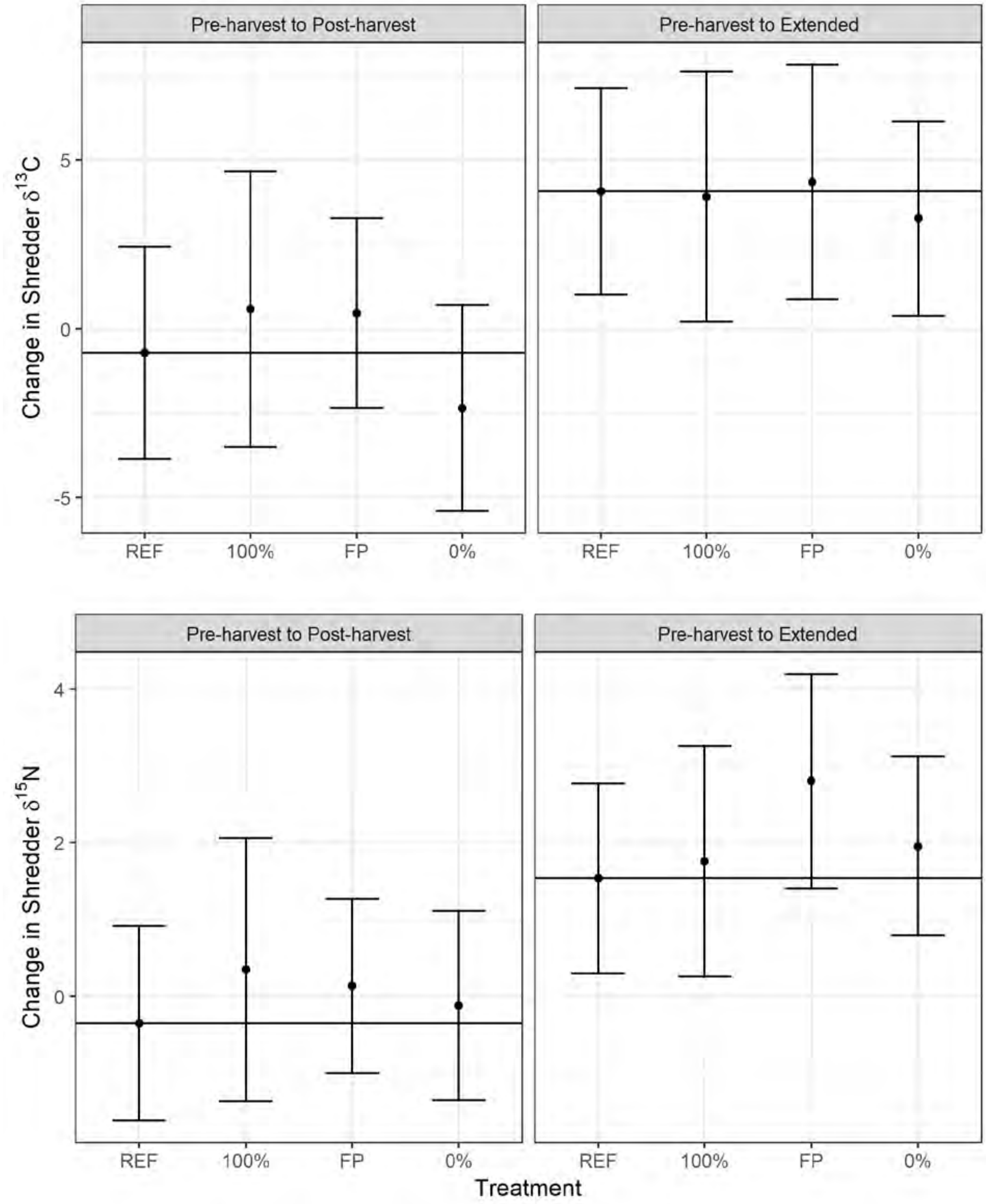

Figure 8-6. The within-treatment estimate of the change and 95% confidence intervals for mean $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for aquatic macroinvertebrate shredders between pre-harvest and post-harvest (Post 1 and Post 2), and pre-harvest and extended (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 8-8. The between-treatment comparison of the change and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for aquatic macroinvertebrate shredders between the pre-harvest and Post 1 and Post 2, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer. Results for $\delta^{13}C$ and $\delta^{15}N$ were not significant.

	$\delta^{13}C$		**$\delta^{15}N$**	
Contrast	**Post-harvest**	**Extended**	**Post-harvest**	**Extended**
100% vs. REF	1.30 (-3.85, 6.44)	-0.16 (-4.96, 4.65)	0.70 (-1.45, 2.84)	0.22 (-1.71, 2.16)
FP vs. REF	1.18 (-3.05, 5.40)	0.28 (-4.35, 4.9)	0.48 (-1.21, 2.17)	1.26 (-0.62, 3.14)
0% vs. REF	-1.63 (-6.01, 2.75)	-0.80 (-4.99, 3.4)	0.23 (-1.52, 1.98)	0.42 (-1.27, 2.1)
0% vs. FP	-2.81 (-6.96, 1.34)	-1.07 (-5.58, 3.44)	-0.25 (-1.9, 1.39)	-0.84 (-2.66, 0.98)
0% vs. 100%	-2.93 (-8.02, 2.16)	-0.64 (-5.34, 4.06)	-0.47 (-2.63, 1.69)	0.19 (-1.69, 2.08)
FP vs. 100%	-0.12 (-5.07, 4.84)	0.43 (-4.65, 5.52)	-0.22 (-2.31, 1.87)	1.04 (-1.01, 3.09)

8-4.1.4. *Torrent Salamanders*

We did not find clear evidence that mean torrent salamander $\delta^{13}C$ or $\delta^{15}N$ varied among treatments for larvae or post-metamorphs ($P > 0.10$;). See **Table 8-9**, **Table 8-10**, **Figure 8-7**, **Figure 8-8**, **Figure 8-9**, and **Figure 8-10**.

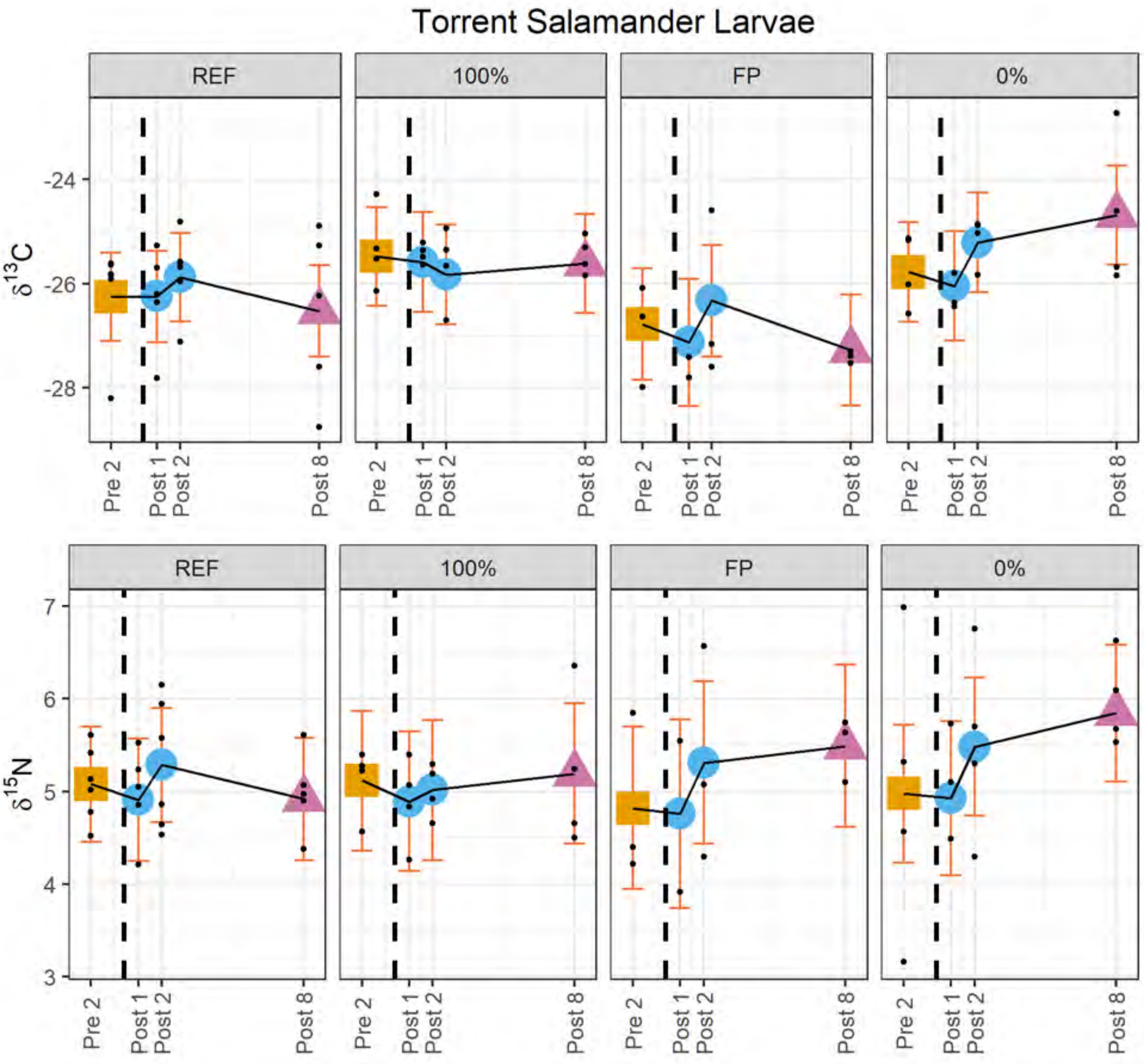

Figure 8-7. Means for $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for torrent salamander larvae by sample year (where pre-harvest includes Pre 2 and post-harvest includes Post 1, Post 2, and Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at riparian buffer treatment sites. Pre-harvest, Post 1 and Post 2, and Post 8 estimates are orange squares, blue circles, and pink triangles, respectively. Site means are dots.

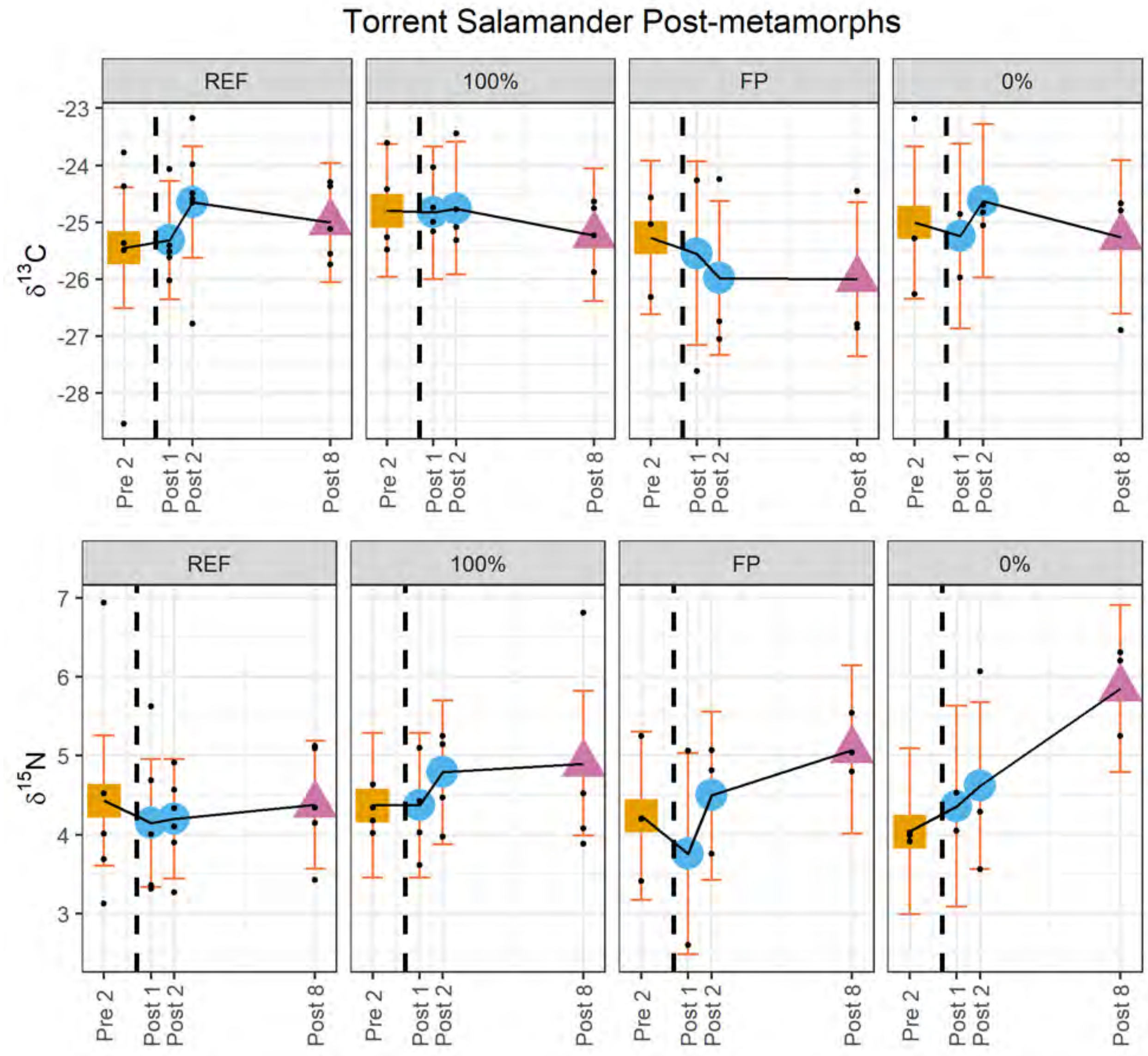

Figure 8-8. Means for $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for torrent salamander post-metamorphs by sample year (where pre-harvest includes Pre 2 and post-harvest includes Post 1, Post 2, and Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at riparian buffer treatment sites. Pre-harvest, Post 1 and Post 2, and Post 8 estimates are orange squares, blue circles, and pink triangles, respectively. Site means are dots.

Table 8-9. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean δ^{13}C and δ^{15}N for torrent salamander larvae and post-metamorphs between pre-harvest and Post 1 and Post 2, and pre-harvest and Post 8.

	δ^{13}C		**δ^{15}N**	
Treatment	**Pre-Post 1 & 2**	**Pre-Post 8**	**Pre-Post 1 & 2**	**Pre-Post 8**
		Larvae		
REF (n = 6)	0.20 (-0.50, 0.89)	-0.27 (-1.10, 0.56)	0.02 (-0.58, 0.61)	-0.16 (-0.88, 0.55)
100% (n = 4)	-0.23 (-1.06, 0.61)	-0.13 (-1.09, 0.83)	-0.16 (-0.87, 0.55)	0.07 (-0.75, 0.90)
FP (n = 3)	0.05 (-0.96, 1.06)	-0.50 (-1.61, 0.61)	0.21 (-0.65, 1.08)	0.67 (-0.28, 1.62)
0% (n = 4)	0.15 (-0.71, 1.01)	1.10 (0.10, 2.09)	0.23 (-0.51, 0.97)	0.87 (0.00, 1.73)
		Post-metamorphs		
REF (n = 6)	0.47 (-0.62, 1.56)	0.45 (-0.84, 1.73)	-0.26 (-1.11, 0.60)	-0.05 (-1.06, 0.95)
100% (n = 4)	0.00 (-1.21, 1.21)	-0.43 (-1.83, 0.97)	0.21 (-0.74, 1.15)	0.53 (-0.56, 1.62)
FP (n = 3)	-0.49 (-1.96, 0.97)	-0.73 (-2.34, 0.88)	-0.11 (-1.26, 1.04)	0.84 (-0.42, 2.10)
0% (n = 4)	0.07 (-1.46, 1.6)	-0.25 (-1.96, 1.45)	0.45 (-0.75, 1.65)	1.81 (0.47, 3.16)

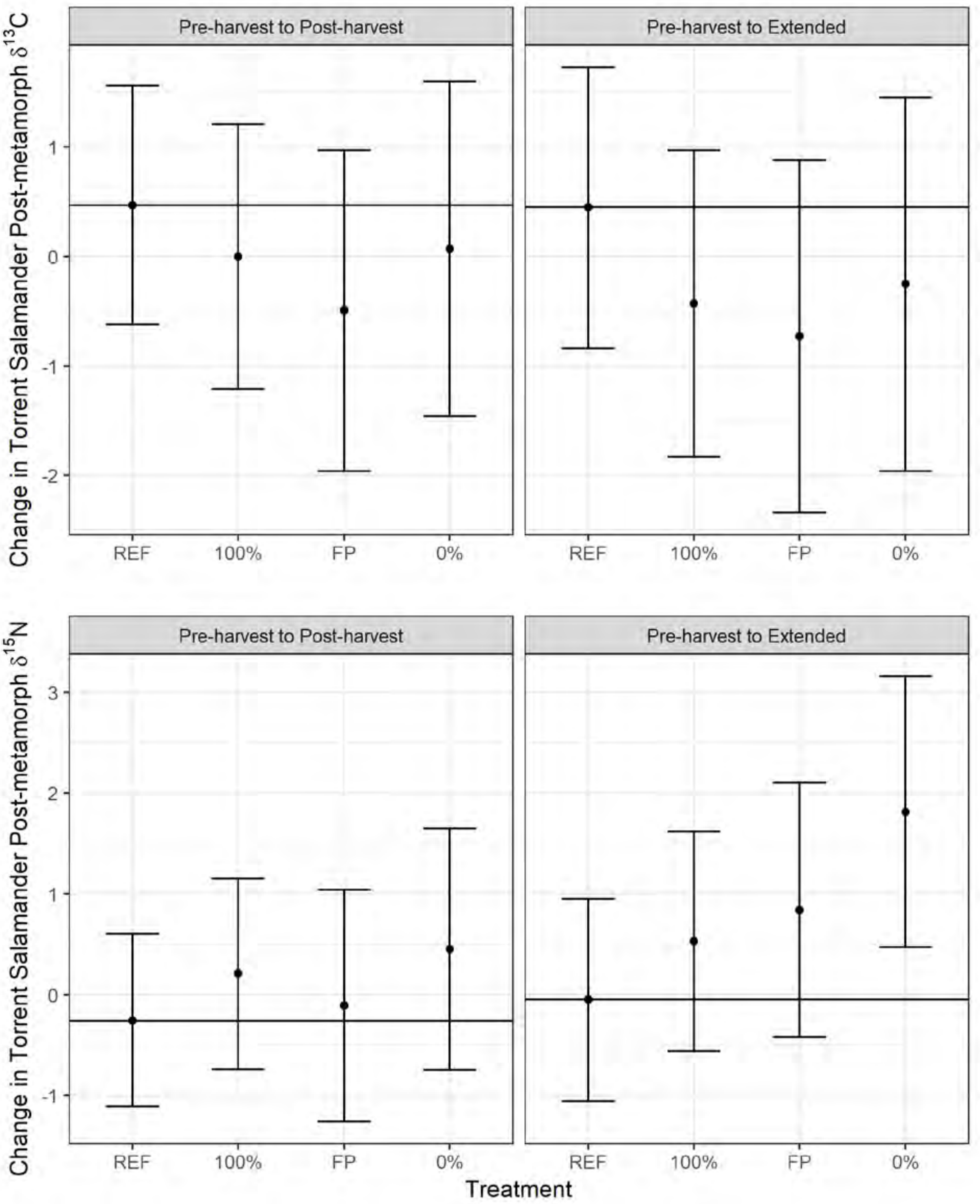

Figure 8-9. The within-treatment estimate of the change and 95% confidence intervals for mean $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for torrent salamander larvae between pre-harvest and post-harvest (Post 1 and Post 2), and pre-harvest and extended (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

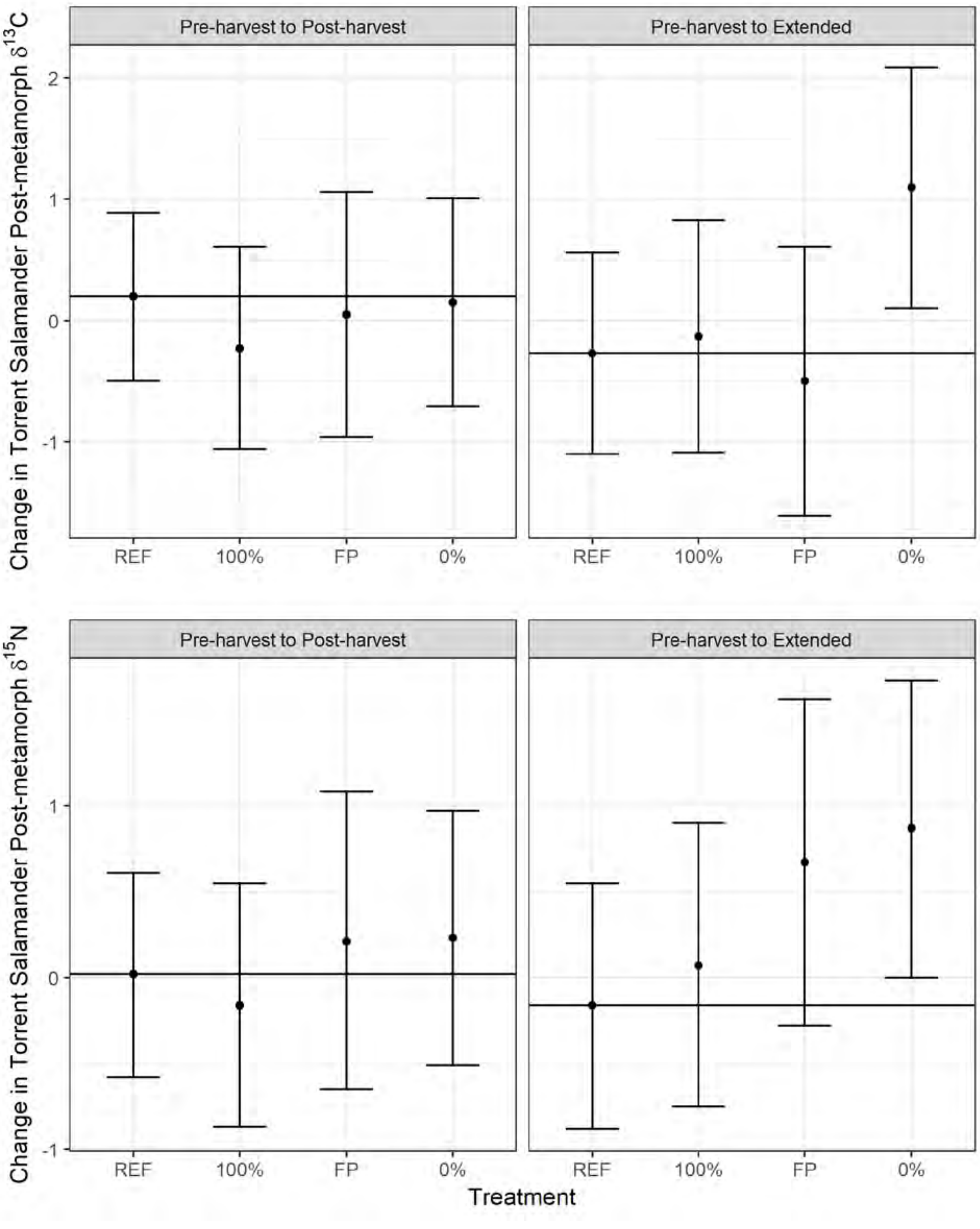

Figure 8-10. The within-treatment estimate of the change and 95% confidence intervals for mean $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for torrent salamander post-metamorphs between pre-harvest and post-harvest (Post 1 and Post 2), and pre-harvest and extended (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 8-10. The between-treatment comparison of the change and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for torrent salamander larvae and post-metamorphs between the pre-harvest and Post 1 and Post 2, and Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer. Results for $\delta^{13}C$ and $\delta^{15}N$ were not significant for either life stage.

	$\delta^{13}C$		**$\delta^{15}N$**	
Contrast	**Post 1 & 2**	**Post 8**	**Post 1 & 2**	**Post 8**
Larvae				
100% vs. REF	-0.42 (-1.50, 0.66)	0.14 (-1.13, 1.41)	-0.18 (-1.11, 0.75)	0.24 (-0.85, 1.33)
FP vs. REF	-0.15 (-1.37, 1.08)	-0.23 (-1.62, 1.15)	0.20 (-0.85, 1.25)	0.83 (-0.36, 2.02)
0% vs. REF	-0.04 (-1.15, 1.06)	1.37 (0.07, 2.66)	0.21 (-0.74, 1.16)	1.03 (-0.09, 2.15)
0% vs. FP	0.10 (-1.22, 1.43)	1.60 (0.11, 3.08)	0.01 (-1.12, 1.15)	0.20 (-1.09, 1.48)
0% vs. 100%	0.38 (-0.82, 1.57)	1.23 (-0.15, 2.61)	0.39 (-0.64, 1.41)	0.79 (-0.40, 1.98)
FP vs. 100%	0.27 (-1.03, 1.58)	-0.37 (-1.83, 1.10)	0.38 (-0.74, 1.50)	0.60 (-0.66, 1.85)
Post-metamorphs				
100% vs. REF	-0.47 (-2.10, 1.16)	-0.88 (-2.78, 1.02)	0.46 (-0.81, 1.74)	0.58 (-0.90, 2.07)
FP vs. REF	-0.96 (-2.80, 0.87)	-1.18 (-3.24, 0.88)	0.15 (-1.28, 1.58)	0.89 (-0.72, 2.50)
0% vs. REF	-0.40 (-2.29, 1.49)	-0.70 (-2.85, 1.45)	0.71 (-0.78, 2.19)	1.87 (0.18, 3.55)
0% vs. FP	0.56 (-1.54, 2.67)	0.48 (-1.87, 2.82)	0.56 (-1.09, 2.21)	0.97 (-0.87, 2.82)
0% vs. 100%	0.07 (-1.88, 2.02)	0.18 (-2.03, 2.38)	0.24 (-1.28, 1.77)	1.28 (-0.45, 3.01)
FP vs. 100%	-0.49 (-2.40, 1.41)	-0.30 (-2.43, 1.83)	-0.32 (-1.80, 1.17)	0.31 (-1.36, 1.97)

8-4.1.5. *Giant Salamanders*

We found statistical evidence that mean $\delta^{13}C$ in Post 1 and Post 2 varied among treatments for larval ($P < 0.001$) and neotenic giant salamanders ($P = 0.04$). We also found statistical evidence that mean $\delta^{15}N$ in Post 1 and Post 2 varied among treatments for neotenic giant salamanders ($P = 0.095$). We did not find clear evidence of variation among treatments for any comparisons in Post 8 ($P > 0.10$). See **Table 8-11**, **Table 8-12**, **Figure 8-11**, **Figure 8-12**, **Figure 8-13** and **Figure 8-14**.

In Post 1 and Post 2, we estimated a 1.14 ($P = 0.06$) increase and a 1.46 ($P = 0.04$) decrease in mean $\delta^{13}C$ for giant salamander larvae in the 100% and FP treatments, respectively, compared with pre-harvest, after controlling for temporal changes in the references. We also estimated a 1.93 ($P = 0.01$) increase in the 0% compared to the FP treatment and a 2.60 ($P = 0.001$) decrease in the FP treatment compared to the 100% treatment, after adjusting for pre- harvest differences among the treatment sites.

For neotenes in Post 1 and Post 2, we estimated a 1.31 ($P = 0.02$) increase in mean $\delta^{13}C$ in the FP treatment, compared with the pre-harvest period, after controlling for temporal changes in the references. We also estimated a 1.63 (P = 0.01) increase in the FP treatment compared to the 100% treatment, after adjusting for pre-treatment differences among the treatment sites. For mean δ^{15}N, we estimated a 1.44 (P = 0.02) increase in the FP treatment, compared with pre-harvest, after controlling for temporal changes in the references. We also estimated a 1.48 increase (P = 0.04) and 1.52 (P = 0.02) decrease in the FP treatment compared to the 100% and 0% treatments, respectively, after adjusting for pre-harvest differences among the treatment sites.

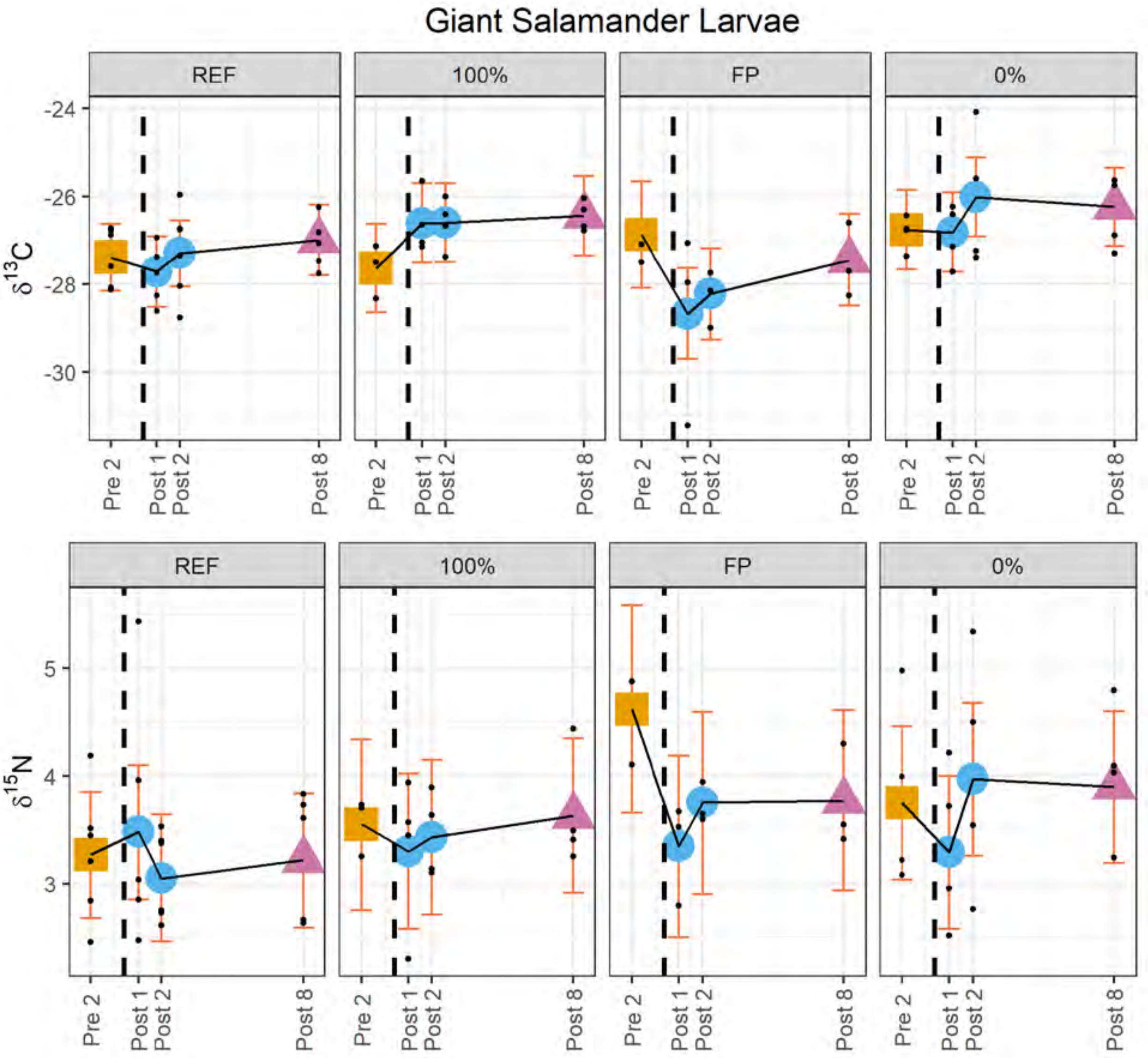

Figure 8-11. Mean δ^{13}C (top panel) and δ^{15}N (bottom panel) for giant salamander larvae by sample year (where pre-harvest includes Pre 2 and post-harvest includes Post 1, Post 2, and Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at riparian buffer treatment sites. Pre-harvest, Post 1 and Post 2, and Post 8 estimates are orange squares, blue circles, and pink triangles, respectively. Site means are dots.

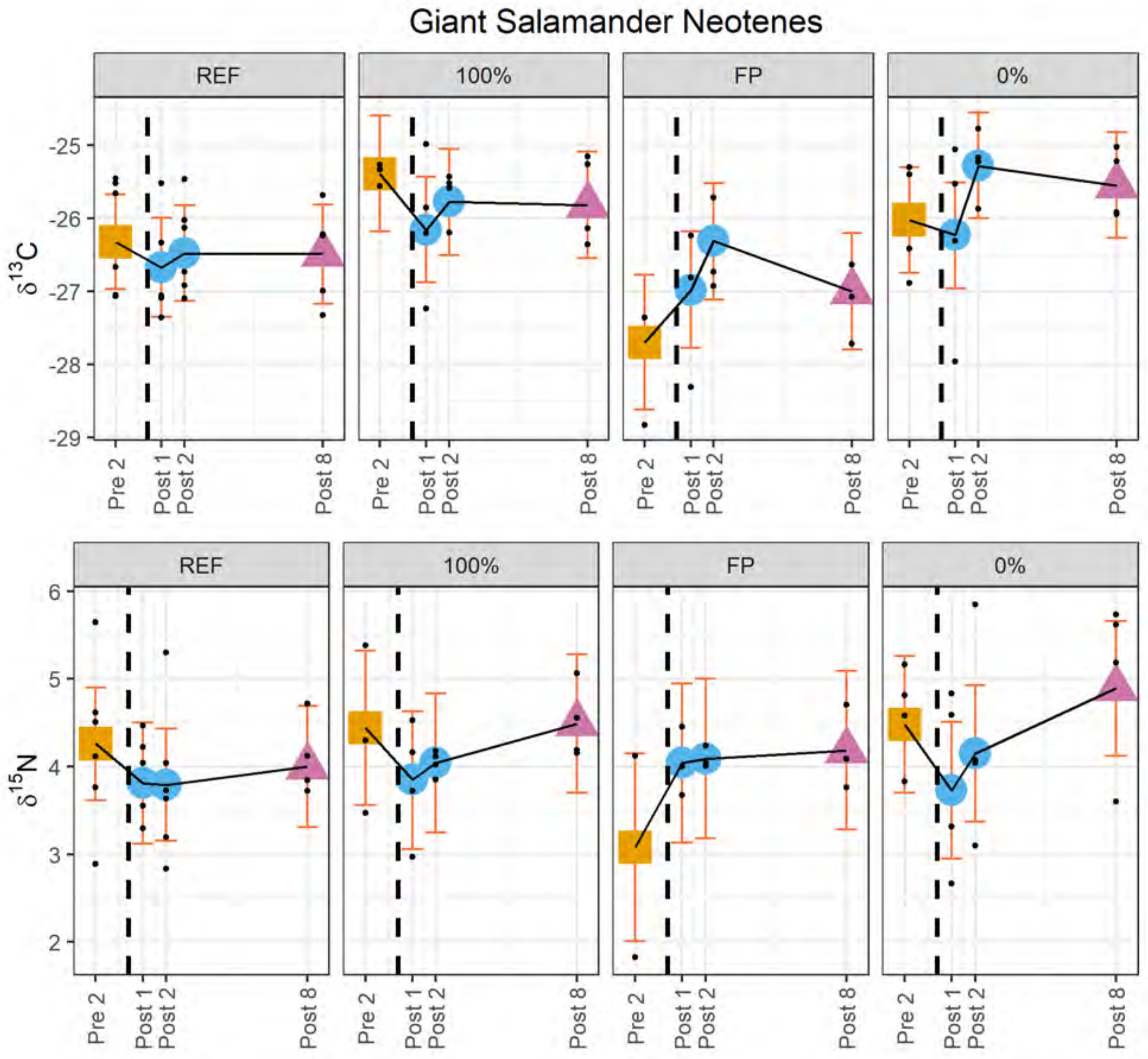

Figure 8-12. Mean $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for giant salamander neotenes by sample year (where pre-harvest includes Pre 2 and post-harvest includes Post 1, Post 2, and Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at riparian buffer treatment sites. Pre-harvest, Post 1 and Post 2, and Post 8 estimates are orange squares, blue circles, and pink triangles, respectively. Site means are dots.

Table 8-11. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for giant salamander larvae and neotenes between pre-harvest and Post 1 and Post 2, and pre-harvest and Post 8.

	$\delta^{13}C$		**$\delta^{15}N$**	
Treatment	**Pre-Post 1 & 2**	**Pre-Post 8**	**Pre-Post 1 & 2**	**Pre-Post 8**
Larvae				
REF (n = 6)	-0.12 (-0.82, 0.59)	0.39 (-0.45, 1.24)	0.00 (-0.54, 0.54)	-0.05 (-0.70, 0.60)
100% (n = 4)	1.02 (0.07, 1.98)	1.19 (0.12, 2.25)	-0.19 (-0.92, 0.54)	0.08 (-0.74, 0.90)
FP (n = 3)	-1.58 (-2.73, -0.42)	-0.58 (-1.86, 0.71)	-1.08 (-1.97, -0.19)	-0.85 (-1.84, 0.13)
0% (n = 4)	0.35 (-0.49, 1.19)	0.52 (-0.49, 1.54)	-0.12 (-0.77, 0.52)	0.15 (-0.64, 0.93)
Neotenes				
REF (n = 6)	-0.26 (-0.83, 0.32)	-0.17 (-0.86, 0.52)	-0.46 (-1.10, 0.19)	-0.26 (-1.03, 0.52)
100% (n = 4)	-0.58 (-1.35, 0.20)	-0.43 (-1.30, 0.45)	-0.50 (-1.37, 0.37)	0.05 (-0.93, 1.03)
FP (n = 3)	1.05 (0.11, 1.99)	0.70 (-0.35, 1.75)	0.98 (-0.08, 2.04)	1.10 (-0.08, 2.28)
0% (n = 4)	0.26 (-0.43, 0.96)	0.48 (-0.33, 1.28)	-0.54 (-1.31, 0.23)	0.41 (-0.52, 1.34)

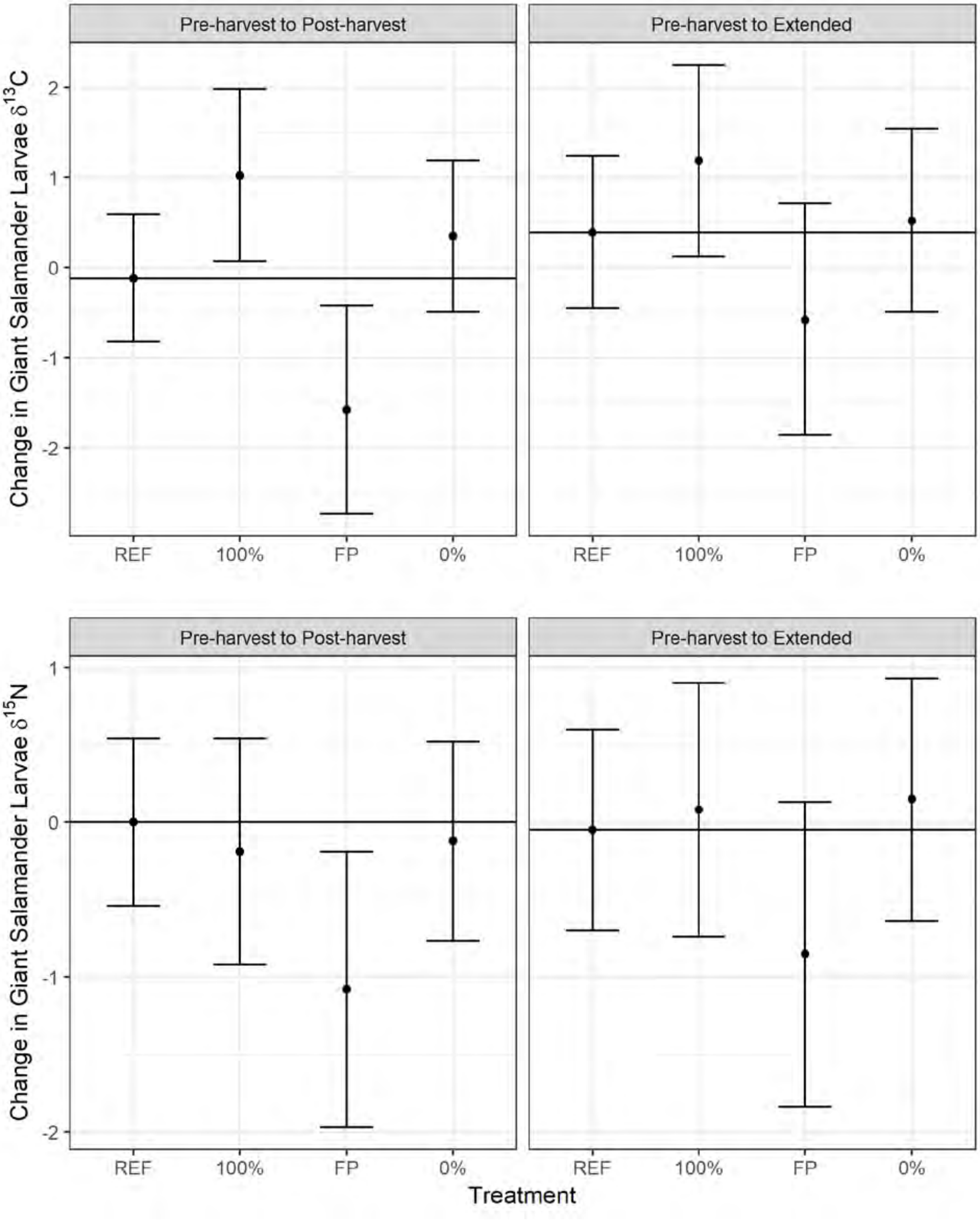

Figure 8-13. The within-treatment estimate of the change and 95% confidence intervals for mean $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for giant salamander larvae between pre-harvest and post-harvest (Post 1 and Post 2), and pre-harvest and extended (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

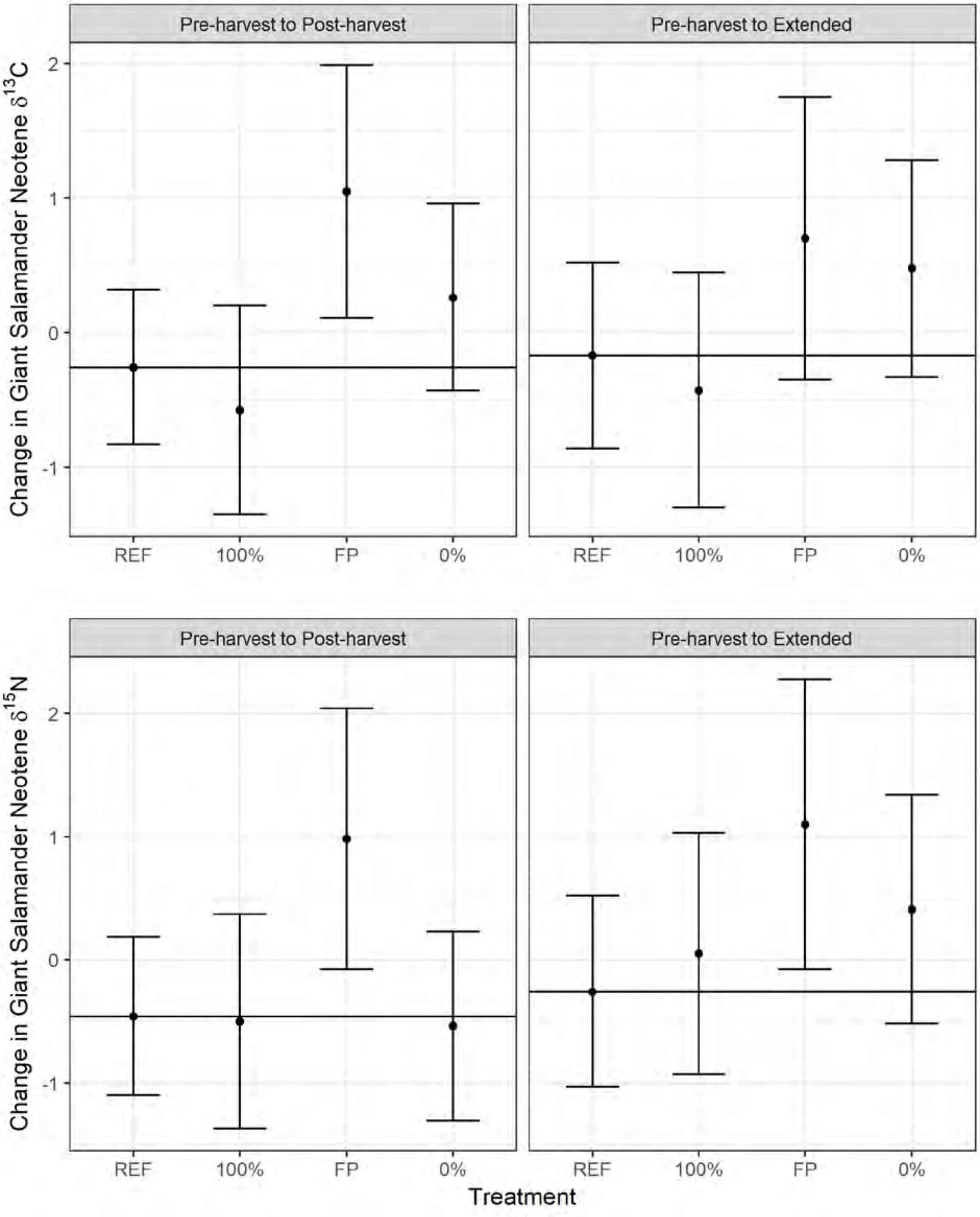

Figure 8-14. The within-treatment estimate of the change and 95% confidence intervals for mean $\delta^{13}C$ (top panel) and $\delta^{15}N$ (bottom panel) for giant salamander neotenes between pre-harvest and post-harvest (Post 1 and Post 2), and pre-harvest and extended (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 8-12. The between-treatment comparison of the change and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for giant salamander larvae and neotenes between pre-harvest and Post 1 and Post 2, and Post 8. P-values for estimates are only shown when the treatment × period interaction was significant (≤0.10). Estimates for contrasts with a significant P-value are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	$\delta^{13}C$			**$\delta^{15}N$**		
	Post 1 & 2		**Post 8**	**Post 1 & 2**		**Post 8**
Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**	**Estimate (CI)**	**P**	**Estimate (CI)**
			Larvae			
100% vs. REF	**1.14**	**0.06**	0.79	-0.19		0.13
	(-0.04, 2.32)		(-0.57, 2.15)	(-1.09, 0.72)		(-0.91, 1.18)
FP vs. REF	**-1.46**	**0.04**	-0.97	-1.08		-0.81
	(-2.81, -0.11)		(-2.51, 0.57)	(-2.12, -0.04)		(-1.99, 0.38)
0% vs. REF	0.47	0.39	0.13	-0.12		0.19
	(-0.63, 1.56)		(-1.19, 1.45)	(-0.96, 0.72)		(-0.83, 1.21)
0% vs. FP	**1.93**	**0.01**	1.10	0.95		1.00
	(0.50, 3.36)		(-0.54, 2.74)	(-0.14, 2.05)		(-0.26, 2.26)
0% vs. 100%	-0.67	0.29	-0.66	0.06		0.06
	(-1.94, 0.60)		(-2.14, 0.81)	(-0.91, 1.04)		(-1.07, 1.20)
FP vs. 100%	**-2.60**	**0.001**	-1.76	-0.89		-0.94
	(-4.09, -1.10)		(-3.43, -0.10)	(-2.04, 0.26)		(-2.22, 0.35)
			Neotenes			
100% vs. REF	-0.32	0.51	-0.26	-0.04	0.94	0.30
	(-1.29, 0.65)		(-1.37, 0.85)	(-1.12, 1.04)		(-0.94, 1.55)
FP vs. REF	**1.31**	**0.02**	0.87	**1.44**	**0.02**	1.36
	(0.20, 2.41)		(-0.39, 2.12)	**(0.20, 2.68)**		(-0.05, 2.77)
0% vs. REF	0.52	0.25	0.64	-0.08	0.87	0.66
	(-0.39, 1.43)		(-0.42, 1.71)	(-1.09, 0.92)		(-0.55, 1.87)
0% vs. FP	-0.79	0.18	-0.22	**-1.52**	**0.02**	-0.69
	(-1.96, 0.38)		(-1.55, 1.10)	**(-2.83, -0.21)**		(-2.19, 0.81)
0% vs. 100%	0.84	0.11	0.90	-0.04	0.94	0.36
	(-0.20, 1.89)		(-0.29, 2.09)	(-1.21, 1.12)		(-0.99, 1.71)
FP vs. 100%	**1.63**	**0.01**	1.13	**1.48**	**0.04**	1.05
	(0.42, 2.84)		(-0.23, 2.49)	**(0.11, 2.85)**		(-0.48, 2.58)

8-4.2. POST-HARVEST POST 8 TREATMENT COMPARISON

Since some metrics were not sampled in the pre-harvest period or Post 1 and Post 2, we conducted a separate analysis comparing post-harvest differences among treatments (i.e., not a part of the BACI analysis). In our post-harvest Post 8 comparison, we did not find clear evidence that mean $\delta^{13}C$ or $\delta^{15}N$ for litterfall varied among treatments ($P > 0.10$ for all comparisons; **Table 8-13**). We also did not find clear statistical evidence that mean $\delta^{15}N$ for instream detritus varied among treatments ($P > 0.10$ for all comparisons), however, we lacked sufficient sample for analysis of mean $\delta^{15}N$ for instream deciduous detritus in the FP treatment. We did find statistical evidence that mean $\delta^{13}C$ for instream deciduous detritus varied among treatments ($P = 0.08$). In Post 8, we estimated that mean $\delta^{13}C$ for instream deciduous detritus in the 0% treatment was 3.90$^{o}/_{oo}$ higher ($P = 0.03$), and FP treatment was 4.39$^{o}/_{oo}$ higher ($P = 0.05$) than in the 100% treatment (**Table 8-14**).

We did not find clear evidence that mean $\delta^{13}C$ or $\delta^{15}N$ for aquatic or terrestrial macroinvertebrate predators (stoneflies and spiders), terrestrial gatherers (springtails) or aquatic scrapers varied among treatments in Post 8 ($P > 0.10$ for all comparisons; **Table 8-15**).

We did have evidence that mean $\delta^{15}N$ for Coastal Tailed Frog larvae varied among treatments in Post 8 ($P = 0.02$). We estimated that mean $\delta^{15}N$ was 2.56$^{o}/_{oo}$ ($P < 0.01$) and 2.20$^{o}/_{oo}$ ($P = 0.01$) higher in the 100% and 0% treatments, respectively, than in the reference. We had no statistical evidence that mean $\delta^{15}N$ for Coastal Tailed Frog post-metamorphs varied among treatments ($P > 0.10$). We had statistical evidence that mean $\delta^{13}C$ for Coastal Tailed Frog varied among treatments for both life stages ($P < 0.001$ for larvae and $P = 0.09$ for post-metamorphs). We estimated that mean $\delta^{13}C$ for larvae was 5.20$^{o}/_{oo}$ ($P < 0.001$) and 5.63$^{o}/_{oo}$ ($P < 0.001$) higher in the 100% and 0% treatments, respectively, than in the reference. We also estimated that mean $\delta^{13}C$ for larvae was 6.37$^{o}/_{oo}$ lower ($P = 0.002$) and 6.81$^{o}/_{oo}$ ($P = 0.001$) higher in the 100% and 0% treatments, respectively, than in the FP treatment. Mean $\delta^{13}C$ for post-metamorphs was 2.31$^{o}/_{oo}$ ($P = 0.05$) and 2.84$^{o}/_{oo}$ ($P = 0.02$) lower in the FP treatment than in the 100% treatment and reference, respectively (**Table 8-16**).

When Post 8 results for $\delta^{13}C$ and $\delta^{15}N$ were considered simultaneously, we found that, in general, organic matter (biofilm, litterfall and instream detritus), aquatic and terrestrial gatherer macroinvertebrates, and aquatic scraper macroinvertebrates exhibited lower mean $\delta^{13}C$ and $\delta^{15}N$ values than aquatic and terrestrial predator macroinvertebrates, aquatic shredder macroinvertebrates, and stream-associated amphibians, which tended to have higher values for both $\delta^{13}C$ and $\delta^{15}N$ (**Figure 8-15**). The higher isotope values for the amphibians were consistent across all study sites and treatments.

Table 8-13. The between-treatment difference and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for instream coniferous, deciduous, and wood litterfall. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer. Results for $\delta^{13}C$ and $\delta^{15}N$ were not significant for any contrasts.

	$\delta^{13}C$	**$\delta^{15}N$**
Post 8 Contrast	**Estimate (CI)**	**Estimate (CI)**
Coniferous		
100% vs. REF	-0.26 (-1.90, 1.38)	0.05 (-1.27, 1.37)
FP vs. REF	-0.11 (-1.94, 1.72)	-1.34 (-2.87, 0.19)
0% vs. REF	0.00 (-1.64, 1.64)	0.07 (-1.26, 1.39)
0% vs. FP	0.11 (-1.72, 1.94)	1.41 (-0.13, 2.94)
0% vs. 100%	0.26 (-1.38, 1.90)	0.01 (-1.31, 1.34)
FP vs. 100%	0.15 (-1.68, 1.98)	-1.39 (-2.93, 0.14)
Deciduous		
100% vs. REF	0.56 (-1.48, 2.59)	0.12 (-0.42, 0.65)
FP vs. REF	-0.48 (-2.83, 1.88)	0.50 (-0.09, 1.10)
0% vs. REF	-0.48 (-2.51, 1.56)	-0.13 (-0.67, 0.40)
0% vs. FP	0.00 (-2.36, 2.36)	-0.64 (-1.23, -0.04)
0% vs. 100%	-1.03 (-3.07, 1.00)	-0.25 (-0.79, 0.28)
FP vs. 100%	-1.03 (-3.39, 1.33)	0.38 (-0.21, 0.98)
Wood		
100% vs. REF	-1.41 (-4.34, 1.53)	-0.08 (-1.50, 1.34)
FP vs. REF	0.63 (-2.78, 4.04)	-1.28 (-2.93, 0.37)
0% vs. REF	-0.48 (-3.41, 2.45)	-0.21 (-1.63, 1.20)
0% vs. FP	-1.11 (-4.52, 2.30)	1.07 (-0.59, 2.72)
0% vs. 100%	0.92 (-2.01, 3.85)	-0.14 (-1.55, 1.28)
FP vs. 100%	2.03 (-1.38, 5.45)	-1.20 (-2.85, 0.45)

Table 8-14. The between-treatment difference and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for instream coniferous, deciduous, and wood detritus. P-values for estimates are only shown when the P-value for the treatment comparison was ≤.1. Estimates for contrasts with a significant P-value are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer. The small sample size for $\delta^{15}N$ for instream deciduous detritus in the FP treatment prevented analysis.

	$\delta^{13}C$		**$\delta^{15}N$**
Post 8 Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**
Coniferous			
100% vs. REF	1.09 (0.11, 2.06)		0.08 (-1.80, 1.96)
FP vs. REF	0.68 (-0.3, 1.65)		-1.62 (-3.50, 0.26)
0% vs. REF	0.52 (-0.35, 1.40)		-0.26 (-1.89, 1.37)
0% vs. FP	-0.15 (-1.13, 0.82)		1.36 (-0.52, 3.24)
0% vs. 100%	-0.57 (-1.54, 0.41)		-0.34 (-2.22, 1.54)
FP vs. 100%	-0.41 (-1.48, 0.65)		-1.70 (-3.85, 0.44)
Deciduous			
100% vs. REF	-2.06 (-6.58, 2.45)	0.27	-0.33 (-4.58, 3.93)
FP vs. REF	2.33 (-3.20, 7.86)	0.31	--
0% vs. REF	1.83 (-2.68, 6.35)	0.32	-0.23 (-4.49, 4.03)
0% vs. FP	-0.49 (-5.01, 4.02)	0.78	--
0% vs. 100%	**3.90 (0.71, 7.09)**	**0.03**	0.09 (-2.74, 2.93)
FP vs. 100%	**4.39 (-0.12, 8.91)**	**0.05**	--
Wood			
100% vs. REF	-1.93 (-4.9, 1.04)		0.90 (-0.65, 2.44)
FP vs. REF	-1.35 (-4.78, 2.09)		-0.20 (-1.98, 1.58)
0% vs. REF	-0.95 (-3.92, 2.02)		0.47 (-1.08, 2.02)
0% vs. FP	0.40 (-3.04, 3.83)		0.67 (-1.11, 2.45)
0% vs. 100%	0.98 (-1.99, 3.95)		-0.43 (-1.97, 1.12)
FP vs. 100%	0.58 (-2.85, 4.02)		-1.10 (-2.88, 0.68)

Table 8-15. The between-treatment difference and 95% confidence intervals (CI) for mean δ^{13}C and δ^{15}N for terrestrial and aquatic predator, terrestrial gatherer, and aquatic scraper macroinvertebrates. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer. Results for δ^{13}C and δ^{15}N were not significant for any contrasts.

	δ^{13}C	**δ^{15}N**
Post 8 Contrast	**Estimate (CI)**	**Estimate (CI)**
Predator - Spiders		
100% vs. REF	0.20 (-3.02, 3.42)	-1.01 (-3.38, 1.35)
FP vs. REF	0.04 (-3.18, 3.27)	-0.24 (-2.60, 2.13)
0% vs. REF	0.13 (-3.09, 3.35)	0.68 (-1.68, 3.05)
0% vs. FP	0.09 (-3.13, 3.31)	0.92 (-1.44, 3.28)
0% vs. 100%	-0.07 (-3.29, 3.15)	1.70 (-0.67, 4.06)
FP vs. 100%	-0.16 (-3.38, 3.07)	0.78 (-1.59, 3.14)
Predator – Stoneflies		
100% vs. REF	-0.10 (-1.74, 1.55)	1.00 (-0.20, 2.19)
FP vs. REF	-0.68 (-2.59, 1.22)	0.34 (-1.04, 1.72)
0% vs. REF	-0.63 (-2.28, 1.02)	1.16 (-0.03, 2.35)
0% vs. FP	0.05 (-1.85, 1.96)	0.82 (-0.56, 2.20)
0% vs. 100%	-0.53 (-2.18, 1.11)	0.16 (-1.03, 1.36)
FP vs. 100%	-0.58 (-2.49, 1.32)	-0.65 (-2.03, 0.73)
Gatherer - Springtails		
100% vs. REF	-0.41 (-1.56, 0.74)	0.14 (-3.80, 4.08)
FP vs. REF	0.36 (-0.84, 1.56)	-0.73 (-6.14, 4.68)
0% vs. REF	-0.24 (-1.39, 0.91)	-0.14 (-4.08, 3.80)
0% vs. FP	-0.6 (-1.75, 0.55)	0.58 (-4.65, 5.82)
0% vs. 100%	0.17 (-0.81, 1.15)	-0.29 (-3.69, 3.12)
FP vs. 100%	0.77 (-0.38, 1.92)	-0.87 (-6.11, 4.37)
Scraper		
100% vs. REF	1.31 (-3.67, 6.29)	0.43 (-0.80, 1.67)
FP vs. REF	-0.59 (-6.33, 5.14)	0.07 (-1.35, 1.50)
0% vs. REF	-0.01 (-4.98, 4.97)	1.12 (-0.12, 2.35)
0% vs. FP	0.59 (-5.14, 6.32)	1.04 (-0.38, 2.47)
0% vs. 100%	-1.31 (-6.29, 3.66)	0.69 (-0.55, 1.92)
FP vs. 100%	-1.90 (-7.64, 3.83)	-0.36 (-1.79, 1.07)

Table 8-16. The between-treatment difference and 95% confidence intervals (CI) for mean $\delta^{13}C$ and $\delta^{15}N$ for Coastal Tailed Frog larvae and post-metamorphs. P-values for estimates are only shown when the treatment comparison was $P \leq .1$. Estimates for contrasts with a significant P-value are in bold font. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

	$\delta^{13}C$		**$\delta^{15}N$**	
Post 8 Contrast	**Estimate (CI)**	**P**	**Estimate (CI)**	**P**
Coastal Tailed Frog – Larvae				
100% vs. REF	**5.20 (3.20, 7.19)**	**<0.001**	**2.56 (0.98, 4.15)**	**0.006**
FP vs. REF	-1.17 (-4.17, 1.82)	0.393	0.66 (-1.71, 3.04)	0.538
0% vs. REF	**5.63 (3.64, 7.63)**	**<0.001**	**2.20 (0.61, 3.78)**	**0.013**
0% vs. FP	**6.81 (3.65, 9.96)**	**0.001**	1.54 (-0.97, 4.04)	0.195
0% vs. 100%	0.44 (-1.80, 2.67)	0.664	-0.37 (-2.14, 1.40)	0.646
FP vs. 100%	**-6.37 (-9.53, -3.22)**	**0.002**	-1.90 (-4.40, 0.60)	0.118
Coastal Tailed Frog – Post-metamorphs				
100% vs. REF	-0.53 (-1.89, 0.83)	0.375	0.37 (-1.14, 1.88)	
FP vs. REF	**-2.84 (-5.04, -0.63)**	**0.019**	-1.20 (-3.47, 1.07)	
0% vs. REF	-1.25 (-2.84, 0.35)	0.105	0.04 (-1.69, 1.77)	
0% vs. FP	1.59 (-0.80, 3.99)	0.153	1.24 (-1.30, 3.78)	
0% vs. 100%	-0.72 (-2.45, 1.01)	0.343	-0.33 (-2.22, 1.56)	
FP vs. 100%	**-2.31 (-4.61, -0.01)**	**0.049**	-1.57 (-3.96, 0.82)	

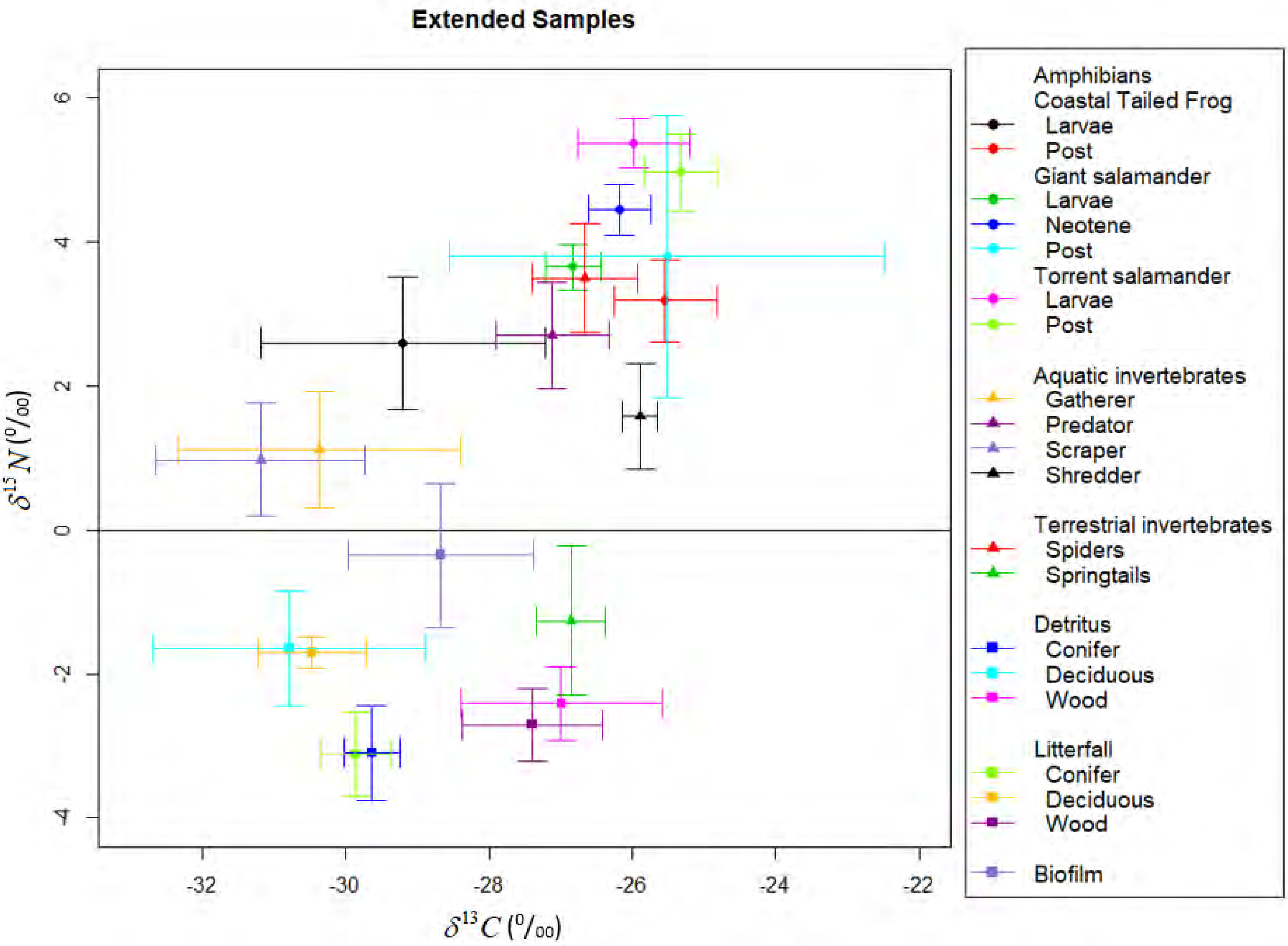

Figure 8-15. Plot of δ^{13}C versus δ^{15}N for all sample categories analyzed for Post 8. This figure illustrates the divergence between the organic matter (squares) and macroinvertebrate (triangles) sample categories, clustered towards the lower left of the plot, and the amphibian values (circles) towards the upper right. The isotope values for each sample type are shown with 95% confidence intervals.

8-5. DISCUSSION

Modification of forest canopy density has often been associated with shifts in trophic system organization along fish-bearing streams in the coastal rainforests of the Pacific Northwest (Murphy *et al.* 1981; Bilby and Bisson 1992; Kiffney *et al.* 2014). Reductions in riparian vegetation, as often occurs with timber harvest, has the potential to influence food web organization (Richardson *et al.* 2005), where increased light may increase instream algal production (Murphy *et al.* 1981), which may increase abundance of invertebrate scrapers that ingest algae (Newbold *et al.* 1980; Hawkins *et al.* 1982). Increased growth rate, production and/or density following reductions in riparian canopy cover has been reported for Cutthroat Trout, *Oncorhynchus clarkii* (Murphy and Hall 1981), juvenile Coho Salmon, *O. kisutch* (Bilby and Bisson 1992; Kiffney *et al.* 2014) and Steelhead, *O. mykiss* (Wilzbach *et al.* 2005).

Nearly all research on trophic response to canopy modifications to date has focused on fish-bearing streams. In these larger stream systems, it has been demonstrated that increased light can increase the proportion of algae in streambed biofilm sufficiently to affect stable isotope values (Mulholland *et al.* 2000). Very little evaluation of the response of small headwater channels to canopy modification has been attempted. In Washington, Riparian Management Zone (RMZ) buffering requirements for small non-fish-bearing streams are less rigorous than those for fish-bearing streams (WFPB 2001). Hence, small streams may be more likely to experience changes in trophic organization related to canopy modification than larger stream reaches that experience only minor increases in light input following timber harvest (Groom *et al.* 2011).

Contrary to our expectations, we found limited evidence of an alteration in the isotopic composition of biofilm in response to riparian buffer treatments. The biofilm isotope values were variable, likely reflecting spatial and temporal differences in biofilm composition. Nonetheless, the stable isotope signals suggest that the organic matter sources supporting the biofilm were not appreciably changed as a result of buffer treatments. This interpretation of the biofilm isotope data is supported by the periphyton response we evaluated through two years post-harvest (McIntyre *et al.* 2018, Chapter 13 – *Biofilm and Periphyton*). In this evaluation, post-harvest changes in periphyton biomass and chlorophyll *a* content did not vary among treatments, including the reference, in the two years following harvest. We initially intended to couple the response of biofilm biomass and chlorophyll *a* levels with our stable isotope data to determine if a reduction in canopy cover caused an increase in availability of autotrophic organic matter in headwater streams. However, the lack of response of biofilm and periphyton in the two years following harvest contributed to our choice to discontinue this sampling past two years post-harvest. As such, we are unable to compare our stable isotope results for algae to a response of periphyton biomass and chlorophyll *a* eight years post-harvest. We were also not able to obtain a sample of pure algal material to assess changes in autotrophic support of the food web as a part of our stable isotope analysis; we were only able to sample biofilm, which is a mixture of allochthonous and autochthonous organic matter sources (France 1995). The lack of an isotope value for one of the key end members of the trophic web prevented us from being able to evaluate if algal consumption by primary consumers was affected by the riparian buffer treatments. As a result, we were only able to assess whether there was any consistent change in biofilm isotopic values related to the buffer treatments.

Though the treatment effect on biofilm response was limited, we did observe a decline in canopy cover in riparian buffer treatments immediately post-harvest (see Chapter 4 – *Stream Temperature and Cover* in this report). Canopy cover in the 100% treatment, which averaged 93% pre-treatment, declined to 91% immediately post-harvest, continued to decline to 88% four years post-harvest, and then increased to 96% eight years post-harvest (which coincides with the timing of data collection for our stable isotope analysis). In the FP treatment, we observed a decline from a pre-treatment value of 98% to 92% immediately post-harvest, followed by a continued decline to 76% three years post-harvest, followed by an increase to 91% eight years post-harvest. Reductions in canopy cover were most dramatic in the 0% treatment, where we observed a decline from 95% to 51% immediately post-harvest, with an erratic increase to 80% eight years post-harvest. Reductions in stream shading as a result of forest harvest has been well documented in the literature (Janisch *et al.* 2012; Rex *et al.* 2012; Schuett-Hames *et al.* 2012). Though we observed some recovery of canopy cover in all riparian buffer treatments during the years after harvest, canopy cover in the FP and 0% sites remained below pre-treatment values. However, high wood loading from logging slash and windthrow from riparian buffers in harvested sites (see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report) may have contributed shading to the stream channel that may have been partially responsible for the lack of autotrophic response in our analysis of stable isotopes.

Though we did not detect a difference in the biofilm isotopic values between the pre- and post-harvest period, we did observe a decrease in mean ^{13}C for giant salamander larvae in the FP treatment and an increase in the 100% treatment in the two years following harvest, after controlling for temporal changes in the references (McIntyre *et al.* 2018, Supplement 2 – *Stable Isotopes Analysis*). Over this same period, we observed a decrease in mean ^{15}N for gatherer invertebrates in the FP and 0% treatments. However, because we found no evidence of an increase in algal content in the biofilm, these changes did not support our hypothesis that canopy modification resulted in increased trophic support from autotrophic sources. Furthermore, the lack of difference in isotope values between biofilm and leaf litter suggested that, in our streams, the biofilm derives primarily from terrestrial material.

The biofilm δ^{13}C values eight years post-harvest were similar to those for the three types of litterfall and instream detritus sampled (conifer, deciduous, wood; see **Figure 8-15**). δ^{15}N values for biofilm were slightly higher than those for the litterfall and instream detritus samples, perhaps due to fractionation (alteration in isotope ratio due to biological activity) from microbial processing of the organic material forming the biofilm. Though we observed an increase in δ^{13}C for biofilm eight years after harvest in the FP and 100% treatments relative to the reference, we detected no change in the 0% treatment. This difference in response among buffer treatments could have been caused by a change in the forms of dissolved organic matter being delivered to the FP and 100% treatments following timber harvest. However, the lack of response at the 0% treatment is inconsistent with this hypothesis.

We did find evidence that isotope values varied among treatments for several of the other sample categories. However, these results should be interpreted with an appreciation of the uncertainty associated with analyzing invertebrates by functional feeding group rather than individual taxa. We used feeding groups because we were unable to consistently obtain enough individuals of a single taxon in each feeding group on each sample date. Lumping samples by feeding group undoubtedly added some variability to the isotope values, especially for predators, which were only included in the eight year post-harvest comparison among treatments.

δ^{15}N for macroinvertebrate gatherers varied among treatments both two and eight years post-harvest. However, between-treatment differences two years post-harvest did not appear to be associated with the intensity of the buffer treatment, i.e., there was no relationship between the length of the riparian buffer retained in the RMZ and isotope values. The greatest decline in δ^{15}N relative to the reference was in the FP treatment two years post-harvest, at which time there was no evidence of a difference between the 100% or 0% treatments and the reference. Eight years after harvest, however, we had evidence of a difference among treatments, with an increase observed in the 0% treatment but no change in the reference, 100% and FP treatments. Here again, it is unlikely that the changes in δ^{15}N levels were due to an increase in autotrophic production in the streams since δ^{15}N values for algae are typically higher than those for terrestrial organic matter (Mulholland *et al.* 2000).

We also found evidence that isotope values varied among treatments for Coastal Tail Frog; however, we could only evaluate between-treatment differences eight years post-harvest due to small sample sizes for this taxon pre-harvest and in the two years post-harvest. Eight years post-harvest, Coastal Tailed-Frog larvae and post-metamorphs both exhibited differences among buffer treatments, especially for δ^{13}C values of the larvae. As Coastal Tailed Frog larvae feed primarily on biofilm, changes in isotope values in the frogs might be expected to be consistent with changes in the isotope values of the biofilm. We observed an increase in δ^{13}C values for biofilm eight years after harvest for the 100% and FP treatments, but not at the 0% sites. The Coastal Tailed Frog larvae response was seen at the 100% and 0% sites, but not the FP sites. As the biofilm and Coastal Tailed Frog larvae responses in δ^{13}C do not completely correspond, we cannot attribute the difference in δ^{13}C in the Coastal Tailed Frog larvae to a change in the isotopic values of their diet. However, samples sizes for Coastal Tailed Frog larvae in all treatments and years was very small, which undoubtedly contributed to variability and possibly limited our ability to detect a difference among treatments where one may or may not have existed.

Although we found a limited and inconsistent treatment response among the amphibians, the δ^{13}C versus δ^{15}N comparison of stable isotope data for eight years post-harvest (**Figure 8-15**) did provide us with some indication of their diet. In the comparison of stable isotope ratios for pre- and two years post-harvest, δ^{13}C and δ^{15}N values for salamanders and adult Coastal Tailed Frogs appeared to be too high for them to be relying on a diet of the aquatic invertebrates we had collected (McIntyre *et al.* 2018, Supplement 2 – *Stable Isotope Analysis*). However, the stable isotope samples taken pre- and two years post-harvest did not include some potentially important food items for the amphibians. To provide a clearer indication of potential food sources, we sampled a broader array of invertebrates eight years post-harvest. These samples included two types of terrestrial invertebrates that were not previously sampled: springtails and spiders.

The stable isotope data eight years post-harvest did indicate that Coastal Tailed Frog larvae were ingesting primarily biofilm as the δ^{13}C values were similar in biofilm and the frog larvae and the δ^{15}N values were approximately 3.5‰ higher in the larvae than in biofilm, a difference consistent with fractionation associated with trophic exchanges (DeNiro and Epstein 1978) (**Figure 8-15**). The post-metamorphic Coastal Tailed Frogs, torrent salamanders and giant salamanders all exhibited stable isotope values that suggested that aquatic gatherers and scrapers and terrestrial springtails were not a significant component of their diet (**Figure 8-15**). Both the δ^{15}N and δ^{13}C values in the amphibians were much higher than those found in these invertebrates. Typically, δ^{15}N values will increase about 3‰ with each trophic exchange and δ^{13}C values about 1‰, due to fractionation (DeNiro and Epstein 1978). The difference in isotope ratios between the amphibians and aquatic gatherer and scraper invertebrates and terrestrial springtails is too great to be attributed solely to fractionation. The amphibian values were consistent with a diet of aquatic predators and shredders, and terrestrial spiders. Aquatic predators and shredders, and terrestrial spider invertebrates all have δ^{13}C values similar to the amphibians and δ^{15}N values approximately 1 to 4‰ lower than the amphibians, a difference consistent with changes due to fractionation.

Our results are supported by results from an evaluation of diet for larval and post-metamorphic Columbia Torrent Salamander, that found that both aquatic and terrestrial prey were common for this species (O'Donnell and Richart 2012). Consistent with our results, O'Donnell and Richart (2012) found that terrestrial spiders were common prey for post-metamorphs; however, their findings contradict our own in that terrestrial springtails were also a common prey item identified for the species. Consistent with field observations, the amphibian stable isotope values also would be consistent with the possibility that some of the amphibians were preying on other amphibians (Dimitrie *et al.* 2016). It is also possible that the amphibians were utilizing a food resource that we did not sample. For example, we observed a larval Columbia Torrent Salamander regurgitating sawfly larvae (*Neodiprion*) in one of our study sites (Hicks *et al.* 2008). Diets of our focal amphibians are not well studied, however, O'Donnell and Richart (2012) note that their findings for Columbia Torrent Salamander differ from findings for another non-focal species of the same genera (i.e., Southern Torrent Salamander, *R. variegatus*; Bury and Martin 1967). Our interpretation of results is likely complicated by the fact that the analysis of torrent salamanders includes three distinct species and for giant salamanders two distinct species. Coupling stable isotope analysis with a direct evaluation of amphibian diets in future studies would provide a more definitive characterization of trophic relationships of these animals.

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CHAPTER 9 - STREAM-ASSOCIATED AMPHIBIANS

Aimee McIntyre, Reed Ojala-Barbour, Marc Hayes, Jay Jones, and Timothy Quinn

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9-1. ABSTRACT

Amphibians are often considered one of the most susceptible organisms to environmental change. Many headwater streams in the Pacific Northwest support abundant amphibian populations compared to larger streams. Because headwater streams commonly receive less protection during forest management than larger streams, it is important to evaluate stream-associated amphibian response to timber harvest. We evaluated the effect of timber harvest, with variable riparian buffer treatments and an unharvested reference, on three stream-breeding amphibian groups (Coastal Tailed Frog [*Ascaphus truei*], and torrent [*Rhyacotriton*] and giant [*Dicamptodon*] salamanders) in western Washington. We compared the response of amphibian density and body condition to clearcut timber harvest in a long-term (2006–2016), spatially blocked and replicated Before-After Control-Impact (BACI) study in western Washington. We included three alternative riparian buffer treatments and references that were not harvested. Buffer treatment basins were clearcut outside of a riparian management zone (RMZ) and received one of three riparian buffer treatments: 100% treatment (two-sided 50-ft width riparian buffer along the entire length of the Type Np stream network), Forest Practices (FP) treatment (two-sided 50-ft width riparian buffer along at least 50% of the Type Np stream network, according to current Forest Practices Rules), and 0% treatment (harvested to the stream edge with no riparian buffer). To estimate stream network-wide density, we used a combination of light-touch sampling and rubble rouse techniques and employed replicate sampling on a subset of stream reaches to adjust counts for imperfect detection. To estimate density in the lower Np reach, we used rubble-rouse sampling in the downstream-most 200 m reach directly upstream from the fish end point. We sampled for amphibians in 17 study sites during the summer low flow period in two years preceding harvest (Pre 3 & 2), the two years immediately following harvest (Post 1 & 2), and seven and eight years after harvest (Post 7 & 8).

In Post 1 & 2, we estimated a 106% increase in stream network-wide larval Coastal Tailed Frog density in the FP treatment compared to the pre-harvest period after controlling for temporal changes in the reference. Conversely, we estimated a decline in larval density in the lower Np reach for the FP and 0% treatments, respectively, over the same period. In Post 7 & 8, we estimated a 65%, 93% and 84% decline in stream network-wide larval tailed frog density in the 100%, FP and 0% treatments, respectively, with similar declines in the lower Np reach. For stream network-wide post-metamorphic tailed frog density, we estimated a 57% decrease in the 100% treatment, and a 961% increase in the 0% treatment in Post 1 & 2; however, there was large uncertainty in all estimates for pairwise comparisons that included the 0% treatment. We did not find clear evidence of a change in post-metamorphic tailed frog density in the lower Np reach over the same period. In Post 7 & 8, we estimated a 71% and 97% decline in stream network-wide post-metamorphic density in the 100% and FP treatments, respectively, but again did not find clear evidence of a change in the lower Np reach.

In Post 1 & 2, we estimated a 198% increase in stream network-wide torrent salamander density in the 0% treatment compared to the pre-harvest period after controlling for temporal changes in the reference. However, in Post 7 & 8 the initial increase in the 0% treatment was no longer evident, and we estimated a 64% decline in torrent salamander density in the FP treatment. In Post 1 & 2, we also estimated a 64% decline in stream network-wide giant salamander density in

the FP treatment. We did not find clear evidence of a difference in torrent or giant salamander density in the lower Np reach for any treatment or period.

Results for stream network-wide and lower Np reach densities were somewhat consistent, especially in Post 7 & 8, however, there were some notable exceptions. The two approaches evaluated amphibian density at different spatial scales. Notably, there was the potential for migration into or out of the lower Np reach, which may have been confounded with treatment. As such, we rely on our stream network-wide estimates of amphibian densities to draw conclusions since they are based on estimates of abundance throughout the entire Type Np stream network. Furthermore, estimates that control for imperfect detection have less bias than classic approaches that do not account for detection (e.g., rubble-rouse).

Our study was designed to evaluate treatment effects, not the mechanisms behind potential changes in amphibian abundance. However, amphibian abundance has been associated with stream temperature, overstory canopy, primary productivity, wood loading, sediment retention, flow dynamics, stream and bank morphology, and nutrients, all of which have the potential to be affected by timber harvest. Based on our results from the broader study, two possible reasons for the decrease in tailed frog and torrent salamander abundance in some treatments and years may be related to increased stream temperature or sedimentation. Additionally, the region experienced a drought that started in 2013 and lasted through the summer of 2015 (Post 7). It is possible that drought conditions exacerbated the effects of buffer treatment by increasing egg or larval mortality, reducing reproduction, or inducing adult emigration from buffer treatment sites. Continued monitoring of the amphibian response to treatment is strongly recommended to expand on our understanding of the long-term impacts of timber harvest and variable length buffers on stream-associated amphibians.

9-2. INTRODUCTION

Pacific Northwest headwater streams support stream-associated amphibian abundances that are greater than in larger, fish-bearing river systems (see Richardson and Danehy 2007). Fish densities decline in smaller streams, offering amphibians a refuge from fish predators common in higher-order streams (Richardson and Danehy 2007). In fact, stream-associated amphibians often replace fish as the dominant vertebrate predators in and along headwater streams (Burton and 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 and Steventon 1999; Grialou *et al.* 2000; Stoddard and Hayes 2005) and water temperature (Bury 2008; Pollett *et al.* 2010), are affected by timber harvest and associated activities (Grizzel and Wolff 1998; Johnson and Jones 2000; Jackson *et al.* 2001; Moore *et al.* 2005; Janisch *et al.* 2012; Araujo *et al.* 2013). Stream-associated amphibians may be particularly predisposed to large variations in population size or local extirpation because of disturbance, including timber harvest (Bury and Corn 1988; Fagan 2002). Once extirpated, opportunities for recolonization from adjacent headwater streams may be restricted by larger downstream reaches (Lowe and Bolger 2002; Richardson and 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 as a result 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 and colleagues (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 and colleagues (2007) found that giant salamander and Coastal Tailed Frog populations declined in the several years immediately following timber harvest. Conversely, others have not detected a correlation between amphibian abundance and forestry activities, including for Coastal Giant Salamander and Coastal Tailed Frog (Murphy and 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 (Welsh and Ollivier 1998; Lawler *et al.* 2010), 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). Furthermore, Cope’s and Coastal Giant Salamanders are extremely difficult to differentiate in the field (Nussbaum 1970, 1976; Good 1989; Foster and Olson 2014), and hybridization is known to occur (Spear *et al.* 2011; Spear *et al.* 2019), so Coastal Giant Salamander had to be included by default.

There is substantial uncertainty regarding the effectiveness of the FPHCP buffer strategy for Np streams as it relates to impacts on stream-associated amphibians. To address these uncertainties, we used a basin-scale approach to compare changes in stream-associated amphibian densities and body condition in response to buffering strategies that varied in proportion of stream length buffered eight years post-harvest. Treatments included no buffering (0% treatment), partial buffering using the FPHCP prescription (FP treatment) and complete buffering (100% treatment).

We also evaluated the response of Coastal Tailed Frog and Cope’s and Coastal Giant Salamander genetics. Genetic monitoring provides a complementary approach to demographic monitoring and can provide additional information on a population’s response to disturbance. Though stream amphibians are most easily detected as larvae (Spear and Storfer 2008; Kroll *et al.* 2010), larval numbers may not accurately represent adult population status since a high number of larvae can be produced by only a few adults (Goldberg and Waits 2010). Genetic data can assess levels of effective population size or reductions in population size that are not immediately obvious demographically (Luikart *et al.* 1998; Garza and Williamson 2001). We measured pre-harvest genetic diversity and genetic differentiation within and among populations to provide insight into trends in population size and identify the levels of migration among sites (Spear *et al.* 2011). We evaluated the response of amphibian genetic diversity and differentiation to our variable length riparian buffer treatments in the seven and eight years following harvest (reported on separately; Spear *et al.* 2019). If there was a treatment effect on stream-associated amphibian genetics, one of the most immediate impacts we thought we might observe genetically was an increase in the number of full siblings relative to total sample size. We hypothesized such an effect because a reduction in population size should lead to fewer breeding individuals and thus fewer family groups represented (Spear *et al.* 2019).

We also explored stable isotopes in headwater streams as a method of evaluating whether variable length riparian buffer treatments affected the trophic position of stream-associated amphibians or caused changes in the primary energy source supporting food webs in small streams. Modification of forest canopy density along fish-bearing streams has often been associated with shifts in trophic system organization, a response that has been consistently reported for streams in the coastal rainforests of the Pacific Northwest (Murphy and Hall 1981; Bilby and Bisson 1992; Kiffney *et al.* 2014). However, nearly all research on trophic response to canopy modifications has focused on fish-bearing streams, where it has been demonstrated that increased light can increase the proportion of algae in streambed biofilm sufficiently to affect stable isotope values (Mulholland *et al.* 2000). Conversely, very little evaluation of canopy modification impacts to stable isotopes has been attempted in small headwater channels. In Washington, riparian management zone (RMZ) buffering requirements for small non-fish-bearing streams are less rigorous than those for fish-bearing streams (WFPB 2001). Hence, small streams may be more likely to experience changes in trophic organization related to canopy modification than larger stream reaches that experience relatively minor increases in light input following timber harvest (Groom *et al.* 2011). Vertebrate animals such as fish and stream-associated amphibians may also respond to reductions in riparian vegetation with an increase in growth rate, production, and/or density (Murphy and Hall 1981; Bilby and Bisson 1992; Wilzbach *et al.* 2005; Kiffney *et al.* 2014; Kaylor and Warren 2017). Stable isotope analysis was included in this study as a method of evaluating whether variable length riparian buffer treatments caused changes in the primary energy source supporting food webs in small streams (see Chapter 8 – *Stable Isotopes Analysis* in this report).

9-3. METHODS

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).

9-3.1. DATA COLLECTION

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.

9-3.1.1. Light-touch Sampling

Researchers commonly use the light-touch method (Lowe and Bolger 2002) for headwater amphibians in the Pacific Northwest to establish occupancy or abundance (Steele *et al.* 2003; Russell *et al.* 2004; Quinn *et al.* 2007). 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 visually 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 9-3.1.1.a.** *Detection estimation*).

9-3.1.1.a. 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 9-1**). 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. We accomplished our goal 90.2% of the time; however, due to schedules and other activities that limited site accessibility (e.g., road closures), more than 1 day did fall between repeat visits for the remaining 9.8% of surveys. Specifically, there were two days between surveys for 1.1% of passes, 4 days for 6.5% of passes, 5 days for 1.7% of passes, 6 days for 0.2% of passes and 7 days for 0.4% 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 ± 1°C).

Table 9-1. 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.

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

9-3.1.1.b. Obstructed reaches

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 (hereafter, obstructed reaches; **Figure 9-1**; *note that obstructed reaches were identified independently for wood loading and amphibian density data collection and analyses*). We conducted a basin-wide census to determine the locations and lengths of obstructed reaches. When encountered, we did not attempt to sample obstructed reaches for amphibians but recorded the location and length for each sample interval. Locations of obstructed reaches were identified by the individual conducting amphibian sampling and was based on professional judgement related to whether accessibility or vision was impaired such that the survey did not meet protocol. Even though obstructed reaches prevented us from sampling some intervals or parts of intervals, we met our minimum sample length requirement for light-touch sampling in all study sites and years without modifying the intensity of our sample effort. For a given site and year, we sampled a minimum of 72% of the stream length for basins ≤800 m long and 26% of the stream length for basins >800 m long.

Figure 9-1. Examples of obstructed reaches at Hard Rock Study sites for unbuffered stream reaches in Post 1 (left panel) and Post 7 (right panel).

9-3.1.2. Rubble-rouse Sampling

In addition to light-touch sampling, we conducted a more intensive rubble-rouse sampling on a subset of stream meters (see below) following standard procedures, which assumes a detection probability of 1 (Bury and Corn 1991). Rubble-rouse sampling was conducted after light-touch sampling was complete. We blocked off a section of the stream with nets placed at the downstream and upstream boundaries of the plot. While searching for animals, we removed all coarse substrate large gravel-sized or larger (≥32 mm diameter) from within the wetted channel until only unconsolidated fines and small gravel (<32 mm diameter) remained, and to a depth of 30 cm or until bedrock was reached. We sifted the remaining unconsolidated fine substrates and carefully removed the nets, examining them for animals. We replaced substrates in the channel and returned animals to the stream.

9-3.1.2.a. Lower Np reach rubble-rouse sampling

We conducted rubble-rouse sampling in the 200 m stream reach immediately upstream of the F/N break, hereafter, lower Np reach, in Pre 2, Post 1, Post 2, Post 7 and Post 8. We used a stratified random sampling approach, attempting to establish one, 1-m long rubble-rouse plot within each of 20 contiguous 10-m long sample intervals . For some sites and years hydrological conditions or obstructed reaches required installation of multiple plots in the same 10-m stream interval (consecutive plots were at least 1 m apart) or relocation of plots farther upstream. This resulted in less than 20 plots, as well as placement of plots more than 200 m upstream of the F/N break, for some sites and years. At each plot location, we measured the stream wetted width to the nearest decimeter to calculate the area (A) of each plot as:

$$A = l * w \qquad \textbf{(Eq. 9-1)}$$

where: l is the length of the rubble-rouse plot, and
w is the wetted stream width at the plot location.

9-3.1.2.b. Obstructed reach rubble-rouse sampling

To estimate amphibian density in obstructed reaches (i.e., obstructed by downed trees or logging slash that prevented access to the stream or made it impossible to see under cover objects; see **Section 9-3.1.1.b.** *Obstructed reaches* and **Figure 9-1**), we sampled a systematic subset of reaches using rubble-rouse sampling (hereafter, obstructed plots). The target obstructed plot length was 3 m. However, obstruction features such as the size and density of wood pieces resulted in plot lengths that ranged from 2 to 4 m to avoid disturbance of the stream prior to net placement. We prioritized placement of the downstream nets followed by the upstream nets with as little wood disturbance or removal as possible to avoid amphibian movement into or out of our sample plots. For each plot, we recorded the length and estimated density as animals per linear meter. We removed in-channel wood using handheld saws, loppers and clippers (**Figure 9-2**). Unlike sampling of rubble-rouse plots in the lower Np reach, we removed step channel units. We replaced all removed substrates (including wood) back in the stream prior to releasing amphibians. We placed obstructed plots throughout study sites based on the length of obstructed stream. Our goal was to sample two plots for streams with 5% to 10% of the stream length obstructed, three for streams with >10% to 20% obstructed, four for streams with >20% to 40% obstructed, and six when >40% was obstructed. We distributed plots by buffer type (reference, buffered or unbuffered) and stream order (first- and second-/third-order; Strahler 1952; **Table 9-2**).

In Post 1 and Post 2, density for some taxa appeared to be greater in obstructed reaches than in nearby reaches that were not obstructed by instream wood and slash (McIntyre *et al.* 2018, Chapter 15 – *Stream-associated Amphibians*). Based on this observation, we evaluated whether amphibian density in obstructed reaches in Post 7 and Post 8 differed from reaches that were not obstructed. To do this, we sampled rubble-rouse plots in unobstructed reaches (hereafter, unobstructed plots) using the same rubble-rouse methodology as in obstructed plots. Each obstructed plot was paired with an unobstructed plot located between 15 m and 100 m away and in an otherwise similar reach based on tributary, stream order, and buffer type, with a few exceptions.[1] We sampled a total of 43 plot pairs located in 10 study sites in Post 7 and Post 8 (**Table 9-2**). In Post 7 and Post 8, we had very few tailed frog observations, including only a single larva and three post-metamorphic individuals; as a result, we were not able to conduct statistical comparisons for this species.

Figure 9-2. Modified sampling approach used to determine amphibian density in stream reaches that were obstructed by downed trees or logging slash preventing access to the stream or making it impossible to see under cover objects in the Hard Rock Study. In this approach, rubble-rouse sampling was conducted in "obstructed plots" for a subset of obstructed stream reaches. Installation of an obstructed plot (left panel) and removal of wood and logging slash to enable sampling (right panel).

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[1] Due to limited first-order plot locations, the unobstructed paired plot for two obstructed plots were located downstream in a second-order reach. For the same reason, the unobstructed paired plot for one was located in a nearby, but different, first-order tributary.

Table 9-2. The number of obstructed plots sampled using rubble-rouse in the post-harvest period by site, year (Post 1, Post 2, Post 7, Post 8), and stream order (1st, 2nd/3rd). We did not sample obstructed plots in sites when less than 5% of the stream channel length was obstructed in a given year.

Block	Treatment	Post 1		Post 2		Post 7		Post 8	
		1st	2nd/3rd	1st	2nd/3rd	1st	2nd/3rd	1st	2nd/3rd
OLYM	FP	0	0	2	0	2	2	0	1
	0%	2	4	3	3	2	0	2	0
WIL1	REF	0	0	0	0	1	0	1	0
	100%	1	2	1	0	1	2	2	2
	FP	4	0	4	0	3	0	2	0
	0%	2	2	2	2	2[2]	1	1	1
WIL2	REF	0	0	0	0	2	1	0	0
	100%	0	0	1	2	0	3	1[2]	0
	0%	2	1	2	2	2	2	2	2
CASC	FP	2	0	2	0	0	0	0	0
Total		**13**	**9**	**17**	**9**	**15**	**11**	**11**	**6**

[1] In the Post 7 and Post 8 we sampled an unobstructed plot paired with each obstructed plot.

9-3.2. 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 sites.

We collected small tissue samples for all taxa. Our target sample size was 40 samples per site for Coastal Tailed Frog for use in genetic diversity and stable isotope analyses. We collected tissue samples for all giant salamanders for the purpose of genetic differentiation between the species, and in use in our genetic diversity and stable isotopes analyses. The exception was for sites in the Olympic Block, where we detected only Cope's giant salamander in the pre-harvest period, so we used a sample size of 40 per site in the post-harvest period. Since we did not include torrent salamanders in our analysis of genetic diversity, our target sample size for this taxon was 10. In general, we collected tissue samples from all individuals as they were encountered until our minimum sample size was met. 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 all 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 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.

9-3.3. OCCUPANCY DETERMINATION

We summarized amphibian species occupancy by site and year for the focal taxa detected with our stream network-wide light-touch and rubble-rouse sampling in the lower Np reach. Our intent was to summarize the sites in which we detected each taxon (i.e., 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 these surveys only in the post-harvest period and plots were not equally distributed across all study sites and treatments. In study results, we note observations that confirm occupancy for a species in the rare case that it was detected only in obstructed plots or incidentally during surveys targeted at increasing tissue sample sizes for genetic or stable isotope analyses.

9-3.4. DENSITY ESTIMATION

We estimated amphibian density at two spatial scales: lower Np reach, based on rubble-rouse sampling (hereafter, lower Np density), and stream network-wide, based on light-touch sampling (hereafter, stream network-wide density). 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 and Wake 1992) and the three species use habitats similarly (Jones *et al.* 2005).

9-3.4.1. Lower Np Density

We used data from our 1-m rubble-rouse plots to calculate amphibian density in the lower Np reach by site and year. We calculated amphibian density (D_{spp}) as:

$$D_{spp\ -=} C_{spp} / A \qquad \textbf{(Eq. 9-2)}$$

where: C_{spp} is the sum of captures by species (and stage for Coastal Tailed Frog) across all plots by site and year, and

A is the sum of the areas from all plots by site and year.

We assumed that detection probability in obstructed plots sampled using rubble-rouse methods was 1. This method has been cited by others as providing the most complete census of animals and has been shown to detect more individuals than other common amphibian sampling techniques, including light-touch (Quinn *et al.* 2007). We did not measure stream wetted width for rubble-rouse plots in 2006 and so were unable to calculate amphibian densities for 2006 rubble-rouse counts. As a result, those data are not included in the formal analysis.

9-3.4.2. Stream Network-wide Density

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 (see Chapter 4 – *Stream Temperature and Cover* in this report). 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. 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. The purpose of stream temperature data collection 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 × 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 × 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 and Post 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:

$$\tilde{N}_{ijh} = C \cdot \frac{\sum_k \tilde{N}_{ijhk}}{\sum_k c_{ijhk}} \qquad \textbf{(Eq. 9-3)}$$

where: k indexes plot,
$\tilde{N}_{ijhk}$ is the adjusted plot abundance,
c_{ijhk} is the plot length, and
C = 30 m.

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

$$\tilde{N}_{ij} = \sum_h w_{ijh} \cdot \tilde{N}_{ijh} \qquad \textbf{(Eq. 9-4)}$$

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

9-3.5. BODY CONDITION

We used body length and mass data gathered during rubble-rouse sampling to calculate a scaled mass index (SMI) for individual amphibians (Peig and Green 2009). The SMI accounts for the allometric relationship between mass and a body structure measure (e.g., length) by removing covariation between body size and body components. This in turn allows for the comparison of condition of a given individual with individuals of the same size. MacCracken and Stebbings (2012) verified the utility of the SMI for use with amphibians, concluding that SMI values accurately reflected amphibian energy stores. We calculated the SMI of body condition ($\widehat{M}_i$) as follows:

$$\widehat{M}_i = M_i \left[\frac{L_0}{L_i}\right]^{b_{SMA}} \qquad \textbf{(Eq. 9-5)}$$

where: Mi and Li are the body mass and the linear body measurement of individual *i* respectively,
b_{SMA} is the scaling exponent estimated by the SMI regression of M on L,
L_0 is an arbitrary value of L (e.g., the arithmetic mean for the study population), and
$\widehat{M}_i$ is the predicted body mass for individual *i* when the linear body measure is standardized to L_0.

We used total length (TL) measures for SMI calculations of torrent and giant salamanders because tails are important sites for fat storage in many species (Sheridan and Kao 1998). We did not include giant salamander post-metamorphs in the analysis. Due to small sample sizes, we were unable to conduct SMI comparisons for larval or post-metamorphic tailed frogs. We had length and weight data for 2,435 torrent and 1,505 giant salamanders. We did not include animals in the analysis that were injured (e.g., missing part of a leg or tail, which could indicate prior sampling) or gravid, including 118 (5%) torrent and 166 (11%) giant salamanders.

To calculate SMI, we examined scatter plots of mass versus length for each taxon to identify and remove outliers. We identified outliers as values that were improbable due to biological considerations; for example, an individual of a given length with several times the mass of individuals of a similar length. We identified 53 (2%) torrent and 17 (1%) giant salamanders as outliers. Once outliers were removed, we fit a line to pre-harvest mass and length data on a natural log-log scale and calculated the scaling exponent (b_{SMA}) for the SMI calculation as the slope of the regression divided by Pearson's correlation coefficient *r* (LaBarbera 1989). We used the average pre-harvest length as our L_0 value (Peig and Green 2009, 2010). We calculated the SMI of body condition for each individual. Due to small sample sizes for some taxa, site, and sample year combinations, we averaged the SMI for each taxon and site across years within the pre-harvest (Post 1 and Post 2) and post-harvest (Post 7 and Post 8) periods. We did not have data for torrent salamanders in eight sites/years, four in the pre-harvest period and four in Post 7 and Post 8 (two references, one FP and one 0% treatment in each period).

9-3.6. STATISTICAL ANALYSIS

Analysis of amphibian density and body condition evaluated the generalized null hypothesis:

$$\Delta T_{\mathrm{REF}} = \Delta T_{100\%} = \Delta T_{\mathrm{FP}} = \Delta T_{0\%} \qquad \textbf{(Eq. 9-6)}$$

Where: ΔT_{REF} is the change (post-harvest − pre-harvest) in the reference, and $\Delta T_{100\%}$, $\Delta\ T_{\mathrm{FP}}$, and $\Delta T_{0\%}$ are the changes in the 100%, FP and 0% buffer treatments, respectively.

We used a generalized linear mixed effects model to evaluate this pre- versus post-harvest hypothesis (McDonald *et al.* 2000). We used the Mixed Procedure in SAS (SAS Institute Inc. 2013) for the analysis of amphibian body condition and the GLIMMIX Procedure for analyses of density (i.e., lower Np and stream network-wide density). In both models, block and site were random effects and the fixed effects were year, treatment, and the treatment × year interaction. We evaluated the null hypothesis with a Wald-type test using linear contrasts of the model fixed effects. We constructed the contrasts to test the difference in mean response for pre- and post-harvest periods, where period corresponded to all years in either the pre- or post-harvest condition. If the period × treatment contrast had a P-value ≤0.1, we examined pairwise contrasts to test for differences among the 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%. If the period × treatment contrast was >0.1, we did not report test results for these terms. We used the default containment method for estimating the denominator degrees of freedom with the SAS Mixed Procedure for the analysis of body condition. The uneven distribution of treatments among blocks required using the Kenward-Roger method (Kenward and Roger 1997) for estimating the denominator degrees of freedom in the density analyses (GLIMMIX Procedure). We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals and found no evidence of heteroscedasticity.

The GLIMMIX Procedure for analyses of density reports results on the natural log (ln) scale. Exponentiating the difference in the natural logs of post- and pre-harvest values gives an estimate of the proportional change in the variable 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 results on a natural log scale and, for results that were statistically significant, we present the back-transformed proportional differences in the discussion.

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.

9-4. RESULTS

9-4.1. SUMMARY OF AMPHIBIAN OCCUPANCY

Over the entire study we made 21,194 amphibian observations using light-touch and rubble-rouse techniques in the lower Np reach, of which 98% were focal amphibians. We made 1,994 Coastal Tailed Frog observations, detecting the species in 15 study sites (**Table 9-3**). We made 12,989 torrent salamander observations, detecting the species in all study sites in the pre-harvest period, and 16 study sites in the post-harvest period (**Table 9-3**). We made a total of 5,727 giant salamander observations, detecting one or both giant salamander species (Cope’s and Coastal) in all sites both pre- and post-harvest (**Table 9-3**). The genetic analysis confirmed that Cope’s Giant Salamander was present at all study sites (Spear *et al.* 2011). We did not detect Coastal Giant Salamander in any site in the Olympic block or in the WIL1-0%. There were two instances where torrent salamanders were not detected with stream network-wide and lower Np reach sampling in the post-harvest period, but were detected in obstructed reaches or with supplemental sampling conducted to improve tissue sample sizes for genetic or stable isotope analyses.

Table 9-3. Focal amphibian taxa detected during stream network-wide light-touch and lower Np reach rubble-rouse sampling for all study sites and periods (pre-harvest [Pre 1 and Pre 2], post-harvest [Post 1 & Post 2; Post 7 & Post 8]). Symbols indicate where a focal species was not detected with either sampling method (○), was detected with only one of the two sampling methods (● / ○; stream network-wide / lower Np reach), or was detected with both sampling methods (●).

Treatment	**Block**	**Coastal Tailed Frog**			**Giant Salamanders**			**Torrent Salamanders**		
		Pre	**Post 1 & 2**	**Post 7 & 8**	**Pre**	**Post 1 & 2**	**Post 7 & 8**	**Pre**	**Post 1 & 2**	**Post 7 & 8**
REF	OLYM	●	●	●	●	●	●	●/○	●	●/○
	WIL1	●	●	●	●	●	●	●	●	●
	WIL2_1	●	●	●	●	●	●	●	●	●
	WIL2_2	●	●	●	●	●	●	●	●	●
	WIL3	●	●	●	●	●	●	●	●	●
	CASC	●	●	●	●	●	●	●/○	●	●
100%	OLYM	○	●	●	●	●	●	●	●	●
	WIL1	●	●	●/○	●	●	●	●	●	●
	WIL2	●	●	●	●	●	●	●	●	●
	WIL3	●/○	●	●/○	●	●	●	●	●	●
FP	OLYM	●	●	●	●	●	●	●/○	○*	●/○
	WIL1	●	●	●	●	●	●	●	●	●
	CASC	●/○*	○	○	●	●	●	●	●	●
0%	OLYM	●	○/●	●/○	●	●	●	●/○	●/○	○*
	WIL1	●	●	●	●	●	●	●	●	●
	WIL2	●	●	●/○	●	●	●	●	●	●
	CASC	○	○	○	●	●	●	●	●	●

* Instances where a taxon was not detected with stream network-wide and lower Np reach sampling but was detected in rubble-rouse in obstructed reaches, in second/third pass of detection plots or during incidental sampling conducted to improve tissue sample sizes for genetic and/or stable isotope analyses.

9-4.2. DENSITY

9-4.2.1. Coastal Tailed Frog Larvae

9-4.2.1.a. Stream network-wide

Mean annual stream network-wide larval tailed frog densities ranged from 0.0 to 3.1, 0.0 to 4.5, and 0.0 to 1.2 animals/30 m, respectively, in the pre-harvest period, Post 1 and Post 2, and Post 7 and Post 8 (**Figure 9-3**). We found evidence that treatments differed in the magnitude of change over time (**Table 9-4**; **Figure 9-4**; **Table 9-5**). In Post 1 and Post 2, we estimated the between-treatment comparison for the 100% treatment and reference to be 1.36 (95% credible interval: 0.97, 1.89), or in other words a +36% (95% credible interval: -3%, +89%) change in mean density, compared with the pre-harvest period, after controlling for temporal changes in the references. Likewise, for the FP and 0% treatments we estimated a +106% (+30%, +230%) and +44% (-1%, +115%) change in density, respectively. The 95% credible interval included 1 (i.e., 0%) for the 100% and 0% treatment versus reference comparisons, indicating uncertainty in the direction of the treatment effects. In Post 7 and Post 8, we estimated a -65% (-79%, -43%), -93% (-98%, -79%), and -84% (-92%, -73%) change in density in the 100%, FP and 0% treatments, compared with the pre-harvest period. We observed a +79% (+47% to +117%) within-treatment change in the reference in this same period (**Table 9-4**). We also estimated a -79% (-94%, -33%) and -55% (-79%, -10%) change in the FP and 0% treatments, respectively, compared with the 100% treatment, after adjusting for pre-harvest differences among the treatment sites.

9-4.2.1.b. Lower Np reach

Mean annual larval tailed frog site densities in the lower Np reach ranged from 0.00 to 0.12, 0.00 to 0.12, and 0.00 to 0.05 animals per m^2, respectively, in the pre-harvest period, Post 1 and Post 2, and Post 8 (**Figure 9-5**). We found evidence that treatments differed in the magnitude of change over time (**Table 9-6**; **Figure 9-6**; **Table 9-7**). In Post 1 and Post 2, we estimated the between-treatment comparison for the FP treatment to be 0.35 (95% credible interval: 0.19, 0.64), or in other words a -65% (95% credible interval: -81%, -36%) change in density compared with the pre-harvest period, after controlling for temporal changes in the references. Likewise, we estimated a -77% (-86%, -63%) change in density in the 0% treatment. We observed a +124% (+66%, +200%) within-treatment change in density in the reference in the same period. We also estimated a -34% (-62%, +12%) change in the 100% treatment after controlling for temporal changes in the references, although the 95% credible interval for this comparison included 1 (i.e., 0%), indicating uncertainty in the direction of the treatment effect. Finally, we estimated a -66% (-81%, -36%) change in the 0% compared with the 100% treatment, after adjusting for pre-harvest differences among the treatment sites. In Post 8, we estimated a -97% (-99%, -88%), -97% (-100%, -84%), and -89% (-96%, -72%) change in density in the 100%, FP and 0% treatments, respectively, after controlling for temporal changes in the references.

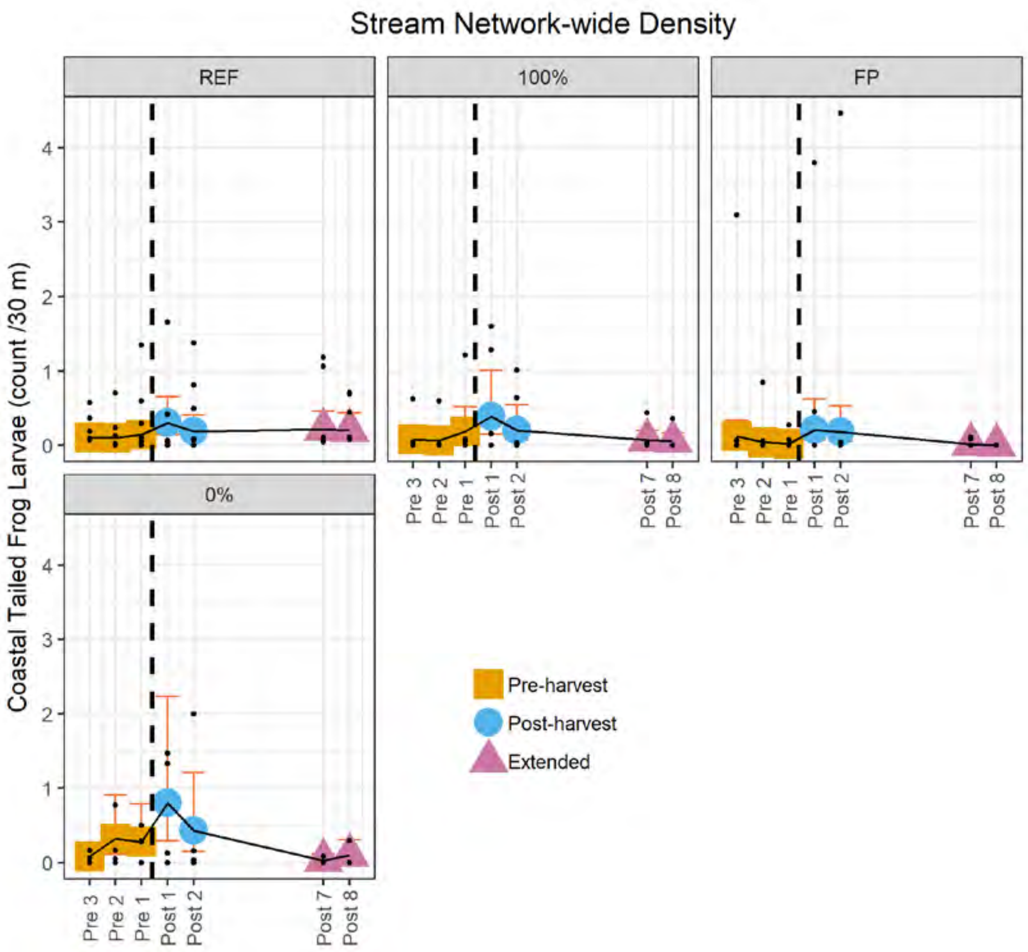

Figure 9-3. Mean stream network-wide larval Coastal Tailed Frog density (animals/30 m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 7 & Post 8). Vertical colored lines show approximate 95% credible intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 9-4. The within-treatment estimate of the proportional change and 95% credible intervals (CI) for mean stream network-wide larval Coastal Tailed Frog density (animals/30 m) between the pre-harvest period and Post 1 & Post 2 and Post 7 & Post 8.

Treatment	**Estimate (CI)**	
	Post 1 & 2	**Post 7 & 8**
REF (n = 6)	2.07 (1.76, 2.44)	1.79 (1.47, 2.17)
100% (n = 4)	2.83 (2.10, 3.75)	0.62 (0.40, 0.98)
FP (n = 3)	4.28 (2.83, 6.68)	0.13 (0.04, 0.38)
0% (n = 4)	2.99 (2.16, 4.29)	0.28 (0.15, 0.48)

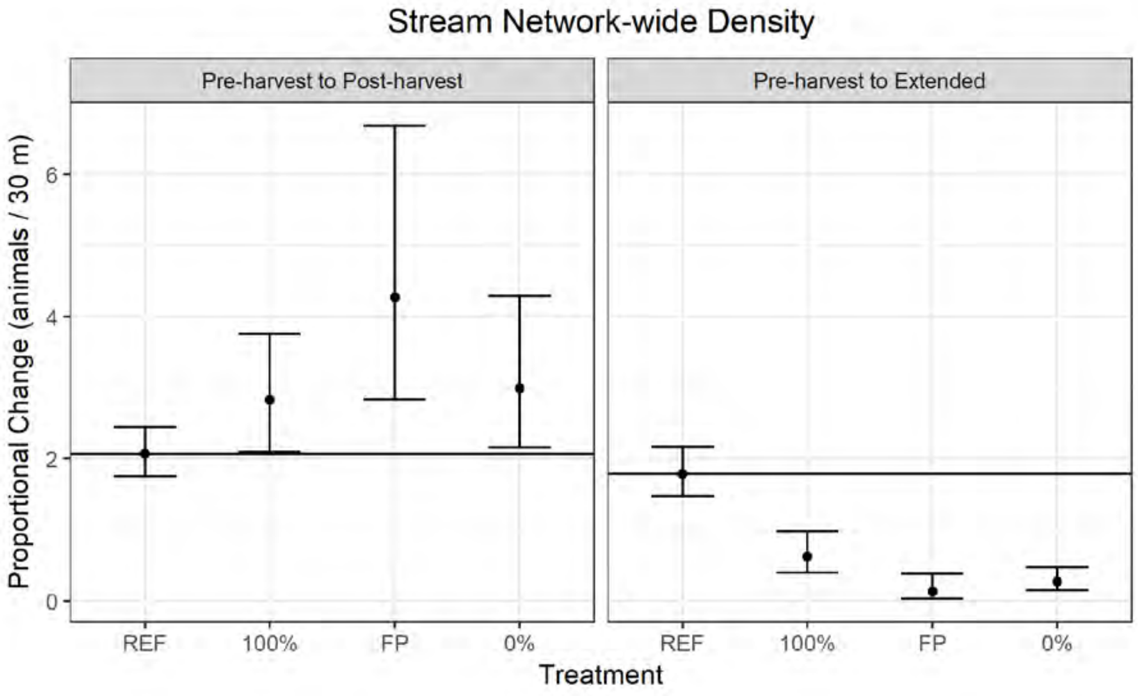

Figure 9-4. The within-treatment estimate of the proportional change and 95% credible intervals for mean stream network-wide larval Coastal Tailed Frog density (animals/30 m; bottom panel) between pre-harvest and post-harvest (Post 1 & Post 2), and extended (Post 7 & Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 9-5. The between-treatment comparison of the proportional change and 95% credible intervals (CI) of the estimates for mean stream network-wide larval Coastal Tailed Frog (animals/30 m) between the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8. Contrasts with credible intervals that do not overlap one 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
100% vs. REF	1.36 (0.97, 1.89)	**0.35 (0.21, 0.57)**
FP vs. REF	**2.06 (1.30, 3.30)**	**0.07 (0.02, 0.21)**
0% vs. REF	1.44 (0.99, 2.15)	**0.16 (0.08, 0.27)**
0% vs. FP	0.70 (0.40, 1.21)	2.08 (0.64, 8.25)
0% vs. 100%	1.06 (0.68, 1.70)	**0.45 (0.21, 0.90)**
FP vs. 100%	1.51 (0.93, 2.58)	**0.21 (0.06, 0.67)**

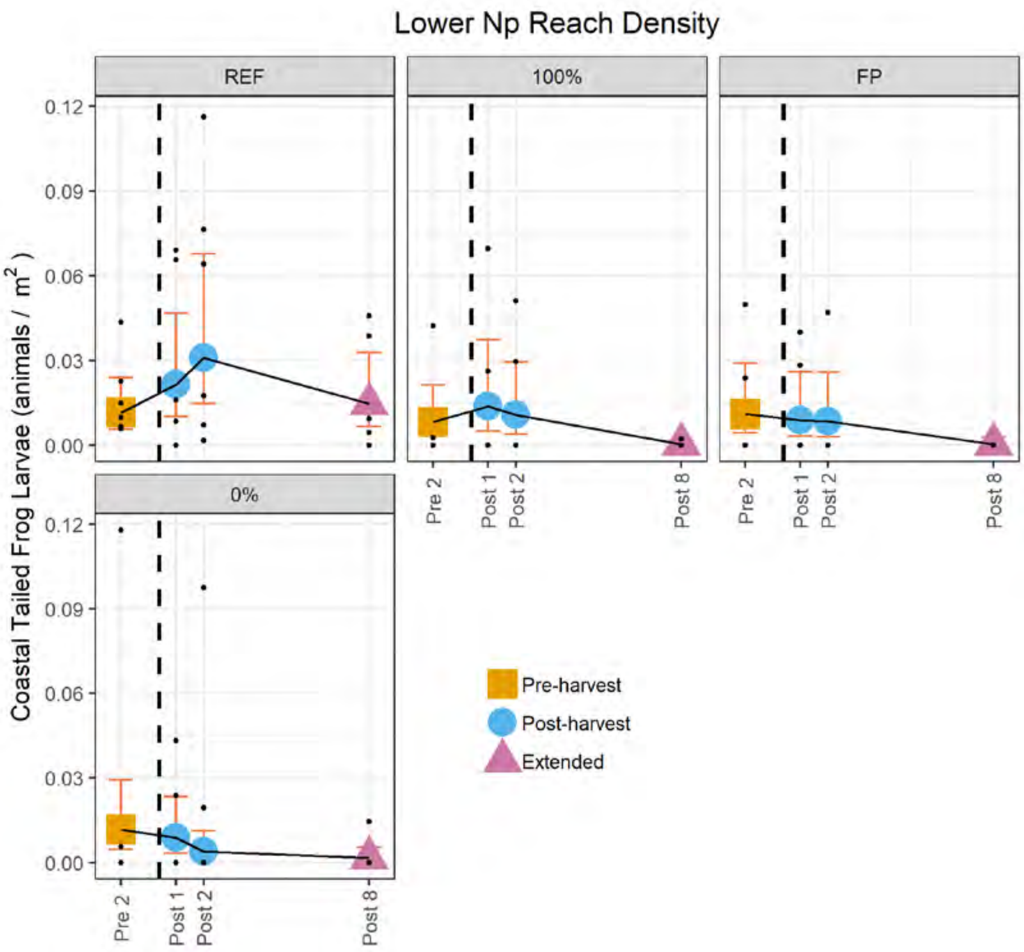

Figure 9-5. Mean lower Np reach larval Coastal Tailed Frog density (animals/m^2) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 8). Vertical colored lines show approximate 95% credible intervals. Vertical dashed lines show the timing of harvest at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 9-6. The within-treatment estimate of the proportional change and 95% credible intervals (CI) for mean lower Np reach larval Coastal Tailed Frog density (animals/m^2) between the pre-harvest period and Post 1 & Post 2, and Post 8.

Treatment	**Estimate (CI)**	
	Post 1 & 2	**Post 8**
REF (n = 6)	2.24 (1.66, 3.00)	1.26 (0.84, 1.97)
100% (n = 4)	1.49 (0.90, 2.32)	0.04 (0.01, 0.15)
FP (n = 3)	0.79 (0.46, 1.34)	0.03 (0.00, 0.19)
0% (n = 4)	0.51 (0.34, 0.75)	0.14 (0.06, 0.32)

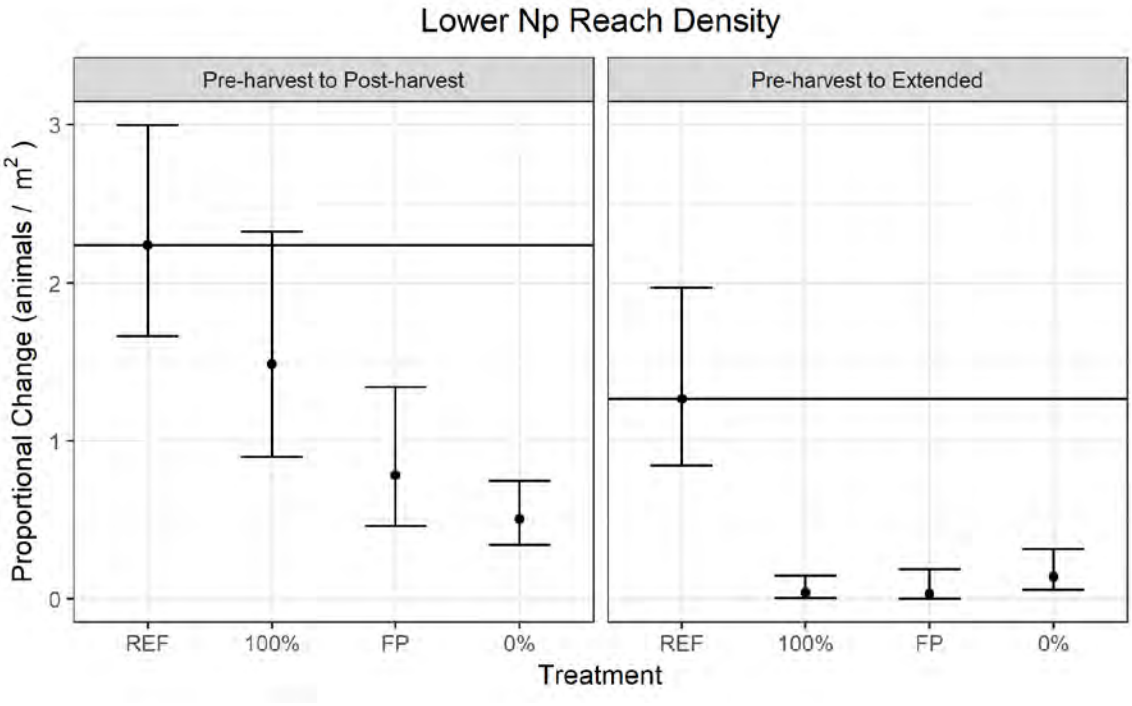

Figure 9-6. The within-treatment estimate of the proportional change and 95% credible intervals for mean lower Np reach larval Coastal Tailed Frog density (animals/m^2) between pre-harvest and post-harvest (Post 1 & Post 2), and extended (Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 9-7. The between-treatment comparison of the proportional change and 95% credible intervals (CI) of the estimates for mean lower Np reach larval Coastal Tailed Frog density (animals/m^2) between the pre-harvest period and Post 1 & Post 2, and Post 8. Lower Np reach contrasts with credible intervals that do not overlap one 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 8**
100% vs. REF	0.66 (0.38, 1.12)	**0.03 (0.01, 0.12)**
FP vs. REF	**0.35 (0.19, 0.64)**	**0.03 (0.00, 0.16)**
0% vs. REF	**0.23 (0.14, 0.37)**	**0.11 (0.04, 0.28)**
0% vs. FP	0.65 (0.33, 1.23)	4.29 (0.58, 50.94)
0% vs. 100%	**0.34 (0.19, 0.64)**	3.62 (0.70, 23.94)
FP vs. 100%	0.53 (0.27, 1.10)	0.85 (0.05, 10.90)

9-4.2.2. Coastal Tailed Frog Post-metamorphs

9-4.2.2.a. Stream network-wide

Mean annual stream network-wide post-metamorphic tailed frog densities ranged from 0.0 to 2.2, 0.0 to 2.5, and 0.0 to 1.3 animals/30 m, respectively, in the pre-harvest period, Post 1 and Post 2, and Post 7 and Post 8 (**Figure 9-7**). We found evidence that treatments differed in the magnitude of change over time (**Table 9-8**; **Figure 9-8**; **Table 9-9**). In Post 1 and Post 2, we estimated the between-treatment comparison for the 100% treatment and reference to be 0.43 (95% credible interval: 0.27, 0.69), or in other words a -57% (95% credible interval: -73%, -31%) change in mean density, compared with the pre-harvest period, after controlling for temporal changes in the references. Conversely, we estimated a +961% (+381%, +2,448%) change in mean density in the 0% treatment. We observed a +94% (+46%, +156%) within-treatment change in the reference in this same period (**Table 9-8**). We also estimated a -49% (-79%, +17%) change in the FP treatment, after controlling for temporal changes in the references; however, the 95% credible interval for this comparison included 1 (e.g., 0%), indicating uncertainty in the direction of the treatment effect. Finally, we estimated a +2,358% (+954%, +5,993%) and +1,972% (+582%, +7,039%) change in the 0% treatment compared with the 100% and FP treatments, after adjusting for pre-harvest differences among the treatment sites. Note the large uncertainty in all estimates for pairwise comparisons that included the 0% treatment.

In Post 7 and Post 8, we estimated a -71% (-82%, -52%) and -97% (-99%, -86%) change in mean post-metamorphic density in the 100% and FP treatments, respectively, compared to the pre-harvest period, after controlling for temporal changes in the reference (**Table 9-9**). We also estimated a -60% (-88%, +38%) change in the 0% treatment; however, the 95% credible interval for this comparison included 1 (e.g., 0%), indicating uncertainty in the direction of this effect. We observed a +146% (+86%, +227%) within-treatment change in the reference in this same period (**Table 9-8**). We estimated a -88% (-98%, -51%) change in the FP treatment compared to the 100% treatment, after adjusting for pre-harvest differences among the treatment sites. Finally, we estimated a +1,051% (+85%, +7,987%) change in the 0% treatment compared with the FP treatment, after adjusting for pre-harvest differences among the treatment sites; however, there was large uncertainty in this estimate.

9-4.2.2.b. Lower Np reach

Mean annual post-metamorphic tailed frog densities in the lower Np reach ranged from 0.00 to 0.03, 0.00 to 0.06, and 0.00 to 0.02 animals per m^2, respectively, in the pre-harvest period, Post 1 and Post 2, and Post 7 and Post 8 (**Figure 9-9**). We did not find clear evidence that density varied among treatments (P = 0.16), however, there was large uncertainty in some of the individual treatment estimates (**Table 9-10**; **Figure 9-10**; **Table 9-11**). This uncertainty was especially large for the 0% treatment in Post 1 and Post 2, for which we estimated a post-harvest change in density of +419% (95% confidence interval: -32%, +3,843%), +122% (-76%, +1,976%), and +1,505% (+60%, +16,019%) compared with the reference, 100%, and FP treatments, respectively, compared with the pre-harvest period, after controlling for temporal changes in the references and adjusting for pre-harvest differences among the treatment sites.

However, the 95% confidence intervals for these comparisons included 1 (i.e., 0%), indicating uncertainty in the direction of this effect.

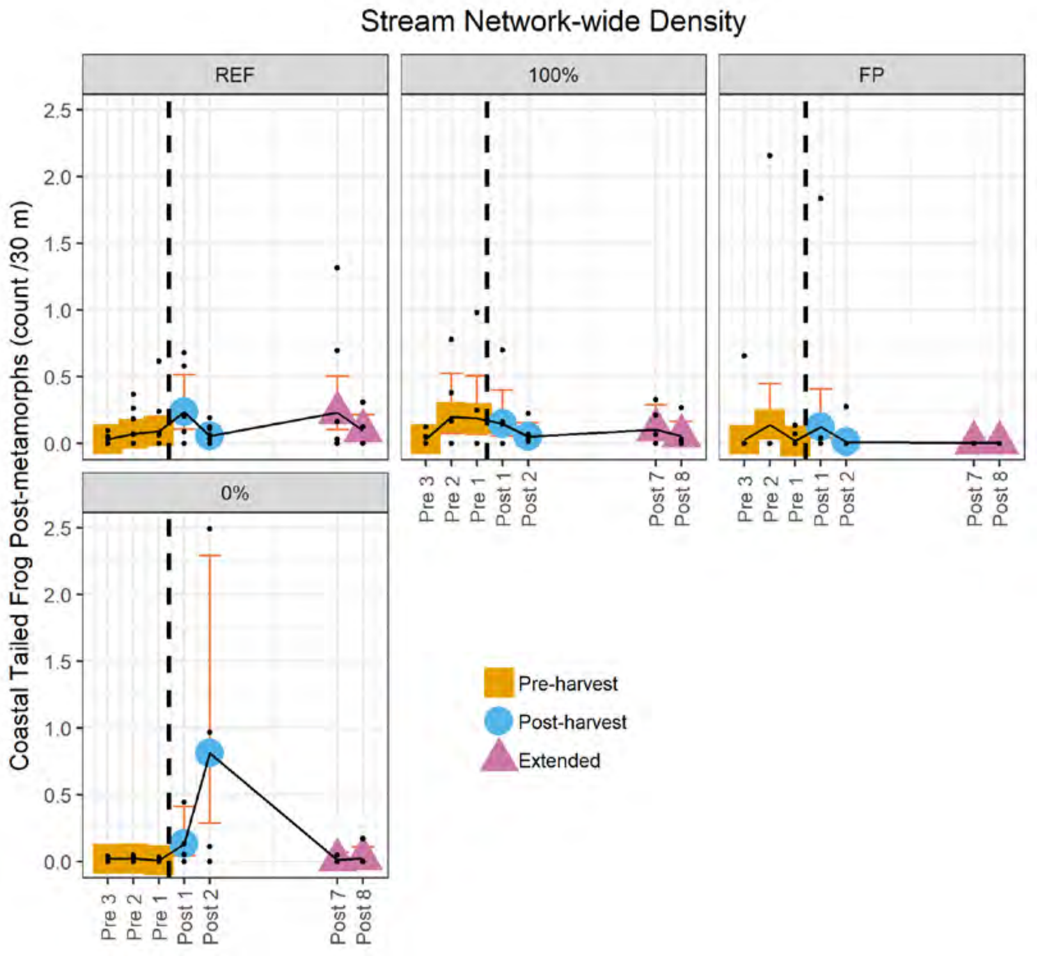

Figure 9-7. Mean stream network-wide post-metamorphic Coastal Tailed Frog density (animals/30 m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 7 & Post 8). Vertical colored lines around treatment means show approximate 95% credible intervals. Vertical dashed lines show the timing of harvest implementation at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 9-8. The within-treatment estimate of the proportional change and 95% credible intervals (CI) for mean stream network-wide post-metamorphic Coastal Tailed Frog density (animals/30 m) between the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8.

Treatment	Estimate (CI)	
	Post 1 & 2	**Post 7 & 8**
REF (n = 6)	1.94 (1.46, 2.56)	2.46 (1.86, 3.27)
100% (n = 4)	0.84 (0.55, 1.22)	0.73 (0.47, 1.11)
FP (n = 3)	0.99 (0.40, 2.22)	0.09 (0.02, 0.33)
0% (n = 4)	20.54 (9.85, 47.60)	0.99 (0.29, 3.26)

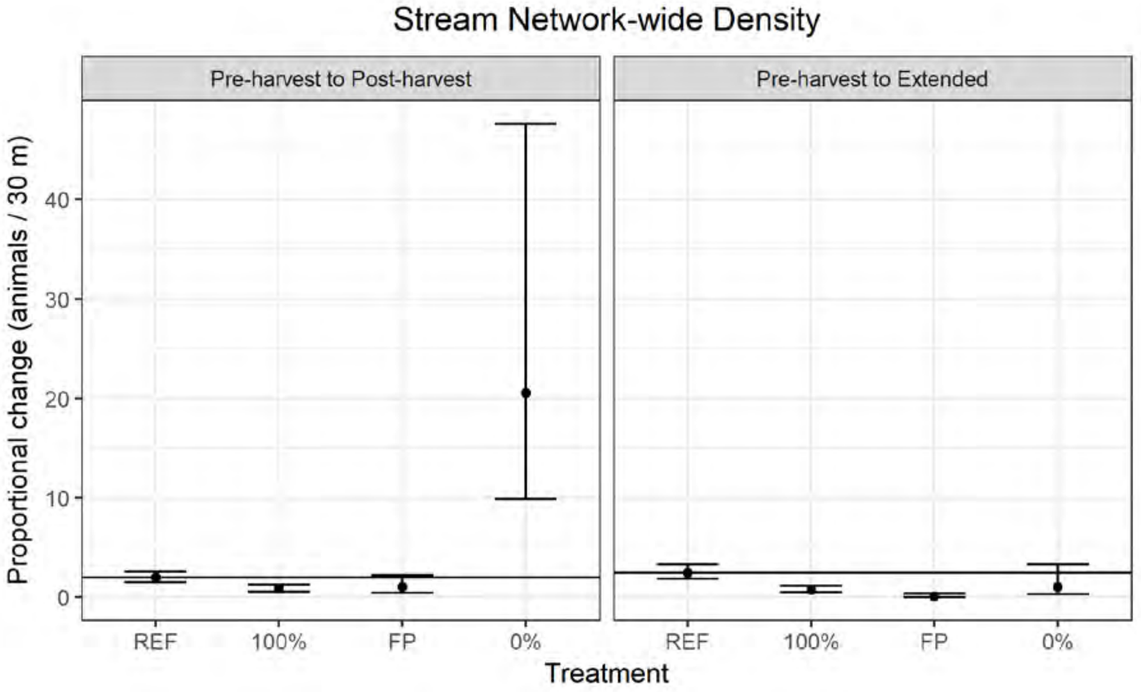

Figure 9-8. The within-treatment estimate of the proportional change and 95% credible intervals for mean stream network-wide post-metamorphic Coastal Tailed Frog density (animals/30 m) between the pre-harvest and post-harvest (Post 1 & Post 2), and extended (Post 7 & Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 9-9. The between-treatment comparison of the proportional change and 95% credible intervals (CI) of the estimates for mean stream network-wide post-metamorphic Coastal Tailed Frog density (animals/30 m) between the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8. Contrasts with credible intervals that do not overlap one 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**
100% vs. REF	**0.43 (0.27, 0.69)**	**0.29 (0.18, 0.48)**
FP vs. REF	0.51 (0.21, 1.17)	**0.03 (0.01, 0.14)**
0% vs. REF	**10.61 (4.81, 25.48)**	0.40 (0.12, 1.38)
0% vs. FP	**20.72 (6.82, 71.39)**	**11.51 (1.85, 80.87)**
0% vs. 100%	**24.58 (10.54, 60.93)**	1.36 (0.38, 4.81)
FP vs. 100%	1.19 (0.46, 2.92)	**0.12 (0.02, 0.49)**

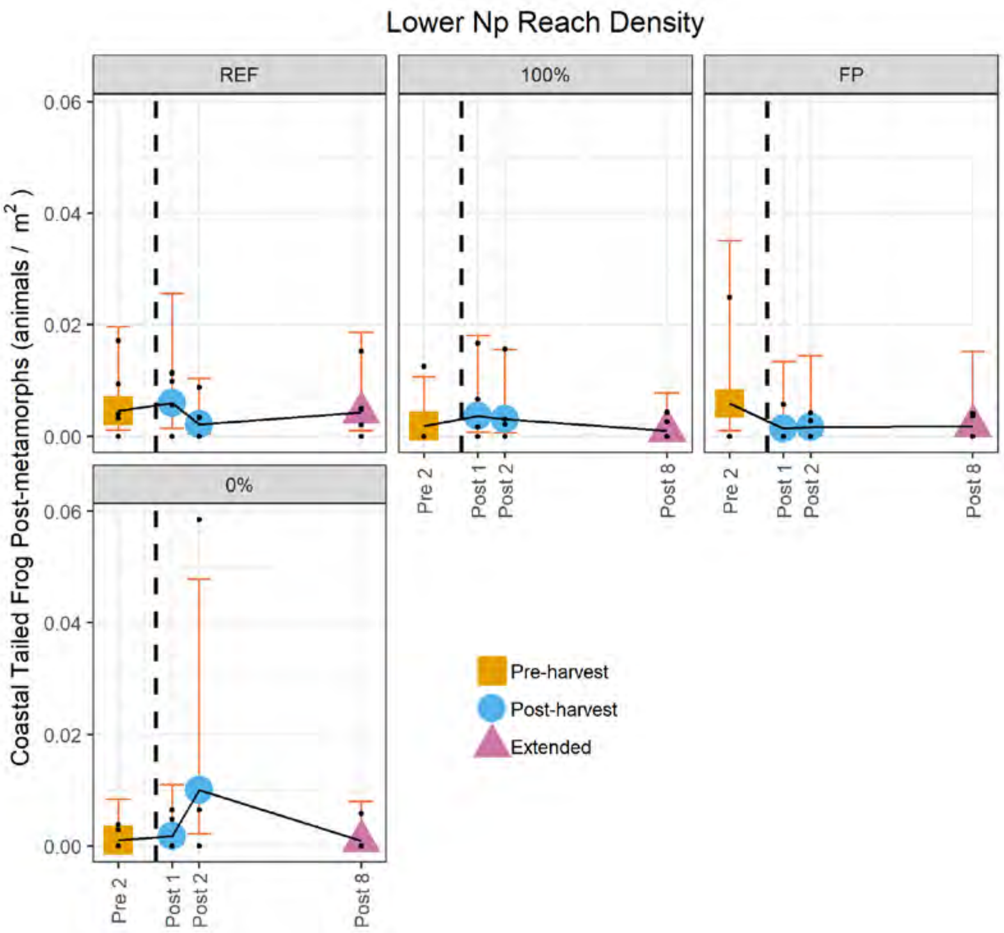

Figure 9-9. Mean lower Np reach post-metamorphic Coastal Tailed Frog density (animals/m^2) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 9-10. The within-treatment estimate of the proportional change and 95% confidence intervals (CI) for mean lower Np reach post-metamorphic Coastal Tailed Frog density (animals/m^2) between the pre-harvest period and Post 1 & Post 2, and Post 8.

Treatment	Estimate (CI)	
	Post 1 & 2	**Post 8**
REF (n = 6)	0.78 (0.29, 2.07)	0.93 (0.30, 2.83)
100% (n = 4)	1.82 (0.47, 7.08)	0.55 (0.07, 4.12)
FP (n = 3)	0.25 (0.06, 1.10)	0.31 (0.05, 1.78)
0% (n = 4)	4.05 (0.68, 23.95)	0.87 (0.08, 9.66)

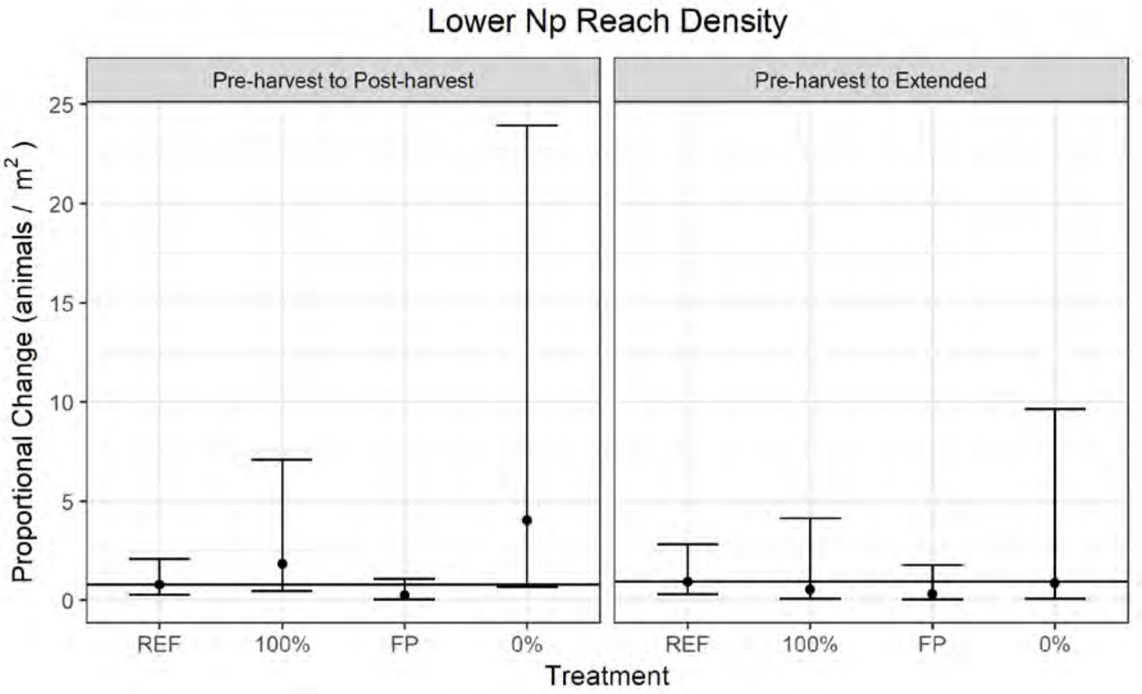

Figure 9-10. The within-treatment estimate of the proportional change and 95% confidence intervals for mean lower Np reach post-metamorphic Coastal Tailed Frog density (animals/m^2) between the pre-harvest and post-harvest periods (Post 1 & Post 2), and extended period (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 9-11. The between-treatment comparison of the proportional change and 95% confidence intervals (CI) of the estimates for mean lower Np reach post-metamorphic Coastal Tailed Frog density (animals/m^2) between the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8. 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 8
100% vs. REF	2.34 (0.44, 12.43)	0.59 (0.06, 5.93)
FP vs. REF	0.32 (0.06, 1.89)	0.33 (0.04, 2.65)
0% vs. REF	5.19 (0.68, 39.43)	0.94 (0.07, 13.34)
0% vs. FP	16.05 (1.60, 161.19)	2.83 (0.14, 55.35)
0% vs. 100%	2.22 (0.24, 20.76)	1.60 (0.07, 36.76)
FP vs. 100%	0.14 (0.02, 1.02)	0.56 (0.04, 8.14)

9-4.2.3. Torrent Salamanders

9-4.2.3.a. Stream network-wide

Mean annual stream network-wide torrent salamander densities adjusted for detection ranged from 0.0 to 48.5, 0.0 to 98.3, and 0.0 to 110.4 animals/30 m, respectively, in the pre-harvest, Post 1 and Post 2, and Post 7 and Post 8 (**Figure 9-11**). We found evidence that treatments differed in the magnitude of change over time (**Table 9-11**; **Figure 9-10**; **Table 9-13**). In Post 1 and Post 2, we estimated the between-treatment comparison for the 0% treatment to be a 2.98 (95% credible interval: 1.18, 7.51), or in other words a +198% (95% credible interval: +18%, +651%) change in mean density, compared with the pre-harvest period, after controlling for temporal changes in the references. We also estimated a +165% (+20%, +487%) and +266% (+55%, +765%) change in the 0% treatment compared to the 100% and FP treatments, respectively, after adjusting for pre-harvest differences among the treatment sites. In Post 7 and Post 8, we estimated a -64% (-86%, -10%) change in mean density in the FP treatment, compared with the pre-harvest period. We observed a +126% (+28%, +299%) within-treatment change in the reference in this same period (**Table 9-12**). We also estimated a -70% (-87%, -33%) change in the FP treatment compared to the 100% treatment.

9-4.2.3.b. Lower Np reach

Mean annual torrent salamander site densities in the lower Np reach ranged from 0.00 to 0.29, 0.00 to 0.37, and 0.00 to 0.30 animals per m^2, respectively, in the pre-harvest period, Post 1 and Post 2, and Post 8 (**Figure 9-13**). We did not find clear evidence that torrent salamander density varied among treatments (P = 0.61; **Table 9-14**; **Figure 9-14**; **Table 9-15**).

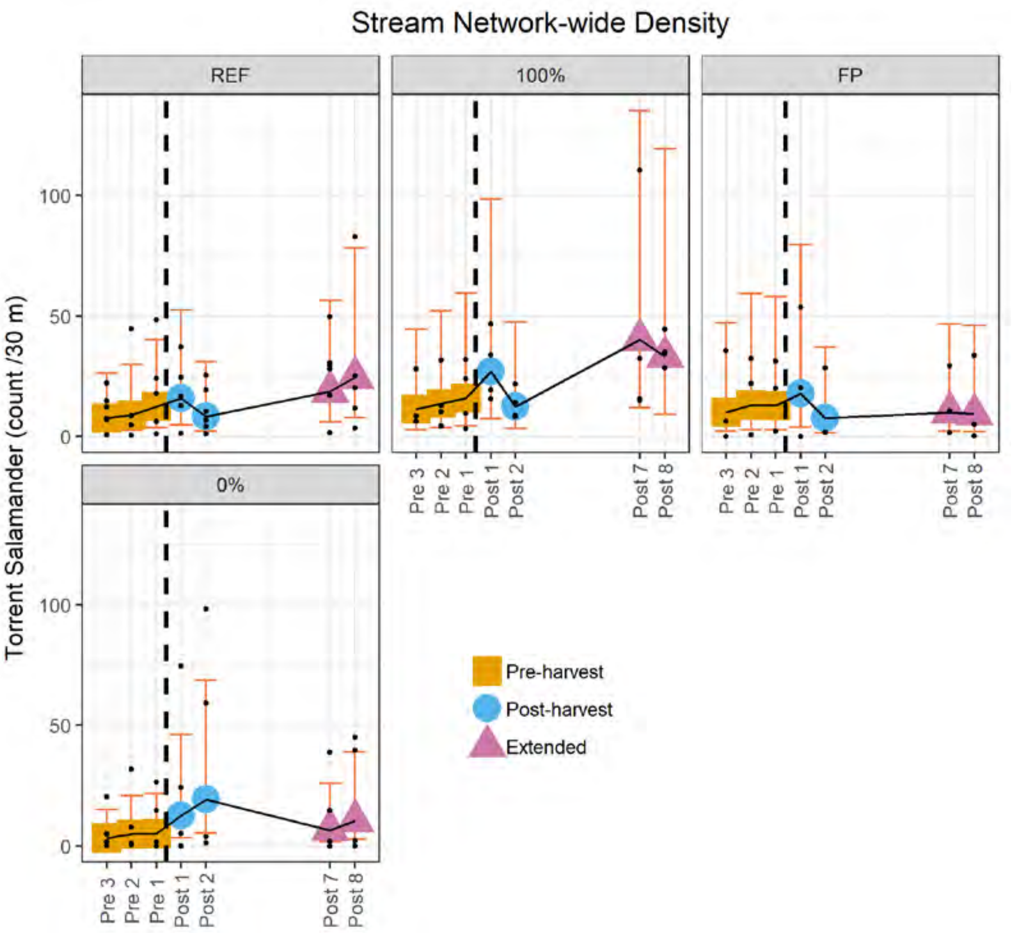

Figure 9-11. Mean stream network-wide torrent salamander density (animals/30 m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 7 & Post 8). Vertical colored lines around treatment means show approximate 95% credible intervals. Vertical dashed lines show the timing of harvest implementation at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 9-12. The within-treatment estimate of the proportional change and 95% credible intervals (CI) for mean stream network-wide torrent salamander density (animals/30 m) between the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8.

Treatment	Estimate (CI)	
	Post 1 & 2	Post 7 & 8
REF (n = 6)	1.20 (0.56, 2.60)	2.26 (1.28, 3.99)
100% (n = 4)	1.35 (0.71, 2.58)	2.70 (1.52, 4.79)
FP (n = 3)	0.98 (0.47, 2.06)	0.81 (0.37, 1.78)
0% (n = 4)	3.59 (1.92, 6.69)	1.90 (0.95, 3.80)

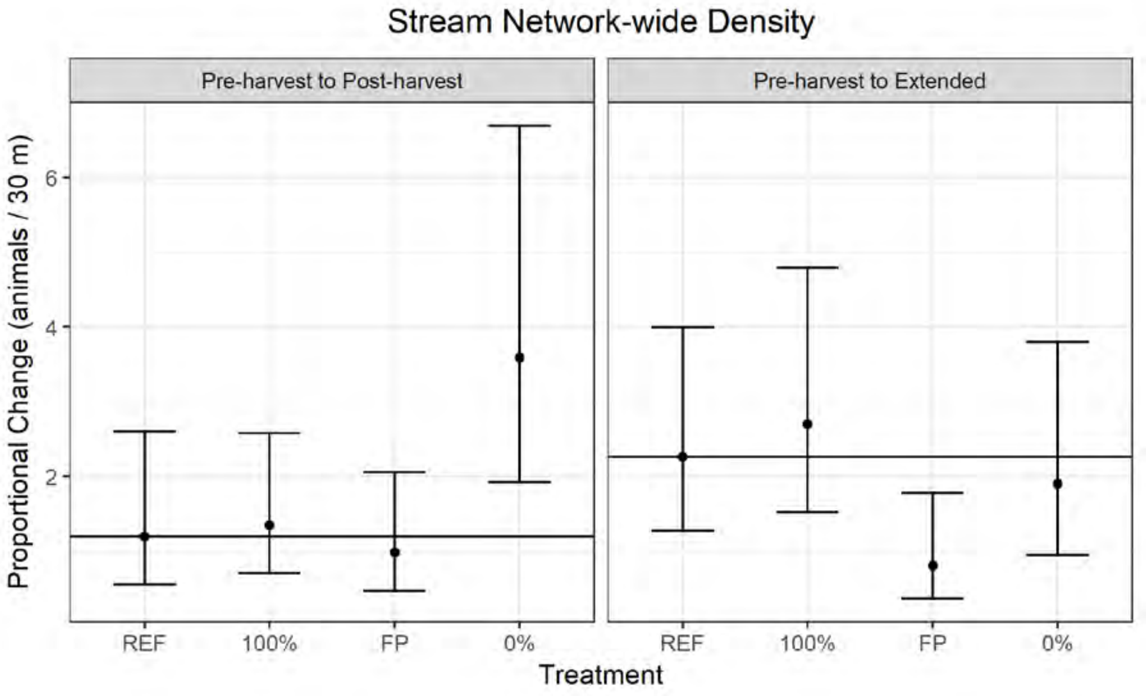

Figure 9-12. The within-treatment estimate of the proportional change and 95% credible intervals for mean stream network-wide torrent salamander density (animals/30 m) between the pre-harvest and post-harvest periods (Post 1 & Post 2), and extended (Post 7 & Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 9-13. The between-treatment comparison of the proportional change and 95% credible intervals (CI) of the estimates for mean stream network-wide torrent salamander density (animals/30 m). Contrasts with credible intervals that do not overlap one 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**
100% vs. REF	1.12 (0.44, 2.88)	1.20 (0.59, 2.43)
FP vs. REF	0.81 (0.28, 2.33)	**0.36 (0.14, 0.90)**
0% vs. REF	**2.98 (1.18, 7.51)**	0.84 (0.37, 1.92)
0% vs. FP	**3.66 (1.55, 8.65)**	2.35 (0.91, 6.02)
0% vs. 100%	**2.65 (1.20, 5.87)**	0.71 (0.32, 1.56)
FP vs. 100%	0.73 (0.31, 1.67)	**0.30 (0.13, 0.67)**

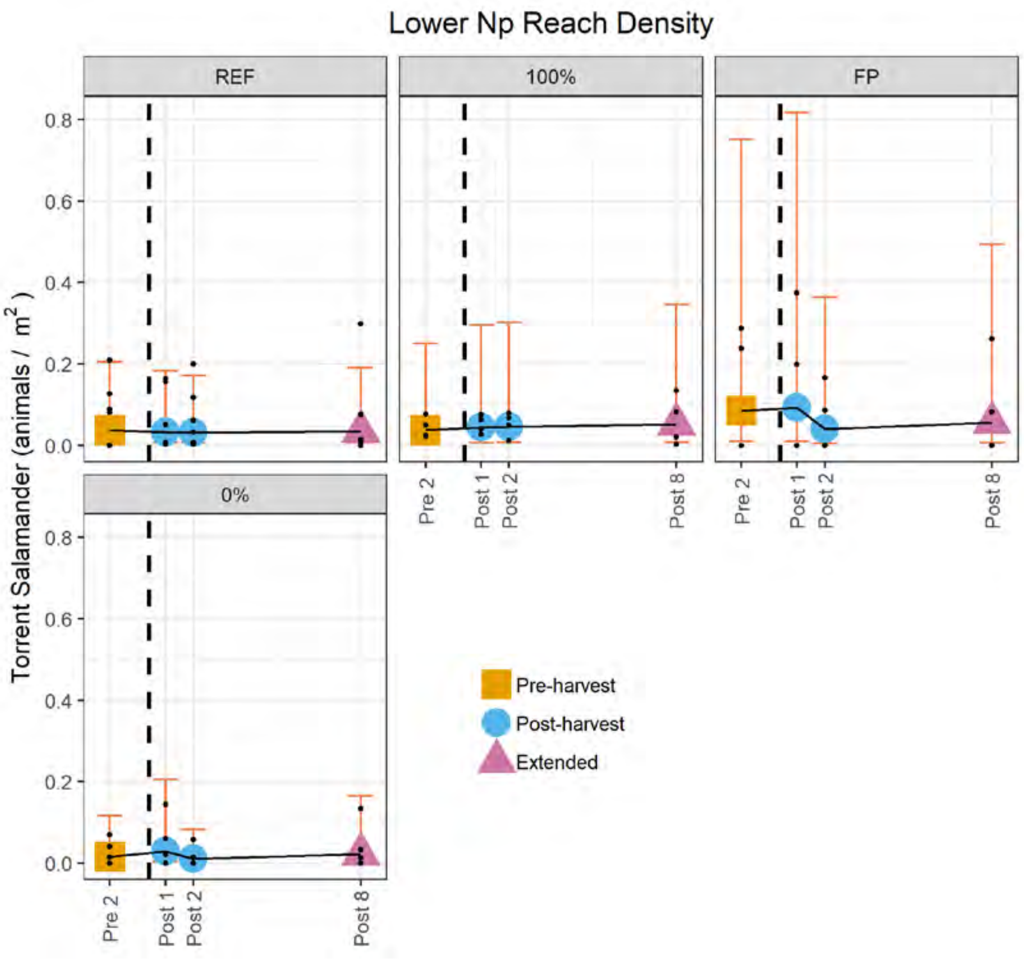

Figure 9-13. Mean lower Np reach torrent salamander density (animals/m^2) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 9-14. The within-treatment estimate of the proportional change and 95% confidence intervals (CI) for mean lower Np reach torrent salamander density (animals/m^2) between the pre-harvest period and Post 1 & Post 2, and the pre-harvest period and Post 8.

Treatment	**Estimate (CI)**	
	Post 1 & 2	**Post 8**
REF (n = 6)	0.91 (0.63, 1.33)	0.81 (0.51, 1.30)
100% (n = 4)	1.21 (0.64, 2.28)	1.30 (0.64, 2.63)
FP (n = 3)	0.70 (0.47, 1.03)	0.51 (0.31, 0.84)
0% (n = 4)	0.72 (0.35, 1.49)	0.77 (0.34, 1.76)

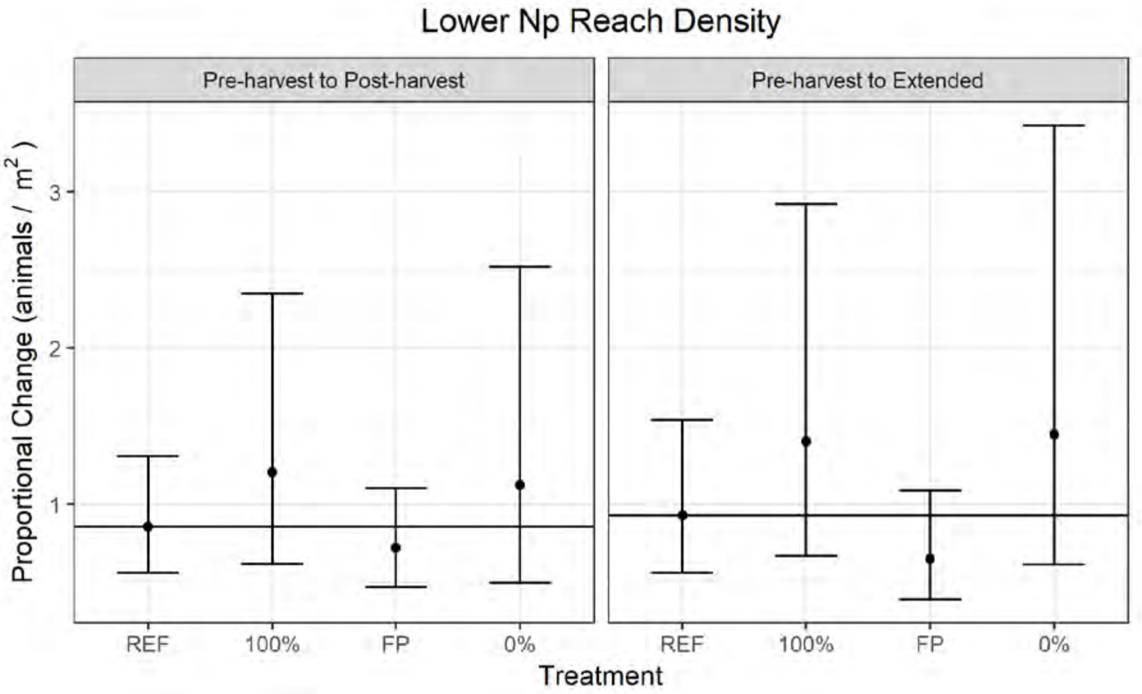

Figure 9-14. The within-treatment estimate of the proportional change and 95% confidence intervals for mean lower Np reach torrent salamander density (animals/m^2) between the pre-harvest and post-harvest periods (Post 1 & Post 2), and extended periods (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 9-15. The between-treatment comparison of the proportional change and 95% confidence intervals (CI) of the estimates for mean lower Np reach torrent salamander density (animals/m^2). 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
100% vs. REF	1.33 (0.64, 2.77)	1.59 (0.68, 3.72)
FP vs. REF	0.76 (0.45, 1.31)	0.62 (0.31, 1.24)
0% vs. REF	0.79 (0.35, 1.79)	0.95 (0.37, 2.45)
0% vs. FP	1.03 (0.45, 2.35)	1.52 (0.58, 3.99)
0% vs. 100%	0.59 (0.23, 1.56)	0.59 (0.20, 1.76)
FP vs. 100%	0.58 (0.27, 1.21)	0.39 (0.16, 0.93)

9-4.2.4. Giant Salamanders

9-4.2.4.a. Stream network-wide

Mean annual stream network-wide giant salamander densities adjusted for detection ranged from 0.3 to 38.5, 0.2 to 22.1, and 1.5 to 59.0 animals/30 m, respectively, in the pre-harvest period, Post 1 and Post 2, and Post 7 and Post 8 (**Figure 9-15**). We found evidence that treatments differed in the magnitude of change over time (**Table 9-16**; **Figure 9-16**; **Table 9-17**). In Post 1 and Post 2, we estimated the between-treatment comparison for the FP treatment to be 0.36 (95% credible interval: 0.14, 0.90), or in other words a -64% (95% credible interval: -86%, -10%) change compared with the pre-harvest period, after controlling for temporal changes in the references. We also estimated a +298% (+65%, +861%) change in the 0% compared to the FP treatment, after adjusting for pre-harvest differences among the treatment sites, however, large uncertainty existed in the estimate. In Post 7 and Post 8, we estimated a -53% (-79%, +6%) change in the FP treatment, compared with the pre-harvest period and relative to the reference; however, the credible interval for this comparison included 1 (i.e., 0%), indicating uncertainty in the direction of these changes. We observed a +198% (+52%, +484%) within-treatment change in the reference in this same period (**Table 9-16**).

9-4.2.4.b. Lower Np reach

Mean annual giant salamander site densities in the lower Np reach ranged from <0.01 to 0.20, 0.00 to 0.19, and <0.01 to 0.12 animals per m^2, respectively, in the pre-harvest period, Post 1 and Post 2, and Post 7 and Post 8 (**Figure 9-17**). We did not find clear evidence that giant salamander density varied among treatments (P = 0.37; **Table 9-18**; **Figure 9-16**; **Table 9-17**).

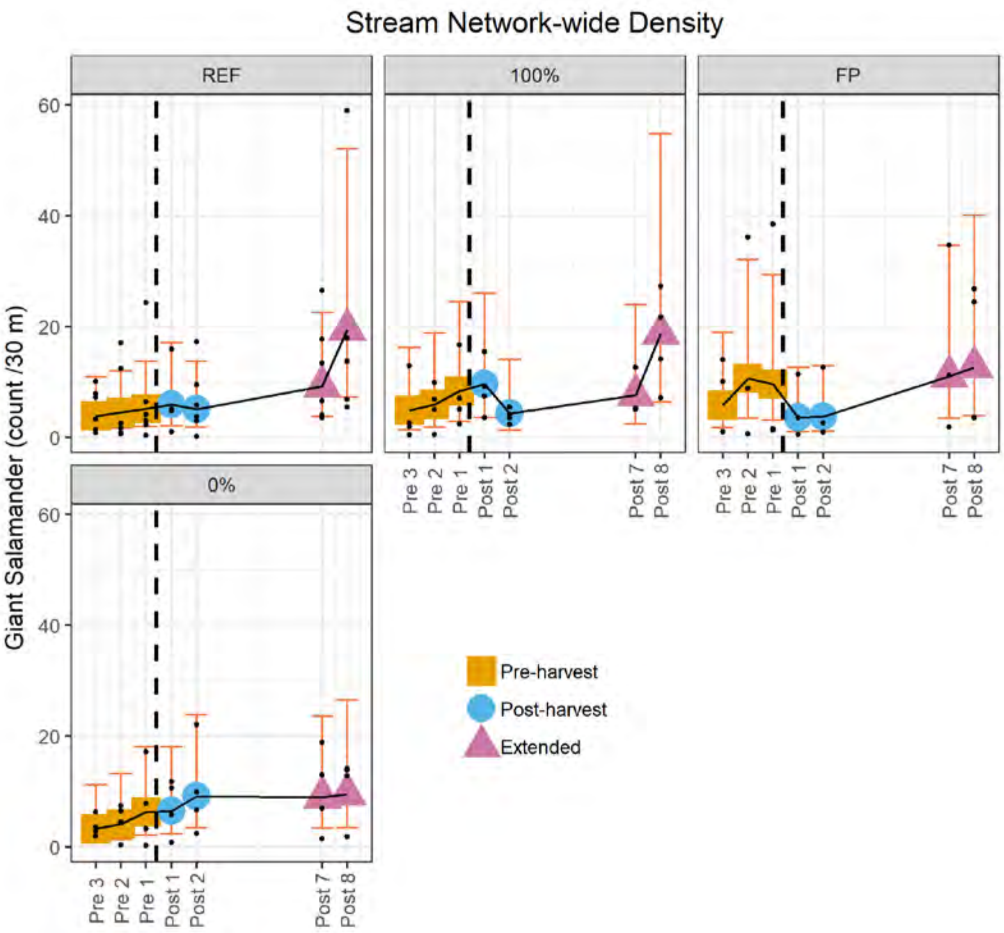

Figure 9-15. Mean stream network-wide giant salamander density (animals/30 m) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 7 & Post 8). Vertical colored lines around treatment means show approximate 95% credible intervals. Vertical dashed lines show the timing of harvest implementation at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 9-16. The within-treatment estimate of the proportional change and 95% credible intervals (CI) for mean stream network-wide giant salamander density (animals/30 m) between the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8.

Treatment	Estimate (CI)	
	Post 1 & 2	Post 7 & 8
REF (n = 6)	1.23 (0.59, 2.56)	2.98 (1.52, 5.84)
100% (n = 4)	1.03 (0.49, 2.16)	1.91 (0.90, 4.04)
FP (n = 3)	0.44 (0.21, 0.94)	1.40 (0.71, 2.78)
0% (n = 4)	1.75 (0.89, 3.46)	2.10 (1.05, 4.20)

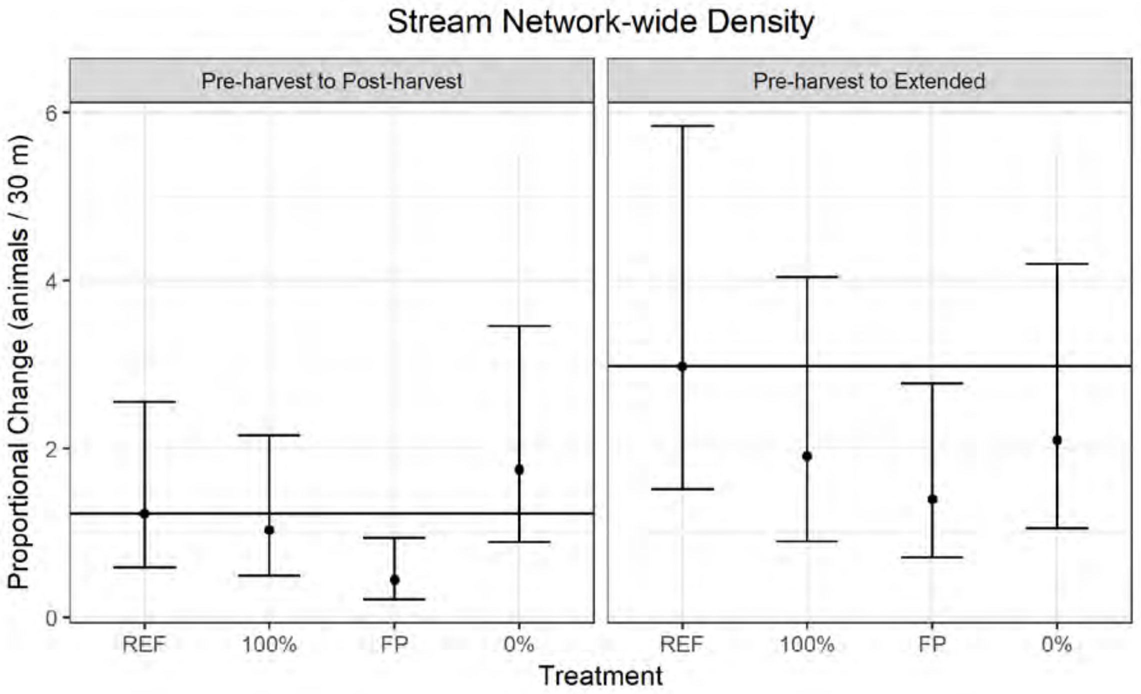

Figure 9-16. The within-treatment estimate of the proportional change and 95% credible intervals for mean stream network-wide giant salamander density (animals/30 m) between the pre-harvest and post-harvest (Post 1 & Post 2), and extended (Post 7 & Post 8) periods. A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 9-17. The between-treatment comparison of the proportional change and 95% credible intervals (CI) of the estimates for mean stream network-wide giant salamander density (animals/30 m) between the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8. Contrasts with credible intervals that do not overlap one 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**
100% vs. REF	0.84 (0.35, 2.00)	0.64 (0.28, 1.48)
FP vs. REF	**0.36 (0.14, 0.90)**	0.47 (0.21, 1.06)
0% vs. REF	1.42 (0.61, 3.34)	0.70 (0.32, 1.55)
0% vs. FP	**3.98 (1.65, 9.61)**	1.49 (0.64, 3.51)
0% vs. 100%	1.70 (0.71, 4.07)	1.10 (0.44, 2.72)
FP vs. 100%	0.43 (0.17, 1.07)	0.74 (0.36, 1.49)

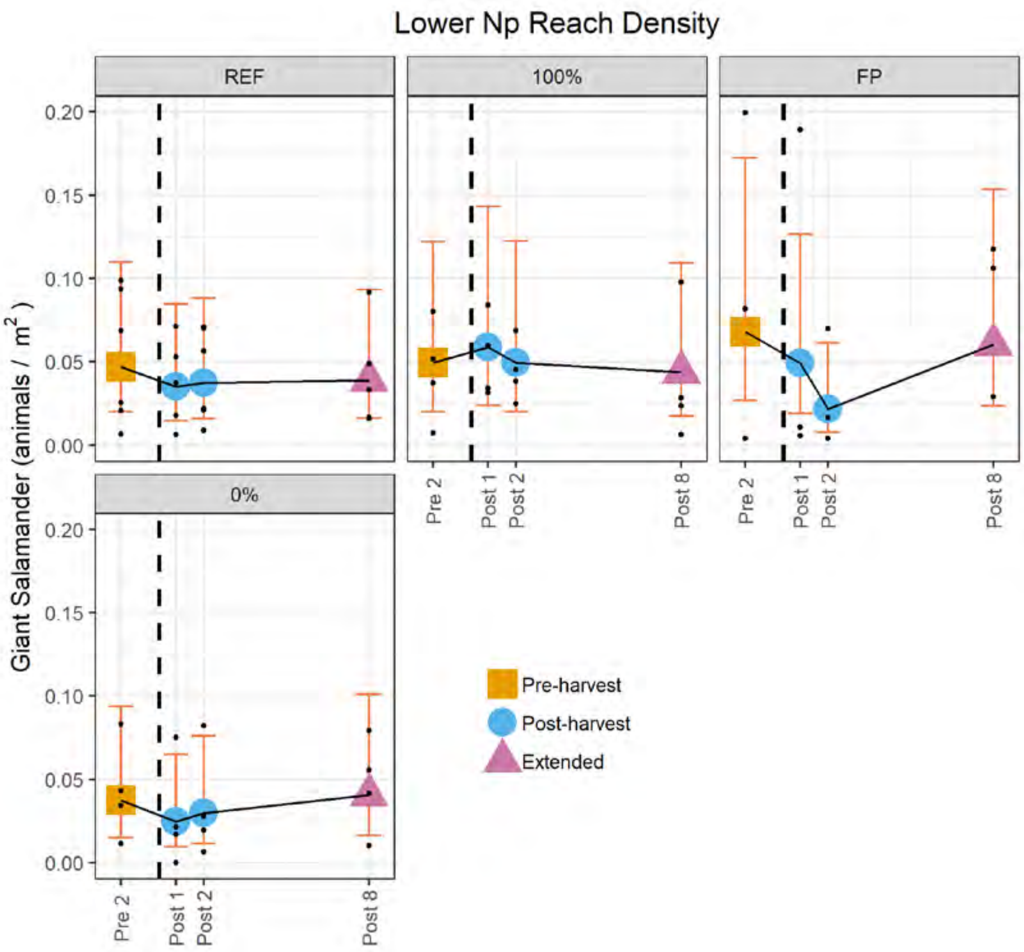

Figure 9-17. Mean lower Np reach giant salamander density (animals/m^2) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 9-18. The within-treatment estimate of the proportional change and 95% confidence intervals (CI) for mean lower Np reach giant salamander density (animals/m^2) between the pre-harvest period and Post 1 & Post 2, and Post 8.

Treatment	**Estimate (CI)**	
	Post 1 & 2	**Post 8**
REF (n = 6)	0.77 (0.46, 1.27)	0.86 (0.47, 1.59)
100% (n = 4)	1.10 (0.62, 1.98)	0.88 (0.43, 1.78)
FP (n = 3)	0.41 (0.23, 0.73)	0.70 (0.39, 1.27)
0% (n = 4)	0.67 (0.35, 1.29)	0.97 (0.49, 1.96)

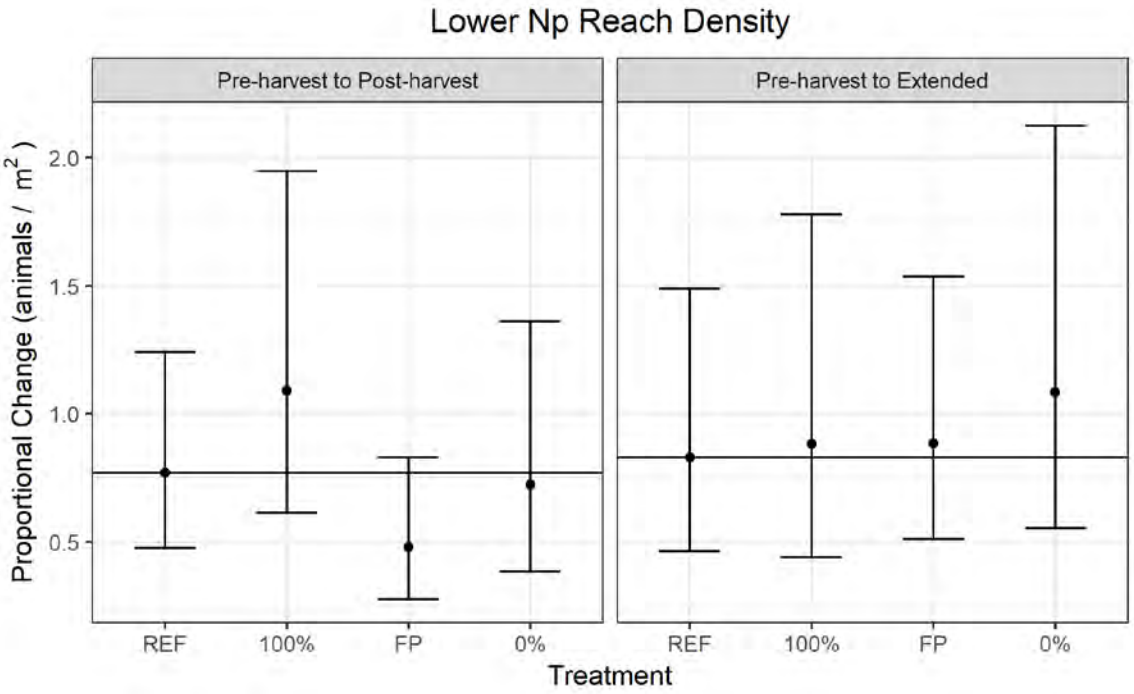

Figure 9-18. The within-treatment estimate of the proportional change and 95% confidence intervals for mean lower Np reach giant salamander density (animal/m^2) between the pre-harvest and post-harvest periods (Post 1 & Post 2), and extended periods (Post 8). A horizontal line placed at the reference treatment value indicates the estimated temporal change under reference conditions.

Table 9-19. The between-treatment comparison of the proportional change and 95% confidence intervals (CI) of the estimates for mean lower Np reach giant salamander density (animals/m^2) between the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8. 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 8**
100% vs. REF	1.44 (0.67, 3.12)	1.02 (0.40, 2.60)
FP vs. REF	0.53 (0.25, 1.15)	0.82 (0.35, 1.91)
0% vs. REF	0.88 (0.39, 2.00)	1.13 (0.45, 2.87)
0% vs. FP	1.65 (0.69, 3.95)	1.39 (0.56, 3.46)
0% vs. 100%	0.61 (0.26, 1.46)	1.11 (0.41, 3.00)
FP vs. 100%	0.37 (0.16, 0.84)	0.80 (0.32, 2.01)

9-4.3. OBSTRUCTED REACH DENSITY

In Post 1 and Post 2, and Post 7 and Post 8, we detected 847 torrent salamanders and 246 giant salamanders in obstructed plots. Torrent salamander density ranged from 0 to 20 animals/m in Post 1 and Post 2 and 0 to 37 animals/m in the Post 7 and Post 8. Giant salamander density ranged from 0 to 6 animals/m in Post 1 and Post 2 and 0 to 3 in Post 7 and Post 8. Results indicated lower giant salamander mean abundance in Post 7 and Post 8 compared with Post 1 and Post 2 for obstructed reaches. The estimated mean abundance for torrent salamanders was also lower in Post 7 and Post 8, but with greater uncertainty regarding the direction of the effect (**Table 9-20**).

In our comparison of mean densities between obstructed and unobstructed reaches in Post 7 and Post 8, torrent salamander density ranged from 0 to 32 animals/m^2 in obstructed plots and 0 to 37 animals/m^2 in unobstructed plots. Giant salamander density in Post 7 and Post 8 ranged from 0 to 7 animals/m^2 in obstructed plots and 0 to 4 animals/m^2 in unobstructed plots. Mean animal densities in obstructed and unobstructed reaches in Post 7 and Post 8 were similar, with no evidence of a difference in densities between reach types for either torrent or giant salamanders (**Table 9-21**).

Table 9-20. Comparison of mean density estimates (ratio of Post 7 & Post 8 to Post 1 & Post 2) and 95% confidence intervals (CI) for torrent and giant salamanders in obstructed reaches between the Post 1 & Post 2, and Post 7 & Post 8.

Taxa	Estimate (CI)
Torrent Salamanders	0.64 (0.36, 1.12)
Giant Salamanders	0.52 (0.28, 0.94)

Table 9-21. Mean estimates, estimated proportional contrasts, and 95% confidence intervals (CI) for torrent and giant salamander density (animals/m^2) in obstructed and unobstructed reaches in Post 7 & Post 8.

Taxa	Unobstructed (CI)	Obstructed (CI)	Proportional Contrast (CI) (obstructed/unobstructed)
Torrent Salamander	4.29 (2.72, 6.74)	4.13 (2.60, 6.55)	0.96 (0.63, 1.05)
Giant Salamander	0.55 (0.33, 0.92)	0.59 (0.35, 0.98)	1.07 (0.70, 1.60)

9-4.4. BODY CONDITION

Mean annual torrent salamander SMI by site ranged from 0.61 to 1.02, 0.25 to 1.07, and 0.56 to 0.69, respectively, in the pre-harvest period, Post 1 and Post 2, and Post 7 and Post 8 (**Figure 9-19**). We did not find clear evidence that torrent salamander SMI varied among treatments (P = 0.40; **Table 9-22**; **Figure 9-21**; **Table 9-23**).

Mean annual giant salamander SMI by site ranged from 1.61 to 2.72, 1.37 to 2.23, and 1.59 to 2.13, respectively, in the pre-harvest period, Post 1 and Post 2, and Post 7 and Post 8 (**Figure 9-20**). We estimated the change within treatments between the pre-harvest period and Post 1 and Post 2, and the pre-harvest period and Post 7 and Post 8. We did not find clear evidence that giant salamander SMI varied among treatments (P = 0.88; **Table 9-22**; **Figure 9-21**; **Table 9-23**).

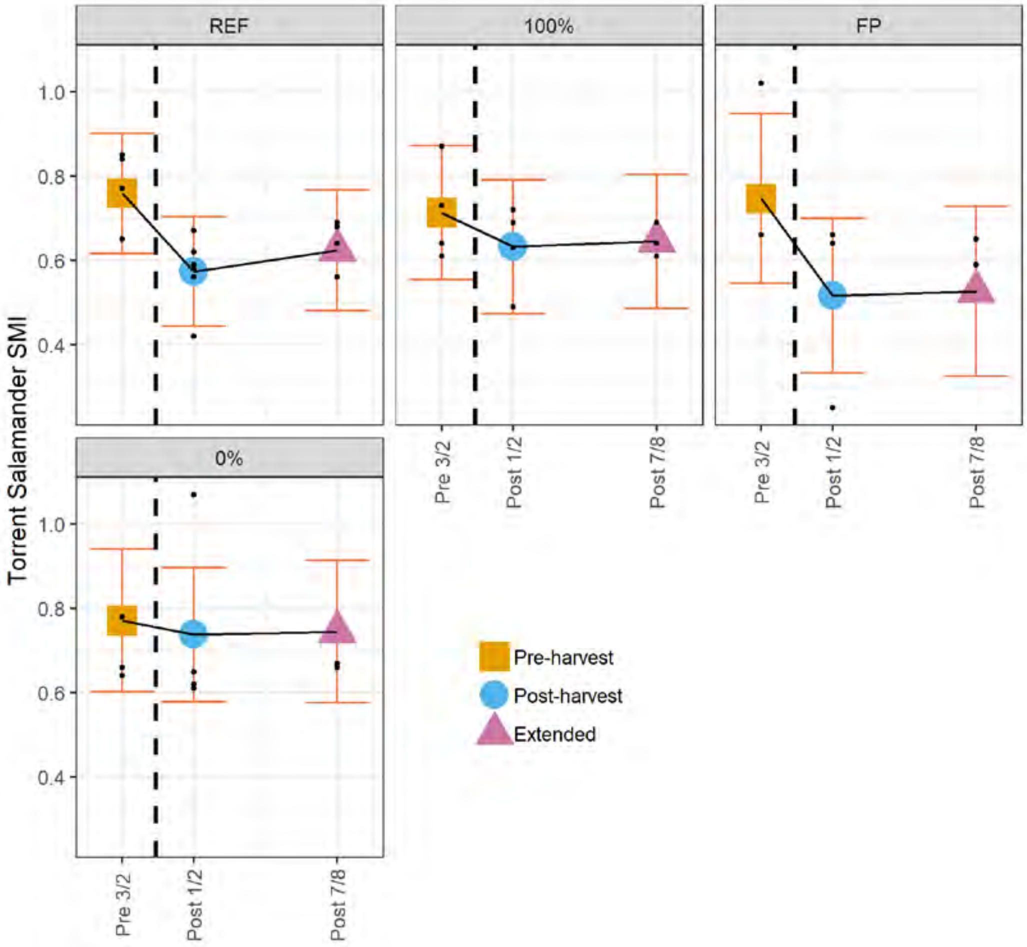

Figure 9-19. Means scaled mass index (SMI) for torrent salamanders by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 7 & Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at buffer treatment sites. Note that torrent salamander estimates were missing for some site/year combinations. Site means are dots; treatment means are colored symbols.

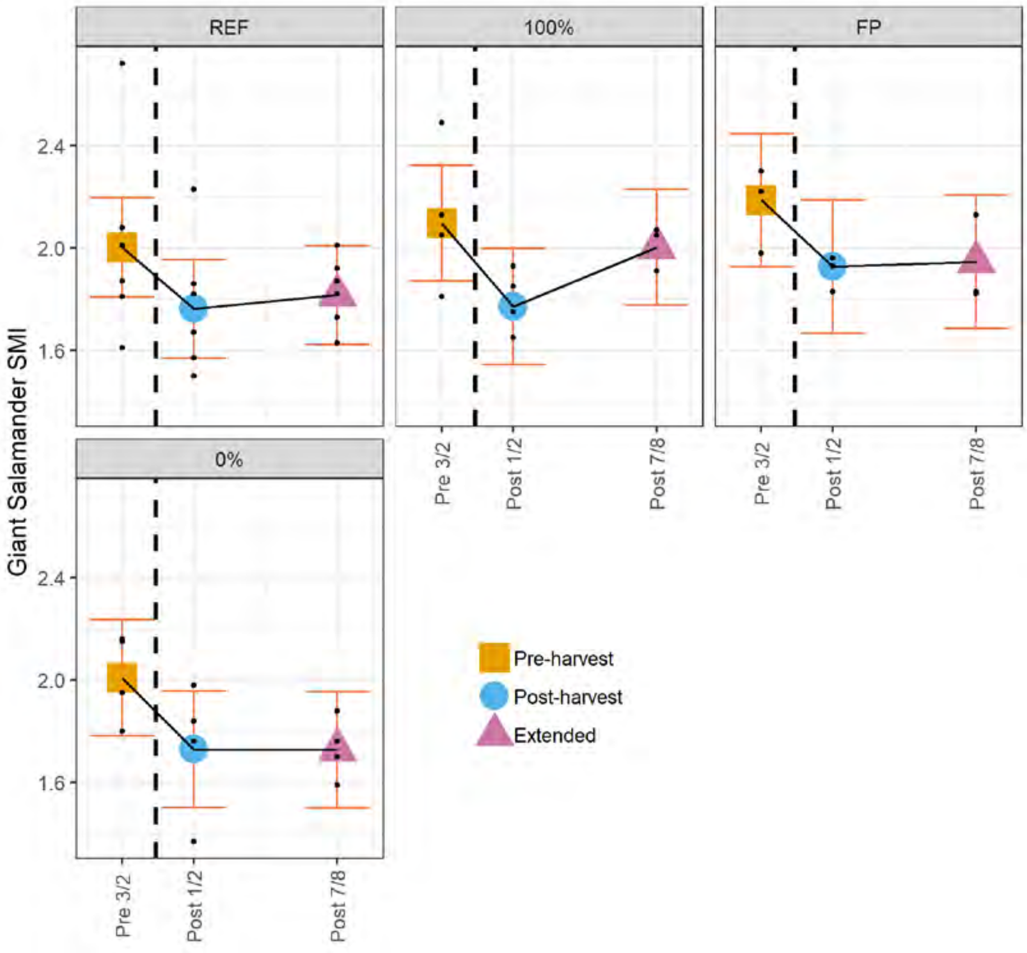

Figure 9-20. Means scaled mass index (SMI) for giant salamanders (bottom panel) by sample year (where pre-harvest includes Pre 3 and Pre 2, post-harvest includes Post 1 & Post 2, and extended includes Post 7 & Post 8). Vertical colored lines around treatment means show approximate 95% confidence intervals. Vertical dashed lines show the timing of harvest implementation at buffer treatment sites. Site means are dots; treatment means are colored symbols.

Table 9-22. The within-treatment estimate of the change and 95% confidence intervals (CI) for mean torrent and giant salamander scaled mass index (SMI) between the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8.

Treatment	Post 1 & 2	Post 7 & 8
	Estimate (CI)	Estimate (CI)
	Torrent Salamander	
REF (n = 6)	-0.19 (-0.31, -0.07)	-0.14 (-0.26, -0.01)
100% (n = 4)	-0.08 (-0.20, 0.04)	-0.07 (-0.19, 0.05)
FP (n = 3)	-0.23 (-0.40, -0.06)	-0.22 (-0.39, -0.05)
0% (n = 4)	-0.03 (-0.17, 0.10)	-0.03 (-0.17, 0.11)
	Giant Salamander	
REF (n = 6)	-0.24 (-0.45, -0.03)	-0.19 (-0.40, 0.03)
100% (n = 4)	-0.33 (-0.59, -0.06)	-0.09 (-0.36, 0.17)
FP (n = 3)	-0.26 (-0.56, 0.04)	-0.24 (-0.54, 0.06)
0% (n = 4)	-0.28 (-0.54, -0.02)	-0.28 (-0.54, -0.02)

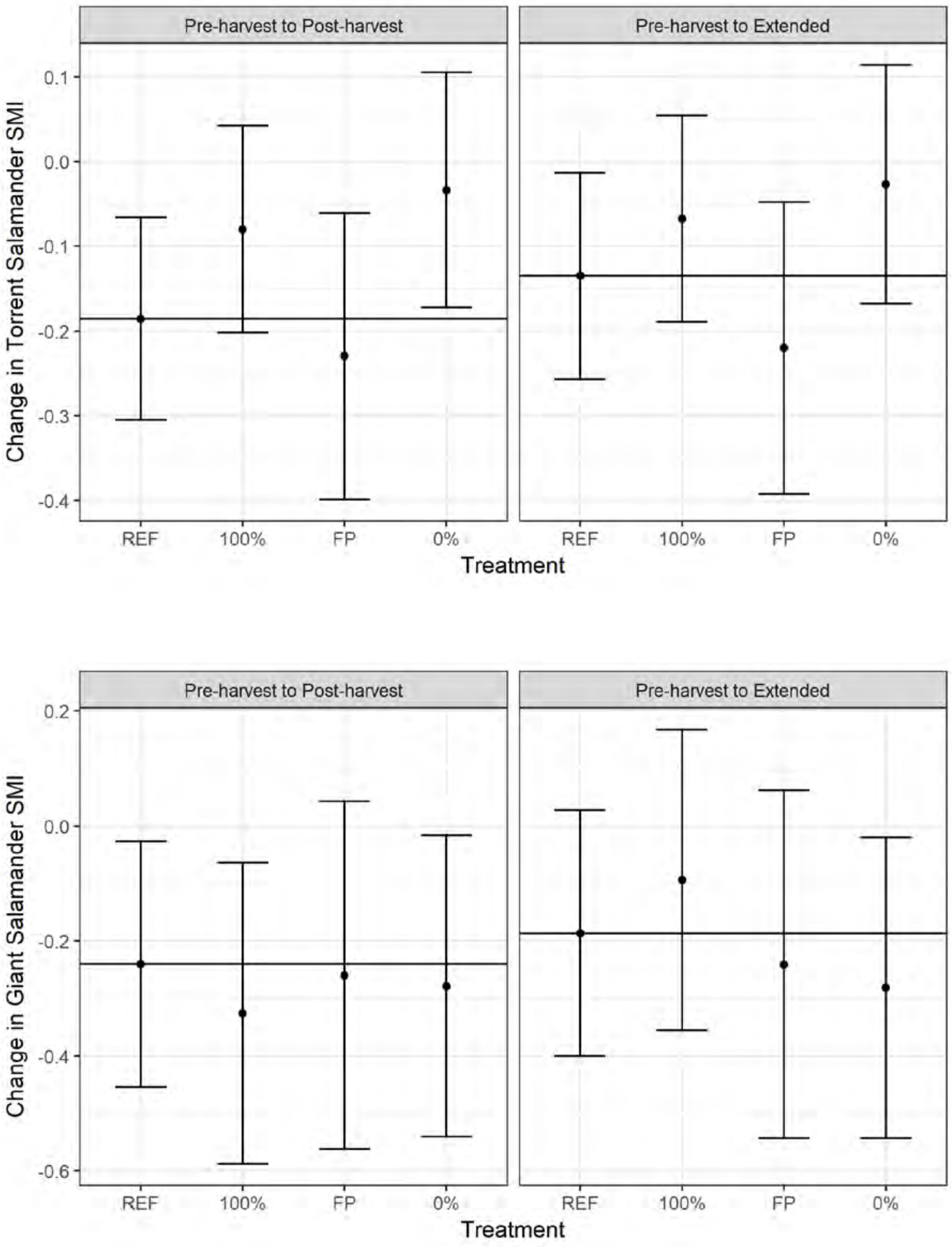

Figure 9-21. The within-treatment estimate in the change and 95% confidence intervals for mean torrent salamander (top panel) and giant salamander (bottom panel) scaled mass index (SMI) between the pre-harvest and post-harvest (Post 1 & Post 2), and extended (Post 7 & Post 8) periods. A horizontal line is placed at the reference treatment value, indicating the estimated temporal change under reference conditions.

Table 9-23. The between-treatment comparison of the change and 95% confidence intervals (CI) of the estimates for mean torrent and giant salamander scaled mass index (SMI) for the pre-harvest period and Post 1 & Post 2, and Post 7 & Post 8. The first treatment listed in each paired comparison is the treatment with fewer trees remaining in the RMZ buffer.

Contrast	Torrent Salamander		Giant Salamander	
	Post 1 & 2	Post 7 & 8	Post 1 & 2	Post 7 & 8
	Estimate (CI)	Estimate (CI)	Estimate (CI)	Estimate (CI)
100% vs. REF	0.11 (-0.07, 0.28)	0.07 (-0.10, 0.24)	-0.09 (-0.42, 0.25)	0.09 (-0.25, 0.43)
FP vs. REF	-0.04 (-0.25, 0.16)	-0.09 (-0.30, 0.13)	-0.02 (-0.39, 0.35)	-0.05 (-0.42, 0.32)
0% vs. REF	0.15 (-0.03, 0.33)	0.11 (-0.08, 0.29)	-0.04 (-0.38, 0.30)	-0.10 (-0.43, 0.24)
0% vs. FP	0.20 (-0.02, 0.41)	0.19 (-0.03, 0.42)	-0.02 (-0.42, 0.38)	-0.04 (-0.44, 0.36)
0% vs. 100%	0.05 (-0.14, 0.23)	0.04 (-0.15, 0.23)	0.05 (-0.32, 0.42)	-0.19 (-0.56, 0.18)
FP vs. 100%	-0.15 (-0.36, 0.06)	-0.15 (-0.36, 0.06)	0.07 (-0.33, 0.47)	-0.15 (-0.55, 0.25)

9-5. DISCUSSION

9-5.1. COASTAL TAILED FROG DENSITIES

We observed a consistent and marked decline in larval Coastal Tailed Frog densities in all three buffer treatments seven and eight years post-harvest that we did not observe in the two years post-harvest. Results for larvae were remarkably similar between the two spatial scales, i.e., stream network-wide and lower Np reach. Seven and eight years post-harvest, we observed a 65%, 93% and 84% decline in stream network-wide larval density in the 100%, FP and 0% treatments, respectively, compared with the reference. Similarly, lower Np reach larval density declined by 97%, 97% and 89% in the 100%, FP and 0% treatments, respectively. We also observed a delayed negative response for post-metamorphic tailed frog in all buffer treatments, however, this response was only evident in the stream network-wide analysis. Declines in post-metamorphic tailed frogs were especially evident in the 100% and FP treatments, where density declined by 71% and 97%, respectively.

The decline of post-metamorphic density seven and eight years post-harvest was the greatest in the FP treatment, a relationship we find difficult to explain considering the lack of treatment effect for this species in the 0% treatment. However, samples sizes were relatively small for post-metamorphic tailed frogs, leading to large confidence intervals, especially for the 0% treatment. Furthermore, unlike larvae, post-metamorphic tailed frogs are not restricted to the stream channel, so a decline in post-metamorphic individuals detected by our sampling does 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, that there was very 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 entirely possible that tailed frog post-metamorphs successfully 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 not persist if animals successfully immigrate back into study streams to breed when conditions become more favorable. Note that the period of our study represents a relatively short time for this species, i.e., life span for this species is estimated to be 15 to 20 years (Daugherty and Sheldon 1982). Our findings raise important considerations for a species such as Coastal Tailed Frog, for which larvae are restricted to the stream and post-metamorphs are highly mobile and can navigate overland.

We are not aware of other experimental buffer studies that have observed a delayed response of Coastal Tailed Frog to land management activities, though most have not monitored the post-treatment response for as long as eight years. In a similar experimental study, Jackson and colleagues (2007) concluded that clearcut timber harvest without riparian buffers had an immediate negative effect on Coastal Tailed Frog populations, however, the post-harvest response was measured for only three years following harvest, and study findings were based on limited observations. O'Connell and colleagues (2000) observed no difference in larval tailed frog densities among variable width buffers in a BACI-designed study in western Washington, however, this study only monitored amphibian densities for two years post-harvest.

We are aware of two experimental studies that monitored Coastal Tailed Frog response for more than the two or three years following harvest that had sufficient data from which to draw conclusions. Olson and colleagues (2014) concluded that timber management activities did not result in severe, persistent declines of Coastal Tailed Frog in the 10 years following harvest. However, this study included thinning in the uplands with a continuous riparian buffer that ranged up to 145 m (476 ft) wide, and a species-specific statistical analysis was not possible due to low and variable samples sizes. The second study (Hawkes and Gregory 2012) evaluated tailed frog post-metamorphs in riparian and upland areas in the 10 years following harvest, and found that relative abundance in the uplands was negatively affected by timber harvest, while conclusions for the riparian area were confounded by site-specific variability. Had we relied on our findings from two years post-harvest, we would have missed the decline of Coastal Tailed Frog in eight years post-harvest. Effects of silvicultural treatments on amphibians, particularly those with relatively long lifespans, may not be realized until many years after treatment (Hawkes and Gregory 2012).

Our findings are consistent with some retrospective studies that have concluded that tailed frog is less abundant in stands with a history of timber harvest (Welsh and Lind 2002; Stoddard and Hayes 2005; Ashton *et al.* 2006; Hawkes and Gregory 2012) and those 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 (Richardson and Neill 1998; Matsuda and Richardson 2005). However, Richardson and Neill (1998) evaluated occupancy rather than density, so declines in density would not have been noted. We cannot say with certainty why the findings from these latter studies differ from our own, however, one possible explanation is that they were conducted in sites located farther north, in British Columbia, Canada. It is possible that the response of the species to harvest varies with latitude, i.e., the species may respond differently depending on the location within its geographic range (Hayes and Quinn 2015), and associations with old-growth or late-seral stands may be strongest in the southern range of the distribution (Gilbert and Allwine 1991).

9-5.2. TORRENT SALAMANDER DENSITY

We had evidence of differences in the stream network-wide comparison of torrent salamander density. In the two years post-harvest, we noted a substantial 198% increase in stream network-wide torrent salamander density in the 0% treatment, but there was no evidence of a difference in the other buffer treatments. In contrast, in the seven and eight years post-harvest, we had evidence of a 64% decline in the FP treatment, and the increase we observed in the 0% treatment was no longer evident. We suspect that the increase in stream network-wide torrent salamander density in the 0% treatment one and two years post-harvest 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. In the two years post-harvest, we found relatively high densities of torrent salamanders in wood-obstructed reaches. However, the elevated density we observed for torrent salamanders in these reaches did not persist seven and eight years post-harvest. In fact, we had evidence of a 34% decline in torrent salamander density in wood-obstructed reaches between one and two, and seven and eight years post-harvest.

The increased torrent salamander density we observed in wood-obstructed reaches one and two years post-harvest was not expected. As such, we conducted a paired comparison between wood-obstructed reaches and stream reaches that were not obstructed with wood seven and eight years post-harvest, using the same rubble-rouse approach. At this time, we found no evidence of a difference in torrent salamander density between reach types. Furthermore, the proportion of the stream channel obstructed by wood in the 0% treatment declined between two and eight years post-harvest (though not significantly; see Chapter 3 – *Stand Structure, Tree Mortality, Wood Recruitment and Loading* in this report). Several potential explanations for the lack of a difference in torrent salamander abundance between reach types seven and eight years post-harvest include: (1) abundance did not differ between wood-obstructed and unobstructed reaches, even in the two years post-harvest, and what appeared to be a difference in densities between the reaches was an artifact of the different sampling methods used between reaches; or (2) in response to harvest immediately post-harvest, torrent salamanders congregated in wood-obstructed reaches, a change in habitat use that did not persist seven and eight years post-harvest. Regardless of the mechanism, we observed an increase in torrent salamander densities in wood-obstructed reaches initially, that we did not detect seven and eight years post-harvest.

We did not have evidence of a difference in torrent salamander density in the lower Np reach between treatments or periods. However, confidence intervals were large, especially for the FP treatment, for which we had only three treatment basins. Furthermore, we did not detect torrent salamanders in our evaluation of density in the lower Np reach in any year for one FP treatment site. These considerations likely impacted our power to detect a difference in response among treatments.

Conclusions regarding the impacts of forest management on torrent salamanders from previous studies have been inconsistent. Several retrospective studies have concluded that torrent salamanders occur in lower abundances in managed stands compared to old-growth stands (Corn and Bury 1989; Bury *et al.* 1991; Russell *et al.* 2005). However, other studies, including those with a BACI design, have not supported this claim, concluding that torrent salamanders were not greatly affected by timber harvest or upland forest thinning (Jackson *et al.* 2007; Olson *et al.* 2014). Russell and colleagues (2004) detected no relationship between torrent salamander occupancy or relative abundance and stand age. Still others have found that torrent salamander numbers and occupancy were greatest in mid-rotation stands (Steele *et al.* 2003; Kroll *et al.* 2008).

9-5.3. GIANT SALAMANDER DENSITY

We found evidence of a 64% decline in stream network-wide giant salamander density in the FP treatment in the two years post-harvest, compared with the reference. We estimated a similar 53% decline in the FP treatment in the seven and eight years post-harvest; however, the credible interval for this pairwise comparison included 1, indicating that the post-harvest difference may have been zero and suggesting the potential of some recovery between periods. We did not have evidence of a difference in giant salamander density in the lower Np reach for any treatment or period. Our stream network-wide results are somewhat consistent with the findings of Jackson and colleagues (2007), who found that giant salamanders were sensitive to the immediate impacts of harvest, but that the negative impacts were short-lived (e.g., three years or less), possibly due to recolonization from source populations in downstream fish-bearing reaches. Correlations of giant salamander density and occupancy with stand age or timber harvest history are variable. For example, some have concluded a lack of correlation (Bury *et al.* 1991; Leuthold *et al.* 2012), while others have concluded increased relative abundance in streams in late-seral forests (Ashton *et al.* 2006), or a positive association between occupancy and stand age (Kroll *et al.* 2008). Still others have concluded that the response of giant salamanders to timber harvest is site dependent, with populations in low gradient channels being more likely to respond negatively (Murphy and Hall 1981; Corn and Bury 1989). Olson and colleagues (2014) observed an increasing trend in the number of Coastal Giant Salamanders in their widest buffer treatment (~70–145 m two-sided buffer) in the 10 years following harvest, though upland harvest in this study was timber thinning rather than clearcut harvest.

9-5.4. COMPARISON OF STREAM NETWORK-WIDE AND LOWER NP REACH DENSITIES

Post-harvest results for stream network-wide and lower Np reach densities appear contradictory in the two years post-harvest, especially for Coastal Tailed Frog larvae. This apparent contradiction may be related to differences in patterns of amphibian distributions between lower and upper Np reaches between periods, or movements throughout the stream network that may have resulted from disturbances associated with timber harvest. Stream-associated amphibian distributions vary spatially and temporally throughout headwater reaches (Kelsey 1995; Hunter 1998; Stoddard and Hayes 2005; Hayes *et al.* 2006; Olson and Weaver 2007), which we found to be true in our own study sites (McIntyre *et al.* 2018, Chapter 15 – *Stream-associated Amphibians*). Consequently, our lower Np reach sampling design, which we applied to a standardized length located at the downstream end of the Type N reach, may not reflect stream network-wide population densities, especially if animals migrated throughout the stream network in response to buffer treatments. This issue may be particularly relevant in our FP treatment sites, where riparian buffers were focused on the lower Np reach and reaches farther upstream were inconsistently protected with riparian buffers.

Though results between the two sampling methods were at times inconsistent in the two years post-harvest, results through eight years post-harvest were more consistent. We observed a substantial decline in larval Coastal Tailed Frog density in all treatments at both spatial scales. We also observed a decline in post-metamorphic Coastal Tailed Frog density in the 100% and FP treatments, however, the 95% confidence intervals for the estimates in the lower Np reach included 1, indicating that the post-harvest difference may have been zero. We observed a decline in torrent salamander density in the FP treatment at both scales, however, the estimate for the lower Np reach again included 1. Finally, we did not have support for a difference in giant salamander density among treatments at either spatial scale. While the estimate for stream-network wide giant salamander density indicated a decline, there was again uncertainty in that result. Nonetheless, these consistencies increase our confidence in our longer-term results.

We used multiple survey methods to evaluate responses (i.e., block net in the lower Np reach, stream network-wide light-touch, and genetics). In our stream network-wide analysis, we adjusted our counts from light-touch sampling for the probability of detection, allowing 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 (MacKenzie and Kendall 2002; Mazerolle *et al.* 2007; McIntyre *et al.* 2012; Guillera-Arroita *et al.* 2014; Ficetola *et al.* 2018), and the statistical methods we used to adjust amphibian density have been validated in other amphibians studies (McKenny *et al.* 2006; Chelgren *et al.* 2011; 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, especially for Coastal Tailed Frogs in the 0% treatments seven and eight years post-harvest, led to wide confidence intervals and numerically unstable model fits. Despite that issue, the consistency of our results for Coastal Tailed Frog larvae between the two demographic sampling methods, and consistency of the decline in density among the buffer treatments, bolsters confidence in our result. Low counts seven and eight years post-harvest were almost certainly related to decreased densities at these sites. This conclusion was supported by the fact that additional intensive sampling efforts (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. Even though Coastal Tailed Frog and giant salamanders are highly aquatic, stream-associated species, terrestrial post-metamorphic tailed frog and giant salamander were not fully addressed in this study since our methodology focused on instream sampling. Ultimately, continued monitoring of amphibian densities across all study sites will be necessary to determine whether populations continue to decline, stabilize, or recover through time.

Important differences in amphibian densities between blocks should be considered when interpreting results. Torrent salamander densities tended to be higher in the Willapa Hills, moderate in the South Cascades, and lowest in the Olympics, possibly reflecting differences in the three species, which are distributed geographically. Giant salamander densities tended to be highest in the South Cascades, moderate in the Willapa Hills, and lowest in the Olympics, possibly reflecting differences between Cope’s and Coastal giant salamanders; only Cope’s giant salamander is found in the Olympics, whereas the two species co-occur throughout the Willapa Hills and South Cascades. Region specific trends were less straight forward for Coastal Tailed Frogs. This species was found in by far the lowest densities throughout all study sites and years and showed the greatest variability among regions and sites.

9-5.5. COMPARISON OF DEMOGRAPHIC AND GENETIC RESULTS

We estimated indices of genetic diversity for the three species included in our evaluation of genetic response and determined genetic clusters for each species. Amphibian genetic monitoring was conducted across all study sites concurrent with demographic sampling in post-harvest years seven and eight to determine if there was genetic evidence of declines in population sizes or non-random mating that may lead to future inbreeding depression (see Spear *et al.* 2019 for a complete discussion of genetic results related to this study). We identified genetic clusters across study sites to determine the role of gene flow in the observed genetic response. Overall, we found little evidence for a change in genetic diversity as a result of buffer treatments, with some exceptions.

Levels of genetic diversity were high in Coastal Tailed Frog, with numbers of alleles per locus averaging 13.6 (range 4.7 to 19.0). Genetic diversity as quantified by the number of alleles per locus was intermediate for Cope’s and Coastal Giant Salamanders, averaging 9.4 (range 4.0 to 15.0) and 5.6 (range 2.2 to 7.8), respectively. We did not detect a treatment effect on average number of alleles per locus for Coastal Tailed Frog ($P = 0.24$), or Cope’s ($P = 0.88$) or Coastal ($P = 0.17$) Giant Salamanders. Effective population sizes varied greatly among species, with the largest sizes for tailed frogs, intermediate for Cope’s Giant Salamander, and smallest for Coastal Giant Salamander. Consistent with their broad species range and ability to disperse terrestrially, we detected geographically large genetic clusters at a regional scale for both Coastal Tailed Frog and Coastal Giant Salamander. In contrast, Cope’s Giant Salamander had geographically restricted genetic clusters, whereby individual sites often were genetically distinct from nearby sites.

While we found no evidence of a change in genetic diversity for Coastal Tailed Frog, this may have been at least partially related to the relatively small samples sizes for some sites and years, including zeros. Although not significant for any variable, the genetic results for the 100% and FP treatments consistently showed similar trends: lower overall sample size, which contributed to a lower overall allelic diversity, lower observed heterozygosity, and a higher Wright’s inbreeding co-efficient (F_{IS}). However, allelic richness (i.e., accounting for sample size) was higher in these two treatments. It is not obvious why the 100% and FP treatment sites differed from the 0% treatment sites, however, these results are consistent with our findings for post-metamorphic tailed frog density, where we observed a significant decline in density in the 100% and FP treatments. Given the lack of significance, we interpret these results with caution, however, the similar pattern in both demographic and genetic results do suggest that future monitoring of tailed frogs in these sites would help to elucidate the response for this species.

Tailed frog density and the number of unique family groups declined in all treatments seven and eight years post-harvest, however, this decline mirrored almost exactly a decline in sample size. The average loss of several alleles in all buffer treatments also clearly reflected the reduced sample size. However, we cannot differentiate whether changes in measures of genetic diversity were a result of demographic differences, small sample sizes, or both. Additionally, a high level of heterozygosity is characteristic of tailed frog populations (Spear and Storfer 2008; Spear and Storfer 2010; Spear *et al.* 2012; Aguilar *et al.* 2013), so tailed frog allelic diversity and heterozygosity are unlikely to be highly sensitive to disturbance. Finally, the period of study represents a relatively short time for this species (Daugherty and Sheldon 1982), so a period sufficient to detect differences in genetic diversity may not have passed.

The only potential treatment effect we detected in Cope's Giant Salamander was a decrease in F_{IS} in the 100% treatment (0.06 decline relative to the reference). Decreases in F_{IS} typically indicates mating between more distantly related individuals, consistent with a shift in immigration/emigration dynamics. The P-value (0.105) was slightly above our alpha of 0.10 but was close enough that we felt it worthy of consideration, especially since non-random mating is a potential result of population disturbance. The decrease in the 100% treatment suggests that individuals were more likely to be mating with less related individuals. This outbreeding could be related to an increase in the probability of movement by individuals, which may be facilitated by the continuous riparian buffer. Further research is needed to determine if this hypothesis is likely.

Coastal Giant Salamander was detected in only 11 of 17 study sites, consistent with the fact that the species is not commonly found in the northern Willapa Hills and is absent from the Olympic Peninsula. This, in combination with the fact that they also had the fewest number of loci, resulted in a lower power to detect differences for this species. We did detect a significant decrease ($P = 0.05$) in F_{IS} in the 100% and 0% treatments. We estimated a 0.16 and 0.17 post-harvest decline in F_{IS} in the 100% ($P = 0.02$) and 0% ($P = 0.03$) treatments, relative to the reference. This response was similar to that observed for Cope's Giant Salamander in the 100% treatment, and likely represents migration in or out of the site. This result is not surprising given that we expected a continuous riparian buffer to facilitate movement, while a lack of a buffer was anticipated to impede movement. However, we hesitate to infer too much from our Coastal Giant Salamander results due to limited sample sizes. An increase in sampling intensity and increased number of loci would be warranted in future studies.

Demographic results for giant salamanders indicated an initial decline in density in the FP treatment in the two years post-harvest. However, seven and eight years post-harvest the change in giant salamander density no longer differed from the change in the reference. This is consistent with our genetic findings. In fact, genetic results for the FP treatment were among the most stable with respect to changes for both giant salamander species. We did detect evidence of a new population bottleneck at the WIL1-FP basin, but no evidence at the other two FP sites. The lower number of individuals in the FP treatment does not seem to have had an impact on genetic structure to date. However, results may be partially confounded by the fact that the demographic analysis was done for both giant salamander species combined versus the genetic analysis which differentiated a response between the two species.

Genetic structure for tailed frog and the two giant salamander species is likely influenced by surrounding basins in addition to site-level treatment effects. Although we do not see evidence of a change in genetic diversity due to clearcut timber harvest and alternative buffer treatments, we caution that increased sample size and additional sampling across future generations may be necessary to detect a trend. For example, simulations have demonstrated that changes in genetic structure are not likely to be detected until several generations post-impact (Hoban *et al.* 2013).

9-5.6. IMPLICATIONS OF FOREST MANAGEMENT ACTIVITIES

Several studies have revealed a positive relationship between stream-associated amphibian populations and stand age (Welsh and Lind 2002; Stoddard and Hayes 2005; Welsh *et al.* 2005; Ashton *et al.* 2006; Pollett *et al.* 2010). However, it is likely that forest age alone does not determine amphibian species' occupancy and abundance, but rather the microclimate and microhabitat conditions that tend to vary in relation to forest age (Welsh 1990; Diller and Wallace 1994). Amphibian abundance has been associated with 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 and Béraud 2014; Yeung *et al.* 2017), and the responses we observed largely met our expectations. Results for stream-associated amphibians, however, appear somewhat more complex. This is due in part to the fact that the amphibian species included in the study are long-lived. Immediate impacts to the species would be due to movement in or out of study sites. Longer-term impacts will reflect the additional effect of timber harvest on reproduction and continued survival. Our study was designed to evaluate treatment effects, not the mechanisms behind potential changes in amphibian abundance. However, because out study also evaluated changes in stream temperature, overstory canopy, primary productivity, wood loading, sediment retention, flow dynamics, stream and bank morphology, and nutrients, we are uniquely situated to consider the mechanisms behind the changes we observed in amphibian densities, and the differences observed among treatments.

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 could have beneficial consequences for stream-associated amphibians either directly (for grazing Coastal Tailed Frogs; Kiffney and 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 and Brodie 1972; Nussbaum *et al.* 1983), which could have negative consequences if food availability is limited. We found no changes in biofilm or periphyton in any buffer treatment in the post-harvest period (McIntyre *et al.* 2018; Chapter 13 – *Biofilm and Periphyton*). Consistent with these findings, our analysis of stable isotopes (see 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 channels where canopy modification increases trophic support from autotrophic sources (Kaylor and Warren 2017). Based on our lack of evidence of a change in instream primary production in the post-harvest period, and the delayed response we observed for Coastal Tailed Frog density in buffer treatments, we do not believe that the stream-associated amphibian response we observed was related to 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 (de Vlaming and Bury 1970; Karraker *et al.* 2006; Bury 2008; 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 (see Chapter 4 – *Stream Temperature and Cover* in this report). The critical aspect of stream temperature is whether the degree of temperature increase over pre-harvest conditions translates to a biologically risky condition. Currently, very limited critical thermal maximum or stress temperature information exists for stream-associated amphibians. Of the taxa included in our study, we do have some information for tailed frog. In a laboratory effort, Brown (1975) estimated the critical thermal maximum for Coastal Tailed Frogs to be approximately 18.5°C (65.3°F). In a summary of known oviposition sites, Karraker and colleagues (2006) found that the stream temperature rarely exceeded 14°C (57.2°F). In a laboratory trial of behavioral responses in thermal gradient chambers, de Vlaming and Bury (1970) found that first year Coastal Tailed Frog larvae congregated in water with temperatures below 10°C (50°F). In a limited field observational study conducted at a single study stream, de Vlaming and Bury (1970) noted that larvae avoided areas of the stream exposed to direct sunlight where temperatures varied between 15 and 20°C (59 and 68°F) on a clear and sunny summer day, but were found in nearby shaded areas that varied between 13 and 16°C (55 and 61°F). We did not observe an immediate negative effect of treatment on amphibians in the post-harvest period (except for giant salamanders in the FP treatment) when the post-harvest increase in stream temperatures was the greatest. However, it is possible that the increased temperatures we observed in all buffer treatment streams had negative longer-term consequences that were not immediately apparent, but which may have impacted movement or reproductive success over time, especially for Coastal Tailed Frogs, which had experienced the greatest declines across all buffer treatments seven and eight years post-harvest. Note that pre-harvest temperatures across all study streams averaged less relative to a random sample of western Washington streams on commercial forestland (Washington State Department of Ecology 2019), so we do not know whether the response of focal taxa would have differed under a scenario where the stream temperatures of our study sites more closely reflected those of other, slightly warmer, streams throughout western Washington.

Another immediately apparent effect of timber harvest in our buffer treatment sites was increased in-channel wood loading, primarily a result of logging slash in all buffer treatments and windthrow from the riparian buffer in the 100% and FP treatments. Wood loading has the potential to alter amphibian habitat availability and quality. Increased wood loading may have been responsible for declines in wetted and bankfull widths in the 0% treatment (see **Sections 7.4.1.** *Wetted Width* and **7.4.4.** *Bankfull Width*), which lasted through seven and eight years post-harvest, potentially altering instream amphibian habitat availability. Stream-associated amphibians, including Coastal Tailed Frog, Coastal Giant Salamander, and Columbia Torrent Salamander have been shown to be positively associated with stream width (Stoddard and Hayes 2005).

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 (Hawkins *et al.* 1983; Diller and Wallace 1996; Welsh and Lind 1996; Welsh and Ollivier 1998; Diller and Wallace 1999; Dupuis and Steventon 1999; Stoddard and Hayes 2005). We observed an increase in fine and sand substrates in all buffer treatments in the seven and eight years post-harvest, though the increase was not statistically significant in the 100% treatment (see **Section 7.4.5.** *Fines and Sand Substrates*). Fine sediment can modify grazing surfaces and availability of retreats for Coastal Tailed Frog larvae (Gomi *et al.* 2001; Jackson and Sturm 2002; Hassan *et al.* 2005; Maxa 2009), which are specialized periphyton grazers that preferentially select smooth, exposed rocks greater than 55 mm (2.2 in) in diameter for grazing and daytime retreats (Altig and Brodie 1972). The increase in fine sediments may help explain the lag in response we observed for Coastal Tailed Frog. We did have evidence of greater salamander densities in wood-obstructed reaches two years post-harvest. This observation seems in contradiction with the assumption that sediment negatively impacts amphibians, though it is possible that these wood deposits provided some benefits including shelter from reduced overhead canopy or predators, or greater diversity or abundance of food resources. However, a similar comparison was not possible for Coastal Tailed Frogs, mostly due to the limited observations we had for the species in wood-obstructed reaches. Furthermore, seven and eight years post-harvest, salamander densities in wood-obstructed reaches did not differ from those in reaches that were not obstructed. It is possible that, if the increased abundance in wood-obstructed reaches in the two years post-harvest was real, conditions that made these habitats attractive initially did not persist into the eight years post-harvest. Regardless of the mechanism for the increased salamander densities that we observed in these reaches initially, elevated densities were not maintained.

Where stream algal production is nutrient limited, increased nutrients can increase algal biomass, providing food resources for invertebrate and vertebrate grazers (including larval Coastal Tailed Frogs). Kiffney and Richardson (2001) saw increased periphyton ash-free dry mass and larval Coastal Tailed Frog growth rates in headwater streams supplemented with nutrients. However, in our study, the increased total-N and nitrate-N in the two years post-harvest did not translate to a corresponding increase in biofilm or periphyton (McIntyre *et al.* 2018, Chapter 13 – *Biofilm and Periphyton*), or macroinvertebrate numbers or biomass exported (McIntyre *et al.* 2018, Chapter 14 – *Macroinvertebrate Export*). Though it is possible that the increase we observed for some amphibian taxa in some treatments in the two years post-harvest may have been related to increases in nutrients (McIntyre *et al.* 2018, Chapter 9 – *Nutrient Export*), we have a difficult time mechanistically linking nutrients with the negative response we observed for some amphibians and life stages seven and eight years following harvest.

Amphibian stable isotope values were consistent with a diet comprised of periphyton for grazing Coastal Tailed Frog larvae, and aquatic predators and shredders and terrestrial spiders for giant salamanders, torrent salamanders, and post-metamorphic tailed frogs (see Chapter 8 – *Stable Isotopes* in this report). Stable isotope values also supported the possibility that some amphibians were preying on each other, a phenomenon that was observed for giant salamanders during this study. Though we did observe some post-harvest differences among treatments for ^{13}C for giant salamanders and ^{15}N for gatherer invertebrates, these changes did not support the hypothesis that canopy modification altered energy sources supporting food webs in our streams. In combination, results for periphyton and biofilm, nutrient export and stable isotopes led us to speculate that changes in primary productivity or nutrient availability were unlikely to have facilitated the negative response we saw for some amphibian taxa in some treatments. However, we did not investigate periphyton community composition, which may be more greatly impacted by the effects of harvest than simple measures of biofilm and periphyton production (Naymik *et al.* 2005). It is entirely possible that changes in community composition, that went undetected with our periphyton and biofilm sampling, impacted amphibian reproduction or fitness.

Some researchers have concluded that riparian buffers can ameliorate the impacts of timber harvest on stream-associated amphibians (Dupuis and Steventon 1999; Vesely and McComb 2002; Stoddard and Hayes 2005; Pollett *et al.* 2010). While the size and extent of the riparian buffers clearly matters, we found that substantial and consistent declines in density were evident for tailed frog larvae seven and eight years post-harvest, regardless of buffer treatment. We also observed consistent declines in post-metamorphic tailed frog density in all buffer treatments; however, because the 95% credible interval for the 0% treatment comparison included 1, due in part to small samples sizes, there is uncertainty surrounding the estimate of post-harvest change in that treatment. These findings were not consistent with our expectation of the greatest impact in the 0% treatment. Also inconsistent with our expectation was the response we observed for torrent salamanders, for which we observed a decline in density only in the FP treatment. We also had evidence of a decline for giant salamander in the FP treatment, though this result was not statistically significant seven and eight years post-harvest.

Timber harvest may impact stream-associated amphibian movement, either stream-network wide or between drainages, especially for terrestrial post-metamorphic individuals for which movement may be limited if habitats that provide adequate environmental conditions, such as temperature and moisture, do not exist (Grant *et al.* 2010). Decreases in habitat quantity or quality from timber harvest could affect stream-associated amphibian populations by altering emigration or immigration (Peterman *et al.* 2011; Chelgren and Adams 2017). Animals associated with streams may move along stream channels or terrestrially between streams (Fagan 2002; Grant *et al.* 2007). Previous research has concluded that stream-associated amphibian movement declines with an increasing density of log jams (Wahbe and Bunnell 2001) and that stream-associated amphibians resist movement across even relatively small (i.e., 13-m) gaps in stream channel riparian canopy (Cecala *et al.* 2014). Some researchers have found that post-metamorphic stream-associated amphibians traveled farther from streams in old-growth forests than in recently clearcut sites (Wahbe *et al.* 2004; Matsuda and Richardson 2005). A study of the impacts of clearcut timber harvest on amphibian movement in the southeastern USA found that several species of stillwater breeding amphibians avoided migrations through harvested areas (Todd *et al.* 2009). 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.

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).

Another consideration that merits discussion in terms of the delayed response we observed for some amphibian taxa in some treatments is that the Pacific Northwest experienced a drought that started in 2013 and lasted through the summer of 2015, as evidenced by reductions in annual water yield in even the reference basins (see Chapter 4 – *Stream Temperature and Cover* in this report). This drought resulted in record low spring and summer discharge rates and elevated stream temperatures throughout the area (Mote *et al.* 2016). Precipitation gauges located at nearby sites recorded extremely reduced precipitation levels in 2013, followed by less than average precipitation through the summer of 2015 (PRISM Climate Group 2020). In studies of fish and macroinvertebrates in headwater streams, droughts have been shown to reduce population densities, affect species composition, and alter reproduction (Cowx *et al.* 1984; Hakala and Hartman 2004). It is possible that the drought played a causative role in the delayed decline we observed for some amphibians seven and eight years post-harvest. Kaylor and colleagues (2019) found that Coastal Giant Salamanders in the Oregon Cascade Range were negatively impacted by the same region-wide drought, as reflected by declines in body condition. In a study of the Northern Dusky Salamander (*Desmognathus fuscus*), a stream salamander in the family Plethodontidae, researchers found that larval occupancy declined under drought conditions in North Carolina, which they suggested may have been related to increased mortality of eggs or larvae, or failure of adult females to oviposit (Price *et al.* 2012). In the same study, researchers observed a high rate of temporary adult emigration, which explained the relatively high adult survival rates under drought conditions (Price *et al.* 2012). In our study, under drought conditions seven and eight years post-harvest, we had evidence of increases in stream network-wide amphibian densities for all taxa in the reference. It is possible that drought conditions exacerbated the effects of buffer treatment by increasing egg or larval mortality, reducing reproduction, or inducing adult emigration from buffer treatment sites, especially for Coastal Tailed Frogs, for which post-metamorphs likely have the greatest mobility. The impacts of drought may not be immediately apparent (Hakala and Hartman 2004) and may have protracted reproductive consequences well after the drought has ended (Price *et al.* 2012).

It may seem like the extreme events experienced in the study sites during our investigation, including the drought in the post-harvest period and severe windthrow event in the pre-treatment period, add unwanted variability that make it more difficult to interpret a response to treatment. However, these events, though stochastic in nature, are a natural part of the managed landscape and provide an opportunity to investigate the response of headwater streams to management practices under the full range of natural conditions to which they are exposed. Indeed, if drought conditions did contribute to the negative response we observed for some species in buffer treatment sites, this is a phenomenon we would not have detected had the timing of the event not overlapped with our study period. We did not do a formal statistical analysis comparing amphibian densities before and after the windthrow event, in part because our statistical power to detect a difference would have been limited by the fact that only a subset of sites was impacted. However, all sites in the windthrow-affected areas were impacted regardless of the treatment to which they were assigned, and the most impacted sites were already grouped geographically into blocks for analysis.

We do not believe that the windthrow event greatly affected the response of stream-associated amphibians to treatment; however, the timing of the event may have affected our ability to distinguish differences among treatments.

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 the eight years post-harvest, most notably for Coastal Tailed Frog in all buffer treatments. Only continued monitoring of the amphibian populations at our study sites can provide information on future trajectories of potential change in population demographics.

Though many studies have provided evidence of a connection between management history and stream-associated amphibian occupancy or abundance, others have failed to find a link between harvest and amphibian occupancy or density for at least some of our focal amphibian taxa (Russell *et al.* 2004; Jackson *et al.* 2007; Olson *et al.* 2014). Differences in amphibian responses among studies may be related to variable study conditions and approaches. For example, some other studies have evaluated riparian buffers that were wider than those included in our study, e.g., 60 m (197 ft; Dupuis and Steventon 1999), 64 m (210 ft; Vesely and McComb 2002), and >46 m (151 ft; Stoddard and Hayes 2005), though riparian buffers for two of our 100% treatment sites exceeded 17.2 m (50 ft) along portions of the stream channel due to buffer requirements for unstable slopes. Riparian buffers in some other studies were also continuous, as opposed to the partial buffer in our FP treatment. Furthermore, some apparent differences in the conclusions regarding stream-associated amphibian occupancy or abundance in retrospective studies may be related to differences in the time since harvest, e.g., stand ages up to 15 years (Dupuis and Steventon 1999) and greater than 55 years (Stoddard and Hayes 2005). This premise is at least somewhat supported by the variable response we observed for some taxa and life stages in our study sites through time.

Most previous studies of stream-associated amphibians and land management are retrospective, with unknown historic amphibian distributions or densities. Retrospective studies can be biased if sites with a particular land management history happened to be located on lands more or less likely to support the taxa of interest. For example, Pollett and colleagues (2010) concluded that tailed frogs were less abundant in unbuffered streams than in streams with buffers or second-growth forests, however, they did not detect a single tailed frog in any unbuffered site. Since historic distribution and density of the species in the study sites is not known, non-detection could represent a response to clearcut harvest, or it could indicate that the species was not present in these sites historically. Importantly, none of the previous studies we report on accounted for the probability of detection in their estimates of density, which assumes that detectability was constant over space and time (MacKenzie *et al.* 2002). This assumption may be violated, especially in the post-harvest state, and even more so if there was clearcut harvest in the RMZ, which frequently results in slash and logging debris accumulations in the stream channel (Jackson *et al.* 2007). Finally, many studies have focused on a determination of occupancy, which does not reflect abundance.

The broad distribution of our study sites throughout western Washington gave us a unique and important opportunity to better understand the impacts of forest management actions on stream-associated amphibians over a broad spatial scale. Coupling our amphibian demographic study with an evaluation of genetic structure 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 the eight years post-harvest for some species in some buffer treatments, and for Coastal Tailed Frog in all buffer treatments. These results warrant continued study. Without further investigation, it is unclear if populations will rebound in buffer treatment sites, or whether densities will continue to decline, stabilize, or return to pre-harvest levels. This consideration is especially important given the temporally and spatially stochastic nature of disturbances, both natural and human-made, across western Washington. Species responses to timber management in headwater streams at broad spatial scales may be reflected at fine spatial scales (Stoddard and Hayes 2005). Continued monitoring of the amphibian response to treatment is strongly suggested to expand on our understanding of the long-term impacts of timber harvest and variable length buffers on stream-associated amphibians.

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APPENDIX A. SUMMARY TABLE OF SIGNIFICANT RESPONSES

Table A-1. Summary of statistically significant responses in the BACI analysis (i.e., P ≤0.05 or P ≤0.1 depending on response). Post Year is the number of post-harvest years over which a response was evaluated. The first symbol for each pairwise comparison indicates the direction of the difference in the change between the pre- and post-harvest periods; the second is the direction of the difference between the pre-harvest and extended periods (i.e., last post-harvest year sampled). Arrows indicate the direction of the difference; zeros indicate no difference. Green cells highlight where a difference in the post-harvest period was no longer evident in the extended period; yellow where the direction of the difference did not change between the post-harvest and extended periods; red where a lack of a difference in the post-harvest period developed into a difference in the extended period, or where there was a change in the direction of the difference between the post-harvest and extended periods.

				REF vs.			100% vs.		FP vs.
Response	**Variable**	**Metric**	**Post Year**	**100%**	**FP**	**0%**	**FP**	**0%**	**0%**
Wood	Loading	Small	8	↑ / 0	↑ / 0	↑ / 0	0 / 0	↑ / 0	↑ / 0
		Functional small	8	↑ / 0	↑ / 0	↑ / 0	0 / ↑	0 / ↑	0 / 0
		Large	8	↑ / 0	↑ / ↑	↑ / 0	0 / 0	0 / 0	0 / ↓
		Functional large	8	↑ / 0	↑ / 0	↑ / 0	0 / 0	0 / 0	0 / 0
Stream Temperature and Cover	Riparian Cover	CTD	5	↓ /↓	↓ /↓	↓ /↓	↓ /↓	↓ /↓	↓ /↓
		Effective shade	5	↓ /↓	↓ /↓	↓ /↓	↓ /↓	↓ /↓	↓ /↓
		Canopy closure 1-m	9	0 / 0	↓ / 0	↓ /↓	↓ / 0	↓ /↓	↓ / 0
		Canopy closure 0-m	9	0 / 0	↓ / 0	↓ /↓	↓ / 0	↓ /↓	↓ /↓
	Stream Temperature	Daily max	9	↑ / 0	↑ / ↑	↑ / ↑	0 / 0	↑ / 0	↑ / 0
Nutrient Export		Total-N	8	0 / 0	↑ / 0	↑ / ↑	0 / 0	↑ / 0	0 / 0
		Nitrate-N	8	0 / 0	↑ / 0	↑ / 0	0 / 0	↑ / 0	0 / 0
Stream Channel Characteristics	Hydrology	Wetted width	8	0 / 0	0 / 0	↓ /↓	0 / 0	↓ / 0	↓ /↓
		Bankfull width	8	0 / 0	0 / 0	↓ /↓	0 / 0	↓ /↓	↓ /↓
	Substrate	Fines and sand	8	0 / 0	0 / ↑	↑ / ↑	0 / 0	0 / 0	0 / 0
	Channel Units	Channel rise by steps	8	0 / 0	0 / 0	↓ /↓	0 / 0	↓ / 0	↓ / ↓
Stream-associated Amphibians	Stream Network-wide Density	Tailed frog larval	8	0 / ↓	↑ / ↓	0 / ↓	0 / ↓	0 / ↓	0 / 0
		Tailed frog post-metamorph	8	↓ /↓	0 / ↓	↑ / 0	0 / ↓	↑ / 0	↑ / ↑
		Torrent salamander	8	0 / 0	0 / ↓	↑ / 0	0 / ↓	↑ / 0	↑ / 0
		Giant salamander	8	0 / 0	↓ / 0	0 / 0	0 / 0	0 / 0	↑ / 0

Table A-2. Summary of statistically significant post-harvest responses that were not intended for analysis as part of the BACI design (i.e., $P \leq 0.1$). Post Year is the number of post-harvest years over which a response was evaluated. The first symbol for each pairwise comparison indicates the direction of the difference between treatments in the post-harvest period; the second symbol is the direction of the difference in the extended period (Post 8). Arrows indicate the direction of the difference; zeros indicate no difference. Green cells highlight where a post-harvest difference was no longer evident in the extended period; yellow where the direction of the difference did not change between the post-harvest and extended periods; red where no post-harvest difference developed into one in the extended period. N/A indicates comparisons that were not made because treatment implementation required the complete removal of trees in the 0% treatment.

			Post	REF vs.			100% vs.		FP vs.
Response	**Variable**	**Metric**	**Year**	**100%**	**FP**	**0%**	**FP**	**0%**	**0%**
Riparian Stand	Post-harvest cumulative tree mortality	% basal area/yr RMZ	8	0 / 0	0 / ↑	N/A	0 / 0	N/A	N/A
		% basal area/yr PIP	8	0 / ↑	↑ / ↑	N/A	0 / 0	N/A	N/A
	Post-harvest cumulative change in stand structure	Δ in live basal area RMZ	8	0 / 0	0 / ↑	N/A	0 / ↑	N/A	N/A
		Δ in live basal area PIP	8	0 / ↑	↑ / ↑	N/A	0 / 0	N/A	N/A
Large Wood Recruitment	Post-harvest cumulative large wood recruitment	Recruited pieces/ha PIP	8	↑ / 0	↑ / 0	N/A	0 / 0	N/A	N/A

GLOSSARY

Bankfull width (for streams; WAC 222-16-010): The measurement of the lateral extent of the water surface elevation perpendicular to the channel at bankfull depth. In cases where multiple channels exist, bankfull width is the sum of the individual channel widths along the cross-section.

Equipment limitation zone (*ELZ*; WAC 222-16-010): A 30-ft (9.1-m) wide zone measured horizontally from the outer edge of the bankfull width of all Type Np and Ns Waters.

Experimental treatments: Experimental treatments include three riparian buffer treatments that vary in the length of perennial non-fish-bearing stream length buffered and reference sites:

(a) *0% treatment:* Clearcut harvest with no riparian buffer;

(b) *Forest Practices treatment (FP treatment):* Clearcut harvest with one application of the buffer currently allowable under Forests and Fish Law for perennial non-fish-bearing streams (clearcut harvest with a two-sided 50-ft (15.2-m) buffer along ≥ 50% of the perennial non-fish-bearing stream length including buffers prescribed for sensitive sites: side-slope and headwall seeps, headwater springs, and Type Np intersections);

(c) *100% treatment:* Clearcut harvest with a two-sided 50-ft (15.2-m) buffer along the entire perennial non-fish-bearing stream length; and,

(d) *Reference:* unharvested reference located on a previously harvested site but having no management within the RMZ during the study period.

F/N break: Point at which a Type F (fish-bearing) Water becomes a Type Np (perennial non-fish-bearing) Water.

Forest Practice (*FP*; WAC 222-16-010): Any activity conducted on or directly pertaining to forest land and relating to growing, harvesting, or processing timber.

Habitat Conservation Plan (*HCP*): In Washington State, the Forest Practices (FP) HCP is a 50-year agreement that covers 60,000 miles of stream across 9.3 million acres of private and state forestlands. The goals of the FP HCP are to protect habitat of aquatic species, support economically viable and healthy forests, and create regulatory stability for landowners.

Headwall seep (WAC 222-16-010): A seep located at the toe of a cliff or other steep topographic feature and at the head of a Type Np Water that connects to the stream channel network via overland flow, and is characterized by loose substrate and/or fractured bedrock with perennial water at or near the surface throughout the year. One of the five currently recognized sensitive site categories.

Headwater spring (WAC 222-16-010): A permanent spring at the head of a perennial channel. Where a headwater spring can be found, it will coincide with the uppermost point of perennial flow (i.e., PIP). One of the five currently recognized sensitive site categories.

Mainstem channel: The dominant stream thread as defined by the greatest surface water volume.

Riparian Management Zone (*RMZ*; WAC 222-16-010): For Type Np Waters in western Washington, the area protected on each side of Type Np Waters, measured horizontally from the outer edge of the bankfull width. Areas within the RMZ can be either buffered or unbuffered (see WAC 222-30-021(2)(b) – *Western Washington riparian management zones/Western Washington* *protection for Type Np and Ns Waters/Sensitive site and RMZs protection along Type Np Waters*).

Sensitive sites (WAC 222-16-010): Areas near or adjacent to Type Np Waters that include any of the following five categories: headwall seep, side-slope seep, Type Np intersection, headwater spring, alluvial fan.

Side-slope seep (WAC 222-16-010): A seep within 100 ft (30.5 m) of a Type Np Water located on side-slopes greater than 20%, connected to the stream channel network via overland flow, and characterized by loose substrates and fractured bedrock, excluding muck, with perennial water at or near the surface throughout the year. Water delivery to the Type Np channel is visible by someone standing in or near the stream. One of the five currently recognized sensitive site categories.

Stream order (*e.g., first-, second-, third-order*): Definition of stream size based on hierarchy of tributaries where perennial streams with no upstream tributaries are first-order, streams downstream from the confluence of two first-order streams are second-order, streams downstream from the confluence of two second-order streams are third-order, and so on. Streams of lower order joining a higher order stream do not change the order of the higher-order stream (Strahler 1952).

Study period: Periods during which the study was conducted. The study currently has three study periods:

(a) *Pre-harvest period*: Period of data collection that occurred prior to harvest. For this study, the period commenced in 2006 and continued through 2008, for approximately three years;

(b) *Post-harvest period*: Period of data collection that occurred immediately after harvest. For this study, the period commenced once harvest was complete at a study site (approximately 2009) and continued for two to three years into 2010 or 2011 depending on the response variable and the timing of harvest in each study site; and,

(c) *Extended period*: Period of data collection that commenced after the post-harvest period and continued for up to seven years, until 2017.

Tributary: A secondary stream or channel that joins the mainstem or another tributary at a Type Np intersection.

Type F Water (*i.e., fish-bearing stream;* WAC 222-16-030): Segments of natural waters, other than Type S Waters (i.e., “shorelines of the state” under chapter 90.58 RCW), which are within the bankfull width of defined channels and contain fish habitat.

Type Np intersection (WAC 222-16-10): The intersection of two or more Type Np Waters. One of the five currently recognized sensitive site categories.

Type Np Water (*i.e., perennial non-fish-bearing stream;* WAC 222-16-030): All segments of natural waters within the bankfull width of defined channels that are perennial non-fish habitat streams. Perennial streams are waters that do not go dry at any time of a year of normal rainfall. However, for the purpose of water typing, Type Np Waters include the intermittent dry portions of the perennial channel below the uppermost point of perennial flow (i.e., PIP).

Type Ns Water (*i.e., seasonal, non-fish-bearing stream;* WAC 222-16-030): All segments of natural waters within the bankfull width of the defined channels that are not Type S, F, or Np Waters. These are seasonal, non-fish habitat streams in which surface flow is not present for at least some portion of a year of normal rainfall and are not located downstream from any stream reach that is a Type Np Water. Type Ns Waters must be physically connected by an above-ground channel system to Type S, F, or Np Waters.

Uppermost point of perennial flow (*i.e., perennial initiation point or PIP)*: Point of the start of perennial flow in the stream channel network. PIPs include both headwater spring and headwall seep sensitive site categories.