Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington

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WASHINGTON DEPARTMENT OF NATURAL RESOURCES

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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 Eastside Type F Riparian Effectiveness Program, and was conducted under the oversight of 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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EXECUTIVE SUMMARY

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

Headwater streams, which comprise approximately 65% of the total stream length on forestlands in western Washington, are largely understudied relative to their frequency in the landscape. We evaluated the effectiveness of riparian forest management prescriptions for small non-fish-bearing (Type N) headwater stream basins in western Washington by comparing current prescriptions to alternatives with longer riparian leave-tree buffers and 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 (see Chapter 1 – *Introduction and Background*). The focus of the study was on Forests and Fish-designated species of stream-associated amphibians. We also evaluated riparian processes affecting in-channel wood recruitment and loading, stream temperature and shade, discharge, nutrient export, suspended sediment export (SSE), channel characteristics, litterfall input and detritus export, biofilm and periphyton, macroinvertebrate export, and downstream fish density and population structure (see **Supplement 1** for a complete list of response variables). The results of this study will inform the efficacy of current Forest Practices rules, including how landowners can continue harvesting wood resources while protecting important headwater habitats and associated species.

We used a Before-After Control-Impact (BACI) study design with blocking to examine how harvest treatments influenced resource response. We collected pre-harvest data from 2006 through 2008 and post-harvest data from 2009 into 2011 (see Chapter 2 – *Study Design*). Study sites included 17 Type N stream basins located in managed second-growth conifer forests across western Washington. Sites were restricted to Type N basins less than 54 ha (133 ac) in size with relatively competent lithologies. We evaluated four experimental treatments, including an unharvested **Reference** (i.e., in the harvest rotation but withheld from harvest; n = 6) and three alternative riparian buffer treatments involving clearcut harvest of the entire basin. Riparian buffer treatments included the following: **100% treatment** (a two-sided 50-ft [15.2-m] riparian leave-tree 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 buffer treatments were implemented between October 2008 and August 2009 (see Chapter 3 – *Management Prescriptions*). Results presented in this summary include those that had statistically significant pre- to post-harvest changes that differed between treatments (alpha of 0.05 or 0.1, depending on the response and clarified in each chapter).

We found that harvest of timber in and adjacent to streamside riparian forests directly affected tree mortality, tree fall rates, and large wood recruitment to streams. The highest mortality rates and greatest reductions in density and basal area occurred in the FP treatment RMZ buffers and the buffers surrounding the uppermost points of perennial flow (PIPs; see Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers).* Mortality and tree fall rates in FP treatment RMZs were significantly greater than in either the 100% treatment or reference RMZs. Tree mortality and tree fall were significantly greater in both the 100% and FP treatment PIPs relative to reference rates. Windthrow-associated tree fall in riparian buffers increased large wood (≥10 cm [4 in] diameter) recruitment to channels in the 100% and FP treatments (see Chapter 6 – *Wood Recruitment and Loading*). However, the vast majority of recruited trees were completely suspended above the active stream channel. We observed a significant post-harvest increase in small wood (<10 cm [4 in] diameter) in the channel in the 0% treatment relative to the FP and 100% treatments, and an increase in in-channel large wood in all three buffer treatments relative to the reference. Increases in in-channel wood loading in treated sites may have been responsible for the changes we saw in stream channel characteristics. We observed a significant post-harvest increase in stream pool length in all three riparian buffer treatments (see Chapter 11 – *Stream Channel Characteristics*). The pre- to post-harvest change in stream bankfull and wetted widths, and the proportion of the stream channel rise attributed to steps, was significantly less in the 0% treatment than in any other treatment including the reference.

Shade decreased and water temperature increased in all buffer treatments, with the greatest change in temperature occurring during the July–August period (see Chapter 7 – *Stream Temperature and Cover*). Both maximum and minimum daily temperatures increased significantly in all buffer treatments over some part of the year. The maximum daily temperature showed signs of recovery toward pre-harvest conditions downstream from the harvest unit (i.e., within 100 m downstream of the harvest boundary); however, stream temperature remained above pre-harvest levels at five of the six sites where downstream recovery could be assessed. While we observed post-harvest reductions in canopy across all riparian buffer treatments, that reduction did not result in differences in biofilm ash-free dry mass (AFDM) or chlorophyll *a* by treatment following harvest (see Chapter 13 - *Biofilm and Periphyton*).

We measured discharge, SSE and nutrient export in eight study sites, four each in the Olympic and Willapa Hill ecoregions. Annual runoff increased in all buffer treatment sites as a result of harvest, but the magnitude of change varied by season and return interval (see Chapter 8 – *Discharge*). As expected, total water yield increased as a function of the proportion of the total area of each basin harvested, which was 88% and 94% in the two FP treatments and 45% and 89% in the two 100% treatments. We saw very little change in the 100% treatment site, where only 45% of the basin was harvested. All sites exhibited changes in discharge, and mean discharge increased in the FP and 0% treatment, but not in the 100% treatment. Baseflows decreased in the 100%, were largely unchanged in the FP, and increased in the 0% treatment.

The sites monitored for SSE appeared to be supply limited (i.e., sediment transport was limited by the sediment delivered to the stream from the adjacent uplands) both before and after harvest (see Chapter 10 – *Sediment Processes*). Most of the sediment export occurred during late fall or early winter storm events, and the relative magnitude of export was stochastic across sites and treatments. In four of the six buffer treatment sites, SSE was greater during clearcut harvest implementation or in the two year post-harvest period, but spikes in sediment export were of similar magnitude to those observed in one of the two reference sites during the same periods.

Mean total nitrogen (N) and nitrate-N concentrations increased in all buffer treatments. The estimated change was greatest in the 0%, intermediate in the FP, and lowest in the 100% treatment, consistent with an increase in the proportion of the watershed harvested, but only the 0% differed statistically from the other buffer treatments (see Chapter 9 – *Nutrient Export*).

Overall, total litterfall input was slightly higher after harvest in the 100% treatment, lower in the FP treatment and lowest in the 0% treatment; however, we observed statistical differences only for deciduous inputs between the 0% treatment and the other treatments (see Chapter 12 – *Litterfall Input and Detritus Export*). Total detritus export decreased in the 0% treatment relative to the reference, and in the FP and 0% treatments relative to the 100% treatment.

We observed some changes in macroinvertebrate export after harvest, but did not detect any major reductions in macroinvertebrate export or major shifts in functional feeding groups (see Chapter 14 – *Macroinvertebrate Export*). Collector-gatherer export in biomass per day decreased in the 0% treatment relative to the FP treatment, but increased in the FP treatment relative to the reference and the 100% treatment.

Treatment effects for stream-associated amphibians (Coastal Tailed Frog [*Ascaphus truei*], and torrent [*Rhyacotriton*] and giant [*Dicamptodon*] salamanders) were variable among genera and, for tailed frogs, life stage (see Chapter 15 – *Stream-associated Amphibians*). We found statistical support for a negative effect of buffer treatment on the density of giant salamanders in the FP treatment. We found that larval Coastal Tailed Frog density increased significantly in the 100% and FP treatments relative to the reference and 0% treatment. Post-metamorphic Coastal Tailed Frog density also increased, but only in the 0% treatment. We lacked evidence of a treatment response for torrent salamanders, except when stream reaches that were visibly obstructed by dense matrices of logging slash in the form of downed wood, litter and fines were included in the analysis; here, torrent salamander density increased significantly in the 0% treatment.

Based on results from six study sites, we found that cutthroat trout (*Oncorhynchus clarkii*) density and population structure downstream of study sites were highly variable across sites, months and years (see Chapter 16 – *Downstream Fish*). Variability in total fish abundance was not correlated with physical stream habitat metrics such as gradient and percent pool area. Consistently low recapture rates for passive integrated transponder (PIT)-tagged fish over the course of the study provided evidence of a high level of fish emigration from, and/or mortality within, study reaches.

During the two years post-harvest, the 100% buffer treatment was the most effective in maintaining pre-harvest conditions, the FP was intermediate, and the 0% treatment was least effective compared to reference sites (see Chapter 17 – *Summary and Discussion*). The collective effects of timber harvest, both in terms of statistical significance and magnitude, were most apparent in the 0% treatment. The direction and magnitude of changes for the 100% and FP treatments did not differ statistically for some metrics, including large wood recruitment, wood cover and loading, water temperature, discharge and channel unit metrics, and Coastal Tailed Frog density. However, some differences existed between the 100% and FP treatments, including for tree mortality and stand structure, riparian cover, detritus and macroinvertebrate export and giant salamander density. While post-harvest differences in the response of treatments were readily apparent across a suite of variables, we noted no consistent negative impacts for stream-associated amphibians.

CHAPTER 1 - INTRODUCTION AND BACKGROUND

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1-1. INTRODUCTION

Washington State has a relatively long history of regulating forest management on private and state forestlands. The Forest Practices Act first established regulatory goals for forest practices in 1974. In 1987, after more than a decade of contention over the adequacy of forest practices regulations, the Timber, Fish and Wildlife agreement was finalized (TFW; Washington Forest Practices Board, WFPB 1987). The TFW agreement was not a legal agreement, but rather an agreement to work together to reach consensus to make the best decisions for the management of forest-based natural resources in Washington. The high-level goals of TFW covered fisheries, wildlife, archeological and cultural resources, water quality and quantity, and the forest products industry. All of the major Washington forest practices stakeholders, including environmental groups, state agencies, the timber industry, and Native American tribes, approved the TFW agreement.

A pivotal outcome of TFW was the expansion of riparian protection by establishing Riparian Management Zones (RMZs) near the banks of streams, rivers and lakes. The timber industry agreed to leave trees along fish-bearing streams to provide shade and a source of wood for recruitment to the stream, and to help stabilize stream banks to protect water quality and habitat for fish and wildlife. Another important outcome of TFW was the development of an adaptive management program to use information from ongoing research and monitoring to help fill knowledge gaps to inform potential policy changes. Research and monitoring needs were outlined in a work plan developed by the Cooperative Monitoring, Evaluation and Research (CMER) committee. This work plan recommended scientific projects to answer unresolved technical and scientific questions related to the impacts of forest management on fish, wildlife and water. However, despite these important advancements, the structure of TFW adaptive management was constrained not only by limited funding to address projects, but also by lack of a formal vehicle to either move the process forward, or effectively link it to policy representation among stakeholder groups.

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 listing of hundreds of stream segments with water quality problems under the Clean Water Act (CWA), the Forests and Fish Report became the Forests and Fish Law on 1 July 2001 (WFPB 2001). The Forests and Fish Law not only expanded upon the protections provided by the TFW agreement but also addressed the aforementioned limitations of TFW. These advancements included:

1) The requirement for a well-funded and functional adaptive management program, which was expressly intended to comply with both the federal ESA and the CWA (USFWS 1999), and directed the WFPB to adopt permanent rules meeting those objectives. Forest practice rules under the Forests and Fish Agreement (hereafter Forest Practices rules) were developed through negotiations among federal (National Marine Fisheries Service, US Environmental Protection Agency, and USFWS), state (The Office of the Governor of the State of Washington, Washington State Department of Ecology [hereafter Ecology], Washington Department of Fish and Wildlife [WDFW], and Washington Department of Natural Resources [WADNR]), tribal and county governments, and private forest landowners. The goal was to "develop biologically sound and economically

practical solutions that would improve and protect riparian habitat on non-federal forestlands in the State of Washington” (USFWS 1999). These rules were designed to meet the four focal goals that the WFPB had established:

   a. Provide compliance with the ESA for aquatic and riparian-dependent species (including Forests and Fish-designated stream-associated amphibians),
   b. Restore and maintain riparian habitat to support a harvestable supply of fish,
   c. Meet the requirements of the CWA for water quality, and
   d. Keep the timber industry economically viable in the state of Washington.

2) The requirement for a formal vehicle to move the process forward. This included:
   a. An adaptive management coordinator to administrate the science program within CMER so that robust science needed to answer questions on the impacts of forest management could be developed in an environment insulated from potential policy bias, and to provide a link to a policy stakeholder group to enable reciprocal information flow between policy makers and scientists.
   b. A formal policy stakeholder group that could decide how adaptive management science might alter Forest Practices rules, and inform the adaptive management science program of important questions that science might address.

In effect, Forest Practices rules were designed to maintain diverse riparian functions and features, including large wood recruitment, shade to mediate light inputs and changes in stream temperature, sediment storage, bank stability, nutrient retention and export, litterfall inputs, and other riparian features important to both riparian forest and aquatic system conditions.

1-1.1. MANAGEMENT OF NON-FISH-BEARING STREAMS UNDER FOREST PRACTICES RULES

Timber harvest guidelines prescribed under Forest Practices rules were developed to achieve the Forests and Fish Law resource objectives. New forest management practices included the expansion of riparian protections to include non-fish-bearing streams, improvement of forest roads and culverts, and identification and protection of unstable slopes, among others.

Non-fish-bearing “headwater” streams, or Type N Waters, comprise more than 65% of the total stream length on forestlands in western Washington (Rogers and Cooke 2007). During negotiations leading to the development of current Forest Practices rules, scientists representing the various stakeholder groups had to address which aquatic and riparian-dependent species would be the focus of protection in Type N Waters. Stakeholder scientists ranked molluscs first and amphibians second; however, policy liaisons familiar with the state legislature advised the selection of amphibians as the focal taxon for coverage as aquatic resources. Therefore, stakeholders selected six stream-associated amphibians that were dependent on riparian habitat and presumed to be the most susceptible and/or least resilient to the potential impacts of forest management as compared to other species of amphibians in forestlands in Washington State. Subsequent to this selection, one of these species was classified into two species in 2001, raising the total number of covered species to seven.

1-1.2. PROBLEM STATEMENT – LACK OF INFORMATION ON THE EFFECTIVENESS OF TYPE N WATERS RIPARIAN MANAGEMENT PRESCRIPTIONS

At the time of Forest Practices negotiations, almost no published studies addressed the efficacy of riparian buffers for Type N Waters or provided clear guidance addressing riparian buffer design, most notably for stream-associated amphibians. Moreover, the few studies available (some of which did not have published results until some years after negotiations were finalized) were either retrospective (Bisson *et al.* 2002; Raphael *et al.* 2002), or lacked the power needed to interpret observed responses for the aquatic resources specified in Forest Practices rules (O'Connell *et al.* 2000; Jackson *et al.* 2003). As a consequence, CMER directed a study, the “Type N Experimental Buffer Treatment Study” (hereafter, Type N Study), that would allow more confident conclusions to be drawn about the relative effectiveness of alternative riparian management prescriptions in meeting Forest Practices resource goals for Type N Waters.

1-2. STUDY OBJECTIVE AND CRITICAL QUESTION

Our objective was to evaluate the effectiveness of current westside riparian management prescriptions for Type N Waters under Forest Practices rules by comparing the current riparian buffer prescription to longer and shorter alternatives within the RMZs of Type N Water systems. We evaluated the influence of these alternative riparian management prescriptions on biotic and physical resources and processes in Type N Waters and examined which prescription(s) were the most effective in maintaining species, and stream and riparian processes, to inform the efficacy of Forest Practices rules through the adaptive management process.

We developed an experimental design to answer the following critical question in basaltic lithologies of the coastal areas and the south Cascades of Washington State:

> What is the magnitude, direction (positive or negative), and duration of change in riparian-related inputs (light, litterfall, sediment, and wood) and the response of instream (amphibians, water temperature, habitat) and downstream components (export of nutrients, organic matter, macroinvertebrates, and sediment; water temperature; and fish in the downstream fish-bearing [Type F] reach) associated with a range of experimental timber harvest treatments that vary in the length of riparian buffer retained within RMZs of Type N Waters relative to untreated reference conditions?

The results of the Type N Study will inform the efficacy of current Forest Practices rules, including how landowners can more effectively protect important headwater habitats and associated species while harvesting wood resources that can be used to create a diversity of products.

1-3. REPORT STRUCTURE

The report is a series of chapters, with Chapters 1 through 4 addressing study objectives, design, and implementation, and Chapters 5 through 16 addressing responses of various in- or near-channel, upland, and downstream export variables. We present a summary of results in Chapter 17. Northwest Indian Fisheries Commission, WDFW, Ecology and Weyerhaeuser Company personnel contributed to data collection, analysis and report writing.

1-4. REFERENCES

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CHAPTER 2 - STUDY DESIGN

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2-1. IDENTIFICATION OF RESPONSE VARIABLES

2-1.1. FOREST PRACTICES – FUNCTIONAL OBJECTIVES

The Washington Forest Practices Board (WFPB) developed a series of key questions, Resource Objectives and Performance Targets for adaptive management, outlined in Schedule L-1 of the Forests and Fish Report (USFWS 1999). The Overall Performance Goals defined in Schedule L-1 are to uphold forest practices that will not, either singly or cumulatively, significantly impair the capacity of aquatic habitat to: a) support harvestable levels of salmonids, b) support long-term viability of other covered species, or c) meet or exceed water quality standards. Further, Resource Objectives are defined for key aquatic conditions and processes affected by forest practices. These Resource Objectives are intended to meet the Overall Performance Goals, and consist of both broad statements of objectives for the major watershed functions potentially affected by forest practices (Functional Objectives) and measurable criteria defining specific, attainable target forest conditions and processes (Performance Targets). Our study was designed to evaluate whether Forest Practices rules pertaining to Type N Waters produce forest conditions (and processes that create those conditions) that achieve agreed-upon Resource Objectives. We identified key response variables that would enable us to address Resource Objectives for watershed functions affected by forest practices, which included:

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.

2-1.2. CONCEPTUAL MODELS THAT FACILITATED SELECTION OF RESPONSE VARIABLES

Resource responses can be driven by shifts in the trophic energy pathway as well as physical changes to habitat-forming processes. We utilized energy pathway and landscape conceptual models to aid in the selection of response variables for inclusion in the study; for an in-depth discussion of the energy and landscape pathway conceptual models used, see Appendix II in Hayes *et al.* (2005). For example, the energy pathway conceptual model can be used to illustrate how alternative riparian buffer configurations may affect stream-associated amphibians and downstream exports to Type F (fish-bearing) Waters. Stream-associated amphibians were selected as a key response variable in the study because stakeholders identified them as one of the important biotic resources to be protected in Type N Waters (USFWS 1999). Forest management could affect amphibians and downstream exports through changes to stream temperature (Johnson and Jones 2000), primary productivity (Murphy 1998), or invertebrate composition or abundance (Hawkins *et al.* 1982; Hawkins 1988), among other things.

Selected response variables were related to WFPB Resource Objectives and derived from energy pathway and landscape conceptual models. These included riparian vegetation, wood, water temperature, flow, nutrient export, litterfall and detritus, sediment, channel characteristics, periphyton, macroinvertebrates, stream-associated amphibians, downstream fish, and trophic pathways.

2-2. SITE-SELECTION CRITERIA IMPOSED BY THE STUDY DESIGN

Prior to the selection of sites for inclusion in the study, we identified 10 criteria and associated constraints important to the study design (**Table 2-1**). The inclusion of stream-associated amphibian species as a response variable placed important constraints on site selection. Six of the seven Forest Practices-designated amphibians occur exclusively (n = 5) or largely (n = 1) in Westside forestlands (**Figure 2-1**). We selected sites that supported four of these amphibian species: Coastal Tailed Frog (*Ascaphus truei*) and Olympic, Columbia, and Cascade Torrent Salamanders (*Rhyacotriton olympicus*, *R. kezeri*, and *R. cascadae*). 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, and therefore require different sampling techniques than the species that were a focus of this study. Although Coastal (*Dicamptodon tenebrosus*) and Cope's (*D. copei*) Giant Salamanders are not covered under Forest Practices rules, 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 entire study area and was appropriate for the amphibian genetic component of the study.

Table 2-1. Criteria used and the associated limits for each criterion during the Type N Study site-selection process, 2004–2006.

Step in Process	Criterion	Limit
Study design criteria	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)
	Type N basin size	12–49 ha (30–120 ac)
	Stream order	Second-order stream basins (Strahler 1952)
	Stream network geometry	Minimum of 75 m (246 ft) of stream between the F/N break and nearest downstream tributary intersection
Ownership criteria	Stand age	>70% of stands in study site between 30 and 80 years old during harvest treatment window
	Harvest timing	Buffer treatment sites: harvest Apr 2008–Mar 2009; References: no harvest
	Area owned	>80% owned by single participating landowner

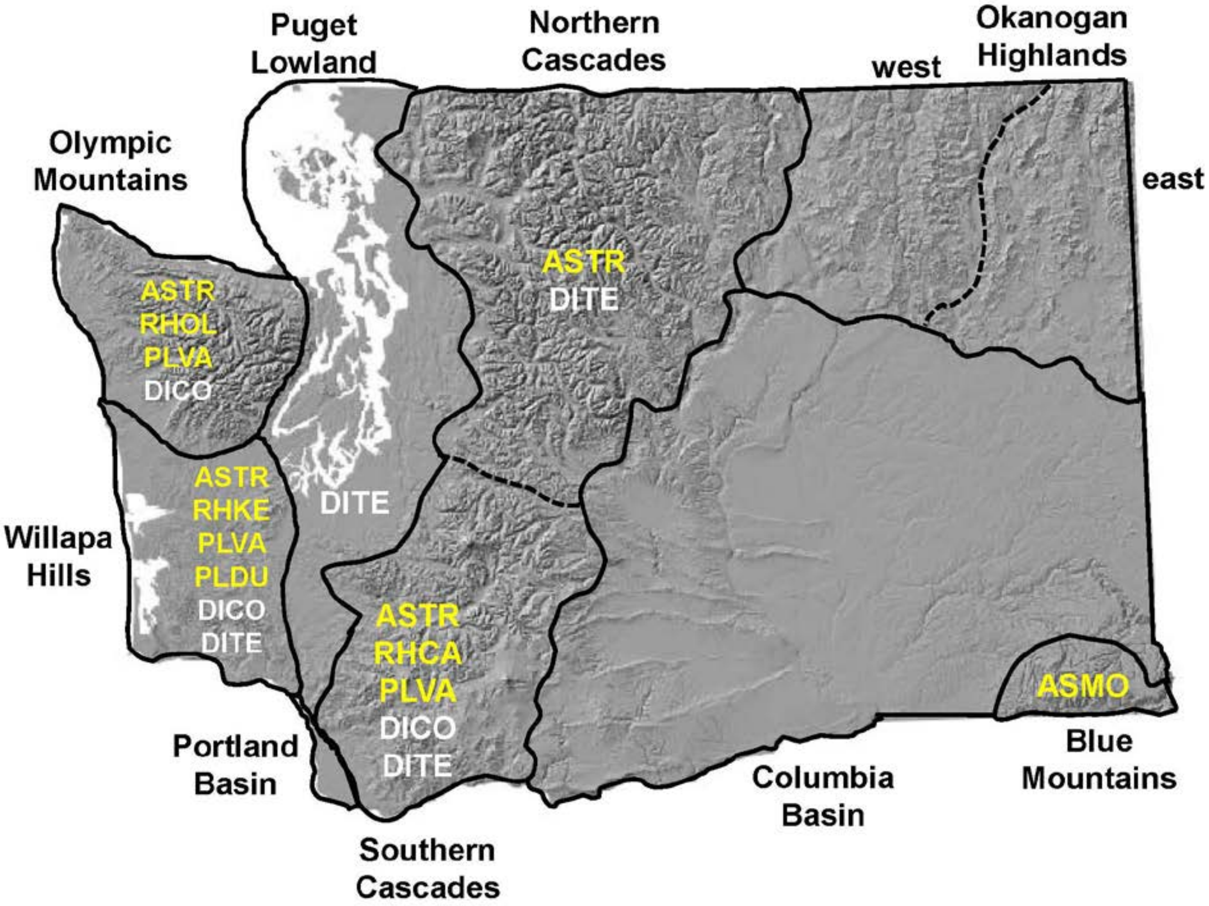

Figure 2-1. Physiographic regions of Washington State and distributions of stream-associated amphibian species by region. Forest Practices-designated amphibians (yellow font) included in our study were Coastal Tailed Frog (*Ascaphus truei*; ASTR) and Olympic (*Rhyacotriton olympicus*; RHOL), Columbia (*R. kezeri*; RHKE) and Cascade (*R. cascadae*; RHCA) Torrent Salamanders. Forest Practices-designated amphibians not included in our study were Rocky Mountain Tailed Frog (*A. montanus*; ASMO), and Dunn's (*Plethodon dunni*; PLDU) and Van Dyke's (*P. vandykei*; PLVA) Salamanders. Coastal (*Dicamptodon tenebrosus* = DITE) and Cope's (*D. copei* = DICO) Giant Salamanders (in white font) are not designated amphibian species under Forest Practices rules but were included in the study.

2-2.1. STUDY DESIGN CRITERIA

We limited our site selection to three physiographic regions: Olympic Mountains, Willapa Hills and Southern Cascades (south of the Cowlitz River), because these regions had the greatest number of Forest Practices-designated amphibians (Jones *et al.* 2005). We further limited selection of study sites based on factors known to influence the distribution of these amphibian species. For example, Forest Practices-designated amphibians rarely occur above 1,219 m (4,000 ft) elevation in Washington State and the upper elevation limit within their range declines slightly with increasing latitude (Dvornich *et al.* 1997). Consequently, we limited sites to those located at elevations less than 1,067 m (3,500 ft) and 1,219 m (4,000 ft) in the Olympic and South Cascade physiographic regions, respectively. 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. Additionally, Coastal Tailed Frogs occur in streams between 5% and 50% (3 to 27 degrees) slope, which also captures almost the entire range of stream gradients over which the other designated species are found (Adams and Bury 2002); therefore, we limited sites to those with a slope in this range. The Coastal Tailed Frog, Columbia Torrent Salamander and giant salamanders also have a greater probability of occurrence on competent lithologies[1] (Dupuis *et al.* 2000; Wilkins and Peterson 2000); therefore, 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. Finally, since Coastal Tailed Frogs rarely reproduce in small (often first-order) basins in western Washington (Hayes *et al.* 2006) we initially restricted site selection to include second-order streams (Strahler 1952); however, we later found it necessary to relax the stream order criteria to include first-, second-, and third-order streams to obtain the desired number of study sites.

To maximize the influence of the buffer treatments and reduce confounding effects we wanted the harvest units to be the size of the entire Type N basin[2] where possible. Additionally, we were interested in studying harvest units that were operationally meaningful (McIntyre *et al.* 2009). Landowners indicated that the minimum unit size typically harvested was about 12 ha (30 ac), while the maximum harvest unit size is limited by Forest Practices to 49 ha (120 ac) without an exception based on review by an interdisciplinary science team (WFPB 2001). In order to maximize the influence of the buffer treatments and reduce confounding effects, we initially constrained sites to Type N basins between 12 and 49 ha (30 to 120 ac). We subsequently found it necessary to relax the Type N basin size to include basins up to 54 ha (133 ac) to obtain the desired number of study sites.

Finally, we required a minimum of 75 m (246 ft) of stream below the F/N break within which to sample fish for the fish portion of the study. We verified that landowners would not harvest along this portion of the stream during our study period and that there were no tributary intersections within this reach. These two requirements were necessary to ensure that the intended experimental treatment and other management activities were not confounded.

2-2.2. OWNERSHIP CRITERIA

Inclusion of study sites relied on commitments from landowners to manage them according to our treatment specifications (i.e., harvest layout and timing). We requested that landowners commit to applying harvest treatments from April 2008 through March 2009. We limited study sites to those with at least 70% of 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 old is infrequent in Washington State. Finally, because multiple ownership of the same study site would greatly complicate the coordination and implementation of treatments, we

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[1] Competent lithologies produce long-lasting, large, durable clasts or coarse grain sizes.

[2] Type N basins are the extent or area of land where surface water from rain and melting snow or ice converge to a single point, in this case at the F/N break, where Type N Waters join the Type F Waters.

limited study sites to those for which more than 80% of the Type N basin had a single landowner.

2-3. FOREST PRACTICES RIPARIAN MANAGEMENT PRESCRIPTIONS FOR WESTSIDE TYPE N WATERS

Among other forest management practices, all shorelines of the state (Type S), Type F, and Type N (including both Np and Ns) Waters in Washington State are protected by a Riparian Management Zone (RMZ) under Forest Practices rules. Type Np Waters are perennial streams with no fish habitat that do not go dry any time of the year, and Type Ns Waters are seasonal streams with no fish habitat and no surface flow for at least some portion of a year of normal rainfall (WAC 222-16-030). Riparian management prescriptions for Type N Waters vary by water type and location, that is, east versus west of the Cascade Mountain crest. The RMZ for Type Np and Ns Waters in western Washington includes the following requirements (WAC 222-30-021 (2)):

1) **Equipment limitation zone (ELZ):** A two-sided 30-ft (9.1-m) wide zone measured horizontally from the outer edge of the bankfull width of Type Np or Ns Water where equipment use and other forest practices are specifically limited. On-site mitigation is required if ground-based equipment, skid trails, stream crossings (other than existing roads), or partially suspended cabled logs exposes the soil on more than 10% of the surface area of the zone. Mitigation measures (e.g., water bars, grass seeding, mulching) must be designed to replace the equivalent of lost functions, especially prevention of sediment delivery.

2) **Riparian protection:** A two-sided 50-ft (15.2-m) wide no-harvest riparian buffer along at least 50% of the Type Np stream length, including:

    a. **Stream buffer:** Required two-sided buffers must start at the F/N break and continue upstream for: (1) a minimum of 500-ft (152.4-m) for Type Np Waters longer than 1000 ft (305 m); (2) at least equal to the greater of 300 ft (91 m) or 50% of the entire length for Type Np Waters greater than 300 ft but less than 1000 ft; or (3) buffered in their entirety for Type Np Waters less than or equal to 300 ft.

    b. **Sensitive site buffers:** No-harvest buffers specific to each sensitive site category (WAC 222-16-010; **Table 2-2**).

The precise distribution of buffered reaches depends on the locations of sensitive sites and other priority features (WFPB 2001). Rules were negotiated to allow flexibility to landowners during forest management activities.

Table 2-2. Sensitive site definitions and RMZ requirements under Forest Practices rules.

Sensitive Site Type	Definition	RMZ Requirement
Headwall seep	A seep located at the toe of a cliff or other steep topographical feature and at the head of a Type Np Water which 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	50-ft (15.2-m) no-harvest buffer around the outer perimeter of the perennially saturated area
Side-slope seep	Seeps within 100 ft (30.5 m) of a Type Np Water located on side-slopes which are >20%, connected to the stream channel network via overland flow, and characterized by loose substrate and fractured bedrock, excluding muck, with perennial water at or near the surface throughout the year	50-ft (15.2-m) no-harvest buffer around the outer perimeter of the perennially saturated area
Type Np intersection	Intersection of two or more Type Np Waters	56-ft (17.1-m) radius no-harvest buffer centered on intersection
Headwater spring	Permanent spring at the head of a perennial channel, coinciding with the uppermost extent of Type Np Waters	56-ft (17.1-m) radius no-harvest buffer centered on spring
Alluvial fan	An erosional land form consisting of a cone-shaped deposit of water-borne, often coarse-sized sediments	No harvest within

2-4. EXPERIMENTAL DESIGN

The design included a pre-harvest period of data collection, the implementation of clearcut harvests with alternative riparian buffer configurations, and a post-harvest period of data collection.

2-4.1. BEFORE-AFTER CONTROL-IMPACT (BACI) DESIGN

We used a Before-After Control-Impact (BACI) design whereby we established baseline conditions across study sites, implemented harvest at buffer treatment sites and continued monitoring response variables of interest after harvests were applied. The BACI design allowed us to compare harvested sites to both their pre-harvest baseline conditions as well as to unharvested references. An advantage of this design is that it controls for the effect of large-scale temporal variation (e.g., annual climate variation) by establishing relationships between the control (i.e., unharvested reference) and impact (i.e., clearcut harvested) sites in the pre- versus post-harvest periods (Smith 2002), allowing us to determine whether observed differences among treatments are associated with environmental variation or forestry practices. The study design incorporated three years of pre-harvest sampling 2006–2008, and two years of post-harvest sampling 2009–2010. The minimum pre- and post-harvest period considered sufficient to capture natural annual variability inherent to forested landscapes in western Washington is two years; however, value always exists to extending sampling over longer timelines. The original intent of the study design was to extend sampling into the next harvest rotation if possible. If results demonstrated no buffer treatment effects in the two year post-harvest period, for example, extended sampling could investigate if there was a lag effect associated with the buffer treatments. Alternatively, if results showed a treatment effect in the two year post-harvest period, then this design would allow sampling over a longer period to monitor recovery during a harvest rotation.

2-4.2. EXPERIMENTAL TREATMENTS

Study sites were Type N basins of second-growth forested stands. To maximize the potential impact of alternative riparian buffer treatments, we requested that clearcut harvests be applied to the entire Type N basin. Landowners were mostly successful in fulfilling this objective, with a few exceptions (see section 3-2.4. *Areas Within Buffer Treatment Sites Where Clearcuts Could not be Applied*). 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 removal. The four experimental treatments included (**Figure 2-2**):

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 riparian leave-tree buffer (i.e., two-sided 50-ft [15.2-m]) throughout the RMZ,

3) **Forest Practices treatment (FP):** clearcut harvest with current Forest Practices riparian leave-tree buffer (i.e., clearcut harvest with a two-sided 50-ft [15.2-m] riparian buffer along ≥50% of the RMZ, including buffers prescribed for sensitive sites—side-slope and headwall seeps, headwater springs, Type Np intersections and alluvial fans), and

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

Alignment of buffer treatments along a gradient, with RMZ riparian buffer lengths both longer (100% treatment) and shorter (0% treatment) than those required, allowed us to evaluate the relative effectiveness of alternative treatments in meeting the four key goals established by the WFPB (see Chapter 1 – *Introduction and Background*).

Harvest followed Forest Practices rules with the exception of the riparian buffer maintained within the RMZ. A 30-ft (9.1-m) ELZ was maintained along all Type Np and Ns Waters, regardless of treatment. During study development, we considered whether exploring differences in length or width of riparian buffers might better inform current Forest Practices rules. Jackson and colleagues (2001) found that riparian buffers along headwater streams protected stream banks, limiting input of logging debris and minimizing bank failure and erosion. Research on riparian buffer effectiveness in western Oregon has shown that most of the change in microclimate from the stream to the upland forest occurs within the first 14 m (45 ft) from the stream (Olson *et al.* 2002). Given that the riparian buffers for Type Np Waters in Washington are required to be 50 ft (15.2 m) wide, we concluded that changing buffer length rather than width had the greatest potential to result in changes that would inform Forest Practices rules.

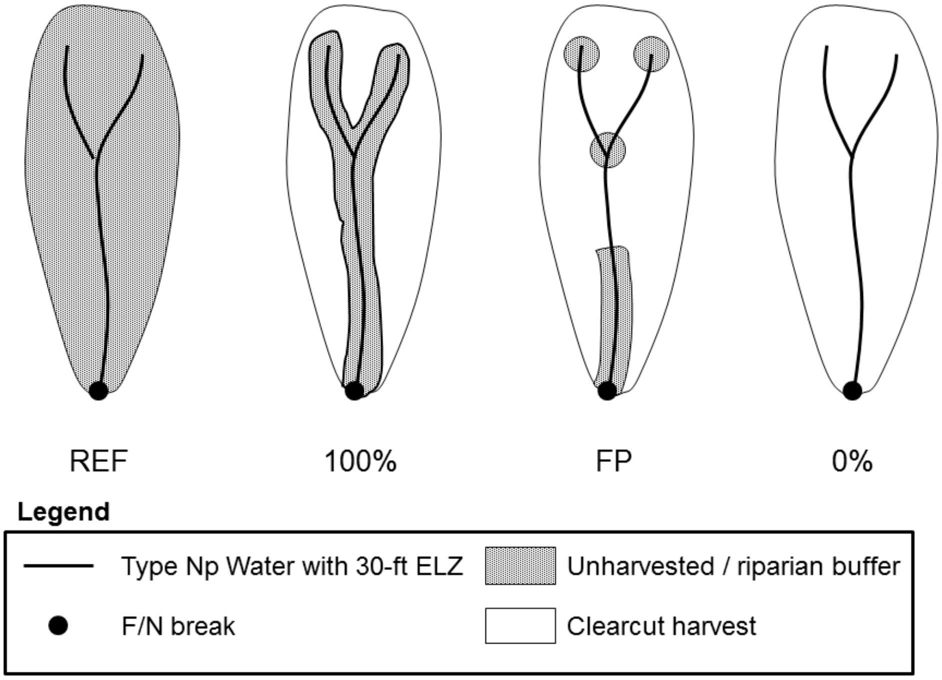

Figure 2-2. Schematic of the four experimental treatments included in the Type N Study. Treatments include unharvested reference sites (REF) and sites receiving a clearcut harvest with one of three riparian buffer treatments along the Type Np Water RMZ: two-sided 50-ft (15.2-m) riparian buffers of 100%, Forest Practice (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).

2-5. SITE SELECTION

2-5.1. IDENTIFICATION OF STUDY SITES MEETING STUDY DESIGN AND OWNERSHIP CRITERIA

Selection of study sites for inclusion in the Type N Study began in June 2004 and continued through August 2006. For a detailed description of selection criteria and the site-selection process, see McIntyre and colleagues (2009). Based on the study design criteria listed in **Table 2-1,** we used a Geographic Information System (GIS), specifically ArcMap (ESRI 2004), first to identify 35,957 Type N basins within our geographic range of interest. Fish distribution endpoints were identified using the Washington Department of Natural Resources (WADNR) GIS hydrolayer recently updated to include an F/N break based on a GIS-logistic regression model. We then applied the rest of the study design criteria, after which 6,125 study sites remained. The site list was further reduced to 496 with the application of ownership criteria (**Table 2-1)**.

2-5.2. ON-SITE VALIDATION OF STUDY DESIGN AND OWNERSHIP CRITERIA

We conducted on-site surveys to validate study design and ownership criteria. Approximately 30% and 25% of study sites visited failed to meet the competent lithology criteria and stream gradient criteria, respectively, and stand age and recent harvest data provided by landowners were accurate in only about 75% of the basins visited. In the end, we identified 131 sites that met study design and ownership criteria (<0.5% of the total Type N basins originally identified within our geographic range of interest). We conducted field sampling at the 131 sites that met study design and ownership criteria to determine if Forest Practices-designated amphibians were present. We detected Forest Practices-designated amphibians at 48 study sites, further reducing the number of potential study sites to 0.1% of the original pool of Type N basins.

On-site electrofishing surveys conducted between December 2005 and June 2006 revealed inaccuracies in the GIS -logistic regression model used to predict the location of the F/N break, or upstream extent of fish distribution, within each basin. The location of the F/N break was determined using specific protocols for conducting presence/absence electrofishing surveys on forestlands in Washington State (WFPB 2002). The F/N break was at the location predicted by the model in only three (6%) of the 48 remaining candidate sites. Thirty-seven (77%) had field verified F/N breaks located downstream of the modeled F/N break locations and eight (17%) had field-verified F/N breaks upstream of the modeled F/N break locations. Since the location of the F/N break determines the corresponding Type N basin size, we recalculated basin sizes in ArcMap based on the field-verified locations of F/N breaks. Seventeen sites were greater than 49 ha (120 ac) and no longer met the Type N basin size criteria, although we decided to retain one 54 ha (133 ac) site that was only slightly larger than the criteria for potential inclusion in the study. Movement of the F/N break downstream at one potential study site expanded the Type N basin to include forest stands that did not meet the minimum stand age criterion of 30 years. Moving the F/N break upstream in one location resulted in dividing the Type N basin into two subbasins, both of which met study criteria. The net result of field validation of the F/N break was that 32 candidate sites remained for potential inclusion in our study.

Field surveys also revealed inaccuracies in the hydrology layer used to determine stream order. The primary reason for restricting site selection to second-order basins was to increase the likelihood that Forest Practices-designated amphibians would be present. We were able to relax our criteria for stream order to include the few remaining first- and third-orders sites in our candidate pool based on the field verification of the presence of Forest Practices-designated amphibians.

2-5.3. LANDOWNER HARVEST TIMING RESTRICTIONS

As part of the study design landowners were required to harvest sites according to treatment specifications and restrict harvest activities at reference sites from April 2008 through March 2009. These landowner restrictions on harvest management further reduced the candidate pool of sites to 20. We removed another two study sites from consideration because of slope instability and resultant harvest restrictions. At the end of the site selection process, we had identified 18 study sites that met all criteria. Negotiations with landowners regarding harvest timing and layout specifications continued through August 2006 when we confirmed that all 18 sites were approved for use in the study and permits allowing access for research purposes were in place.

2-5.4. SITES USED FOR DOWNSTREAM FISH RESPONSE

The study design required at least 75 m of stream below the F/N break to conduct fish sampling. Of the 18 candidate sites available, only six were suitable for the fish component of the study once we considered the proximity of the F/N break to a downstream tributary confluence (e.g., <75 m) and other physical and/or biological constraints. For a full description of the sites that were selected for inclusion and why some sites were not included see McIntyre and colleagues (2009) and Chapter 16 – *Downstream Fish.*

2-6. ASSIGNMENT OF STUDY SITES TO BLOCKS AND TREATMENTS

We blocked (grouped) study sites based on geography to minimize variability and assigned sites within each block to one of the four treatments. Sites within a block were located within the same physiographic region (Olympic, Willapa Hills, and South Cascade). We had one block of four sites in the Olympic region, two blocks of four sites each and one block of two sites in the Willapa Hills region, and one block of three sites in the South Cascade region (**Figure 2-3**).

As noted earlier, all participating landowners committed to allowing pre- and post-harvest monitoring throughout the initial study period, 2006–2010: landowners contributing buffer treatment sites to the study committed to harvest those sites during the period April 2008 through March 2009, and landowners contributing unharvested references to the study committed to restricting harvest through 2010 at a minimum. We also requested that landowners consider restricting future harvest activities in buffer treatment sites until 2020 to accommodate sampling 10 years post-harvest. This would allow us to determine if there were lag effects in any response variables, and to sample amphibian genetics after one generational turnover had occurred (approximately 7 to 8 years). Post-harvest genetics sampling would allow us to determine if timber harvest was associated with any genetic changes, including changes in genetic diversity (see Cornuet and Luikart 1996; Luikart *et al.* 1998).

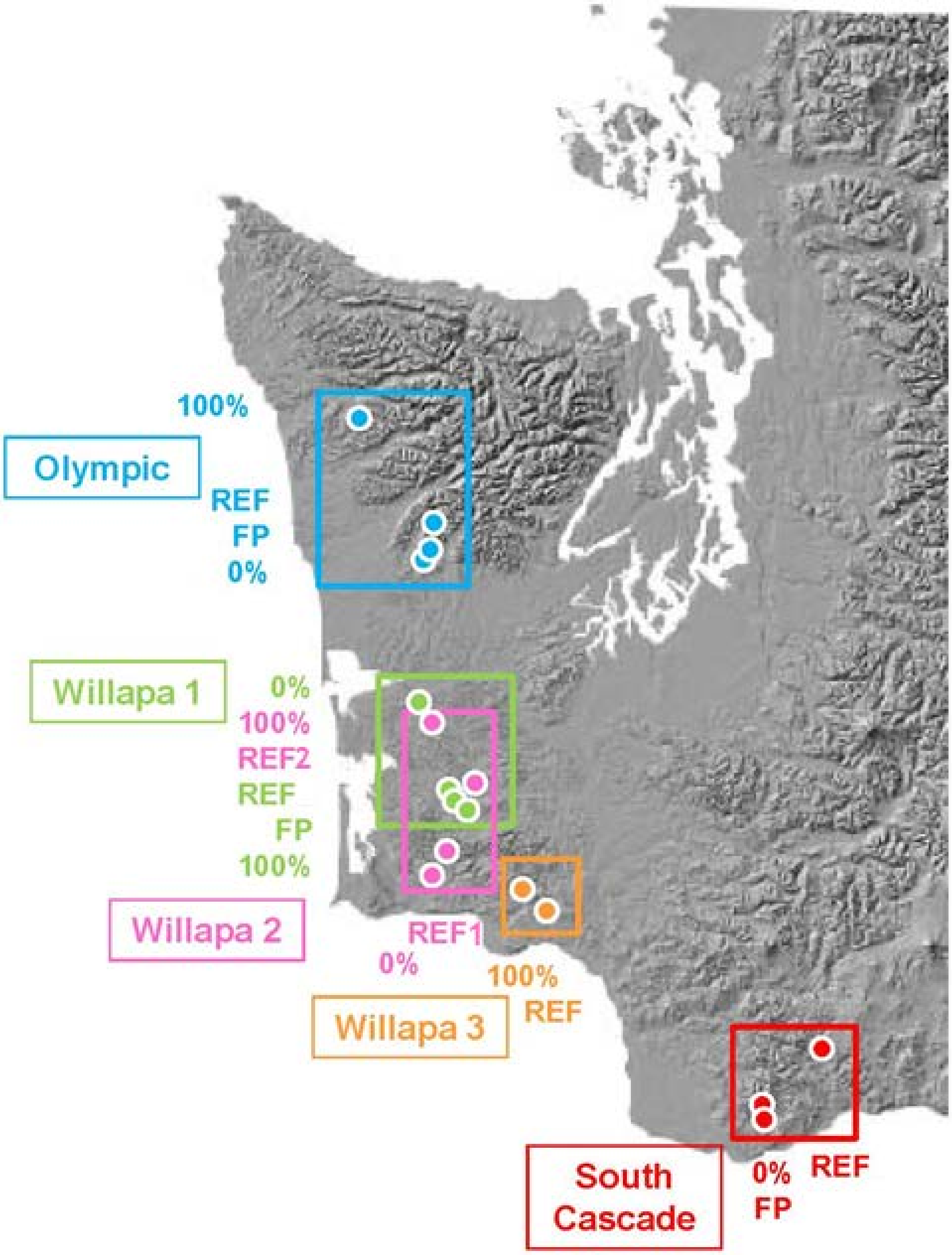

Figure 2-3. Distribution of study sites and treatment allocation for the Type N Study, 2006–2010. Study sites are blocked (grouped) based on geography. The five blocks are color-coded such that sites in a block are the same color. REF = reference sites and 100%, FP, and 0% = 100%, Forest Practices and 0% buffer treatments, respectively.

We were not able to assign some treatment types to particular study sites; unharvested references could only be located on public ownerships because private landowners would not agree to exclude sites from harvest for the duration of the proposed initial study period, and restricted harvest activities on federal forestlands prevented us from prescribing buffer treatments on National Forest sites. As a result, only sites located on state forestlands (WADNR) were available for inclusion as both buffer treatments and references. 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) meant we would be able to measure downstream exports in only eight of the study sites. Finally, only six sites were included in the fish response portion of the study (see McIntyre *et al.* 2009 and Chapter 16 – *Downstream Fish* for details). Given these constraints, we randomized the assignment of treatments in blocks as follows:

1) **Olympic block (OLYM):** We randomly assigned treatments to each of the four study sites available in the Olympic physiographic region. All four study sites were suitable for the assessment of export variables, and two sites (FP and 0% treatment sites) were suitable for assessment of downstream fish response.

2) **Willapa 1 block (WIL1):** Ten study sites were available in the Willapa Hills physiographic region: eight sites were spread throughout the coastal region of the Willapa Hills, and two were located south and east of the others that together constituted the Willapa 3 block. We wanted to have one complete block in the Willapa Hills for use in the fish portion of the study, so we first considered the five sites that were suitable for fish and how to organize one block out of these. Out of these five sites, four were located on state forestland, and one was on privately owned forestland and was only available as a buffer treatment. Of the four state-owned sites, we randomly selected two as unharvested reference sites (one for the Willapa 1 "fish block" and one for the Willapa 2 block). The remaining two sites became buffer treatments. We randomly assigned buffer treatments to each of the three treatment sites within this block. We then randomly selected one of the two unharvested references and grouped it with the three buffer treatment sites to become the Willapa 1 block. All four study sites in the Willapa 1 block were suitable for the assessment of export variables and the downstream fish response.

3) **Willapa 2 block (WIL2):** We randomly assigned buffer treatments to the remaining three study sites in western Willapa Hills, which along with the remaining state-owned reference became the Willapa 2 block. Due to unfavorable economic conditions, harvest at the site that was assigned the FP treatment was not applied so this site acted as a second reference in this block. None of the sites in this block were included in the assessment of export variables or downstream fish response.

4) **Willapa 3 block (WIL3):** The two geographically separated study sites located south and east of the eight coastal sites became the Willapa 3 block. One site was only available as a reference because of constraints imposed by the presence of marbled murrelet habitat, and the other site was available as a clearcut harvest with 100% treatment. Neither of these sites was suitable for inclusion in the assessment of export variables or downstream fish response.

5) **South Cascade block (CASC):** One of the sites in the South Cascade physiographic region was located in the Gifford Pinchot National Forest and was therefore only available as a reference. We assigned buffer treatments randomly to the three remaining sites. When grouped with the reference these sites became the South Cascade block. Due to unfavorable economic conditions, harvest in the 100% treatment was not completed. As a result, we removed this site from the study. None of the sites in this block were included in the assessment of export variables or downstream fish response.

We established an acronym for each study site, based on the combination of the block to which it was assigned and the treatment applied. We will use these acronyms in tables and figures throughout the remainder of the report (**Table 2-3**).

Table 2-3. Blocks, treatments, and study site acronyms used in tables and figures throughout the Type N Study final report.

Block	Treatment Type	Study Site Acronym
Olympic	Reference	OLYM-REF
	100% treatment	OLYM-100%
	Forest Practices treatment	OLYM-FP
	0% treatment	OLYM-0%
Willapa 1	Reference	WIL1-REF
	100% treatment	WIL1-100%
	Forest Practices treatment	WIL1-FP
	0% treatment	WIL1-0%
Willapa 2	Reference 1	WIL2-REF1
	Reference 2	WIL2-REF2
	100% treatment	WIL2-100%
	0% treatment	WIL2-0%
Willapa 3	Reference	WIL3-REF
	100% treatment	WIL3-100%
South Cascade	Reference	CASC-REF
	Forest Practices treatment	CASC-FP
	0% treatment	CASC-0%

2-7. STUDY SITE DESCRIPTIONS

After final considerations based on field verification of study design and ownership criteria, 18 study sites remained for inclusion in the Type N Study. Unfortunately, after commencement of data collection, we had to remove one site from the study because the application of the buffer treatment was not implemented in its entirety due to landowner economic decisions, leaving 17 study sites. These sites included Type N, first-, second- and third-order stream basins located over a large geographic area of western Washington. Drainages included in the study were located along the Clearwater, Humptulips, and Wishkah Rivers in the Olympic physiographic region; the North, Willapa, Nemah, Grays and Skamokawa Rivers and Smith Creek in the Willapa Hills physiographic region; and the Washougal River and Trout Creek in the South Cascade physiographic region (45.81° to 47.65°, −122.26° to −124.20°, elevation 22 to 601 m [72 to 1,972 ft]).

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. Measurable rainfall is recorded for an average of 150 days each year in the interior valleys and for 190 days in the mountains and along the coast, with heavier intensities occurring along the windward slopes of the Cascade Mountains. 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°C to 1.2°C (27.7 to 34.2 Fahrenheit) and 22.2°C to 25.0°C (72.0 to 77 Fahrenheit) 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) across our study sites. Study sites were located in managed second-growth forests dominated by Douglas fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*) on private (Fruit Growers Supply Company, Longview Timber, Rayonier, and Weyerhaeuser Company), state (WADNR), and federal (Gifford Pinchot and Olympic National Forests) forestlands. The 17 study sites (**Figure 2-3**) ranged from 12 to 54 ha (30 to 133 ac) and were composed primarily of stand ages ranging from 30 to 80 years (**Table 2-4**). Average stream-adjacent valley wall slopes ranged from 18% to 65% (10 to 33 degrees; **Table 2-5**), as measured perpendicular to the stream channel along 50-ft (15.2-m) transects in riparian stand vegetation plots (see Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers*). Sites were located in areas dominated by competent lithology types, and with average Np channel gradients ranging from 14% to 35% (8 to 19 degrees). Three study sites had unforested areas in the form of rock quarries (WIL1-FP and CASC-FP) and/or talus slopes (CASC-FP and CASC-0%). We present study design and site-selection criteria for each study site in **Table 2-6**.

Table 2-4. The proportions of each of the 17 study sites within each of seven stand age range categories. Stand ages are presented for the age of the stand near the time of harvest implementation (2008). Proportions of the study site within each stand age category are estimates only since precise calculations were not available for most landowners. “Other” includes rock quarries and talus slopes.

		Stand Age Range							
Block	**Treatment**	**<30**	**30–40**	**>40–50**	**>50–60**	**>60–70**	**>70–80**	**>80**	**Other**
OLYM	REF	-	0.60	-	0.22	-	-	0.18	-
	100%	-	0.20	0.72	-	-	-	0.08	-
	FP	0.01	-	0.33	0.67	-	-	-	-
	0%	0.05	-	0.67	0.28	0.01	-	-	-
WIL1	REF	-	0.01	0.38	0.59	-	0.01	-	-
	100%	-	<0.01	0.98	0.01	-	-	-	-
	FP	-	-	0.44	0.43	-	-	-	0.13
	0%	0.02	0.61	0.37	-	-	-	-	-
WIL2	REF1	-	0.21	0.78	-	-	-	-	-
	REF2	0.01	-	0.07	0.91	-	-	-	-
	100%	0.16	0.31	0.52	-	-	-	-	-
	0%	-	-	1.00	-	-	-	-	-
WIL3	REF	-	-	-	0.01	0.81	0.18	-	-
	100%	-	-	1.00	-	0.00	-	-	-
CASC	REF	-	-	-	-	0.05	0.95	-	-
	FP	-	0.01	0.85	0.07	-	-	-	0.07
	0%	0.02	-	0.20	0.53	-	-	-	0.25

Table 2-5. The average stream-adjacent valley wall percent slope (degrees) and proportions of the valley slope adjacent to the mainstem channel and secondary tributaries within each of six slope range categories at each of the 17 study sites.

			Stream-adjacent Valley Wall Percent Slope (degrees) Range											
			Mainstem						**Secondary Tributaries**					
Block	**Treat-ment**	**Avg % (deg)**	**<20 (<11)**	**20–39 (11–21)**	**40–59 (22–31)**	**60–79 (31–38)**	**80-99 (39-45)**	**>100 (>45)**	**<20 (<11)**	**20–39 (11–21)**	**40–59 (22–31)**	**60–79 (31–38)**	**80–99 (39–45)**	**>100 (>45)**
OLYM	REF	62(32)	0.04	0.15	0.24	0.31	0.22	0.04	0.05	0.10	0.20	0.54	0.11	0.00
	100%	54(28)	0.10	0.15	0.33	0.28	0.08	0.05	0.13	0.11	0.28	0.32	0.15	0.02
	FP	43(23)	0.19	0.19	0.33	0.29	0.00	0.00	0.17	0.27	0.35	0.21	0.00	0.00
	0%	28(16)	0.32	0.54	0.11	0.04	0.00	0.00	0.17	0.67	0.17	0.00	0.00	0.00
WIL1	REF	34(19)	0.12	0.51	0.31	0.06	0.00	0.00	0.60	0.40	0.00	0.00	0.00	0.00
	100%	48(25)	0.02	0.18	0.50	0.30	0.00	0.00	0.04	0.50	0.39	0.07	0.00	0.00
	FP[1]	50(27)	0.03	0.22	0.51	0.24	0.00	0.00	-	-	-	-	-	-
	0%	53(28)	0.05	0.19	0.16	0.43	0.16	0.00	0.06	0.16	0.53	0.24	0.02	0.00
WIL2	REF1[2]	65(33)	0.07	0.09	0.22	0.29	0.26	0.07	-	-	-	-	-	-
	REF2	36(20)	0.05	0.50	0.40	0.05	0.00	0.00	0.12	0.69	0.15	0.04	0.00	0.00
	100%	59(30)	0.06	0.08	0.29	0.27	0.29	0.01	0.03	0.24	0.45	0.24	0.05	0.00
	0%	35(19)	0.15	0.38	0.35	0.12	0.00	0.00	0.75	0.25	0.00	0.00	0.00	0.00
WIL3	REF	22(12)	0.49	0.29	0.16	0.01	0.04	0.00	0.67	0.33	0.00	0.00	0.00	0.00
	100%	18(10)	0.60	0.33	0.08	0.00	0.00	0.00	1.00	0.00	0.00	0.00	0.00	0.00
CASC	REF	32(18)	0.30	0.28	0.28	0.14	0.00	0.01	0.32	0.54	0.14	0.00	0.00	0.00
	FP	30(17)	0.24	0.54	0.20	0.02	0.00	0.00	0.38	0.25	0.38	0.00	0.00	0.00
	0%[1]	32(18)	0.26	0.42	0.21	0.09	0.02	0.00	-	-	-	-	-	-

[1] First order site lacking any secondary tributaries.

[2] Site includes one small secondary tributary insufficient for riparian vegetation plot installation and slope measurements not recorded.

Table 2-6. Block and treatment assignment, landowner, and study design and site selection criteria for 17 study sites included in the Type N Study. The four treatments include unharvested reference sites (REF) and sites receiving a clearcut harvest with one of three riparian buffer treatments along the Type Np Water RMZ: two-sided 50-ft (15.2-m) riparian buffers of 100%, Forest Practice (FP), and 0%. Elevation is the elevation at the field-verified F/N break. Stream gradient refers to the average stream gradient for the entire Type Np stream network as calculated using a 10-meter digital elevation model (DEM) in ArcMap (ESRI 2004). Basin size is the Type N basin size.

Block	**Landowner**	**Treatment**	**Elevation (m [ft])**	**Lithology**	**Stream Gradient (% [°])**	**Stream Order**	**Basin Size (ha [ac])**
OLYM	Olympic NF	REF	163 (535)	Basalt flows and flow breccias	18 (10)	3	54 (133)
	WADNR / Fruit Growers Supply Company[1]	100%	72 (236)	Tectonic breccia	27 (15)	3	28 (68)
	Rayonier	FP	277 (909)	Basalt flows and flow breccias	25 (14)	3	17 (41)
	Rayonier	0%	233 (764)	Basalt flows and flow breccias	31 (17)	2	13 (32)
WIL1	WADNR	REF	200 (656)	Basalt flows and flow breccias	19 (11)	2	12 (29)
	WADNR	100%	198 (650)	Basalt flows and flow breccias	18 (10)	2	31 (76)
	WADNR	FP	197 (646)	Basalt flows and flow breccias	19 (11)	1	15 (37)
	Weyerhaeuser[2]	0%	87 (285)	Terraced deposits	16 (9)	3	28 (69)
WIL2	Weyerhaeuser[2]	REF1[3]	183 (600)	Basalt flows and flow breccias	34 (19)	2	19 (48)
	WADNR	REF2	228 (748)	Basalt flows and flow breccias	18 (10)	2	16 (41)
	Weyerhaeuser[1]	100%	22 (72)	Basalt flows and flow breccias	21 (12)	3	26 (65)
	WADNR	0%	159 (522)	Basalt flows	21 (12)	2	17 (41)

Table 2-6. (continued)

Block	Landowner	Treatment	Elevation (m [ft])	Lithology	Stream Gradient (% [°])	Stream Order	Basin Size (ha [ac])
WIL3	WADNR	REF	241 (791)	Basalt flows	14 (8)	3	37 (92)
	WADNR	100%	351 (1152)	Basalt flows	19 (11)	2	23 (58)
CASC	Gifford Pinchot NF	REF	601 (1972)	Tuffs and tuff breccias	21 (12)	2	49 (120)
	WADNR	FP	450 (1476)	Andesite flows	16 (9)	2	26 (64)
	WADNR	0%	438 (1437)	Andesite flows	29 (16)	1	14 (36)

[1] The downstream 2.2 ha (5.4 ac) of this Type N study site was owned by Fruit Growers Supply Company; however, the portion of the study site under their ownership was not harvested as a part of the buffer treatment application for our study.

[2] Owned by Weyerhaeuser Company during site selection, pre-harvest sampling, harvest application, and the majority of post-harvest sampling. Purchased by Hancock Timber Resource Group in February 2011.

[3] Intended to be a FP treatment, but harvest did not occur due to the economy. See Chapter 3 – *Management Prescriptions.*

2-8. SCOPE OF INFERENCE

Scope of inference is limited by the site selection criteria listed. Inference can only be made to Type N basins located in second-growth forests on lands managed for timber production, dominated by competent lithologies, located in western Washington (including the Olympic, Willapa Hills, and South Cascade (south of the Cowlitz River) physiographic regions), and consistent with our other selection criteria (size, gradient, etc.).

2-9. STATISTICAL ANALYSIS APPROACH

We designed this study to evaluate response differences among treatments at the site scale, not to investigate within-site variability. Though we could evaluate within-site variability for some responses, we do not formally address those comparisons in this report. In general, analyses following the BACI design evaluated the generalized null hypothesis:

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

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.

Randomization during site selection, when possible, helps ensure that there is 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 (detailed in each chapter) 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, basin × time interaction, and basin × 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.

While data for most variables were collected at every study site, flumes with turbidity and flow sensors were only placed in eight study sites in two blocks (the Olympic and Willapa 1 blocks; **Table 2-7)** due to logistical constraints (see **Supplement 1** for a complete list of response variables included in the study). Additionally, because of the limited number of sites with downstream reaches suitable for fish sampling, the fish portion of the study was restricted to only six sites (0% and 50% buffers in the Olympic block and all sites in the Willapa 1 block; **Table 2-7**). Finally, we collected tissues for stable isotope analysis from amphibians across all study sites, when available. We collected samples for fish at the six sites included in the fish component of the study, and only collected periphyton, litterfall, detritus and macroinvertebrates samples in the eight study sites in which turbidity and flow were evaluated. 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 analysis approach used within the BACI design, and presents units of measure in the most appropriate unit (i.e., English or metric), with equivalents in parentheses.

Table 2-7. Response variables identified for inclusion in the Type N Study. Number of sites/blocks indicates the number of sites for which data were collected, as well as the number of blocks in which those sites are contained.

Variable Group	Variable	Total Sites	Block				
			OLYM	WIL1	WIL2	WIL3	CASC
In- or Near-Channel	Amphibian occupancy and density	17	4	4	4	2	3
	Amphibian genetics	17	4	4	4	2	3
	Periphyton standing crop	17	4	4	4	2	3
	Water, air and soil temperature	17	4	4	4	2	3
	Channel gross morphology	17	4	4	4	2	3
	Large wood loading	17	4	4	4	2	3
	Stream substrate	17	4	4	4	2	3
	Bank erosion	17	4	4	4	2	3
Downstream and Export	Fish density and quality	6	2[1]	4	-	-	-
	Stable isotopes						
	Fish	6	2[1]	4	-	-	-
	Amphibians	17	4	4	4	2	3
	All else	8	4	4	-	-	-
	Nutrients	8	4	4	-	-	-
	Macroinvertebrates	8	4	4	-	-	-
	Detritus	8	4	4	-	-	-
	Sediment	8	4	4	-	-	-
	Stream flow	8	4	4	-	-	-
	Water Temperature	17	4	4	4	2	3
Riparian Input	Stand growth/survival	17	5	4	4	2	3
	Large wood recruitment	17	5	4	4	2	3
	Shade	17	5	4	4	2	3
	Litterfall	8	4	4	0	0	0
	Sediment	17	5	4	4	2	3

[1] OLYM-0% and the OLYM-FP sites were included in the fish density, quality and stable isotopes analyses.

As with many scientific studies, and especially those involving ecological processes, our statistical analysis was limited by sample size, variability among plots, sites and blocks, and missing replicates of some treatments in some blocks. We suspect that for many of the comparisons with marginal P-values (0.05 to 0.15), a larger sample size would increase the ability to distinguish differences among the treatments and increase our confidence in interpreting results. It is for these reasons that we set α and β at 0.1 for some variables *a priori* (e.g., Underwood 1997; Welsh and Ollivier 1998). We clarify the alpha level used for each response in individual chapters. Interpretation of results should 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, will be an integral part of appropriately evaluating our results.

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CHAPTER 3 - MANAGEMENT PRESCRIPTIONS

Aimee McIntyre, Marc Hayes, Eric Lund, and Timothy Quinn

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3-1. FOREST PRACTICES RULES FOR TYPE N WATERS

Washington State Forest Practices rules apply to state and private forest landowners lacking a Habitat Conservation Plan (HCP). These regulations are outlined in Title 222 Washington Administrative Code (WAC) – *Forest Practices Rules* and in the Forest Practices Rules, Board Manual and Act (WFPB 2001). Forest Practices rules dictate specific requirements for forest management activities around Type Np and Ns Waters. Both private landowners (Rayonier and Weyerhaeuser) participating in riparian buffer treatment implementation in the Type N Study followed Forest Practices rules; however, Washington Department of Natural Resources (WADNR) lands are covered by an HCP that is more restrictive than Forest Practices rules for timber harvest adjacent to Type N Waters. For the purpose of this study, WADNR agreed to apply our experimental treatments and to follow regulations for forest management activities along Type N Waters as described by Forest Practices rules.

3-2. BUFFER TREATMENT APPLICATION

3-2.1. HARVEST TIMING

Beginning in 2004, we worked with the landowners of the 12 riparian buffer treatment sites to establish agreements for how and when harvest would occur, with the goal of scheduling all harvest activities from April 2008 through March 2009. However, due to limitations associated with the global economic decline that began in December 2007 and took a particularly sharp downward turn in September 2008, harvest in one site (OLYM-0%) was not completed until August 2009 and harvest in one FP treatment site in the Willapa 2 block was postponed indefinitely and retained as an additional unharvested reference (WIL2-REF1). As a result, clearcut harvests with buffer treatments were applied to 11 of 17 sites: 0% treatment in four sites, FP treatment in three sites, and 100% treatment in four sites (**Table 3-1**). Harvests began in July 2008 and were completed by August 2009, and lasted from two to six months, with an average duration of four months.

Table 3-1. Harvest dates and duration of harvest for 11 study sites receiving buffer treatments for the Type N Study, 2008–2009.

Block	Treatment	Harvest Dates		Approximate Duration (months)
		Begin	End	
OLYM	100%	February 2009	March 2009	2
	FP	July 2008	October 2008	4
	0%	June 2009	August 2009	3
WIL1	100%	October 2008	April 2009	6
	FP	October 2008	March 2009	5
	0%	October 2008[1]	January 2009	3
WIL2	100%	January 2009	April 2009	3
	0%	July 2008	November 2008	4
WIL3	100%	July 2008	November 2008[2]	4
CASC	FP	November 2008	March 2009	4
	0%	November 2008	March 2009	4

[1] Approximately 2.4 of 28 ha (6 of 69 ac) of windthrow located in the uppermost extent of the study site was salvaged in April 2008, approximately five months prior to harvest in the rest of the site.

[2] Approximately 2 of 23 ha (5 of 58 ac) located in the uppermost extent of the study site had a delayed harvest that did not occur until August 2009, approximately eight months after the rest of the site had been harvested.

3-2.2. HARVEST IMPLEMENTATION

All timber harvest adhered to the guidelines outlined under Washington State Forest Practices rules (WAC 222-30), with the exception of the length of the riparian buffer in the riparian management zone (RMZ). Timber removal in all study sites was even-aged harvest consisting of ground-based logging systems (including shovel and skidder) and cable yarding, except in the 100% treatment in the Willapa 2 block, where western redcedar (*Thuja plicata*) of a greater age were also removed using helicopter yarding. Other Forest Practices rules adhered to included:

- Ground-based logging under Forest Practices rules requires that the transport of logs across Type Np and Ns Waters minimize the potential for damage to public resources, and that skidding logs and driving ground-based equipment through defined channels with flowing water is not allowed (WAC 222-30-070 (1) (b)).

- Cable yarding was utilized in all harvest units, and where logs were transported across Type Np Waters, they were fully suspended above the water. Reasonable care was taken to minimize damage to the vegetation providing shade to understory vegetation, stumps and root systems where timber was yarded from or across a RMZ or sensitive site (WAC 222-30-060 (4)).

- Uphill yarding was the standard (WAC 222-30-060 (5)).

- Skid trails were kept to a minimum width, were outsloped where practical, and were at least 30 ft (9.1 m) from the outer edge of the bankfull width of the unbuffered portions of Type Np or Ns Waters (WAC 222-303-070 (7)).

- In order to maintain stream bank integrity, operators avoided disturbing brush and stumps (WAC 222-30-030).

- Felling of trees was generally directional away from streams, though trees may be felled into Type Np Water if logs are removed as soon thereafter as practical (WAC 222-30-050 (1)).

- No bucking or limbing was performed on trees or portions of trees lying within the bankfull width of Type Np Waters, in RMZ or in sensitive sites and reasonable care was taken to avoid felling trees into the RMZ (WAC 222-30-050).

- The two-sided 30-ft (9.1-m) equipment limitation zone (ELZ) applied to the entirety of the RMZ, regardless of whether or not a riparian buffer was maintained (WAC 222-30-021 (2)).

- Timber harvest was not conducted on potentially unstable slopes, which resulted in riparian buffers in RMZs wider than the minimum two-sided 50-ft (15.2-m) buffer for two study sites (see 3-2.5. *Riparian Buffer Configuration*).

3-2.3. BUFFER TREATMENT CONFIGURATION

Forest Practices rules specify the minimum riparian buffer length and configuration required in the RMZ, although landowners may leave more than the required minimum. We worked closely with the landowners and logging contractors to set specific guidelines for management activities to reduce variability in the application of treatments (harvest techniques and practices) among sites. Specifically, we requested that land managers adhere to the following practices:

1) Cut all non-merchantable timber (i.e., not suitable for the production of lumber, plywood, pulp or other forest products) in clearcut areas of the RMZ,

2) Locate wildlife reserve and green recruitment trees away from the RMZ,

3) Locate buffers needed to meet the requirement of a minimum of 50% of the Type Np Water buffered in the FP treatment contiguous with, and upstream of, the 500-ft (152.4-m) long stream buffer required upstream from the F/N break, and

4) No harvest downstream of the F/N break.

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. At each study site, we walked upstream along each tributary beginning at the F/N break. We identified Type Np and Ns Waters and locations of all sensitive sites according to Forest Practices rules. Type Np and Ns

Waters and other important features were located in the field using Trimble Global Positioning Systems (GPS), which were differentially corrected using Pathfinder Office software and integrated into a Geographic Information System (GIS; ArcMap). We created maps displaying the Type N Waters and locations of sensitive sites, channel heads, road crossings and other features (**Figures 3-2** through **3-16**), and shared these maps with landowners so that they could easily locate features.

3-2.3.1. F/N Break Identification

Application of all three types of buffer treatments began at the F/N break, which we established at each site with electrofishing surveys (see Chapter 2 – *Study Design*). Locations of F/N breaks were marked with flagging and an aluminum tag.

3-2.3.2. Stream Typing

Waters were typed during the summer low-flow period (typically August and September, but dependent on seasonal precipitation patterns) and identified as Type Np or Ns according to definitions outlined in Forest Practices rules (WAC 222-16-030). Type Np Waters are located downstream of the uppermost point of perennial flow (hereafter PIP, see 3-2.3.3 *Sensitive Site Identification*). Type Ns Waters are located upstream of Type Np Waters between the PIP and channel head (see 3-2.3.4 *Channel Head Identification*). Type Ns Waters are physically connected to other waters via a defined channel system with exposed mineral substrates. All Type Np and Ns Waters were systematically marked with flagging every 10 m (33 ft; slope distance) from the F/N break to each tributary channel head. Type Np Water length varied among sites, ranging from 325 m (1,066 ft) to 2,737 m (8,980 ft) and averaging 1,160 m (3,805 ft; **Table 3-2**).

3-2.3.3. Sensitive Site Identification

We identified sensitive sites during the process of stream typing and marked them with flagging and aluminum tags (for complete definitions of sensitive sites see WAC 222-16-101). We identified the following sensitive sites (**Table 3-3**):

1) **PIP (i.e., uppermost point of perennial flow):** The PIP, which includes both the headwater spring and headwall seep sensitive site categories, is at the point where a Type Np Water becomes a Type Ns Water. We identified PIP locations as the last pool of surface water greater in area than 10 cm^2, a criterion that could be unambiguously identified (Hunter *et al.* 2005). Type Np Waters can be spatially intermittent, especially during the low-flow period, so we continued our search for the PIP upslope to each tributary channel head. The number of PIPs at a study site ranged from 1 to 10 (1 to 9 headwater springs and 0 to 3 headwall seeps).

2) **Side-slope seeps:** Side-slope seeps are similar to headwall seeps with the exception that side-slope seeps are located downstream of the PIP. The number of side-slope seeps at a study site ranged from 0 to 11.

3) **Type Np intersections:** Type Np intersections occur where two or more Type Np Waters intersect. The number of Type Np intersections at a study site ranged from 0 to 9.

4) **Alluvial fans:** Alluvial fans are an erosional land form consisting of a cone-shaped deposit of water-borne, often coarse-sized sediments. We did not observe alluvial fans in any site.

3-2.3.4. Channel Head Identification

The channel head is the termination point of the tributary, where headwaters converge into a single channel, and is located at the uppermost extent of the channel system. At times the channel head and PIP coincide (e.g., when the PIP is a headwall seep); however, the channel head can also occur upslope of the PIP. We marked locations of channel heads with flagging.

Table 3-2. Characteristics of study sites included in the Type N Study. Unharvested areas for buffer treatment sites are the portion of the Type N basin that could not be harvested due to regulatory (N/F break location, stand age, unstable slopes) or logistic (different landowner) constraints (see 3-2.4 *Areas Within Buffer Treatment Sites Where Clearcuts Could not be Applied* for details). Type Np Water length, unharvested length for buffer treatment sites, and resulting length of riparian buffer (percentage of total stream length buffered for FP treatment sites in brackets). Dashes indicate instances where this information is not applicable (i.e., for references).

Treatment	**Block**	**Study site**		**Type Np Water**		
		Area (ha [ac])	**Unharvested Area (ha [ac]) / %**	**Length (m [ft])**	**Unharvested Length (m [ft]) / %**	**Riparian Buffer Length (m [ft])**
REF	OLYM	54 (133)	-	2,737 (8,980)	-	-
	WIL1	12 (29)	-	589 (1,932)	-	-
	WIL2 (1)	19 (48)	-	653 (2,142)	-	-
	WIL2 (2)	16 (41)	-	816 (2,677)	-	-
	WIL3	37 (92)	-	2,513 (8,245)	-	-
	CASC	50 (122)	-	1,080 (3,543)	-	-
100%	OLYM	28 (68)	3.7 (9.1) / 13%	1,949 (6,394)	269 (883) / 14%	1,680 (5,512)
	WIL1	31 (76)	0 (0)	1,029 (3,376)	0 (0)	1,029 (3,376)
	WIL2	26 (65)	2.8 (6.9) / 11%	1,257 (4,124)	179 (587) / 14%	1,078 (3,537)
	WIL3	23 (58)	0 (0)	1,359 (4,459)	0 (0)	1,339 (4,393)
FP	OLYM	17 (41)	0 (0)	1,070 (3,510)	0 (0)	663 (2,175; 62%)
	WIL1	15 (37)	0 (0)	325 (1,066)	0 (0)	236 (774; 73%)
	CASC	26 (64)	0 (0)	822 (2,697)	0 (0)	456 (1,496; 55%)
0%	OLYM	13 (32)	0 (0)	637 (2,090)	0 (0)	0 (0)
	WIL1	28 (69)	0 (0)	1,525 (5,003)	0 (0)	0 (0)
	WIL2	17 (42)	0.4 (1.0) / 2%	933 (3,061)	82 (269) / 9%	0 (0)
	CASC	14 (36)	2.1 (5.2) / 15%	420 (1,378)	90 (295) / 21%	0 (0)

Table 3-3. The number of sensitive sites by type for Type N Study sites in the first pre-harvest sample year (2006). HW seep = headwall seep, SS seep = side-slope seep, TJ = Type Np intersection, HW Spring = headwater spring.

Treatment	Block	Sensitive Sites			
		HW Seep	SS Seep	TJ	HW Spring
REF	OLYM	0	2	8	9
	WIL1	2	1	2	1
	WIL2 (1)	0	0	1	2
	WIL2 (2)	2	10	4	3
	WIL3	0	5	2	3
	CASC	0	0	2	3
100%	OLYM	3	6	9	7
	WIL1	0	3	4	5
	WIL2	0	2	6	7
	WIL3	2	6	7	6
FP	OLYM	0	3	5	6
	WIL1	0	2	0	1
	CASC	0	2	2	3
0%	OLYM	0	1	3	4
	WIL1	1	3	4	4
	WIL2	0	11	2	3
	CASC	0	0	0	1

3-2.4. AREAS WITHIN BUFFER TREATMENT SITES WHERE CLEARCUTS COULD NOT BE APPLIED

We intended to apply clearcut harvests to the entire Type N basin, from the F/N break upstream and including all lands draining the Type Np and Ns Waters. However, regulatory and logistic constraints prevented application of clearcut harvest to the entire site in two 100% and two 0% treatment study sites (**Table 3-2**):

1) **F/N break location** (regulatory constraint): Two types of F/N breaks exist: terminal and lateral. Terminal F/N breaks are defined as those where the last fish occurs within a Type F Water (**Figure 3-1a**) or at the confluence of two Type N Waters (**Figure 3-1b**). Lateral F/N breaks are defined as those that occur where a Type N Water laterally intersects a Type F Water (**Figure 3-1c**). Riparian buffers are required within the RMZs of Type F Waters (WAC 222-30-021 (1)); therefore, when F/N breaks are lateral they are located in the buffered RMZ of the Type F Water. Likewise, F/N breaks can be terminal but still located within the RMZ of Type F Waters. Locations of F/N breaks were within the Type F Water RMZ in two study sites (WIL2-0% (**Figure 3-12**), terminal F/N break, and CASC-0% (**Figure 3-16**), lateral F/N break), resulting in untreated areas ranging from 0.1 to 2.1 ha (0.3 to 5.2 ac) at the downstream ends of the study sites.

2) **Different landowner** (logistic constraint): Fruit Growers Supply Company was unable to harvest the 2.2 ha (5.4 ac) of the 100% treatment in the Olympic block falling on their ownership, from the F/N break and upstream for 244 m (801 ft) to just above the road crossing (**Figure 3-4**).

3) **Stand age** (regulatory constraint): WADNR, who owns the majority (25.3 ha [62.5 ac]) of the 100% in the Olympic block, was unable to harvest 1.5 ha (3.7 ac) of the Type N study site, including one headwall seep, because the stand age was over 50 years old (**Figure 3-4**). This study site is located in the Olympic Experimental State Forest (OESF) where a lawsuit settlement agreement limits WADNR from cutting timber over 50 years old until a review in 2014.

4) **Unstable slopes** (regulatory constraint): Unstable slopes directly upstream of the F/N break at the 100% treatment in the Willapa 2 block could not be harvested under Forest Practices rules without special review for compliance with State Environmental Policy Act (SEPA) and SEPA guidelines (WFPB 2001). As a result, harvest was not implemented in 2.8 ha (6.9 ac), including both sides of the Type Np Water from the F/N break and upstream for 179 m (587 ft) and another 201 m (659 ft) of Type Np Water on the west side of the stream (**Figure 3-11**).

In summary, 0.1 to 3.7 ha (0.3 to 9.1 ac) of five study sites (or less than 1% to 15% of the total Type N basin area) were not harvested due to regulatory and logistic constraints. The resulting length of stream that was not adjacent to harvest ranged from 82 to 269 m (269 to 883 ft), or 9% to 21% of total Type Np Waters length (**Table 3-2**).

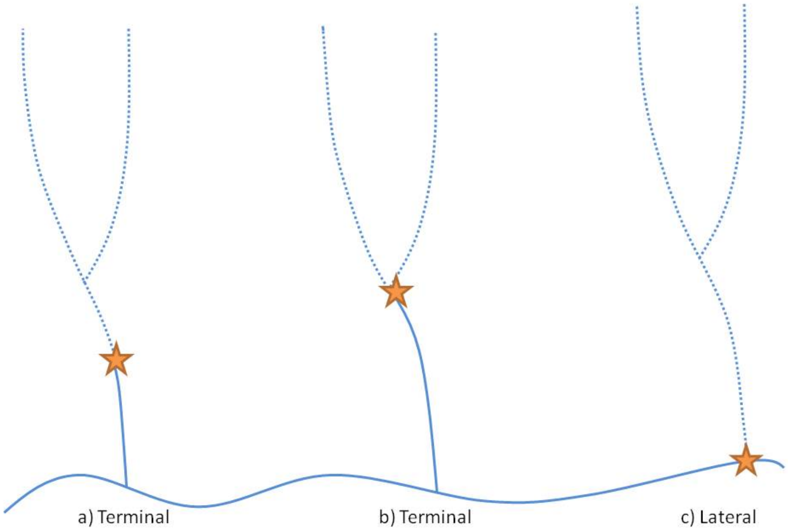

Figure 3-1. Types of F/N breaks: terminal break where the last fish occurs within a Type F Water (a), terminal break where the last fish occurs below the confluence of two Type N Waters (b), and lateral break where a Type N Water laterally intersects a Type F Water (c). Solid blue lines are Type F Waters, dashed blue lines are Type N Waters, and stars indicate F/N breaks.

3-2.5. RIPARIAN BUFFER CONFIGURATION

Timber management activities under Forest Practices rules in Type N basins can be complex given the layout of streams, the number and location of sensitive sites and harvesting logistics. While it is relatively easy to apply Forest and Fish rules in theory, in practice they can result in complicated harvest configurations. For example, buffers are required both along the stream immediately upstream of the F/N break (hereafter, stream buffer) and centered on sensitive sites (hereafter, sensitive site buffer). When features are far apart spatially, the resulting buffer configuration may include stand-alone buffers along sensitive sites. Alternatively, sensitive sites can occur within areas already protected by the stream buffer, resulting in contiguous buffers that may include multiple sensitive sites.

As intended, the buffered length of the RMZ varied by treatment (**Table 3-2**). Similarly, the number of buffered sensitive sites depended on both the treatment and the number of sensitive sites present at a site (**Table 3-3**).

3-2.5.1. 100% Treatment Sites

We identified 26 Type Np intersections, 17 side-slope seeps, 5 headwall seeps and 25 headwater springs across all 100% treatment sites. These sensitive sites were protected by riparian buffers applied according to Forest Practices rules, with one exception: no harvest was applied adjacent to one headwall seep in the 100% buffer treatment in the Olympic block due to regulatory constraints preventing harvest of the surrounding timber (stand age greater than 50 years). Protection of unstable slopes resulted in wider riparian buffers along some portions of the two 100% buffer treatment sites in the Olympic and Willapa 2 blocks (**Figures 3-4** and **3-11**, respectively), although it should be noted that we designed this study to evaluate buffer length, not buffer width (see Chapter 2 – *Study Design*). Further, our intent was to evaluate the effectiveness of the regulatory riparian buffers required by Forest Practices rules in context of the application of those rules broadly across the western Washington landscape, including for sites requiring buffers on unstable slopes. As such, statistical analyses do not directly address the implications of buffers wider than 50 ft (15.2 m).

3-2.5.2. FP Treatment Sites

Several sensitive sites in FP treatments located in the Olympic and South Cascades blocks (**Figures 3-5** and **3-16**, respectively) did not receive stand-alone buffers but rather formed a contiguous buffer with the stream buffer. The features included in these contiguous buffers included five tributary junctions, three seeps, and two PIPs in the FP treatment in the Olympic block, and two tributary junctions, two seeps and one PIP at the FP treatment in the South Cascade block. In fact, among the three FP treatment sites only 2 of 7 side-slope seeps and 7 of 10 headwater springs were protected with stand-alone buffers, respectively; and the two buffered side-slope seeps were encompassed in a single buffer at one site (WIL1-FP, **Figure 3-7**).

3-2.5.3. 0% Treatment Sites

In the 0% treatment sites, sensitive sites were not protected by a riparian buffer. This included nine Type Np intersections, 15 side-slope seeps, one headwall seep, and 12 headwater springs.

3-2.6. FOREST PRACTICES APPLICATIONS

We established one primary contact person representing the landowner at each study site (typically a forester) and worked with them to establish harvest boundary layouts on the ground. We provided maps of study sites that identified the F/N break, streams, and sensitive sites and indicated the desired locations of riparian buffers. We either assisted directly with harvest boundary layout or walked boundaries after they had been established in the field to ensure that treatment specifications were met.

A state permit to harvest timber in Washington State requires a Forest Practice Application (FPA). Once we worked with landowner contacts to develop harvest boundaries, they developed an FPA that we reviewed to ensure that plans met our specifications. Once submitted, foresters informed us of the FPA number and when the FPA was approved. All FPAs were approved as submitted, that is, no conditions were placed on any approved FPA associated with the study.

We maintained communication with the primary contact person for each study site so that we knew when harvest was expected to begin. We held pre-harvest meetings with contractors who were managing timber harvest activities at each study site to explain the study and research questions and emphasize the importance of following timber boundaries as established. Once harvest was ready to begin, we met on-site with timber harvesters to explain again the research and stress the importance of sticking with the plan as outlined. In addition, we worked with foresters to visit sites throughout the timber harvest period to view harvest progress.

3-2.7. REQUIRED FOREST PRACTICES EXEMPTIONS

This study explores the effectiveness of alternative riparian buffers within RMZs of Type N Waters that differ from those allowed under current Forest Practices rules. We were required to obtain exemptions from both Forest Practices rules and the WADNR HCP in order to apply buffer treatments in nine of 11 study sites (**Table 3-4**).

3-2.7.1. Exemption from Forest Practices Rules

A two-sided 50-ft (15.2-m) buffer protects at least 50% of the Type Np Water length under Forest Practices rules. Clearcutting the full length of the RMZ at 0% treatment sites required exemption from Forest Practices rules. To obtain the exemption we utilized pilot rule making through the CR-101 Preproposal Statement of Inquiry process for experimental research treatments (Revised Code of Washington (RCW) 34.05.310). The Forest Practices Board granted Exemption on 15 February 2007.

3-2.7.2. Exemption from WADNR Habitat Conservation Plan

The 10 WADNR-managed study sites are managed under the WADNR HCP (WADNR 1997). Seven of these 10 sites were buffer treatments (two 0%, two FP and three 100% treatments). Exemptions from the HCP riparian conservation strategy, and leave tree and talus slope requirements were necessary to harvest these seven sites according to treatment specifications. USFWS granted exemptions from the WADNR HCP on 17 January 2007.

3-2.7.2.a. Exemption from WADNR HCP riparian conservation strategy

A two-sided 100-ft (30.5-m) riparian buffer is required on Type Np Waters under the WADNR HCP. We obtained exemption from the WADNR HCP riparian buffer strategy for all seven buffer treatment sites located on WADNR-managed forestlands to allow for either two-sided 50-ft (15.2-m) riparian buffers (for 100% and FP treatments) or no riparian buffers (for 0% treatments) along Type Np Waters. Exemption was also required to allow for riparian buffer lengths that did not contain the entire Type Np Water length for the FP and 0% treatment sites.

3-2.7.2.b. Exemption from WADNR HCP leave tree requirement

The WADNR HCP requires retention of at least three snags and five live trees for each acre harvested, on average. Forest Practices rules for western Washington (WAC 222-30-020 (11b)) require leaving three wildlife reserve trees, two green recruitment trees, and two downed logs for each acre harvested. However, under Forest Practices rules, wildlife reserve tree and green recruitment tree retention areas can include RMZs. In other words, the trees remaining in the RMZ as part of the required two-sided 50-ft (15.2-m) buffer count towards the Washington State Forest Practices leave tree requirement. Since the WADNR HCP and the Forest Practices rules are different, we obtained exemption from the eight-tree-per-acre leave tree requirement under the WADNR HCP for all seven harvested study sites located on WADNR-managed forestlands.

3-2.7.2.c. Exemption from WADNR HCP talus slope requirement

Under the WADNR HCP, WADNR provides protection for talus fields. One WADNR-managed study site (CASC-0%; **Figure 3-16**) included a talus slope. The WADNR HCP specifies that a 100-ft (30.5-m) wide timber buffer will be maintained around talus fields. Forest Practices rules do not specify protections for talus slopes so we obtained an exemption. No harvest occurred within the talus slope itself, but a no-harvest buffer was not retained around the talus.

Table 3-4. Exemptions from Forest Practices rules and the WADNR HCP necessary to apply buffer treatments, by treatment and block.

				WADNR HCP Rules		
Treatment	**Block**	**Landowner**	**FP Rules RMZ Buffer**	**RMZ Buffer**	**Leave Tree**	**Talus Slope**
0%	OLYM	Private	X			
	WIL1	Private	X			
	WIL2	State	X	X	X	
	CASC	State	X	X	X	X
FP	OLYM	Private				
	WIL1	State		X	X	
	CASC	State		X	X	
100%	OLYM	State		X	X	
	WIL1	State		X	X	
	WIL2	Private				
	WIL3	State		X	X	

3-2.8. TREATMENT INCONSISTENCIES

Harvest application in the 11 buffer treatment sites went as expected with one minor exception. During harvest of the 100% treatment in the Willapa 1 block (**Figure 3-7**) in October 2008, 15 small trees (including nine western hemlock (*Tsuga heterophylla*), two western redcedar and four red alder (*Alnus rubra*), all less than 30 cm (12 in) diameter at breast height) were felled within the riparian buffer. The trees were located approximately 45 m (148 ft) upstream of the F/N break and just above the road crossing on the mainstem tributary. This oversight was a function of operator error; the trees were within the no harvest riparian buffer area and one tree even had a harvest boundary tag on it. A WADNR Forest Practices Forester and a Type N Study field team independently visited the site and neither one saw any direct impacts to the stream. We observed no evidence of sediment input and no trees felled into or over the stream. Fortuitously, instream water sampling equipment was located above this point.

3-3. ROADS

The construction and use of forest roads can be a major source of sediment in forested basins (Reid and Dunne 1984; Ketcheson and Megahan 1996). Sediment that reaches streams can affect water quality, fish, and other animal communities (Waters 1995). The primary source of sedimentation from roads is via road surface erosion, which occurs on all roads (Dubé *et al.* 2004). Since road crossing locations can be an entry point for road sediment into the stream channel, and sediment input into streams was a response variable of interest, we documented the number of stream road crossings in all study sites. In the pre-harvest period, we verified the number of stream road crossings in each site July through October 2006. The number of stream road crossings of Type N Waters per site during the pre-harvest period ranged from 0 to 4 (**Table 3-5**). Landowners did construct new roads to facilitate harvest in some treatment sites; however, this road construction did not include any crossings of Type N Waters. We discuss road use and construction in Chapter 10 – *Sediment Processes,* along with estimates of road surface contributions to instream sediment in both the pre- and post-harvest periods.

A culvert identified as a blockage to fish passage was replaced at one site (WIL1-100%) in September and October 2006, during the pre-harvest period. Prior to replacement, the F/N break was located approximately 28 m (92 ft) downstream of the culvert; however, the end of fish habitat (potential habitat likely to be used by fish, which could be recovered by restoration or management; WAC 222-16-010) had been field-identified approximately 90 m (295 ft) upstream of the culvert at a 15-m (49-ft) long, 35% (19 degree) gradient cascade presumed to prevent upstream fish migration. During the second year of pre-harvest sampling in May 2007, we observed fish approximately 40 m (131 ft) upstream of the new culvert.

Table 3-5. Number of road crossings over Type N Waters by study site.

Block	Treatment	Road Crossings
OLYM	REF	4
	100%	2
	FP	0
	0%	3
WIL1	REF	1
	100%	2
	FP	0
	0%	2
WIL2	REF1	2
	REF2	0
	100%	2
	0%	1
WIL3	REF	4
	100%	2
CASC	REF	0
	FP	0
	0%	0

3-4. POST-HARVEST MANAGEMENT ACTIVITIES

The objective of the study was to evaluate the effectiveness of alternative riparian buffers where all Forest Practices rules were followed, with the exception of those related to the RMZ. Under Forest Practices rules, a wide range of practices can occur after harvest, so we documented post-harvest management activities, such as herbicide applications and reforestation. Tracking how study sites were treated in the post-harvest period may prove useful if this study continues.

3-4.1. SLASH REMOVAL

3-4.1.1. Upland Slash Removal

Slash is defined under Forest Practices rules as "pieces of woody material containing more than 3 cubic feet resulting from forest practice activities" (WAC 222-16-10). Generally, slash includes large organic material in the form of tree boles and branches that have little commercial value. Some slash is mechanically gathered from uplands after harvest and sold for pulp or biofuel, or burned in slash piles. Mechanical gathering and piling of slash occurred in seven of 11 harvested sites (**Table 3-6**). Slash piles at five sites were subsequently burned. At one site, a contractor removed piles near roads, while piles away from roads were burned. Slash burning occurred in the fall months (September through December) in 2009, 2010, and 2011. Large pieces of slash from piles at one site were sold for pulp making.

3-4.1.2. Instream Slash Removal

Current Forest Practices rules were designed to minimize the amount of woody material that enters the Type N Waters during harvest, including:

1) *Felling and bucking* (WAC 222-30-050): (1) **Falling along water.** (c) Trees may be felled into Type Np Water if logs are removed as soon thereafter as practical and (2) **Bucking or limbing along water**. No bucking or limbing shall be done on trees or portions thereof lying within the bankfull width of Type S, F, or Np Waters, in RMZ core zones, in sensitive sites, or in open water areas of Type A Wetlands, and

2) Forest Practices Board Manual Section 4, *Guidelines for Clearing Slash and Debris from Type Np and Ns Waters*, which includes "…current forest practice rules prohibit the machine piling of slash and debris within 30 ft (9.1 m) of unbuffered stream banks."

When a relatively large amount of slash enters the stream landowners may choose, or be required, to remove it. Not surprisingly, variable amounts of slash existed in our study streams after treatment applications, and total instream slash was correlated with treatment type (see Chapter 6 – *Wood Recruitment and Loading)*. Post-harvest removal of slash in streams occurred at only one study site (CASC-0%). As a result, this particular site had the least amount of instream slash compared to the other 0% treatment sites.

3-4.2. HERBICIDE APPLICATION

Clearcut harvest can result in the recruitment and growth of unwanted vegetation that inhibits the growth of commercial tree seedlings. The control of competing vegetation to allow establishment, survival, and growth of commercial tree species is required under Forest Practices rules (WAC 222-34-010 (3)). A variety of manual, mechanical and chemical methods are available and land management objectives and cost are among the factors landowners consider when choosing the control methods to apply, if any. Herbicides, such as glyphosate, are commonly used to control undesirable vegetation in forestry applications. Amphibian mortality has been linked to products containing glyphosate and surfactants (Relyea 2005); however, the impact may depend on herbicide concentration and the timing of the application (Jones *et al.* 2010). Herbicide was applied in four of 11 harvested sites August 2008 through May 2012 (**Table 3-6**) following the *Forest Chemicals* chapter of the Forest Practices rules (WAC 222-38) which specify that pesticides cannot be applied within 50 ft (15.2 m) of Type Np and Ns Waters with surface water present. In each of the four sites where chemical herbicides were used, glyphosate was included as one of the several herbicides applied, and in all cases a surfactant was used to improve the performance of the herbicide in spray solution by increasing coverage, spray retention, and absorption. We tracked herbicidal application, including products used, concentrations, and timing at the four sites (**Table 3-6**).

3-4.3. REFORESTATION

Reforestation is the natural or planned replanting of trees removed during timber harvest activities. According to Forest Practices rules (WAC 222-34) reforestation is required in clearcut areas that are not being converted to another use, or where conversion is unlikely. Acceptable stocking levels are typically defined as a minimum of 190 well-distributed, vigorous, undamaged seedlings per acre of commercial tree species that have survived for at least one growing season. Land managers for all study sites opted for artificial regeneration (as opposed to natural regeneration). Reforestation was by hand and occurred in study sites April 2009–April 2010 (**Table 3-6**). Restocking levels ranged from 300 to 500 trees per acre. Four study sites were replanted exclusively with Douglas-fir (*Pseudotsuga menziesii*), while others were composed of a mix of conifers including Douglas-fir, as well as one or more of western redcedar, western hemlock, and noble (*Abies procera*), Pacific silver (*A. amabilis*), and grand (*A. grandis*) firs. Information about stocking levels and species was not available for two study sites. Reforestation generally occurred over a single period lasting from one to four months. However, reforestation in the Willapa 1 FP and 100% treatment sites occurred in two discreet periods. The landowner replanted most of these two sites in April 2009; however, a small area (northernmost portions of the 100% and FP treatments and a small strip on the west edge of the FP treatment) was not replanted until Jan–Apr 2010. The majority of the Willapa 3 100% treatment site was replanted in March 2009; however, the northwest corner of the site (~2 ha [5 ac]) was not replanted until Jan–Mar 2010. For sites where stocking information was available, Douglas-firs were 1+1 seedlings and all other species' were P+1 seedlings[1].

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[1] A 1+1 seedling is grown from seed for one year, lifted at the end of the first growing season, and transplanted back into nursery beds and grown for one more year. A P+1 (i.e., Plug+1) seedling is grown in a greenhouse or shelter-house for nine to 12 months and then transplanted into a bareroot nursery for a year. The latter method of seedling husbandry is commonly used for species that are harder to cultivate.

Table 3-6. Post-harvest management activities in buffer treatment sites included in the Type N Study, 2008–2010. ABAM = Pacific silver fir (*Abies amabilis*), ABGR = grand fir (*A. grandis*), ABPR = noble fir (*A. procera*), PSME = Douglas-fir (*Pseudotsuga menziesii*), THPL = western redcedar (*Thuja plicata*), and TSHE = western hemlock (*Tsuga heterophylla*).

		Herbicide Application		**Slash Removal**		**Reforestation**		
Block	**Treatment**	**Date / Method**	**Product**	**Upland**	**Instream**	**Date**	**Stocking Level (TPA)**	**Stocking Species (TPA)**
OLYM	100%	August 2010: ground application	Accord (Glyphosate) Chopper (Imazapyr) Oust Extra (with Polaris SP Sulfomet Extra SYL-TAC (surfactant)	None	None	March 2011	414	PSME (126) TSHE (149) THPL (139)
	FP	None		November 2011: slash pile burning on landings	None	January 2010	455	PSME
	0%	None		November 2011: slash pile burning on landings	None	January 2010	455	PSME
WIL1	100%	None		October 2010: slash pile burning on landings	None	April 2009; March 2010	300	PSME
	FP	None		October 2010: slash pile burning on landings	None	February 2009; March 2010	300	PSME
	0%	August 2008: aerially by helicopter (18 ac only)	Razor Pro (Glyphosate) SFM+MSM Epro (with: SYL-TAC (surfactant) EDT-concentrate)	October–December 2009: slash piles near road removed by contractor, piles away from road were burned	None	January–March 2010	proprietary	proprietary

Table 3-6. (continued)

		Herbicide Application		Slash Removal		Reforestation		
Block	**Treatment**	**Date / Method**	**Product**	**Upland**	**Instream**	**Date**	**Stocking Level (TPA)**	**Stocking Species (TPA)**
WIL2	100%	None		October–December 2009: slash pile burning	None	January–March 2010	proprietary	proprietary
	0%	None		September–October 2009: mechanized removal of large pieces for fiber	None	January 2009	375	PSME (300) THPL (75)
WIL3	100%	None		None	None	March 2009; January–March 2010	412	PSME (300) ABGR (10) ABPR (14) ABAM (21) TSHE (42) THPL (25)
CASC	FP	July 2009 : aerially by helicopter May 2012: aerially by helicopter	July 2009: Accord (Glyphosate), Chopper Gen 2 (Imazapyr), Oust Extra MSO (surfactant) May 2012: Transline (Clopyralid), Spyder (Sulfometuron Methyl)	None	None	January–April 2010	400–500	above the L1500 road: PSME (250) ABPR (250); middle 1/3 of the unit: PSME (400); lowest 20 ac: PSME (350) THPL (50)

Table 3-6. (continued)

		Herbicide Application		Slash Removal		Reforestation		
Block	**Treatment**	**Date / Method**	**Product**	**Upland**	**Instream**	**Date**	**Stocking Level (TPA)**	**Stocking Species (TPA)**
CASC	0%	July 2009: aerially by helicopter May 2012: aerially by helicopter	July 2009: Accord (Glyphosate), Chopper Gen 2 (Imazapyr), Oust Extra MSO (surfactant) May 2012: Oust Extra, Transline	None	May 2009: by hand	January–April 2010	400–500	above the L1500 road: PSME (250) ABPR (250); middle 1/3 of the unit: PSME (400); lowest 20 ac: PSME (350) THPL (50)

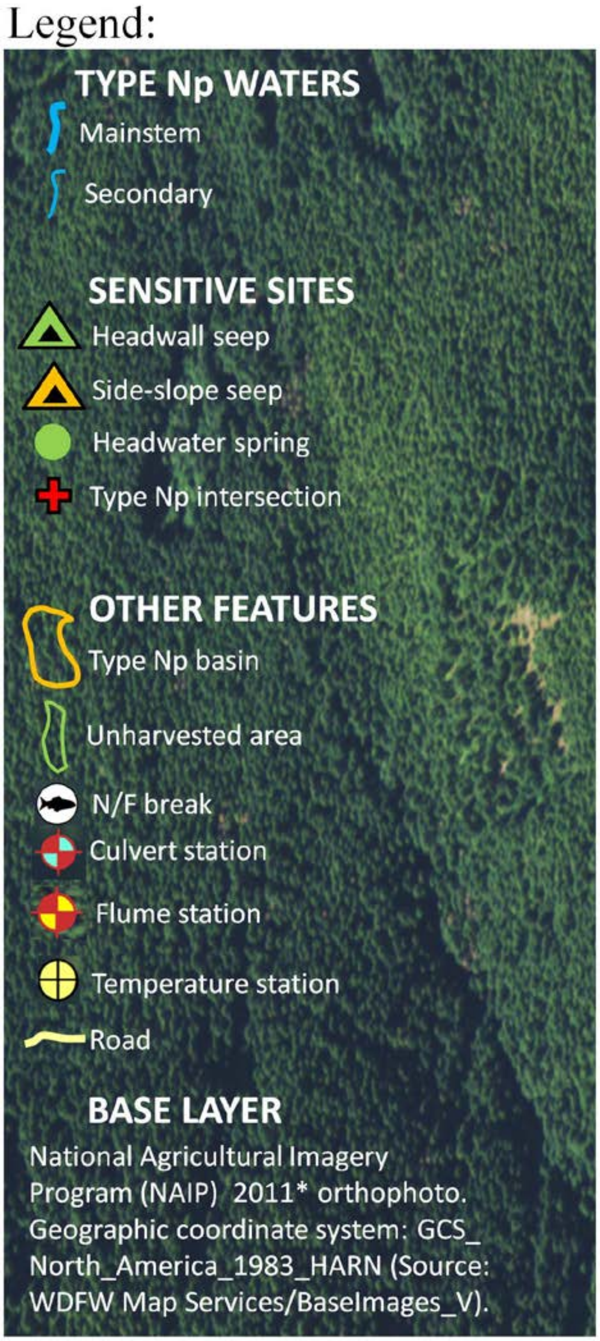

Figure 3-2. Study basin map legend. *Note: the Olympic block 100% treatment (OLYM-100%) map displays the NAIP 2009 orthophoto as its base layer.

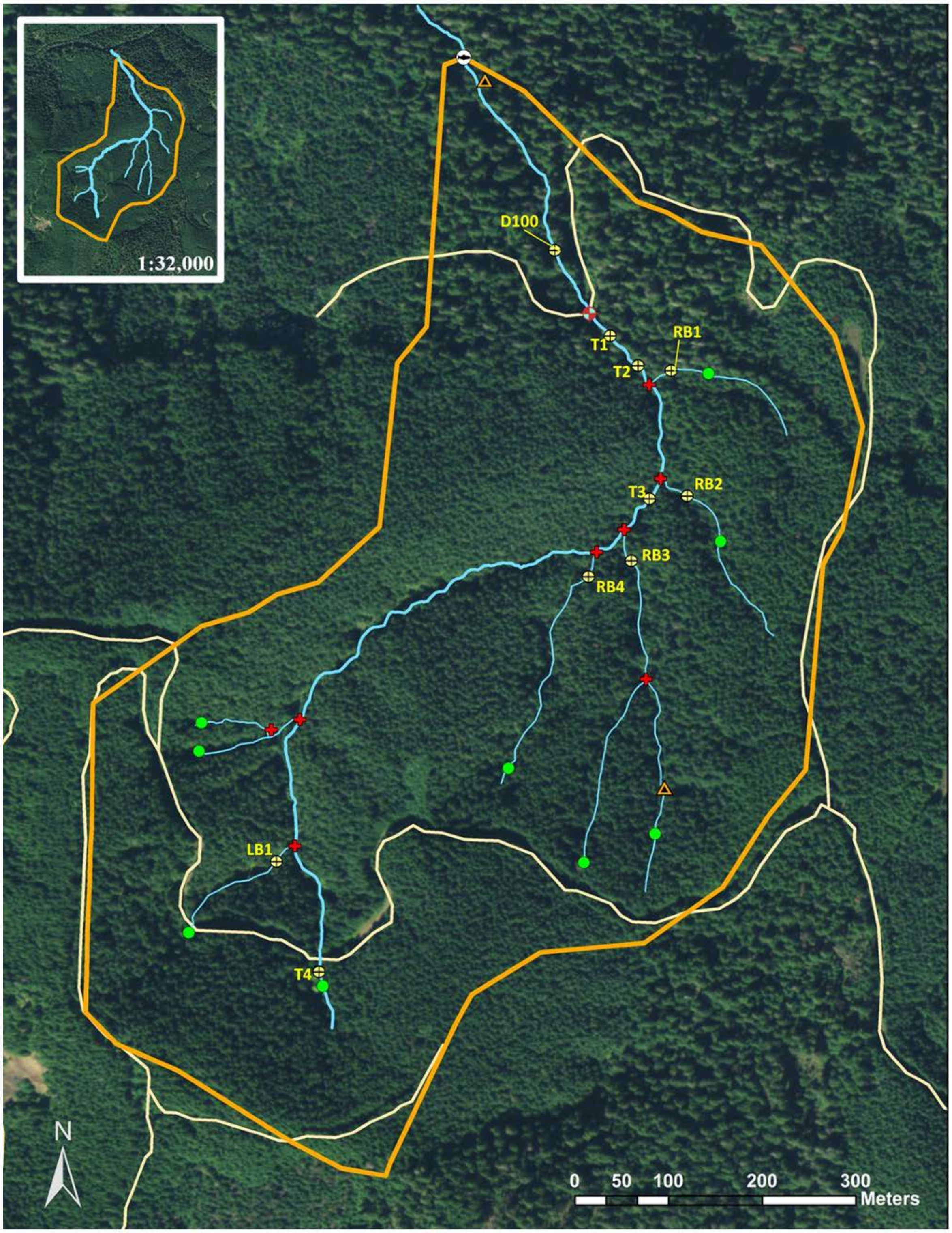

Figure 3-3. Type N Waters, sensitive sites and other features in the Olympic block reference (OLYM-REF).

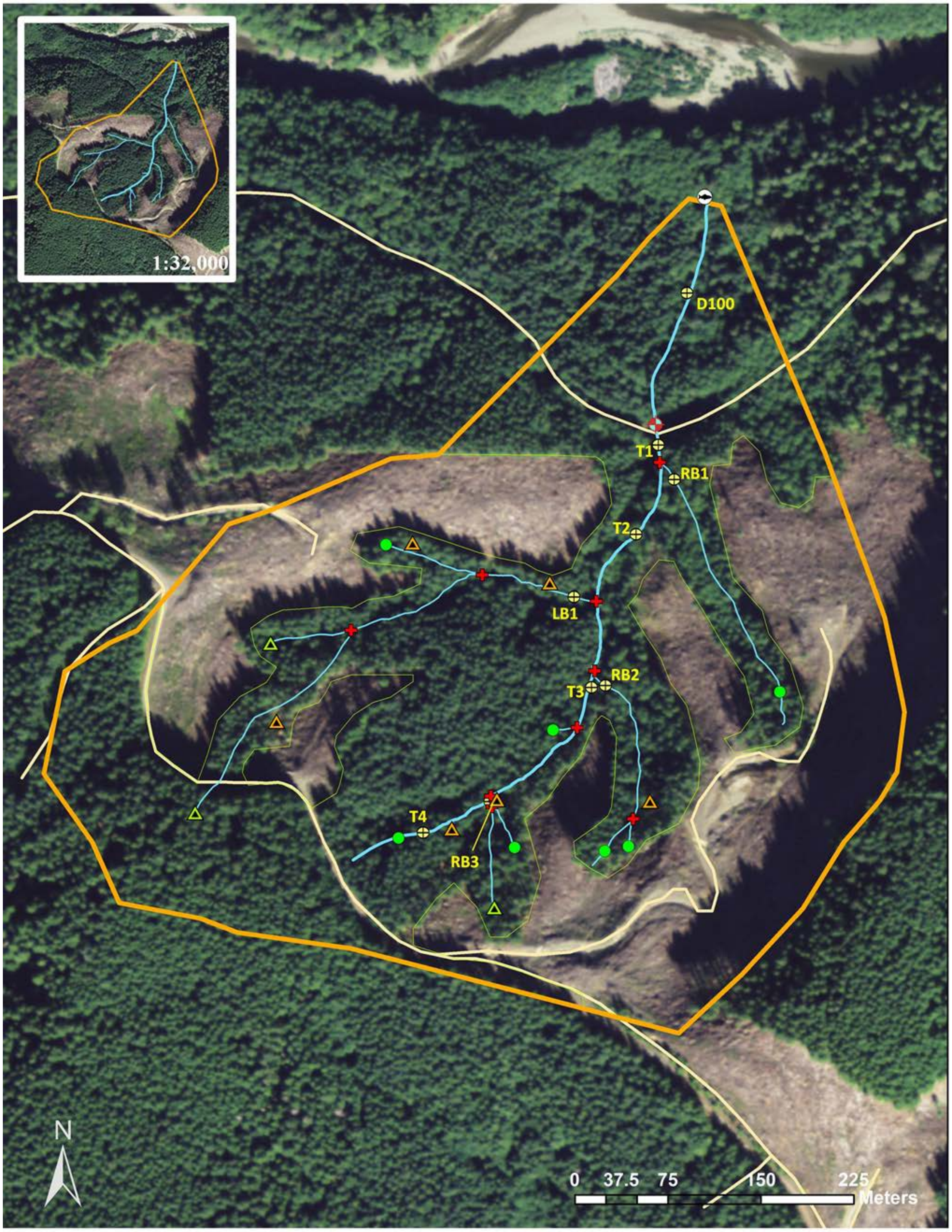

Figure 3-4. Type N Waters, sensitive sites and other features in the Olympic block 100% treatment (OLYM-100%). Base layer is the NAIP 2009 orthophoto.

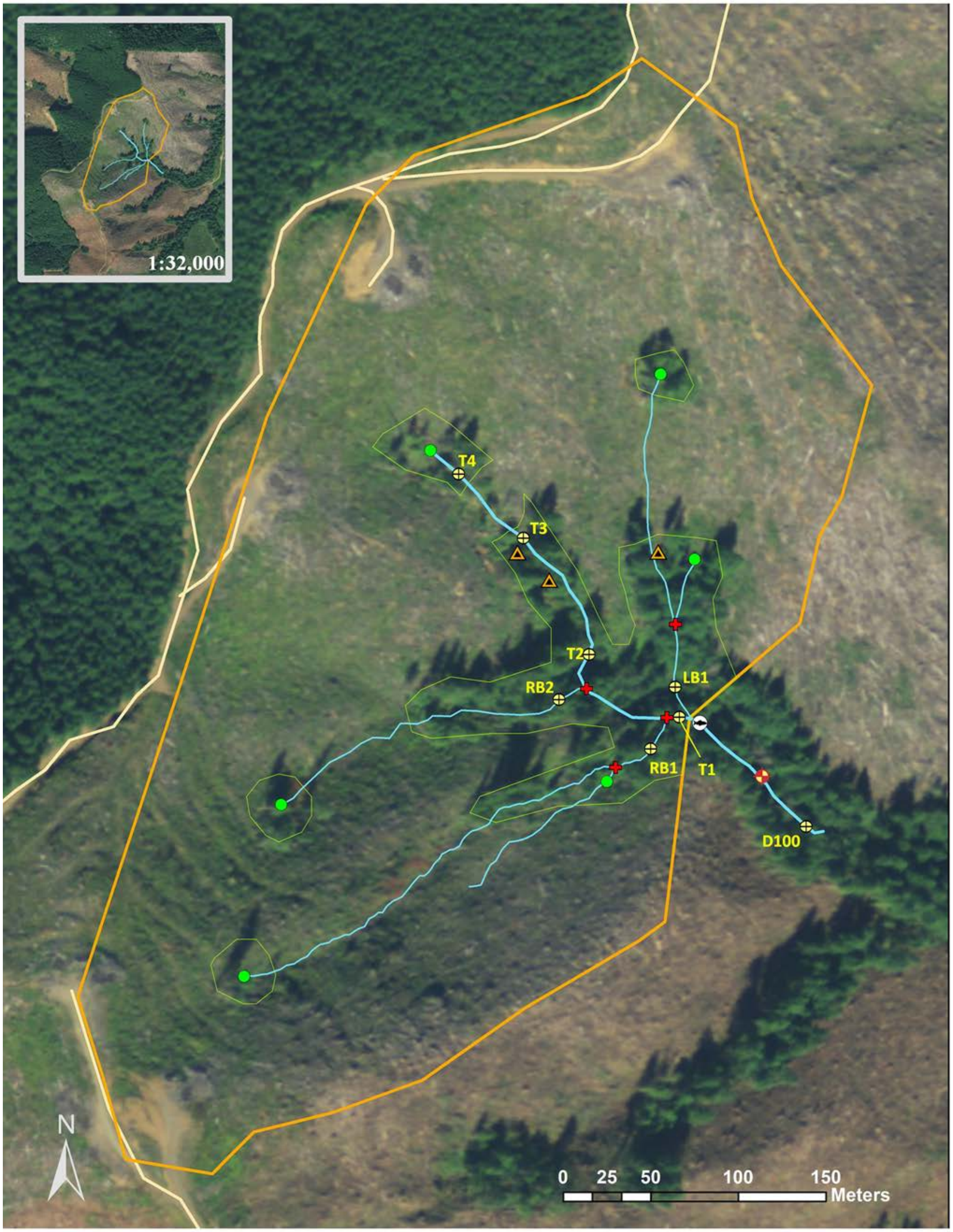

Figure 3-5. Type N Waters, sensitive sites and other features in the Olympic block FP treatment (OLYM-FP).

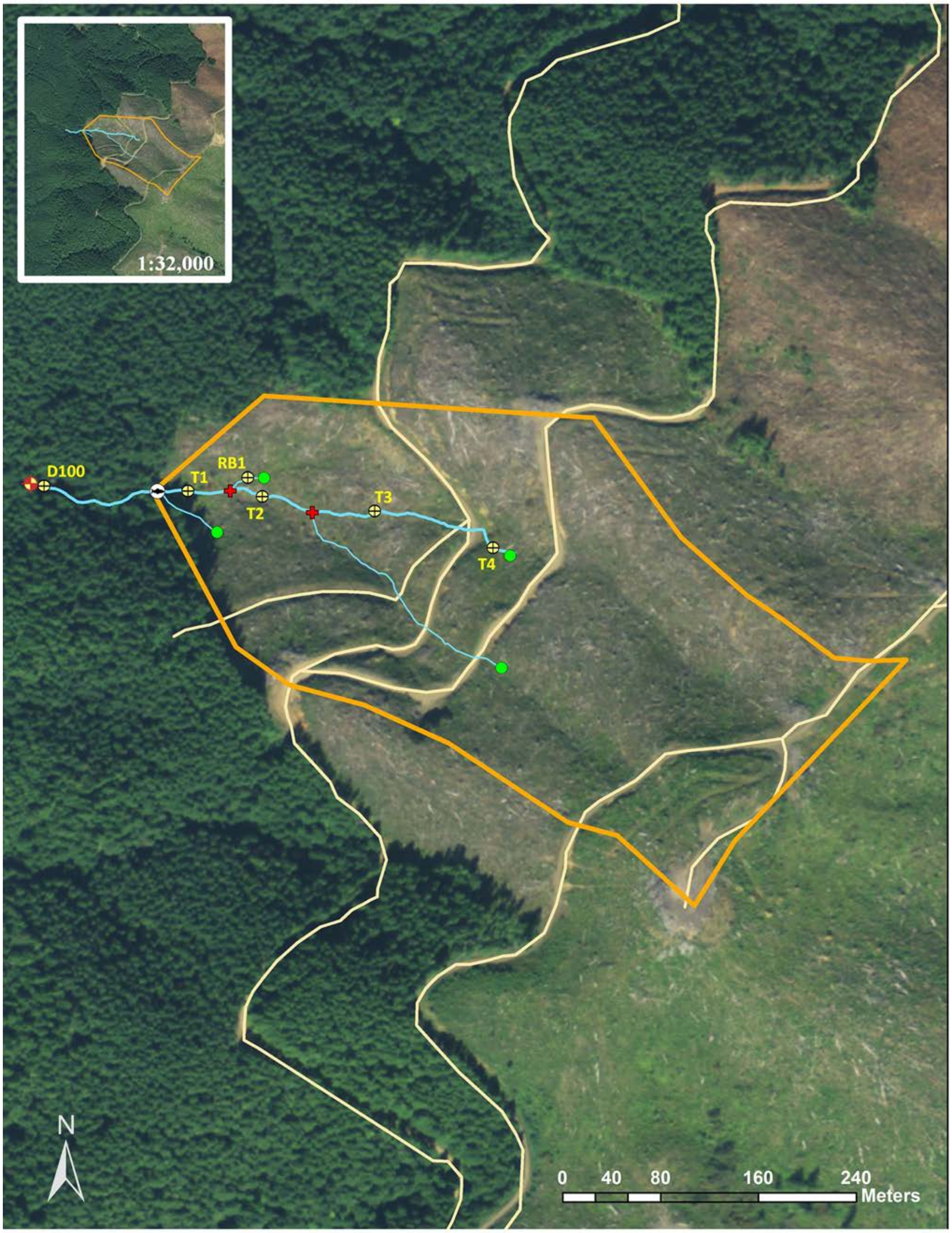

Figure 3-6. Type N Waters, sensitive sites and other features in the Olympic block 0% treatment (OLYM-0%).

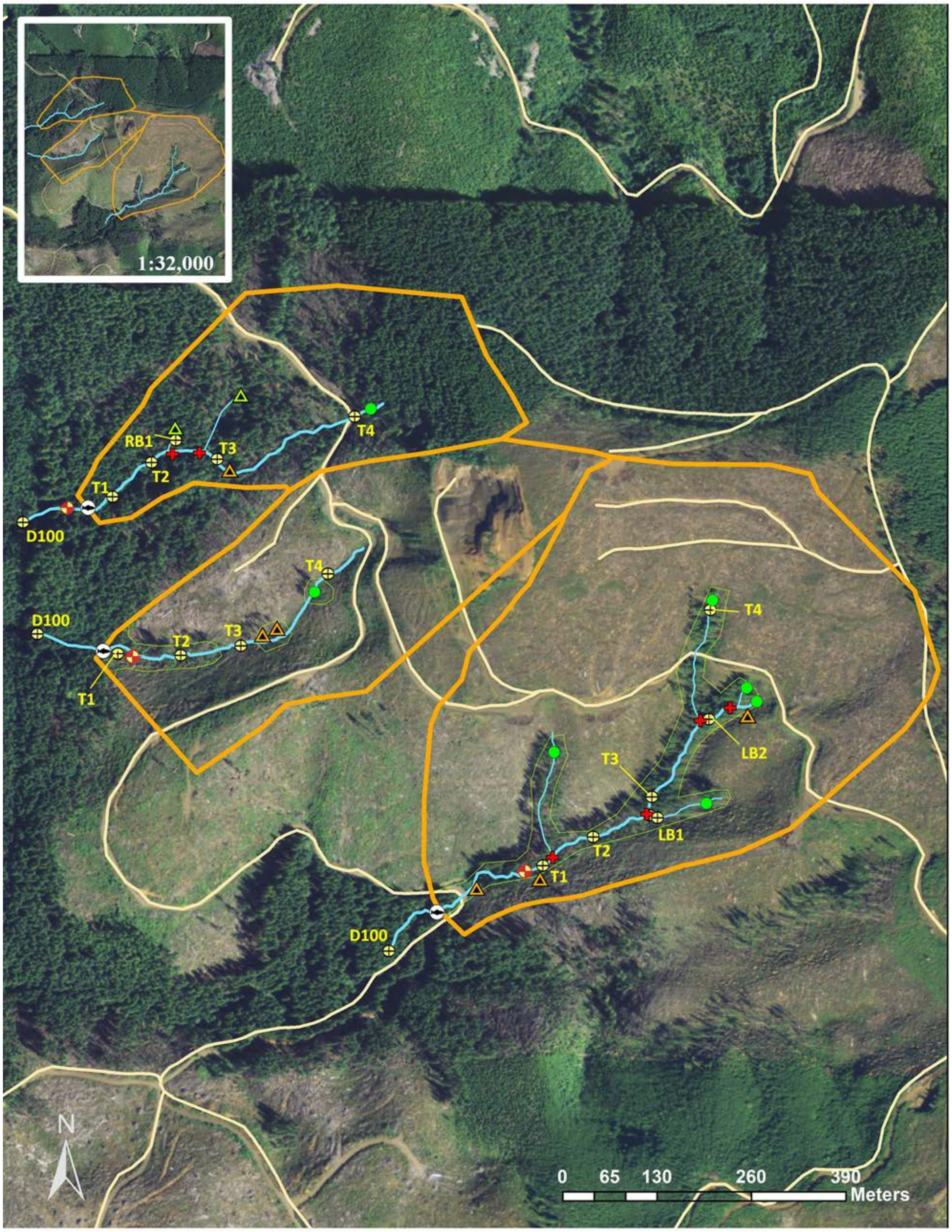

Figure 3-7. Type N Waters, sensitive sites and other features in the Willapa 1 block (from north to south): reference (WIL1-REF), FP treatment (WIL1-FP) and 100% treatment (WIL1-100%).

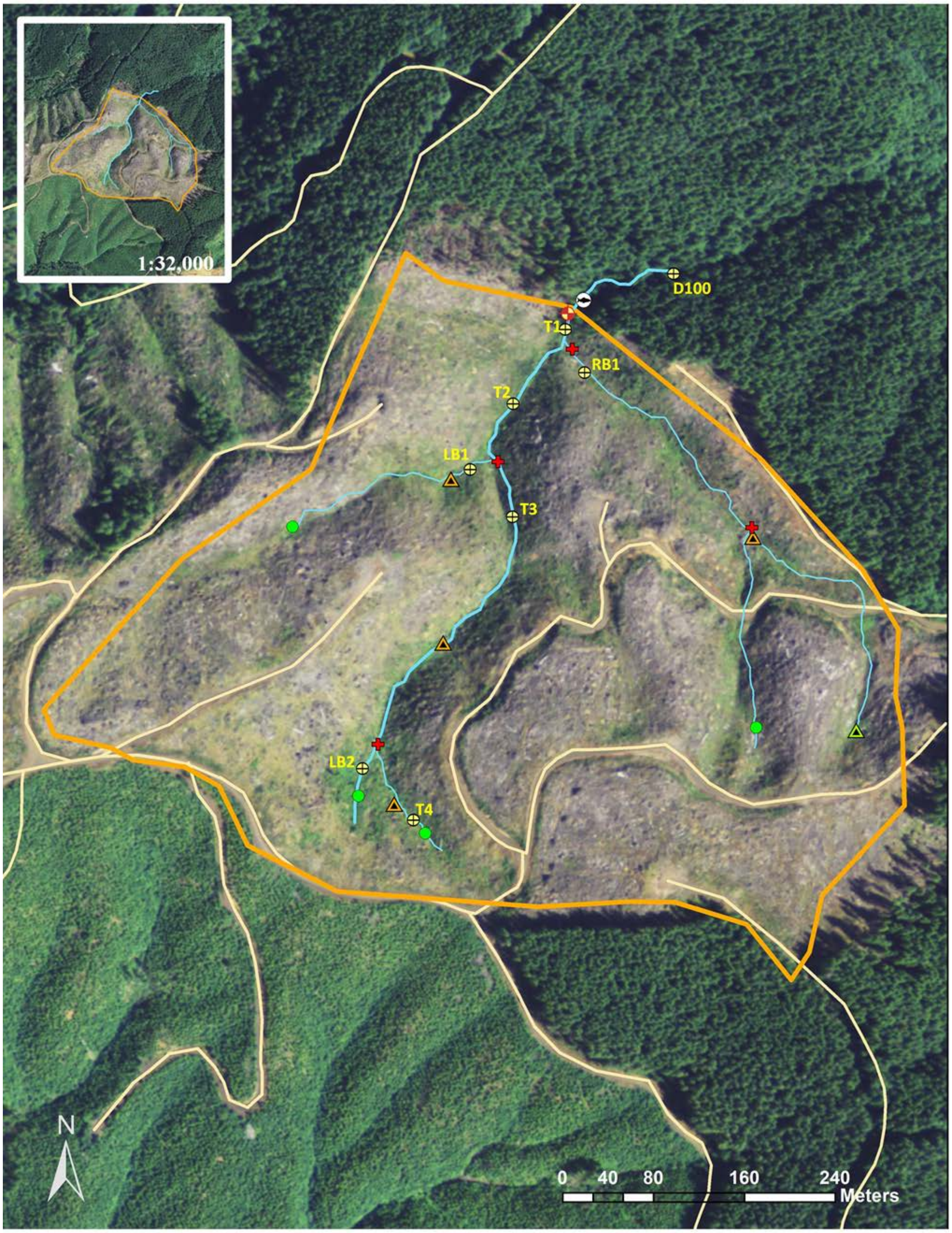

Figure 3-8. Type N Waters, sensitive sites and other features in the Willapa 1 block 0% treatment (WIL1-0%).

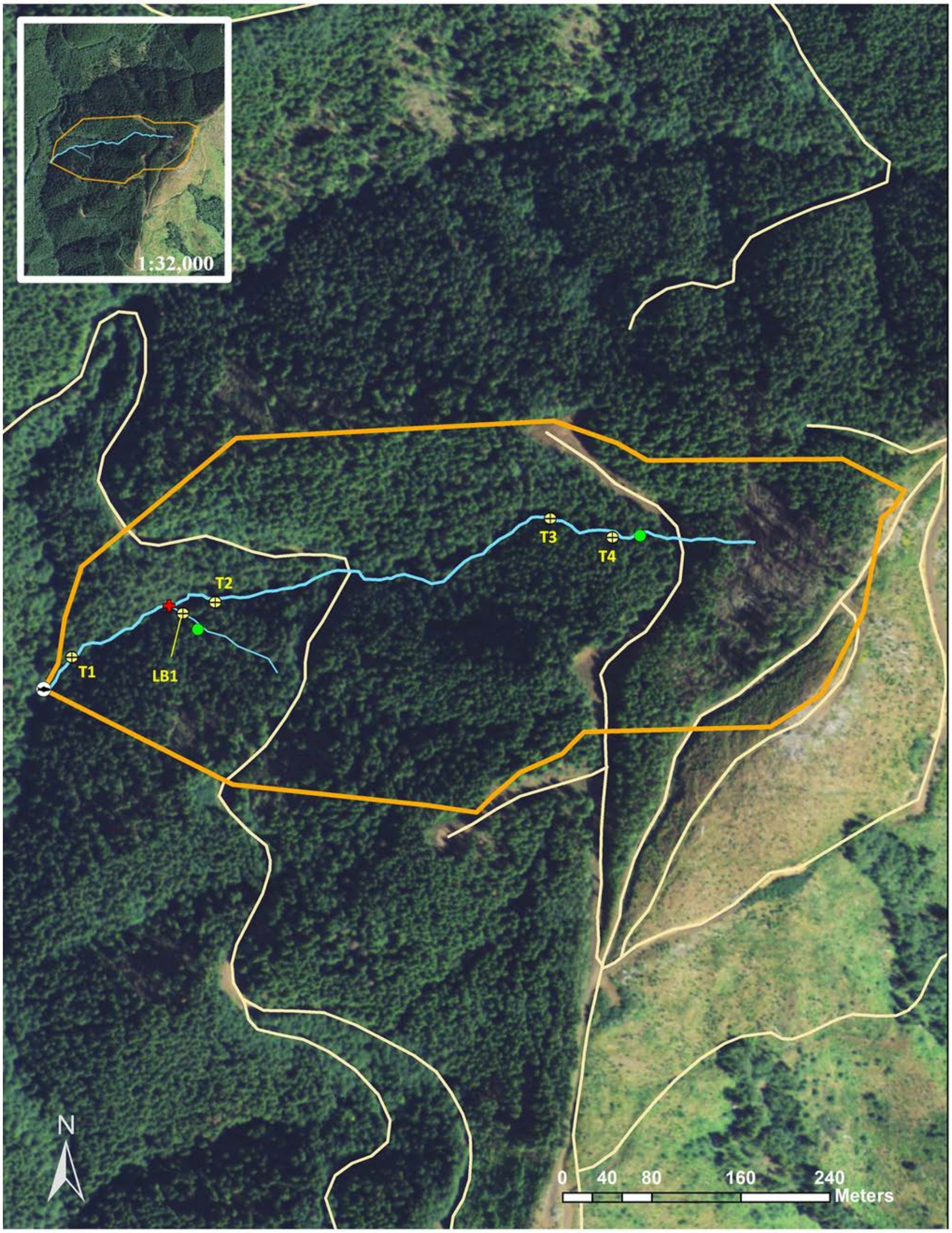

Figure 3-9. Type N Waters, sensitive sites and other features in the Willapa 2 block reference 1 (WIL2-REF1).

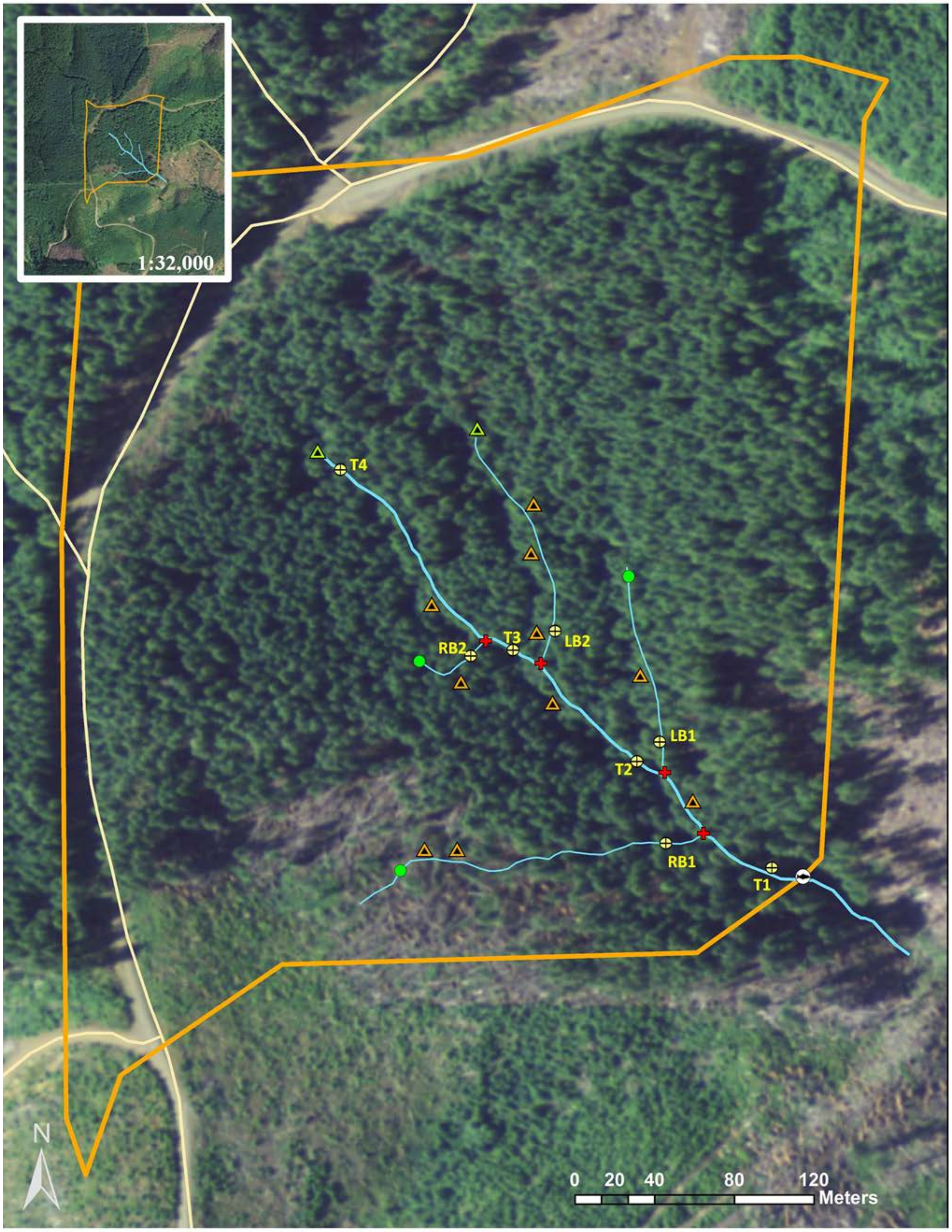

Figure 3-10. Type N Waters, sensitive sites and other features in the Willapa 2 block reference 2 (WIL2-REF2).

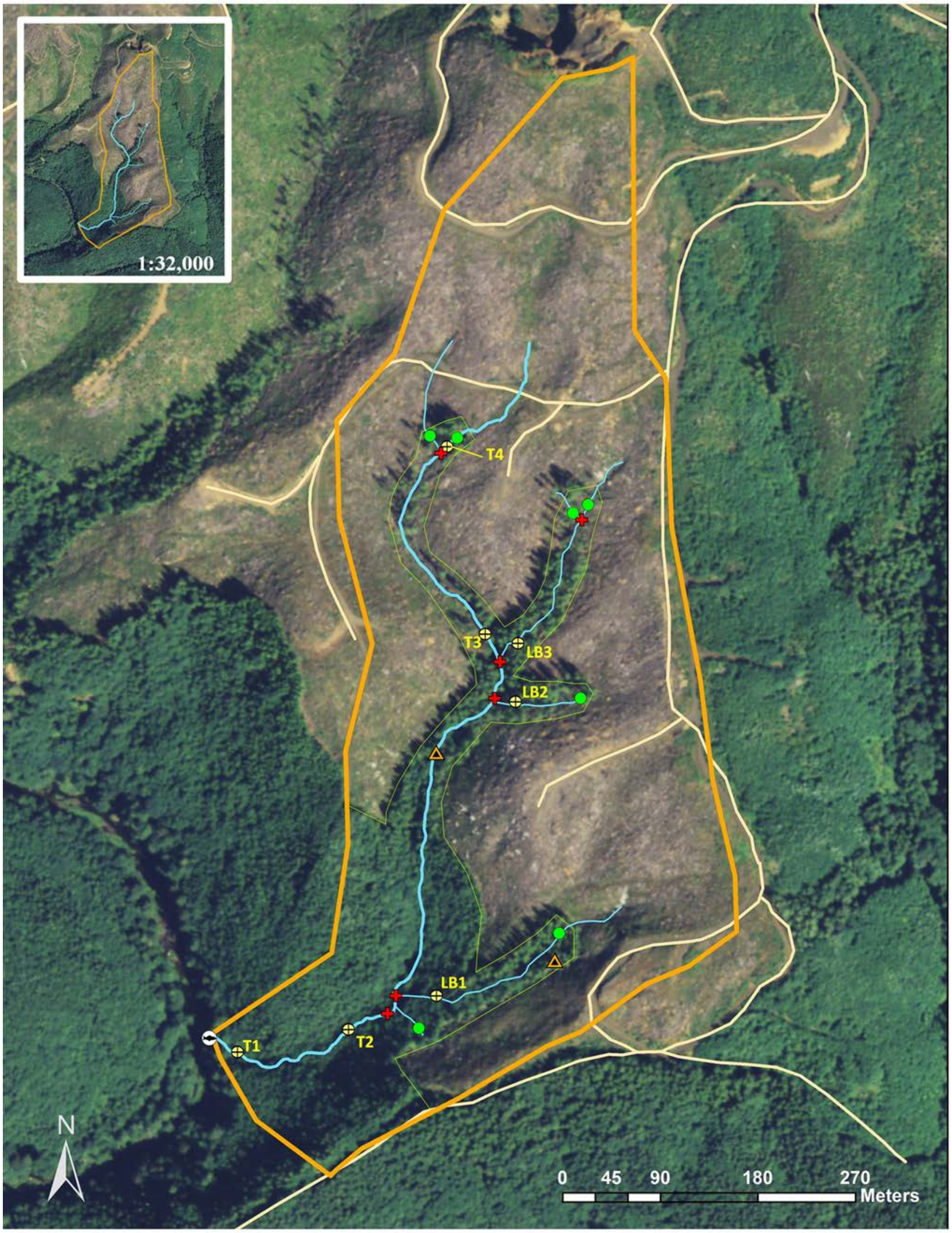

Figure 3-11. Type N Waters, sensitive sites and other features in the Willapa 2 block 100% treatment (WIL2-100%).

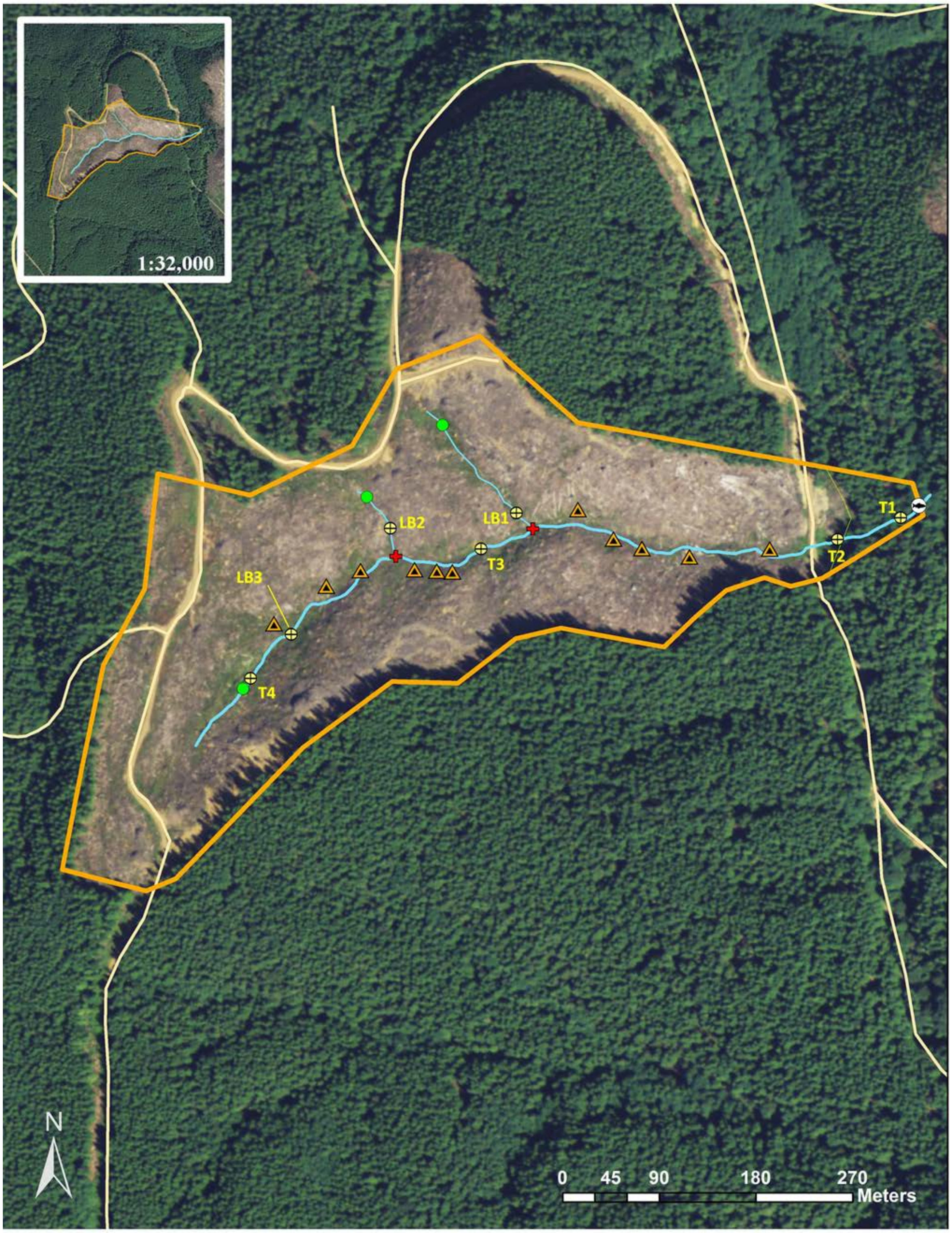

Figure 3-12. Type N Waters, sensitive sites and other features in the Willapa 2 block 0% treatment (WIL2-0%).

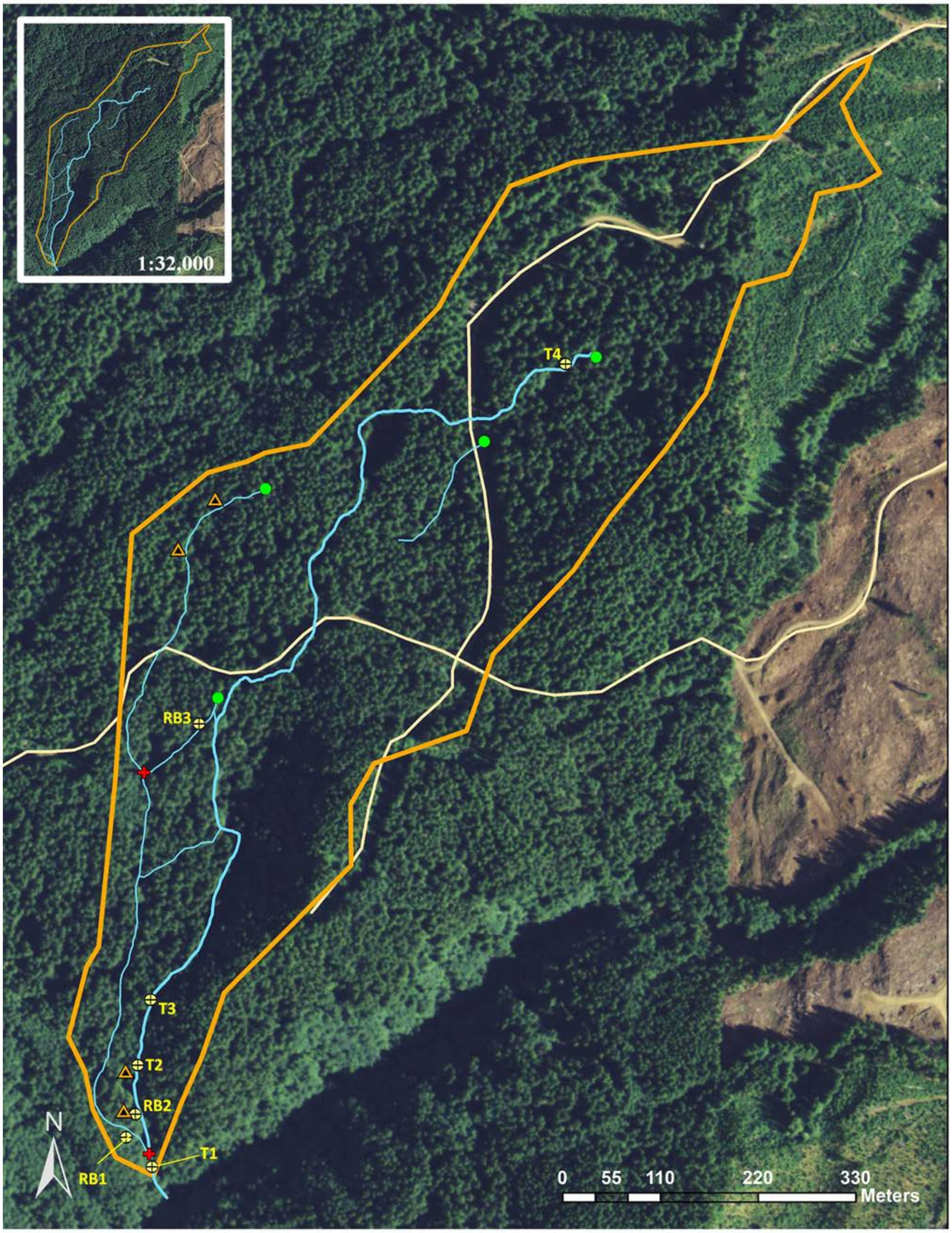

Figure 3-13. Type N Waters, sensitive sites and other features in the Willapa 3 block reference (WIL3-REF).

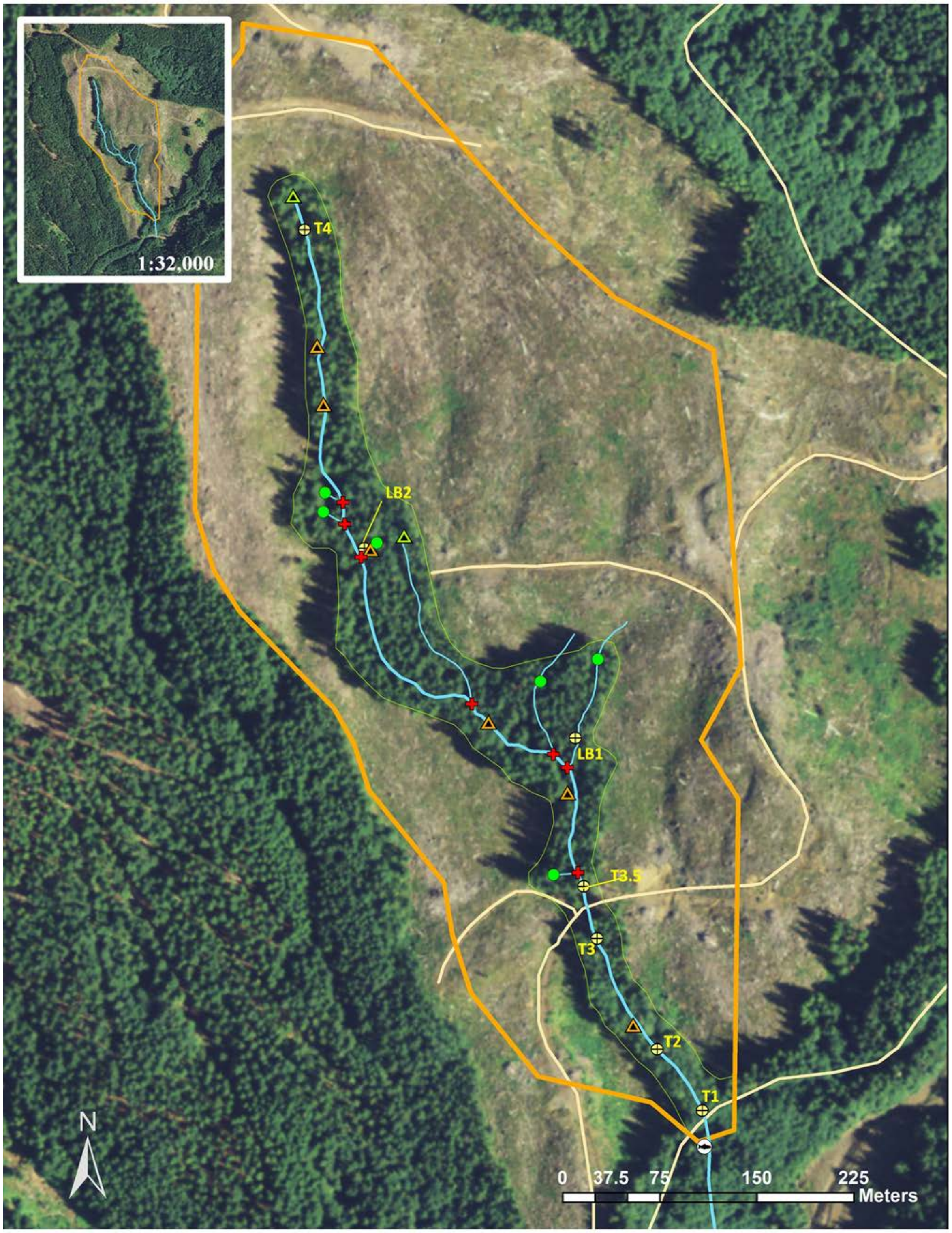

Figure 3-14. Type N Waters, sensitive sites and other features in the Willapa 3 block 100% treatment (WIL3-100%).

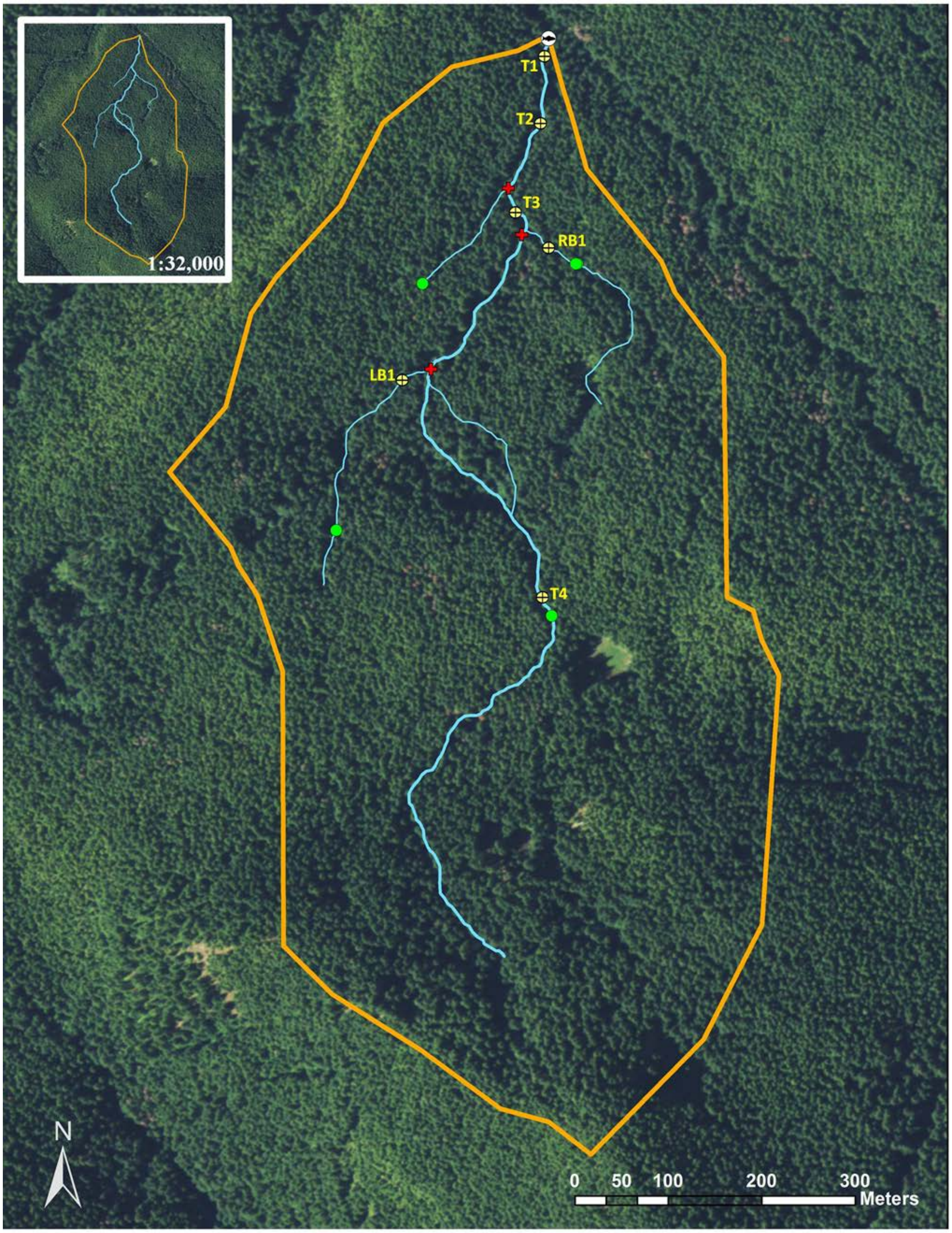

Figure 3-15. Type N Waters, sensitive sites and other features in the South Cascade block reference (CASC-REF).

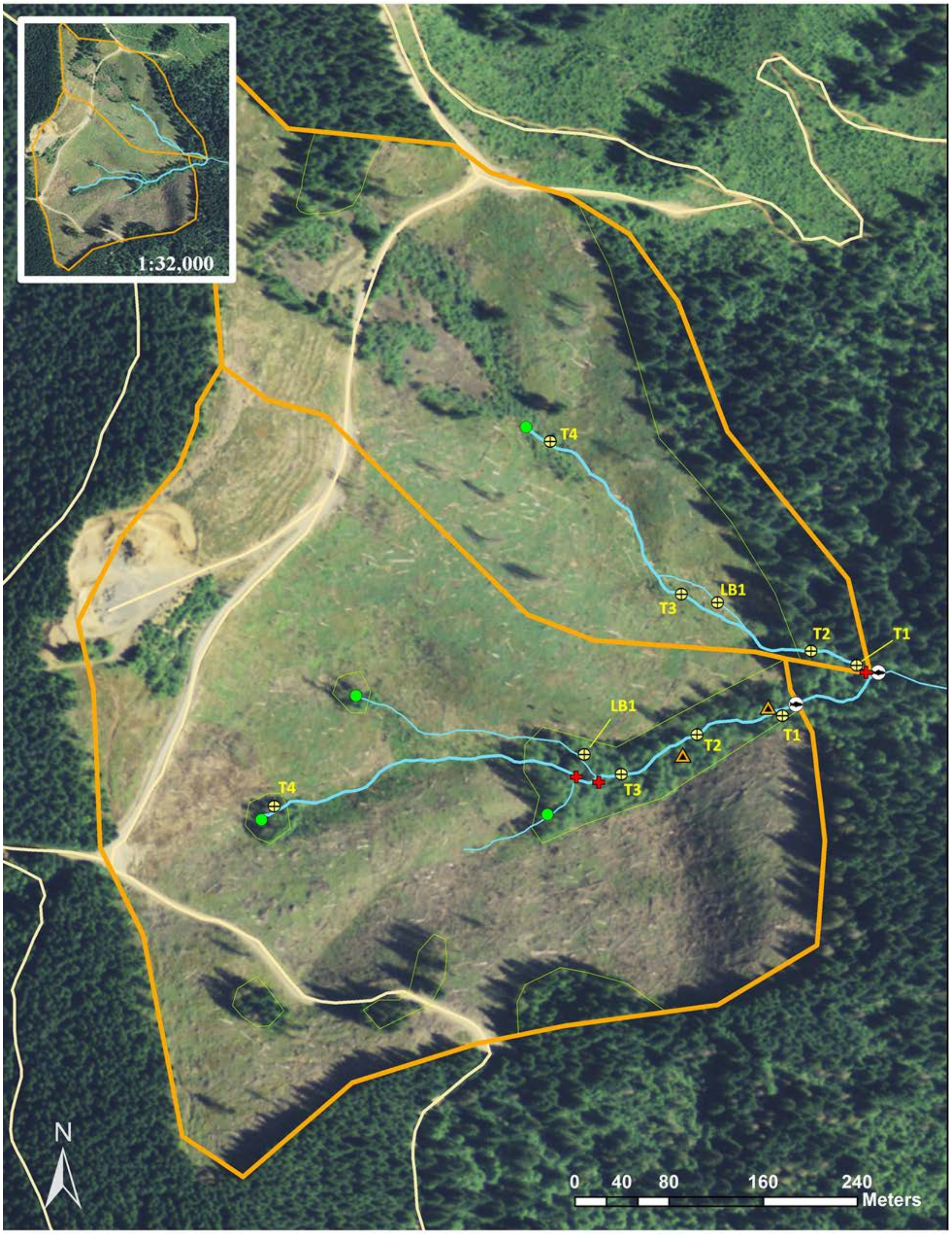

Figure 3-16. Type N Waters, sensitive sites and other features in the South Cascade block (from north to south): 0% treatment (CASC-0%) and FP treatment (CASC-FP).

3-5. REFERENCES

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CHAPTER 4 - UNANTICIPATED DISTURBANCE EVENTS

Aimee McIntyre, William Ehinger, Stephanie Estrella, Eric Lund, and Marc Hayes

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4-1. INTRODUCTION

Disturbances influence both natural and managed forestlands, and are a normal, even integral part of long-term ecological dynamics (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 a five-year period, it is not surprising that disturbance processes other than timber harvest activities impacted some study sites over the course of our investigation. Two disturbance events of note impacted study sites between 2006 and 2010: an extensive windthrow event in December 2007 that affected multiple study sites, and a wildfire in October 2009 that affected two buffer treatment sites located in the South Cascade block.

4-2. WINDTHROW

From 1–4 December 2007, a series of storms caused extensive windthrow throughout western Washington (**Figure 4-1**). Windthrow is defined as a natural process by which trees are uprooted or sustain severe trunk damage by the wind (WAC 222-16-010). The first of three separate storms arrived on 1 December 2007 with some areas receiving up to 360 mm (14 in) of snowfall. On 2 December 2007, a second storm brought more rain and snow, hurricane-force winds and tropically affected temperatures. In as little as two hours, temperatures across the region jumped from near freezing to above 60 degrees. The storm moved northward through Oregon and Washington with heavy rain (over 254 mm [10 in] in some areas) falling in a 24-hour period. The rapid rise in temperature caused the recent snow to melt quickly, with some rivers reaching flood stage early on 3 December 2007. A third storm brought sustained winds of 130 km (80 mi) per hour, with gusts up to 230 km (145 m) per hour. The windstorm resulted in significant damage to forestlands along the Washington coast from Naselle to north of Hoquiam.

Figure 4-1. Images of study site areas impacted by windthrow during December 2007: WIL1-100% (a) and WIL1-0% (b).

This windthrow event occurred within the pre-harvest sampling period prior to the implementation of buffer treatments. We were concerned about the impact of windthrow on pre-harvest conditions across our study sites, in particular that pre-harvest conditions may have changed, so we sampled again in 2008 just prior to harvest.

4-2.1. EVALUATION OF WINDTHROW ACROSS STUDY SITES

Pre-harvest sampling in 2008 included an evaluation of the extent and severity of windthrow across all study sites. We included an evaluation of aerial photos taken of study sites March through April 2008, as well as field estimates of downed tree counts within the bankfull channel.

4-2.1.1. Evaluation of Aerial Photos

To estimate the extent and severity of windthrow from the December 2007 storm throughout each study site, Washington Department of Ecology (Ecology) examined 1:12,000 stereophotos taken by Washington Department of Natural Resources (WADNR) March through April 2008. Stereophotos were available for 15 of 17 study sites (excluding the reference and 100% treatments in the Willapa 3 block). In fall 2008, Ecology delineated each study site on acetate overlying each photo. We identified and delineated areas of windthrow using a mirror stereoscope. We categorized windthrow severity by comparing stem counts between areas of windthrow and nearby areas with no visible windthrow, and assigned areas into one of four numeric categories:

1) None visible,
2) 1%–33% of stems down,
3) 34%–67% of stems down, and
4) >67% of stems down.

While this method is not exact given the variability of windthrow, the results of repeated categorization of the same areas were consistent.

We transferred polygons drawn on acetate to a GIS using landmarks visible on both the stereophotos and recent orthophotos from 2003 and viewed in ArcGIS (ESRI 2004). We calculated an estimate of the study site area impacted based on the location of the F/N break for all study sites (total area impacted). In addition, for the eight study sites with flumes or other stream flow monitoring equipment in the Willapa 1 and Olympic blocks, we calculated an estimate of the impacted area upstream of the equipment (flume area impacted). We estimated the proportion of total and flume areas, where applicable, within each windthrow severity category.

4-2.1.2. Field Evaluation

We conducted field evaluations of newly downed trees that entered within the stream bankfull channel in the approximately one-year period between summer 2007 and 2008. We could not differentiate between trees that fell as a direct result of the December storm versus trees that fell for some other reason; however, we assumed that the majority of newly downed trees were due to the December windthrow event. We identified newly downed trees by the presence of green needles still on branches or freshly disturbed soils around upturned root wads. We conducted post-windthrow sampling at all study sites from the F/N break along all Type Np Waters to each channel head. We identified each newly downed tree that entered into the bankfull channel. We recorded the exact location (tributary and distance upstream from F/N break), diameter (>2–10 cm, 11–25 cm, 26–50 cm, 51–100 cm, and >100 cm), length (0.1–1 m, >1–2 m, >2–4 m, >4–8 m, >8–16 m, and >16 m), decay class (1 = green needles present, 2 = no green needles, appearing to have been dead prior to falling), function (**Table 4-1**), position (**Table 4-2** and **Figure 4-2)**, and the dominant structure type (root, trunk, branch, needle). All evaluations, excluding length, were for the portion of the tree falling within the bankfull channel.

Table 4-1. Instream function categories for windthrown trees evaluated March through July 2008.

Function	Definition
Step	Contributes to step formation
Bank	Does not contribute to step formation; covers or is buried in bank. Does not include pieces lying on top of the bank.
Roughness	Does not provide step or bank function; provides hydraulic roughness that affects flow direction or velocity.
Over Non-Functional	Does not interact with, and is completely suspended over, the bankfull channel.

Table 4-2. Instream position categories for windthrown trees evaluated March through July 2008 (see **Figure 4-2**).

Position	Definition
Spanning	Tree trunk extends from one bank to the other. No portion of the trunk extends into the bankfull channel, however, roots, branches or needles may extend into the bankfull channel.
Hanging	Tree trunk does not extend from one bank to the other, but rather hangs over from one side only. No portion of the trunk extends into the bankfull channel, however, roots, branches or needles may extend into the bankfull channel.
Bank	At least one end of the tree trunk extends into the bankfull channel.

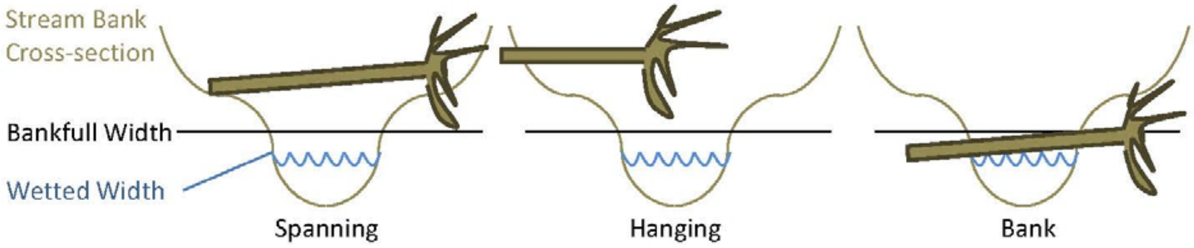

Figure 4-2. Downed tree instream position categories for windthrown trees evaluated March through July 2008 (see **Table 4-2**).

4-2.2. RESULTS OF WINDTHROW EVALUATION

4-2.2.1. Aerial Photo Results

Sites visibly impacted by windthrow as observed using aerial photos included all four study sites in the Willapa 1 block and two of four study sites in the Willapa 2 block (**Table 4-3**). When all windthrow severity classes were considered, windthrow exceeded 25% of the basin area in the WIL1-REF, -100% and -FP sites (**Figure 4-3**), and was less than 10% for the remaining sites. We did not note any windthrow on the stereophotos for sites located in the Olympic or South Cascade blocks. For the eight study sites with instream flow monitoring equipment, four had windthrow discernable using aerial photos (**Table 4-3**). Values for total and flume areas impacted are comparable.

4-2.2.2. Field Evaluation Results

In our field evaluation of newly downed trees, we observed 2,380 downed trees across our 17 study sites. The number of newly downed trees at a site ranged from one in the CASC-0% to 458 in the WIL1-100% site. The WIL1-REF and -100% sites had the largest number of downed trees, 380 and 458 respectively (**Table 4-3**). However, the total number of downed trees does not correspond directly to the proportion of the stream channel impacted. To understand the proportion of a site impacted relative to Type Np Water length, we divided each stream into consecutive 10 m intervals and calculated the proportion of 10 m intervals with at least one newly downed tree. The proportion of Type Np Water length impacted ranged from 0.02 in the CASC-0% to 0.76 in the WIL1-REF site. The WIL1-REF and -FP, and the WIL2-0% sites had the greatest proportion of Type Np Water length impacted (0.76, 0.54 and 0.54, respectively). The average number of downed trees per interval ranged from 0.02 trees per 10 m interval in the CASC-0% to 6.13 trees per 10 m interval in the WIL1-REF site. The WIL1-100% site had the highest number of downed trees in a single 10 m interval (36).

Similarities existed in the results from the aerial photo and field evaluations of windthrow. Both evaluations indicate that the study sites that were most highly impacted by windthrow were located in the Willapa 1 and Willapa 2 blocks. However, the aerial photo evaluation was not able to discern individual trees or even very small patches of downed trees in some study sites, including most notably sites located in the Olympic block and the reference in the South Cascade block. In general, it appears that when either the proportion of Type Np Water impacted was less than 20%, or the number of downed trees per 10 m interval was less than 1.25, the aerial photo evaluation was not able to detect windthrow. However, the WIL2-0% site had 1.99 windthrown trees per 10 m interval and we estimated that 54% of the Type Np Water length was impacted, while we did not detect any windthrow with the aerial photo evaluation.

Table 4-3. Results of field and aerial photo evaluations of windthrow for study sites included in the Type N Study. Field evaluation includes the total number of downed trees tallied along the entire Type Np Water length (# downed trees), the proportion of Type Np Water length with one or more downed trees observed (Type Np Water impacted), and the average number of downed trees per 10-m stream interval (downed trees/10 m). The aerial photo evaluation includes the proportion of the total study site area (from F/N break) and the flume area (from flow monitoring equipment, when applicable) impacted by windthrow considering all three severity classes. Aerial photos were not available for the WIL3-REF or -100% sites.

Block	Treatment	Field Evaluation			Aerial Photo Evaluation	
		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	-
	REF2	223	0.42	2.62	0.08	-
	100%	190	0.36	1.16	0.00	-
	0%	207	0.54	1.99	0.00	-
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	-
	FP	3	0.03	0.03	0.00	-
	0%	1	0.02	0.02	0.00	-

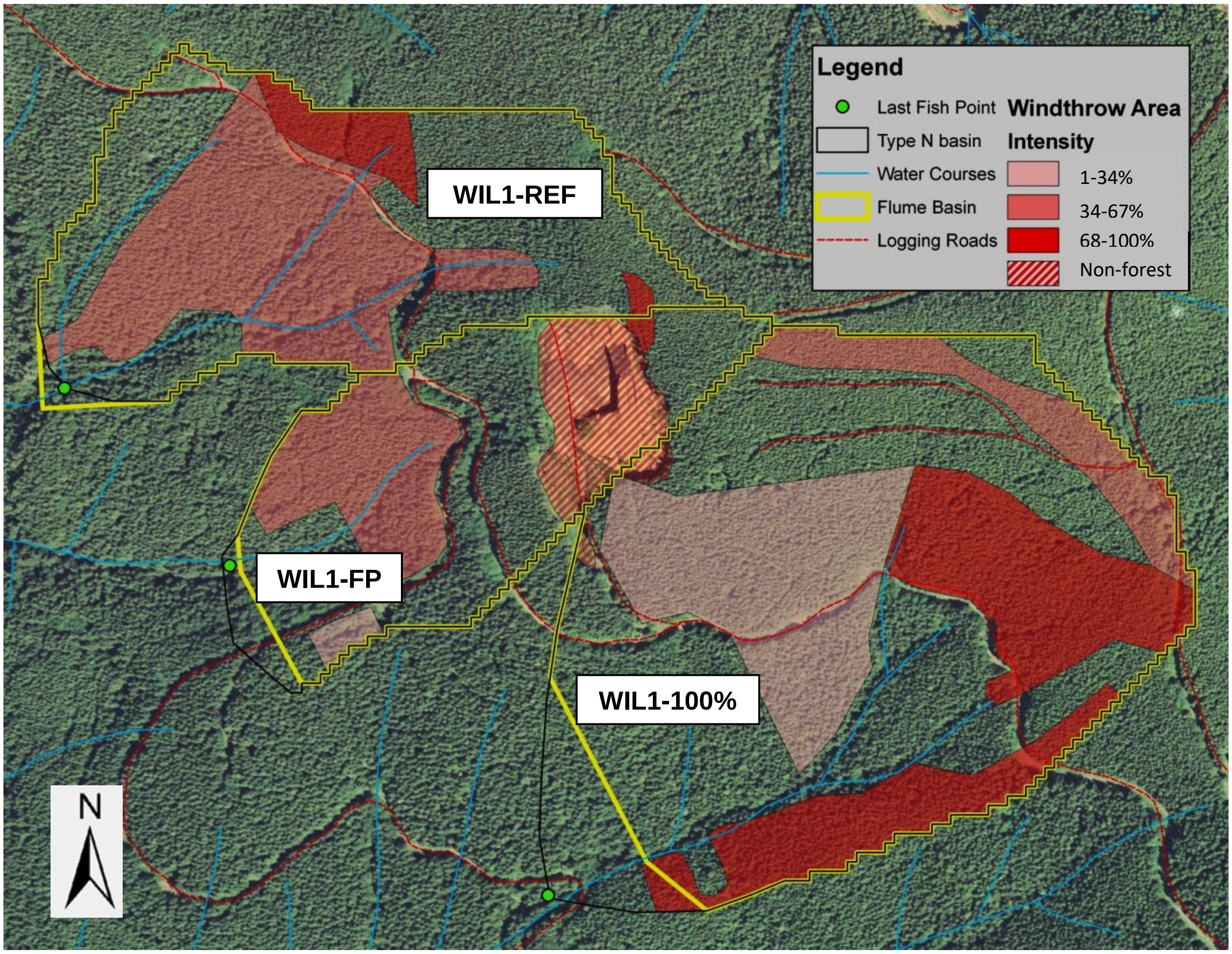

Figure 4-3. Windthrow severity (windthrow area intensity) classes for the Willapa 1 block reference, FP and 100% treatment sites. Area labeled 'non-forest' is a rock quarry and adjacent road. The background image is an orthophoto taken in 2003 before the windthrow occurred. Black outline is the total study site delineation (total area). Yellow outline is the basin flume area, or area above instream flow monitoring equipment.

4-2.3. POTENTIAL IMPLICATIONS OF WINDTHROW ON THE INTERPRETATION OF STUDY RESULTS

Fortunately, since the windthrow event occurred prior to harvest, we had the opportunity to collect additional pre-harvest data after the windthrow event. Our pre-harvest data reflect the broad range of disturbances that occur throughout the managed forestlands of western Washington. Windthrow occurred across sites in all treatment assignments, including references. Regardless of the metric considered (including both aerial photo and field evaluations), our data indicate that the severity of windthrow was greatest for sites located in the Willapa 1 and 2 blocks, while windthrow severity was comparably moderate for sites located in the Olympic block, and was generally minimal for sites in the Willapa 3 and South Cascade blocks. We assigned sites to blocks *a priori* to reduce known variability by grouping sites that were similar; fortunately, the severity of windthrow among sites in a block was more similar than for sites between blocks. Nonetheless, data analysis and interpretation for response variables, especially for riparian vegetation and wood, will require careful consideration of the timing and severity of the windthrow event, as well as the disproportionate impact across blocks and treatments. We discuss the potential effect of the windthrow event for each response variable where appropriate.

4-3. WILDFIRE

In October 2009, a post-harvest wildfire burned portions of two study sites (CASC-FP and -0% sites) harvested November 2008 through March 2009. The fire was extinguished with water from fire engines and helicopter bucket drops by 14 October 2009, with the exception of a few hot spots that were not a threat for potential spread. No bulldozers or fire retardants were used. The fire had no impact to future management plans for the study sites.

The fire affected approximately 12.5 hectares (31 acres) across the two study sites (combined area 40 ha [100 ac]). A site visit on 22 October 2009 revealed that the 0% site was impacted more than the FP site. Approximately 6.6 of 14 ha (16 of 36 ac) in the 0% site were affected. The fire crossed the Type Np Water from approximately 130 m upstream of the F/N break to 40 m below the uppermost point of perennial flow, for a length of approximately 260 m (**Figure 4-4**). Some areas along the stream burned to the level of soil and ashes, and approximately 200 m of stream length contained charred logs and wood within the channel. The fire did not directly affect any sensitive sites. Approximately 5.9 of 26 ha (15 of 64 ac) in the FP site were impacted. The fire did not cross the stream, and therefore did not greatly affect the riparian buffers, but did come to within feet of the stream in several places (**Figure 4-4**). While the riparian buffers remained intact, some buffer trees had minor scorching.

4-3.1. POTENTIAL IMPLICATIONS OF FIRE ON THE INTERPRETATION OF STUDY RESULTS

We do not anticipate that the forest fire in two study sites will impair our ability to interpret study results or assign cause and effect to potential pre- to post-harvest changes among treatments. The forest fire only affected one of three FP and one of four 0% treatments, and both sites are within the same block. Since the application of harvest treatments had already occurred, very little forest was available to burn. There were trees only in the riparian buffers of the FP treatment. They were not greatly impacted.

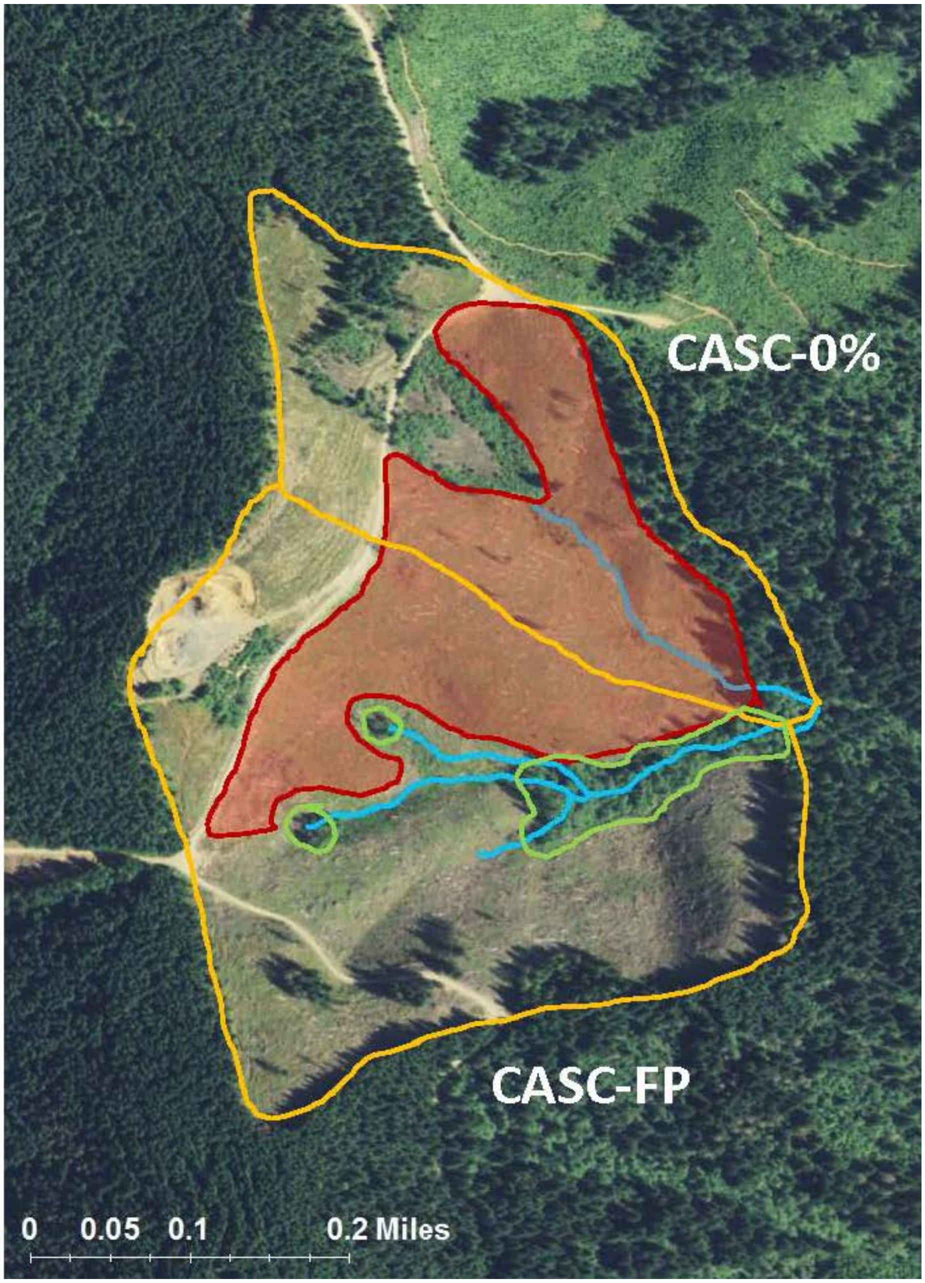

Figure 4-4. Area impacted (in red) by the October 2009 forest fire in the FP and 0% treatment sites in the South Cascade block (study sites outlined in orange). Green outlines delineate riparian buffers.

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CHAPTER 5 - STAND STRUCTURE AND TREE MORTALITY RATES IN RIPARIAN BUFFERS

Dave Schuett-Hames and Greg Stewart

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5-1. ABSTRACT

This study evaluated tree mortality rates and changes in stand structure in riparian buffers in response to different buffering strategies for non-fish-bearing, perennial (Type Np) streams on western Washington forestland. Three treatments that differed in the proportion of stream network buffered were compared with unharvested references. Treatments included the 0% treatment (no buffers), the Forest Practices rule (FP) treatment (minimum of 50% of Np network buffered) and the 100% treatment (entire Np network buffered). Treatment and reference sites were blocked geographically. The 50-ft (15.2-m) wide riparian management zone (RMZ) and 56-ft (17.1-m) radius perennial initiation point (PIP) buffers were sampled before and two years after harvest. General linear mixed-effect models were used for between-treatment and treatment-reference comparisons.

Prior to harvest, most stream-adjacent stands consisted of dense second-growth conifers. Tree density and basal area were lower and broadleaf trees were more abundant in the South Cascades block. A pre-harvest December 2007 storm generated hurricane-force winds in coastal southwestern Washington, causing extensive, but patchy, mortality in two of three coastal blocks. Little pre-harvest mortality occurred elsewhere. During the first two years post-harvest, there were significant differences in tree mortality among treatments. The highest mortality and greatest reductions in density and basal area occurred in FP treatment buffers. Post-harvest mortality in the FP treatment RMZs was ~18% of initial density and basal area, over twice the rates in the reference and 100% treatments. There was no significant difference in mortality between the reference and 100% treatment RMZs. For PIP buffers, mortality rates for the FP and the 100% treatments were eight times and over four times the reference rates, respectively. Differences in mortality between FP and 100% treatment PIPs were equivocal. These results are consistent with earlier studies, including the Westside Type N Buffer Characteristics, Integrity and Function Study, which documented higher mortality rates in FP RMZs and PIPs compared to unharvested reference sites. Post-harvest stand structure differed among experimental treatments due to harvest and subsequent mortality. Among RMZs, post-harvest basal area was highest in the reference and 100% treatments. Higher mortality in the FP treatment resulted in a lower mean and range of density and basal area values compared to the reference or 100% treatments.

The FP treatment was less effective than the 100% treatment in maintaining stable stand structure in riparian buffers over the short term. Approximately 75% of FP treatment RMZs and 40% of PIPs are likely to develop as single-age conifer stands through the stem-exclusion phase. In the remaining buffer stands, lower densities (due to wind or fire disturbance) and reduced competition are likely to produce multi-age conifer stands if conifer regeneration is successful. If conifer regeneration is poor, the understory may become dominated by broadleaf trees or shrubs. Local and regional differences in susceptibility to wind can create a mosaic of riparian buffer stand structure across the landscape. Since wind and fire are episodic disturbances, additional monitoring is needed (and planned) to document changes in mortality rates, determine whether differences among treatments persist over time, and track patterns of stand development over a longer timeframe.

5-2. INTRODUCTION

Riparian forests are the interface between the terrestrial environment and aquatic ecosystems, and play an important role in the productivity of headwater stream channels and aquatic resources (Gregory *et al.* 1991; MacDonald and Coe 2007). Stream-adjacent forests affect the productivity of aquatic systems by providing: (1) shade that reduces solar energy input to water and modulates heat exchange (Dent *et al.* 2008); (2) wood that creates cover, forms habitat, and stores sediment (Bilby and Ward 1989); (3) leaf litter that contributes nutrients to the aquatic food chain (Fisher and Likens 1973); and (4) by creating a barrier to delivery of sediment and slash resulting from timber harvest of adjacent uplands (Jackson *et al.* 2001; Litschert and MacDonald 2009).

Harvest of timber in forests adjacent to streams can affect inputs of solar radiation (Gomi *et al.* 2006a), litter and nutrients (Richardson *et al.* 2005), and large wood (Gomi *et al.* 2006b). The nature and magnitude of these changes depends on the type and intensity of harvest, site conditions, and weather. Riparian buffers consisting of strips of leave trees adjacent to the stream have been used to reduce the effects of timber harvest. However, tree mortality can be extensive when the edges of these buffers are exposed to the wind (Grizzel and Wolff 1998; Grizzel *et al.* 2000; Liquori 2006; Schuett-Hames *et al.* 2012), resulting in loss of shade and future input of wood to the stream.

This study examined the short-term effects of three experimental riparian buffer treatments on tree mortality and associated changes in stand structure. The buffer treatments represent three points along a continuum of riparian buffering strategies for the network of perennial non-fish bearing streams (Np streams) in headwater basins. They range from no buffers on any portion of the network (0% treatment) to complete buffering of the network (100% treatment). The intermediate, or Forest Practices (FP) treatment, represents the current Forest Practices rules on private forestlands in western Washington, in which a minimum of 50% of the network is buffered. Tests of the effects of these treatments on residual stand structure are critical to understanding the broader effects of riparian buffering strategies on inputs to streams and the responses of aquatic habitat and biota. Analyses of tree mortality patterns and resulting stand structures contribute to a broader evaluation of riparian buffering strategies on Type Np streams. The analyses have two primary purposes: (1) to document the magnitude of tree mortality and change in forest structure associated with a range of buffering strategies, and (2) to provide stand structural data as potential covariates in analyzing responses of other biotic components or ecological processes in these systems.

5-3. OBJECTIVES

The primary objectives of our analyses were to: (1) characterize rates of tree mortality and stand structure in riparian buffers both prior to and after timber harvest; and (2) determine whether post-harvest mortality and stand structure differ among experimental treatments defined by the proportion of the stream network that was buffered. We addressed the following questions in our analyses:

1) What were the rates of tree mortality and magnitudes of change in riparian buffer stand structure during the pre-harvest period?

2) What were the rates of tree mortality and the magnitudes of change in riparian buffer stand structure during the post-harvest period?

3) Did rates of tree mortality and changes in riparian buffer stand structure differ among experimental treatments and unharvested reference sites in the post-harvest period?

4) What were the dominant mortality agents and characteristics of trees that died during the pre- and post-harvest periods?

5-4. METHODS

This section provides an overview of the sampling strategy and methods for data collection and analysis. We detailed information on the overall study design in Chapter 2 – *Study Design*.

5-4.1. SAMPLING STRATEGY

We evaluated two riparian management strategies defined by state Forest Practices rules (WFPB 2012). Riparian management zones (RMZs) are 50-ft (15.2-m) wide bands adjacent to both sides of Type Np streams. Perennial initiation points (PIPs) are sensitive sites located at the uppermost point of perennial flow, surrounded by a 56-ft (17.1-m) radius management zone. In this study, management of the RMZs and PIPs varied among the four basin-scale experimental treatments as follows:

1) <u>0% treatment</u>: the entire length of the Type Np stream network and all PIPs were clearcut to the edge of the stream. The uplands were also clearcut.

2) <u>Forest Practice (FP) treatment</u>: Approximately 50% of the length of the Type Np stream network received a 50-ft (15.2-m) wide buffer; the remainder of the RMZ was clearcut. The buffered portion is referred to as the FPB treatment and the clearcut portion as FPU treatment. All PIPs in the FP treatment received a 56-ft (17.1-m) radius buffer (no trees removed). Adjacent uplands were clearcut.

3) <u>100% treatment</u>: A 50-ft (15.2-m) wide buffer was retained along the entire length of the Type Np stream network and a 56-ft (17.1-m) radius buffer was retained around all PIPs. No trees were removed from these buffers. Adjacent uplands were clearcut.

4) <u>Reference sites</u>: No trees were cut in or adjacent to the RMZ or PIPs.

In portions of some sites, it was not possible to harvest adjacent to the RMZ or PIPs due to the potential for mass wasting, buffer requirements for adjacent Type F streams, or logistical constraints. In areas where harvest did not occur, plots in the 0%, FP, or 100% treatments were not included in the analysis.

We collected data at 17 sites consisting of non-fish-bearing headwater basins located in competent lithologies (largely basaltic) across western Washington (see Chapter 2 – *Study Design*). We blocked sites by geographic location due to regional variability in climatic and environmental conditions. The Olympic block was located in the southwestern foothills of the Olympic Mountains, three blocks were in the Willapa Hills, and one block was in the southern Cascade Range just north of the Columbia River Gorge. We describe site characteristics in Chapter 2 – *Study Design*.

5-4.1.1. Riparian Management Zone Sampling Strategy

Riparian zones present a unique sampling situation due to their shape (narrow linear features) and extensive variability in stand structure and mortality both along and perpendicular to the stream. We based our RMZ sampling strategy on the recommendations of Marquardt and colleagues (2010), who found that a series of small strip plots oriented perpendicular to the stream performed well in characterizing stand structure in headwater stream RMZs. We used a systematic design to establish permanent sample plots in RMZs on the mainstem and on all tributaries that were long enough to accommodate a plot. Plot density varied with basin size due to temporal constraints on sampling: at sites with <5,000 ft (<1,524 m) of RMZ (n = 11), we established plots every 100 ft (30.48 m), thus sampling ~50% of the RMZ. At sites with >5,000 ft (>1,524 m) of RMZ (n = 6), we established plots every 150 ft (45.72 m), sampling ~33% of the RMZ. As a result, the number of plots varied substantially among sites (**Table 5-1**).

Table 5-1. Number of RMZ and PIP plots by block x treatment (site).

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 established plots systematically in a downstream direction from the upper end of the RMZ. On each stream, we selected a starting point of either 0 or 50 ft (0 or 15.2 m) from the downstream end of the PIP buffer at random. Each plot was 50 x 100 ft (15.2 x 30.5 m), with the long axis perpendicular to the stream channel, extending out 0 ft (15.2 m) from each side of the stream, for a total area of 0.115 ac (0.0046 ha). We measured plot dimensions in horizontal distance using a laser rangefinder with foliage filter. Once we established plots on the mainstem, we used the same procedure to establish plots on each tributary. We did not sample plots that overlapped with plots in adjacent RMZs or that intersected roads.

5-4.1.2. Perennial Initiation Point (PIP) Sampling Strategy

The availability of PIPs varied among sites and treatments due to differences in basin size and numbers of tributaries. One site had no PIPs, and we did not sample PIPs at the Willapa 3 sites due to uncertainty about inclusion of this block in the study. We collected data at 34 PIPs in 14 sites (**Table 5-1**). We established PIP buffer plots using a laser rangefinder with a radius of 56 ft (17.1 m; horizontal distance) and an area of 0.226 ac (0.092 ha). As with RMZs, we excluded plots that overlapped with those from adjacent streams or that intersected roads.

5-4.2. DATA COLLECTION

In each plot, we collected data on all standing live and dead trees with diameters ≥4 in (≥10.2 cm) at breast height (4.5 ft [1.37 m] above ground). Data recorded included the condition (live or dead), species, and diameter at breast height (DBH). We recorded the canopy class (overstory, understory, or open) for live trees and the mortality agent for trees that died (e.g., wind, erosion, suppression, fire, insects, disease, and physical damage) when it was possible to determine. Detailed information on field methods is available in Schuett-Hames and Roorbach (2010).

We collected pre-harvest data during summer 2007 and 2008, with one exception: we did not sample Willapa 3 block sites in 2007, so we reconstructed 2007 stand conditions from data on standing and fallen trees in 2008. We collected post-harvest data during the first and second summers following harvest (2009 and 2010) at all sites.

5-4.3. DATA ANALYSIS PROCEDURES

5-4.3.1. Metrics

We calculated tree mortality as a percentage of initial live tree count (% of stems/yr) and initial live basal area (% of basal area/yr) for each plot. We expressed rates on an annual basis using a compounding formula (Sheil *et al.* 1995):

$$m = 100[1-(N_t/N_0)^{1/t}] \tag{5-1}$$

where: m is the annual mortality rate (%/yr),

N_0 is the initial density (or basal area) of live trees,

N_t is the density (or basal area) of live trees at the end of the measurement period; and

t is the length of the measurement period (yr).

Ingrowth (trees present at the end, but not at the start of the measurement period) were not included in the calculation. We calculated stand structural metrics separately for live and dead trees for each plot on each sample date. We calculated density (trees/ac), basal area (ft^2/ac), percent conifer basal area (% conifer basal area as the percentage of total basal area), and quadratic mean diameter (QMD, the square root of the mean basal area for the plot divided by 0.005454).

Metrics for change in stand structure were calculated in the pre-harvest (2007–2008) and post-harvest (2008–2010) periods. We computed proportional change in stem count (%Δ in stems) or basal area (%Δ in basal area) in a plot as (initial value − final value)/initial value.

5-4.3.2. Analyses

We addressed the following four questions:

1) What were the pre-harvest tree mortality rates and change in stand structure?

We expressed annual mortality and proportional change in stand structure during the pre-harvest period as both the proportion of original stems (% of stems/yr) and basal area (% of basal area/yr). We averaged plot values by site for each plot type (RMZ and PIP), and site values by block. We examined frequency distributions of plot values for pre-harvest tree mortality rates (basal area/yr) and changes in stand structure (basal area) to evaluate plot-scale variability for each plot type.

2) What were the post-harvest tree mortality rates and change in stand structure?

We analyzed annual mortality and proportional change in stand structure during the post-harvest period in a similar fashion to the pre-harvest data, using 2008 and 2010 as the initial and final values, respectively. However, we included only the reference, 100%, and the buffered portions of the FP treatments, because nearly all trees were harvested in the 0% and unbuffered portions of the FP treatments. We produced frequency distributions similarly.

3) What were the post-harvest differences among experimental treatments and reference sites?

To determine post-harvest differences among experimental treatments and reference sites we used post-harvest tree mortality data (% of stems/yr and % of basal area/yr) and changes in stand structure (%Δ in stems and %Δ in basal area), excluding the 0% treatment, unbuffered portions of the FP treatment, and any other plots that had no trees at the beginning of the post-harvest period.

We conducted all statistical modeling with Generalized Linear Mixed Models (GLMM) using the GLIMMIX procedure in the SAS ® 9.2 software (SAS 2013). We conducted separate analyses for plots in RMZs and PIPs because their prescriptions differed. We first averaged values from RMZ plots for each site, since RMZs are linear features and plots are not independent. This also helped meet the distributional assumptions of the linear model. We did not average PIP plots within sites because we assumed they were spatially independent. Mixed models account for the correlation implicit in hierarchical nesting of PIP plots within sites and sites within blocks. An added advantage of mixed models is that they accommodate missing data if data are missing at random (SAS 2013).

GLMM can be used to fit data that derive from non-normal distributions with monotonic link transformations. In GLIMMIX, it is assumed that a link function to the error distribution can be used to model data from a set of exponential distributions including the binomial distribution. The link functions used in this analysis were identity and logit. We used the identity link for analyses where the residuals (errors) were approximately normally distributed, and the logit link for tree mortality, which was analyzed as events/trials and was characterized by a binomial distribution (**Table 5-2**).

All the statistical models incorporated treatment (including reference) as a fixed effect and block as a random effect. For the PIP plots, site was nested within block as a random effect. We estimated model parameters using Restricted Maximum Likelihood for data with a Gaussian distribution, and we estimated tree mortality by restricted pseudo-likelihood. Random effects are assumed to be normally distributed (SAS 2013). 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. The covariance matrices for random effects were based on compound symmetry, which is a form that arises naturally with nested random effects, as when sub-sampling error is nested within experimental error. We visually assessed residuals to see if they met model assumptions. PIP tree mortality models initially exhibited over-dispersion, which we corrected by including a multiplicative over-dispersion parameter. For each model, we compared the reference, 100% and FPB treatments with a set of pairwise contrasts.

Table 5-2. Description of metrics used in the analysis.

Type	**Observed Response**	**Scale**	**n**	**Distribution/Link**	**Degrees of Freedom**		
					100%-REF	**FPB-REF**	**FPU-100%**
RMZ	%Δ in stems	Site	17	Gaussian/Identity	9.3	9.7	9.9
RMZ	%Δ in basal area	Site	17	Gaussian/Identity	9.4	9.8	10.0
RMZ	Tree mortality- % of stems/yr	Site	13[1]	Binomial/Logit	6.2	6.3	7.2
RMZ	Tree mortality- % of basal area/yr	Site	13[1]	Binomial/Logit	6.9	6.5	7.1
PIP	%Δ in stems	Plot	32	Gaussian/Identity	8.4	8.0	8.2
PIP	%Δ in basal area	Plot	32	Gaussian/Identity	8.8	7.3	8.1
PIP	Tree mortality- % of stems/yr	Plot	24[1]	Binomial/Logit	7.8	7.0	7.6
PIP	Tree mortality- % of basal area/yr	Plot	24[1]	Binomial/Logit[2]	7.4	6.9	7.3

[1] Excludes the 0% treatment, unbuffered portion of the FP treatment, and plots that had no trees in 2008.
[2] Models included an over-dispersion parameter.

4) What were the dominant mortality agents and characteristics of trees that died?

To determine the dominant mortality agents and characteristics of trees that died we summarized mortality data by agent, tree crown class, mean diameter, and species. To assess the relationship between mortality and timing of windstorms, we used daily data (NOAA National Climate Data Center 2013a) from three weather stations closest to the study blocks: Hoquiam, Washington (near the Olympic block); Astoria, Oregon (near Willapa 1 and 2); and Portland, Oregon (near Willapa 3 and South Cascades). Daily records of the highest five-second wind speeds were used to determine the number of days before and after harvest when speeds exceeded the criteria for storm-force winds (24.6–33.1 m/s) or hurricane-force winds (33.1–50.1 m/s) according to the Beaufort wind scale (NOAA National Weather Service 2013b) (**Appendix Table 5-A-1**).

5-5. RESULTS

5-5.1. PRE-HARVEST TREE MORTALITY AND STAND STRUCTURE

5-5.1.1. Initial Stand Structure

There was substantial variation in stand structure among sites at the beginning of the study. Second-growth conifers dominated most stands. In the RMZs, initial live density ranged from 91.0 to 385.5 trees/ac (224.9 to 952.6 trees/ha; **Appendix Table 5-A-2**). Initial basal area ranged from 113.8 to 317.8 ft^2/ac (26.2−73.1 m^2/ha). Conifers accounted for 5.2–100% of the basal area. Among PIPs, initial live density ranged from 48.6 to 307.3 trees/ac (120.1−759.3 trees/ha; **Appendix Table 5-A-3**). Initial basal area ranged from 53.2 to 303.0 ft^2/ac (12.2−69.7 m^2/ha). Conifers accounted for 8.6–100% of the basal area.

Initial stand structure varied among blocks (**Table 5-3**). For both RMZs and PIPs, mean live tree density was highest in the Willapa 2 block. Basal area was comparable among the three Willapa blocks, but lower in the Olympics and South Cascades blocks. The Willapa 3 and South Cascades block RMZs had the lowest mean density and largest QMDs. Stand density in both RMZs and PIPs was much lower in the South Cascades block than in other blocks. Live basal area was also lower in the South Cascades block, although the difference was not as great because the mean diameter of trees in the South Cascades block was higher than in most other blocks. Conifers contributed less to total basal area in South Cascades than in the other blocks.

Table 5-3. Initial live density, basal area, quadratic mean diameter and percent conifer basal area for RMZs and PIPs. Values are site averages within blocks.

Block	Live Tree Density trees/ac (trees/ha)	Live Tree Basal Area ft^2/ac (m^2/ha)	Live Tree QMD inches (cm)	% Conifer Basal Area
		RMZs		
OLYM	210.9 (521.0)	182.8 (42.0)	12.9 (32.7)	87.4
CASC	111.7 (276.1)	158.5 (36.4)	16.6 (42.1)	53.6
WIL1	231.1 (571.0)	233.2 (53.6)	13.9 (35.3)	96.8
WIL2	273.9 (676.9)	226.2 (51.9)	12.7 (32.2)	92.1
WIL3	178.6 (441.3)	235.2 (54.0)	16.4 (41.5)	89.1
		PIPs		
OLYM	173.4 (428.5)	202.3 (46.4)	14.7 (37.4)	99.4
CASC	102.7 (253.7)	141.6 (32.5)	15.5 (39.4)	61.5
WIL1	222.4 (549.6)	241.9 (55.5)	14.3 (36.3)	97.4
WIL2	247.8 (612.3)	248.8 (57.1)	13.5 (34.4)	98.1

Initial stand structure also varied within blocks. Among RMZs, site-to-site variation in basal area was high in the Willapa 1 and South Cascade blocks and low in the Olympic block. Among PIPS, there was relatively little variation in initial basal area among sites in the Olympic block compared to the other blocks (**Figure 5-1**).

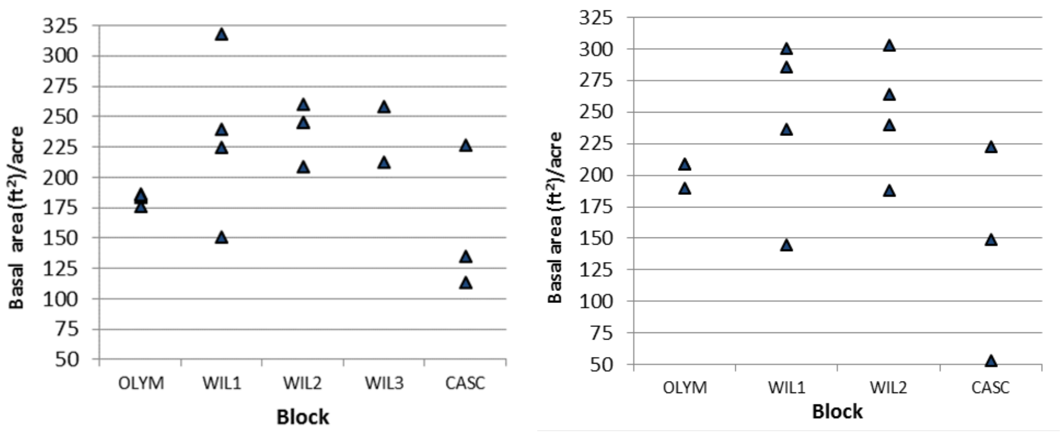

Figure 5-1. Variation in initial live basal area/ac among sites within blocks for RMZs (left panel) and PIPs (right panel).

We observed variability among blocks in the initial density and characteristics of dead trees (**Table 5-4; Appendix Tables 5-A-4 and 5-A-5**). Dead trees were most abundant in the South Cascades block and least abundant in the Olympic block. Conifers contributed much less to the basal area of dead trees in the South Cascades block than elsewhere, paralleling the pattern for live conifer basal area.

Table 5-4. Variation among blocks in initial (2007) dead tree density, basal area, quadratic mean diameter and percent conifer basal area for RMZs and PIPs. Values are site averages within each block. Metric units are shown in parentheses.

Block	Dead Tree Density in trees/ac (trees/ha)	Dead Basal Area in ft^2/ac (m2/ha)	Dead QMD in in (cm)	% Conifer Basal Area
		RMZs		
OLYM	16.6 (41.1)	9.1 (2.1)	9.4 (23.8)	83.2
CASC	41.9 (103.5)	104.8 (24.1)	19.8 (50.4)	49.9
WIL1	28.6 (70.7)	18.9 (4.3)	9.6 (24.4)	89.8
WIL2	34.2 (84.5)	23.4 (5.4)	9.2 (23.4)	94.4
WIL3	32.7 (80.9)	43.8 (10.1)	11.5 (29.1)	89.9
		PIPs		
OLYM	18.7 (46.1)	9.7 (2.2)	9.7 (24.6)	99.4%
CASC	34.9 (86.2)	55.5 (12.8)	13.2 (33.5)	32.4%
WIL1	25.2 (62.3)	13.2 (3.0)	9.8 (25.0)	94.9%
WIL2	21.0 (51.9)	9.0 (2.1)	8.8 (22.4)	89.0%

5-5.1.2. Pre-Harvest Tree Mortality Rates

During the pre-harvest period, 838 of 9,938 total live trees died (8.4%). Annual mortality rates pre-harvest (2007–2008) varied substantially among sites (**Appendix Table 5-A-6**). In RMZs, annual mortality rates ranged from 0% to 39.8% of live stems (average of 8.6%) and from 0% to 38.9% of live basal area (average of 8.0%). Mean annual mortality rates in PIPs were somewhat higher and more variable, ranging from 0% to 50.9% of live stems (average of 12.1%) and from % to 46.1% of live basal area (average of 11.2%).

Among RMZs, the vast majority of sites experienced little (<10%/yr) or no mortality (**Figure 5-2**). Within PIPs, a greater proportion of sites (35%) experienced moderate rates of mortality (>10–20%/yr for both stems and basal area). Few sites had mortality rates >20%/yr for either mortality metric.

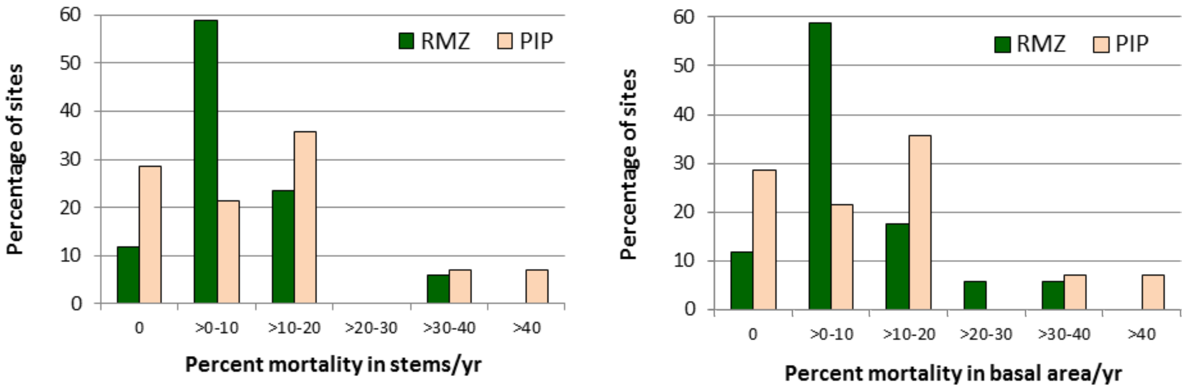

Figure 5-2. Distributions of mean pre-harvest mortality rates among sites, expressed as a percentage of initial live stem count (left panel) and live basal area (right panel) per year.

Among geographic locations, mortality in % of basal area/yr was notably higher in the two coastal blocks (means of 21.5% in Willapa 1 and 9.7% in Willapa 2; **Figure 5-3**).

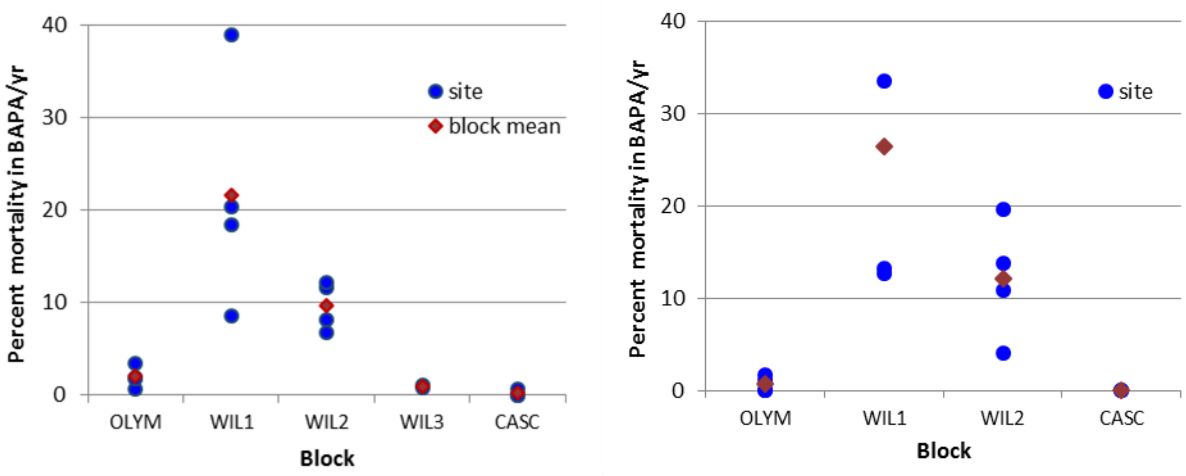

Figure 5-3. Pre-harvest mortality rates (% of basal area/yr) for RMZs (left panel) and PIPs (right panel) by block. Values are site means.

Among plots, the frequency distributions of percent mortality were similar for the Olympic, South Cascades and Willapa 3 blocks (**Figure 5-4**). Most plots had no mortality and many others lost <10% of initial basal area. In contrast, fewer (~30%) of the RMZ plots in the Willapa 1 and 2 blocks had no mortality and some plots experienced over 50% mortality. The pattern was generally similar for plots in PIPs, although rates did not exceed 50% (**Figure 5-4**).

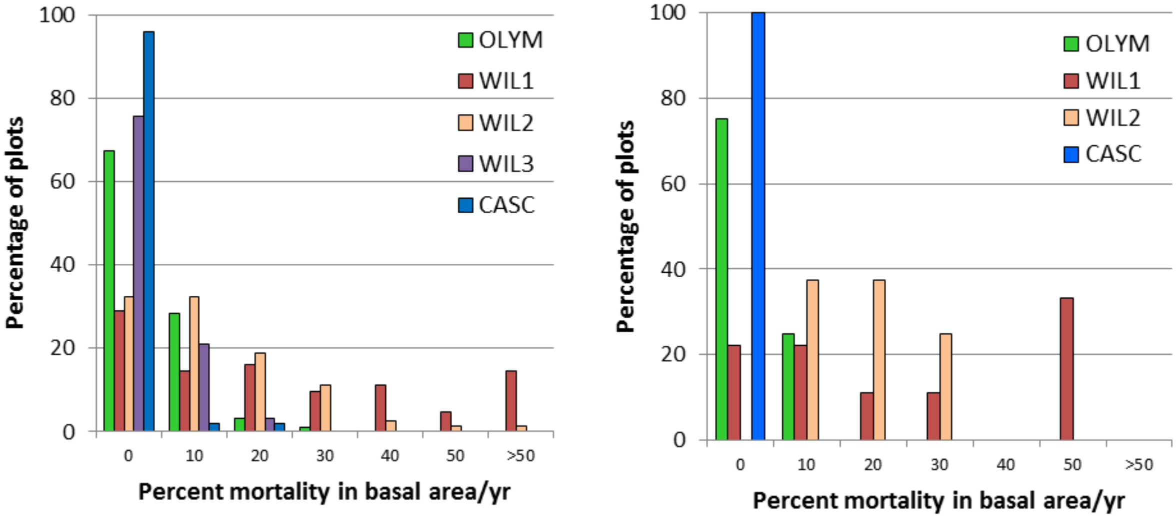

Figure 5-4. Frequency distributions of pre-harvest tree mortality rates (% of basal area/yr) in RMZ plot (left panel) and PIP plot (right panel) by block.

5-5.1.3. Pre-Harvest Change in Stand Structure

Most sites experienced reductions in density and basal area during the pre-harvest period in response to mortality. In the RMZs, density and basal area decreased at 15 of 17 sites. Mean density declined from 209.2 trees/ac (516.9 trees/ha) in 2007 to 189.9 trees/ac (469.2 trees/ha) in 2008. Mean basal area decreased from 206.8 ft^2/ac (47.6 m^2/ha) in 2007 to 189.8 ft^2/ac (43.7 m^2/ha) in 2008. In the PIPs, density and basal area decreased at 10 of 14 sites. Mean density declined from 193.5 trees/ac (478.1 trees/ha) in 2007 to 166.0 trees/ac (410.2 trees/ha) in 2008. Mean basal area declined from 213.9 ft^2/ac (49.2 m^2/ha) in 2007 to 186.2 ft^2/ac (42.4 m^2/ha) in 2008.

Differences among blocks in the degree to which stand structure changed pre-harvest were largely attributable to mortality that occurred during the December 2007 windstorm. Notably, basal area decreased substantially in many sites in the Willapa 1 and Willapa 2 blocks, but showed little change in the Olympic, Willapa 3 and South Cascades blocks (**Figure 5-5**). In the Olympic block, which experienced little wind, pre-harvest mortality rates were low and the distribution of basal area among RMZ plots showed little change (**Figure 5-6**, left panel). In the Willapa 1 block, extensive wind-induced mortality reduced the basal area in many plots, shifting the distribution to the left (**Figure 5-6**, right panel).

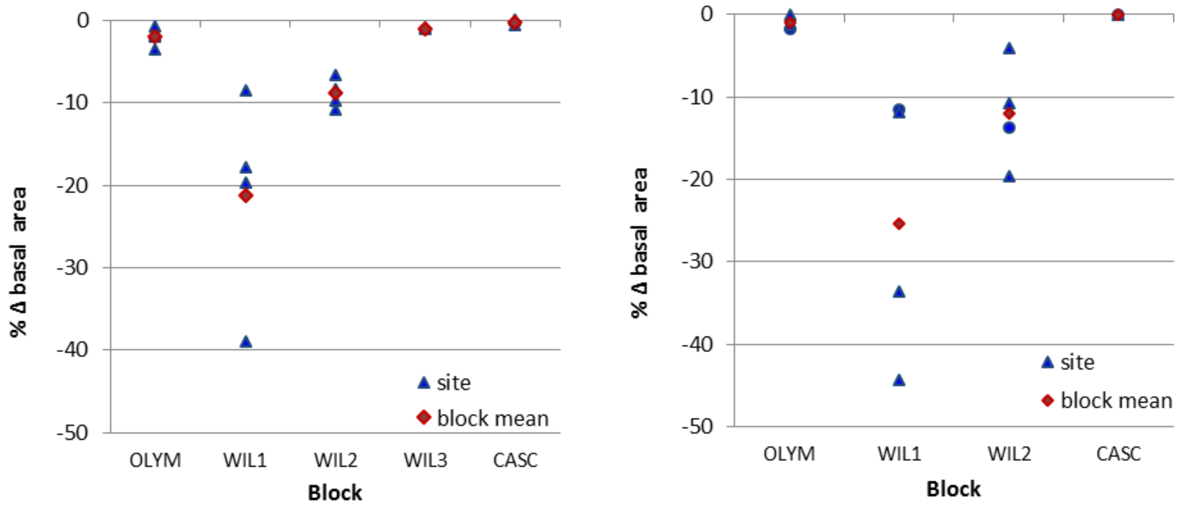

Figure 5-5. Percent change in live basal area during the pre-harvest period for RMZs (left panel) and PIPs (right panel) by block. Values are site means.

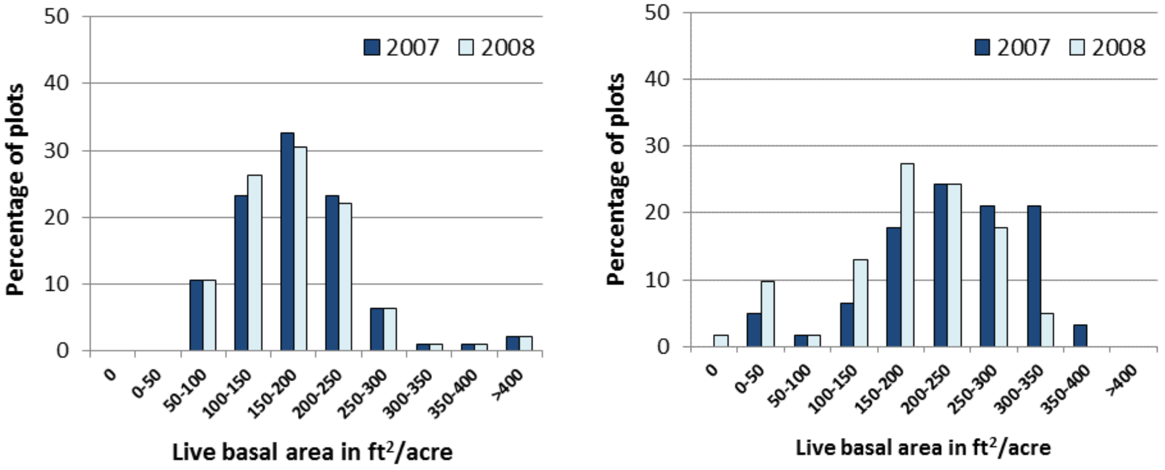

Figure 5-6. Changes in distribution of live basal area in RMZ plots during the pre-harvest period for the Olympic (left panel) and Willapa 1 (right panel) blocks.

There was no consistent change in density of dead standing trees during the pre-harvest period. In the RMZs, density of dead trees decreased at five sites, increased at seven, and showed no change at five. Basal area of dead trees increased at 10 of 17 sites, but the magnitude of increase was small (mean of 1.0 ft^2/ac [0.2 m^2/ha]). In the PIPs, density of dead trees decreased at three sites, increased at three, and remained the same at eight. Basal area decreased to a small degree at 13 of 14 sites (mean of 3.6 ft^2/ac [0.8 m^2/ha]).

5-5.2. POST-HARVEST TREE MORTALITY AND STAND STRUCTURE

5-5.2.1. Post-Harvest Tree Mortality

During the first two years following harvest, 1,129 of 6,981 live trees died (16.2%). There was large variation in mortality among sites during the post-harvest period (**Appendix Table 5-A-7**). In RMZs, annual rates of mortality ranged from 2.3% to 34.2% of stems/yr and from 1.7% to 38.5% of basal area/yr (**Figure 5-7**). Variability in mortality by basal area was greatest among the FPB sites (2.3–38.5%/yr), lower in the reference sites (1.8–20.5%/yr) and least variable in the 100% treatment sites (1.7–15.2% /yr). In PIPs, annual mortality ranged from 1.0% to 36.4% of stems/yr and from 0.5% to 24.8% of basal area/yr (**Figure 5-7**). Variability among sites was lowest in the reference, greater in the FPB treatment, and highest in the 100% treatment. In general, mortality rates in the Willapa 1, Willapa 2 and Olympic blocks were higher than in the South Cascade and Willapa 3 for both RMZs and PIPs.

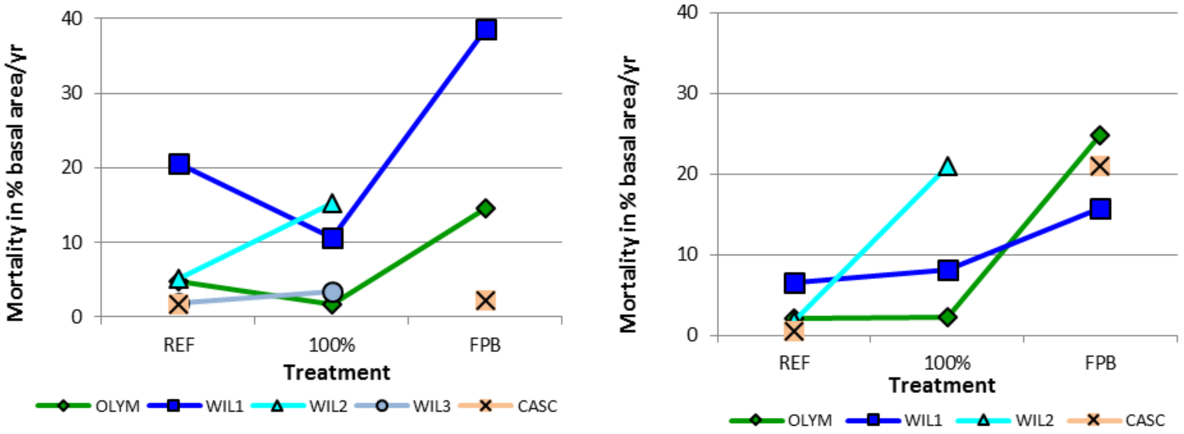

Figure 5-7. Variation in post-harvest mortality rates (% of basal area/yr) among treatments within blocks for RMZs (left panel) and PIPs (right panel).

5-5.2.2. Experimental Treatment Contrasts

Post-harvest mortality rates differed among experimental treatments (**Table 5-5**). In the RMZs, annual post-harvest mortality rates in the FPB treatment were more than twice those of the reference and 100% treatments (**Table 5-5**). The FPB-REF contrasts were significant for both % of stems and % of basal area, whereas the FPB-100% contrasts were equivocal. In the PIPs, annual post-harvest mortality rates were significantly greater in the 100% and FPB treatments than in the reference sites (four times and eight times higher, respectively).

Table 5-5. Mean annual post-harvest mortality rates for RMZ and PIP plots as percent of stems and percent of basal area by experimental treatment with P-values for treatment contrasts. Significant results (P <0.10) are in bold font.

Plot Type	Metric	REF	100%	FPB	Treatment Contrast P-values		
		Mean[1] (SE)	Mean[1] (SE)	Mean[1] (SE)	100%-REF	FPB-REF	FPB-100%
RMZ[2]	% stems/yr	6.3 (1.8)	9.9 (2.4)	20.8 (7.7)	0.157	**0.024**	0.105
	% basal area/yr	3.9 (1.7)	7.8 (2.6)	18.7 (8.2)	0.106	**0.013**	**0.092**
PIP[3]	% stems/yr	3.9 (1.8)	12.3 (3.7)	31.0 (6.7)	**0.071**	**0.004**	**0.039**
	% basal area/yr	2.4 (1.3)	10.2 (3.2)	21.0 (5.1)	**0.054**	**0.008**	0.110

[1] Mean mortality rates are GLIMMIX mixed-effect model estimates.

[2] Denominator degrees of freedom for RMZ contrasts ranged from 6.2 to 6.8 on a sample size of 13.

[3] Denominator degrees of freedom for PIP contrasts ranged from 6.8 to 7.8 on a sample size of 24.

The treatment effect is also evident at the plot scale. Most reference site RMZ plots had <10% mortality (**Figure 5-8**). The 100% treatment distribution was similar, except for a slightly greater percentage of plots with higher mortality rates. In contrast, the FPB treatment had fewer plots with <10% mortality and more plots with mortality >20%. In the PIPs, all reference site plots had mortality rates <10%, whereas the majority of plots in the 100% treatment had rates >0% but <10% and the FPB treatment had more plots with higher mortality (>10–40%).

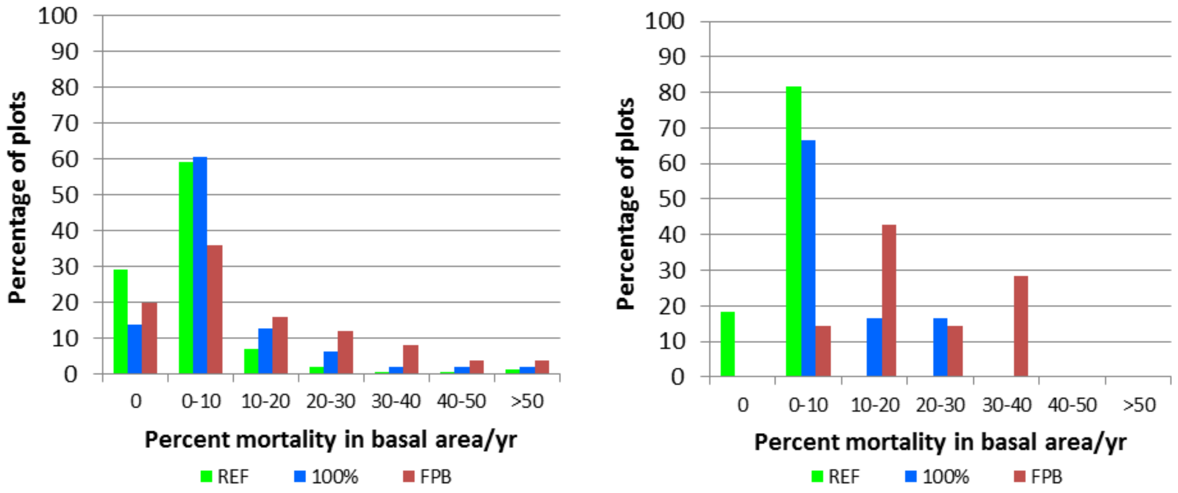

Figure 5-8. Frequency distributions of post-harvest tree mortality rates (% of basal area/yr) by treatment for plots in the RMZs (left panel) and PIPs (right panel).

5-5.2.3. Changes in Stand Structure during the Post-Harvest Period

Live tree density and basal area declined at nearly all reference, 100%, and FPB treatment sites during the post-harvest period. In RMZ plots, density declined at all sites and basal area at all but one site. In PIP plots, density declined at all sites and basal area at seven of 11 sites (**Figure 5-9**).

We report site mean values for change in live density and live basal area following harvest in **Appendix Table 5-A-8** (RMZs) and **Appendix Table 5-A-9** (PIPs).

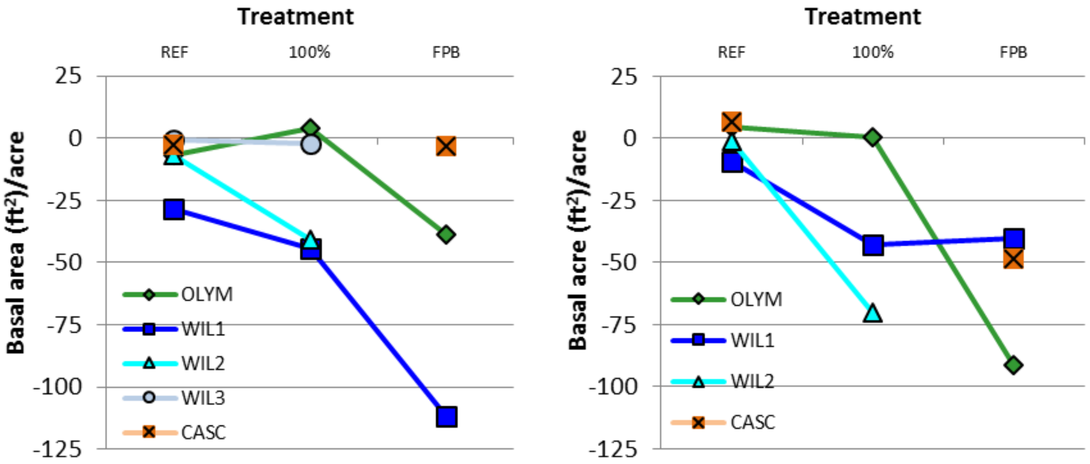

Figure 5-9. Post-harvest change in basal area (ft^2/ac) for RMZs (left panel) and PIPs (right panel) by experimental treatment and block.

5-5.2.3.a Experimental Treatment Contrasts

In the RMZs, the proportional changes in stem count (%Δ in stems) and basal area (%Δ in basal area) were similar for the reference and 100% treatment (**Table 5-6**). In contrast, the magnitude of decrease was significantly greater in the FPB treatment than in either the reference or 100% treatment. The pattern was similar in the PIPs, with the exception that there was a greater decrease in % basal area in the 100% treatment than in the reference sites (**Table 5-6**).

Table 5-6. Mean proportional change in stem count and basal area by experimental treatment with P-values for treatment contrasts. Significant results (P <0.10) are in bold font.

Plot Type	Metric	REF	100%	FPB	Treatment Contrast P-values		
		Mean[1] (SE)	Mean[1] (SE)	Mean[1] (SE)	100%-REF	FPB-REF	FPB-100%
RMZ[2]	%Δ in stems	−11.8 (5.3)	−3.8 (5.9)	−29.6 (6.5)	0.723	**0.017**	**0.042**
	%Δ in basal area	−6.9 (5.4)	−6.7 (6.0)	−24.4 (6.7)	0.983	**0.024**	**0.034**
PIP[3]	%Δ in stems	−7.3 (6.3)	−19.8 (8.1)	−48.7 (8.1)	0.255	**0.004**	**0.035**
	%Δ in basal area	0.0 (4.1)	−17.1 (5.3)	−37.0 (5.3)	**0.029**	**<0.001**	**0.029**

[1] Mean mortality rates are GLIMMIX mixed-effect model estimates.
[2] Denominator degrees of freedom for RMZ contrasts ranged from 6.2 to 6.8 on a sample size of 13.
[3] Denominator degrees of freedom for PIP contrasts ranged from 6.8 to 7.8 on a sample size of 24.

5-5.2.3.b Stand Structure Two Years Post-Harvest

Variation in initial stand structure, buffer treatments, and post-harvest mortality produced a wide range of stand structures two years post-harvest (**Figure 5-10**). In the RMZs, post-harvest basal area tended to be highest in reference and 100% treatments. However, basal area in the Willapa 1 reference site was lower than other reference sites, reflecting mortality from the intense pre-harvest windstorm. The FPB RMZs had lower basal area than nearly all reference and 100% RMZs. In the PIPs, post-harvest basal area tended to be highest in the reference and 100% treatments, and lower in the FPB treatment.

The distribution of plot-scale stand structures diverged among experimental treatments two years post-harvest (**Figure 5-11**). Live density distributions of RMZ and PIP plots were similar in the reference and 100% treatments, with most values between 120 and 240 trees/ac. For the FP treatment, the combined effect of post-harvest mortality in the buffered portions of the network and clearcut harvest in the remainder produced a distribution where most plots had <160 trees/ac in the RMZs and <80 trees/ac in the PIPs.

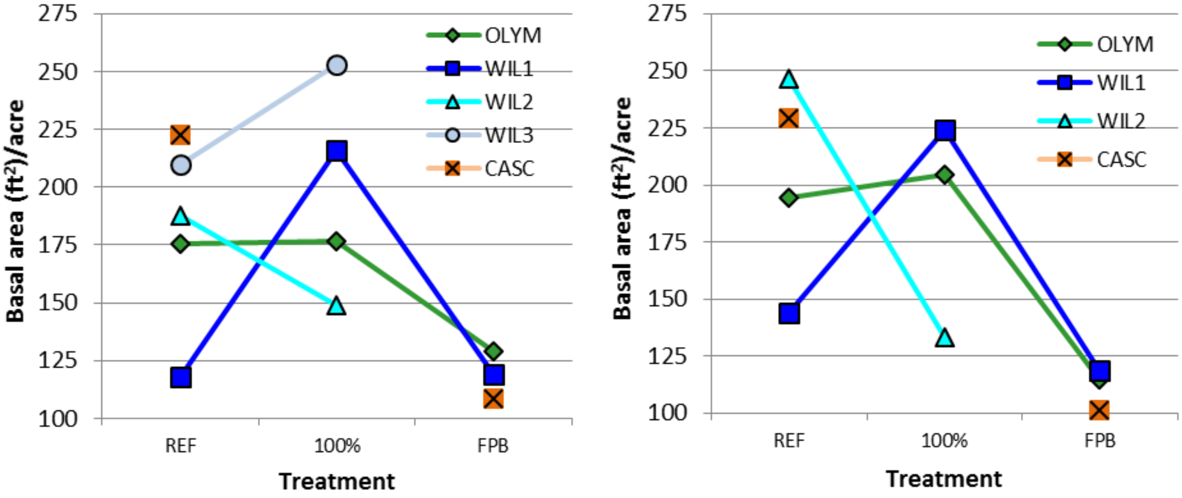

Figure 5-10. Basal area (ft^2/ac) the second summer post-harvest by experimental treatment for RMZs (left panel) and PIPs (right panel).

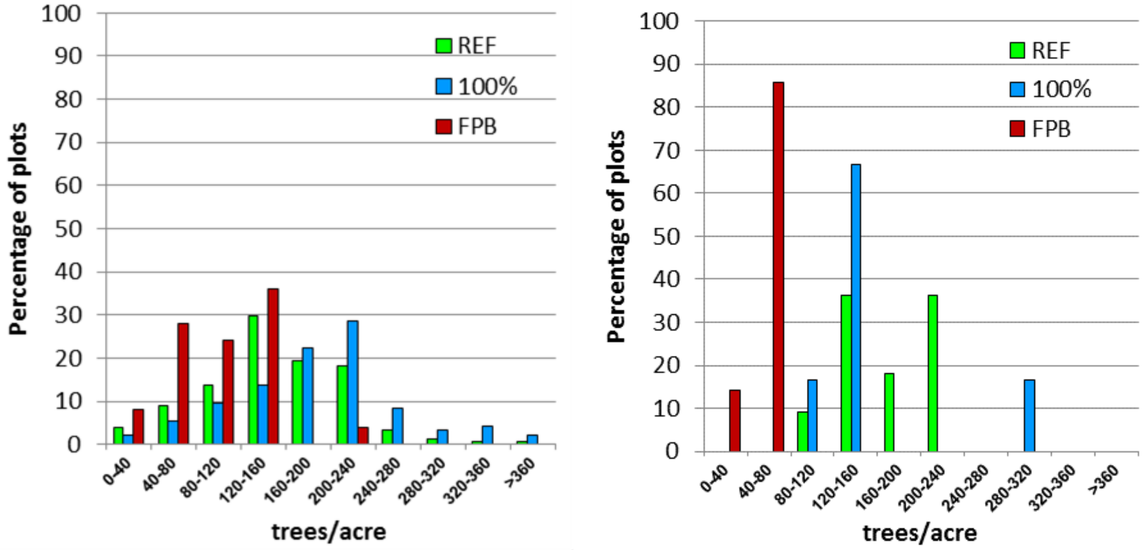

Figure 5-11. Distributions of RMZ (left panel) and PIP (right panel) plot values for live tree density (trees/ac) by experimental treatment, two years after harvest.

Among RMZ and PIP plots with buffers (FPB and 100% treatments), density and basal area varied widely two years post-harvest (**Figure 5-12**). In RMZ buffers, densities ranged from <40 to >360 trees/ac with the majority of values between 120 and 240 trees/ac. The majority of PIP buffers had densities <160 trees/ac and basal areas <200 ft^2/ac.

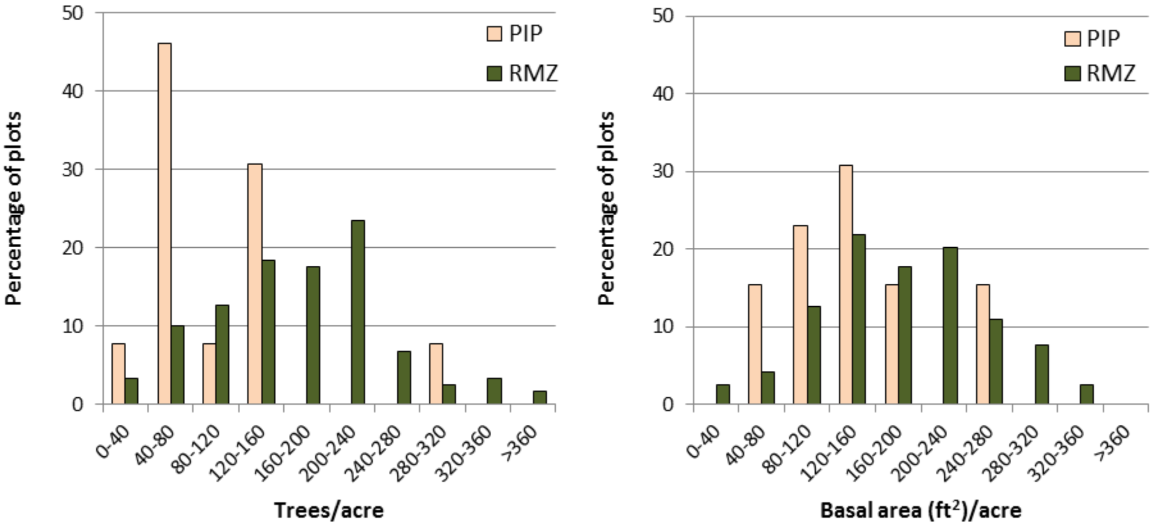

Figure 5-12. Distributions plot values for live density (left panel) and live basal area/ac (right panel) for RMZ and PIP buffer plots (FPB and 100% treatments), two years after harvest.

5-5.3. MORTALITY AGENTS AND CHARACTERISTICS OF DEAD TREES

Pre-harvest, mechanical damage attributable to wind was the dominant cause of mortality, accounting for 98% of stems that died (**Table 5-7**). The remainder was attributable to other mortality agents including suppression, damage from insect and diseases, erosion and slope failure, and unidentified causes. Both the proportion of overstory trees and the mean diameter were greater for trees killed by mechanical damage than for trees killed by other agents. Western redcedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*) had the highest pre-harvest mortality rates (~12%; **Table 5-8**).

Table 5-7. Characteristics of pre-harvest mortality, by mortality agent.

Mortality Agent	% Total Mortality (Stem Count)	Mean Diameter inches (cm)	Percentage of Overstory Trees
Mechanical	98.0	12.1 (30.7)	73.0
Other	2.0	9.7 (24.6)	45.5

Table 5-8. Proportion of live trees that died during the pre-harvest period by species.

Species	Scientific Name	Initial Live Count	Mortality Count	% Mortality
Western redcedar	*Thuja plicata*	401	49	12.2
Western hemlock	*Tsuga heterophylla*	5827	677	11.6
True fir[1]	*Abies spp.*	153	11	7.2
Sitka spruce	*Picea sitchensis*	242	13	5.4
Red alder	*Alnus rubra*	1041	37	3.6
Cascara	*Rhamnus purshiana*	137	4	2.9
Douglas-fir	*Pseudotsuga menziesii*	2,082	46	2.2
Willow	*Salix spp.*	38	0	0.0

[1] True fir includes grand fir (*Abies grandis*) and Pacific silver fir (*Abies amabilis*).

Following harvest, mechanical damage was also the dominant mortality agent, accounting for ≥70% of trees that died (**Figure 5-13**). The "other" category accounted for ~20% of mortality in the 100% treatments and references, while a fire of anthropogenic origin at one site contributed about 12% of the FPB mortality. Mortality of buffer trees from felling and yarding activity associated with the adjacent harvest was minimal.

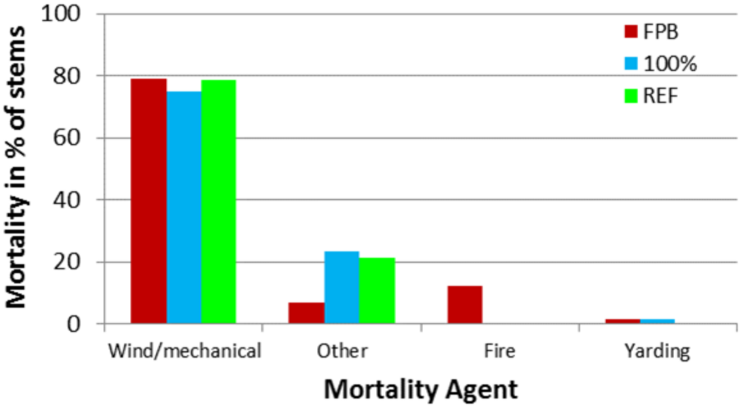

Figure 5-13. Proportion of post-harvest mortality, by mortality agent and treatment.

In the reference sites, trees that died post-harvest had smaller diameters and fewer came from the overstory crown class than those in the other treatments (**Table 5-9**). In contrast, in the 100% and FPB treatments, ~70% of trees that died were from the overstory crown class and their mean diameters were 1 and 2 in (2.5 and 5.1 cm) greater than those in the reference sites, respectively. Trees that died due to wind/mechanical damage or fire during the post-harvest period had larger mean diameters and a higher proportion of overstory trees than did trees dying from other causes (**Table 5-10**).

Table 5-9. Mean diameter of trees that died during the post-harvest period and proportion of total mortality consisting of trees from the overstory crown class by treatment.

Treatment	Mean Diameter inches (cm)	Percentage of Total Mortality from Overstory Crown Class
FPB	12.2 (31.0)	73.3
100%	11.2 (28.5)	69.1
REF	10.3 (26.2)	59.0

Table 5-10. Mean diameter of trees that died during the post-harvest period and proportion of total mortality from the overstory crown class by mortality agent.

Mortality Agent	Mean Diameter inches (cm)	Percentage of Total Mortality from Overstory Crown Class
Fire	12.4 (31.5)	88.9
Wind/mechanical	11.9 (30.2)	71.9
Other	9.3 (23.6)	41.7
Yarding	8.9 (22.6)	41.7

In the FPB treatments, post-harvest mortality was ~45% for western red cedar, western hemlock and cascara, slightly lower for Douglas-fir, and lowest for Sitka spruce and red alder (~25%). The pattern was similar for the 100% treatment, but ~20 percentage points lower. In the reference sites, mortality rates for all species were ~15% or less (**Figure 5-14**).

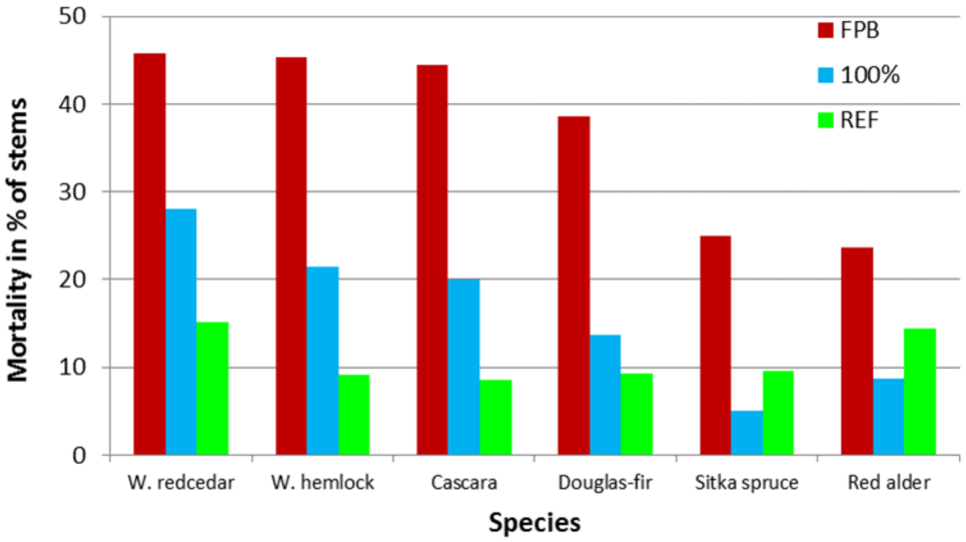

Figure 5-14. Proportion of live trees that died (% of stems) during the post-harvest period by species and treatment. Species that did not occur in all three treatment groups are not shown.

5-6. DISCUSSION

5-6.1. TREE MORTALITY RATES

The short-term mortality rates we documented were higher than long-term rates reported in the literature for both young and mature forests in the Pacific Northwest. Mean post-harvest mortality in buffered RMZs in the FPB and 100% treatments (7.7 and 18.2% of stems/yr, respectively) were considerably higher than long-term rates (1.0–5.3% of stems/yr) reported for young forests in the Cascade Range of Oregon (Lutz and Halpern 2006) and western Washington and Oregon (1.0−1.7% of stems/yr; Pollock and Beechie 2014). Our rates were also higher than that reported for old-growth forests in the Washington Cascades (0.75%; Franklin and DeBell 1988). However, they were consistent with the higher short-term mortality rates for newly established riparian buffers in western Washington reported in previous studies (Mobbs and Jones 1995; Grizzel and Wolff 1998; Grizzel *et al.* 2000; Liquori 2006; Schuett-Hames *et al.* 2012).

Cumulative post-harvest mortality rates in our FPB treatment RMZs were within the range reported for 50-ft (15.2-m) no-harvest buffers on Type N streams in western Washington (Jackson *et al.* 2007; Schuett-Hames *et al.* 2012) and were similar to rates for variable-width (typically narrower) buffers established under previous rules (Grizzel and Wolff 1998; **Table 5-11**). Both this study and Schuett-Hames and colleagues (2012) documented statistically significant increases in mortality in FPB RMZs compared to reference sites. Cumulative mortality for the 100% treatment RMZs in this study was lower than the values for 50-ft (15.2-m) buffers in previous studies. Mortality in the PIP buffers in this study was higher than mortality for PIP buffers reported in Schuett-Hames and colleagues (2012; **Table 5-11**).

Table 5-11. Comparison of cumulative post-harvest tree mortality reported in studies of RMZ buffers on Type N streams in western Washington.

Study	Years After Harvest	RMZs		PIPs	
		Cumulative Mean Mortality (% of stems)	Range (%)	Cumulative Mean Mortality (% of stems)	Range (%)
FPB treatment this study	2	30	8–52	48	14–74
100% treatment this study	2	17	5–29	20	3–50
Schuett-Hames *et al.* (2012)	3[1]	20.5	1–69	35	12–63
Jackson *et al.* (2007)	2	47	33–64	-	-
Grizzel and Wolff (1998)	1–3	33	2–92	-	-

[1] Data from first three years after harvest.

The higher annual mortality rates in our study reflect short-term response to episodic wind disturbance following harvest, in contrast to long-term average rates that integrate low rates of chronic background mortality with elevated rates punctuated by infrequent or episodic disturbance-related mortality (e.g., Franklin and DeBell 1988). Mechanical damage associated with wind was the most prominent cause of mortality during both the pre- and post-harvest periods, although a fire during the post-harvest period caused substantial mortality at one site. Elevated mortality from wind has been observed consistently across the maritime areas of the Pacific Northwest in newly established riparian buffers (Mobbs and Jones 1995; Grizzel and Wolff 1998; Grizzel et. al. 2000; Liquori 2006; Martin and Grotefendt 2007; Jackson *et al*. 2007; Bahuguna *et al.* 2010; Schuett-Hames *et al.* 2012), along harvest unit boundaries, and in variable-retention harvest units (Rollerson *et al.* 2009; Urgenson *et al.* 2013).

We observed spatial and temporal variation in mortality associated with the magnitude and frequency of windstorms. Winds were stronger and more frequent at the two coastal weather stations (Astoria and Hoquiam) than inland (Portland) in both the pre- and post-harvest periods (**Table 5-12**).

Table 5-12. Number of days with storm-force and hurricane-force winds (five-second wind speed) by weather station.

Station	Pre-harvest Period		Post-harvest Period	
	Storm-force Days	Hurricane-force Days	Storm-force Days	Hurricane-force Days
Astoria	4	2	7	0
Hoquiam	4	1	10	0
Portland	0	0	0	0

During the pre-harvest period, both Astoria and Hoquiam recorded hurricane-force winds for one or more days and storm-force winds for four additional days, whereas winds further east at Portland did not reach storm-force during the same period. The highest intensity windstorm occurred in December 2007, producing hurricane-force winds in coastal southwestern Washington and northern Oregon peaking at 94 mph (42.0 m/s) in Astoria; however, winds were not as strong further east, peaking at 41 mph (18.3 m/s) in Portland on the same date. The hurricane-force winds in the pre-harvest period resulted in extensive mortality in unharvested riparian stands embedded in continuous second-growth forest at the coastal sites, while mortality was lower further inland.

In the post-harvest period, both the Astoria and Hoquiam stations had multiple days with storm-force winds, while Portland recorded no storm-force winds during the post-harvest period. This regional variation in wind frequency and magnitude affected experimental treatments differently. Mortality rates among RMZs tended to be highest in the newly established RMZ and PIP buffers in the FPB and 100% treatments in the Willapa 1 and Willapa 2 blocks that were exposed to storm-force winds. The combination of newly established buffers and storm-force winds resulted in the greatest contrast between riparian buffers and reference sites. This was also observed in the Westside Type N Buffer Characteristics, Integrity and Function study during the first three years after harvest (Schuett-Hames *et al.* 2012). In our study, mortality rates were low in the absence of storm-force winds, except for the one site that burned.

We expected substantial mortality in newly exposed FPB buffers based on previous studies. However, the significant difference between higher mortality rates in the FPB treatment RMZs and the lower rates in the 100% treatment RMZs was unexpected, because both had 50-ft (15.2-m) wide buffers and differed only in the percentage of stream length buffered. It is unclear why mortality was lower in the 100% treatment or whether this difference will persist over time. The small size of the PIP buffers and their increased exposure to winds following clearcut harvest may explain the higher rates of mortality in the FPB and 100% treatments than in the reference site PIPs surrounded by unharvested forest. Other research suggests that vulnerability to wind damage is greater for small patches of leave trees located higher on hill slopes and those with large fetch distances across adjacent clearcut land (Ruel *et al.* 2001; Rollerson *et al.* 2009). One possible explanation for lower mortality in the 100% treatment PIPs compared to the FPB treatment PIPs is that the former were connected to RMZ buffers, whereas the latter were separated from the downstream buffer and surrounded by clearcut areas.

As expected, we observed substantial variability in mortality among and within blocks and sites. The combination of wind patterns, site factors and stand characteristics result in varying susceptibility to windthrow at local and landscape scales, resulting in mosaics of stand conditions as forests respond to windstorms over time (Nowacki and Kramer 1998; Sinton *et al.* 2000; Kramer *et al.* 2001; Harcombe *et al*. 2004). The variation in mortality among blocks appears related to regional differences in exposure to high winds associated with low pressure storms originating in the Pacific Ocean. Other examples of landscape-scale patterns in mortality associated with recurring windstorms produced by large-scale weather systems have been reported along the coast of SE Alaska due to recurring cyclonic storms (southeast gales) from the North Pacific (Harris 1999; Kramer *et al.* 2001), and where strong pressure gradients between interior and coastal areas result in strong easterly winds, such as in the Columbia Gorge in

Washington and Oregon (Sinton *et al.* 2000) or the Taku and Stikine River valleys in southeast Alaska (Harris 1999).

Variability in mortality among and within sites was likely due to local topographic effects and physical site conditions. Complex interactions with topographic features can cause local increases in wind speed, increasing the likelihood of windthrow at some sites (Moore 1977; Nowacki and Kramer 1998; Harris 1999; Sinton *et al.* 2000; Kramer *et al.* 2001; Mitchell *et al.* 2001; Ruel *et al.* 2001; Harcombe *et al.* 2004) as do harvest units that create openings with edges exposed to winds (Moore 1977; Sinton *et al.* 2000; Kramer *et al.* 2001; Mitchell *et al.* 2001; Liquori 2006; Rollerson *et al.* 2009; Urgenson *et al.* 2013). Susceptibility to windthrow can also be affected by the characteristics of trees (e.g., height-diameter ratio, rooting structure), soil depth and composition, soil moisture and groundwater saturation (Moore 1977; Harris 1999; Scott and Mitchell 2005; Rollerson *et al.* 2009). However, the data from this study were not conducive to an evaluation of topographic or other local effects on buffer tree mortality.

5-6.2. CHANGE IN RIPARIAN BUFFER STANDS

Patchy mortality in the riparian buffers caused substantial plot-scale variation in stand structure (reduced density and basal area) that sets the stage for divergent patterns of stand development in the future. Approximately 75% of RMZ buffer plots and 40% of PIP plots in the FPB and 100% treatments had densities exceeding 120 trees/ac (296.5 trees/ha) two years after harvest. These densities are typical of second-growth riparian stands in western Oregon and Washington (Pollock and Beechie 2014). We do not expect future harvest in the FPB and 100% treatment buffers, which we expect will develop as even-age stands, self-thinning to lower densities during the stem exclusion stage of development (Liquori 2000; Pollock and Beechie 2014).

The trajectories for the remaining 25% of RMZ plots and 60% of PIP plots with densities <120 trees/ac (296.5 trees/ha) are uncertain. It is likely some will deviate from the conventional model and develop along an alternative open canopy pathway (Donato *et al.* 2012). Stands at the lower end of the range, as low as 40 trees/ac (138.8 trees/ha), are at densities similar to those created by two-age shelterwood harvest strategies applied to upland stands (Curtis *et al.* 2004). There is no requirement for landowners to plant trees following wind disturbance in the riparian buffers, so natural successional processes will determine whether conifers reestablish. Stands affected by wind disturbance are characterized by spatial heterogeneity, with clumps of trees intermixed with gaps, the latter characterized by increased light and nutrient availability where rates of regeneration may be enhanced (Edmonds *et al.* 2005). Gaps created by windthrow are characterized by pits and mounds of exposed soil, and abundant down wood, which provide suitable sites for tree regeneration (Edmonds *et al*. 2005). We expect that these stands should develop into multi-aged conifer stands if natural conifer regeneration is successful (Agee 1993). Shade-tolerant species such as western hemlock may dominate regeneration in cooler, moister locations (Lutz and Halpern 2006). Although initially open in structure, multi-age stands should provide shade and wood-recruitment potential over the long term as the understory cohort develops. However, conifer regeneration may not be successful if there is not an adequate source of seeds (Beach and Halpern 2001) or if understory shrub cover is dense. If broadleaf tree regeneration occurs, a multi-age stand of mixed composition may result, with gaps filled by an understory of broadleaf trees such as red alder in moist locations, which can increase aquatic productivity by providing litter inputs high in nitrogen. Shrubs may dominate in situations where there is an existing shrub understory and a conifer seed source is lacking. In these cases, an open stand condition may persist for decades, although the remnant conifer stand should eventually take on mature and old-growth structural characteristics (Donato *et al.* 2012).

A third scenario will play out in stream-adjacent stands in the clearcut (unbuffered) sections of the FP treatment. The Washington Forest Practices rules require reforestation at a minimum density of 190 trees/ac (470 trees/ha) in these areas. These stands can be harvested repeatedly over time, so we expect them to alternate between stand initiation, young forest, and subsequent harvest.

In addition to natural regeneration processes, additional mortality from wind and other agents will have an effect on future stand structure in riparian buffers. It is uncertain to what extent stands disturbed by wind during the first two years post-harvest will continue to experience wind damage over time. The outcome may depend upon their susceptibility to wind damage. In mature conifer forests in the Oregon Coast Range, small patches created by windthrow increased in size and severity over a 40-year period as trees along the patch edges and residual trees within the patch fell during subsequent windstorms (Harcombe *et al.* 2004). However, in other cases, leave trees in variable-retention harvest units appear to stabilize after an initial windthrow event, with little additional mortality from wind (Urgenson *et al.* 2013).

Changes in riparian management philosophy and practices over the last several decades are transforming stand structure and dynamics on perennial headwater streams on forestland in western Washington. Retention of buffers along a substantial portion of the perennial stream network is providing an opportunity for stands to develop to maturity without further management intervention. Our preliminary data from the first two years post-harvest indicate that the 50-ft (15.2-m) no-cut prescription is maintaining existing stand structure in about 75% of the RMZ plots and 40% of the PIP plots, while wind disturbance is causing substantial changes in the remaining plots. Thus, it appears that regional and local variation in susceptibility to windthrow will produce a mosaic of stand structures and developmental pathways. Further research and monitoring will be necessary to determine how the distribution of stand structures will change over time, and whether the resulting diversity of structure will achieve the resource protection goals of the Forest Practice HCP.

5-6.3. LIMITATIONS OF THE ANALYSES

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. Denominator degrees of freedom, which is an estimate of the number of independent observations used in estimating the population parameters, ranged from 6.2 to 10. Larger sample sizes would likely increase the precision of the parameter estimates and provide greater confidence in our interpretation of treatment contrasts.

The two-year post-harvest sampling period provides a snapshot of the response of the Type N buffers, but it is too short to adequately characterize effects of episodic disturbances on long-term rates of tree mortality and corresponding changes in stand structures. Stochastic disturbance processes such as wind, fire, insects, and disease are highly variable in space and time, thus more time is necessary to understand responses to a wider range of climatic and disturbance conditions. Monitoring of these stands is scheduled for another six years, providing data from which we can begin to assess these longer-term processes.

Our results demonstrate the importance of recognizing and addressing spatial and temporal variability in disturbance (and in stand conditions) when designing and interpreting the results of riparian prescription effectiveness studies. Had this study been limited to either coastal or inland locations, our interpretations would have differed. The broad geographic distribution of sites in this study resulted in a wide range of disturbance effects, providing an opportunity to document the context-dependence of these riparian prescriptions. Because disturbance processes such as wind and fire occur stochastically, increasing the spatial distribution of sites and the length of post-harvest monitoring increases the likelihood of obtaining a representative estimate of the variation in treatment response and helps to elucidate the causes of this variation.

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APPENDIX 5-A. DATA TABLES

Appendix Table 5-A-1. Maximum wind speed in miles per hour (mph) and meters per sec (m/s) at the Astoria, Hoquiam and Portland weather stations on days when winds reached storm- or hurricane-force (bolded) magnitude during the study period.

Date	Period	Astoria mph (m/s)	Hoquiam mph (m/s)	Portland mph (m/s)
2007-10-18	2007–2008	60 (26.8)	56 (25.0)	36 (16.1)
2007-11-12	2007–2008	63 (28.2)	71 (31.7)	46 (20.6)
2007-12-02	2007–2008	**77 (34.4)**	67 (30.0)	33 (14.8)
2007-12-03	2007–2008	**94 (42.0)**	No data[1]	41 (18.3)
2007-12-19	2007–2008	61 (27.3)	38 (17.0)	38 (17.0)
2008-02-06	2007–2008	56 (25.0)	46 (20.6)	44 (19.7)
2008-02-07	2007–2008	51 (22.8)	56 (25.0)	36 (16.1)
2008-12-12	2008–2009	60 (26.8)	58 (25.9)	35 (15.6)
2009-11-16	2009–2010	66 (29.5)	71 (31.7)	45 (20.1)
2009-11-18	2009–2010	49 (21.9)	59 (26.4)	37 (16.5)
2009-11-19	2009–2010	55 (24.6)	62 (27.7)	24 (10.7)
2009-11-22	2009–2010	58 (25.9)	60 (26.8)	51 (22.8)
2010-01-15	2009–2010	52 (23.2)	61 (27.3)	29 (13.0)
2010-01-18	2009–2010	60 (26.8)	62 (27.7)	51 (22.8)
2010-03-28	2009–2010	60 (26.8)	53 (23.7)	29 (13.0)
2010-03-29	2009–2010	66 (29.5)	60 (26.8)	44 (19.7)
2010-04-02	2009–2010	52 (23.2)	58 (25.9)	47 (21.0)
2010-05-19	2009–2010	53 (23.7)	60 (26.8)	45 (20.1)

Note: According to the National Weather Service descriptions, storm-force winds range from 55 to 74 mph and can topple shallow rooted trees; hurricane-force winds range from 74 to 122 mph and can uproot or snap trees (NOAA National Weather Service, 2013).

[1] Anemometer stopped operating, but we assumed that wind speed exceeded the 74 mph threshold for hurricane-force classification.

Appendix Table 5-A-2. Mean pre-harvest site values for live tree density (trees/ac), live tree basal area (ft^2/ac), and percentage of live conifer basal area in RMZs. Metric units (trees/ha, m^2/ha) are shown in parentheses.

Block	Treatment	Live Tree Density			Live Tree Basal Area			% Live Conifer Basal Area		
		2007	2008	Change	2007	2008	Change	2007	2008	Change
OLYM	REF	224.5 (554.7)	222.0 (548.6)	−2.6 (−6.4)	183.3 (42.2)	181.9 (41.8)	−1.4 (−0.3)	99.2	99.2	0.0
	100%	198.6 (490.7)	190.8 (471.5)	−7.8 (−19.3)	176.0 (40.5)	172.5 (39.7)	−3.5 (−0.8)	84.3	84.4	0.0
	FP	193.7 (478.6)	183.8 (454.2)	−10.0 (−24.7)	186.1 (42.8)	180.7 (41.6)	−5.3 (−1.2)	88.1	88.5	0.4
	0%	226.5 (559.7)	220.4 (544.6)	−6.1 (−15.1)	185.8 (42.7)	182.4 (42.0)	−3.4 (−0.8)	78.0	78.1	0.2
WIL1	REF	224.0 (553.5)	135.0 (333.6)	−89.0 (−219.9)	239.8 (55.2)	146.7 (33.7)	−93.1 (−21.4)	96.2	96.0	−0.2
	100%	323.7 (799.9)	262.7 (649.1)	−61.0 (−150.7)	317.8 (73.1)	260.6 (59.9)	−57.3 (−13.2)	99.2	99.4	0.2
	FP	180.5 (446.0)	168.0 (415.1)	−12.4 (−30.6)	224.8 (51.7)	208.5 (48.0)	−16.3 (−3.7)	100.0	100.0	0.0
	0%	196.2 (484.8)	175.9 (434.6)	−20.3 (−50.2)	150.6 (34.6)	129.8 (29.9)	−20.7 (−4.8)	91.8	91.2	−0.6
WIL2	REF1	216.7 (535.5)	188.9 (466.8)	−27.8 (−68.7)	245.4 (56.4)	219.5 (50.5)	−25.9 (−6.0)	99.2	99.1	−0.1
	REF2	219.0 (541.1)	186.4 (460.6)	−32.5 (−80.3)	190.2 (43.7)	169.2 (38.9)	−21.1 (−4.9)	90.2	90.2	0.0
	100%	274.6 (678.5)	250.5 (619.0)	−24.0 (−59.3)	208.9 (48.0)	189.9 (43.7)	−19.0 (−4.4)	83.0	83.1	0.2
	0%	385.5 (952.6)	357.6 (883.6)	−27.9 (−68.9)	260.4 (59.9)	244.3 (56.2)	−16.1 (−3.7)	95.9	95.9	−0.1
WIL3	REF	129.3 (319.5)	126.9 (313.6)	−2.4 (−5.9)	212.4 (48.9)	210.5 (48.4)	−1.9 (−0.4)	88.9	88.9	0.0
	100%	227.9 (563.1)	224.8 (555.5)	−3.1 (−7.7)	258.0 (59.3)	255.4 (58.7)	−2.5 (−0.6)	89.3	89.1	−0.1
CASC	REF	152.1 (375.8)	151.5 (374.4)	−0.6 (−1.5)	226.4 (52.1)	225.2 (51.8)	−1.2 (−0.3)	88.8	88.8	0.0
	FP	92.1 (227.6)	92.1 (227.6)	0.0	113.8 (26.2)	113.8 (26.2)	0.0	5.2	5.2	0.0
	0%	91.0 (224.9)	91.0 (224.9)	0.0	135.2 (31.1)	135.2 (31.1)	0.0	66.9	66.9	0.0
Mean		209.2 (516.9)	189.9 (469.2)	−19.3 (−47.7)	206.8 (47.6)	189.8 (43.7)	−17.0 (−3.9)	84.9	84.9	0.0

Appendix Table 5-A-3. Mean pre-harvest site values for live tree density (trees/ac), live tree basal area (ft^2/ac), and percentage of live conifer basal area in PIPs. Metric units (trees/ha, m^2/ha) are shown in parentheses.

Block	Treatment	Live Tree Density			Live Tree Basal Area			% Live Conifer Basal Area		
		2007	2008	Change	2007	2008	Change	2007	2008	Change
OLYM[1]	REF	207.8 (513.5)	207.8 (513.5)	0.0 (0.0)	189.9 (43.7)	189.9 (43.7)	0.0 (0.0)	100.0	100.0	0.0
	100%	154.7 (382.3)	150.3 (371.4)	−4.4 (−10.9)	208.5 (48.0)	204.1 (46.9)	−4.3 (−1.0)	98.1	98.1	0.0
	FP	157.7 (389.7)	154.7 (382.3)	−2.9 (−7.2)	208.5 (48.0)	206.0 (47.4)	−2.5 (−0.6)	100.0	100.0	0.0
WIL1	REF	218.9 (540.9)	150.3 (371.4)	−68.5 (−169.3)	236.7 (54.4)	153.6 (35.3)	−83.1 (−19.1)	90.0	87.7	−2.4
	100%	307.3 (759.3)	267.5 (661.0)	−39.8 (−98.3)	300.4 (69.1)	267.1 (61.4)	−33.3 (−7.7)	100.0	100.0	0.0
	FP	234.3 (579.0)	115.0 (284.2)	−119.4 (−295.0)	285.4 (65.6)	159.1 (36.6)	−126.3 (−29.0)	100.0	100.0	0.0
	0%	129.1 (319.0)	116.7 (288.4)	−12.4 (−30.6)	145.2 (33.4)	133.3 (30.7)	−11.9 (−2.7)	99.5	100.0	0.5
WIL2	REF1	247.6 (611.8)	199.0 (491.7)	−48.6 (−120.1)	303.0 (69.7)	241.6 (55.6)	−61.4 (−14.1)	100.0	100.0	0.0
	REF2	243.2 (600.9)	225.5 (557.2)	−17.7 (−43.7)	264.4 (60.8)	253.7 (58.4)	−10.7 (−2.5)	100.0	100.0	0.0
	100%	249.8 (617.3)	210.0 (518.9)	−39.8 (−98.3)	239.5 (55.1)	203.8 (46.9)	−35.7 (−8.2)	99.9	99.8	0.0
	0%	250.5 (619.0)	219.6 (542.6)	−30.9 (−76.4)	188.4 (43.3)	170.5 (39.2)	−18.0 (−4.1)	92.7	91.4	−1.3
CASC	REF	147.4 (364.2)	147.4 (364.2)	0.0 (0.0)	222.2 (51.1)	222.2 (51.1)	0.0 (0.0)	92.1	92.1	0.0
	FP	112.0 (276.8)	112.0 (276.8)	0.0 (0.0)	149.5 (34.4)	149.5 (34.4)	0.0 (0.0)	8.6	8.6	0.0
	0%	48.6 (120.1)	48.6 (120.1)	0.0 (0.0)	53.2 (12.2)	53.2 (12.2)	0.0 (0.0)	83.8	83.8	0.0
Mean		193.5 (478.1)	166.0 (410.2)	−27.5 (−68.0)	213.9 (49.2)	186.2 (42.8)	−27.7 (−6.4)	90.3	90.1	−0.2

[1] We excluded plots that intersected roads, which resulted in a lack of PIP plots sampled in the OLYM-0% treatment.

Appendix Table 5-A-4. Mean pre-harvest site values for dead tree density (trees/ac) and dead tree basal area (ft^2/ac) in RMZs. Metric units (trees/ha, m^2/ha) are shown in parentheses.

Block	Treatment	Dead Tree Density			Dead Tree Basal Area		
		2007	2008	Change	2007	2008	Change
OLYM	REF	9.9 (24.5)	10.1 (25.0)	0.2 (0.5)	11.3 (2.6)	11.4 (2.6)	0.1 (0.02)
	100%	13.5 (33.4)	13.9 (34.3)	0.4 (1.0)	6.0 (1.4)	6.1 (1.4)	0.1 (0.02)
	FP	27.4 (67.7)	27.8 (68.7)	0.4 (1.0)	10.6 (2.4)	10.7 (2.5)	0.2 (0.05)
	0%	15.7 (38.8)	15.7 (38.8)	0.0 (0.0)	8.6 (2.0)	8.6 (2.0)	0.0 (0.0)
WIL1	REF	21.2 (52.4)	21.8 (53.9)	0.6 (1.5)	4.4 (1.0)	7.9 (1.8)	3.5 (0.8)
	100%	49.7 (122.8)	48.4 (119.6)	−1.3 (−3.2)	13.9 (3.2)	17.8 (4.1)	3.9 (0.9)
	FP	27.4 (67.7)	29.9 (73.9)	2.5 (6.2)	28.2 (6.5)	33.6 (7.7)	5.3 (1.2)
	0%	16.2 (40.0)	16.6 (41.0)	0.4 (1.0)	28.9 (6.6)	29.5 (6.8)	0.6 (0.14)
WIL2	REF1	27.8 (68.7)	23.4 (57.8)	−4.4 (−10.9)	9.2 (2.1)	10.3 (2.4)	1.0 (0.2)
	REF2	19.2 (47.4)	20.9 (51.6)	1.7 (4.2)	9.5 (2.2)	11.7 (2.7)	2.1 (0.5)
	100%	23.1 (57.1)	21.6 (53.4)	−1.5 (−3.7)	61.0 (14.0)	60.6 (13.9)	−0.5 (−0.1)
	0%	66.6 (164.6)	65.8 (162.6)	−0.9 (−2.2)	13.8 (3.2)	13.8 (3.2)	0.0 (0.0)
WIL3	REF	24.0 (59.3)	24.0 (59.3)	0.0 (0.0)	67.7 (15.6)	67.7 (15.6)	0.0 (0.0)
	100%	41.5 (102.5)	40.1 (99.1)	−1.4 (−3.5)	19.9 (4.6)	19.5 (4.5)	−0.4 (−0.1)
CASC	REF	98.6 (243.6)	98.6 (243.6)	0.0 (0.0)	221.9 (51.0)	222.3 (51.1)	0.4 (0.1)
	FP	10.6 (26.2)	10.6 (26.2)	0.0 (0.0)	33.0 (7.6)	33.0 (7.6)	0.0 (0.0)
	0%	16.5 (40.8)	16.5 (40.8)	0.0 (0.0)	59.5 (13.7)	59.5 (13.7)	0.0 (0.0)
Mean		29.9 (73.9)	29.7 (73.4)	−0.2 (−0.5)	35.7 (8.2)	36.7 (8.4)	1.0 (0.2)

Appendix Table 5-A-5. Mean pre-harvest site values for dead tree density (trees/ac) and dead tree basal area (ft^2/ac) in PIPs. Metric units (trees/ha, m^2/ha) are shown in parentheses.

Block	Treatment	Dead Tree Density			Dead Tree Basal Area		
		2007	2008	Change	2007	2008	Change
OLYM[1]	REF	16.2 (40.0)	16.2 (40.0)	0.0 (0.0)	10.8 (2.5)	10.8 (2.5)	−5.4 (−1.2)
	100%	13.3 (32.9)	13.3 (32.9)	0.0 (0.0)	9.4 (2.2)	9.4 (2.2)	−3.9 (−0.9)
	FP	26.5 (65.5)	26.5 (65.5)	0.0 (0.0)	9.0 (2.1)	9.0 (2.1)	−17.5 (−4.0)
WIL1	REF	15.5 (38.3)	15.5 (38.3)	0.0 (0.0)	9.9 (2.3)	12.0 (2.8)	−3.5 (−0.8)
	100%	28.7 70.9)	37.6 (92.9)	8.8 (21.7)	25.3 (5.8)	31.5 (7.2)	−6.1 (−1.4)
	FP	48.6 (120.1)	39.8 (98.3)	−8.8 (−21.7)	14.0 (3.2)	14.0 (3.2)	−25.8 (−5.9)
	0%	8.0 (19.8)	6.2 (15.3)	−1.8 (−4.4)	3.4 (0.8)	2.9 (0.7)	−3.3 (−0.8)
WIL2	REF1	30.9 (76.4)	30.9 (76.4)	0.0 (0.0)	14.5 (3.3)	15.0 (3.5)	−16.0 (−3.7)
	REF2	26.5 (65.5)	22.1 (54.6)	−4.4 (−10.9)	6.4 (1.5)	5.2 (1.2)	−16.9 (−3.9)
	100%	8.8 (21.7)	13.3 (32.9)	4.4 (10.9)	8.7 (2.0)	10.1 (2.3)	−3.2 (−0.7)
	0%	17.7 (43.7)	19.2 (47.4)	1.5 (3.7)	6.5 (1.5)	8.4 (1.9)	−10.8 (−2.5)
CASC	REF	79.6 (196.7)	79.6 (196.7)	0.0 (0.0)	150.4 (34.6)	150.4 (34.6)	70.9 (16.3)
	0%	13.3 (32.9)	13.3 (32.9)	0.0 (0.0)	10.0 (2.3)	10.0 (2.3)	−3.2 (−0.7)
	FP	11.8 (29.2)	11.8 (29.2)	0.0 (0.0)	6.2 (1.4)	6.2 (1.4)	−5.6 (−1.3)
Mean		24.7 (61.0)	24.7 (61.0)	0.0 (0.0)	20.3 (4.7)	21.1 (4.9)	−3.6 (−0.8)

[1] We excluded plots that intersected roads, which resulted in a lack of PIP plots sampled in the OLYM-0% treatment.

Appendix Table 5-A-6. Mean annual mortality rates among sites expressed as percentage of initial density and basal area for RMZ and PIP plots for the pre-harvest period.

		RMZ Plots		PIP Plots	
Block	**Treatment**	**Annual Mortality Rate (% of initial density)**	**Annual Mortality Rate (% of initial basal area)**	**Annual Mortality Rate (% of initial density)**	**Annual Mortality Rate (% of initial basal area)**
OLYM	REF	1.1	0.7	0.0	0.0
	100%	3.7	1.9	2.9	1.7
	FP	5.6	3.5	2.2	1.2
	0%	2.2	1.8	-	-
WIL1	REF	39.8	38.9	31.2	33.6
	100%	19.6	18.3	15.6	12.6
	FP	8.9	8.5	50.9	46.1
	0%	17.6	20.3	13.7	13.2
WIL2	REF1	14.2	11.6	19.8	19.7
	REF2	13.0	12.1	7.3	4.0
	100%	8.4	8.2	13.8	13.7
	0%	8.3	6.8	12.6	10.9
WIL3	REF	1.4	0.8	-	-
	100%	1.2	1.1	-	-
CASC	REF	0.8	0.6	0.0	0.0
	FP	0.0	0.0	0.0	0.0
	0%	0.0	0.0	0.0	0.0
Mean		8.6	8.0	12.1	11.2

Appendix Table 5-A-7. Mean annual tree mortality rates among sites expressed as percentage of initial density and basal area in for RMZ and PIP plots in the post-harvest period.

		RMZ		PIP	
Block	**Treatment**	**Annual Mortality Rate (% of initial density)**	**Annual Mortality Rate (% of initial basal area)**	**Annual Mortality Rate (% of initial density)**	**Annual Mortality Rate (% of initial basal area)**
OLYM	REF	7.0	4.8	2.5	2.0
	100%	2.6	1.7	2.2	2.2
	FPB	17.7	14.6	36.4	24.8
WIL1	REF	21.8	20.5	9.6	6.5
	100%	13.8	10.7	10.7	8.2
	FPB	34.2	38.5	24.0	15.8
WIL2	REF1	5.5	5.5	4.9	2.1
	REF2	6.5	4.8	1.0	1.3
	100%	17.2	15.2	20.5	21.0
WIL3	REF	2.6	1.9	-	-
	100%	4.4	3.5	-	-
CASC	REF	2.3	1.8	1.1	0.5
	FPB	4.0	2.3	23.5	21.0

Appendix Table 5-A-8. Mean post-harvest site values for live tree density (trees/ac) and live tree basal area (ft^2/ac) in RMZs. Metric units (trees/ha, m^2/ha) are shown in parentheses.

Block	Treatment	Live Tree Density			Live Tree Basal Area		
		2008	**2010**	**Change**	**2008**	**2010**	**Change**
OLYM	REF	222.0	193.8	−28.1	181.9	175.3	−6.6
		(548.6)	(478.9)	(−69.4)	(41.8)	(40.3)	(−1.5)
	100%	190.8	182.5	−8.3	172.5	176.6	4.1
		(471.5)	(451.0)	(−20.5)	(39.7)	(40.6)	(0.9)
	FPB	175.0	118.3	−56.6	168.0	129.1	−38.9
		(432.4)	(292.3)	(−139.9)	(38.6)	(29.7)	(−8.9)
	0%	220.4	0.0	−220.4	182.4	0.0	−182.4
		(544.6)	(0.0)	(−544.6)	(42.0)	(0.0)	(−42.0)
WIL1	REF	135.0	106.4	−28.6	146.7	118.0	−28.6
		(333.6)	(262.9)	(−70.7)	(33.7)	(27.1)	(−6.6)
	100%	262.7	198.6	−64.0	260.6	216.0	−44.6
		(649.1)	(490.7)	(−158.1)	(59.9)	(49.7)	(−10.3)
	FPB	189.9	92.3	−97.6	230.9	119.1	−111.8
		(469.2)	(228.1)	(−241.2)	(53.1)	(27.4)	(−25.7)
	0%	175.9	2.1	−173.8	129.8	1.4	−128.5
		(434.6)	(5.2)	(−429.5)	(29.9)	(0.3)	(−29.6)
WIL2	REF1	188.9	174.8	−14.2	219.5	213.0	−6.4
		(466.8)	(431.9)	(−35.1)	(50.5)	(49.0)	(−1.5)
	REF2	186.4	164.9	−21.5	169.2	162.1	−7.1
		(460.6)	(407.5)	(−53.1)	(38.9)	(37.3)	(−1.6)
	100%	250.5	189.9	−60.7	189.9	149.0	−40.9
		(619.0)	(469.2)	(−150.0)	(43.7)	(34.3)	(−9.4)
	0%	357.6	2.6	−355.0	244.3	1.1	−43.2
		(883.6)	(6.4)	(−877.2)	(56.2)	(0.3)	(−55.9)
WIL3	REF	126.9	120.1	−6.8	210.5	209.7	−0.9
		(313.6)	(296.8)	(−16.8)	(48.4)	(48.2)	(−0.2)
	100%	224.8	206.0	−18.8	255.4	253.1	−2.3
		(555.5)	(509.0)	(−46.5)	(58.7)	(58.2)	(−0.5)
CASC	REF	151.5	144.1	−7.5	225.2	222.4	−2.8
		(374.4)	(356.1)	(−18.5)	(51.8)	(51.2)	(−0.6)
	FPB	84.9	77.3	−7.6	111.6	108.5	−3.1
		(209.8)	(191.0)	(−18.8)	(25.7)	(25.0)	(−0.7)
	0%	91.0	12.6	−78.4	135.2	11.1	−124.1
		(224.9)	(31.1)	(−193.7)	(31.1)	(2.6)	(−28.5)

Appendix Table 5-A-9. Mean post-harvest site values for live tree density (trees/ac) and live tree basal area (ft^2/ac) in PIPs. Metric units (trees/ha, m^2/ha) are shown in parentheses.

Block	**Treatment**	**Live Tree Density**			**Live Tree Basal Area**		
		2008	**2010**	**Change**	**2008**	**2010**	**Change**
OLYM	REF	207.8 (513.5)	197.5 (488.0)	−10.3 (−25.5)	189.9 (43.7)	194.6 (44.8)	4.7 (1.1)
	100%	150.3 (371.4)	143.7 (355.1)	−6.6 (−16.3)	204.1 (46.9)	204.7 (47.1)	0.5 (0.1)
	FPB	154.7 (382.3)	59.0 (145.8)	−95.8 (−236.7)	206.0 (47.4)	114.9 (26.4)	−91.2 (−21.0)
WIL1	REF	150.3 (371.4)	119.4 (295.0)	−30.9 (−76.4)	153.6 (35.3)	144.2 (33.2)	−9.4 (−2.2)
	100%	267.5 (661.0)	214.4 (529.8)	−53.1 (−131.2)	267.1 (61.4)	224.1 (51.5)	−43.0 (−9.9)
	FPB	115.0 (284.2)	66.3 (163.8)	−48.6 (−120.1)	159.1 (36.6)	118.7 (27.3)	−40.3 (−9.3)
	0%	116.7 (288.4)	0.0 (0.0)	−116.7 (−288.4)	133.3 (30.7)	0.0 (0.0)	−133.3 (−30.7)
WIL2	REF1	199.0 (491.7)	181.3 (448.0)	−17.7 (−43.7)	241.6 (55.6)	235.9 (54.3)	−5.7 (−1.3)
	REF2	225.5 (557.2)	221.1 (546.3)	−4.4 (−10.9)	253.7 (58.4)	257.2 (59.2)	3.5 (0.8)
	100%	210.0 (518.9)	132.6 (327.7)	−77.4 (−191.3)	203.8 (46.9)	133.4 (30.7)	−70.4 (−16.2)
	0%	219.6 (542.6)	0.0 (0.0)	−19.6 (−542.6)	170.5 (39.2)	0.0 (0.0)	−170.5 (−39.2)
CASC	REF	147.4 (364.2)	144.4 (356.8)	−2.9 (−7.2)	222.2 (51.1)	229.0 (52.7)	6.9 (1.6)
	FPB	112.0 (276.8)	64.8 (160.1)	−47.2 (−116.6)	149.5 (34.4)	101.1 (23.3)	−48.4 (−11.1)
	0%	48.6 (120.1)	22.1 (54.6)	−26.5 (−65.5)	53.2 (12.2)	6.0 (1.4)	−47.2 (−10.9)

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CHAPTER 6 - WOOD RECRUITMENT AND LOADING

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

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6-1. ABSTRACT

Wood plays an important functional role in streams throughout the Pacific Northwest, influencing channel morphology and hydraulics, storage and routing of sediment and organic matter, the formation of habitat, and food resources supporting biotic communities. Tree fall from the streamside forest is an important source of large wood to small headwater streams. Clearcut harvest of the streamside forest typically results in large inputs of mostly small wood and long-term reductions in large wood inputs through removal of the stream-adjacent forest. Retention of riparian buffers provides a source of future wood recruitment; however, increased tree fall rates in wind-affected reaches may result in short-term increases in wood input and loading with uncertain long-term consequences.

We compared the response of large wood recruitment rates and wood loading in headwater streams among several alternative riparian buffer treatments during the first two years following clearcut harvest. Treatments differed in the length of a two-sided 50-ft (15.2-m) riparian buffer maintained in the Riparian Management Zone (RMZ) and included a reference (REF), full buffer (100%), ≥50% buffered (following current Forest Practices rules [FP]), and no buffer (0%). We observed substantial post-harvest tree fall in the FP treatment buffers, with mean tree fall rates that were over 5 times the reference rate and nearly 3 times the 100% treatment rate. We also found that tree fall was significantly greater in buffered areas surrounding the uppermost points of perennial flow (PIPs) in the 100% and FP treatments compared with the reference. The primary factor contributing to tree fall was windthrow. Mean large wood recruitment volume was generally highest in the 100% treatment, lower in the FP treatment, and lowest in the reference for RMZs and PIPs, though there were few statistically significant differences. There was a substantial reduction in wood recruitment in the 0% treatment since nearly all streamside trees were removed from the RMZ by design.

Post-harvest changes in the amount and characteristics of in-channel wood (i.e., loading) differed among treatments, with an increase in the number of large wood pieces in all three buffer treatments. We observed a significant increase in the number of small wood pieces (<10 cm diameter) in all buffer treatments. In-channel wood cover (i.e., shading) differed between treatments in the first year post-harvest, with new wood cover in the 100%, FP and 0% treatments estimated to be 3, 8, and 9 times greater than in the reference. Due to differences in the riparian stand conditions and wood input regimes between buffered and clearcut reaches of the FP treatment RMZ, we anticipate extensive reach-scale variation in wood loading to develop in headwater basins managed under Forest Practices rules. Based on simulation modeling, we expect lower wood loading in the clearcut RMZ, with wood load oscillating in response to periodic inputs of logging debris, as well as mass wasting or debris flow disturbance. We expect the wood to be greater in the buffered reaches, with the pattern determined by the characteristics of the stand and the magnitude and timing of disturbance. In the absence of episodic disturbances, wood loading in the buffered reaches should gradually increase over time. In buffered reaches experiencing episodic disturbance, we expect wood loading to increase in the short-term and then decline over time as the streamside forest regenerates.

6-2. INTRODUCTION

The amount, characteristics and function of wood in stream channels (wood loading) is affected by the wood supply available, the magnitude and frequency of tree mortality and disturbance processes that deliver and distribute wood through the stream system, and the characteristics of the watershed, valley and channel network (Montgomery *et al.* 2003; Hassan *et al.* 2005; Wohl and Cadol 2011). Stream size and topographic setting exert a strong influence on processes that deliver and redistribute wood to streams (May and Gresswell 2003a) and the functions of wood in the channel (Bilby and Ward 1989). Small headwater streams confined by narrow valleys in mountainous terrain, such as those in the Cascades and Coast Ranges of the Pacific Northwest, have high connectivity with the adjacent hillslopes and upland forests (May and Gresswell 2003a). The dominant natural processes delivering wood to headwater streams include mortality and tree fall from adjacent streamside stands, and mass wasting processes or snow avalanches that deliver wood from upslope forests (Keller and Swanson 1979; Andrus *et al.* 1988; May and Gresswell 2003a). Input from bank erosion is limited because headwater channels are typically confined by valley walls, limiting the potential for lateral movement (Martin and Benda 2001; May and Gresswell 2003a). The wood recruited to small streams from the adjacent forest tends to remain in place because the pieces are typically longer than the narrow channels, are oriented perpendicular to the channel (Bilby and Ward 1989) and become jammed in the channel or supported by the banks and valley side slopes (Gurnell 2003). In addition, headwater streams with small drainage areas typically lack the flow capacity to move large wood (LW) by floatation (Gurnell 2003). Consequently, LW tends to accumulate in small channels over time (Keller and Swanson 1979; May and Gresswell 2003a, b), with reach-scale variation in wood load dependent on the location of input sources and residence time determined by transport and decay rates (Gurnell 2003). Many steep headwater streams are periodically affected by debris flows that scour wood from the bed and banks and deposit it in low gradient reaches or tributary junctions (Keller and Swanson 1979; May and Gresswell 2003a, b; Nakamura and Swanson 2003).

Wood plays an important functional role in headwater stream channels (Bilby and Bisson 1998). The abundance and characteristics of in-channel wood influences channel morphology and hydraulics, the storage and routing of sediment and organic matter, and the formation of habitat and food sources supporting biotic communities (Harmon *et al.* 1986). Wood in small streams influences channel form by obstructing and directing flow, causing localized scour of the bed and banks and sorting of bed material (Keller and Swanson 1979; Montgomery *et al.* 2003). Step-pool sequences formed by wood in small streams are important features that impound water upstream of the obstruction and scour pools below the drop (Keller and Swanson 1979; Bilby and Ward 1989). A substantial proportion of the flow energy in headwater streams is dissipated in these step-pool sequences due to the form resistance of the wood and spill resistance associated with the hydraulic drop into the pool below the step (Curran and Wohl 2003; MacFarlane and Wohl 2003; Wilcox *et al.* 2011).

Wood pieces and associated steps trap and store sediment, smaller wood and particulate organic material (Bilby and Ward 1989; Gomi *et al.* 2002; Hassan *et al.* 2005). Depositional areas associated with wood are more frequent in small streams due to the greater frequency of wood pieces that accumulate sediment (Bilby and Ward 1989). Channel-spanning jams in headwater streams are important sites for organic carbon storage and processing, and increase the residence time of fine sediments and particulate organic matter (Bilby and Ward 1989; Beckman and Wohl 2014). Wood-associated storage sites, particularly debris jams, can account for a large proportion of total sediment storage in small streams, providing substantial storage for long periods in the absence of debris flow disturbance (Keller and Swanson 1979). The loss of wood in headwater streams can lead to channel incision and the mobilization of large amounts of sediment (Beschta 1979; Montgomery *et al.* 2003). Wood pieces, which play an important role in pool formation, result in a large proportion of pools in small stream channels (Andrus *et al.* 1988; Montgomery *et al.* 1995). Wood creates cover that provides refuge for fish and stream-associated amphibian species (Wilkins and Peterson 2000; Rundio and Olson 2007) and influences the composition of invertebrate assemblages (Anderson *et al.* 1978).

The functions provided by wood depend upon the characteristics of the wood and the stream channel (Gomi *et al.* 2001; Maxa 2009). While most studies have focused on LW, small wood (SW, i.e., ≤10 cm diameter) can be abundant in small streams where transport is limited by channel size and stream power (Bilby and Ward 1989; Maxa 2009). Small wood may play a short-term functional role in smaller stream channels by storing sediment and influencing channel morphology (Gomi *et al.* 2001; Maxa 2009). For example, Jackson and Sturm (2002) found that SW and organic debris were major step-forming elements in non-fish-bearing streams in the Pacific Northwest. However, SW appears to be less persistent due to more rapid decay and susceptibility to downstream transport (Wallace *et al.* 2000; Scherer 2004).

The characteristics of the streamside forest determines the abundance and characteristics of in-channel wood (Beechie *et al.* 2000). There are countervailing processes that increase and decrease wood loading through recruitment of wood from adjacent forests or upstream reaches, transportation of wood downstream or onto the floodplain, and decomposition of wood through time (Keller and Swanson 1979; Benda and Sias 2003; Hassan *et al.* 2005). The nature and timing of disturbance processes have a major effect on wood loading over time (Spies *et al.* 1988; Bragg 2000). Wood input processes can be characterized as chronic or episodic. Chronic input refers to mortality and recruitment of individual trees (or small groups) from streamside forests that occurs gradually over long periods due to suppression, wind, insects or disease. Episodic (i.e., catastrophic) input refers to mortality and input of larger numbers of trees over short periods of time due to disturbance events such as high winds, fire, insect outbreaks, landslides or snow avalanches (Bragg 2000; Benda *et al.* 2003). With a chronic process, wood inputs from streamside stands provide a relatively stable supply of wood to the channel, producing a wood loading regime that gradually increases with stand age (Hedman *et al.* 1996; Warren *et al.* 2009). In contrast, episodic disturbance results in large inputs of wood at irregular intervals, producing extensive variation in wood loading over time. Wood input and loading is typically high during the first few decades after the disturbance, as newly killed trees recruit to the channel, augmenting the pre-disturbance wood load. This period of peak wood abundance is followed by an extended period of decreasing wood abundance when in-channel wood is depleted and little wood is recruited following the disturbance. Finally, if no further disturbance occurs, wood loading increases as the forest matures and wood is recruited from chronic tree mortality associated with suppression and small-scale disturbance (Spies *et al.* 1988; Bragg 2000).

Timber harvest and riparian management practices (e.g., riparian buffers) affect wood input and loading over time (Boyer *et al.* 2003; Gregory *et al.* 2003; Meleason *et al.* 2003; Burton *et al.*

2016). Clearcut harvest of streamside forests typically results in large inputs of mostly SW during the harvest. In a study of the impacts of clearcut harvest on headwater streams in the coastal range of Washington, over 98% of the stream length was covered with organic debris (i.e., logs, branches, twigs and needles) that was 1 to 2 m deep (Jackson *et al.* 2001). Harvest of streamside trees can result in long-term reductions in LW inputs by decreasing future wood recruitment from tree fall in the stream-adjacent forest (Beechie *et al.* 2000), affecting mass-wasting processes on unstable slopes (May 2001) and accelerating debris flows that transport wood out of headwater channels (Nakamura *et al.* 2000; May 2001). These processes can reduce wood input and loading for many decades while a new forest becomes established (Beechie *et al.* 2000; Bragg 2000; Meleason *et al.* 2003). The current Washington Forest Practices rules for timber harvest along Type N streams in western Washington require partial buffering of the perennial headwater stream network (WFPB 2016). One of the objectives of maintaining streamside riparian buffers is to provide a long-term source of wood recruitment that will contribute to wood loading over time. However, if the buffers are affected by windthrow, increased tree fall rates may result in short-term increases in wood input and loading in headwater streams (Grizzel and Wolff 1998; Jackson *et al.* 2001; Schuett-Hames *et al.* 2012), with uncertain effects on future wood input and loading. The purpose of this study was to evaluate the response of tree fall, LW recruitment rates, and in-channel wood loading (including SW and LW) along headwater streams in western Washington to timber harvest with variable-length riparian buffers.

6-3. OBJECTIVES

Our primary goal was to evaluate the response of wood recruitment and loading in Type N streams in western Washington to different riparian buffer treatments. We did this by examining the magnitude, direction (positive or negative) and duration of change in riparian-related tree fall, wood recruitment and in-channel wood loading associated with a range of experimental riparian buffering treatments that varied in the length of riparian buffer retained relative to untreated reference conditions. To achieve this goal, we addressed the following objectives and research questions:

Objective 1: Tree Fall and Recruitment of Large Wood to the Channel: Evaluate the response of tree fall and LW recruitment rates to buffer treatments by estimating riparian tree fall and LW recruitment rates during the pre-and post-harvest periods, and comparing post-harvest rates between treatments.

1) What were the tree fall and in-channel LW recruitment rates during the pre-harvest period and what factors influence pre-harvest rates?

2) What were the tree fall and in-channel LW recruitment rates during the post-harvest period and what factors influence post-harvest rates?

3) Were there differences in post-harvest tree fall and in-channel LW recruitment rates between treatments (including buffer treatments and the reference)?

4) What were the characteristics of wood recruited to the channel?

Objective 2: In-channel Wood Loading: Evaluate the response of in-channel wood loading to buffer treatments.

1) Were there differences in in-channel wood loading (i.e., numbers of pieces) between treatments?
2) Were there differences in the functional roles (e.g., contributions to step formation, bank stability or hydraulic roughness) of in-channel wood pieces between treatments?

Objective 3: Newly Recruited Wood Cover: Characterize the in-channel cover, or shading, provided by wood newly recruited in the post-harvest period.

1) Were there differences in the in-channel cover provided by newly recruited post-harvest wood between treatments?

6-4. METHODS

We collected data at 17 study sites consisting of Type N headwater basins located in competent lithologies (largely basaltic) across western Washington. We blocked sites by geographic location to account for regional variability in climatic and environmental conditions (see Chapter 2 – *Study Design*). The Olympic block was located in the southwestern foothills of the Olympic Mountains, three blocks were in the Willapa Hills, and one block was in the southern Cascade Range just north of the Columbia River Gorge.

We assigned one of four basin-scale riparian buffer treatments to each site as follows:

1) Reference: No trees were cut in the uplands or in the Riparian Management Zone (RMZ).
2) 100% treatment: Uplands were clearcut and the RMZ maintained a 50-ft (15.2-m) wide riparian buffer along the entire length of the Type Np stream network, with 56-ft (17.1-m) radius buffers surrounding the uppermost points of perennial flow (PIPs).
3) Forest Practice (FP) treatment: Uplands and a maximum of 50% of the length of the Type Np stream RMZ were clearcut to the edge of the stream; the remainder of RMZ maintained a 50-ft (15.2-m) wide riparian buffer, with 56-ft (17.1-m) radius buffers surrounding the PIPs. We refer to the buffered portion as the FPB.
4) 0% treatment: Uplands and the entire RMZ were clearcut to the edge of the stream.

The methods for data collection and analysis depended on the response metric evaluated. The following sections describe the sample timing, sampling strategy, data collection methods and metric estimation for tree fall and LW recruitment and channel wood loading. This is followed by a section that describes the statistical analysis procedures for all metrics.

6-4.1. TREE FALL AND LARGE WOOD RECRUITMENT

6-4.1.1. Sample Timing

We estimated tree fall and LW recruitment rates for two periods. We compared a survey of fallen trees and LW recruitment done in the summer 2008 with an inventory of standing trees done in the summer of 2007 to estimate tree fall and LW recruitment rates for a one-year pre-harvest period. The pre-harvest period included tree fall and wood recruitment from the December 2007 windstorm that caused extensive damage to riparian stands in study sites in the Willapa 1 and Willapa 2 blocks (see Chapter 4 – *Unanticipated Disturbance Events* and Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers*). We compared a survey of fallen trees and LW recruitment done in the summer of 2010 with an inventory of standing trees in the summer of 2008 to estimate tree fall and LW recruitment rates for the two-year post-harvest period. The post-harvest period includes the harvest window, a transition period from July 2008 through August 2009. The rationale was that the tree fall and LW recruitment that occurred during the transition period were largely due to extensive windthrow that occurred immediately post-harvest in the newly established buffers. Ignoring tree fall and LW recruitment that occurred during this period would confound the estimates of post-harvest tree fall and LW recruitment in the 100% and FPB treatment buffers.

6-4.1.2. Sampling Strategy

We evaluated two riparian management strategies defined by state Forest Practices rules (WFPB 2016): the two-sided 50-ft (15.2-m) wide RMZ buffer adjacent to both sides of the perennial Type N stream, and the 56-ft (17.1-m) radius PIP buffers located around the uppermost points of perennial flow. We estimated tree fall and LW recruitment rates separately for each. We collected data in the same plots established for riparian vegetation sampling (see Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers*).

6-4.1.2.a. RMZ sampling strategy

Riparian zones present a unique sampling situation due to their shape (narrow linear features) and extensive variability in stand structure and mortality both longitudinally and perpendicular to the stream. We based our RMZ sampling strategy on the recommendations of Marquardt and colleagues (2010), who found that a series of small strip plots oriented perpendicular to the stream performed well in characterizing stand structure in headwater stream RMZs. We established permanent sample plots systematically along the mainstem and all tributaries long enough to accommodate a plot (i.e., ≥30.5 m [100 ft]). We established plots spaced every 30.5 m (100 ft) for sites with less than or equal to 1524 m (5,000 ft) of RMZ (n = 11), sampling approximately 50% of the RMZ length. We established plots spaced every 45.7 m (150 ft) for sites with over 1524 m (5,000 ft) of RMZ (n = 6), sampling approximately 33% of the RMZ.

Plots were established in a downstream direction from the upper end of the RMZ of each tributary, with the first plot randomly located either 0 or 15.2 m (0 or 50 ft) downstream from the edge of the PIP buffer. Each plot was a 15.2 by 30.4 m (50 by 100 ft) rectangle, with the long axis perpendicular to, and bisected by, the stream channel (15.2 m [50 ft] on each side), for a total area of 0.05 ha (0.115 acres). We measured plot dimensions in horizontal distance using a laser rangefinder with foliage filter. Plots that intersected roads or that partially overlapped with plots in adjacent streams were not sampled, with priority given to plots located along the mainstem.

6-4.1.2.b. PIP sampling strategy

We established PIP buffer plots with a laser rangefinder. Each plot had a radius of 56 ft (17.1 m) horizontal distance and an area of 0.09 ha (0.226 ac). The number of PIPs was equal to the number of tributaries and varied among sites. We collected data at 34 PIPs in 14 study sites. One site had no PIPs. We did not sample PIPs in the two Willapa 3 block sites due to uncertainty about inclusion of this block in the study. As with RMZs, we excluded PIP plots that partially overlapped plots from adjacent streams or that intersected roads.

6-4.1.3. Data Collection

We defined fallen trees as those that originated within the plot boundaries and had fallen since the last survey. We classified fallen trees into two groups: uprooted (i.e., those that toppled over with the roots still attached) and broken (i.e., those sheared off along the stem). For broken trees, we included only trees where the broken portion had a diameter equal to or greater than 10 cm (4 inches) at the large end. When the upper portion of a tree broke off but the stem remained standing and was at least 1.4 m (4.5 ft) high, we treated the standing portion as a standing tree and the broken portion as a broken tree, if it was large enough to qualify. For each fallen tree, we recorded the condition (live/dead), species, breast height diameter, distance from stream, the process that caused the tree to fall, and the number of pieces. We counted the number of trees that reached the edge of the bankfull channel and their diameter at the bankfull channel edge, and whether they entered the bankfull channel or extended above the channel and were supported on one or both sides of the stream (suspended or spanning, respectively).

We defined LW recruitment as trees or pieces of trees that originated from within the boundary of a plot and fell so that a portion intruded into or over the bankfull channel during the study period. We counted each piece only once when it first recruited. To qualify as LW for this analysis, pieces had to be at least 10 cm (4 in) in diameter and 30 cm (1 ft) long. We counted all pieces that intersected the edge of the bankfull channel, including pieces in the following recruitment classes: those that intruded into the bankfull channel (bankfull), those extending over the channel and supported on both sides (spanning), and those extending over the channel but supported on only one side (suspended). For each LW piece we recorded: piece type (with or without attached rootwad); length, mid-point diameter, recruitment class, and whether it contributed to pool formation (decrease in water velocity and/or increase in depth), step formation (a vertical drop of at least 20 cm [8 in]), sediment retention (a deposit of sediment likely resulting from the presence of the identified piece, or that would become mobilized if the piece was removed) or formation of a functional debris jam (accumulation of wood pieces within the bankfull channel).

6-4.1.4. Metric Estimation

We calculated tree fall rates for each RMZ and PIP plot as a percentage of initial standing tree count (%STC/yr), initial standing density in trees/ac (%TPA/yr) and initial standing basal area (%basal area/yr) that fell during the pre- and post-harvest periods. We expressed rates on an annual basis using a compounding formula (Sheil *et al.* 1995):

$$m = 100[1 - (N_t/N_0)^{1/t}] \qquad (6\text{-}1)$$

where: m is the annual tree fall rate (%/yr),
N_0 is the initial density (or basal area) of standing trees,
N_t is the count, density or basal area of standing trees still present at the end of the measurement period; and
t is the length of the measurement period (yr).

We calculated pre- and post-harvest LW recruitment rates for each RMZ and PIP plot as the piece count or volume divided by plot size in acres divided by the length of the measurement period in years (i.e., pieces/acre/yr and ft^3/acre/yr, respectively). We then averaged values by site. Nearly all trees were harvested in the clearcut portions of FP treatments, creating a bi-modal distribution of tree fall and LW recruitment rates in the post-harvest period. Consequently, we included only plots from the buffered portions of the FP treatments (FPB) in the calculation of post-harvest rates. We examined frequency distributions of pre- and post-harvest tree fall and LW recruitment rates for RMZs and PIPs to evaluate plot-scale variability. While we included only plots from the FPB in analyses, we address the implications of the overall FP treatment in the discussion.

6-4.1.5. Approach to Answer Research Questions

To answer Objective 1, research questions 1 and 2 concerning pre- and post-harvest tree fall and LW recruitment rates, we reported site mean annualized tree fall and LW recruitment rates by plot type (RMZs and PIPs) for the pre-harvest and post-harvest periods, respectively. We examined frequency distributions of plot values to evaluate plot-scale variability for each plot type. To answer Objective 1, research question 3 concerning differences in tree fall and LW recruitment rates among treatments, we compared post-harvest tree fall and LW recruitment rates in the reference, 100% treatment, and FPB RMZs and PIPs. To answer Objective 1, research question 4, we reported the percentage of fallen trees that recruited to the channel by treatment, the distribution of fallen trees by source distance, and present information on their position relative to the channel (in or above) and in-channel functions.

6-4.2. CHANNEL WOOD LOADING AND NEWLY RECRUITED WOOD COVER

6-4.2.1. Sample Timing

We conducted sampling April–October. We collected channel wood loading data in each of two years in the pre-harvest (2006−2007) and post-harvest (2009−2010) periods. Consequently, post-harvest wood loading values include wood recruited both during the pre-harvest windthrow event and during and immediately post-harvest. We collected newly recruited wood cover data for the post-harvest period only.

6-4.2.2. Sampling Strategy

6-4.2.2.a. Channel wood loading

We evaluated channel wood loading along the mainstem channel of each study site. Ralph and colleagues (1994) proposed that the clumped distribution of organic debris dams within harvested headwater streams (Bilby and Likens 1980), and the resulting high variance in wood frequency, necessitates sampling longer stream reaches than is typical of many studies to obtain a reliable estimate of in-channel wood. Initially, in 2006, we evaluated wood loading along the entire mainstem channel, from F/N point to PIP, in every study site. Beginning in 2007, we adjusted this strategy in response to the large amount of wood encountered. From 2007 on, we conducted sampling along a subset of the mainstem channel, including the contiguous 200 m (656 ft) of stream immediately upstream of the F/N break, as well as additional sampling based on the total length of the mainstem channel. We delineated sample reaches as follows: for sites with a mainstem length <300 m (984 ft; n = 1) we sampled the entirety of the mainstem, for those 300 (984 ft) to 800 m (2,625 ft; n = 10) we sampled a minimum of 50% of the remaining length, and for those >800 m (2,625 ft; n = 6) we sampled a minimum of 25% of the remaining length. When we sampled less than the entire mainstem length, we conducted sampling in alternating 20 m (66 ft) stream segments (i.e., two consecutive 10 m [33 ft] sample reaches) systematically distributed throughout the remainder of the mainstem channel and above the 200-m (656-ft) contiguous reach sampled above the F/N break.

We did not sample a study site for one year in two instances. We did not sample the OLYM-REF in 2006 because it had not been approved for inclusion in the study. For that reason, in 2007, rather than sampling only 25% of the remaining mainstem channel based on length, we sampled it entirely (consistent with the first sample of all other study sites in 2006). We did not sample the (WIL2-REF1) in 2009 because we had intended it to be the FP treatment in the Willapa 2 block; when the site was not harvested, we included it as a second reference in the Willapa 2 block.

6-4.2.2.b. Newly recruited wood cover

In the post-harvest period, we found that newly recruited wood (i.e., wood pieces recruited from adjacent uplands, windthrow, or logging debris, including tree tops, branches and broken boles) was patchily distributed throughout the basin dependent on the stream riparian condition (i.e., buffered or not buffered), stream dendritic pattern, and topography among other things. Therefore, while we conducted our standard protocol only along the mainstem channel, we evaluated newly recruited wood basin wide, from the F/N break and upstream along all tributaries to the channel heads.

6-4.2.3. Data Collection

6-4.2.3.a. Channel wood loading

We collected wood loading data according to methodology derived from Veldhuisen and colleagues (2007). Within each sample reach, we identified all wood pieces greater than 2 cm average diameter, >10 cm long, and located within or directly over the stream within the bankfull channel. We tallied each piece as it was encountered, including only pieces that first entered the bankfull channel in the sample reach (i.e., we did not include pieces that first intersected the bankfull channel downstream of our sample reach). We classified each piece by diameter size class SW (2–10 cm), and LW (11–25 cm; 26–50 cm; 51–100 cm; >100 cm) and function (**Table 6-1**). We adapted the SW diameter size class from Schuett-Hames and colleagues (2012).

Table 6-1. Stream function categories for wood pieces. Categories were hierarchical, with each piece classified as the highest function to which it contributed (e.g., a wood piece that contributed to hydraulic roughness, bank stability, and step formation was classified as only a step).

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

In some buffer treatments, newly recruited wood restricted access to the stream and limited our ability to identify individual wood pieces. We developed an alternative method for evaluating newly recruited wood in reaches where we could not apply our standard protocol. We found that we could effectively conduct our standard protocol when the ocular estimate of newly recruited wood in 10 m (33 ft) sample reaches was less than 70% (hereafter, unobstructed stream reaches; e.g., **Figure 6-1a**). However, we found that we could not effectively apply our standard protocol when estimates of newly recruited wood were greater than 70% (hereafter, obstructed stream reaches; e.g., **Figure 6-1b**).

In study sites where obstructed stream reaches comprised more than 5% of the stream length, we applied a modified sampling protocol whereby we established 3-m long plots randomly located within obstructed reaches. To determine the number and locations of obstructed plots for a study site, we estimated the total proportion of the entire stream network length comprised of 10-m long obstructed reaches. For sites with 5 to 10% of the stream network length obstructed, we sampled two plots in obstructed reaches; for those 10 to 20% obstructed, we sampled three plots; for those 20 to 40% obstructed, four plots; and those 40% obstructed, six plots (**Table 6-2**). For each plot, we randomly selected a 10 m obstructed reach and then randomly selected the starting point of the plot within the obstructed portion of the sample reach.

Figure 6-1. Examples of an unobstructed and obstructed stream reaches in which (a) the standard sampling protocol was applied during both the pre- and post-harvest periods, and (b) a modified sampling approach was applied during the post-harvest period. Photo credit Frithiof T. Waterstrat (a) and Aimee P. McIntyre (b).

We conducted sampling in obstructed stream reaches in six and eight study sites in 2009 and 2010, respectively. In some instances, we could still sample stream reaches meeting the 70% or greater obstruction criteria with our standard sampling strategy; in other cases, the presence of large legacy logs inhibited sampling in obstructed reaches. As a result, the number of plots sampled in some sites was fewer than that intended based on the length of stream meeting the obstructed definition. Since our intent was to sample more than just wood in obstructed reaches (see Chapter 15 – *Stream-associated Amphibians*), we randomized our selection of sample reaches to the entire basin (mainstem and tributaries) and by stream order to ensure our sample was distributed throughout the entire stream network.

Within each plot, we temporarily removed wood from above and within the stream bankfull channel by moving it aside or, when necessary, cutting through and removing wood pieces with hand tools (e.g., handsaws, clippers). We tallied each piece prior to removal and assigned it to a diameter and function class. Following the completion of sampling, we replaced all wood into the channel. Our sampling in obstructed stream reaches in 2009 could have influenced piece size and transport, effecting wood loading estimates in 2010; however, we did not sample the same obstructed plots in 2009 and 2010. Moreover, because wood in obstructed stream reaches formed a dense mat, transport out of these reaches is unlikely without a debris flow or other high flow event; we had no evidence of these types of events in this year-long period. Finally, the length of the stream sampled with this method was minimal (maximum of 18 m for a single site and year) compared with the overall channel length sampled.

Table 6-2. The proportion of 10 m sample reaches throughout the entire stream network length (mainstem and tributaries) meeting the definition of an obstructed stream reach (i.e., >70% newly recruited wood cover) and number of resulting plots sampled in obstructed reaches for each of the two post-harvest years (2009 and 2010).

Block	Treatment	Proportion Obstructed		Plots	
		2009	2010	2009	2010
OLYM	REF	2%	0%	0	0
	100%	0%	5%	0	0
	FP	25%	20%	0	2
	0%	61%	57%	6	6
WIL1	REF	2%	2%	0	0
	100%	14%	14%	3	1
	FP	25%	15%	4	4
	0%	26%	29%	4	4
WIL2	REF1	0%	1%	0	0
	REF2	0%	2%	0	0
	100%	16%	23%	0	3
	0%	18%	25%	3	4
WIL3	REF	0%	0%	0	0
	100%	1%	2%	0	0
CASC	REF	0%	0%	0	0
	FP	14%	8%	2	2
	0%	2%	0%	0	0

6-4.2.3.b. Newly recruited wood cover

Since we were interested in the actual cover (i.e., shading) provided by newly recruited wood, regardless of wood size, we combined SW and LW in our evaluation. We estimated newly recruited wood cover in consecutive 10 m sample reaches as an ocular estimate of the percentage of the surface area within or over the bankfull channel that was covered by new wood, to the nearest 10%. Newly recruited wood included all fallen or cut trees, branches, twigs and leaves (if still attached) that appeared to have entered the stream during the preceding year, including green wood and weathered wood. “Greed wood” included pieces with green leaves and intact bark that lacked evidence of aging as identified by faded wood color and/or weathering. “Weathered wood” included pieces that did not meet the definition of green wood but were located on or above pieces identified as green wood, suggesting recent entry into the stream. Two individuals conducted these surveys each year following extensive training and calibration in order to minimize variability.

6-4.2.4. Metric Estimation

6-4.2.4.a. Channel wood loading

We calculated an average number of wood pieces per stream meter for the following groups: total SW, functional SW (categorized as step, bank or rough), total LW, and functional LW. Since wood loading estimates at sites with obstructed stream reaches were a combination of wood tallies from unobstructed and obstructed reaches, we determined the proportion of the sampled stream that was obstructed for each site and year. While applying our standard wood loading protocol, we recorded the locations and lengths of sample intervals to which we could not apply the standard protocol due to obstructions. The length of the intended sample intervals that could not be sampled in a site due to obstructions ranged from 23 to 144 m in 2009 and 2 to 114 m in 2010 (**Table 6-3**).

Table 6-3. Mainstem stream channel length that could not be sampled due to wood obstructions in the post-harvest period, 2009–2010.

Block	**Treatment**	**Survey Obstructions (m [ft])**	
		2009	**2010**
OLYM	REF	0 (0)	0 (0)
	100%	0 (0)	17 (55.8)
	FP	0 (0)	2 (6.6)
	0%	144 (472.4)	114 (374.0)
WIL1	REF	0 (0)	0 (0)
	100%	26 (85.3)	46 (150.9)
	FP	29 (95.1)	19 (62.3)
	0%	53 (173.9)	68 (223.1)
WIL2	REF1	0 (0)	10 (32.8)
	REF2	0 (0)	20 (65.6)
	100%	0 (0)	34 (111.6)
	0%	50 (164.0)	83 (272.3)
WIL3	REF	0 (0)	0 (0)
	100%	0 (0)	0 (0)
CASC	REF	0 (0)	0 (0)
	FP	0 (0)	16 (52.5)
	0%	23 (75.5)	30 (98.4)

We calculated the frequency of wood per meter in unobstructed and obstructed stream reaches as the total numbers of pieces by size class (i.e., SW and LW) divided by the total unobstructed and obstructed stream length sampled, respectively. We then calculated an overall "weighted" average of the number of wood pieces per stream meter in each group (WD/m_{weight}) as the sum of the wood pieces per meter in both unobstructed (WD/m_{unobs}) and obstructed (WD/m_{obs}) reaches based on the proportion of the sampled stream length that was unobstructed and obstructed:

$$WD/m_{weight} = (WD/m_{unobs} * \%\ \text{unobstructed}) + (WD/m_{obs} * \%\ \text{obstructed}) \quad (6\text{-}2)$$

We excluded data collected from the unharvested reaches of buffer treatments that were not harvested in their entirety (see **Table 3-2** in Chapter 3 – *Management Prescriptions*).

6-4.3. STATISTICAL ANALYSES

Analysis of tree fall and LW recruitment rates, and newly recruited wood cover, evaluated the null hypothesis:

$$T_{REF} = T_{100\%} = T_{FP} = T_{0\%} \quad (6\text{-}3)$$

where: T_{REF}, $T_{100\%}$, T_{FP}, and $T_{0\%}$ are post-harvest rates in the reference, 100%, FP and 0% treatments, respectively. For newly recruited wood cover, we predicted that the response would differ based on the number of years since buffer treatment application, so we conducted two analyses, one for each post-harvest year.

Analysis of wood loading evaluated the generalized null hypothesis:

$$\Delta T_{REF} = \Delta T_{100\%} = \Delta T_{FP} = \Delta T_{0\%} \quad (6\text{-}4)$$

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 conducted statistical analyses using a generalized linear mixed effects model with either the MIXED or GLIMMIX procedure in SAS (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 though the use of monotonic link transformations. An added benefit of mixed models is that they accommodate missing data as long as 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 wood loading under the BACI design, the models also incorporated site as a random effect with year and a treatment × year interaction term as fixed effects. For analyses conducted at the plot scale, 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. In some cases, we transformed data (e.g., log transformation) prior to analysis to obtain an approximately Gaussian error distribution while in other cases we analyzed non-normal data directly using a link function in a GLMM (**Table 6-4**). We characterized binomial models as events/trials and analyzed them using a binomial distribution and log link using restricted pseudo-likelihood. PIP % fallen tree count models initially exhibited over dispersion, which we corrected by including a multiplicative over-dispersion parameter.

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 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% with the exception of the fallen tree models which did not include the 0% treatment. 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 6-4. Description of metrics, scale of analysis, number of samples, and distribution/link for responses included in the tree fall, LW recruitment and wood loading analysis of response for the Type N Study (NA = not applicable).

Response variable	Scale	N	SAS Procedure	Data Transformation	Distribution/link
RMZ %fallen tree count (%STC/yr)	Site	13[1]	GLIMMIX	NA	Binomial/Logit
RMZ LW count (pieces/acre/yr)	Site	17	GLIMMIX	NA	Gaussian/Identity
RMZ LW volume (ft^3/acre/yr)	Site	17	GLIMMIX	NA	Lognormal/Identity
PIP %fallen tree count (%STC/yr)	Plot	24[2]	GLIMMIX	NA	Binomial/Logit[a]
PIP LW count (pieces/acre/yr)	Plot	33	GLIMMIX	NA	Gaussian/Identity
PIP LW volume (ft^3/acre/yr)	Plot	33	GLIMMIX	NA	Lognormal/Identity
SW total pieces	Site	17	GLIMMIX	NA	Poisson/Log
SW functional pieces	Site	17	GLIMMIX	NA	Poisson/Log
LW total pieces	Site	17	GLIMMIX	NA	Poisson/Log
LW functional pieces	Site	17	GLIMMIX	NA	Poisson/Log
Newly recruited wood (1st year post)	Site	17	MIXED	LOG	Gaussian/NA
Newly recruited wood (2nd year post)	Site	17	MIXED	LOG	Gaussian/NA

[1] Models included an over dispersion parameter.

[2] Fallen tree data were not collected in the 0% treatment.

For results reported on the natural log (ln) scale (from GLIMMIX Procedure in SAS and natural log transformed data), 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 −1.5 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. For results that were statistically significant, we present the back-transformed proportional differences in summary by section.

6-5. RESULTS

6-5.1. TREE FALL AND LARGE WOOD RECRUITMENT RATES

6-5.1.1. Pre-harvest Rates

We found extensive between-site variation in pre-harvest tree fall rates for both RMZs and PIPs (**Figure 6-2**; **Appendix Table 6-A-1**). Site mean tree fall rates for RMZs ranged from 0 to 239.9 trees/ha/yr (0–97.1 trees/acre/yr) and 0 to 22.5 m^2/ha/yr of basal area (0–97.8 ft^2/acre/yr). Variability was greater in the PIPs, ranging from 0 to 360.5 trees/ha/yr (0–145.9 trees/acre/yr) and 0 to 34.9 m^2/ha/yr of basal area (0–151.9 ft^2/acre/yr). The largest proportion of sites had tree fall rates of 49.4 trees/ha/yr (20 trees/acre/yr) and <4.6 m^2/ha/yr of basal area (20 ft^2/acre/yr), but we observed much higher rates in a small proportion of the sites. The block mean tree fall rates for RMZs and PIPS, and the variability between sites within blocks, were highest in the two coastal blocks (Willapa 1 and Willapa 2; **Figure 6-3**).

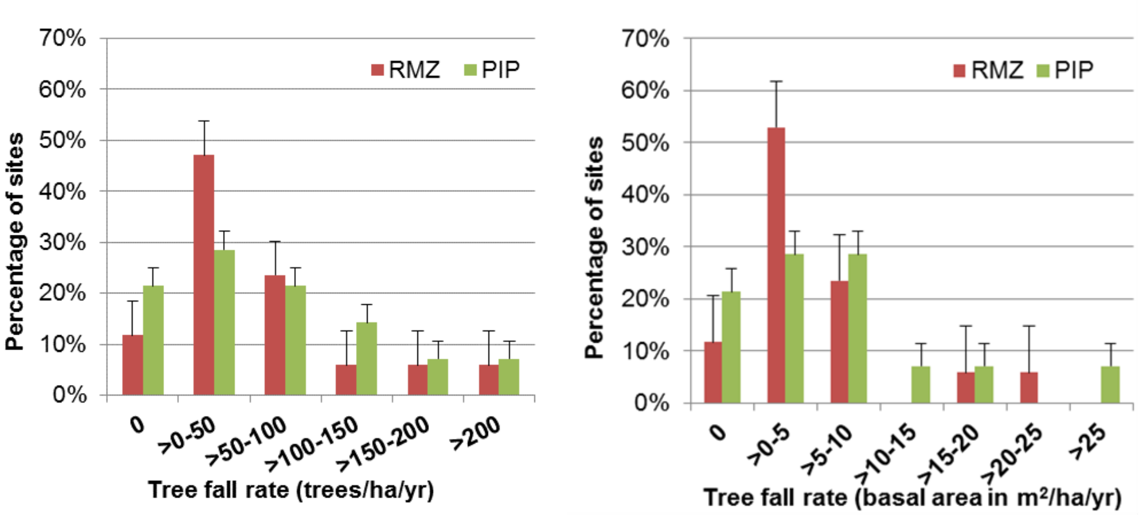

Figure 6-2. Percentage of sites by mean tree fall rate in trees/hectare/yr (left panel) and basal area in m^2/hectare/yr (right panel) for RMZs and PIPs in the pre-harvest period.

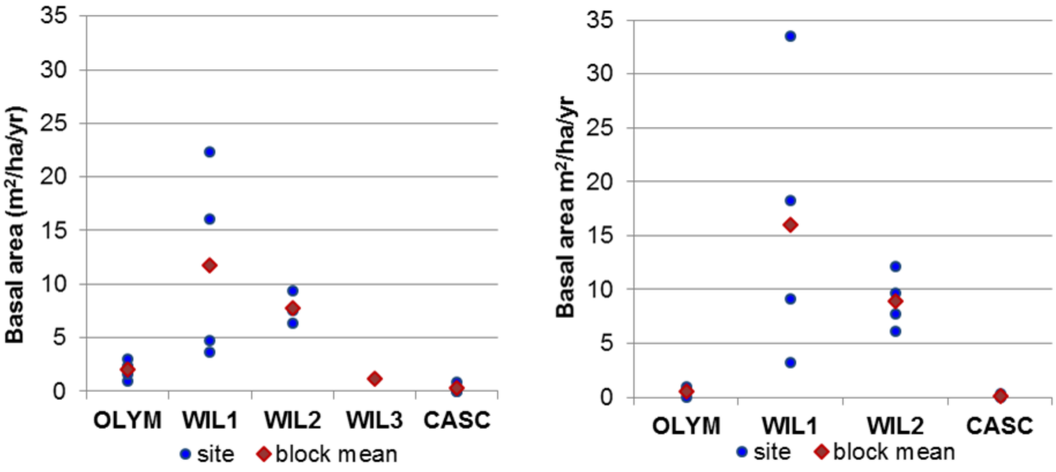

Figure 6-3. Pre-harvest site mean tree fall rates in basal area for RMZs (left panel) and PIPs (right panel) by block.

LW recruitment rates during the pre-harvest period were highly variable (**Appendix Table 6-A-2**). LW recruitment rates for RMZs ranged from 0 to 121.6 pieces/ha/yr (0–49.2 pieces/acre/yr) and recruitment volume ranged from 0 to 18.0 m^3/ha/yr (0–256.6 ft^3/acre/yr). LW recruitment rates were similar in the PIPs, with counts ranging from 0 to 131.2 pieces/ha/yr (0–1 pieces/acre/yr) and volumes from 0 to 16.2 m^3/ha/yr (0–232.1 ft^3/acre/yr). Recruitment rates for most sites were ≤49.4 pieces/ha/yr (20 pieces/acre/yr) and ≤3.5 m^3/ha/yr (50 ft^3/acre/yr) for both RMZs and PIPs (**Figure 6-4**).

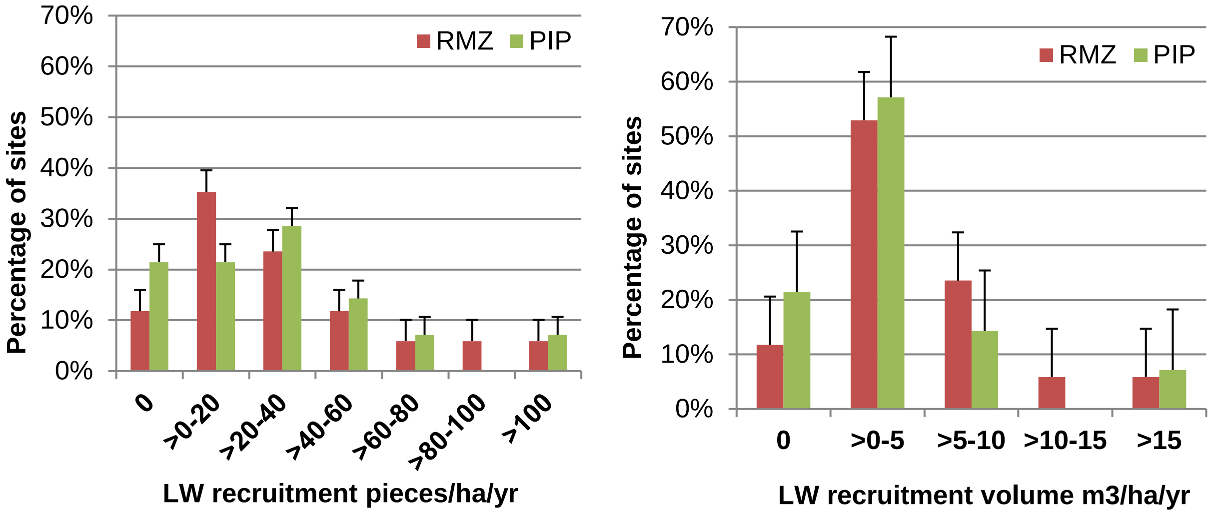

Figure 6-4. Percentage of sites by LW recruitment piece count (left panel) and volume (right panel) for RMZs and PIP in the pre-harvest period.

The differences in pre-harvest LW recruitment rates between blocks followed a pattern similar to that for tree fall rates (**Figure 6-5**), probably because LW recruitment results from the subset of fallen trees that reach the channel. LW recruitment rates were highest in the Willapa 1 and Willapa 2 blocks where extensive tree fall occurred during the December 2007 windstorm.

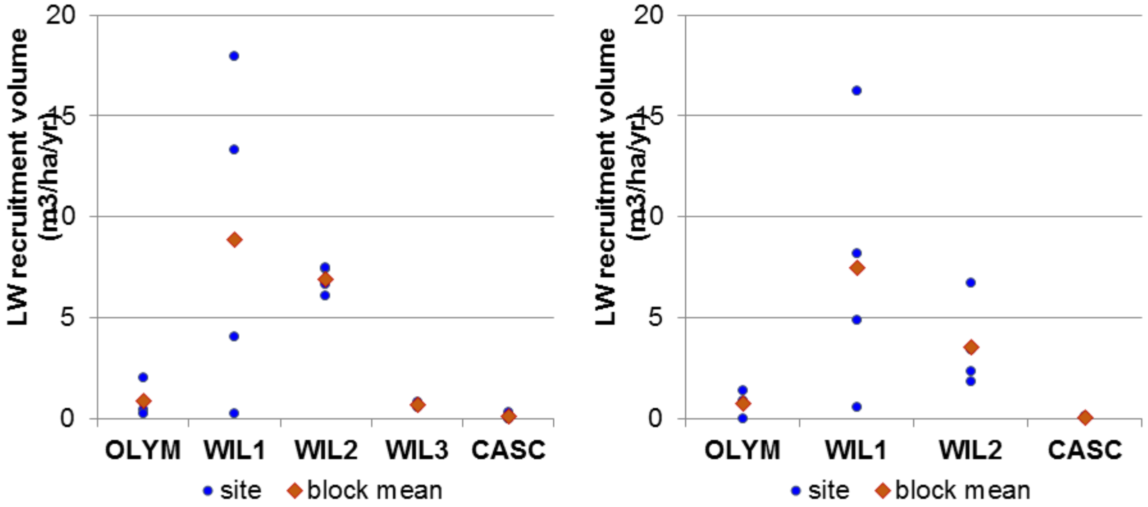

Figure 6-5. Site mean LW recruitment rates for RMZs (left panel) and PIPs (right panel) by block for the pre-harvest period.

6-5.1.2. Post-Harvest Rates and Treatment Comparisons

Post-harvest tree fall rates in the reference RMZs were lower and less variable than in the buffer treatments (**Appendix Table 6-A-3)**, ranging from 5.9 to 37.3 trees/ha/yr (2.4–15.1 trees/acre/yr). Tree fall rates in the 100% treatment RMZs were more variable, ranging from 7.7 to 76.4 trees/ha/yr (3.1–30.9 trees/acre/yr), with the highest values in the Willapa 1 and Willapa 2 blocks (**Figure 6-6**, left panel). We observed the greatest variability in the FPB RMZs, where tree fall rates ranged from 4.2 to 152.5 trees/ha/yr (1.7–61.7 trees/acre/yr). Tree fall rates in reference RMZs were lower than in 100% treatment RMZs in three of four blocks, and lower than FPB RMZs in two of three blocks (**Figure 6-6**, right panel). The tree fall rates in 100% RMZs were less than FPB RMZs in the two blocks where both treatments were present. Differences in tree fall rates between blocks were also evident. In general, post-harvest tree fall rates were lower in the Willapa 3 and South Cascade blocks and higher in the Willapa 1, Willapa 2 and Olympic blocks.

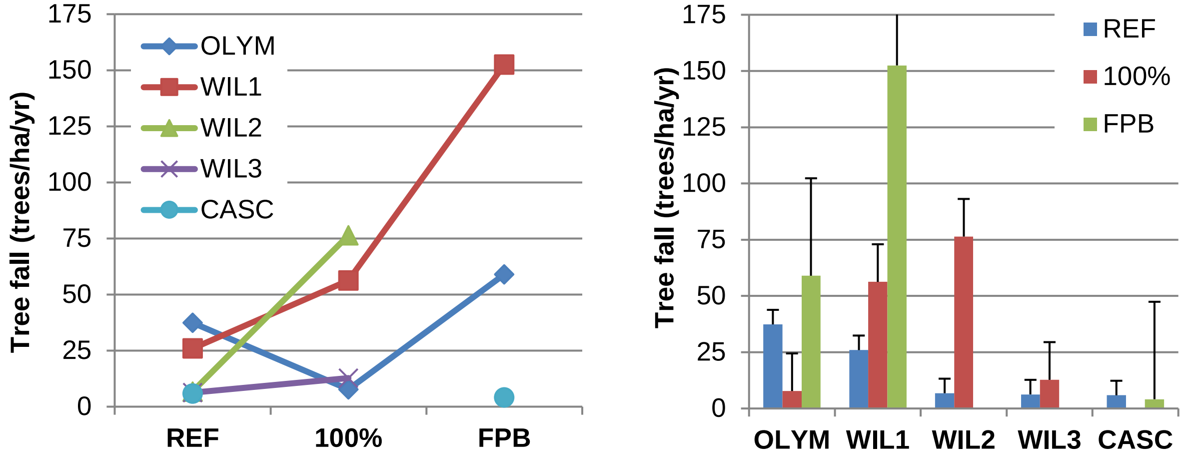

Figure 6-6. Post-harvest RMZ tree fall rates by treatment (left panel) and block (right panel).

Post-harvest tree fall rates in the reference PIPs were lower and less variable than in the buffer treatments, ranging from 1.7 to 11.1 trees/ha/yr (0.7–4.5 trees/acre/yr; **Figure 6-7**). Tree fall in the 100% PIPs was more variable, ranging between 8.4 to 93.2 trees/ha/yr (3.4–37.7 trees/acre/yr), with the highest values in the Willapa 1 and Willapa 2 blocks. Tree fall rates for the FPB PIPs were highly variable, ranging from 3.7 to 102.3 trees/ha/yr (1.5–41.4 trees/acre/yr).

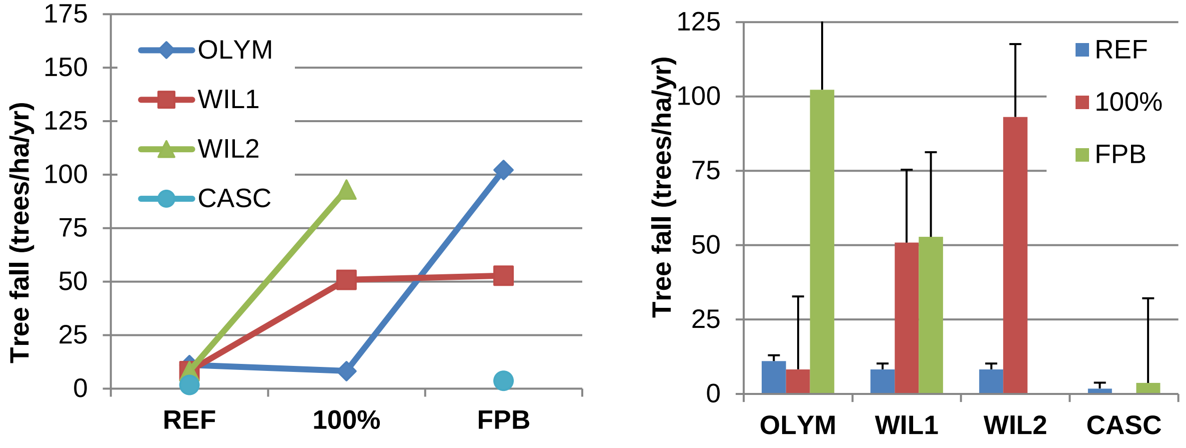

Figure 6-7. Post-harvest tree fall rates in PIPs by treatment (left panel) and block (right panel).

The mean post-harvest tree fall rate for RMZs, measured as tree count per year (%STC/yr), was lowest for the reference, intermediate for the 100% treatment, and highest for the FPB (**Table 6-5**). The P-value for the reference vs. 100% treatment comparison was 0.34, although the mean tree fall rate for the 100% treatment RMZs was nearly double the mean for reference RMZs. The mean tree fall rate for FPB RMZs was five times greater than for the references (P = 0.02) and nearly three times greater than for the 100% treatment (P = 0.06).

Table 6-5. Mean post-harvest tree fall rates as percent of standing tree count per year (%STC/yr) RMZs and PIPs by treatment, with standard errors (SE) and P-values for treatment comparisons.

Plot Type	REF		100%		FPB		Treatment Contrast P-values		
	Mean (%)	SE (%)	Mean (%)	SE (%)	Mean (%)	SE (%)	100% vs. REF	FPB vs. REF	FPB vs. 100%
RMZ	3.0	1.6	5.6	2.3	16.4	5.4	0.34	0.02	0.06
PIP	1.3	0.9	9.2	3.0	15.7	4.8	0.03	0.01	0.27

Mean post-harvest tree fall rates for PIPs followed the same pattern as for RMZs, with the lowest rates for the reference PIPs, intermediate rates for the 100% treatment PIPs, and highest rates for the FPB PIPs (**Table 6-5**). The mean tree fall rates for the 100% treatment and FPB PIPs were seven and 12 times the reference rate, respectively (P ≤0.03). Although mean percent tree fall by count was nearly two times greater in FPB PIPs compared to 100% PIPs, there was high variability and the rates were not significantly different (P = 0.27).

Post-harvest LW recruitment volumes in reference RMZs were relatively low, ranging from 0.7 to 2.2 m^3/ha/yr (10–32 ft^3/acre/yr; **Figure 6-8**). Post-harvest LW recruitment volumes were generally higher and more variable in the 100% and FPB RMZs, ranging from 0.3 to 14.0 m^3/ha/yr (5–200 ft^3/acre/yr) in the 100% treatment and 0 to 7.6 m^3/ha/yr (0–109 ft^3/acre/yr) in the FPB. Mean LW recruitment volume in the 100% RMZs was greater than the reference rate in two of four blocks and the mean recruitment volume in the FPB RMZs was greater than the reference rate in two of three blocks. Post-harvest recruitment volumes in the reference PIPs were low, ranging from 0 to 0.3 m^3/ha/yr (0–4.3 ft^3/acre/yr) and with little variability. Recruitment volumnes in the 100% and FPB PIPs were more variable, ranging from 0.1 to 5.2 m^3/ha/yr (0.9–74 ft^3/acre/yr) and 0 to 3.9 m^3/ha/yr (0–56 ft^3/acre/yr), respectively (**Figure 6-8**). Site mean post-harvest LW recruitment rates are shown in **Appendix Table 6-A-4**.

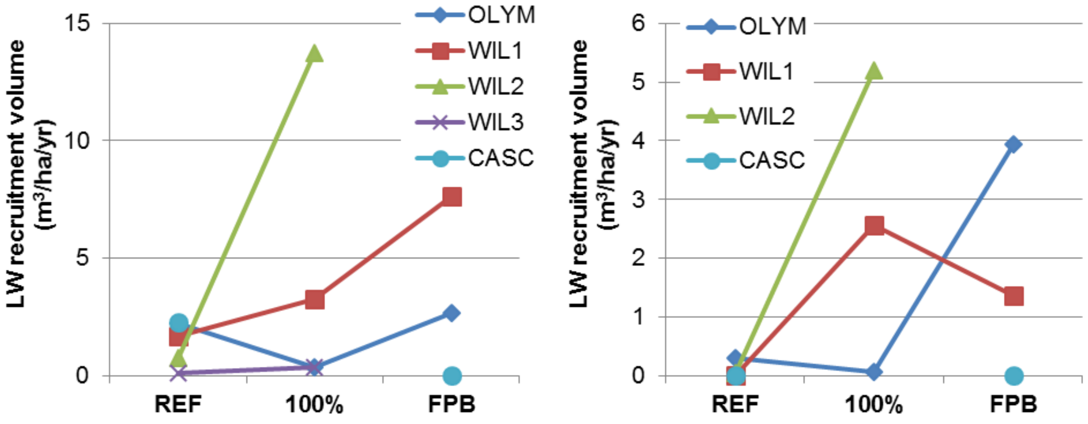

Figure 6-8. Site mean post-harvest LW recruitment volume for RMZs (left panel) and PIPs (right panel) by block and treatment.

LW recruitment rates in the 100% treatment RMZs were about twice the reference rates by piece count and volume (**Table 6-6**). A high value for the WIL2-100% influenced the overall 100% treatment mean (**Figure 6-8**). The recruited piece count for the FPB RMZs was nearly three times the reference rate, but the recruited volume was similar to the reference rate, indicating that recruited LW pieces in the FPB were greater in number but smaller in volume than those in the reference RMZs. There was substantial variability in LW recruitment rates among sites in both the 100% treatment and FPB. Consequently, the P-values for the 100% vs. reference, FPB vs. reference and FPB vs. 100% comparisons were generally high (P ≥0.35), except for the FPB vs. reference comparison for piece count (**Table 6-7**). LW recruitment volume was lower in the 0% RMZs than in all other treatments (P ≤0.04). LW recruitment by piece count was also lower in the 0% treatment than the 100% treatment (P = 0.09) and FPB (P = 0.03).

Table 6-6. Mean post-harvest LW recruitment for RMZs and PIPs by piece count and median volume by treatment. SE is the standard error.

Treatment	RMZs			PIPs		
	Mean Count pieces/ha/yr (pieces/ac/yr)	SE	Median Volume m^3/ha/yr (ft^3/ac/yr)	Mean Count pieces/ha/yr (pieces/ac/yr)	SE	Median Volume m^3/ha/yr (ft^3/ac/yr)
REF	10.1 (4.1)	5.9	1.20 (17.21)	1.7 (0.7)	4.9	0.075 (1.07)
100%	19.3 (7.8)	7.2	1.97 (28.2)	21.7 (8.8)	6.2	1.49 (21.25)
FP	27.7 (11.2)	8.6	1.32 (18.83)	15.8 (6.4)	6.2	0.92 (13.12)
0%	0.2 (0.1)	7.2	0.025 (0.36)	0.0 (0.0)	6.2	0.027 (0.39)

Table 6-7. P-values for post-harvest comparisons of mean LW recruitment rates among treatments for RMZs and PIPs.

Contrast	RMZ P-values		PIP P-values	
	Piece Count	Volume	Piece Count	Volume
100% vs. REF	0.35	0.62	0.03	0.04
FPB vs. REF	0.13	0.94	0.11	0.08
0% vs. REF	0.31	0.02	0.84	0.70
0% vs. FPB	0.03	0.04	0.10	0.06
0% vs. 100%	0.09	0.01	0.03	0.03
FPB vs. 100%	0.48	0.74	0.51	0.69

For PIPs, LW recruitment in the 100% treatment was over 12 times the reference rate by piece count ($P = 0.03$) and 30 times the reference rate by volume ($P = 0.04$). Recruitment in the FPB PIPs was also high, over nine times the reference rate by piece count ($P = 0.08$) and 18 times the reference rate by volume ($P = 0.11$). LW recruitment rates for the FPB and 100% treatments were similar ($P > 0.5$ for recruitment by piece count and volume). We recorded no LW recruitment in the 0% PIPs and very little in the reference PIPs, with no difference in recruitment rates detected for either piece count of volume ($P \geq 0.70$). The LW recruitment rates for the 100% PIPs were greater than for the 0% treatment ($P = 0.03$ for piece count and volume). Similarly, LW recruitment rates for the FPB were higher than the rates for the 0% treatment ($P = 0.10$ and 0.06, for piece count and volume, respectively).

6-5.1.3. Summary of Treatment Effects on Wood Recruitment

There was evidence of a treatment effect on tree fall rates in both RMZs and PIPs. In the RMZs, the FPB rates were significant higher than the reference rates, while in the PIPs tree fall rates in both the 100% treatment and FPB were significantly higher compared to the reference ($P < 0.05$ for both comparisons). There was also evidence of a higher rate in FPB RMZs compared to the 100% treatment ($P = 0.06$); however, there was no significant difference in rates between the 100% and FPB PIPs (**Table 6-8**).

We observed significantly higher rates of LW recruitment by piece count in 100% treatment than the reference PIPs ($P < 0.05$). We saw evidence of higher post-harvest LW recruitment rates in both the 100% treatment and FPB than in the 0% treatment where all trees were removed; however, we detected no difference between the 100% treatment and FPB. The patterns were similar for LW recruitment by volume, with higher rates for both the 100% treatment and FPB compared to the reference rates ($P < 0.05$ and < 0.10, respectively).

Table 6-8. Summary results for statistical comparisons of wood recruitment metrics between experimental treatments. ***alpha ≤0.05, ** alpha >0.05 and ≤0.10, * alpha >0.10 and ≤0.20; + indicates that the estimated value for the treatment with less buffering is significantly more than the treatment with more buffering; - indicates that the estimated value for the treatment with less buffering is significantly less than the treatment with more buffering.

Response		**100%**	**FPB†**	**0%**	**Response**		**100%**	**FPB†**	**0%**
Tree Fall - RMZ	REF		+***	na	Tree Fall - PIP	REF	+***	+***	na
	100%		+**	na		100%			na
	FPB			na		FPB			na
LW Recruitment (count) RMZ	REF		+*		LW Recruitment (count) -PIP	REF	+***	+*	
	100%			-**		100%			-***
	FPB			-***		FPB			-**
LW Recruitment (volume) - RMZ	REF			-***	LW Recruitment (volume) - PIP	REF	+***	+**	
	100%			-***		100%			-***
	FPB			-***		FPB			-**

6-5.2. FACTORS AFFECTING TREE FALL AND LARGE WOOD RECRUITMENT RATES

6-5.2.1. Pre-Harvest Period

Wind accounted for over 90% of the tree fall observed during the pre-harvest period. The differences in tree fall and LW recruitment rates among blocks appear to be associated with regional differences in the frequency and magnitude of windstorms during the pre-harvest period, as was the case for tree mortality (see Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers*). Weather station data showed that windstorms were stronger and more frequent along the coast than further inland during the pre-harvest period. Both Astoria and Hoquiam (coastal stations) recorded hurricane-force winds (33–50 m/s [74–112 mph]) during the December 2007 storm and storm-force winds (25–33 m/s [55–73 mph]) on four other days, while winds further inland in Portland did not reach storm-force during the pre-harvest period. Sites in the Willapa 1 and Willapa 2 blocks that were most affected by the December 2007 windstorm had a greater proportion of plots with LW recruitment, higher variability in recruitment rates, and a greater number of plots with recruitment rates greater than 7.0 m^3/ha/yr (100 ft^3/acre/yr; **Figure 6-9**). On average, tree fall rates were approximately seven times higher, and LW recruitment was about five and a half times higher by piece count and 14 times higher by volume, in the eight wind-affected sites compared to the nine remaining sites (**Table 6-9**).

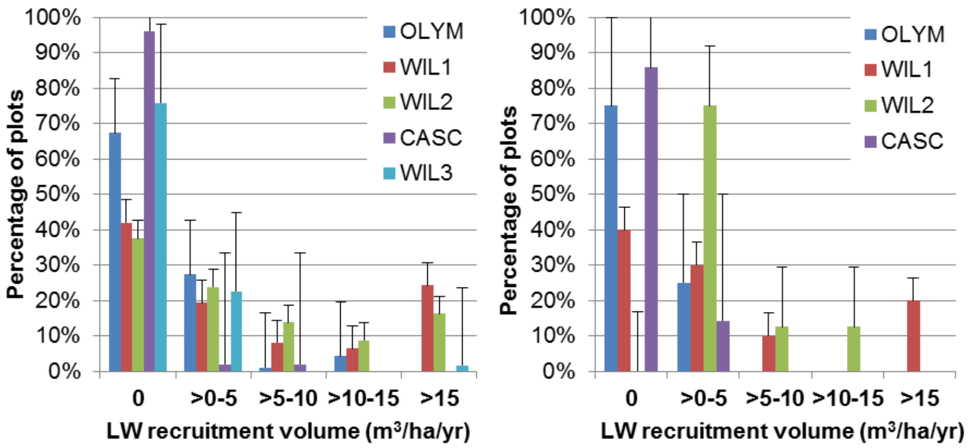

Figure 6-9. Distribution of pre-harvest LW recruitment volume for RMZ (left panel) and PIP plots (right panel) by block.

Table 6-9. Comparison of mean tree fall and LW recruitment rates in sites most vs. least affected by the December 2007 windstorm.

Metric	Least Affected	Most Affected
Fallen trees/ha	12.57	90.98
Fallen trees/100 m	3.83	27.73
% fallen trees	2.3	16.5
Fallen basal area m^2/ha	0.90	6.27
Fallen basal area m^2/100 m	0.27	1.91
% fallen basal area	1.5	12.0
LW pieces/ha	9.85	55.54
LW pieces/100m	3.00	16.93
LW volume m^3/ha	0.56	7.91
LW volume m^3/100 m	0.17	2.41

6-5.2.2. Post-Harvest Period

LW recruitment rates in the post-harvest period differed by treatment and block (**Figure 6-10**) due to differences in mortality and tree fall rates. In the 100% treatment and the FPB RMZs, post-harvest LW recruitment rates tended to be highest in the coastal blocks (Willapa 1 and Willapa 2), while rates in the three non-coastal blocks were lower and similar to pre-harvest rates. Overall, RMZ LW recruitment rates increased in two of three FPB sites following harvest, with large increases in the WIL1-FP site and a small increase in the OLYM-FP. We did not observe any LW recruitment in the South Cascade FPB RMZs during either interval. Post-harvest LW recruitment rates decreased sharply in the WIL1-REF and WIL2-REF. In contrast, post-harvest rates were higher at the OLYM-REF and CASC-REF, and we observed little change in the WIL3-REF.

PIP LW recruitment rates followed a similar pattern. In FPB PIPs, post-harvest LW recruitment rates increased in the OLYM-FP, remained the same in the CASC-FP, and decreased in the WIL1-FP. Large wood recruitment rates decreased in the WIL1-100% and OLYM-100% PIPs, and increased in the WIL2-100% PIPs. Post-harvest rates in the reference PIPs decreased in the coastal blocks, while there was little change in the non-coastal blocks. Post-harvest LW recruitment rates were very low in the 0% sites because few trees remained to provide potential wood recruitment; however, post-harvest rates do not include wood input that occurred during timber harvest.

There was extensive variation in LW recruitment volume among plots. The plot-scale distributions for the 100% treatment and FPB differed from that of references, having a greater percentage of plots with higher LW recruitment rates. This difference was more distinct for the PIPs **(Figure 6-11)**.

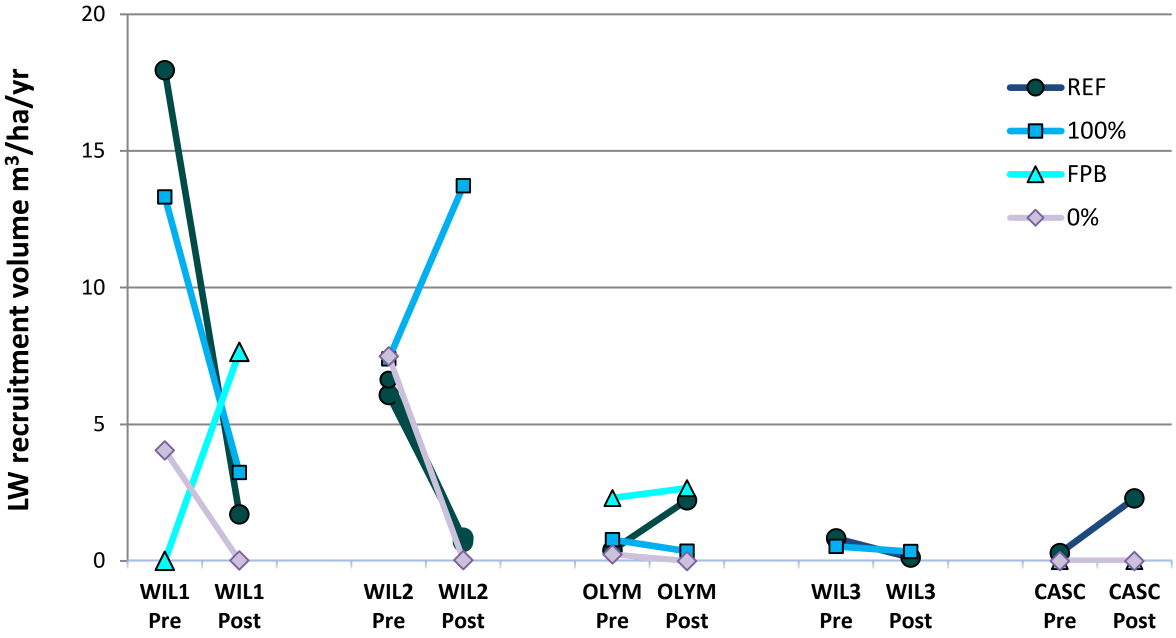

Figure 6-10. Changes in RMZ LW recruitment volume from the pre- to the post-harvest period by treatment and block.

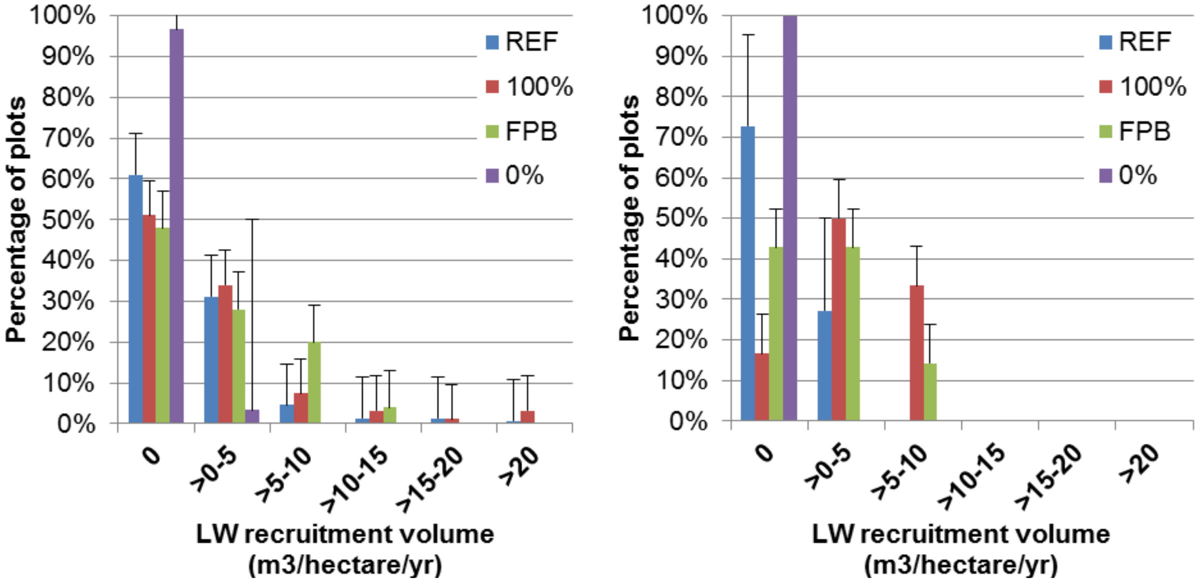

Figure 6-11. Distribution of post-harvest LW recruitment rates for RMZ (left panel) and PIP plots (right panel) by treatment for recruited volume in m^3/ha/yr.

6-5.3. CHARACTERISTICS OF FALLEN TREES AND LARGE WOOD RECRUITMENT

The proportion of fallen trees that recruited to the bankfull channel was just below 50% during the pre-harvest period (**Table 6-10**). The proportion of trees that recruited was similar by count during the post-harvest period, while the percent by basal area increased by around 8%, indicating that the trees that recruited to the channel during the post-harvest period were somewhat larger than the trees that did not.

Table 6-10. Mean percent of total fallen trees that recruited to the bankfull channel in the pre- and post-harvest periods.

	Pre-harvest		**Post-harvest**	
	Count	**Basal Area**	**Count**	**Basal Area**
Recruited	49.4%	48.2%	51.1%	56.2%
Not recruited	50.6%	51.8%	48.9%	43.8%

There was a contrast among treatments in the contribution of recruiting trees with distance from stream. Source distance patterns for the pre-harvest RMZ and the REF RMZs in the post-harvest period were similar (**Figure 6-12**). In both of these cases, where the RMZ was embedded in an unharvested second growth forest, over 70% of the fallen trees that recruited came from within 7.6 m of the stream. In the 100% treatment and FPB, a much smaller proportion of the recruited trees came from within 7.6 m (55% and 35 %, respectively) and the contribution from the outer half of the buffer (7.6−15.2 m) was more substantial (**Figure 6-12**).

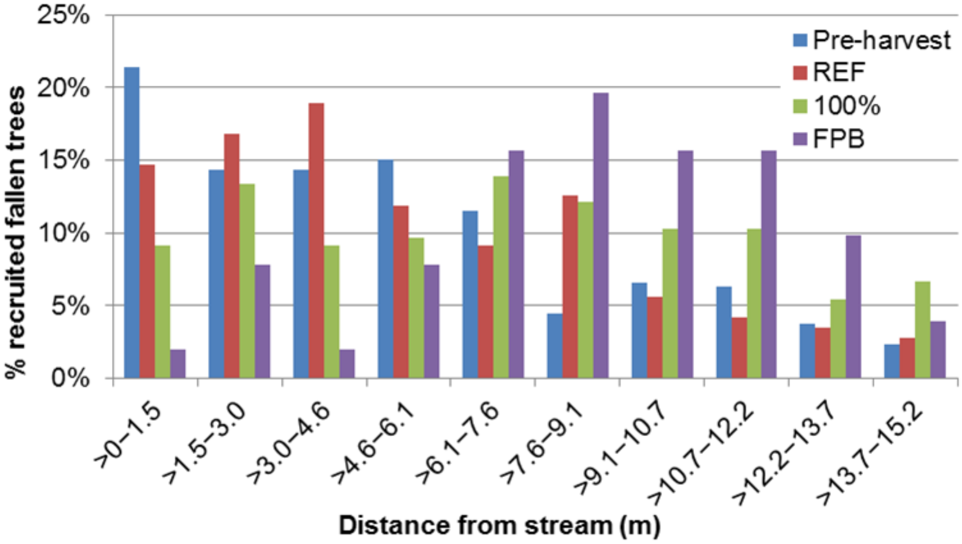

Figure 6-12. Source distances for recruited fallen trees in 15.2-m (50-ft) RMZs (percentage of total recruited fallen trees by 1.5-m [5-ft] distance increments), comparing pre-harvest values with post-harvest values for the REF, 100% treatment and FPB.

Of those trees that reached the channel, only around 10% came to rest so that a portion of the stem intruded within the bankfull channel, while approximately 90% came to rest with the entire tree suspended or spanning above the channel (**Table 6-11**). The proportion intruding within the bankfull channel was slightly higher in the post-harvest period.

Table 6-11. Mean percent of recruited trees that came to rest within the channel versus suspended or spanning above the channel in the pre- and post-harvest periods.

	Pre-harvest		Post-harvest	
	Count	**Basal Area**	**Count**	**Basal Area**
Within bankfull channel	9.6%	10.3%	11.3%	14.7%
Above bankfull channel	90.4%	89.7%	88.7%	85.3%

The proportion of fallen trees that recruited within the bankfull channel during the post-harvest period was similar between the reference and 100% treatment, and lower for the FPB (**Table 6-12**). Lower mean values for the FPB were driven by a single low value for the FPB in the South Cascade block, where only five trees fell during the post-harvest period and none recruited. The proportion by basal area was greater for all treatments, indicating a tendency for larger trees to reach the channel.

Table 6-12. Mean percent of fallen trees that recruited in the post-harvest period by treatment.

Treatment	Count	Basal Area
REF	51.5%	56.3%
100%	47.0%	53.4%
FPB	26.6%	33.1%

Over three quarters of all trees that fell during the pre-harvest period were uprooted, while the remainder were broken off along the stem. Uprooted trees made up an even greater proportion (84%) of the fallen trees that recruited to the bankfull channel (**Table 6-13**). Uprooted trees made up a high proportion of the fallen trees that came to rest entirely over the channel (87%), and a much lower proportion of fallen trees that intruded within the bankfull channel (57%). Overall, uprooted trees were more likely to be suspended or spanning the channel, while broken stems were more likely to intrude within the channel. In the post-harvest period, uprooted trees made up a greater proportion of total tree fall in the 100% treatment and FPB compared to the reference (**Table 6-14**).

Table 6-13. Pre-harvest recruitment contributions by fall type.

	All Fallen Trees	Total Recruited	Within Channel	Over Channel
Uprooted	76.7%	84.1%	57.4%	86.9%
Broken	23.3%	15.9%	42.6%	13.1%

Table 6-14. Post-harvest recruitment contributions by fall type.

	REF	100%	FPB
Uprooted	79.6%	85.6%	87.3%
Broken	20.4%	14.4%	12.7%

Table 6-15. Percent of recruited LW intruding within the bankfull channel by piece count and volume for RMZs and PIPs, pre- and post-harvest.

Metric	RMZs		PIPs	
	Pre-harvest	Post- harvest	Pre-harvest	Post-harvest
Recruited LW piece count	13.8%	16.3%	9.2%	4.2%
Recruited LW volume	6.9%	9.4%	7.1%	0.8%

Most newly recruited LW pieces were suspended or spanning above the bankfull channel. The percentage of recruited LW volume that intruded within the bankfull channel was lower than the percentage by piece count for both RMZs and PIPs (**Table 6-15**), likely because only a small portion of a piece typically intruded within the bankfull channel. The percentages of recruited pieces and volume that intruded within the bankfull channel were greater for the RMZs than for the PIPs, perhaps because channels were typically narrower near the PIP.

Only a small percentage of newly recruited LW pieces performed an in-channel function (**Table 6-16**), consistent with the fact that the percentage of pieces intruding within the bankfull channel was small. There was an increase in the percentages of LW contributing to pool, step and debris jam formation in RMZs in the post-harvest period. Conversely, there was a decrease in the percentage of LW contributing to sediment retention and debris jam formation in the PIPs in the post-harvest period.

Table 6-16. Mean percentage of newly recruited LW pieces performing in-channel functions in RMZs and PIPs in the pre- and post-harvest periods.

Function	RMZs		PIPs	
	Pre-harvest	Post-harvest	Pre-harvest	Post-harvest
Pool formation	1.1%	5.6%	2.8%	0.0%
Sediment retention	9.6%	9.3%	12.7%	6.3%
Step formation	1.1%	4.4%	0.0%	0.0%
Debris jam formation	5.1%	12.9%	9.9%	6.3%

To document the effects of the intense windstorms that occurred in December 2007, we collected additional data on all trees recruited to the stream channel during the winter of 2007–2008. Forty-nine percent (49%) of the trees that were recruited during the storm were 16 m or greater in length and 28% were between 8 and 16 m in length. Forty-six percent (46%) of recruited trees were in the greater than 10 to 25 cm diameter class (mid-point diameter), and 99% were between 2 and 50 cm in diameter. Only 1% of recruited trees were greater than 50 cm in diameter. About half (53%) of the recruited trees did not provide in-channel functions because they were suspended over the bankfull channel. Forty-four percent (44%) of the recruited trees contributed to hydraulic roughness that affected stream flow direction or velocity, but in most cases the stems were suspended over the channel and branches extending into the channel provided in-channel function. When only the trunk of the tree was considered (not including branches and roots), 83% of recruited trees were positioned above the bankfull channel (55% spanning and 27% suspended). The lower portion of the tree stem with few to no branches was most often spanning the bankfull channel (47%), followed by the uppermost portion of the tree with attached branches and needles (37%), the upper portion of tree with attached leafless branches (13%) and the rootwad of the tree (4%). Only 3% of recruited trees contributed to bank stability or step formation. Ninety-five percent (95%) were decay class 1, which suggests that those trees were living prior to falling.

6-5.4. CHANNEL WOOD LOADING

6-5.4.1. Small Wood

We estimated the pre- to post-harvest change within treatment for the natural log (ln) of SW pieces per stream meter (**Table 6-17** and **Figure 6-13**). The amount of change in the number of SW pieces per meter depended on treatment (P <0.001). We estimated the change in the 100%, FP and 0% treatments to be different from the change in the reference (P <0.05, 0.07 and <0.0001, respectively; **Table 6-18**). We estimated the change in the 0% treatment to be different from the changes in the FP (P = 0.08) and 100% treatments (P = 0.01).

Table 6-17. The within-treatment change, standard error (SE) of the estimate and 95% confidence intervals (CI) for the natural log (ln) of the number of SW and functional SW pieces per meter, i.e., [ln(post) − ln(pre)].

Treatment	SW/m				Functional SW/m			
	Estimate	SE	95% CI		Estimate	SE	95% CI	
			Lower	Upper			Lower	Upper
REF	0.06	0.16	−0.26	0.39	−0.04	0.15	−0.34	0.27
100%	0.52	0.15	0.20	0.83	0.48	0.15	0.18	0.78
FP	0.58	0.23	0.12	1.04	0.50	0.22	0.06	0.94
0%	1.07	0.15	0.78	1.37	0.79	0.15	0.49	1.09

The pre- to post-harvest change within treatment was estimated for the natural log (ln) of the number of functional (i.e., contributing to step formation, bank stability and/or hydraulic roughness) SW pieces per stream meter (**Table 6-17** and **Figure 6-13**). The amount of change in the number of functional SW pieces per meter depended on treatment (P <0.01). We estimated the change in the 100%, FP, and 0% treatments to be different from the change in the reference (P = 0.02, 0.05 and <0.001, respectively; **Table 6-18**).

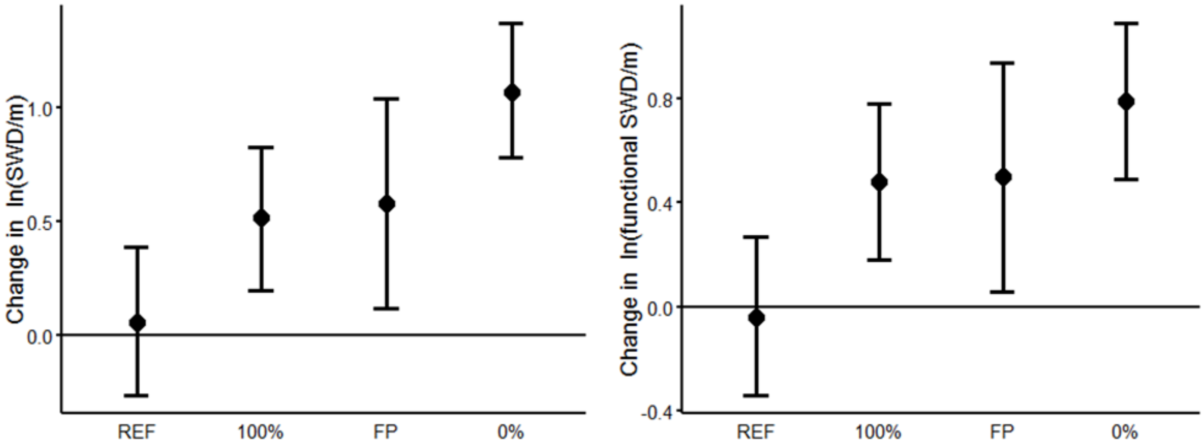

Figure 6-13. The within-treatment change and 95% confidence intervals (CI) for the natural log (ln) of the number of SW pieces per meter (left panel) and number of functional SW pieces per meter (right panel), i.e., [ln(post) − ln(pre)].

Table 6-18. The between-treatment difference in the change, standard error (SE) of the estimate and 95% confidence intervals (CI) of the natural log (ln) of the number of SW and functional SW pieces per stream meter, i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	SW/m					Functional SW/m				
	Estimate	SE	P-value	95% CI		Estimate	SE	P-value	95% CI	
				Lower	Upper				Lower	Upper
100% vs. REF	0.45	0.22	0.05	0.00	0.91	0.52	0.21	0.02	0.09	0.94
FP vs. REF	0.52	0.28	0.07	−0.05	1.08	0.54	0.26	0.05	0.00	1.07
0% vs. REF	1.01	0.22	<0.0001	0.57	1.45	0.83	0.21	<0.001	0.40	1.26
0% vs. FP	0.49	0.27	0.08	−0.05	1.04	0.29	0.26	0.27	−0.24	0.83
0% vs. 100%	0.56	0.21	0.01	0.12	0.99	0.31	0.21	0.14	−0.11	0.73
FP vs. 100%	0.06	0.27	0.82	−0.49	0.62	0.02	0.26	0.95	−0.51	0.55

6-5.4.2. Large Wood

We estimated the pre- to post-harvest change within treatment for the natural log (ln) of the number of LW pieces per stream meter (**Table 6-19** and **Figure 6-14**). The amount of change in the number of LW pieces per meter depended on treatment ($P < 0.01$). We estimated the changes in 100%, FP and 0% treatments to be different from the change in the reference ($P < 0.001$, 0.03 and <0.01, respectively; **Table 6-20**).

We estimated the pre- to post-harvest change within treatment for the natural log (ln) of the number of LW functional pieces per stream meter (**Table 6-19** and **Figure 6-14**). The amount of change in the number of functional LW pieces per meter depended on treatment ($P < 0.01$). We estimated the change in the 100%, FP and 0% treatments to be different from the change in the reference ($P < 0.001$, 0.05 and 0.03, respectively; **Table 6-20**).

Table 6-19. The within-treatment change, standard error (SE) of the estimate and 95% confidence intervals (CI) for the natural log (ln) of the number of LW and functional LW pieces per meter, i.e., [ln(post) – ln(pre)].

Treatment	**LW/m**				**Functional LW/m**			
	Estimate	**SE**	**95% CI**		**Estimate**	**SE**	**95% CI**	
			Lower	**Upper**			**Lower**	**Upper**
REF	0.13	0.09	−0.05	0.31	0.03	0.08	−0.14	0.2
100%	0.63	0.09	0.45	0.81	0.5	0.08	0.33	0.67
FP	0.49	0.14	0.21	0.76	0.34	0.13	0.08	0.59
0%	0.51	0.1	0.31	0.70	0.31	0.09	0.12	0.49

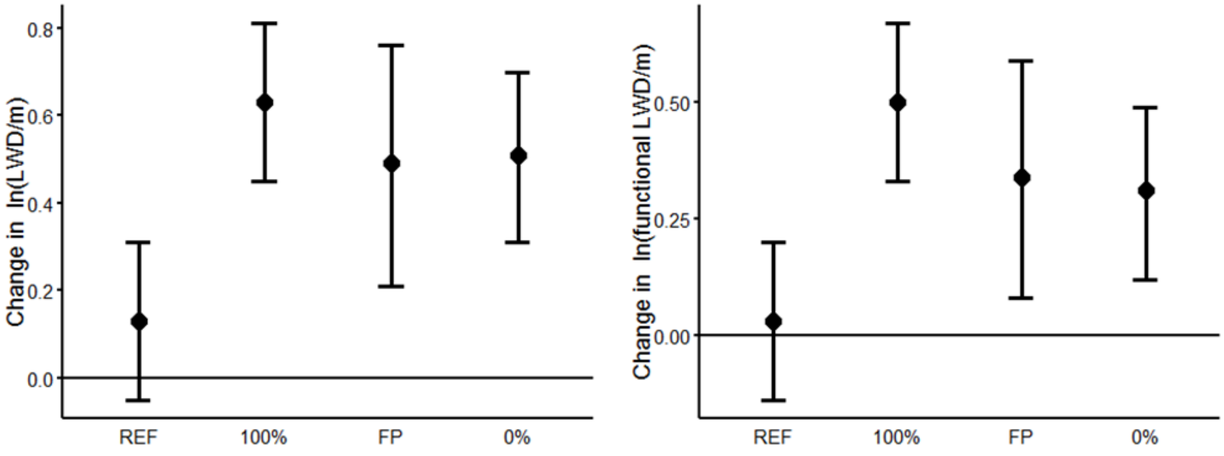

Figure 6-14. The within-treatment change, standard error (SE) of the estimate and 95% confidence interval (CI) for the natural log (ln) of the number of LW pieces per meter (left panel) and number of functional LW pieces per meter (right panel), i.e., [ln(post) – ln(pre)].

Table 6-20. The between-treatment difference in the change, standard error (SE) of the estimate and 95% confidence intervals (CI) of the natural log (ln) of the number of LW and functional LW pieces per stream meter, i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	**LW/m**					**Functional LW/m**				
	Estimate	**SE**	**P-value**	**95% CI**		**Estimate**	**SE**	**P-value**	**95% CI**	
				Lower	**Upper**				**Lower**	**Upper**
100% vs. REF	0.50	0.13	<0.001	0.24	0.76	0.47	0.12	<0.001	0.23	0.71
FP vs. REF	0.36	0.16	0.03	0.03	0.69	0.31	0.15	0.05	0.00	0.61
0% vs. REF	0.38	0.13	0.01	0.11	0.65	0.28	0.12	0.03	0.03	0.53
0% vs. FP	0.02	0.17	0.92	−0.32	0.36	−0.03	0.16	0.86	−0.35	0.29
0% vs. 100%	−0.12	0.13	0.36	−0.39	0.15	−0.19	0.12	0.13	−0.44	0.06
FP vs. 100%	−0.14	0.16	0.40	−0.47	0.19	−0.16	0.15	0.29	−0.47	0.14

6-5.5. CHARACTERISTICS OF WOOD CONTRIBUTING TO LOADING

In addition to our primary study objective of comparing post-harvest changes in SW and LW loading by treatment, we collected secondary data to describe the proportion of pieces by diameter class and stream function, and compare between areas that were and were not obstructed by inputs of newly recruited wood from logging debris and windthrow.

6-5.5.1. Small Wood Characteristics

The site-wide average number of SW pieces per stream meter ranged from 2 to 8 in the pre-harvest period. In the post-harvest period, SW ranged from 4 to 8 in the reference, 6 to 19 in the 100% treatment, 4 to 10 in the FP treatment, and 6 to 42 in the 0% treatment (**Appendix Table 6-A-5**). Average number of pieces per meter by treatment ranged from 4 to 6 in the pre-harvest period, whereas the post-harvest increase in the numbers of pieces per meter averaged 0 in the reference, 4 in the 100%, 3 in the FP and 11 in the 0% treatment (**Figure 6-15**). Though the total number of SW pieces increased in all buffer treatments in the post-harvest period, the proportion of total wood pieces classified as SW was similar across treatments and periods, ranging from 78% of total wood pieces in the reference in the pre-harvest period to 84% in the 0% treatment in the post-harvest period (**Figure 6-16**).

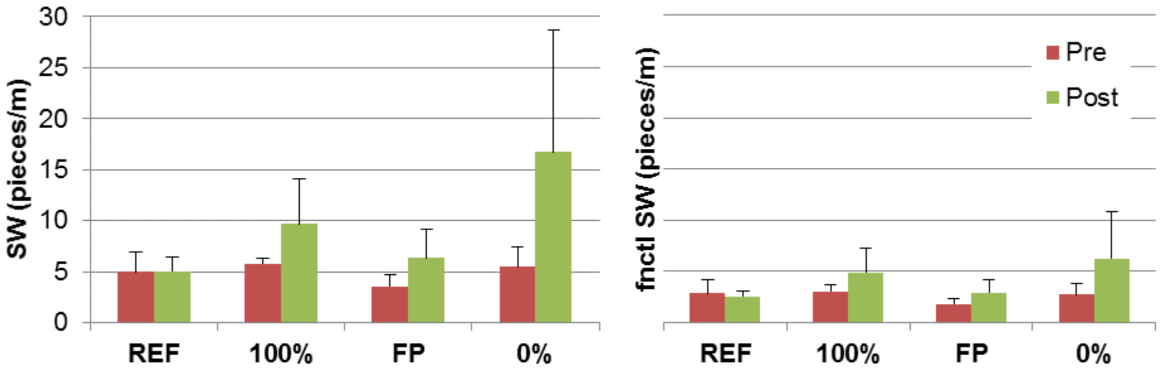

Figure 6-15. The average number of SW pieces/m (left panel) and functional SW pieces/m (right panel) by treatment for the pre- and post-harvest periods.

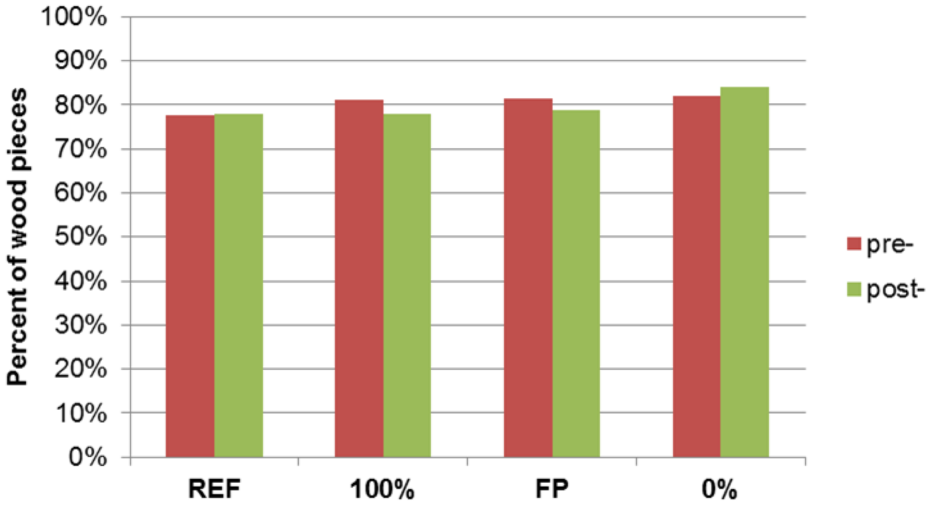

Figure 6-16. Percent of all wood pieces meeting the definition of SW for the pre- and post-harvest periods by treatment.

The site-wide average number of functional SW pieces per stream meter ranged from 1 to 5 in the pre-harvest period. In the post-harvest period, functional SW ranged from 2 to 4 in the reference, 3 to 11 in the 100% treatment, 2 to 5 in the FP treatment, and 2 to 17 in the 0% treatment (**Appendix Table 6-A-5**). Average number of functional pieces per meter by treatment ranged from 2 to 3 in the pre-harvest period, whereas the post-harvest change in the numbers of pieces per meter averaged 0 in the reference, 2 in the 100%, 1 in the FP and 4 in the 0% treatment (**Figure 6-15**). We noted an increase in the proportion of SW pieces contributing to hydraulic roughness in all buffer treatments (**Figure 6-17**), while the proportion of pieces that were loose in the stream (i.e., not anchored) decreased.

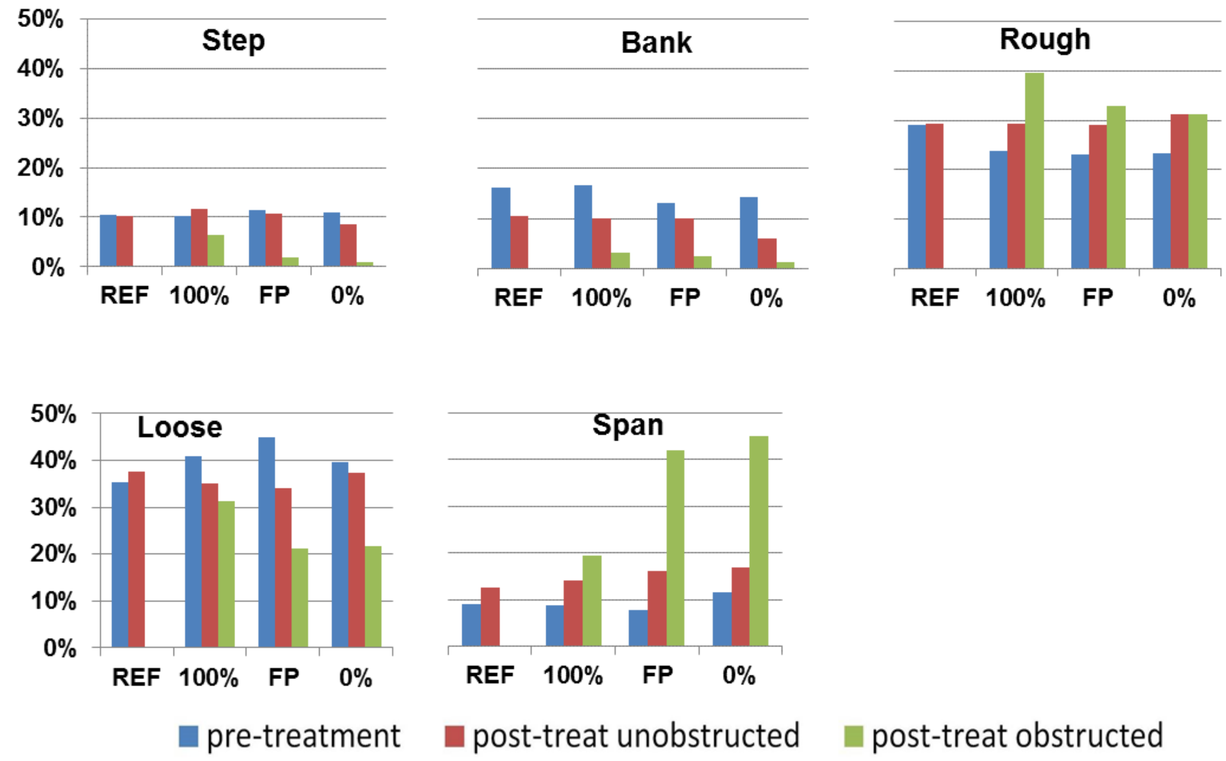

Figure 6-17. Percent of SW contributing to instream function in the pre-harvest period, and stream reaches unobstructed and obstructed by logging debris and windthrow in the post-harvest period, by treatment. Note the absence of obstructed reaches in references.

6-5.5.2. Large wood characteristics

The average number of LW pieces per stream meter ranged from 1 to 2 in the pre-harvest period. In the post-harvest period, LW ranged from 1 to 2 in the reference, 1 to 5 in the 100% treatment, 1 to 2 in the FP treatment, and 1 to 5 in the 0% treatment (**Appendix Table 6-A-5**). The average number of pieces per meter was 1 for all treatments in the pre-harvest period, whereas the post-harvest increase in the numbers of pieces per meter averaged 0 in the reference versus 1 in all three buffer treatments (**Figure 6-18**). By far the greatest proportion of LW observed in both the pre- and post-harvest periods was in the smallest diameter class (10–25 cm), followed by the 26–50 cm; 50–100 cm; and >100 cm classes, regardless of treatment (**Figure 6-19**). The proportion of pieces in the smallest LW diameter class ranged from 68% to 72% across treatments in the pre-harvest period and from 70% to 74% in the post-harvest period.

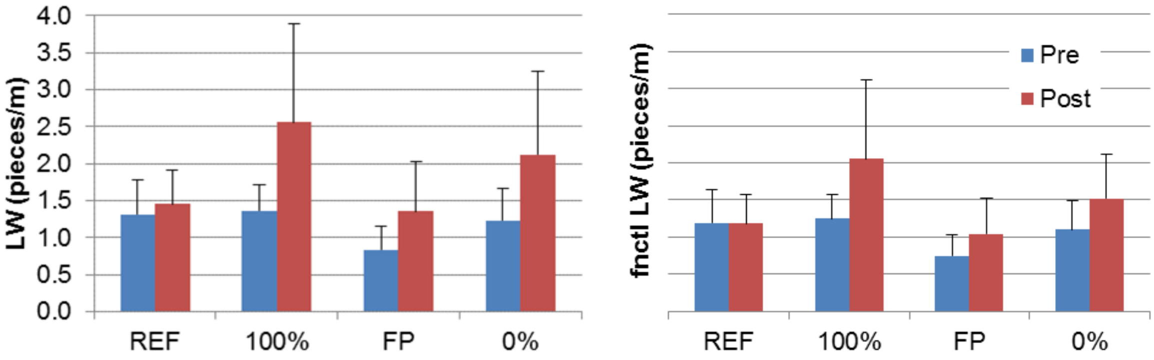

Figure 6-18. The average number of LW pieces/m (left panel) and functional LW pieces/m (right panel) by treatment for the pre- and post-harvest periods.

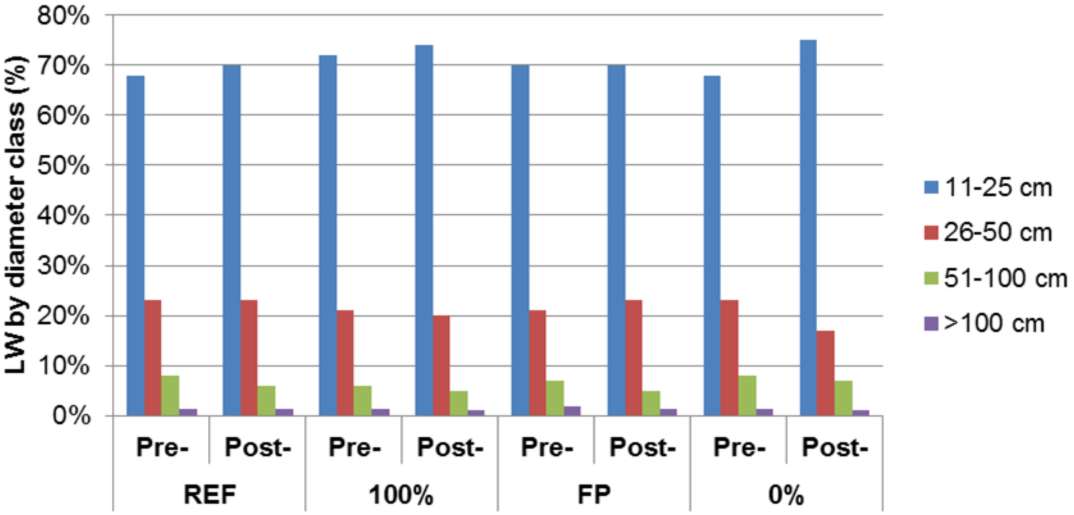

Figure 6-19. Percent of LW by diameter class and treatment for the pre- and post-harvest periods.

The greatest proportion of LW pieces in reaches that were not obstructed by logging debris and windthrow contributed to bank stability in both the pre- and post-harvest periods (**Figure 6-20**). Overall, we observed a decrease in the proportion of LW pieces contributing to bank stability and loose in the stream in the post-harvest period, regardless of treatment. Conversely, there was an increase in the number of pieces contributing to hydraulic roughness and spanning the stream.

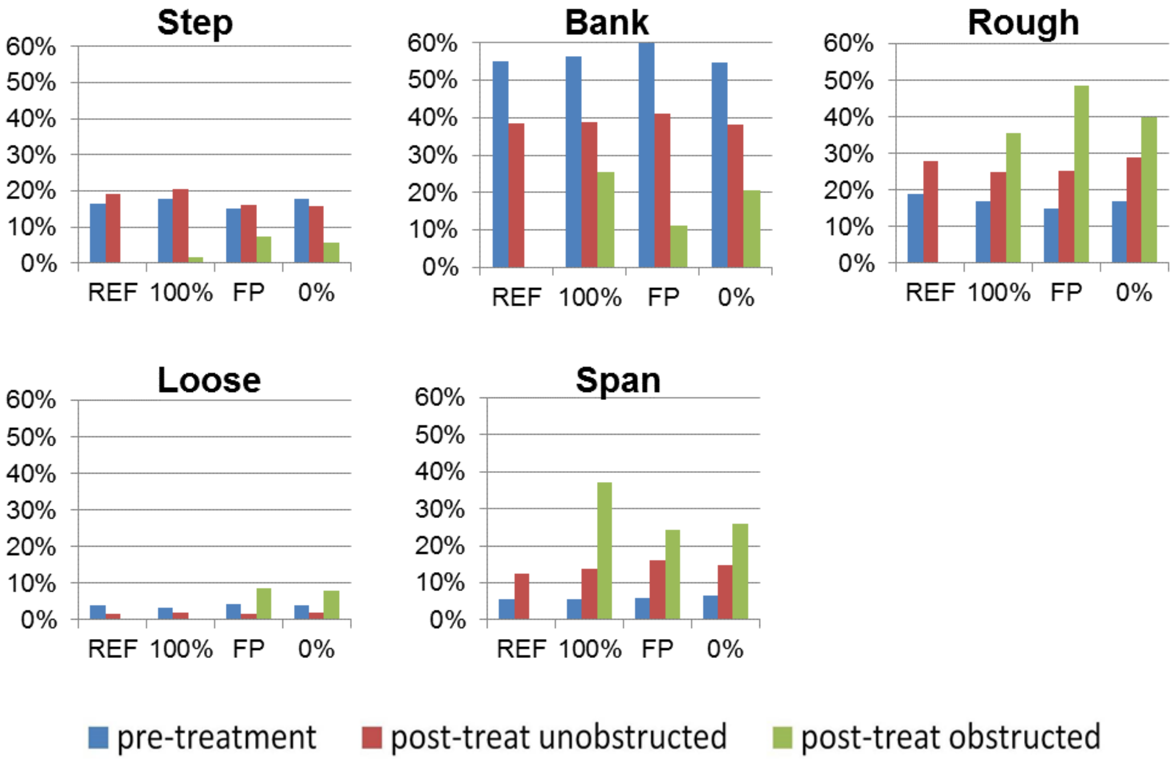

Figure 6-20. Percent of LW contributing to instream function in the pre-harvest period, and stream reaches unobstructed and obstructed by logging debris and windthrow in the post-harvest period, by treatment. Note the absence of obstructed reaches in references.

6-5.5.3. Wood Accumulations in Obstructed Stream Reaches

Stream reaches obstructed by logging debris and windthrow only occurred in buffer treatments in the post-harvest period. The length of the stream channel impacted varied by treatment and study site (see **Table 6-2**). We classified 91% of all wood pieces in obstructed reaches as SW. We estimated the number of SW pieces per meter to be up to eight times greater in obstructed than unobstructed reaches in the first post-harvest year and up to 15 times greater in the second post-harvest year (**Table 6-21**). The proportion of SW pieces contributing to hydraulic roughness and spanning the channel was greater in obstructed reaches (**Figure 6-17**). Conversely, the proportion of pieces contributing to step formation and bank stability, and that were loose in the stream, was less.

We estimated the number of LW pieces per meter to be up to 2.5 times greater in obstructed than unobstructed stream reaches in the first post-harvest year and up to 5 times greater in the second post-harvest year (**Table 6-21**). The proportion of LW pieces contributing to step formation and bank stability was less in obstructed reaches, while those contributing to hydraulic roughness, loose, and spanning the stream were greater in obstructed reaches (**Figure 6-20**). Greater than 80% of LW pieces in obstructed reaches were in the smallest diameter class (10–25 cm diameter).

Table 6-21. The number of SW and LW pieces per stream meter in unobstructed (unobs) and obstructed (obs) stream reaches for study sites with obstructed reaches in the first (2009) and second (2010) post-harvest years. NA indicates that the obstructed length of the stream channel was less than 5% of the total length; as a result, we did not sample any plots.

		SW (pieces/m)				**LW (pieces/m)**			
		First Post-harvest Year		**Second Post-harvest Year**		**First Post-harvest Year**		**Second Post-harvest Year**	
Treatment	**Block**	**Unobs**	**Obs**	**Unobs**	**Obs**	**Unobs**	**Obs**	**Unobs**	**Obs**
100%	WIL1	6.7	17.7	5.7	21.7	2.0	2.3	2.0	3.0
	WIL2	NA	NA	14.3	38.7	NA	NA	3.6	6.3
FP	OLYM	NA	NA	4.7	27.5	NA	NA	1.2	1.8
	WIL1	5.7	47.8	4.6	72.0	1.9	4.8	1.6	8.1
	CASC	3.5	7.2	2.6	50.3	0.7	0.7	0.6	3.2
0%	OLYM	6.9	29.6	7.7	39.6	1.1	2.3	1.5	3.7
	WIL1	8.8	21.9	5.6	35.1	1.4	2.1	1.2	3.3
	WIL2	9.9	10.3	15.3	78.8	1.6	2.4	3.6	3.2

6-5.5.4. Newly Recruited Wood Cover

We anticipated different responses in amounts of newly recruited wood cover by treatment and by year (first- or second-year post-harvest). For the first post-harvest year, we hypothesized that there would be the greatest amount of newly recruited wood in the 0% treatment, lesser amounts in the FP and 100% treatments, and the lowest levels in the references. Alternatively, in the second post-harvest year, we hypothesized that newly recruited wood would be greatest in the 100% treatment, followed by the FP treatment, reference and 0% treatment, respectively.

The percent of the stream channel length covered by newly recruited wood in the first post-harvest year ranged from 1 to 10% in the reference, 4 to 32% in the 100% treatment, 24 to 38% in the FP treatment, and 23 to 43% in the 0% treatment (**Appendix Table 6-A-6**). The natural log (ln) of the percent of the stream channel length covered differed between treatments ($P = 0.001$; **Table 6-22** and **Figure 6-21**). We observed the greatest amount of newly recruited wood in the 0% treatment, followed by the FP treatment, 100% treatment and reference, respectively. The percent of stream channel covered by new wood was estimated to be greater in the FP and 0% treatments than the reference ($P < 0.01$ for both comparisons; **Table 6-23**).

The percent of the stream channel length covered by newly recruited wood in the second post-harvest year ranged from 0 to 11% in the reference, 1 to 15% in the 100% treatment and 0 to 10% in the FP treatment, and was 0% in all four of the 0% treatments (**Appendix Table 6-A-6**). The natural log (ln) of the percent of the stream channel length covered differed between treatments (P <0.01; **Table 6-22** and **Figure 6-21**). We observed the least amount of newly recruited wood in the 0% treatment, followed by the reference, FP and 100% treatments, respectively. The percent of stream channel covered by new wood differed between the 0% treatment and the reference (P = 0.03), 100% (P <0.01), and FP treatments (P = 0.03; **Table 6-23**).

Table 6-22. The natural log (ln) of the percent of the stream channel length covered by newly recruited wood (<1 year old), standard error (SE) of the estimate and 95% confidence intervals (CI) for the first and second post-harvest years, by treatment.

Treatment	**First Post-harvest Year**				**Second Post-harvest Year**			
	Estimate	**SE**	**95% CI**		**Estimate**	**SE**	**95% CI**	
			Lower	**Upper**			**Lower**	**Upper**
REF	1.38	0.29	0.75	2.00	0.76	0.50	−0.49	2.00
100%	2.51	0.35	1.75	3.28	1.58	0.54	0.30	2.85
FP	3.46	0.41	2.58	4.35	1.05	0.59	−0.29	2.38
0%	3.56	0.35	2.80	4.33	−0.77	0.55	−2.05	0.51

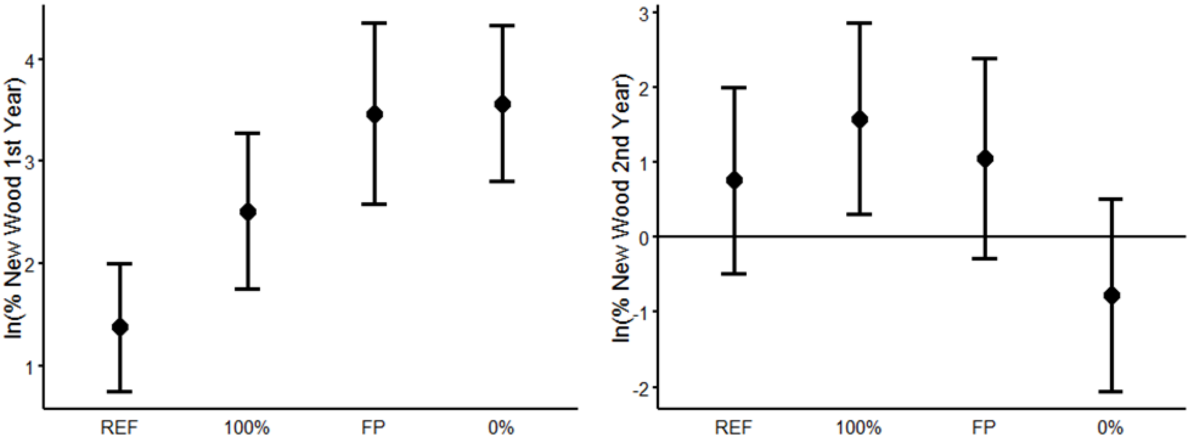

Figure 6-21. The natural log (ln) of the percent of the stream channel length covered by newly recruited wood (<1 year old) and 95% confidence intervals (CI) for the first and second post-harvest years.

Table 6-23. The between-treatment difference in the natural log (ln) of the percent of the stream channel length covered by newly recruited wood (<1 year old), standard error (SE) of the estimate and 95% confidence intervals (CI) for the first and second post-harvest years.

Contrast	First Post-harvest Year					Second Post-harvest Year				
	Est.	SE	P-value	95% CI		Est.	SE	P-value	95% CI	
				Lower	Upper				Lower	Upper
100% vs. REF	1.14	0.46	0.11	−0.20	2.48	0.82	0.43	0.30	−0.53	2.17
FP vs. REF	2.09	0.50	<0.01	0.62	3.55	0.29	0.50	0.94	−1.27	1.85
0% vs. REF	2.19	0.46	<0.01	0.85	3.52	−1.53	0.44	0.03	−2.89	−0.17
0% vs. FP	0.10	0.54	1.00	−1.49	1.68	−1.82	0.51	0.03	−3.41	−0.23
0% vs. 100%	1.05	0.50	0.21	−0.42	2.51	−2.35	0.49	<0.01	−3.88	−0.81
FP vs. 100%	0.95	0.54	0.33	−0.63	2.53	−0.53	0.55	0.77	−2.23	1.18

6-5.6. SUMMARY OF TREATMENT EFFECTS ON WOOD LOADING

On average, we estimated a between-treatment increase of 60% (95% CI: 0–150%), 70% (95% CI: 0–190%) and 170% (95% CI: 80–330%) in the number of SW pieces per stream meter in the 100%, FP and 0% treatments compared with the reference, respectively. Likewise, we estimated a between-treatment increase of 60% (95% CI: 30–110%), 40% (95% CI: 0–100%) and 50% (95% CI: 10–90%) in the number of LW pieces per stream meter in the 100%, FP and 0% treatments compared with the reference, respectively. This appears to be due to wood inputs associated with post-harvest tree fall in buffered reaches and timber harvest operations in the clearcut reaches of the RMZ.

We also noted a between-treatment increase in total SW in the 0% treatment that was 70% greater (10–170%) than the change in the 100% treatment and 60% greater (95% CI: −10% to 180%) than the change in the FP treatment. This pattern appears to be explained by differences among buffer treatments in the proportion of the RMZ where clearcut harvest occurred. We did not observe a similar difference among buffer treatments for LW pieces, apparently because the input of LW from tree fall in the buffered reaches offset the input of LW from harvest operations in the clearcut reaches. We detected no differences between the 100% and FP treatments for either total SW or total LW loading.

The pattern for abundance of functional SW pieces was similar to total SW pieces, with significant increases across all buffer treatments in the post-harvest period (P <0.01). Compared with the change in the reference, there was a 70% (95% CI: 10–160%), 70% (95% CI: 0–190%) and 130% (95% CI: 50–250%) increase in functional SW pieces in the 100%, FP and 0% treatments, respectively. Although all buffer treatments had more functional post-harvest SW than the reference, there was some evidence that the 0% and 100% treatments may have also differed (P = 0.14), with functional SW estimated to be 40% (95% CI: −10% to 110%) greater in the 0% than 100% treatment. The estimated increase in functional wood was due primarily to an increase in the number of pieces contributing to hydraulic roughness, especially in stream reaches obstructed due to high levels of new wood recruitment associated with harvest

The pattern for functional LW was consistent with our observations for total LW. We noted a between-treatment increase in functional LW pieces in all buffer treatments relative to the reference, with a 60% (95% CI: 30–100%), 40% (95% CI: 0–80%), and 30% (95% CI: 0–70%) increase in the 100%, FP and 0% treatments, respectively. There was also some evidence of a difference among buffer treatments, with a lesser amount of post-harvest functional LW in the 0% than in the 100% treatment (P = 0.13; **Table 6-24**).

The greatest differences in newly recruited wood cover in the post-harvest period were observed in the first post-harvest year (P = 0.001). We observed an increase in newly recruited wood between the reference and the FP and 0% buffer treatments (P <0.05). We estimated that wood cover in the 0%, FP and 100% treatments was 8.9 (95% CI: 2.3–33.9; P = 0.001), 8.1 (95% CI: 1.9–35.0; P = 0.001), and 3.1 (95% CI: 0.8–11.9; P = 0.11) times greater than in the reference, respectively. Increases in wood cover were typically due to the input of logging debris in the clearcut portions of the RMZ and windthrow in stream reaches with riparian buffers. Newly recruited wood decreased in the second post-harvest year; however, input of new wood cover was significantly less in the 0% treatment than in the reference, 100% or FP treatments (P <0.05). The amount of wood cover in the second post-harvest year was only 10% (95% CI: 0 - 40%) to 20% (95% CI: 0 - 80%) of the levels observed in all other treatments, including the reference.

Table 6-24. Summary results for statistical comparisons of wood loading metrics between experimental treatments. Results for the period × treatment contrasts were statistically significant (P <0.01) for all responses. *** alpha ≤0.05, ** alpha >0.05 and ≤0.10, * alpha >0.10 and ≤0.20; + indicates that the estimated value for the treatment with less buffering is significantly more than the treatment with more buffering; - indicates that the estimated value for the treatment with less buffering is significantly less than the treatment with more buffering.

Response		**100%**	**FP**	**0%**	**Response**		**100%**	**FP**	**0%**
SW total pieces	REF	+***	+**	+***	LW total pieces	REF	+***	+***	+***
	100%			+***		100%			
	FP			+**		FP			
SW functional pieces	REF	+***	+***	+***	LW functional pieces	REF	+***	+***	+***
	100%			+*		100%			-*
	FP					FP			
New Wood Cover 1st year	REF	+*	+***	+***	New Wood Cover 2nd year	REF			-***
	100%					100%			-***
	FP					FP			-***

6-6. DISCUSSION

6-6.1. SHORT-TERM TREATMENT EFFECTS

6-6.1.1. Tree Fall and Wood Recruitment

We anticipated that post-harvest tree fall and LW recruitment rates would be similar for FPB and 100% treatment RMZs because the only difference between buffer treatments was the proportion of the RMZ length buffered. However, the results for RMZs were not as expected. Tree fall rates in the 100% and reference RMZs were similar, and both were significantly lower than in the FPB RMZs. The same pattern was observed in tree mortality (see Chapter 5 − *Stand Structure and Tree Mortality Rates in Riparian Buffers*). We have no explanation for this unexpected pattern and are unsure whether it will persist through time. We expected rates to be lower in reference RMZs and PIPs because they were embedded in unharvested second-growth stands more protected from wind. As expected, tree fall and LW recruitment were significantly higher in the FPB and 100% PIPs compared with reference PIPs, with no significant difference between the two buffer treatments. Post-harvest LW recruitment rates in the 0% treatment were near zero for both RMZs and PIPs. Tree fall and LW recruitment rates during the first two post-harvest years followed a pattern similar to that observed during the first three post-harvest years in the Westside Type N Buffer Characteristics, Integrity and Function (BCIF) study (Schuett-Hames *et al.* 2012). Both studies documented higher tree fall (**Table 6-25)** and LW recruitment (**Table 6-26)** in RMZs and PIPs buffered according to Forest Practices rules than in unharvested references. However, the tree fall and LW recruitment rates documented in this study were higher than those in the BCIF study for both reference and FPB RMZs. The only instance in which a rate was higher in the BCIF study was for LW recruitment rate in the FPB PIPs.

Table 6-25. Comparison of tree fall rates (percent of standing trees per acre per year) in this study with the Westside Type N Buffer Characteristics, Integrity and Function (BCIF) study.

	Reference RMZs	**FPB RMZs**	**Reference PIPs**	**FPB PIPs**
BCIF Study	0.6%	6.4%	-	12.2%
Type N Study	3.0%	16.4%	1.3%	15.7%

Table 6-26. Comparison of LW recruitment rates in this study with the Westside Type N Buffer Characteristics, Integrity and Function (BCIF) study.

	Reference RMZs pieces/ha/yr (pieces/ac/yr)	**FPB RMZs pieces/ha/yr (pieces/ac/yr)**	**Reference PIPs pieces/ha/yr (pieces/ac/yr)**	**FPB PIPs pieces/ha/yr (pieces/ac/yr)**
BCIF Study	2.0 (0.8)	16.8 (6.8)	-	32.9 (13.3)
Type N Study	10.1 (4.1)	27.7 (11.2)	1.7 (0.7)	15.8 (6.4)

The pre-harvest windstorm in December 2007 provided an unexpected opportunity to document the effects of hurricane-force winds on tree fall and wood recruitment in young (~40–60 year old) streamside stands embedded in continuous second-growth forests, and to contrast these effects with the response of trees in post-harvest riparian buffers. The pre-harvest windstorm caused extensive tree fall from windthrow and stem breakage in the Willapa 1 and Willapa 2 blocks, resulting in much higher rates of tree fall and LW recruitment than in the remaining blocks. On average, the pre-harvest tree fall rates in sites most affected by the storm were seven times higher than those in unaffected sites, and LW recruitment rates were five and 14 times higher by piece and volume, respectively. A similar response was observed in unharvested references from the BCIF study that were affected by the same storm (Schuett-Hames *et al.* 2012).

Wind disturbance resulted in extensive within-site variability. Plot-scale LW recruitment rates for sites most affected by the pre-harvest windstorm had a bimodal distribution: around 40% of plots had no LW recruitment, while 30–40% had LW recruitment rates in excess of 7.0 m^3/ha/yr (100 ft^3/ac/yr). There was also extensive variability in post-harvest tree fall and LW recruitment at the plot-scale in buffers affected by windthrow.

As with the tree mortality rates (see Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers*), it appears the combined effect of treatment with windstorm frequency and magnitude affected the pattern of response. Little tree fall occurred in buffers in the absence of storm-force winds. However, in sites that experienced storm-force winds, tree fall rates generally increased in exposed buffers, but remained low in unharvested references. In contrast, when hurricane-force winds occurred, tree fall rates increased in riparian buffers and in riparian stands embedded in unharvested forests. A similar pattern was observed in the BCIF study. Storm-force winds during the first three years after harvest resulted in tree fall and LW recruitment rates in FP buffers that were significantly greater than in the riparian areas of unharvested references. However, the hurricane-force winds of the December 2007 windstorm, which occurred four years after harvest in the BCIF study, resulted in increased tree fall and LW recruitment rates in both the FP treatment and reference RMZs (Schuett-Hames *et al.* 2012).

Most (>80%) of newly recruited wood pieces spanned, or were suspended over, the channel. Wood often spans the channel in headwater streams due to relatively narrow channel width and confinement. Direct input into the channel may not occur until a log is either broken or fragmented (Nakamura and Swanson 1993). We observed a post-harvest increase in the numbers of LW that were spanning the bankfull channel in all treated sites, especially in clearcut stream reaches obstructed with high amounts of logging debris. Obstructed stream reaches had a higher proportion of pieces that spanned the stream channel, up to 37% of pieces compared to an average of 6% in the pre-harvest period and 14% in reaches that were not obstructed by logging debris in the post-harvest period. Most spanning and suspended pieces did not immediately contribute to in-channel hydraulic functions such as pool formation, sediment storage, or step formation, and did not interact with water flowing in the channel, except where branches protruding from the stems intruded into the channel. In the near-term, these spanning and suspended pieces provide shade and cover to the channel, as well as a source of SW as the bark and branches slough into the channel. Over a longer period, the stems are likely to drop into the bankfull channel; however, there is uncertainty about how long this will take, how decayed the wood will be when it reaches the channel, and what functions it will provide.

6-6.1.2. Wood Loading

Consistent with the results from other studies, we anticipated that wood load and cover would increase with decreasing riparian buffer length (Jackson *et. al* 2001; Olson and Rugger 2007). We observed the greatest increase in newly recruited wood cover in the 0% treatment and clearcut portions of the FP treatment in the first year post-harvest. We estimated that an average of 35% of the stream length was completely covered by new wood. Others have documented higher levels of logging debris immediately following harvest along headwater streams without riparian buffers. For example, Jackson and colleagues (2001) found that 98% of the channel length was buried after logging in headwaters with clearcut harvest to the stream edge. Differing results between the studies may be associated with site or stand conditions (e.g., site slope), harvest techniques and yarding systems, or differences in the implementation of Forest Practices guidelines intended to minimize logging debris inputs during harvest.

Jackson and colleagues (2001) found that maintaining a riparian buffer along headwater streams limited recruitment of logging debris to the stream following timber harvest. Maxa (2009) and Schuett-Hames and colleagues (2012) also observed lower levels of wood in Type N streams with 50-ft (15.2-m) buffers compared to channels with adjacent clearcuts. While we also observed the greatest post-harvest increase of wood cover where treated sites lacked a riparian buffer, we also observed an increase in wood cover in some buffered reaches due to windthrow.

We observed that the numbers of total functional SW and LW pieces responded similarly, with significant increases across all buffer treatments in the post-harvest period. The estimated increase in functional wood in buffer treatments was due primarily to an increase in the number of pieces contributing to hydraulic roughness, especially in stream reaches obstructed due to high levels of new wood recruitment associated with harvest. Approximately 10% of SW pieces contributed to step formation. Other research has also shown that SW provides a functional role in small streams, including step formation (Chesney 2000). In stream reaches obstructed with logging debris and windthrow, we observed a decrease in the number of SW pieces contributing to step formation and bank stability. Conversely, we found an increase in the numbers of SW pieces contributing to hydraulic roughness and spanning the bankfull channel in these reaches.

Our results differed somewhat from those of Gomi and colleagues (2001), who found no significant difference in their retrospective comparison of SW piece counts among clearcut (3-year-old), young conifer (37-year-old) and old-growth stands. This could be due to many factors, including differences in site or stand characteristics, timber harvest methods, or sampling methods. For example, our criterion for SW diameter included smaller pieces (minimum 2 cm vs. 3 cm diameter in the Gomi study) and we excavated dense wood piles in stream reaches highly obstructed by logging debris and windthrow to obtain a more accurate estimate of abundance than could be obtained by visual estimation.

We did not observe a shift in the overall distribution of LW among our diameter class categories in response to buffer treatment. However, we did observe an increase in the number of LW pieces in obstructed stream reaches. While this did not affect our overall site-wide results, it seems reasonable that, as the proportion of stream channel length obstructed by logging debris increases, the relative influence of these obstructed reaches on the overall LW loading may also increase.

6-6.2. POTENTIAL LONG-TERM EFFECTS

Simulation modeling studies have been used to examine the long-term effects of differences in wood input from streamside stands on wood loading patterns. These studies indicate that chronic mortality and episodic disturbances (including timber harvest) affect the amount and timing of wood recruitment to the channel. The magnitude and timing of these inputs, together with the processes of decay and wood transport, are responsible for the abundance and characteristics of in-channel wood (Gregory *et al.* 2003). In the absence of episodic disturbance, wood loading tends to increase over time as the streamside stand matures (Hedman *et al.* 1996; Meleason *et al.* 2003). In contrast, following major episodic disturbance, wood recruitment and loading follow a U-shaped pattern over time. An initial surge in wood input occurs during the first few decades as trees killed by the disturbance fall and recruit to the channel. Wood input then decreases as the remaining trees killed in the disturbance are depleted. With little recruitment potential from the newly established forest, wood load decreases as in-channel wood decays or is transported out of the reach. As a new forest establishes, grows and matures, chronic wood input increases loading over time (Spies *et al.* 1988; Bragg 2000).

The magnitude and timing of wood inputs vary depending on the severity and frequency of disturbance (Benda and Sias 2003). Severe disturbances result in greater variability, with a higher initial peak due to input of many trees over a short period and a lower trough during the period when there is little recruitment from the young forest. The pattern is modulated if the disturbance is less severe. Less initial input results in a smaller spike in wood load, and the decrease over time is offset by continued recruitment from the remnant trees that survive the initial disturbance (Spies *et al.* 1988; Bragg 2000). Although there are greater oscillations in wood load, cumulative wood input and loading over long periods is greatest in stands with episodic disturbances (Bragg 2000) having a greater recurrence interval (Benda *et al.* 2003). Harvest of streamside stands results in the greatest reduction in wood input and loading, since removal of streamside trees eliminates potential wood recruitment. Although there is input of logging debris during harvest, input over time is much less than for unharvested stands (either disturbed or undisturbed), resulting in low wood loading levels that vary depending on the stand characteristics; frequency, intensity and method of harvest; and the presence and width of a riparian buffer (Beechie *et al.* 2000; Meleason *et al.* 2003).

We used the concepts from the simulation models developed by Spies and colleagues (1988) and Bragg (2000) to predict future wood loading in our study sites over time based on differences in disturbance and buffer treatments. Overall, the models indicate that, over time, we might expect the highest cumulative wood inputs in unharvested stands with episodic disturbance, lower inputs in unharvested stands with chronic mortality, and the lowest inputs when trees are removed with periodic harvest (Bragg 2000). For FPB and 100% treatments with minimal tree mortality in the riparian buffers, there was little LW recruitment during the first two post-harvest years. In the absence of episodic disturbance, chronic mortality should provide limited wood recruitment over the next few decades. We expect that wood load would remain relatively stable for several decades as the limited input offsets depletion by decay and transport, followed by a gradual increase over time as the trees in the riparian buffer grow and stands pass through the stem-exclusion phase of development (Spies *et al.* 1988; Bragg 2000). In FPB and 100% treatments with greater tree mortality due to windthrow, tree fall during the first two post-harvest years provided a pulse of wood to adjacent streams, which could prove beneficial for streams with low wood loading due to past harvest or debris flow disturbance. In these cases, simulation models indicate that the pattern of wood input will depend on the magnitude of the disturbance. Where the disturbance fells most trees simultaneously, we expect a pronounced oscillation in wood load over time, with high input during or immediately after the disturbance and little additional wood recruitment for many years as the new stand becomes established. Where the disturbance is less severe, felling some trees but leaving many standing, the initial wood loading is not as great and additional recruitment from the standing trees that survive the disturbance will continue over time, dampening the decrease in wood load while the new stand becomes established. For RMZs with clearcut harvest to the stream, most trees will be harvested repeatedly over time, limiting wood input from these streamside stands. There will be periodic increases in wood loading due to inputs of unmerchantable pieces, tops and branches during harvest, followed by a decrease in loading as the logging debris decays or is transported downstream. Overall, we expect that RMZ harvest will result in lower levels of wood loading over time compared to sites with a riparian buffer.

While simulation models are useful for comparing different scenarios, the actual amount and characteristics of wood will depend on a host of factors. These factors include characteristics of the pre-existing wood load, the structure and composition of the streamside stand, input during harvest, the level and type of post-harvest stand disturbance (chronic or episodic), and the potential for wood recruitment and transport from other sources (e.g., mass wasting, debris flows, snow avalanches). Though we were able to detect differences in large and small wood between treatments, these differences would have likely been greater without the pre-harvest (December 2007) windstorm, which added substantial wood to impacted streams regardless of treatment.

6-6.3. FACTORS POTENTIALLY AFFECTING RESULTS

Several factors influenced the rates of wood recruitment and the quantity and characteristics of wood loading reported in this study. Our study took place in second-growth forests managed for timber production, which has undoubtedly influenced the composition and structure of the riparian stands and amount and characteristics of in-channel wood. Stand age variation may have affected LW recruitment volumes. For example, older stands in the Olympic and South Cascade references may have contributed to higher LW recruitment volumes, since the larger trees would produce wood pieces with larger diameters.

The December 2007 windstorm with hurricane-force winds affected only a subset of study sites (see Chapter 4 – *Unanticipated Disturbance Events* and Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers*). 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, and likely effecting future wood recruitment and loading. We were able to distinguish between LW recruitment associated with the December 2007 windstorm and post-harvest recruitment. However, pre-harvest sampling for wood loading occurred prior to the 2007 storm and post-harvest sampling occurred after that time. Consequently, post-harvest wood loading values include wood recruited during the pre-harvest windthrow event and during and immediately post-harvest. Fortunately, the windthrow event reflects the natural variability that occurs throughout western Washington. Furthermore, we detected treatment effects in spite of 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.

While timber harvest was conducted in a manner consistent with current Forest Practices rules, input of logging debris varied between and within harvest units depending on site and stand characteristics, equipment and logistics, among other factors. One should consider this variability when comparing our results with other studies. At one site, the operator removed wood from the channel post-harvest. A fire also burned across this study site in October 2009, between our first and second post-harvest years of sampling (see Chapter 4 – *Unanticipated Disturbance Events*). While fire can affect wood loading in streams (Benda *et al.* 2003; Chen *et al.* 2005), we had no evidence that this event decreased the number of wood pieces in the stream.

6-7. CONCLUSIONS

The results of this study have important implications for wood recruitment and loading in headwater streams in second-growth forests. Most old-growth forests on state and private lands in western Washington were harvested early in the 20th century and harvest of second-growth stands is now underway. Previous studies of western Washington streams have documented rapid loss of in-channel LW volume following the initial harvest (Bilby and Ward 1991), and continued depletion over the first three to four decades following harvest. Although the number of pieces remained similar over time, the volume of smaller diameter wood recruited from the young second-growth forests was not great enough to offset the continued depletion of larger wood from the original old-growth forest (McHenry *et al.* 1998).

The management practices applied to young riparian forests will have a major influence on wood recruitment and loading in streams in second-growth forests such as ours. Much of the wood recruited from the FPB and 100% treatment buffers due to windthrow consisted either of entire uprooted trees or large stems with branches attached. This wood input is potentially beneficial, since the volume of wood in many streams has decreased over time following the initial harvest of old-growth forests (McHenry *et al.* 1998). Since the buffers required along Type N streams under Forest Practices rules will not be harvested in the future, simulation studies indicate that wood recruitment will increase over time, resulting in an increase in the volume of wood loading and the size of wood pieces. However, recovery to pre-harvest levels may take as long as 250 years or more (Murphy and Koski 1989). Many of the fallen trees in riparian buffers came to rest spanning or suspended over the channel. Eventually they will break or decay and fall into the channel, but there is uncertainty concerning the timeframe for this process.

Inputs of logging debris into stream reaches lacking riparian buffers often occurred in accumulations that filled or buried our small study channels. This wood can provide functions in headwater streams, including shading, habitat formation, energy dissipation and sediment storage; however, there is uncertainty about the length of time smaller pieces will persist in the channel. The results of the simulation studies and the empirical observations of McHenry and colleagues (1998) indicate that there will be continued depletion of LW over time where there is repeated harvest of streamside trees.

The FP treatment requires buffering of a minimum of 50% of the Type Np stream length, with the remainder of the streamside stand available for harvest. Consequently, the FP prescription produces two dramatically different riparian stand conditions, with buffered RMZs concentrated in the downstream portions of the Np stream network and clearcut RMZs further upstream. Due to differences in the wood input regimes between the buffered and clearcut reaches of stream, we anticipate extensive reach-scale variation in wood loading to develop in headwater basins managed under the Forest Practices rules. Based on simulation modeling, we expect lower wood loading in the clearcut portions of the Np network, with wood load oscillating in response to periodic inputs of logging debris, as well as mass wasting or debris flow disturbance. We expect the wood load will be greater in the buffered reaches, with the pattern determined by the characteristics of the stand and the magnitude and timing of disturbance. In the absence of episodic disturbances, wood loading in the buffered reaches should gradually increase over time. In buffered reaches experiencing episodic disturbance, we expect wood loading would increase in the short-term and then decline as the streamside forest regenerates.

We observed substantial regional variability in tree fall rates. The timing and magnitude of wood inputs was inconsistent, resulting in considerable variability between and within sites, especially in the FP treatment. The implication of increased variability is unclear; however, it should result in a greater diversity of channel conditions, which could affect habitat abundance and suitability for a variety of aquatic species.

Our study informs many of the information gaps concerning headwater streams and wood loading and function, as well as the impacts of timber management on headwater streams. Results from this study will allow managers to consider the multiple processes responsible for delivery of wood to streams, both natural and human-derived. Results from the first two years post-harvest document the short-term response to riparian buffer treatments. However, the response of wood input and loading will continues for decades, so our results only provide an initial snapshot of the disturbance and recovery patterns. Future monitoring will be necessary to address the uncertainties associated with the effect of buffer treatments over a longer timeframe.

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APPENDIX 6-A. DATA TABLES

Appendix Table 6-A-1. Mean total tree fall rates in trees per acre per year (TPA/yr) and basal area per acre per year (BAPA/yr) for RMZ and PIP plots compiled by site for the pre-harvest period.

Block	Treatment	RMZs		PIPs	
		Trees/ac/yr (trees/ha/yr)	Basal Area ft^2/ac/yr (m^2/ha/yr)	Trees/acre/yr (trees/ha/yr)	Basal Area ft^2/ac/yr (m^2/ha/yr)
OLYM	REF	4.2 (10.4)	1.9 (0.4)	0.0 (0.0)	0.0 (0.0)
	100%	10.5 (25.9)	5.0 (1.2)	4.4 (10.9)	4.3 (1.0)
	FP	13.3 (32.9)	7.5 (1.7)	2.9 (7.2)	2.5 (0.6)
	0%	7.0 (17.3)	4.0 (0.9)	-	-
WIL1	REF	97.1 (239.9)	97.8 (22.5)	79.6 (196.7)	90.9 (20.9)
	100%	70.1(173.2)	61.9 (14.2)	39.8 (98.3)	35.0 (8.1)
	FP	16.2 (40.0)	19.4 (4.5)	145.9 (360.5)	151.9 (34.9)
	0%	20.7 (51.1)	20.2 (4.6)	14.1 (34.8)	13.9 (3.2)
WIL2	REF1	33.1 (81.8)	24.9 (5.7)	26.5 (65.5)	12.4 (2.9)
	REF2	40.8 (100.8)	38.4 (8.8)	53.1 (131.2)	59.0 (13.6)
	100%	27.6 (68.2)	20.5 (4.7)	42.0 (103.8)	39.1 (9.0)
	0%	33.5 (82.8)	17.6 (4.0)	33.9 (83.8)	18.6 (4.3)
WIL3	REF	4.9 (12.1)	4.7 (1.1)	-	-
	100%	5.2 (12.8)	3.1 (0.7)	-	-
CASC	REF	3.7 (9.1)	5.6 (1.3)	1.5 (3.7)	3.2 (0.7)
	FP	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)
	0%	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)
Mean		22.8 (56.3)	19.6 (4.5)	31.7 (78.3)	30.8 (7.1)

Appendix Table 6-A-2. LW recruitment rates during the pre-harvest period (2007–2008) by piece count and volume for RMZs and PIPs.

Block	Treatment	RMZ		PIP	
		Count pieces/ac/yr (pieces /ha/yr)	Volume ft^3/ac/yr (m^3/ha/yr)	Count pieces/ac/yr (pieces /ha/yr)	Volume ft^3/ac/yr (m^3/ha/yr)
OLYM	REF	2.8 (6.9)	5.6 (0.4)	0.0 (0.0)	0.0 (0.0)
	100%	12.6 (31.1)	11.1 (0.8)	13.3 (32.9)	19.5 (1.4)
	FP	10.8 (26.7)	28.9 (2.0)	2.9 (7.2)	12.5 (0.9)
	0%	2.6 (6.4)	3.4 (0.2)	-	-
WIL1	REF	49.2 (121.6)	256.6 (18.0)	13.3 (32.9)	69.9 (4.9)
	100%	37.0 (91.4)	190.3 (13.3)	13.3 (32.9)	116.3 (8.1)
	FP	1.2 (3.0)	3.5 (0.2)	53.1 (131.2)	232.1 (16.2)
	0%	10.4 (25.7)	57.8 (4.0)	4.4 (10.9)	8.2 (0.6)
WIL2	REF1	23.4 57.8)	86.8 (6.1)	11.1 (27.4)	33.3 (2.3)
	REF2	24.4 (60.3)	94.7 (6.6)	17.7 (43.7)	25.9 (1.8)
	100%	19.8 (48.9)	105.7 (7.4)	26.5 (65.5)	48.2 (3.4)
	0%	12.6 (31.1)	107.1 (7.5)	16.2 (40.0)	95.8 (6.7)
WIL3	REF	2.8 (6.9)	11.7 (0.8)	-	-
	100%	3.8 (9.4)	7.7 (0.5)	-	-
CASC	REF	1.6 (4.0)	4.1 (0.3)	2.9 (7.2)	1.7 (0.1)
	FP	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)
	0%	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)
Mean		12.7 (31.4)	57.3 (4.0)	12.5 (30.9)	47.4 (3.3)

Appendix Table 6-A-3. Mean total tree fall rates in trees/acre/yr (TPA/yr) and basal area/acre/yr (BAPA/yr) for RMZ and PIP plots compiled by site for the post-harvest period.[1]

Block	Treatment	RMZ		PIP	
		Trees/ac/yr (trees/ha/yr)	Basal Area ft^2/ac/yr (m^2/ha/yr)	Trees/ac/yr (trees/ha/yr)	Basal Area ft^2/ac/yr (m^2/ha/yr)
OLYM	REF	15.1 (37.3)	6.8 (1.6)	4.5 (11.1)	2.1 (0.5)
	100%	3.1 (7.7)	2.5 (0.6)	3.4 (8.4)	2.1 (0.5)
	FP	23.9 (59.1)	19.8 (4.6)	41.4 (102.3)	37.5 (8.6)
WIL1	REF	10.5 (25.9)	12.9 (3.0)	3.4 (8.4)	5.1 (1.2)
	100%	22.8 (56.3)	19.5 (4.5)	20.6 (50.9)	17.1 (3.9)
	FP	61.7 (152.5)	75.1 (17.3)	21.4 (52.9)	14.9 (3.4)
WIL2	REF1	2.7 (6.7)	2.0 (0.5)	4.5 (11.1)	4.4 (1.0)
	REF2	2.8 (6.9)	3.7 (0.9)	2.2 (5.4)	1.1 (0.3)
	100%	30.9 (76.4)	26.2 (6.0)	37.7 (93.2)	39.4 (9.1)
WIL3	REF	2.5 (6.2)	1.6 (0.4)	-	-
	100%	5.2 (12.8)	2.6 (0.6)	-	-
CASC	REF	2.4 (5.9)	6.8 (1.6)	0.7 (1.7)	0.3 (0.1)
	FP	1.7 (4.2)	0.7 (0.2)	1.5 (3.7)	0.7 (0.2)

[1] We did not calculate values for the 0% sites and the clearcut portions of the FP sites because there were few or no trees remaining.

Appendix Table 6-A-4. Piece count and volume of LW recruitment during the post-harvest period (2008–2010) by for RMZ and PIP reaches.

Block	Treatment	RMZ		PIP	
		Count pieces/ac/yr (pieces /ha/yr)	Volume ft^3/ac/yr (m^3/ha/yr)	Count pieces/ac/yr (pieces/ha/yr)	Volume ft^3/ac/yr (m^3/ha/yr)
OLYM	REF	10.0 (24.7)	31.8 (2.2)	2.2 (5.4)	4.3 (0.3)
	100%	2.0 (4.9)	5.1 (0.4)	1.1 (2.7)	0.9 (0.1)
	FP	13.4 (33.1)	38.2 (2.7)	12.5 (30.9)	56.3 (3.9)
	0%	0.0 (0.0)	0.0 (0.0)	-	-
WIL1	REF	5.3 (13.1)	24.3 (1.7)	0.0 (0.0)	0.0 (0.0)
	100%	10.9 (26.9)	46.3 (3.2)	9.9 (24.5)	36.7 (2.6)
	FP	20.0 (49.4)	109.2 (7.6)	6.6 (16.3)	19.3 (1.4)
	0%	0.2 (0.5)	0.2 (0.01)	0.0 (0.0)	0.0 (0.0)
WIL2	REF1	1.7 (4.2)	11.8 (0.8)	0.0 (0.0)	0.0 (0.0)
	REF2	2.2 (5.4)	9.9 (0.7)	1.1 (2.7)	1.5 (0.1)
	100%	15.9 (39.3)	196.2 (13.7)	15.5 (38.3)	74.4 (5.2)
	0%	0.2 (0.5)	0.4 (0.03)	0.0 (0.0)	0.0 (0.0)
WIL3	REF	0.8 (2.0)	1.7 (0.1)	-	-
	100%	2.4 (5.9)	4.9 (0.3)	-	-
CASC	REF	4.7 (11.6)	32.7 (2.3)	0.0 (0.0)	0.0 (0.0)
	FP	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)
	0%	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)	0.0 (0.0)

Appendix Table 6-A-5. Estimated number of total (tot) and functional (fnctl) pieces per stream meter for SW and LW by block, treatment and year. Period indicates which years were pre-harvest (2006 and 2007) and post-harvest (2009 and 2010).

Block	Treatment	Period	Year	SW		LW	
				tot/m	fnctl /m	tot/m	fnctl /m
OLYM	REF	Pre-	2007	3.03	1.90	1.39	1.29
		Post-	2009	7.78	3.15	2.27	1.91
			2010	5.23	3.02	1.9	1.63
	100%	Pre-	2006	6.48	3.20	1.67	1.49
			2007	5.36	3.52	1.51	1.44
		Post-	2009	11.38	4.88	3.03	2.49
			2010	7.77	4.42	2.72	2.25
	FP	Pre-	2006	3.93	1.65	0.88	0.79
			2007	3.55	2.23	0.7	0.63
		Post-	2009	5.24	2.83	1.26	1.13
			2010	4.65	2.33	1.18	0.93
	0%	Pre-	2006	6.43	2.66	1.71	1.43
			2007	4.91	2.94	1.67	1.59
		Post-	2009	19.78	6.12	1.78	1.33
			2010	24.70	8.14	2.69	1.70
WIL1	REF	Pre-	2006	4.52	1.96	1.28	1.05
			2007	6.02	3.29	1.26	1.11
		Post-	2009	4.75	1.91	1.59	1.15
			2010	4.34	2.40	1.64	1.10
	100%	Pre-	2006	5.89	2.36	1.65	1.47
			2007	5.17	3.12	1.2	1.10
		Post-	2009	7.48	3.27	1.99	1.43
			2010	7.67	3.95	2.14	1.57
	FP	Pre-	2006	5.30	2.10	1.26	1.11
			2007	3.68	2.41	1.17	1.03
		Post-	2009	10.16	4.73	2.21	1.72
			2010	9.65	3.87	2.11	1.44
	0%	Pre-	2006	4.52	1.89	1.00	0.80
			2007	7.54	4.08	1.43	1.29
		Post-	2009	11.03	4.59	1.54	1.26
			2010	12.24	5.42	1.71	1.47

Appendix Table 6-A-6. (continued)

Block	Treatment	Period	Year	SW		LW	
				tot/m	fnctl /m	tot/m	fnctl /m
WIL2	REF1	Pre-	2006	6.50	3.77	2.11	1.94
			2007	8.22	5.02	2.23	2.10
		Post-	2010	6.13	3.50	2.06	1.75
	REF2	Pre-	2006	5.44	2.88	1.16	1.03
			2007	7.90	5.07	1.30	1.25
		Post-	2009	5.24	2.78	1.16	0.97
			2010	3.48	1.90	1.17	0.97
	100%	Pre-	2006	5.22	1.82	1.50	1.32
			2007	6.26	3.71	1.67	1.58
		Post-	2009	11.64	5.38	4.04	3.42
			2010	19.14	10.45	4.62	3.57
	0%	Pre-	2006	6.98	3.55	1.50	1.37
			2007	7.70	3.75	1.31	1.16
		Post-	2009	9.97	4.05	1.76	1.38
			2010	42.06	16.62	4.72	2.90
WIL3	REF	Pre-	2006	3.10	1.54	0.81	0.71
			2007	3.00	2.04	0.83	0.80
		Post-	2009	4.43	2.26	1.02	0.91
			2010	3.77	2.05	1.07	0.91
	100%	Pre-	2006	6.50	3.36	0.94	0.82
			2007	5.23	3.08	0.80	0.74
		Post-	2009	6.38	3.53	0.88	0.76
			2010	5.82	3.30	1.07	0.92
CASC	REF	Pre-	2006	4.17	1.94	1.09	0.92
			2007	2.81	1.49	1.01	0.89
		Post-	2009	6.76	2.86	1.17	0.94
			2010	3.82	2.00	0.99	0.77
	FP	Pre-	2006	3.35	1.30	0.52	0.48
			2007	1.73	0.98	0.49	0.44
		Post-	2009	3.49	1.50	0.65	0.52
			2010	4.72	2.29	0.75	0.54
	0%	Pre-	2006	2.85	1.18	0.58	0.54
			2007	2.80	1.39	0.67	0.59
		Post-	2009	6.13	1.59	1.16	0.91
			2010	7.87	3.46	1.60	1.23

Appendix Table 6-A-7. The percentage of the bankfull channel length covered by new wood (i.e., recruited in the past year) within 10-m stream reaches, averaged by site and year for the two post-harvest years immediately following harvest (2009 and 2010).

Block	Treatment	2009 (%)	2010 (%)
OLYM	REF	10	2
	100%	4	8
	FP	33	6
	0%	34	0
WIL1	REF	8	11
	100%	28	8
	FP	38	10
	0%	43	0
WIL2	REF1	3	3
	REF2	1	3
	100%	32	15
	0%	35	0
WIL3	REF	5	0
	100%	6	1
CASC	REF	1	0
	FP	24	0
	0%	28	0

CHAPTER 7 - STREAM TEMPERATURE AND COVER

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7-1. 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 at least two years post-harvest.

Our results indicated that riparian shade decreased post-harvest in all treatments relative to unharvested reference sites. By the second year post-harvest canopy and topographic density decreased 6, 23 and 48%; effective shade decreased by 10, 36, and 72%; canopy closure at 1 m height decreased by 5, 27, and 78%; and canopy closure at the water surface decreased by 2, 15, and 45% in the 100%, FP, and 0% treatments, respectively. Water temperature increased post-harvest, with the greatest change occurring during the July–August period. Maximum daily temperature increased at most locations throughout the streams and was often elevated from April through October. Minimum daily water temperature increased at all sites over some part of the year. Diel range in water temperature also increased in a pattern similar to the maximum daily temperature change but to a lesser degree.

The mean buffer treatment effect on the July–August seven-day average maximum daily temperature immediately below the harvest unit was 1.2°C in the 100%, 1.2°C in the FP, and 3.1°C in the 0% treatments over the first two years post-harvest. Seven-day average minimum temperature increased by 0.8, 0.4, and 0.9°C in the 100%, FP, and 0% treatments, respectively. The seven-day average diel range increased 0.8, 1.0, and 3.0°C in the 100%, FP, and 0% treatments, respectively.

Below the harvest unit, maximum daily temperature showed signs of recovery toward pre-harvest conditions with the mean monthly maximum daily temperature decreasing by 0.3°C to 3.2°C after flowing through approximately 100 m of unharvested forest. However, stream temperature was still elevated above pre-harvest levels at five of the six sites where recovery could be assessed.

7-2. 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.* 2005a). Radiative exchanges include direct and diffuse solar radiation inputs and long-wave radiation exchange with the surrounding atmosphere, vegetation, and terrain. In forested environments, shade provided by riparian vegetation attenuates incoming solar radiation and is the single most important variable influencing summer stream temperature (Brown 1969; Johnson and Jones 2000; Danehy *et al.* 2005; Groom *et al.* 2011). There are several pathways for heat exchange in the stream environment: latent heat exchange is associated with the evaporation or condensation of water; sensible heat exchange between the water and overlying air depends upon the temperature difference between the two; and 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.* 2005b) while estimates of bed heat exchange are 10% of net radiation for a step-pool stream (Moore *et al.* 2005b) 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 temperatures (Webb and Zhang 1999). Hyporheic exchange of water between the stream and the underlying substrate typically moderates temperature extremes and can be an important factor in local and reach-scale temperatures in headwater streams (Johnson 2004; Moore *et al.* 2005b).

Early studies of the direct effects of forest harvest on stream temperature demonstrated dramatic decreases in shade and increased summer stream temperature after harvest (Brown and Krygier 1970; Harris 1977; Feller 1981; Holtby and Newcombe 1982; 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). However, Moore and colleagues (2005a) reported more modest temperature increases of 2.5°C to 5.0°C from studies of riparian buffers following contemporary forest practices. They suggest that much of the variability in results is 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 the sensitivity of a stream to increased temperatures is related to stream width and depth, flow velocity and volume, and site aspect and elevation (Beschta *et al.* 1987; Isaak and Hubert 2001; Poole and Berman 2001; Moore *et al*. 2005b).

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. An explicit goal of the law is to maintain cool water temperatures within the non-fish-bearing stream as well as downstream in fish-bearing waters and there is an implicit assumption that increases in stream temperature in the unbuffered reaches will not be transmitted downstream after flowing through a riparian buffer or intact forest. Burton and Likens (1973) noted rapid increases and decreases in stream temperature as it flowed through alternating unbuffered and buffered stream reaches, and similar heating and cooling has been observed in other studies in forested streams (Caldwell *et al.* 1991; Storey and Cowley 1997; Keith *et al.* 1998; Storey *et al.* 2003; Wilkerson *et al.* 2006; Gravelle and Link 2007), although energy budget models suggest this is due to inputs of cooler water rather than more shading (Brown *et al.* 1971; Storey *et al.* 2003; Garner *et al.* 2014).

In terms of stream temperature effects on fish species, most studies to date have focused on the effects of changes in one metric, the summer maximum daily stream temperature. Modeling by Leach and colleagues (2012) suggests that while higher summer mean temperatures can depress growth rates of cutthroat trout, this may be offset in some cases by higher growth rates in the fall and spring. In addition, changes in diel temperature variation can alter salmon egg development (Steel *et al.* 2012). This variability in fish responses illustrates the need for the evaluation of multiple daily temperature statistics on a year-round basis.

In this chapter, we report the effects of clearcut forest harvest following the current Washington State Forest Practices rules for non-fish-bearing perennial streams and two alternative buffer treatments on riparian shade and the year-round effect on the daily maximum, minimum, and diel range in stream temperature.

7-3. OBJECTIVES

The objective of this study was to estimate the effects of clearcut timber harvest with three different riparian buffer treatments on riparian cover and water temperature. Specific questions 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, daily minimum, and diel range in stream temperature following harvest within the non-fish-bearing stream?

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

7-4. METHODS

7-4.1. *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 include 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 were located in 30–80 year old managed Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*)-dominated second-growth forests on private, state, and federal forestlands. Sites were located in areas dominated by competent lithology types with average Type Np channel gradients ranging from 14% to 34% with catchment areas ranging from 12 to 49 ha (**Table 7-1**). We present site-selection criteria in Chapter 2 – *Study Design*.

Table 7-1. Study site characteristics. BFW = bankfull width; Elev = elevation.

Block	Treatment	Elev (m)	Lithology	Stream Gradient (%)	Type Np Stream Length (m)	BFW (m)	Area (ha)
OLYM	REF	163	Basalt flows and flow breccias	18	2,737	2.6	44
	100%	72	Tectonic breccia	27	1,949	2.1	28
	FP	277	Basalt flows and flow breccias	25	1,070	1.1	17
	0%	233	Basalt flows and flow breccias	31	637	1.5	13
WIL1	REF	200	Basalt flows and flow breccias	19	589	1.4	12
	100%	198	Basalt flows and flow breccias	18	1,029	2.1	31
	FP	197	Basalt flows and flow breccias	19	325	1.5	15
	0%	87	Terraced deposits	16	1,525	1.8	28
WIL2	REF1	183	Basalt flows and flow breccias	34	653	1.9	19
	REF2	228	Basalt flows and flow breccias	18	816	1.3	16
	100%	22	Basalt flows and flow breccias	21	1,257	1.9	26
	0%	159	Basalt flows	21	933	2.2	17
WIL3	REF	241	Basalt flows	14	2,513	1.8	37
	100%	351	Basalt flows	19	1,359	2.2	23
CASC	REF	601	Tuffs and tuff breccias	21	1,080	2.0	49
	FP	450	Andesite flows	16	822	1.5	26
	0%	438	Andesite flows	29	420	1.6	14

7-4.2. *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 (**Figure 7-1**).
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 safety, 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 (ELZ).

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%, 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 (**Appendix Figure 7-A-2**).
- In the WIL2-100%, 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 (**Appendix Figure 7-A-4**).
- In the WIL3-100%, the F/N break (T1) was 10 m inside the fish-bearing stream buffer at its downstream confluence (**Appendix Figure 7-A-5**).

Similarly, we intended the 0% treatment to have no buffer; however:

- In the CASC-0%, the F/N break (T1) was located 85 m inside the fish-bearing buffer of the adjacent CASC-FP (**Appendix Figure 7-A-7**).

- In the WIL2-0%, no harvest occurred along the 50 m of stream immediately above the F/N break (T1) (**Appendix Figure 7-A-10**).

7-4.3. *DATA COLLECTION*

7-4.3.1. *Riparian Cover*

We calculated four metrics of riparian cover using data collected with hemispheric canopy photos (two metrics) and a spherical densiometer (two metrics). We used hemispheric canopy photos taken at 1 m above the water surface to estimate:

- Canopy and topographic density (CTD)
- Effective shade

We used a spherical densiometer (Lemmon 1956) to measure:

- Canopy closure at 1 meter height (CC-1m)
- Canopy closure at the water surface (CC-0m)

7-4.3.1.a. 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. We took photographs at 10 stations per site, except in the OLYM-0%, where only nine stations were established. Stations were equally spaced longitudinally along the main 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 would be 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 instances, we took photos at, or within 2 m of, the original station.

We took photographs between 6 June and 9 August annually in all study sites from 2007 through 2011, with the following exceptions:

- In 2009, photographs in the OLYM-0% were taken on 30 September 2009, immediately post-harvest but before deciduous leaf fall.
- In 2009, we did not take photographs in what would have been the WIL2-FP because harvest was delayed. We resumed taking photographs in 2010 after we decided to keep this site in the study as an unharvested reference (now WIL2-REF2).

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, dark blue sky, and brightly lit vegetation 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 determine CTD and effective shade, for which we used the mean value for the entire seventh solar month (June 22–July 21), 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).

From these values, we calculated two metrics:

- 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 Bilheimer 2008).

7-4.3.1.b. 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 (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. We made measurements at 1 m from 2007 through 2010, and those at the water surface from 2008 through 2010.

7-4.3.2. *Temperature*

We measured water temperature at 30-minute intervals using StowAway TidbiT thermistors (Onset Computer Corporation, Bourne, Massachusetts). At each location, we installed a narrow range (−5°C to 37°C) 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 might have been 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 7-1**). The intent was to measure water temperature at multiple locations along the main perennial channel from the F/N break to the uppermost point of perennial flow. We monitored comparable locations along the main channel in the other treatments. We installed TidbiTs in all perennial tributaries near the confluence with, but above the influence of, the main 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 and to provide redundancy in the event of missing data (e.g., in case of missing data at the F/N break, we could use the next location upstream). To monitor temperature recovery after flowing through intact forest, we also monitored a location downstream from the harvest unit in the six buffer treatment sites having at least 100 m of stream flowing through a fish-bearing stream reach with no perennial tributaries.

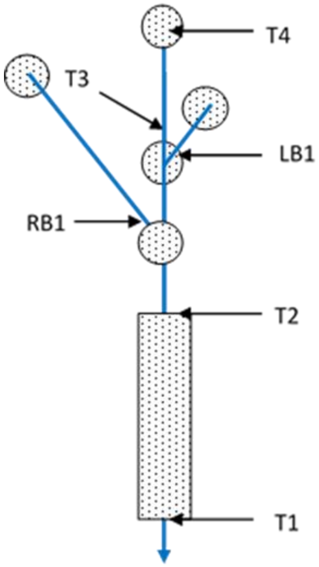

Figure 7-1. Conceptual layout of Forest Practices riparian buffers and temperature monitoring locations.

We monitored air temperature along the main stream channel at the same locations as water temperature using wide range (−20°C to 50°C) TidbiTs. We placed sensors 1 m above the ground, adjacent to the stream channel and protected from direct sunlight.

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

Prior to use, all TidbiTs passed a calibration check, where we compared them to a National Institute of Standards and Technology (NIST) thermometer in an ice bath and in ambient water baths (~18°C). We did not use TidbiTs that deviated by more than 0.2°C from the NIST thermometer for the narrow temperature range model or by more than 0.4°C for the wide temperature range model. In 2010, we replaced all TidbiTs manufactured before 2007 and ran them through the same calibration check. All of the wide-range TidbiTs passed the post-deployment calibration check and 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.

7-4.4. *ANALYSIS*

7-4.4.1. *Riparian Cover*

Analyses evaluated the generalized null hypothesis:

$$\Delta S_{\mathrm{REF}} = \Delta S_{100\%} = \Delta S_{\mathrm{FP}} = \Delta S_{0\%} \tag{7-1}$$

where: ΔS_{REF} is the change (post-harvest minus pre-harvest) in shade 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 used generalized linear mixed-effects models (GLMM) that incorporate both fixed and random effects for hypothesis testing. In matrix form, this model can be represented as:

$$Y = X\beta + Z\gamma + \epsilon \tag{7-2}$$

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.

Site was included as a random effect and the 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 according to the method of Kenward and Roger (1997), which is recommended for imbalanced designs. We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals and found no evidence of either.

When the interaction term was significant ($P < 0.05$), pairwise comparisons were used to estimate the effect size for each buffer treatment relative to the reference treatment in each post-harvest year and to estimate the difference in response among the three buffer treatments. We present estimates of the effect size and the associated 95% confidence intervals. We did not adjust the P-values for multiple comparisons.

7-4.4.2. *Temperature*

We address questions two and three in section 7-3 by calculating a daily temperature response (TR) for each monitoring location in the buffer treatment streams, then describing the magnitude and pattern of temperature change in terms of daily and mean monthly TR. We estimated the buffer treatment effect using the seven-day average TR. We also describe the longitudinal pattern in maximum daily stream temperature and temperature change along the mainstem channel in the context of post-harvest shade and surface stream flow.

7-4.4.2.a. Calculation of Daily Temperature Response (TR)

We calculated daily temperature response at each location in each treatment using an approach similar to that advocated by Watson and colleagues (2001) and modified by Gomi and colleagues (2006). This method involves two steps:

Step 1. We used a generalized least squares (GLS) regression of treatment vs. reference maximum daily temperature using the pre-harvest period data (**Equation 7-3**).

$$y_t = \beta_0 + \beta_1 x_t + \beta_2 x_t^2 + \sin(2\pi t/365) + \cos(2\pi t/365t) + \varepsilon_t \qquad (7\text{-}3)$$

where: y_t is the temperature in the treatment site on day t,
x_t is the 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:

$$\mathrm{E}_t = \phi_1 \varepsilon_{t-1} + \cdots + \phi_p \varepsilon_{t-p} + a_t \tag{7-4}$$

where: 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:

$$\varepsilon_t = \theta_1 a_{t-1} + \cdots + \theta_q a_{t-q} + a_t \tag{7-5}$$

where: 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:

$$\sum_{i=1}^{p} \phi_i \varepsilon_{t-1} + \sum_{j=1}^{q} \theta_j a_{t-j} + a_t \tag{7-6}$$

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 ($P < 0.05$), positive, lag one autocorrelation and the residuals were homoscedastic, relative to the predicted value and to time, and were approximately normally distributed.

If these conditions could not be 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. In all cases presented there was no significant positive lag one autocorrelation and few, if any, significant autocorrelation terms out to lag 20.

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

$$R^2{}_{LR} = 1 - \exp\left({}^{-2}\!/_{n} * \left(\log Lik(x) - \log Lik(0)\right)\right) \tag{7-7}$$

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 r^2, with $R^2 = 0$ indicating that the model explains no additional variation and $R^2 = 1$ 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 R^2 reflects the adequacy of the prediction model.

All water and air temperature GLS regressions exhibited significant lag one or greater autocorrelation in the residuals. Autoregressive lag terms in final models for maximum daily water temperature ranged from one to four. Twenty-six locations exhibited lag one, 27 with lag two, 14 with lag three and three with lag four AR terms. The MA term was needed in only seven locations. The regressions of the minimum and diel range of water temperatures and the maximum and minimum air temperature were similar, with the majority of the models requiring a lag one or lag two AR term. Pseudo R^2 values varied widely with low values often 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 analysis described in section 7-4.4.5 had pseudo R^2 values ranging from 0.398 in the CASC-0% to 0.930 in the WIL1-100%.

Step 2. Calculate the daily temperature response (TR) as the observed temperature minus the predicted temperature in the treatment stream for both post-harvest years.

$$TR = (y_t - \hat{y}_t) \tag{7-8}$$

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 in any of the regressions because of poor model fit. In the WIL1-REF, this may have been due to the windthrow caused by the December 2007 storm. The WIL3-REF had no surface flow in much of the channel during the summer and the maximum daily stream temperatures were low and temporally stable relative to all other sites. This is probably because hyporheic flows in summer attenuated the temperature fluctuations. We paired the WIL1, WIL2, and WIL3 buffer treatment sites with the WIL2-REF1 site.

7-4.4.2.b. Calculation of Mean Monthly Temperature Response (MMTR)

Although other methods of comparing the significance of the temperature response are available, for example, derived algebraic expressions (Som *et al.* 2012) or Monte Carlo simulation (Leach *et al.* 2012; Guenther *et al.* 2014), 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 two post-harvest years. We included an AR term in the model to account for the autocorrelation present and used the weights = VarIdent option to allow the variance to vary by month.

$$y_i = \beta_j + \varepsilon_{ij} \tag{7-9}$$

where: y_i is the daily temperature response,

β_j are the monthly mean responses for months j=1…12, and

ε_{ij} are the errors.

The errors are modeled using an AR1 correlation structure:

$$\varepsilon_{ij} = \phi_1\varepsilon_{t-1} + \cdots + \phi_p\varepsilon_{t-p} + a_{i,j} \tag{7-10}$$

where: ε_{t-1} is the error term for the day before,

ϕ_p is the lag p autocorrelation coefficient, and

a_{ij} is white noise centered at 0 and assumed to be independent of previous observations.

Each month is allowed to have a different error variance:

$$Var(\varepsilon_{ij}) = \sigma^2\delta^2{}_j \tag{7-11}$$

where: $\delta^2{}_j$ is the variance parameter with $\delta_1 = 1$, and

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

We plotted the daily TR values to illustrate the patterns over the entire year. The large number of comparisons (months) and the large number of locations increases the likelihood of Type II error so it is inappropriate to emphasize any single monthly estimate. Rather, we focused on patterns in the direction, magnitude, and seasonal variability of the monthly estimates.

We completed this analysis for the daily maximum, daily minimum, and diel range in water temperature and for daily maximum and minimum air temperature at the buffer treatment locations to illustrate the seasonal pattern of temperature response at the location that best represents the buffer treatment (**Appendix 7-C**). We also ran this analysis on the daily maximum water temperature at all locations at each buffer treatment site to illustrate the pattern along a longitudinal gradient from initiation of perennial flow through the harvest unit (**Appendix 7-D**).

Large or multiple gaps in the temporal sequence of the data can produce spurious results when correcting for serial autocorrelation. Likewise calculating MMTR for months with few observations can exaggerate the confidence intervals making comparison with other months difficult. We calculated MMTR only for years with at least 300 consecutive days of data to avoid spurious results and only for months with at least 20 days of data to reduce the effects of sample size on MMTR confidence intervals.

7-4.4.3. *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 will be detected in the treatment sites.

We used the same method described above to fit a regression model of daily maximum water temperature between locations in the WIL2-REF2, OLYM-REF, and CASC-REF and analogous locations in the WIL2-REF1 site in order to examine the stability of the calibration relationship over time in our reference sites. Daily TR values were calculated and plotted for the pre- and post-harvest period. We also plotted MMTR as a gauge of the sensitivity of the analysis of location-specific changes in monthly average maximum daily stream temperature. We used data collected prior to 1 September 2008 to calibrate the regression model and considered data collected on or after 1 September 2008 to be post-harvest. The WIL1-REF and WIL3-REF were not included for the reasons described above.

7-4.4.4. *Longitudinal Patterns in Water Temperature, Riparian Cover, and Flow*

We used graphical analyses to illustrate longitudinal patterns in pre- and post-harvest seven-day average daily maximum stream temperature (Max7D) and July MMTR relative to canopy cover at the water surface, effective shade, riparian buffers, and presence of surface water. We report only July MMTR for brevity, but the patterns are very similar using July, August, or September values.

7-4.4.5. *Statistical Analysis of Buffer Treatment Effect on Temperature*

Two criteria included in the water quality standards are based on the seven-day average maximum daily water temperature and the magnitude of human-caused changes in this metric. We used the maximum seven-day average TR (7DTR) during July–August to represent human-caused change in stream temperature. Although not referenced in the water quality standards, we also calculated 7DTR for minimum daily water temperature, diel range in water temperature, and both maximum and minimum daily air temperature, for comparison. We used these 7DTR values, one value per year for each pre-harvest and each post-harvest year, in the analyses described below.

The analyses evaluated the generalized null hypothesis:

$$\Delta 7DTR_{100\%} = \Delta 7DTR_{\mathrm{FP}} = \Delta 7DTR_{0\%} \tag{7-12}$$

where: $\Delta 7DTR_{100\%}$, $\Delta 7DTR_{\mathrm{FP}}$, and $\Delta 7DTR_{0\%}$ are the post-harvest change 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 can be represented as:

$$Y = X\beta + Z\gamma + \epsilon \tag{7-13}$$

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.

Site was included as a random effect and the 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 imbalanced designs. We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals and found no evidence of heteroscedasticity.

We used pairwise comparisons to estimate post-harvest changes in 7DTR for each buffer treatment and to estimate differences among the treatments. Estimates of the mean effect size, expressed in degrees Celsius, and the associated 95% confidence interval (CI) are presented in **Appendix 7-B**. The groups differed if the CI does not include zero. We chose 95% CI by convention and we elected not to adjust for multiple comparisons. The combination of small numbers 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 rather than any specific pairwise comparison.

Forest policy stakeholders were interested in the effects of the specific buffer treatments on water temperature where it discharged into fish-bearing waters. As noted above in section 7-4.2, some sites were not harvested as planned; i.e., the harvest unit did not always extend to the F/N junction. 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%, where we used the next location upstream because of missing temperature data at the F/N break (**Table 7-2**).

2) Buffer Treatment (BT) 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. In six of the 11 sites, this was the same location as used in the F/N break analysis. In the remaining five sites (OLYM-100%, WIL2-100%, WIL3-100%, WIL2-0%, and CASC-0%), we used other locations further upstream where the actual riparian buffer matched the buffer treatments (**Table 7-2**).

Table 7-2. Temperature monitoring location within each treatment basin that is: (a) nearest the Type F/N junction or, (b) best represents the intended buffer treatment. Notes describe the reason why the two monitoring locations differ.

Treatment	Block	F/N Analysis	Buffer Treatment	Notes
100%	OLYM	T1	RB1	Reach below road crossing is dry part of the year so F/N break was set at just above the road crossing. Riparian buffer width was much wider than 50 ft (15.2 m) because of unstable slope buffers, except on RB1 tributary.
	WIL1	T2	T2	Missing post-harvest data at T1.
	WIL2	T1	LB3	Unstable slope buffer precluded harvest over much of the lower basin. T3 and LB3 both represent the buffer treatment well, but summer data are missing at T3.
	WIL3	T1	T1	
FP	OLYM	T1	T1	62% of perennial channel is buffered.
	WIL1	T1	T1	73% of perennial channel is buffered.
	CASC	T1	T1	55% of perennial channel is buffered.
0%	OLYM	T1	T1	
	WIL1	T1	T1	
	WIL2	T1	T3	No harvest along lower 50 m of stream (T1). Missing data at T2 in Post 1.
	CASC	T1	T3	F/N break was within fish-bearing buffer of adjacent stream. T2 is missing pre-harvest summer data.

7-5. RESULTS

7-5.1. *RIPARIAN COVER*

Mean pre-harvest shade levels were high, exceeding 80% in both years at all sites (**Figure 7-2; Appendix Tables 7-B-1** to **7-B-4**). Three WIL1 sites (REF, 100%, FP) were substantially impacted by the December 2007 windstorm, with shade decreasing from approximately 90% to 80–84% in 2008 (**Table 7-3**).

Post-harvest shade loss, regardless of the metric, was greater in the 0% treatment than in either the 100% or the FP treatment (**Figure 7-3**). CTD decreased from an average of 95% pre-harvest to 86, 71, and 43% by Post 2 in the 100%, FP, and 0% buffer treatments, respectively. Effective shade decreased to 77, 52, and 14% by Post 2, in the 100%, FP, and 0% buffer treatments, respectively. Canopy closure-1m decreased to 85, 67, and 9% by Post 2 in the 100%, FP, and 0% buffer treatments, respectively. Canopy closure-0m changed less post-harvest, decreasing to 91, 85, and 52% in the 100%, FP, and 0% buffer treatments, respectively, due to overlying slash and windthrow, especially in the unbuffered stream reaches.

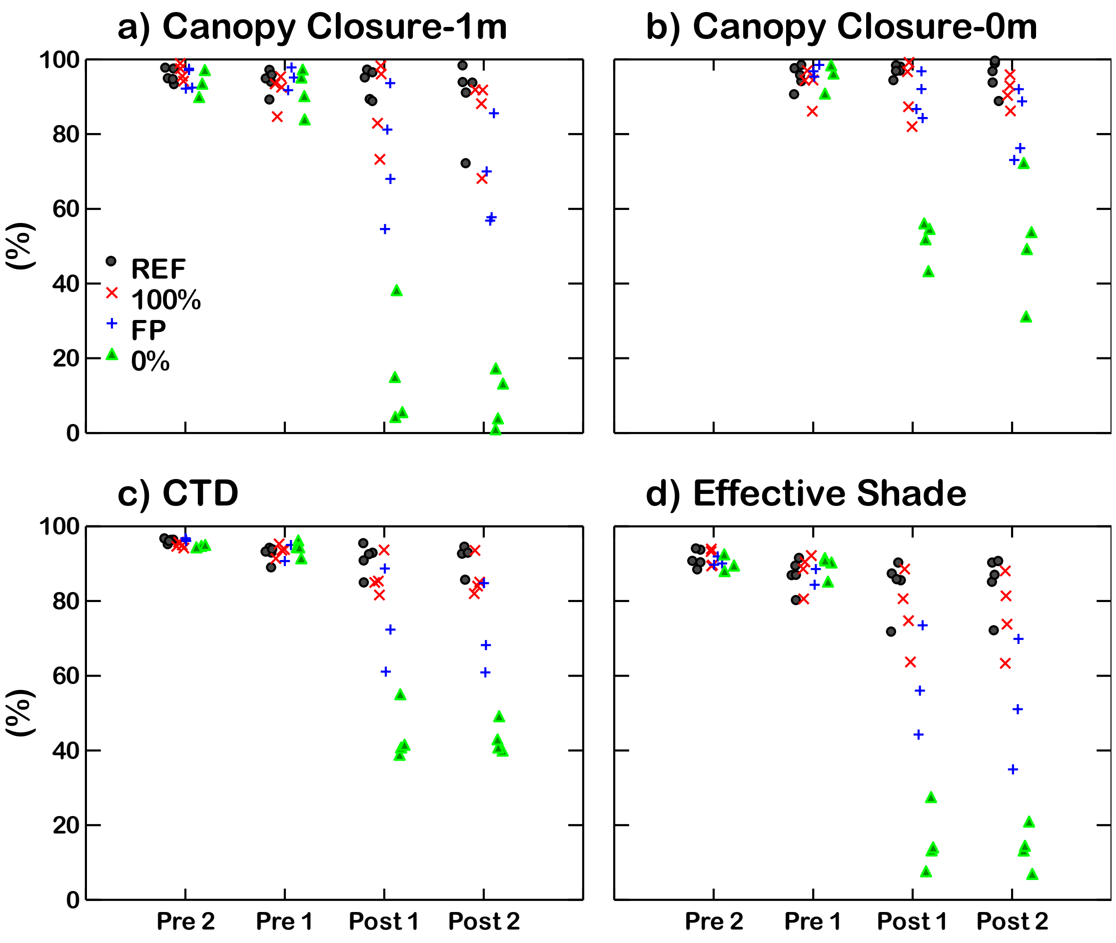

Figure 7-2. Mean site values for each stream cover metric. We did not calculate CTD (Canopy and topographic density) and effective shade for WIL1-0% in 2008. We did not measure canopy closure-0m until 2008 (Pre 1 year above).

Table 7-3. Mean values for the four shade metrics presented by treatment and year. Canopy closure-0m measurements began in 2008. Numbers in **bold type** indicate a significant (P <0.05) decrease in that metric for that post-harvest year based upon the post-hoc comparison. CTD-Canopy and Topographic Density.

Block	Treatment	Percent			
		2007-Pre 2	2008-Pre 1	2009-Post 1	2010-Post 2
CTD	REF	96	92	91	92
	100%	95	94	86	**86**
	FP	96	93	**74**	**71**
	0%	94	95	**44**	**43**
Effective Shade	REF	90	87	84	85
	100%	92	88	**77**	**77**
	FP	91	86	**58**	**52**
	0%	89	91	**16**	**14**
Canopy Closure-1m	REF	96	94	93	91
	100%	97	92	88	85
	FP	96	96	**72**	**67**
	0%	91	94	**16**	**9**
Canopy Closure-0m	REF		95	97	95
	100%		93	91	91
	FP		98	92	**85**
	0%		95	**51**	**52**

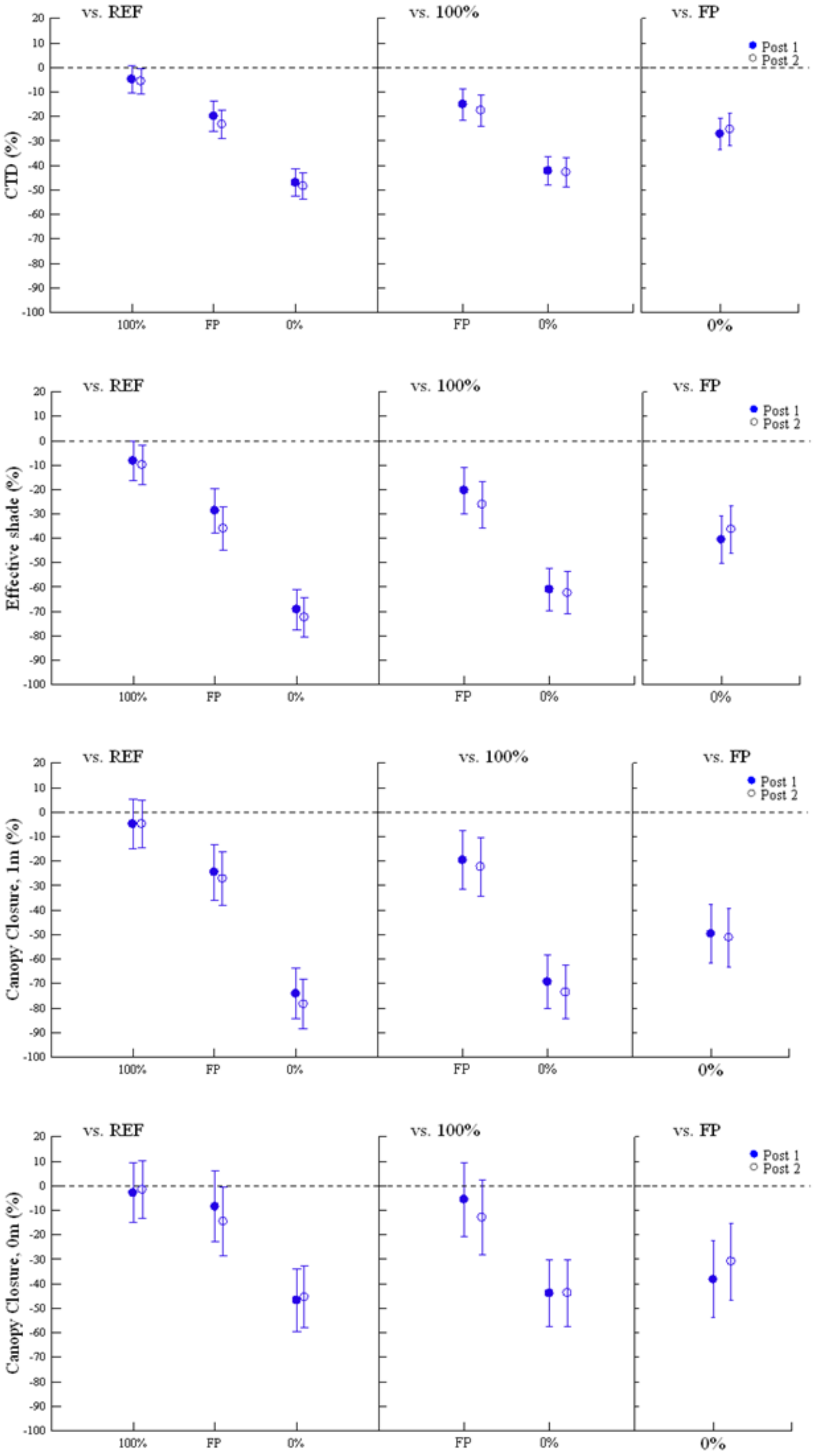

Figure 7-3. Changes in CTD, effective shade, canopy closure-1m and canopy closure-0m (with 95% confidence intervals (CI) based on the post-hoc pairwise comparisons. The confidence intervals that overlap the dotted horizontal line (0%) indicate that there was no significant (P <0.05) difference between those two treatments.

There was a significant treatment × period interaction effect in the GLMM analysis for each of the four shade metrics (**Table 7-4)**. Relative to the reference treatment, we observed significant decreases in CTD and effective shade in all three buffer treatments post-harvest (**Figure 7-3**, **Appendix Table 7-B-5**). By Post 2 CTD decreased by 6, 23, and 49% in the 100%, FP, and 0% buffer treatments, respectively. Effective shade losses by Post 2 were even greater at 10, 36, and 72%, in the 100%, FP, and 0% buffer treatments, respectively.

Table 7-4. Type 3 tests for fixed effects in the generalized linear mixed-effects model for shade metrics. Significant (P <0.05) treatment × period interaction term indicates pre- to post-harvest differences among treatments. Num DF = numerator degrees of freedom. Den DF = denominator degrees of freedom.

Shade Metric	Fixed Effect	Num DF	Den DF	F Value	Pr > F
CTD	Treatment	3	13.0	54.13	<0.001
	Period	2	39.7	260.50	<0.001
	Treatment × Period	6	39.7	73.91	<0.001
Effective Shade	Treatment	3	13.3	50.50	<0.001
	Period	2	40.0	287.35	<0.001
	Treatment × Period	6	40.0	78.72	<0.001
Canopy Closure	Treatment	3	13.0	58.34	<0.001
	Period	2	39.7	146.48	<0.001
	Treatment × Period	6	39.7	55.03	<0.001
Canopy Closure 0m	Treatment	3	12.0	42.10	<0.001
	Period	2	23.7	18.84	<0.001
	Treatment × Period	6	23.6	11.53	<0.001

Relative to the reference treatment, we observed significant decreases in CC-1m only in the FP and 0% buffer treatments (**Appendix Table 7-B-6**). By Post 2, we observed decreases of 5, 27, and 78% in the 100%, FP, and 0% buffer treatments, respectively. Likewise, Post 2 CC-0m decreased significantly in the FP and 0% buffer treatments. By Post 2, we noted decreases of 2, 15, and 45% in the 100%, FP, and 0% buffer treatments, respectively.

7-5.2. *STREAM TEMPERATURE*

The highest monthly average July–August daily summer maximum, minimum, and diel range in water temperatures at the buffer treatment locations are shown in **Table 7-5**. Average pre-harvest maximum daily July–August temperatures ranged from 9.0°C in the WIL3-REF to 13.8°C in the CASC-0%. The mean pre-harvest maximum daily July–August temperatures by treatment were 11.1, 12.3, 10.6, and 11.7°C for the REF, 100%, FP, and 0% treatments, respectively. The WIL3-REF had the lowest pre-harvest maximum and minimum temperatures. This site has extensive reaches with no summer surface flow and only 100 to 200 m of stream immediately above the F/N break with contiguous surface flow in the summer months. The summer of 2009, the first year post-harvest, was warmer at all sites, except the WIL3-REF and CASC-0%, as indicated by July–August maximum and minimum daily temperatures.

Table 7-5. Highest July–August monthly mean of daily maximum, minimum, and diel range of water temperature measured at the buffer treatment location.

Treatment	Block	Average Maximum			Average Minimum			Average Diel Range		
		Pre	Post 1	Post 2	Pre	Post 1	Post 2	Pre	Post 1	Post 2
REF	OLYM	10.7	11.3	10.6	10.0	10.5	9.8	1.0	1.1	0.9
	WIL1	11.9	12.8	12.2	10.8	11.1	10.8	1.3	2.2	1.6
	WIL2-1	11.5	11.8	11.4	10.8	11.0	10.7	0.9	1.1	1.0
	WIL2-2	11.7	12.2	11.4	11.3	11.8	11.1	0.5	0.7	0.6
	WIL3	9.0	8.9	9.3	8.7	8.5	8.6	0.7	1.0	0.7
	CASC	11.6	12.6	11.1	10.4	11.2	9.8	1.5	2.3	1.6
	Average	**11.1**	**11.6**	**11.0**	**10.3**	**10.7**	**10.1**	**1.0**	**1.4**	**1.1**
100%	OLYM	12.5	12.9	12.5	11.5	12.0	11.5	0.9	1.0	0.9
	WIL1	12.1	13.0	12.8	11.0	11.1	11.3	1.2	1.9	1.7
	WIL2	11.5	12.6	12.3	10.8	11.4	10.9	0.8	1.7	1.5
	WIL3	13.0	15.0	14.0	12.1	13.2	12.0	1.5	2.5	2.2
	Average	**12.3**	**13.4**	**12.9**	**11.4**	**11.9**	**11.4**	**1.1**	**1.8**	**1.6**
FP	OLYM	10.1	11.0	10.6	9.6	10.0	9.7	0.7	1.3	1.2
	WIL1	10.4	12.6	11.7	10.1	10.9	10.3	0.5	1.9	1.4
	CASC	11.2	11.7	11.2	10.3	10.3	10.0	1.1	1.7	1.4
	Average	**10.6**	**11.8**	**11.2**	**10.0**	**10.4**	**10.0**	**0.8**	**1.6**	**1.3**
0%	OLYM	9.6	11.0	10.6	9.1	9.5	9.9	0.8	1.6	1.3
	WIL1	11.2	15.1	14.2	10.6	11.4	10.8	0.8	4.0	3.5
	WIL2	12.2	15.2	13.8	11.3	12.2	11.8	1.3	3.4	2.6
	CASC	13.8	14.9	17.2	13.0	12.3	14.0	1.3	3.0	4.8
	Average	**11.7**	**14.1**	**14.0**	**11.0**	**11.4**	**11.6**	**1.1**	**3.0**	**3.1**

7-5.2.1. *Stationarity of Reference Sites and Sensitivity of the Method*

Daily TR values for the maximum water temperature calculated in the reference-to-reference regressions were approximately centered about zero, small in magnitude and displayed no temporal trends that would indicate non-stationarity of the reference sites. Mean post-harvest TR was 0.05, 0.00, and 0.01°C in WIL2-REF2, OLYM-REF, and CASC-REF, respectively (**Figure 7-4**). The mean of the absolute value of TR over all locations in the WIL2-REF2 was 0.27°C and the maximum was 1.52°C. Over 90% of the absolute values were less than 0.58°C and only 1.5% of the 1,982 records exceeded 1.0°C. The mean absolute value of TR in the OLYM-REF was 0.46°C and the maximum was 5.9°C. Only 10% of the 2,170 absolute values exceeded 1.00°C. In the CASC-REF the mean absolute value was 0.59°C, the maximum was 4.03°C, and less than 10% exceeded 1.0°C.

Values of MMTR ranged from −0.41°C to 0.43°C in the WIL2-REF2, from −1.12°C to 0.66°C in the OLYM-REF, and from −0.50°C to 0.93°C in the CASC-REF. Based on the uncorrected P-values, a higher-than-expected proportion of the maximum daily temperature MMTR values differed significantly from zero: 29 of 60 in the WIL2-REF, 18 of 48 in the OLYM-REF, and five of 48 in the CASC-REF (**Figure 7-5**). However, MMTR was nearly equally distributed between positive (80 cases) and negative (76 cases) values.

These results indicate that reference sites were stationary over time and that our method should reliably detect a mean monthly temperature change as small as 0.5 to 1.0°C with well-matched reference-treatment sites (e.g., WIL2-REF1 and WIL2-REF2 in the example above). We assume that the regressions for the OLYM and CASC blocks are comparably powerful because the sites within the blocks were in close proximity and the regression fits were similar to the WIL2 regressions.

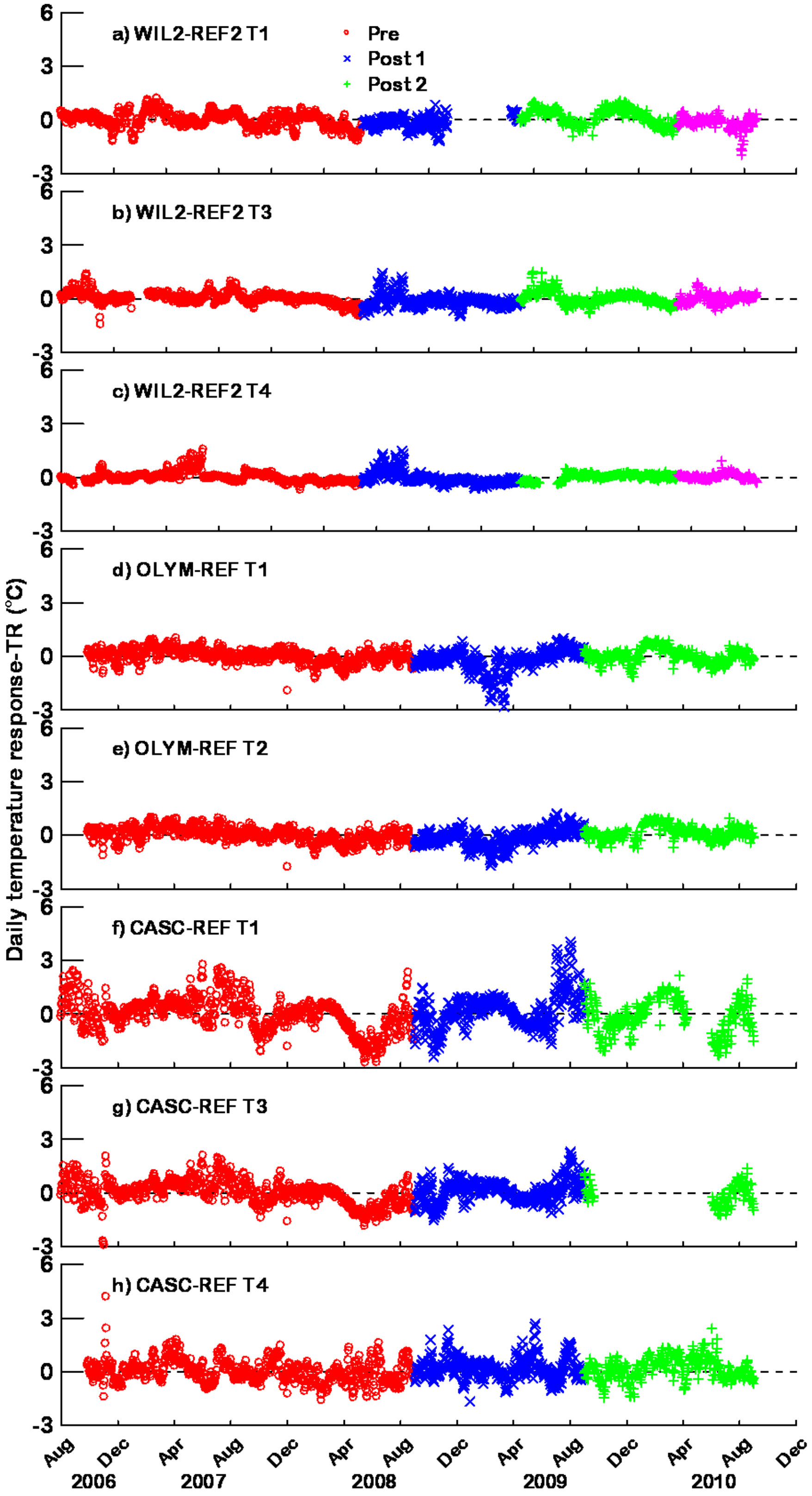

Figure 7-4. Daily temperature response (TR) values for maximum daily temperature calculated from reference to reference comparisons.

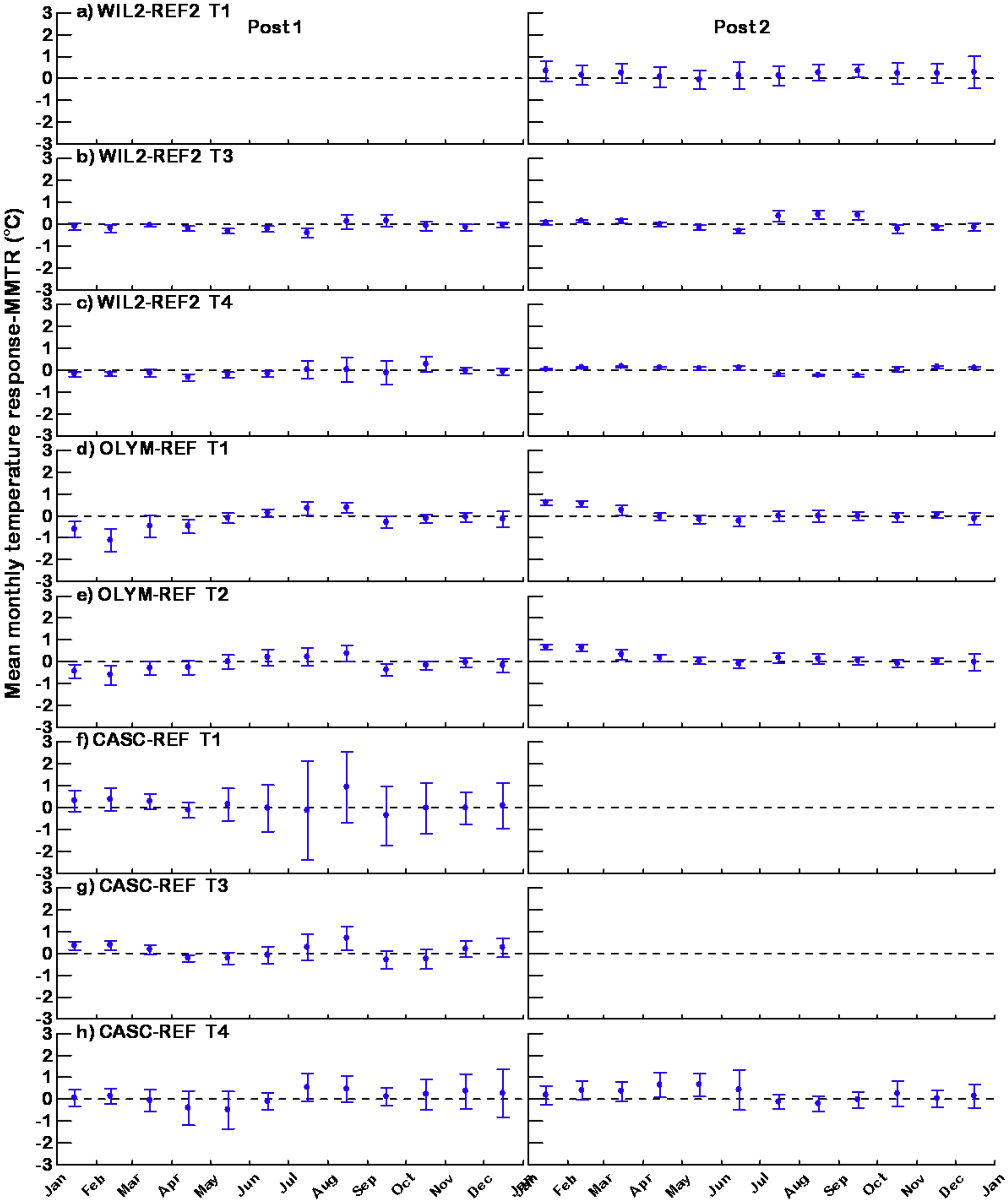

Figure 7-5. Mean monthly temperature response (MMTR) for maximum daily temperature and 95% confidence intervals (CI) calculated from reference-to-reference comparisons for locations in WIL2-REF2 (a-c), OLYM-REF (d-e) and CASC-REF (f-h). We did not calculate MMTR for years with less than 300 consecutive days, or months with less than 20 days, of data.

7-5.2.2. *Response of Water and Air Temperature to Harvest*

Maximum daily water temperatures increased post-harvest in all but one of the harvested sites and was elevated over much of the year at most of the sites. The typical pattern seen in maximum daily temperature response is shown for the WIL1-FP (**Figure 7-6**, top panel). Figures for all sites are included in **Appendix 7-C**. Daily TR increased in late winter or early spring, reached a maximum in July–August and was still elevated well into the fall. We observed this pattern at most of the sites. At two Willapa sites, MMTR was significantly elevated over the entire year. **Table 7-6** shows the MMTR for each site by month for the first year post-harvest (except the WIL2-0% and CASC-0%, where there were missing data from the first year and only second year post-harvest is shown). For the Buffer Treatment locations shown in **Table 7-6**, 94 of the 131 calculated MMTRs were significant and 91 of these significant responses were positive. In comparison, only 52 of 156 MMTR values calculated for the reference sites were significant and these were nearly evenly split with 25 positive and 27 negative responses. This strongly suggests that the pattern of post-harvest increases in daily maximum water temperature is real even though the magnitude of some of the individual MMTRs is relatively small (<0.5°C).

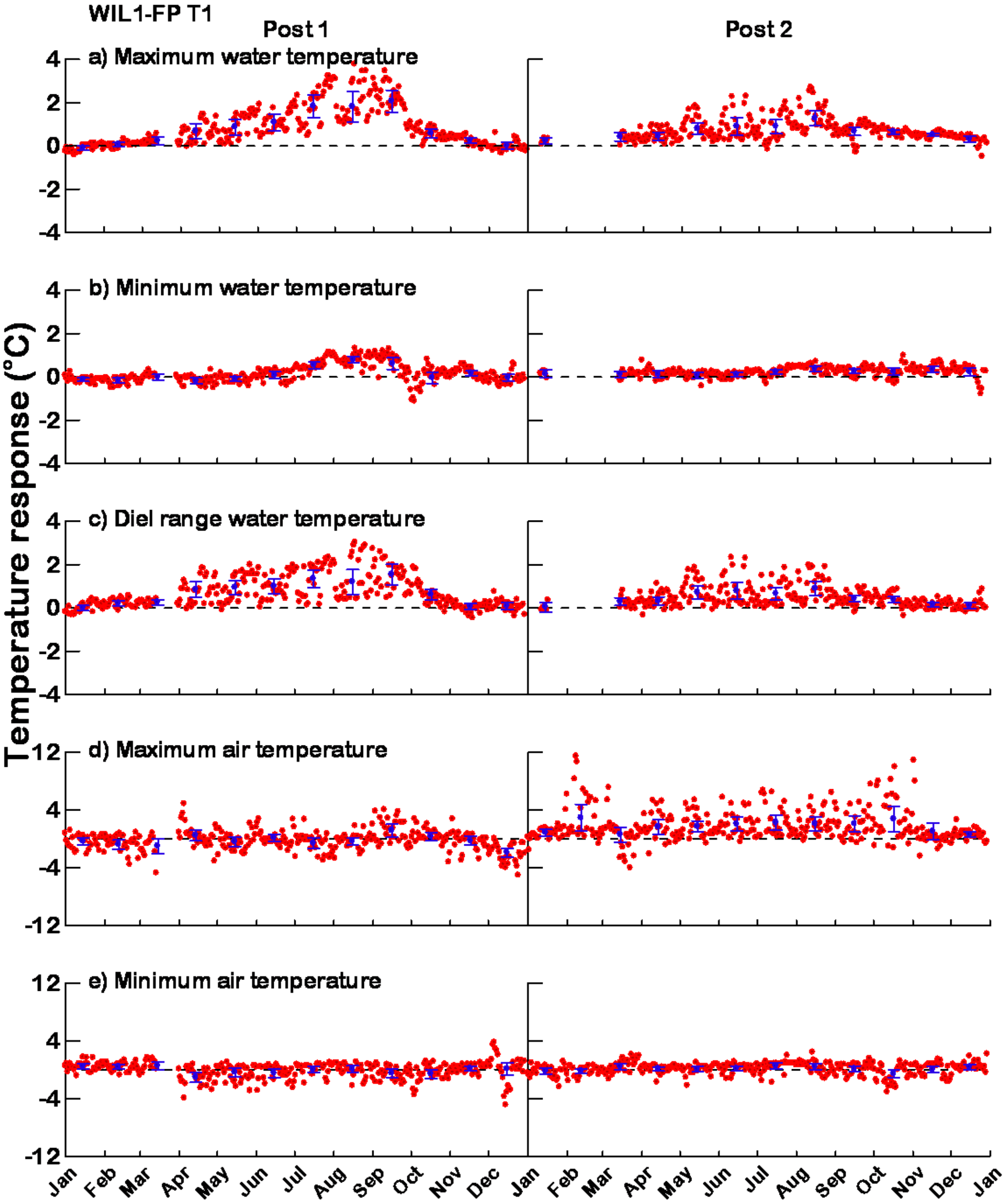

Figure 7-6. Daily TR (red) and mean monthly TR with 95% confidence intervals (CI; blue) for maximum, minimum, and diel range in daily water temperature and maximum and minimum daily air temperature at WIL1-FP T1. We did not calculate MMTR for years with less than 300 consecutive days, or months with less than 20 days, of data.

Table 7-6. Mean monthly temperature response (MMTR) at the Buffer Treatment locations for maximum, minimum and diel range in daily water temperature and maximum and minimum daily air temperature. Red-shaded cells indicate MMTR >0 (P <0.05), that is, higher temperature post-harvest. Blue-shaded cells indicate MMTR <0 (P <0.05) or lower temperature post-harvest. Blocks marked with asterisks (*) indicate that there were missing data from the first post-harvest year and only second post-harvest year data are shown.

Maximum Daily Water Temperature

Treatment	Block	J	F	M	A	M	J	J	A	S	O	N	D
100%	OLYM	0.6	0.6	0.7	0.1	0.2	0.5	0.5	0.8	0.1	−0.3	0.0	0.0
	WIL1	0.3	0.5	0.7	0.4	0.5	0.4	0.9	0.8	0.9	0.5	0.2	0.2
	WIL2	0.3	0.4	0.6	0.5	0.3	0.5	1.0	1.0	1.1	0.2	0.2	0.1
	WIL3	0.4	0.3	0.7	0.9	0.8	0.9	2.3	1.7	1.0	0.0	0.0	0.1
FP	OLYM	0.2	0.4	0.5	0.9	0.4	0.5	0.5	0.3	0.4	0.3	0.1	−0.1
	WIL1	−0.1	0.1	0.2	0.7	0.9	1.1	1.8	1.8	2.1	0.6	0.3	0.0
	CASC	−0.1	0.3	−0.1	0.6	0.3	−0.2	−0.4	−0.1	0.7	0.5	0.2	−0.5
0%	OLYM	0.0	0.1	0.5	0.7	0.9	1.0	1.4	1.0	0.3	0.4	0.1	0.0
	WIL1	0.4	0.2	0.9	1.4	1.8	2.4	3.4	2.4	2.3	0.9	0.5	−0.1
	WIL2*	0.7	0.9	1.8	2.0	2.3	2.3	2.2	1.9	1.4	0.7	0.5	−0.3
	CASC*	0.9		1.3	1.5	1.4	1.2	3.0	3.5	2.2	0.9	0.0	0.3

Minimum Daily Water Temperature

Treatment	Block	J	F	M	A	M	J	J	A	S	O	N	D
100%	OLYM	0.7	0.6	0.7	−0.6	0.2	0.6	0.6	−0.2	0.8	0.0	0.4	0.0
	WIL1	0.1	0.1	0.1	0.1	0.1	0.2	0.4	0.4	0.4	0.4	0.2	0.0
	WIL2	0.2	0.2	0.2	0.1	0.0	0.0	0.3	0.5	0.4	0.0	0.3	0.1
	WIL3	−0.1	−0.4	−0.1	0.0	0.1	0.6	1.3	1.0	0.1	−0.3	−0.1	−0.6
FP	OLYM	0.5	1.0	0.6	0.1	0.1	0.2	0.3	0.3	0.3	0.1	−0.1	0.1
	WIL1	−0.1	−0.2	0.0	−0.2	−0.1	0.1	0.5	0.8	0.6	0.0	0.2	0.0
	CASC	0.2	0.2	−0.3	−0.3	−0.1	0.0	−0.3	−0.4	0.1	−0.1	−0.1	−0.6
0%	OLYM	0.2	0.5	0.5	0.5	0.4	0.4	0.4	0.2	−0.1	0.2	−0.1	0.0
	WIL1	0.1	−0.2	0.2	−0.1	0.0	0.2	0.7	0.7	0.3	−0.1	0.2	0.1
	WIL2*	0.0	−0.2	0.8	0.7	1.2	1.5	1.6	1.5	0.6	0.2	0.4	0.4
	CASC*	−0.3		0.2	−0.1	−0.6	−1.0	−0.1	0.8	0.2	−0.3	−0.5	−0.4

Diel Range Water Temperature

Treatment	Block	J	F	M	A	M	J	J	A	S	O	N	D
100%	OLYM	−0.1	0.0	0.0	0.5	0.1	−0.2	−0.3	−0.9	−0.6	0.0	0.0	0.0
	WIL1	0.1	0.2	0.3	0.2	0.3	0.2	0.6	0.4	0.5	0.1	−0.1	0.0
	WIL2	0.1	0.2	0.3	0.4	0.3	0.5	0.7	0.5	0.7	0.3	−0.2	0.0
	WIL3	0.4	0.2	−0.1	0.7	0.8	0.4	0.7	0.7	0.9	0.6	0.2	0.3
FP	OLYM	−0.2	−0.4	0.0	0.5	0.4	0.5	0.5	0.4	0.2	0.1	0.0	0.0
	WIL1	0.0	0.2	0.3	0.8	0.9	1.0	1.3	1.2	1.5	0.6	0.1	0.1
	CASC	0.0	0.3	0.3	0.9	0.5	−0.1	0.1	0.5	0.8	0.4	0.1	0.0
0%	OLYM	0.0	0.0	0.1	0.3	0.5	0.5	1.0	0.9	0.3	0.1	0.1	0.0
	WIL1	0.2	0.5	1.0	1.6	1.7	2.2	3.0	1.9	2.1	0.9	0.2	0.0
	WIL2*	−0.1	−0.2	−0.3	0.0	0.8	0.8	1.1	0.9	0.9	0.4	0.0	−0.2
	CASC*	0.7		1.2	1.7	2.0	2.4	3.6	2.3	1.9	1.8	0.6	0.5

Table 7-6. (continued)

		Maximum Daily Air Temperature											
Treatment	**Block**	**J**	**F**	**M**	**A**	**M**	**J**	**J**	**A**	**S**	**O**	**N**	**D**
100%	OLYM	0.0	0.1	0.6	0.4	−0.3	0.0	0.6	0.8	1.2	0.1	−0.4	0.0
	WIL1	0.2	1.6	2.5	4.9	4.2	3.9	4.9	3.9	4.9	3.1	0.8	-0.1
	WIL2	1.6	3.7	4.7	5.4	4.3	4.3	4.1	3.9	5.1	3.1	1.5	2.1
	WIL3	1.9	2.3	1.7	4.4	2.5	2.0	2.9	3.6	4.7	3.7	1.6	1.3
FP	OLYM	0.9	2.5	1.9	3.3	2.6	2.4	2.4	1.7	3.3	2.4	0.5	0.3
	WIL1*	0.9	2.9	0.6	1.6	1.7	2.0	2.3	2.1	2.1	2.7	1.0	0.6
	CASC	0.9	2.4	0.6	0.0	0.4	0.3	0.7	0.3	3.8	2.9	1.0	−0.1
0%	OLYM*	−0.5	0.1	0.6	2.2	1.7	1.4	1.1	2.5	3.0	1.4	0.5	−0.1
	WIL1	1.4	3.6	3.9	4.8	3.9	4.4	4.6	3.8	5.3	4.1	1.9	0.4
	WIL2*	0.4	1.0	2.8	3.3	4.0	4.5	4.7	5.4	3.8	2.6	0.6	−0.6
	CASC*	0.3	0.9	0.2	−0.9	1.9	2.2	4.1	2.4	3.9	1.8	0.8	−0.1

Two sites, OLYM-100% and CASC-FP, deviated from this pattern. The OLYM-100% included unstable slope buffers that resulted in very wide stream buffers and only a 3% loss in effective shade post-harvest (**Appendix Table 7-B-4**) and few MMTR values were significantly greater than zero. The CASC-FP stream included a 75 m reach with no surface flow located immediately above the monitoring location in **Table 7-6** and two side-slope seeps located 23 m and 119 m upstream. It is likely that a combination of hyporheic flow and groundwater from seeps moderated water temperature. Overall, in the entire study, we observed a pattern of significantly higher maximum daily water temperatures at 15 of the 23 monitoring locations in the 100% treatment sites, 12 of 15 locations in the FP treatment sites, and 21 of 23 locations in the 0% treatment sites (see **Figure 7-7** and **Appendix 7-D**). It was generally more pronounced at the locations on the mainstem channel, for example, locations T1, T2, T3, and T4, but was sometimes observed even on short tributaries.

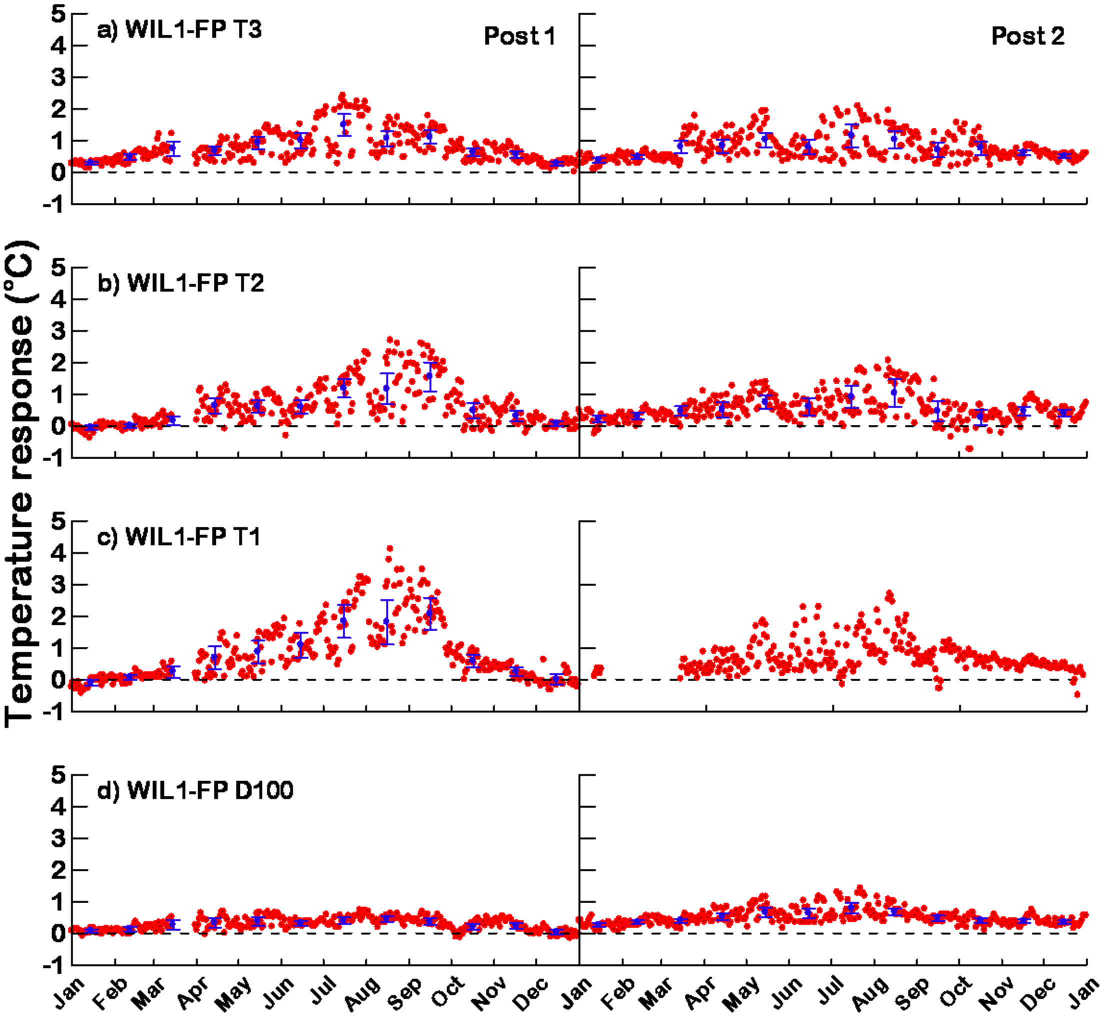

Figure 7-7. Daily TR (in red) and MMTR (in blue) with 95% confidence interval (CI) values for locations in WIL1-FP treatment. We did not calculate MMTR for years with less than 300 consecutive days, or months with less than 20 days, of data.

Warming tended to be greatest in July or August with MMTR ranging from 0.5°C to 2.3°C in the 100%, −0.4°C to 1.8°C in the FP, and 1.0°C to 3.5°C in the 0% treatments. The magnitude of summer warming tended to be lower in the OLYM sites than in the WIL1, WIL2, and WIL3 sites, while the two CASC sites were variable. Warming in the October–March period was less than 1.0°C (MMTR), except for the 0% buffer sites in the WIL2 and CASC block, where March MMTR was 1.8°C and 1.3°C, respectively.

The response of minimum daily water temperature was similar but less pronounced than the maximum. July–August MMTR was less than 1.0°C, except at WIL3-100% and WIL2-0%. Only the CASC-FP showed significant cooling in the summer. Overall, 64 of the 130 MMTR values in **Table 7-6** were significant and 56 of these were positive, suggesting warmer daily minimum temperatures mainly in the summer months.

Changes in the diel range in water temperature were very similar to the changes in the maximum daily temperature (**Table 7-6, Figure 7-6**). Except for the OLYM-100%, MMTR was significantly elevated in most months from March through October. Of the 96 significant responses in **Table 7-5,** 85 were positive.

Maximum daily air temperature increased from April through September, except at OLYM-100% and CASC-FP with the WIL sites showing higher temperatures both earlier and later in the year. Summertime air MMTR values tended to be higher in the WIL sites, regardless of buffer treatment, ranging from 2.1°C to 5.4°C in July and August. Minimum daily air temperatures changed little post-harvest (**Figure 7-6, Appendix 7-C**).

7-5.2.3. *Longitudinal Patterns in Water Temperature, Riparian Cover, and Flow*

Pre-harvest, the July–August Max7D increased in a downstream direction at 10 of the 11 buffer treatment sites (e.g., **Figure 7-8**, see **Appendix 7-E** for all sites). The WIL1-100% was the exception with highest Max7D at the most upstream location (**Appendix Figure 7-E-2**). As a result of higher water temperature post-harvest, especially at upstream locations, Max7D decreased going downstream at four sites: WIL1-100%, WIL3-100%, OLYM-FP, and OLYM-0%. At the other locations, the post-harvest pattern in Max7D was similar to pre-harvest, although at a higher temperature.

Post-harvest, Max7D was higher at 36 of the 40 locations within the harvest units across all 11 buffer treatment sites regardless of presence or absence of a buffer, buffer width, and longitudinal location along the stream (**Appendix 7-E**). Changes in Max7D at the uppermost monitoring location ranged from −1.4°C (decrease) in CASC-0% to 6.2°C in the OLYM-FP and at the lowermost location from −0.8°C at the WIL3-100% to 5.2°C at the WIL1-0%.

Relative to the unharvested sites, there were summertime temperature increases throughout the stream length and across all buffer treatment sites. Of the 40 locations within the harvested portion of the treatment sites (**Appendix 7-E**) 37 had sufficient data to calculate the July MMTR and 30 of these were significantly greater than zero. MMTR was less than zero, indicating significant cooling, at only three locations, and all of these were located in the CASC-FP or CASC-0%. In the 100% sites, MMTR tended to be greater high in the watershed and decreased in a downstream direction. Although we present July MMTR, the pattern is similar for the July through September period.

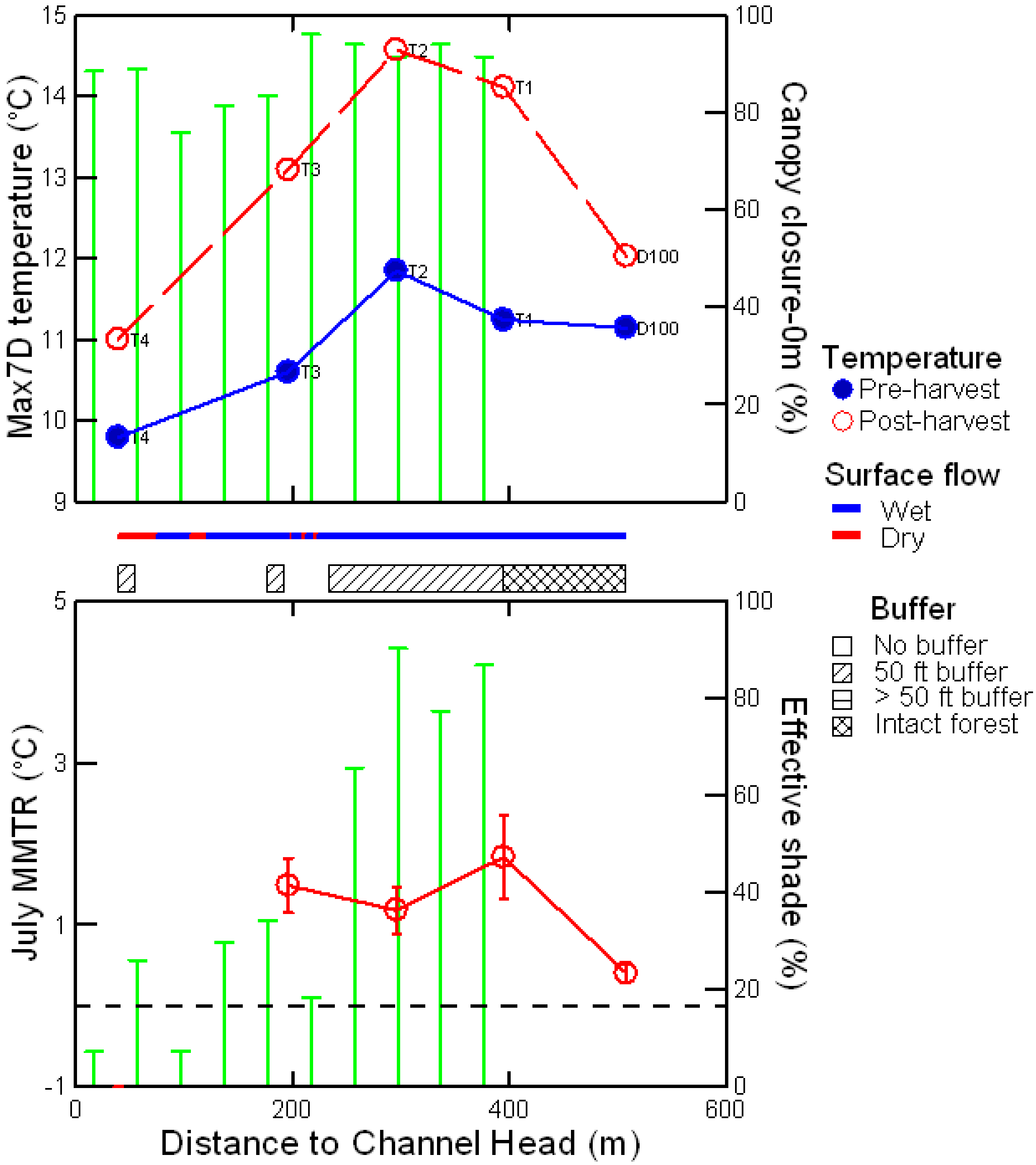

Figure 7-8. Pre- and post-harvest seven-day maximum water temperatures and post-harvest canopy closure along the main stream channel in WIL1-FP (top panel) and July mean monthly temperature response (MMTR) and post-harvest effective shade (bottom panel). Presence (blue)/absence (red) of surface flow is shown in the horizontal bar. Riparian buffer type is shown in the patterned bar. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).

Stream temperature was elevated downstream of the harvest unit compared to pre-harvest conditions, although maximum daily stream temperature did cool (i.e., MMTR increased less post-harvest than the next location upstream) below the harvest unit as the stream flowed through an intact forest. Specifically:

- WIL1-100% July MMTR decreased from 0.9°C at T2 to 0.3°C at D100 after flowing through 100 m of unharvested forest (**Appendix Figure 7-E-2**).
- WIL2-100% MMTR decreased from 0.7°C at T2 to 0.4°C at T1 after flowing through 138 m of unharvested forest (**Appendix Figure 7-E-3**).
- WIL1-FP July MMTR decreased from 1.8°C at T1 to 0.4°C at D100 after flowing through 108 m of unharvested forest (**Appendix Figure 7-E-6**).
- OLYM-0% July MMTR decreased from 1.4°C at T1 to 0.9°C at D100 after flowing through 104 m of unharvested forest (**Appendix Figure 7-E-8**).
- WIL1-0% July MMTR decreased from 3.4°C at T1 to 1.6°C at D100 after flowing through 124 m of unharvested forest (**Appendix Figure 7-E-9**).
- CASC-0% July MMTR decreased from 3.4°C at T2 to 0.2°C at T1 after flowing through 50 m of unharvested forest (**Appendix Figure 7-E-11**); however, in the summer the stream channel was sometimes dry between these locations indicating that flow at T1 was largely hyporheic or groundwater driven.

7-5.2.4. *Effect of Harvest on Seven-Day Average Temperature Response*

7-5.2.4.a. Buffer treatment locations

Period was significant in the GLMM analysis of all three water temperature metrics at the buffer treatment locations and for maximum air temperature (**Table 7-7**) indicating significant post-harvest temperature increases in each buffer treatment. The 7DTR for maximum daily water temperature was 1.2°C higher in each post-harvest year in the 100% treatment, 1.4°C and 1.0°C higher in Post 1 and Post 2, respectively, in the FP treatment, and 3.4°C and 3.0°C higher in Post 1 and Post 2, respectively, in the 0% treatment (**Figure 7-9**; **Appendix Table 7-B-7**).

Table 7-7. Type 3 tests for fixed effects in the generalized linear mixed-effects model for July–August seven-day average temperature metrics at the buffer treatment locations. Significant (P <0.05) period term indicates pre- to post-harvest differences within a treatment. Num DF = numerator degrees of freedom. Den DF = denominator degrees of freedom.

Fixed Effects	Num DF	Den DF	F Value	Pr > F
7DTR July–Aug Maximum Daily Water Temperature				
Treatment	2	8.9	4.50	0.045
Period	2	31.5	32.98	<0.001
Treatment × Period	4	31.5	4.21	0.008
7DTR July–Aug Minimum Daily Water Temperature				
Treatment	2	8.3	1.61	0.256
Period	2	31.1	24.49	<0.001
Treatment × Period	4	31.1	1.61	0.198
7DTR July–Aug Diel Range Water Temperature				
Treatment	2	8.8	10.11	0.005
Period	2	30.1	35.13	<0.001
Treatment × Period	4	30.1	6.92	<0.001
7DTR July–Aug Maximum Daily Air Temperature				
Treatment	2	9.5	5.63	0.024
Period	2	30.6	39.98	<0.001
Treatment × Period	4	30.6	2.72	0.048
7DTR July–Aug Minimum Daily Air Temperature				
Treatment	2	8.4	0.74	0.507
Period	2	31.0	0.87	0.430
Treatment × Period	4	31.0	1.65	0.187

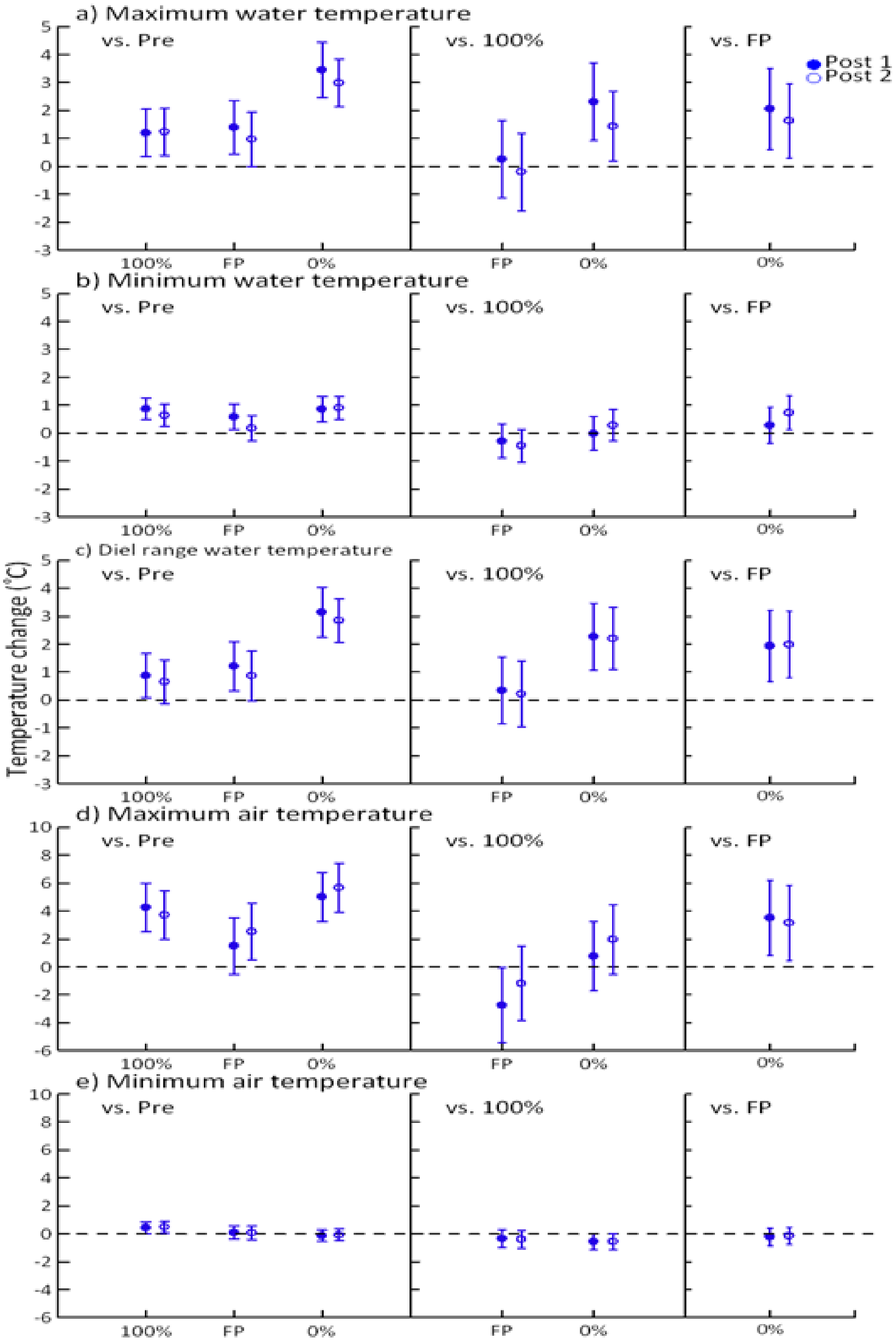

Figure 7-9. Pairwise comparisons of buffer treatment effects on seven-day average temperature response for July–August water and air temperature metrics measured at the buffer treatment locations. Error bars are 95% confidence intervals (CI).

Minimum water temperature 7DTR increased significantly by 0.6–0.9°C in all buffer treatments (**Figure 7-9; Appendix Table 7-B-8**). There was no significant difference in response among treatments, except for FP vs. 0% in Post 2 (**Appendix Table 7-B-8**). Significant increases in the diel temperature range of 0.9°C and 1.2°C were seen in the 100% and FP treatments, respectively. The increase at the 0% treatment was significantly greater than the other buffer treatments, 3.1°C and 2.8°C in Post 1 and Post 2, respectively (**Appendix Table 7-B-8**).

Maximum air temperature 7DTR increased by 4.2°C and 3.7°C in the 100% treatment in the first two years post-harvest, 1.5°C and 2.5°C in the FP treatment, and 5.0°C and 5.7°C in the 0% treatment (**Figure 7-9; Appendix Table 7-B-9**). Minimum air temperature 7DTR response post-harvest was inconsistent, and significant only in the 100% treatment where it increased 0.4°C and 0.5°C in Post 1 and Post 2, respectively.

7-5.2.4.b. F/N break locations

We observed significant post-harvest increases in maximum water 7DTR in all treatments (**Table 7-8; Figure 7-10; Appendix Table 7-B-7**). The 7DTR was 0.9°C and 0.6°C higher in each post-harvest year in the 100% treatment, 1.4°C and 1.0°C higher in Post 1 and Post 2, respectively, in the FP treatment, and 3.1°C and 2.7°C higher in Post 1 and Post 2, respectively, in the 0% treatment.

Table 7-8. Type 3 tests for fixed effects in the generalized linear mixed-effects model for July–August seven-day average maximum daily water temperature at the F/N break locations. Significant (P <0.05) period term indicates pre- to post-harvest differences within a treatment. Num DF = numerator degrees of freedom. Den DF = denominator degrees of freedom.

Fixed Effects	Num DF	Den DF	F Value	Pr>F
Treatment	2	8.4	4.95	0.038
Period	2	32.8	34.13	<0.001
Treatment × Period	4	32.8	6.18	<0.001

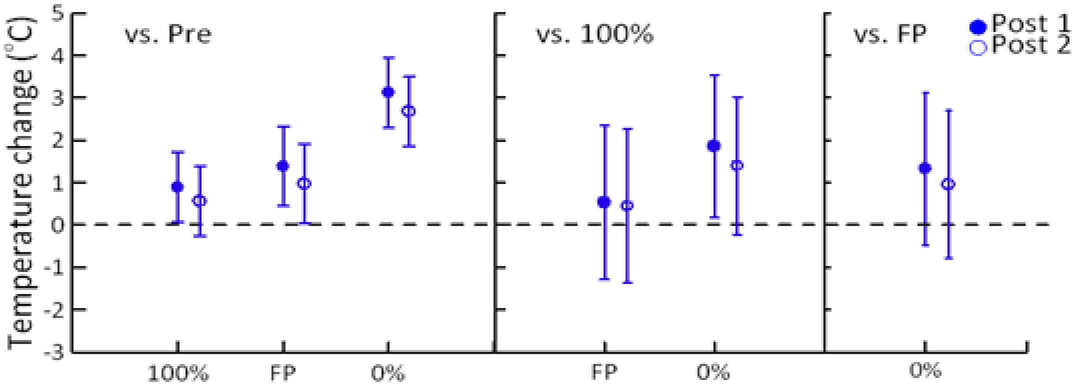

Figure 7-10. Pairwise comparison of treatment effects on the seven-day average maximum daily July–August water temperature at the F/N break.

7-6. DISCUSSION

7-6.1. *EFFECTIVENESS AT MAINTAINING STREAM SHADE*

We found only two studies that reported shade for same-sized streams with similar width and height riparian buffers that did not allow some thinning within the buffer. Mean pre-harvest CTD was 94% and 95% for Janisch and colleagues (2012) and this study, respectively. Their mean first year post-harvest CTD for a continuous, 50-foot wide buffer (86%), patch-cut buffer (75%) and clearcut (53%) harvest treatment were very similar to our analogous 100% (86%), FP (74%), and 0% (44%) treatments. Mean pre-harvest CC-1m in this study was slightly higher than Schuett-Hames and colleagues (2012) measurement of unharvested Type Np streams in western Washington State, 95% and 89%, respectively. Post-harvest mean CC-1m in the FP treatment in this study and Schuett-Hames and colleagues (2012) decreased to 72% and 76%, a decrease of 23% and 13%, respectively.

Buffer width exceeded 50 ft (15.2 m) at 18 of the 109 locations in the 11 buffer treatment sites where we measured shade: 14 locations in the 100% and two each in the FP and 0% treatment sites. We examined the canopy data across all 17 sites by the actual width of buffer at individual measurement locations: unharvested reference, greater than 50 ft (15.2 m), equal to 50 ft (15.2 m), and no buffer (**Figure 7-11**). These data indicate that very little difference exists between the locations within unharvested reference sites and locations within harvested sites where buffers were much wider than 50 ft (15.2 m). Locations with 50-ft (15.2-m) wide buffers had substantially lower shade values and, as expected, the unbuffered locations had the lowest. The results show similar patterns for all four shade metrics. The greatest potential impact was in the 100% treatment where 14 of the 40 shade measurement locations had buffers wider than the intended 50 ft (15.2 m) and our shade estimates are likely higher than if the treatments had been applied as intended. The effect on the FP and 0% treatments is likely minor because only two locations did not match the intended buffer treatment.

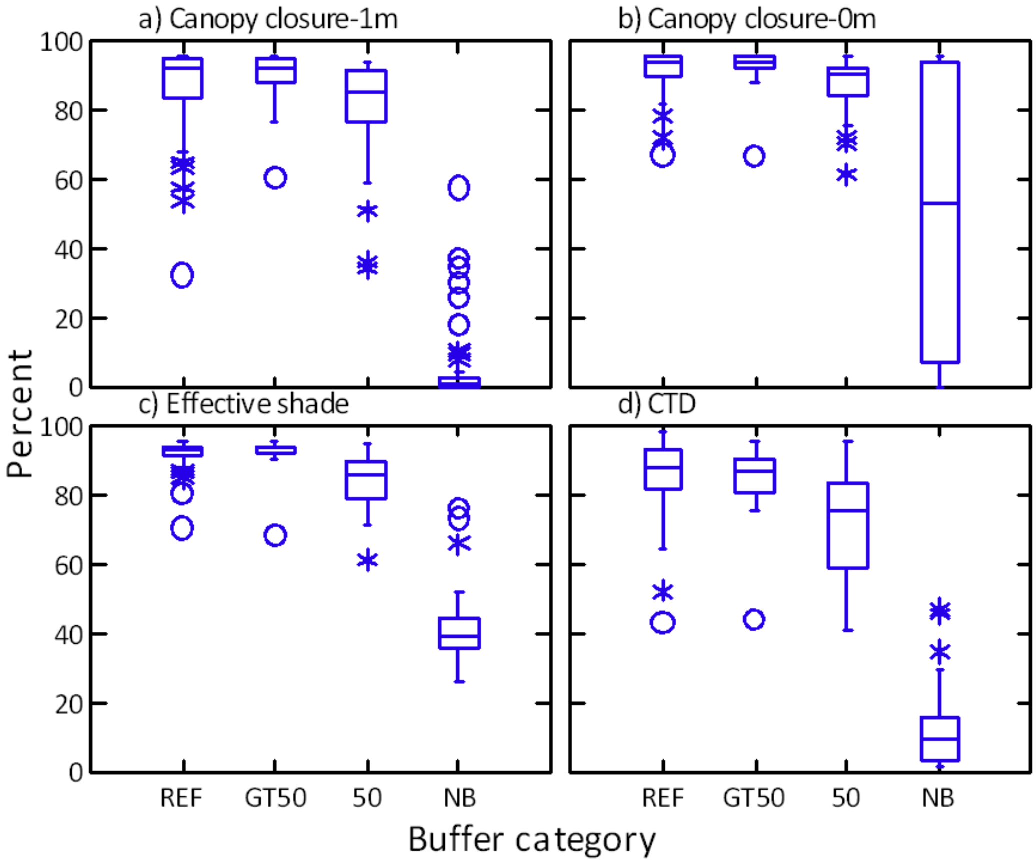

Figure 7-11. Distribution of post-harvest shade values by buffer width category. REF = reference site, GT50 = buffer greater than 50 ft (15.2 m), 50 = buffer equal to 50 ft (15.2 m), NB = no buffer.

7-6.2. *STREAM AND AIR TEMPERATURE RESPONSE TO HARVEST*

7-6.2.1. *Summer Water and Air Temperature Near Bottom of Harvest Unit*

7-6.2.1.a. Maximum water temperature

None of the three buffer treatments in our study were successful in preventing significant increases in maximum stream temperature. These higher temperatures persisted from April–October at most locations along the stream, with the greatest increase usually seen in July or August. Few published studies used methods similar to ours to calculate temperature change so we are unable to make direct comparisons with much of the earlier research. However, the magnitude and direction of change we observed toward higher stream temperature post-harvest, regardless of buffer treatment, has been described elsewhere. For example, Macdonald and colleagues (2003) reported a mean increase of 0.5°C in daily maximum stream temperature in their only stream with a patch retention buffer (similar to our FP buffer except with a 66-ft (20.1-m) wide buffer along the lower 60% of the stream length). Likewise, Wilkerson and colleagues (2006) reported an increase in the seven-day average maximum temperature of 1.4–4.4°C after harvest in the no buffer treatment.

There were three recent studies using similar methods on similar-sized streams with which we can directly compare our results. Janisch and colleagues (2012) reported the mean July–August temperature increased by 0.61°C and 1.06°C in treatments analogous to our 100% and FP treatments, respectively, very similar to the 0.7 and 1.1°C change in the 100% and FP buffer treatments 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, slightly lower than the 3.2°C increase we observed in 0% treatments. Guenther and colleagues (2014) observed mean July–August temperature increases of 1.64–3.00°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 canopy closure near the bottom of the harvest unit, the inverse of our FP treatment (little or no buffer in the upper reach and a 50-ft (15.2-m) buffer in the lower portion) which may have affected the outcome.

The magnitude of change in 7DTR for maximum daily water temperature in this study was correlated with the average post-harvest value of each of the four shade metrics (**Figure 7-12**). This is consistent with numerous studies of larger streams over the past four decades that show that riparian buffers (shade) can mitigate the effects of forest harvest on stream temperature (Brown and Krygier 1970; Castele and Johnson 2000; Johnson 2004; Moore *et al.* 2005a; Groom *et al.* 2011). However, it contrasts with Janisch and colleagues (2012) who found no correlation between temperature change and CTD in very small (<8.5 ha) streams with spatially intermittent summer flow. However, Janisch and colleagues (2012) noted, but did not quantify, slash accumulation up to 1 m deep over the unbuffered reaches of channel that may have confounded their temperature-shade relationship and their study included two very distinct lithologies.

We expected that canopy closure measurements that included slash and understory (e.g., Jackson *et al.* 2001) would have provided a better predictor of temperature change than canopy closure at 1 m, and that estimating effective shade along the solar path would be a better predictor than CTD; however, based on the strength of the correlations and the associated P-values, all shade metrics were roughly equivalent.

It is important to note that the OLYM-100%, with very wide (>50 feet) buffers and very little change in riparian shade after harvest (−4% to +3.5%, depending upon the shade metric), was the only site that did not exhibit significant warming over much of the year at multiple locations along the stream. In spite of this, we observed a significant increase in stream temperature in the 100% treatment where the mean decrease in shade was less than 10%. This is consistent with Groom and colleagues (2011) who observed a detectable increase in maximum daily stream temperature with a post-harvest decrease in shade of only 6%.

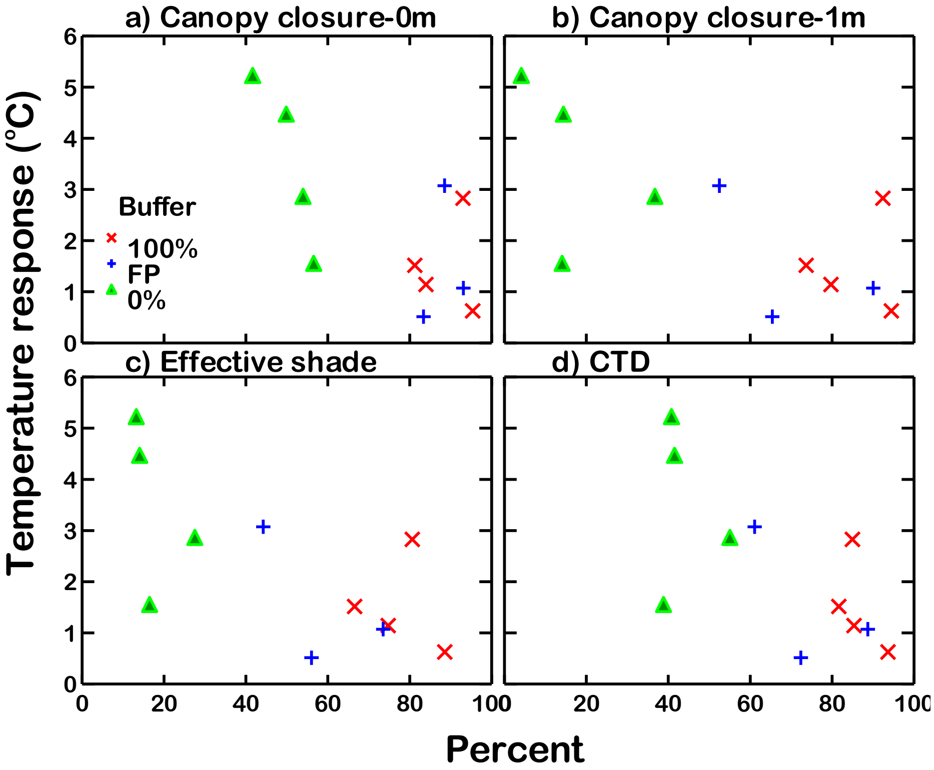

Figure 7-12. Correlation between seven-day average temperature response (7DTR) and shade metrics. Correlation coefficients and P-values are: CC-0m (r = −0.697, P = 0.017); CC-1m (r = −0.697, P = 0.017); effective shade (r = −0.686, P = 0.041), and CTD (r = −0.728, P = 0.011).

We examined the length of contiguous wetted channel above the F/N break location, total length of wetted channel, distance to channel head, and the estimated change in solar energy reaching the stream (based on the channel length, width, and estimates of solar radiation calculated in HemiView) and found no correlation (P >0.05) with change in stream temperature. However, when examined by harvest treatment (**Figure 7-13** upper panel), temperature change did increase with greater length of contiguous surface flow above the monitoring location in all three treatments and decreased with increasing proportion of dry channels in the summer months. This is consistent with the Janisch and colleagues (2012) findings in small streams. The length of surface flow above the monitoring station provides an index of the stream area that could be exposed to increased solar radiation after harvest. However, it was not possible to separate the effects of length of surface flow and percent dry channel because the metrics are related. However, it is clear that there is no threshold of minimum wetted channel length below which a measureable temperature response was not detected.

Another consideration is the impact of large (relative to surface flow) inputs of groundwater and hyporheic exchange on stream temperature (Johnson and Jones 2000; Storey *et al.* 2003; Wondzell 2006). In spite of the spatially intermittent (i.e., dry) reaches in many of our streams (where flow immediately downstream is 100% groundwater or hyporheic flows), our results do not show a consistent pattern of less temperature change downstream of dry reaches (**Appendix 7-C**). However, the temporal variability in the surface flow (we surveyed surface flow only two times during the entire study) could have missed location-specific responses.

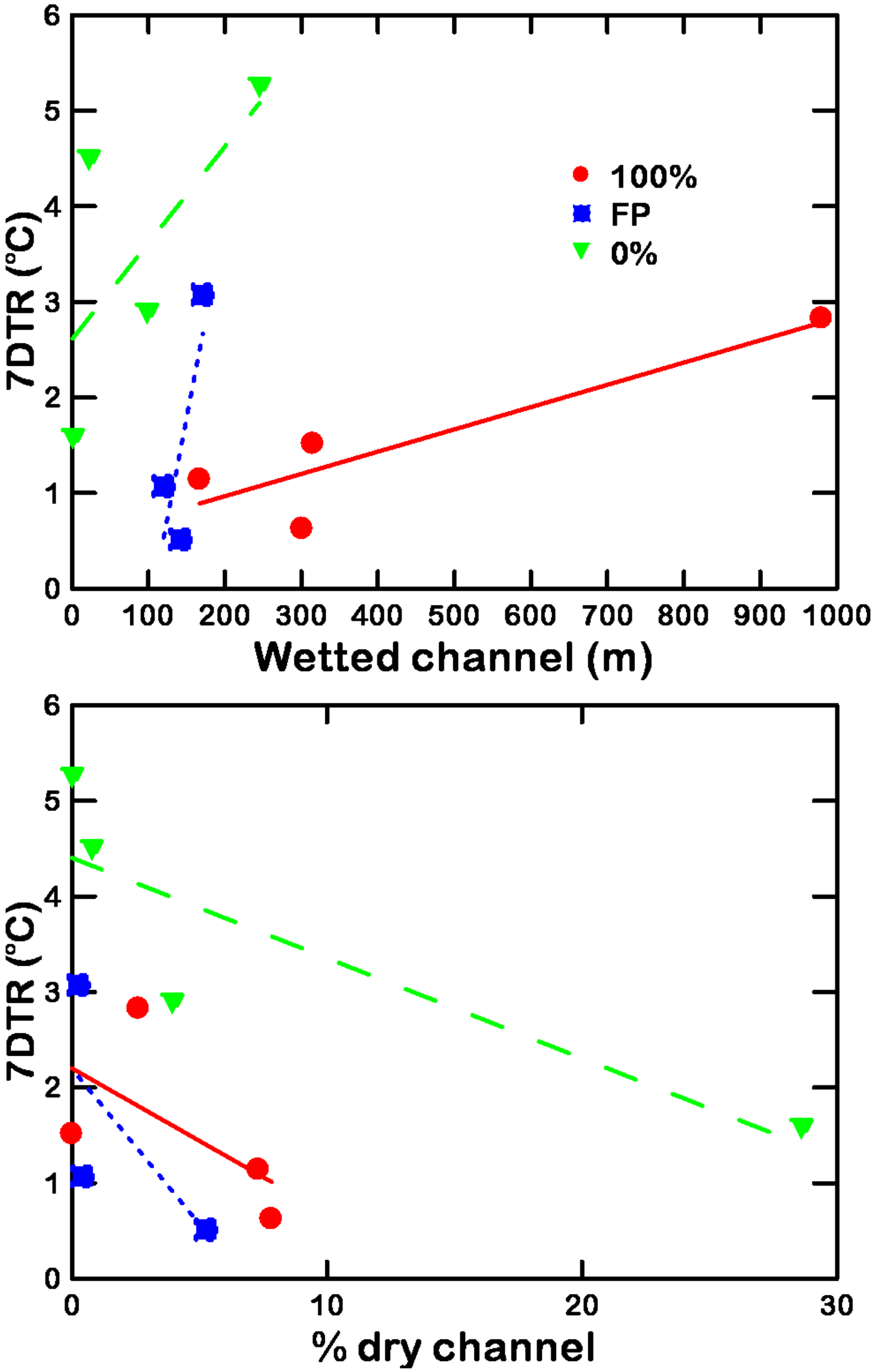

Figure 7-13. Change in 7DTR plotted against length of wetted channel above the monitoring location (top panel) and percent of the perennial Type N channel that had no summer surface flow (bottom panel). The temperature effect data are from the first post-harvest year. The % dry channel data are an average of surveys taken in the summers of 2006 and 2010.

7-6.2.1.b. Minimum water temperatures

We observed higher July–August minimum stream temperatures at eight of the 11 treatment sites and lower temperatures at only one site with MMTR ranging from −0.4°C to 1.6°C. In the only study using similar analytical methods, Guenther and colleagues (2014) reported mean July–August minimum water temperature increased by 0.59–1.04°C after a partial harvest that decreased canopy closure by 14%. Johnson and Jones (2000) also reported increases of 1.8–2.0°C in mean weekly summer minimum temperature in an unbuffered stream in a clearcut catchment in the Oregon Coast Range, and a change ranging from 0.1°C decrease to 1.0°C increase over the first three years after a debris flow. However, this response is not consistent, for example, Mellina and colleagues (2002) reported a decrease in mean August minimum daily temperature of 0.2–1.1°C in an interior British Columbia stream with a 30 m thinned buffer. The different responses in these studies may be due to the differences in water residence time, hyporheic exchange, climate, harvest, or buffer type.

7-6.2.1.c. Diel range in water temperature

Only a few studies have reported on the effects of timber harvest with various buffer treatments on the diel range of water temperature. Holtby and Newcombe (1982) reported increases of 1.4–2.5°C on unbuffered streams within clearcut catchments, and Johnson and Jones (2000) reported a diel range of 6–8°C in a clearcut harvested bedrock-dominated basin compared to only 1–2°C in the unharvested reference streams. We found a small increase in the July–August diel range in water temperature averaged across July–August (1.7°C) in the 0% buffer treatment (and only 0.3°C and 0.6°C in the 100% and FP treatments).

7-6.2.1.d. Maximum daily air temperature

We observed increased maximum daily air temperature at nearly all harvested sites over much of the year. Mean monthly maximum daily temperature response across all sites ranged from 2.4°C to 3.7°C April–October, higher than the 2°C reported by Guenther and colleagues (2012); however, Guenther and colleagues (2012) estimated the change relative to 25°C air temperature at the reference site. Our maximum daily air temperatures exceeded 25°C at most of our sites over this period and may account for some of the difference.

A number of studies in the Pacific Northwest (Chen *et al.* 1993a 1993b 1995, Brosofske *et al.* 1997) and elsewhere (Raynor 1971, Young and Mitchell 1994, Cadenasso *et al.* 1997, Davies-Colley *et al.* 2000, Hagan and Whitcomb 2000, Spittlehouse *et al.* 2004) suggest that most of the change in maximum air temperature occurs within 15–60 m of the buffer edge. This is consistent with our observation that the only site where air temperature did not increase was OLYM-100%, which had buffers much wider than the prescribed 50 ft (15.2 m) width.

7-6.2.2. *Longitudinal Variability in Stream Temperature Response to Harvest*

Post-harvest, stream temperature increased throughout the stream network from spring through fall at nearly all treatment sites, but the pattern of the response varied among the study sites. At five sites, warming was greatest higher in the catchment. As a result, temperature changed from gradual warming in downstream direction at 10 of 11 sites before harvest to cooling in a downstream direction at four of 11 sites after harvest. Dent and colleagues (2008) saw variable patterns in pre-harvest downstream temperature trends in larger streams but at a slightly larger spatial scale (0.5–1.7 km long) than ours, as did Poole and Berman (2001), both in the Oregon coast range, and Torgerson and colleagues (1999) and Ebersole and colleagues (2003) in northeastern Oregon. Pre-harvest longitudinal trends were likely a result of tributary inflow and the influx of cooler groundwater or hyporheic flows (Beschta *et al.* 1987; Ebersole *et al.* 2003; Wondzell 2006) because riparian canopy cover was uniformly high in all watersheds. The post-harvest changes we observed were likely due to differential heating of the water along the stream length and the degree to which groundwater and hyporheic flows influenced each stream.

7-6.2.3. *Downstream Recovery*

There was a clear pattern of decreasing maximum daily stream temperature and a smaller increase in MMTR for maximum daily water temperature after leaving the harvest unit and flowing through unharvested forest. Reported post-harvest temperature responses downstream of harvest units are variable in direction and magnitude (Caldwell *et al.* 1991; Zwienieki and Newton 1999; Storey *et al.* 2003; Garner *et al.* 2014). Caldwell and colleagues (1991) noted lower temperature in well-shaded reaches below some unbuffered headwater streams. Zwienieki and Newton (1999) and Storey and colleagues (2003) observed cooling within a few hundred meters below the harvest unit. Gravelle and Link (2007) noted an increase in headwater stream temperatures after a two-sided harvest but no effect at a downstream location. Kibler and colleagues (2013) also detected no cumulative effect on downstream temperature after harvest along headwater streams in Hinkle Creek in western Oregon. However, in the latter two studies the harvested area comprised a small proportion of the catchment area where downstream effects were measured. Wilkerson and colleagues (2006) found a 2.5°C higher maximum stream temperature 100 m below their unbuffered harvest treatment and no measureable change with buffer widths of 11 m or more. However, the sensitivity of the analysis may have been compromised because over half of their sites were dry for at least some portion of the June 15 to August 15 study period. Keith and colleagues (1998) observed greater cooling downstream when upstream temperature was high than on cool days when water temperature was lower, and Storey and Cowley (1997) observed greater downstream cooling in streams with higher water temperatures consistent with our observations.

Energy budgets (Brown *et al.* 1971; Storey *et al.* 2003; Garner *et al.* 2014) indicate there are net energy gains to the stream even under a forest canopy, suggesting that downstream cooling is due to inputs of cooler groundwater or hyporheic flows. However, Johnson (2004) observed substantial cooling across a 150 m reach of bedrock channel after it was artificially shaded which they attributed to the relatively large influence of net longwave radiation and evaporative cooling under shaded conditions. We did not measure groundwater or hyporheic inputs or estimate an energy budget, but given the low summer discharges in these streams (often less than 1 L s^{-1}) and the presence of seeps, it is likely that our observed cooling trends are partially due to groundwater inputs and hyporheic exchange.

7-6.2.4. *Implications of the 2007 Windstorm*

The effects of the December 2007 windstorm on the WIL-REF, WIL-100% and WIL1-FP can be seen in lower values for CTD, effective shade, and CC-1m in 2008 relative to 2007 (**Tables 7B-1 to 7B-3**). The likely impacts on this study's results are:

- Mean pre-harvest shade values were lower and the pre-harvest, within-treatment variance was greater because of the difference between 2007 and 2008 values. Higher variance may have reduced the statistical power of the analysis to detect a change.
- Mean post-harvest shade values may be lower due to the pre-harvest windthrow within the buffer.
- Pre-harvest stream temperatures may have been higher at sites with extensive windthrow. If so, this would have resulted in greater variability in the pre-harvest TR values, which may have reduced the power to detect a change in temperature.

We did detect relatively small, but significant post-harvest changes in stream cover and stream temperature, which suggests that the effects of the windthrow event were small compared to the buffer treatment effects observed.

7-7. CONCLUSIONS

1) Buffer widths greater than 50 ft (15.2 m) are needed to prevent shade loss.
2) None of the three buffer treatments tested were effective at preventing increases in maximum water temperature after harvest, although both the 100% and the FP treatments were clearly more effective than the 0% treatment.
3) Maximum daily water temperature increased at most locations along all streams with measureable shade loss.
4) Increases in maximum water temperature were greatest during July and August; however, small, but significant, increases often persisted from spring through the fall and, at some sites, year round.
5) There were small, but significant, increases in the minimum daily temperature and in the diel temperature range in all treatments.
6) Maximum water temperature decreased below the harvest unit after flowing through approximately 100 m of intact forest, but was still elevated compared to pre-harvest conditions.

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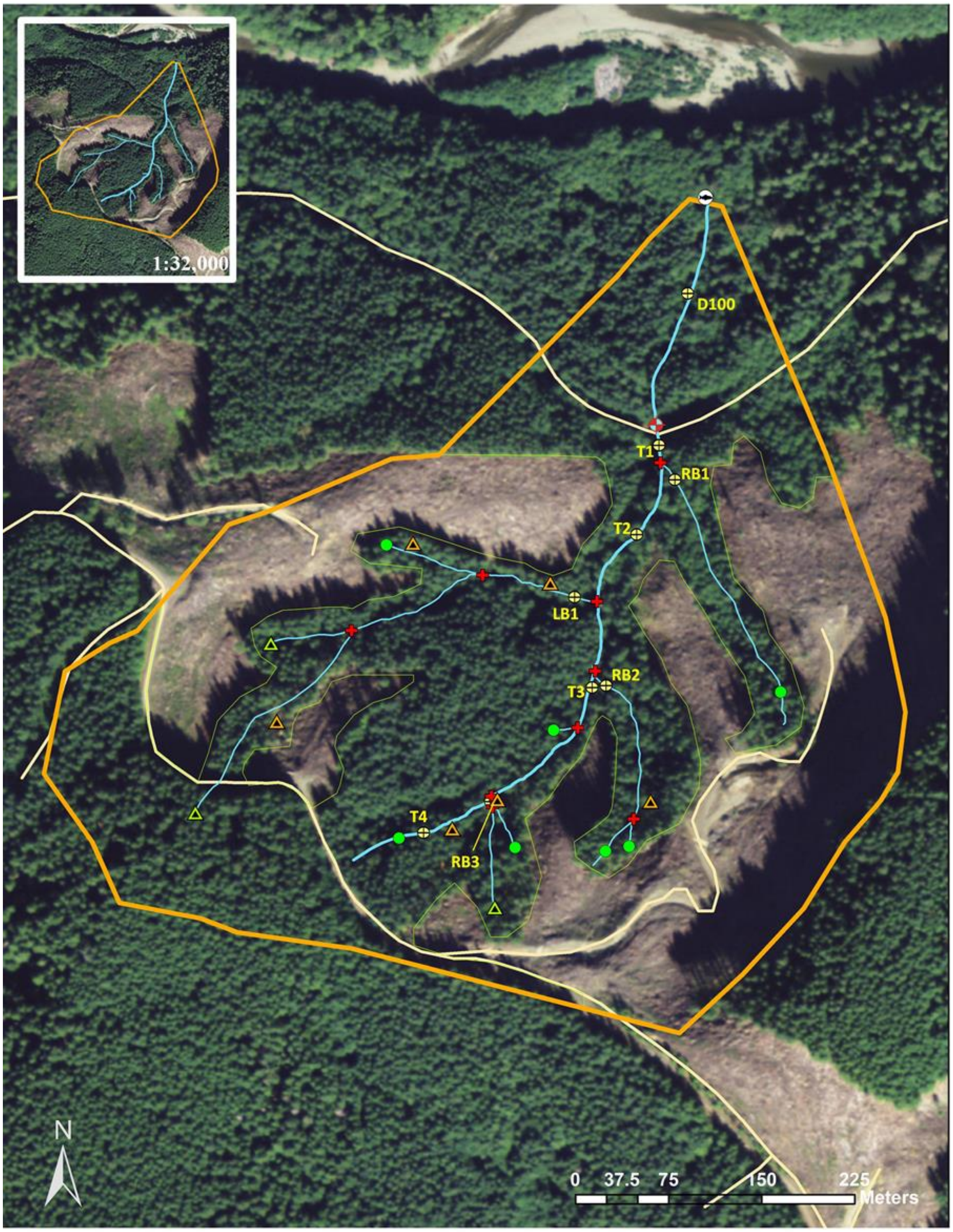
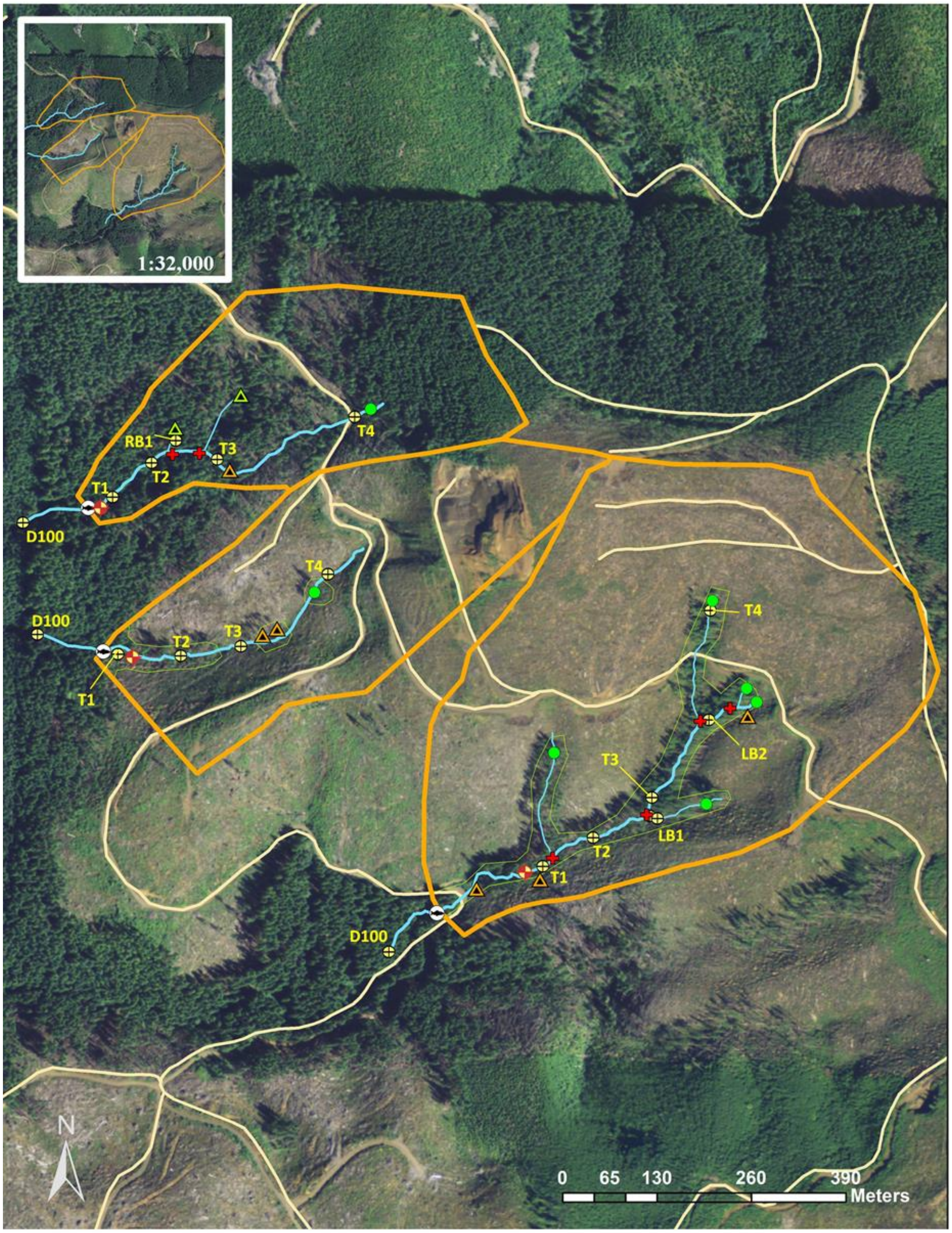
APPENDICES

APPENDIX 7-A. SITE PHOTOGRAPHS

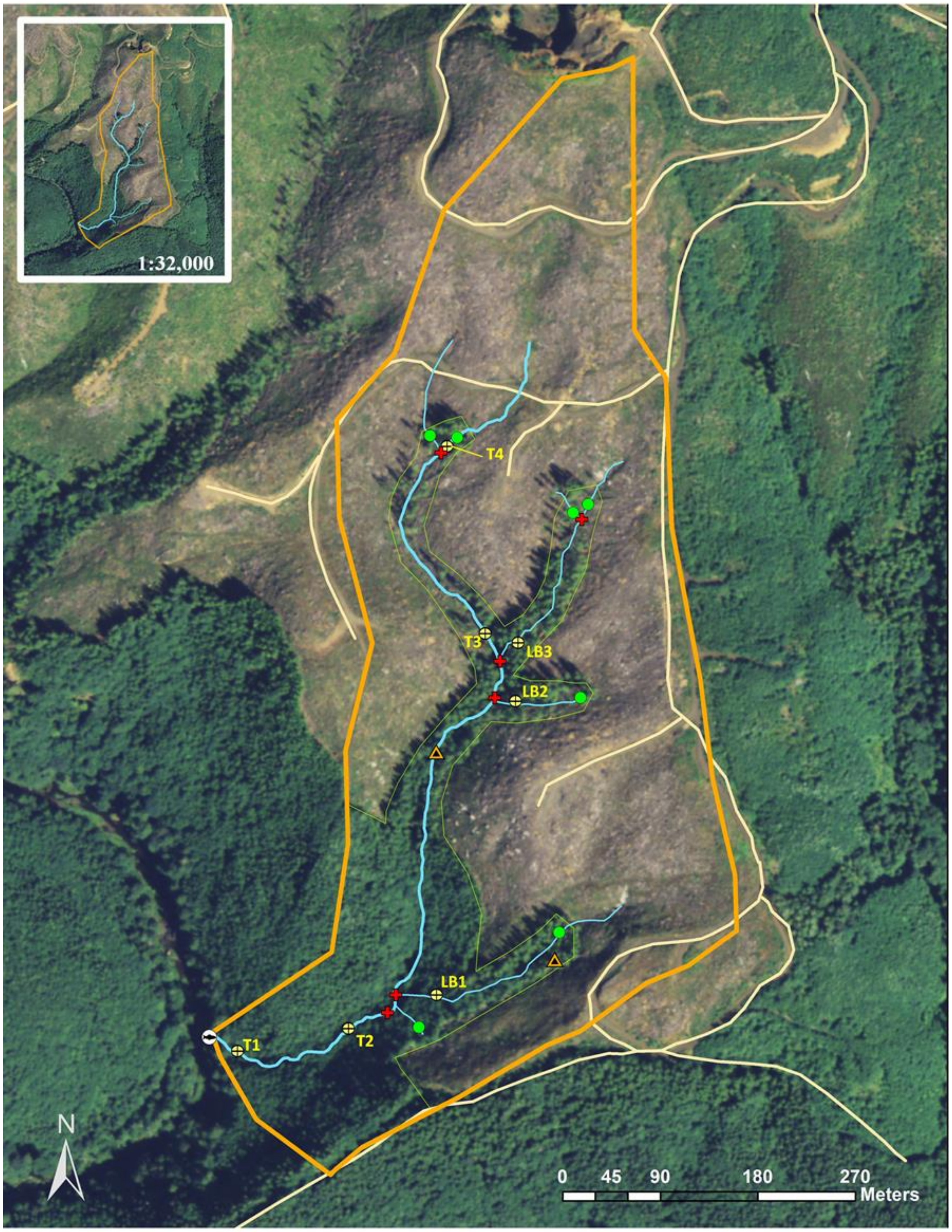
Appendix Figure 7-A-1. Legend for site photographs.



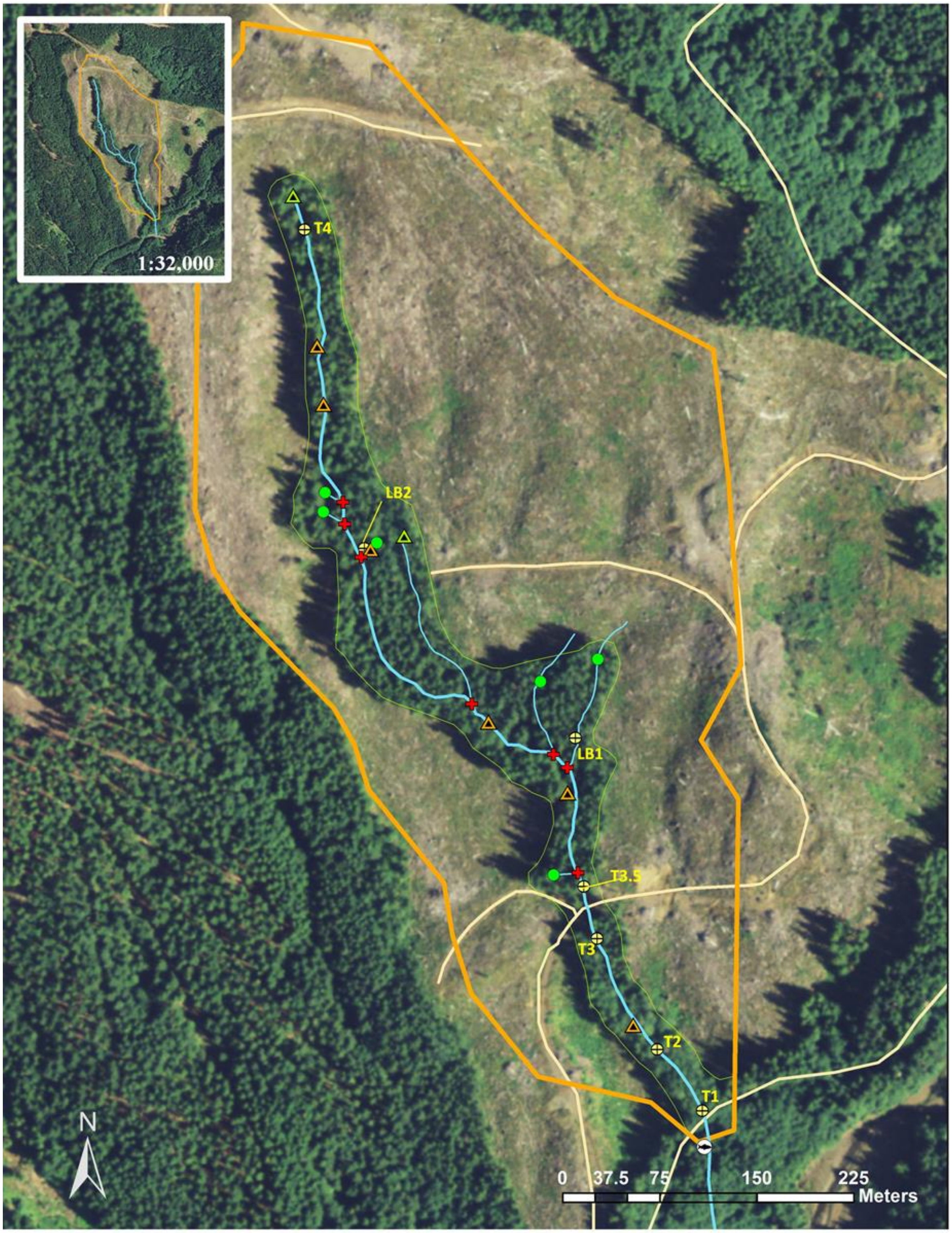
Appendix Figure 7-A-2. Monitoring locations in OLYM-100% site. Unstable slope buffers resulted in buffers much wider than 50 ft (15.2 m).



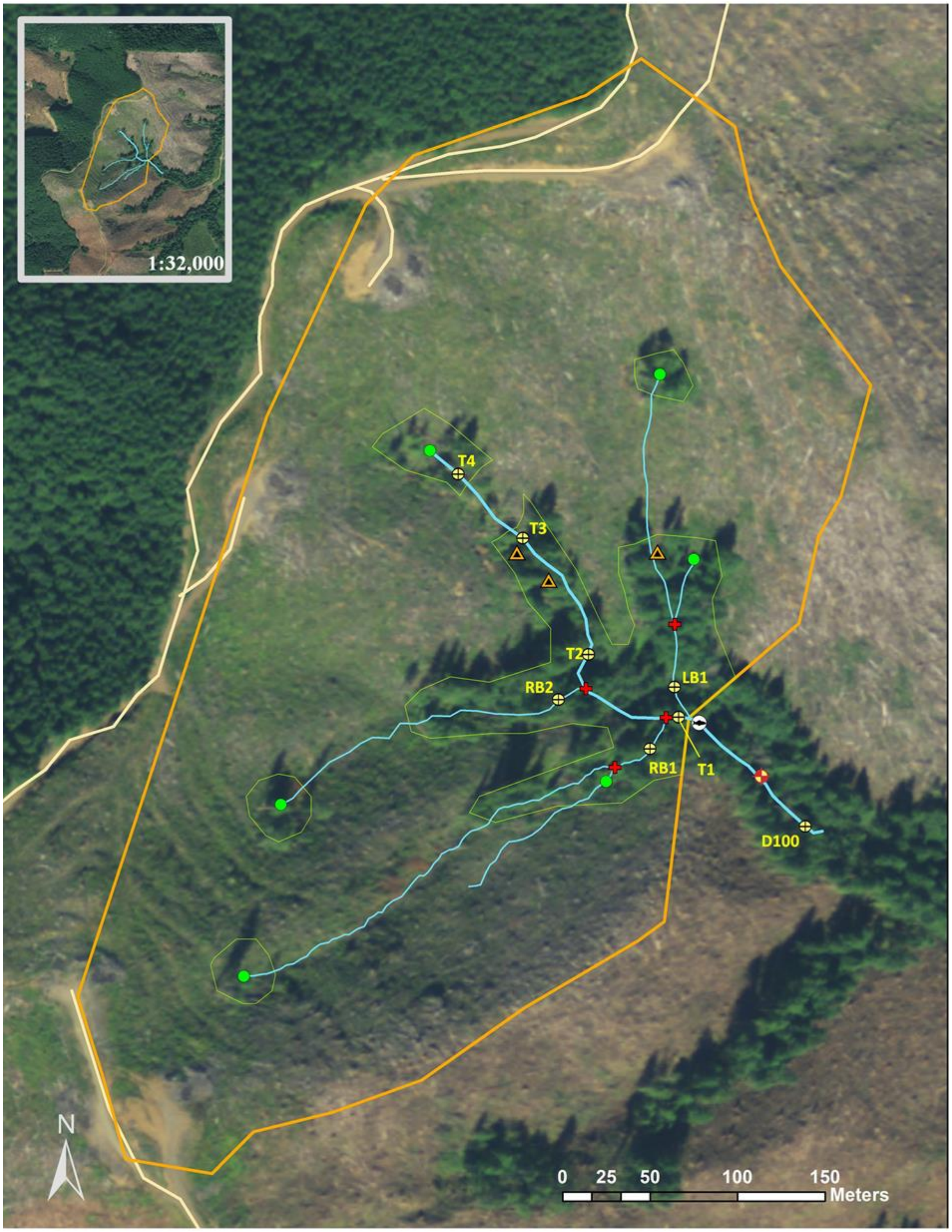
Appendix Figure 7-A-3. Monitoring locations for WIL1-REF (top), WIL1-FP (middle) and WIL1-100% (bottom) sites.



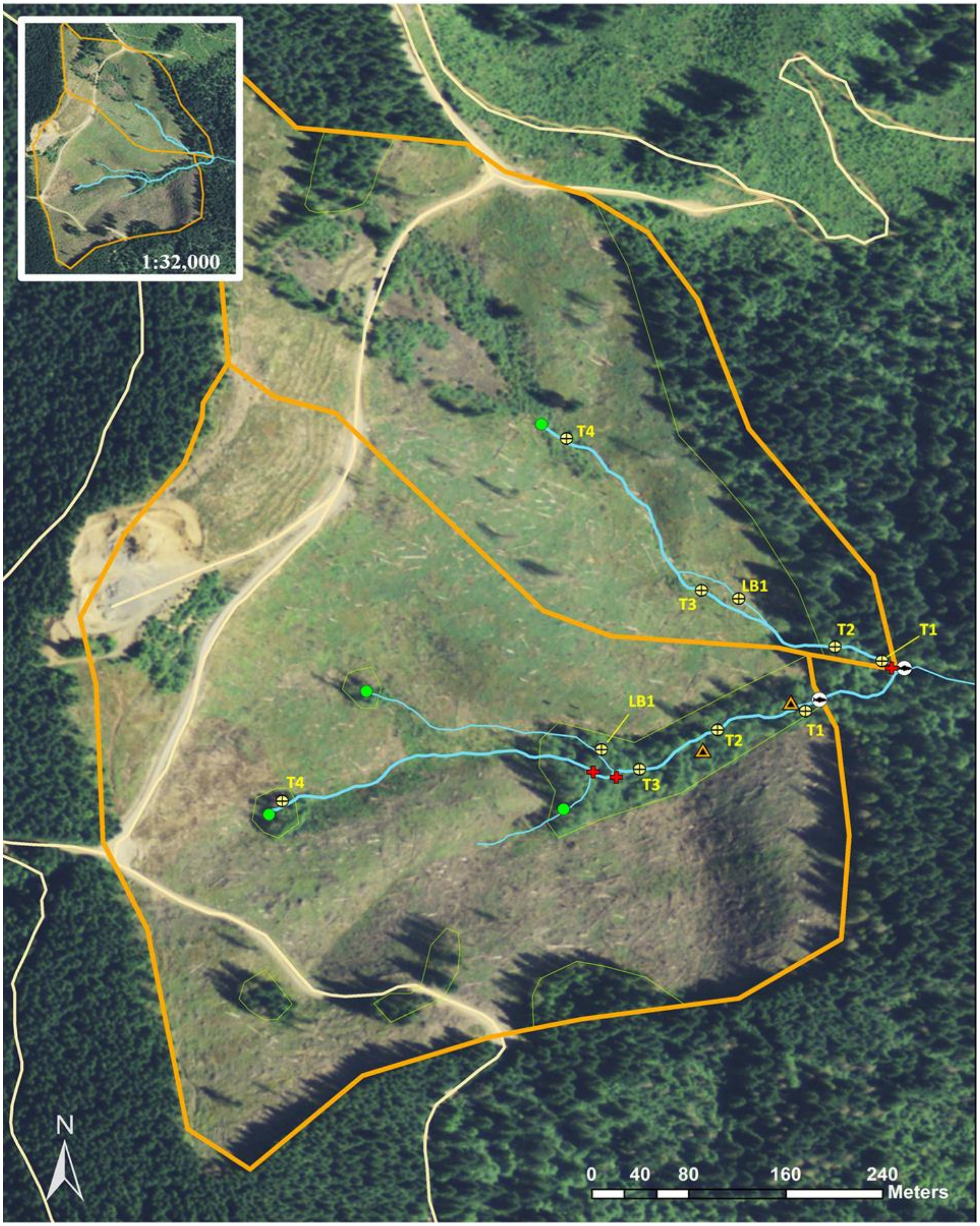
Appendix Figure 7-A-4. Monitoring locations in WIL2-100% site.



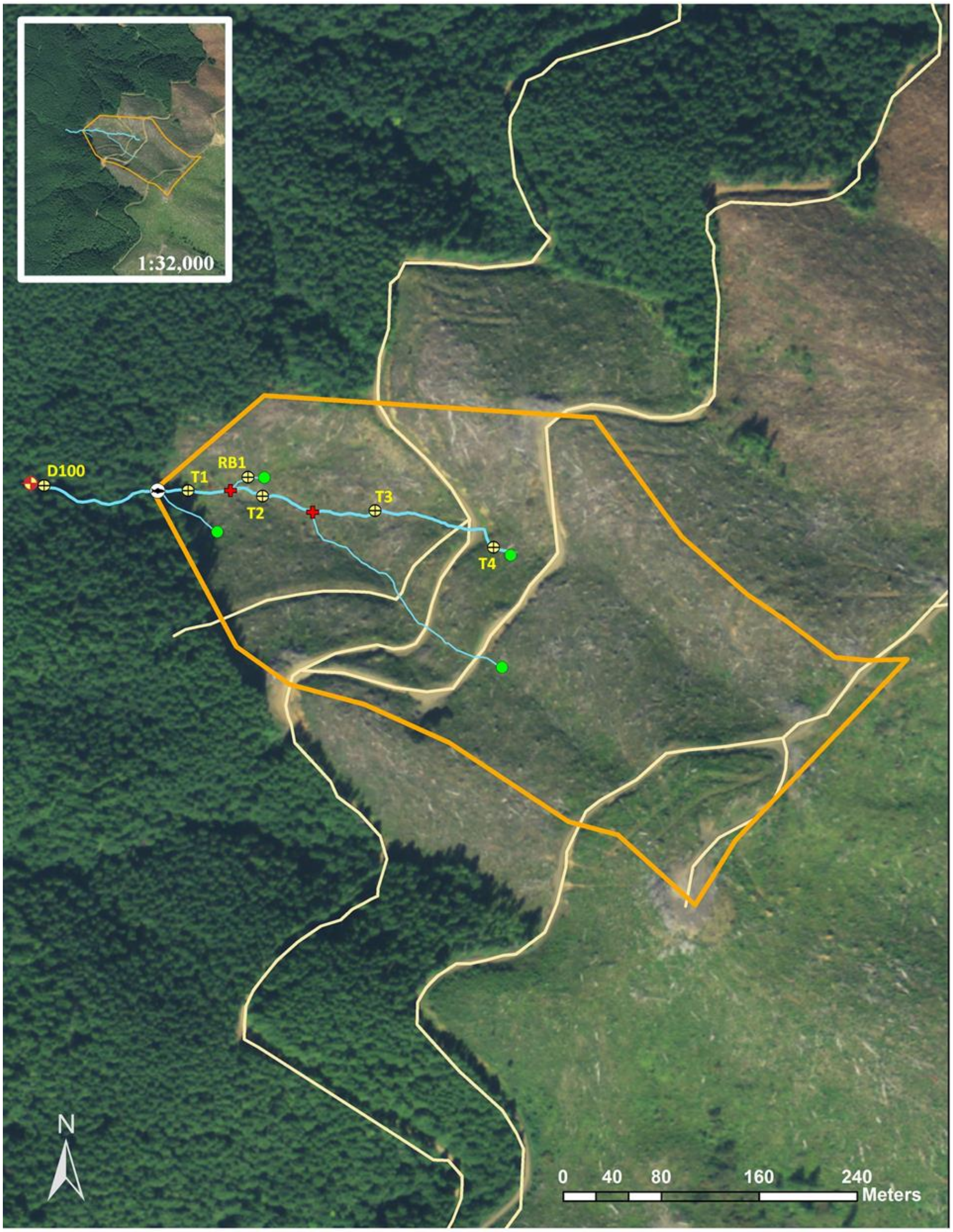
Appendix Figure 7-A-5. Monitoring locations in WIL3-100% site.



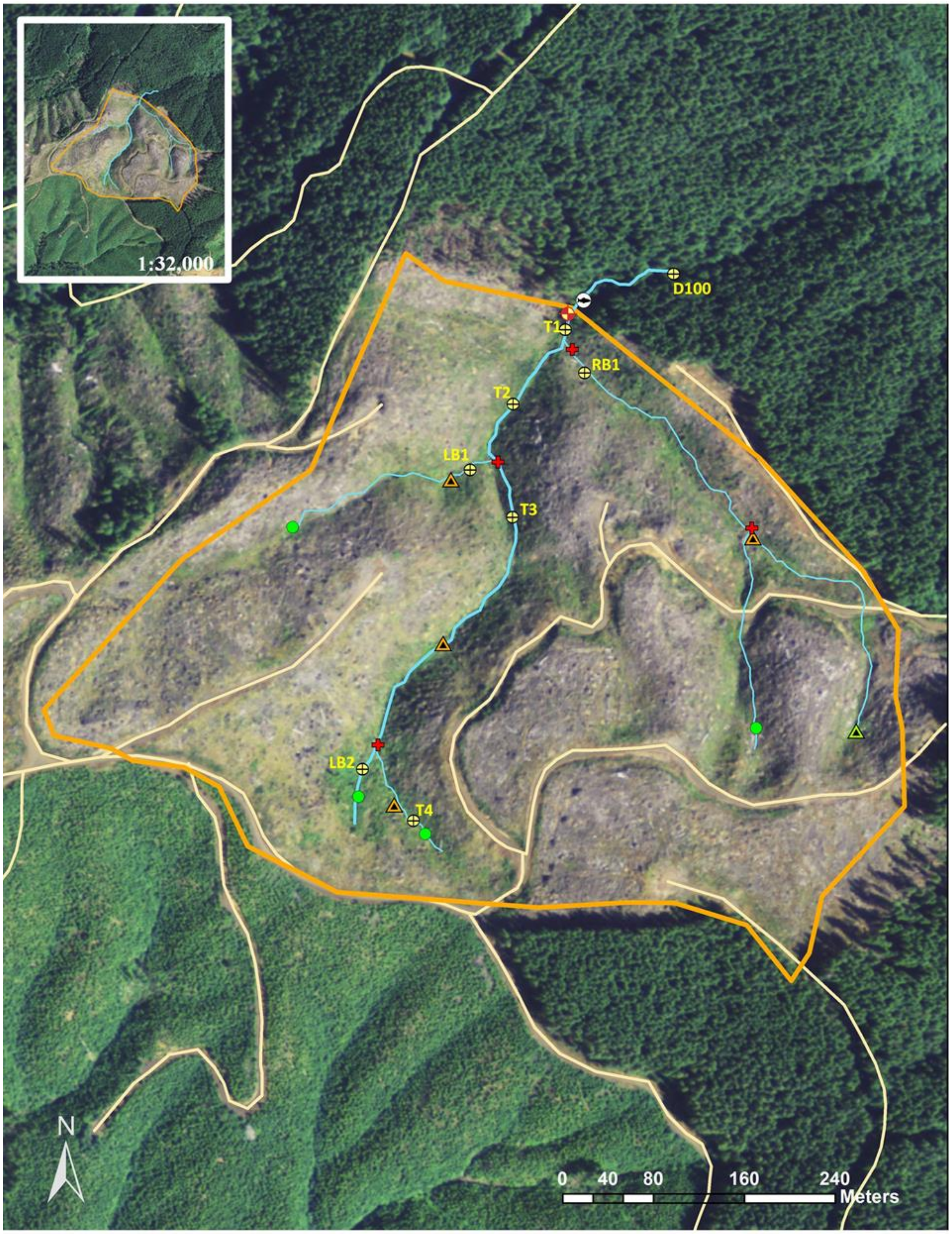
Appendix Figure 7-A-6. Monitoring locations in OLYM-FP site.



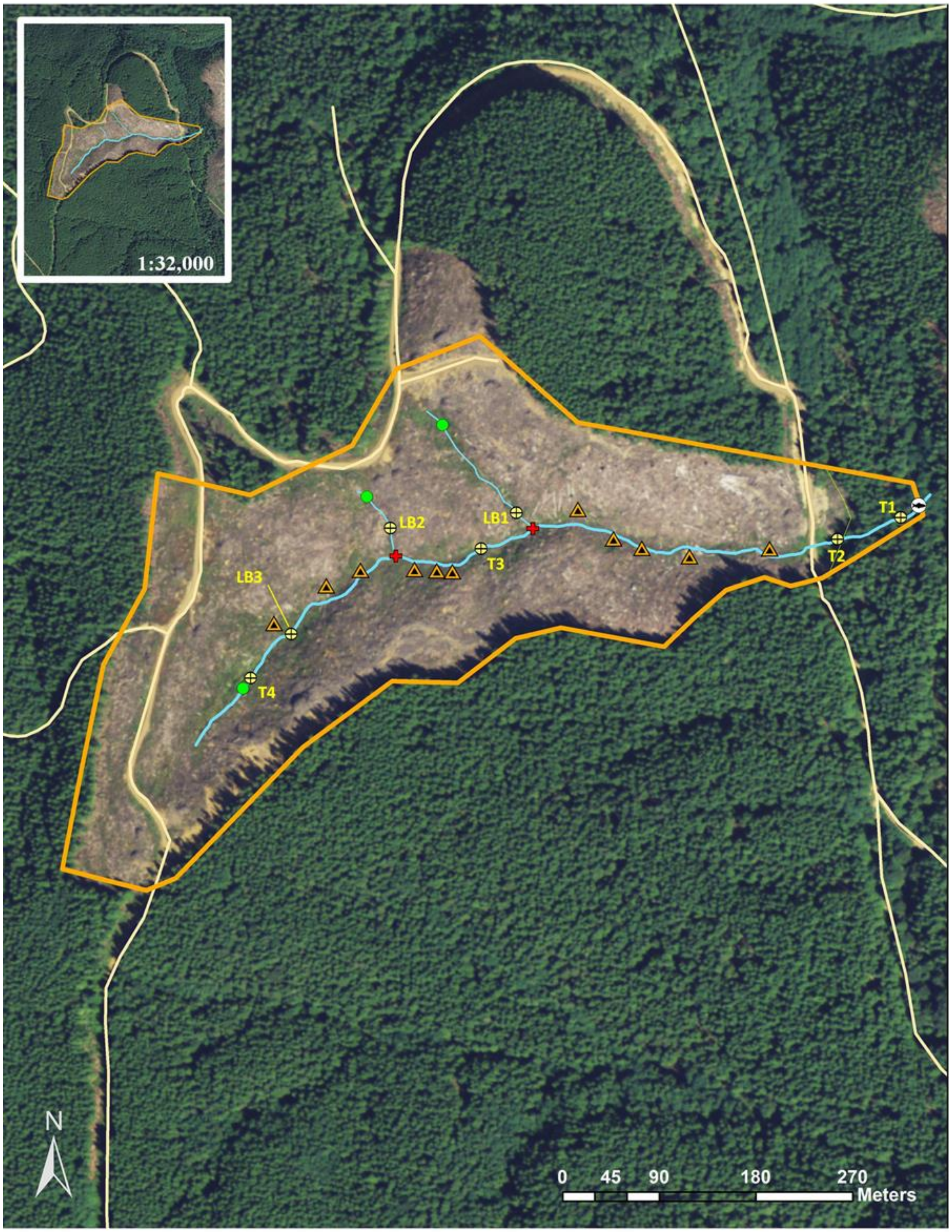
Appendix Figure 7-A-7. Monitoring locations in CASC-FP (lower) and CASC-0% (upper) sites.



Appendix Figure 7-A-8. Monitoring locations in OLYM-0% site.



Appendix Figure 7-A-9. Monitoring locations in WIL1-0% site.



Appendix Figure 7-A-10. Monitoring locations in WIL2-0% site.

APPENDIX 7-B. TABLES

Appendix Table 7-B-1. Mean canopy and topographic density (CTD) by site by year.

Block	Treatment	CTD (%)			
		2007-Pre	**2008-Pre**	**2009-Post 1**	**2010-Post 2**
CASC	REF	95	93	95	94
OLYM	REF	96	94	91	93
WIL1	REF	96	89	85	85
WIL2	REF1	97	93	93	92
WIL2	REF2	95	92		92
WIL3	REF	96	94	92	93
	Average	**96**	**92**	**91**	**92**
OLYM	100%	94	94	94	94
WIL1	100%	95	91	82	82
WIL2	100%	95	94	85	84
WIL3	100%	96	95	85	85
	Average	**95**	**94**	**86**	**86**
CASC	FP	96	95	72	68
OLYM	FP	97	97	89	85
WIL1	FP	97	91	61	61
	Average	**96**	**93**	**74**	**71**
CASC	0%	95	94	55	49
OLYM	0%	95	96	39	40
WIL1	0%	91		41	41
WIL2	0%	94	94	42	43
	Average	**94**	**95**	**44**	**43**

Appendix Table 7-B-2. Mean effective shade calculated for the seventh solar month (June 22–July 21) by site by year.

Block	Treatment	Mean Effective Shade (%)			
		2007-Pre	**2008-Pre**	**2009-Post 1**	**2010-Post 2**
CASC	REF	90	89	90	90
OLYM	REF	93	91	85	90
WIL1	REF	90	80	72	72
WIL2	REF1	94	87	87	85
WIL2	REF2	84	85		86
WIL3	REF	88	87	86	87
	Average	**90**	**87**	**84**	**85**
OLYM	100%	93	92	89	88
WIL1	100%	90	81	64	63
WIL2	100%	89	90	75	74
WIL3	100%	94	89	81	81
	Average	**92**	**88**	**77**	**77**
CASC	FP	90	89	56	51
OLYM	FP	90	90	74	70
WIL1	FP	92	84	44	35
	Average	**91**	**86**	**58**	**52**
CASC	0%	88	91	28	21
OLYM	0%	90	92	8	7
WIL1	0%	85		13	13
WIL2	0%	92	90	14	14
	Average	**89**	**91**	**16**	**14**

Appendix Table 7-B-3. Mean canopy closure measured at 1 m (CC-1m) by site by year.

Block	Treatment	Mean Canopy Closure (%)			
		2007-Pre	**2008-Pre**	**2009-Post 1**	**2010-Post 2**
CASC	REF	95	97	96	94
OLYM	REF	93	94	89	91
WIL1	REF	98	89	89	72
WIL2	REF1	97	95	97	98
WIL2	REF2	97	94		96
WIL3	REF	94	96	95	94
	Average	**96**	**94**	**93**	**91**
OLYM	100%	94	93	98	92
WIL1	100%	99	85	73	68
WIL2	100%	95	95	83	88
WIL3	100%	98	93	96	92
	Average	**97**	**92**	**88**	**85**
CASC	FP	92	98	68	58
OLYM	FP	98	90	94	86
WIL1	FP	97	95	55	57
	Average	**96**	**96**	**72**	**67**
CASC	0%	90	97	38	13
OLYM	0%	94	95	6	4
WIL1	0%	84		4	1
WIL2	0%	97	90	15	17
	Average	**91**	**94**	**16**	**9**

Appendix Table 7-B-4. Mean canopy closure measured at the water surface (CC-0m) by site by year.

Block	Treatment	Mean Canopy Closure (%)		
		2008-Pre	**2009-Post 1**	**2010-Post 2**
CASC	REF	97	97	94
OLYM	REF	94	98	97
WIL1	REF	90	94	89
WIL2	REF1	96	98	99
WIL2	REF2	97		96
WIL3	REF	98	97	99
	Average	**95**	**97**	**95**
OLYM	100%	94	99	96
WIL1	100%	86	82	86
WIL2	100%	97	87	90
WIL3	100%	94	97	93
	Average	**93**	**91**	**91**
CASC	FP	99	87	73
OLYM	FP	91	97	92
WIL1	FP	97	92	89
	Average	**98**	**92**	**85**
CASC	0%	98	56	31
OLYM	0%	96	55	54
WIL1	0%		43	72
WIL2	0%	91	52	49
	Average	**95**	**51**	**52**

Appendix Table 7-B-5. Results of the 12 pairwise *post hoc* comparisons including the estimated difference, 95% confidence interval (CI), and the associated probability values for CTD and effective shade. A negative estimate indicates that the parameter decreased in the second treatment listed, relative to the first, in that post-harvest year. Comparisons with P <0.05 are in **bold type**.

Comparison	Difference (°C)	Lower	Upper	P-value
Canopy and Topographic Density (CTD) %				
REF vs. 100%-Post 1	−4.9	−10.4	0.6	0.078
REF vs. FP-Post 1	**−19.9**	**−26.0**	**−13.8**	**<0.001**
REF vs. 0%-Post 1	**−47.1**	**−52.7**	**−41.6**	**<0.001**
REF vs. 100%-Post 2	**−5.7**	**−11.0**	**−0.4**	**0.036**
REF vs. FP-Post 2	**−23.2**	**−29.2**	**−17.3**	**<0.001**
REF vs. 0%-Post 2	**−48.5**	**−53.9**	**−43.1**	**<0.001**
100% vs. FP-Post 1	**−15.0**	**−21.4**	**−8.6**	**<0.001**
100% vs. FP-Post 2	**−17.5**	**−23.9**	**−11.1**	**<0.001**
100% vs. 0%-Post 1	**−42.2**	**−48.1**	**−36.3**	**<0.001**
100% vs. 0%-Post 2	**−42.8**	**−48.7**	**−36.9**	**<0.001**
FP vs. 0%-Post 1	**−27.2**	**−33.7**	**−20.7**	**<0.001**
FP vs. 0%-Post 2	**−25.3**	**−31.7**	**−18.8**	**<0.001**
Effective Shade %				
REF vs. 100%-Post 1	**−8.2**	**−16.4**	**−0.1**	**0.047**
REF vs. FP-Post 1	**−28.6**	**−37.7**	**−19.5**	**<0.001**
REF vs. 0%-Post 1	**−69.2**	**−77.5**	**−61.0**	**<0.001**
REF vs. 100%-Post 2	**−9.9**	**−17.8**	**−2.1**	**0.015**
REF vs. FP-Post 2	**−36.0**	**−44.9**	**−27.2**	**<0.001**
REF vs. 0%-Post 2	**−72.4**	**−80.4**	**−64.4**	**<0.001**
100% vs. FP-Post 1	**−20.4**	**−29.9**	**−10.9**	**<0.001**
100% vs. FP-Post 2	**−26.1**	**−35.6**	**−16.6**	**<0.001**
100% vs. 0%-Post 1	**−61.0**	**−69.7**	**−52.2**	**<0.001**
100% vs. 0%-Post 2	**−62.5**	**−71.2**	**−53.7**	**<0.001**
FP vs. 0%-Post 1	**−40.6**	**−50.2**	**−31.0**	**<0.001**
FP vs. 0%-Post 2	**−36.4**	**−46.0**	**−26.7**	**<0.001**

Appendix Table 7-B-6. Results of the pairwise comparisons including the estimated difference (%) and 95% confidence intervals for CC-1m and CC-0m. A negative difference indicates that canopy closure decreased in the second treatment listed, relative to the first, in that post-harvest year. Comparisons with P <0.05 are in **bold type**.

Comparison	Change	95% CI Lower	95% CI Upper	P-value
Canopy Closure-1m				
REF vs. 100%-Post 1	−4.9	−15.0	5.3	0.338
REF vs. FP-Post 1	**−24.5**	**−35.8**	**−13.2**	**<0.001**
REF vs. 0%-Post 1	**−74.2**	**−84.5**	**−63.8**	**<0.001**
REF vs. 100%-Post 2	−4.9	−14.7	5.0	0.326
REF vs. FP-Post 2	**−27.2**	**−38.3**	**−16.1**	**<0.001**
REF vs. 0%-Post 2	**−78.4**	**−88.4**	**−68.4**	**<0.001**
100% vs. FP-Post 1	**−19.6**	**−31.5**	**−7.7**	**0.002**
100% vs. FP-Post 2	**−22.3**	**−34.2**	**−10.4**	**<0.001**
100% vs. 0%-Post 1	**−69.3**	**−80.2**	**−58.3**	**<0.001**
100% vs. 0%-Post 2	**−73.5**	**−84.5**	**−62.6**	**<0.001**
FP vs. 0%-Post 1	**−49.6**	**−61.7**	**−37.6**	**<0.001**
FP vs. 0%-Post 2	**−51.2**	**−63.2**	**−39.2**	**<0.001**
Canopy Closure-0m				
REF vs. 100%-Post 1	−2.9	−15.0	9.2	0.632
REF vs. FP-Post 1	−8.5	−22.9	5.9	0.240
REF vs. 0%-Post 1	**−46.7**	**−59.5**	**−34.0**	**<0.001**
REF vs. 100%-Post 2	−1.6	−13.5	10.2	0.779
REF vs. FP-Post 2	**−14.6**	**−28.8**	**−0.4**	**0.044**
REF vs. 0%-Post 2	**−45.4**	**−57.9**	**−32.9**	**<0.001**
100% vs. FP-Post 1	−5.6	−20.8	9.5	0.456
100% vs. FP-Post 2	−12.9	−28.1	2.2	0.092
100% vs. 0%-Post 1	**−43.9**	**−57.4**	**−30.3**	**<0.001**
100% vs. 0%-Post 2	**−43.8**	**−57.4**	**−30.2**	**<0.001**
FP vs. 0%-Post 1	**−38.2**	**−53.9**	**−22.6**	**<0.001**
FP vs. 0%-Post 2	**−30.9**	**−46.5**	**−15.2**	**<0.001**

Appendix Table 7-B-7. Estimated change (°C), 95% confidence intervals, and the associated probability values for the seven-day average temperature response (7DTR) calculated for July–August maximum water temperature at the fish-bearing/non-fish-bearing (F/N) break and buffer treatment locations. A positive estimate indicates that stream temperature was higher in the second period/treatment listed, relative to the first. P-values were not adjusted for multiple comparisons. Comparisons with P <0.05 are in **bold type**.

Comparison	Change	95% CI		P-value
		Lower	Upper	
F/N Break				
Pre vs. Post 1-100%	**0.9**	**0.1**	**1.7**	**0.035**
Pre vs. Post 2-100%	0.6	−0.3	1.4	0.176
Pre vs. Post 1-FP	**1.4**	**0.4**	**2.3**	**0.005**
Pre vs. Post 2-FP	**1.0**	**0.0**	**1.9**	**0.044**
Pre vs. Post 1-0%	**3.1**	**2.3**	**3.9**	**<0.001**
Pre vs. Post 2-0%	**2.7**	**1.9**	**3.5**	**<0.001**
100% vs. FP-Post 1	0.5	−0.8	1.7	0.425
100% vs. 0%-Post 1	**2.2**	**1.1**	**3.4**	**<0.001**
100% vs. FP-Post 2	0.4	−0.8	1.7	0.508
100% vs. 0%-Post 2	**2.1**	**1.0**	**3.3**	**0.001**
FP vs. 0%-Post 1	**1.7**	**0.5**	**3.0**	**0.008**
FP vs. 0%-Post 2	**1.7**	**0.5**	**3.0**	**0.009**
Buffer Treatment				
Pre vs. Post 1-100%	**1.2**	**0.4**	**2.0**	**0.007**
Pre vs. Post 2-100%	**1.2**	**0.4**	**2.1**	**0.006**
Pre vs. Post 1-FP	**1.4**	**0.4**	**2.3**	**0.006**
Pre vs. Post 2-FP	**1.0**	**0.0**	**1.9**	**0.049**
Pre vs. Post 1-0%	**3.4**	**2.5**	**4.4**	**<0.001**
Pre vs. Post 2-0%	**3.0**	**2.1**	**3.8**	**<0.001**
100% vs. FP-Post 1	0.2	−1.1	1.5	0.767
100% vs. 0%-Post 1	**2.3**	**1.0**	**3.5**	**0.001**
100% vs. FP-Post 2	−0.3	−1.5	1.0	0.680
100% vs. 0%-Post 2	**1.7**	**0.5**	**2.9**	**0.006**
FP vs. 0%-Post 1	**2.1**	**0.7**	**3.4**	**0.005**
FP vs. 0%-Post 2	**2.0**	**0.7**	**3.3**	**0.003**

Appendix Table 7-B-8. Estimated change (°C), 95% confidence intervals, and the associated probability values for the seven-day average temperature response (7DTR) calculated for July–August minimum daily water temperature and diel range at the buffer treatment locations. A positive estimate indicates that stream temperature was higher in the second period/treatment listed, relative to the first. P-values were not adjusted for multiple comparisons. Comparisons with P <0.05 are in **bold type**.

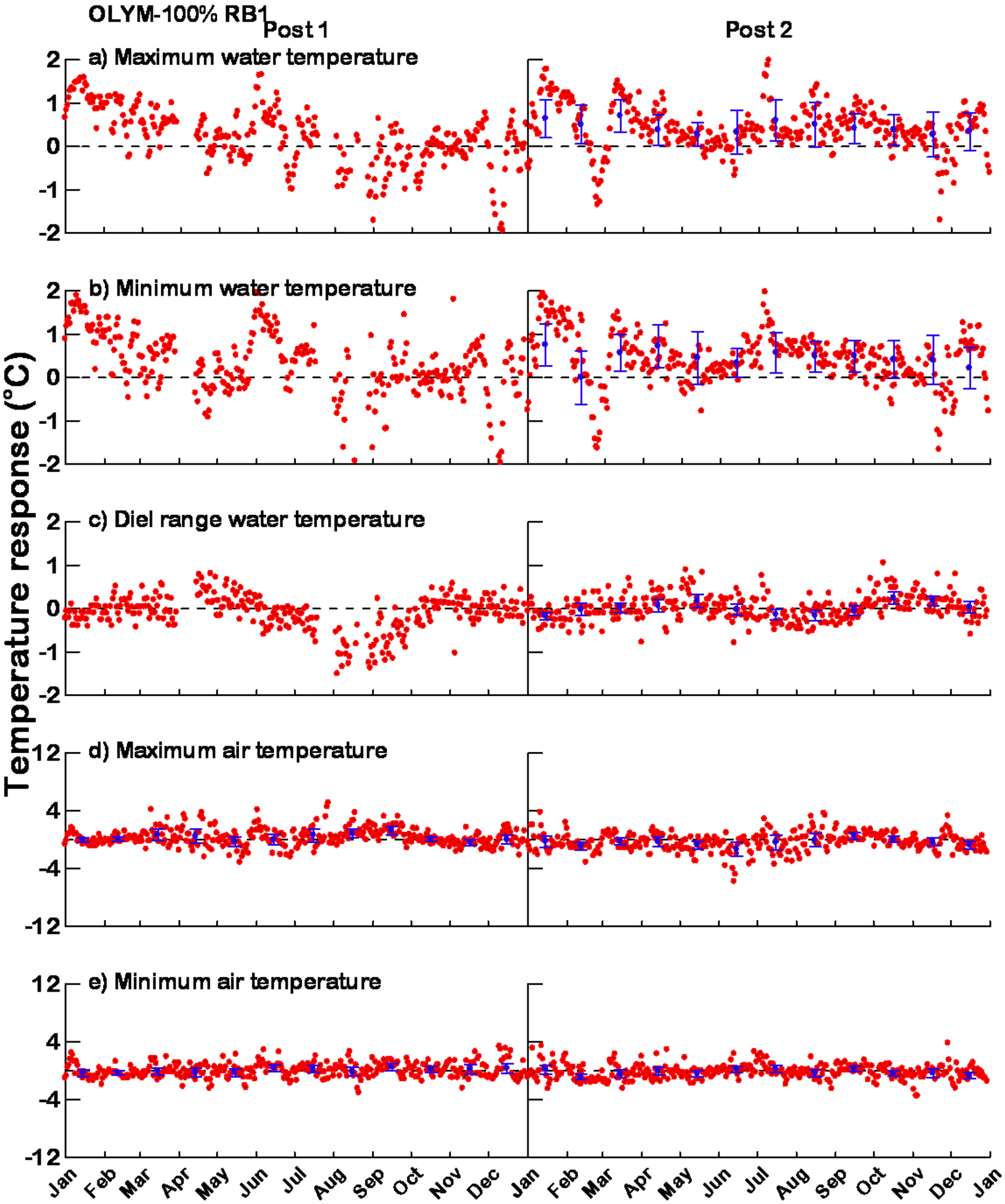
Comparison	Change	95% CI		P-value
		Lower	Upper	
Minimum Water Temperature				
Pre vs. Post 1-100%	**0.9**	**0.5**	**1.3**	**<0.001**
Pre vs. Post 2-100%	**0.6**	**0.2**	**1.0**	**0.003**
Pre vs. Post 1-FP	**0.6**	**0.1**	**1.0**	**0.013**
Pre vs. Post 2-FP	0.2	−0.3	0.6	0.435
Pre vs. Post 1-0%	**0.9**	**0.4**	**1.3**	**0.001**
Pre vs. Post 2-0%	**0.9**	**0.5**	**1.3**	**<0.001**
100% vs. FP-Post 1	−0.3	−0.9	0.3	0.335
100% vs. 0%-Post 1	0.0	−0.6	0.6	0.966
100% vs. FP-Post 2	−0.5	−1.1	0.1	0.132
100% vs. 0%-Post 2	0.3	−0.3	0.8	0.332
FP vs. 0%-Post 1	0.3	−0.4	0.9	0.392
FP vs. 0%-Post 2	**0.7**	**0.1**	**1.3**	**0.020**
Diel Range Water Temperature				
Pre vs. Post 1-100%	**0.9**	**0.1**	**1.7**	**0.031**
Pre vs. Post 2-100%	0.7	−0.1	1.4	0.102
Pre vs. Post 1-FP	**1.2**	**0.3**	**2.1**	**0.009**
Pre vs. Post 2-FP	0.9	0.0	1.7	0.057
Pre vs. Post 1-0%	**3.1**	**2.2**	**4.0**	**<0.001**
Pre vs. Post 2-0%	**2.8**	**2.1**	**3.6**	**<0.001**
100% vs. FP-Post 1	0.3	−0.8	1.5	0.566
100% vs. 0%-Post 1	**2.3**	**1.1**	**3.5**	**0.001**
100% vs. FP-Post 2	0.2	−1.0	1.4	0.718
100% vs. 0%-Post 2	**2.2**	**1.1**	**3.3**	**0.000**
FP vs. 0%-Post 1	**1.9**	**0.7**	**3.2**	**0.004**
FP vs. 0%-Post 2	**2.0**	**0.8**	**3.2**	**0.002**

Appendix Table 7-B-9. Estimated change (°C), 95% confidence intervals, and the associated probability values for the seven-day average temperature response (7DTR) calculated for July–August maximum and minimum daily air temperature 7DTR at the buffer treatment locations. A positive estimate indicates that stream temperature was higher in the second period/treatment listed, in that post-harvest year. P-values were not adjusted for multiple comparisons. Comparisons with P <0.05 are in **bold type**.

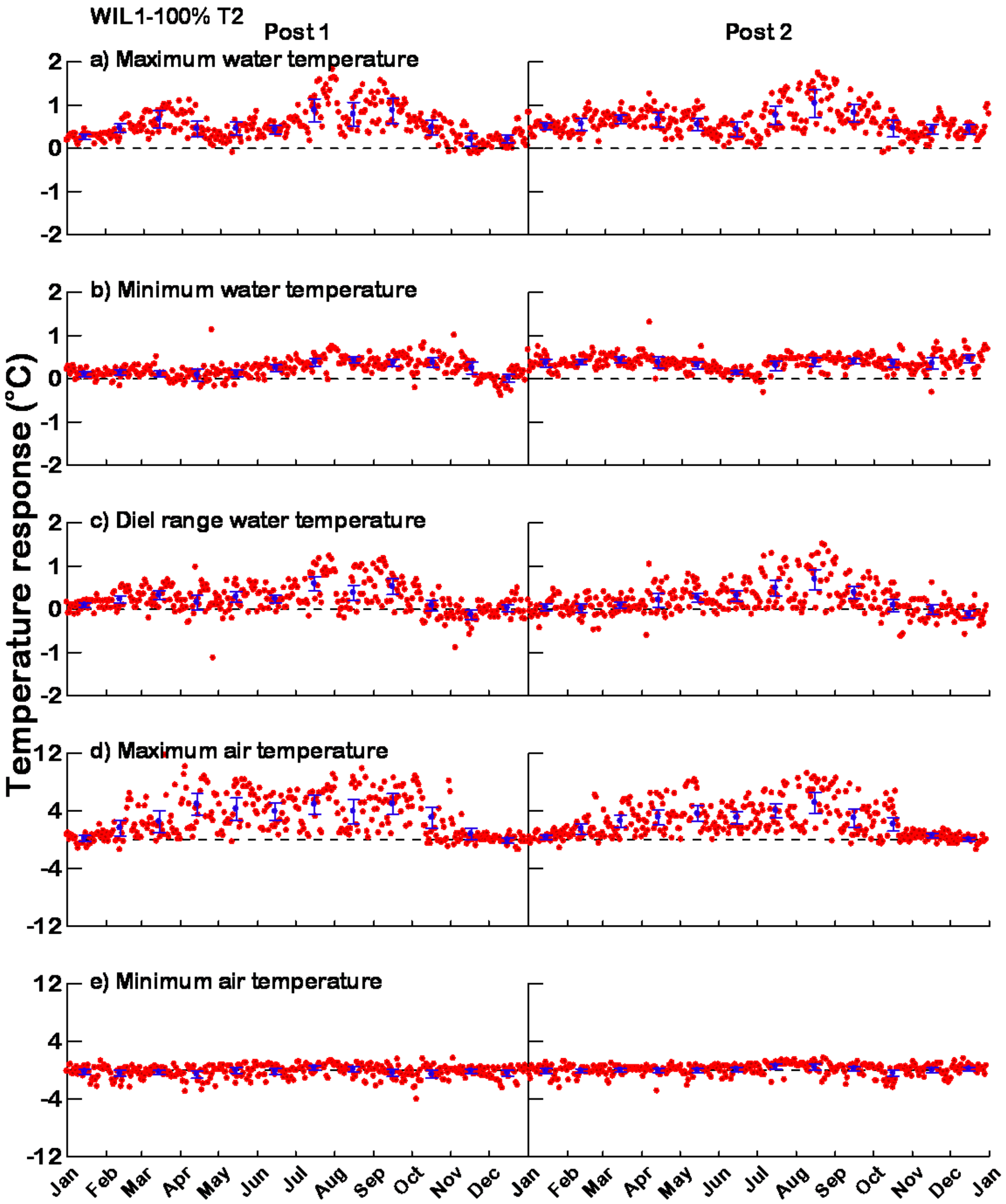
Comparison	Change	95% CI		P-value
		Lower	Upper	
Maximum Air Temperature				
Pre vs. Post 1-100%	**4.2**	**2.5**	**6.0**	**<0.001**
Pre vs. Post 2-100%	**3.7**	**2.0**	**5.5**	**0.000**
Pre vs. Post 1-FP	1.5	−0.5	3.5	0.143
Pre vs. Post 2-FP	**2.5**	**0.5**	**4.6**	**0.017**
Pre vs. Post 1-0%	**5.0**	**3.2**	**6.8**	**<0.001**
Pre vs. Post 2-0%	**5.7**	**3.9**	**7.4**	**<0.001**
100% vs. FP-Post 1	**−2.8**	**−5.4**	**−0.1**	**0.044**
100% vs. 0%-Post 1	0.8	−1.7	3.2	0.539
100% vs. FP-Post 2	−1.2	−3.9	1.5	0.372
100% vs. 0%-Post 2	2.0	−0.5	4.4	0.118
FP vs. 0%-Post 1	**3.5**	**0.8**	**6.2**	**0.013**
FP vs. 0%-Post 2	**3.1**	**0.4**	**5.8**	**0.024**
Minimum Air Temperature				
Pre vs. Post 1-100%	**0.4**	**0.02**	**0.83**	**0.040**
Pre vs. Post 2-100%	**0.5**	**0.1**	**0.9**	**0.025**
Pre vs. Post 1-FP	0.1	−0.4	0.5	0.761
Pre vs. Post 2-FP	0.06	−0.4	0.5	0.793
Pre vs. Post 1-0%	−0.2	−0.6	0.3	0.451
Pre vs. Post 2-0%	−0.1	−0.5	0.3	0.613
100% vs. FP-Post 1	−0.4	−1.0	0.3	0.255
100% vs. 0%-Post 1	**−0.56**	**−1.2**	**0.0**	**0.049**
100% vs. FP-Post 2	−0.4	−1.0	0.2	0.192
100% vs. 0%-Post 2	−0.6	−1.2	0.0	0.052
FP vs. 0%-Post 1	−0.2	−0.9	0.4	0.469
FP vs. 0%-Post 2	−0.2	−0.8	0.5	0.597

APPENDIX 7-C. AIR AND WATER TEMPERATURE RESPONSE

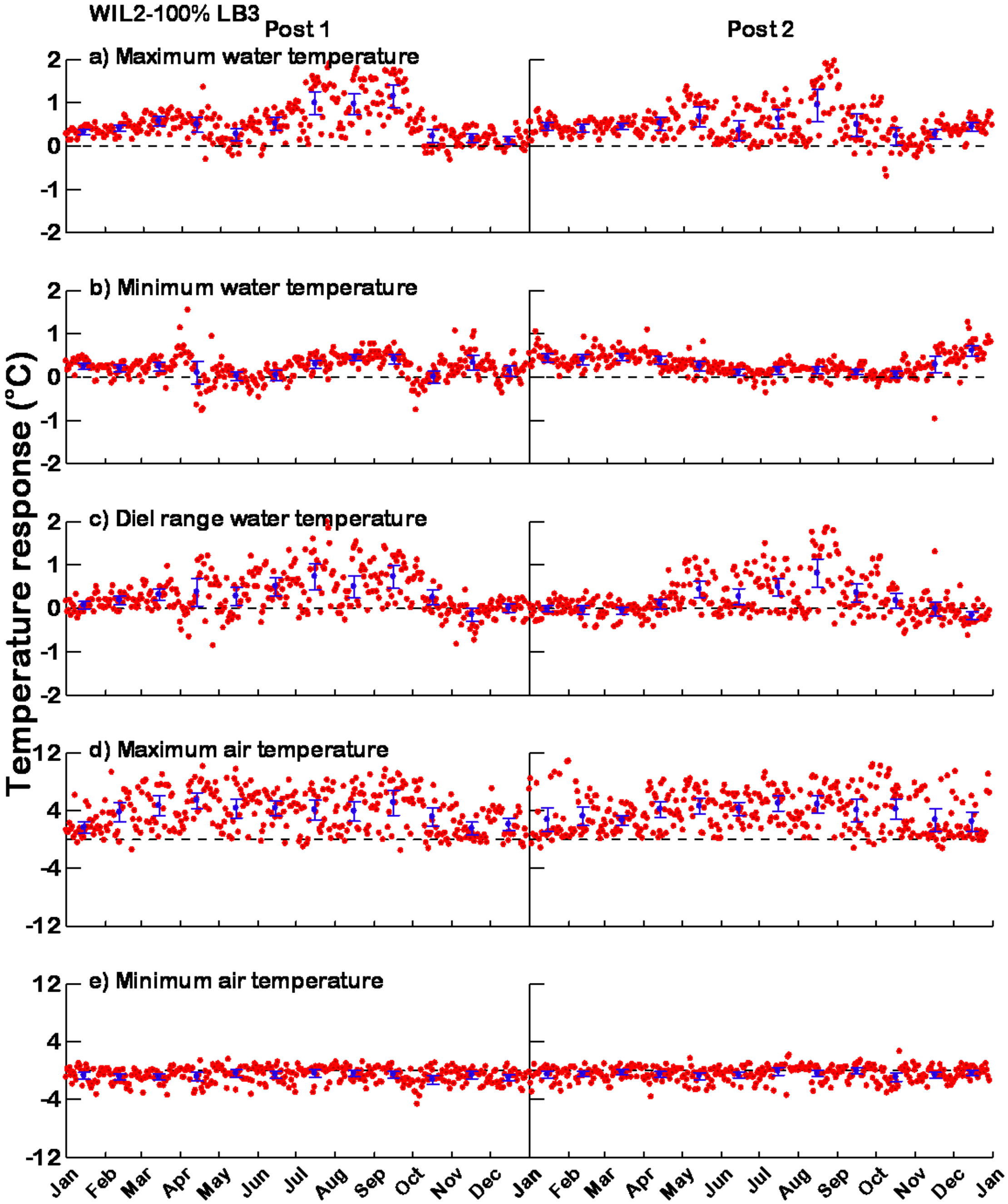
For each buffer treatment site, post-harvest daily TR for maximum, minimum, and diel range in water temperature and for daily maximum and minimum air temperatures was plotted vs. time. The corresponding MMTR and 95% confidence intervals (CI) are plotted by month. MMTR was not calculated if the annual data record was less than 300 continuous days. MMTR was not plotted for individual months with less than 20 records.



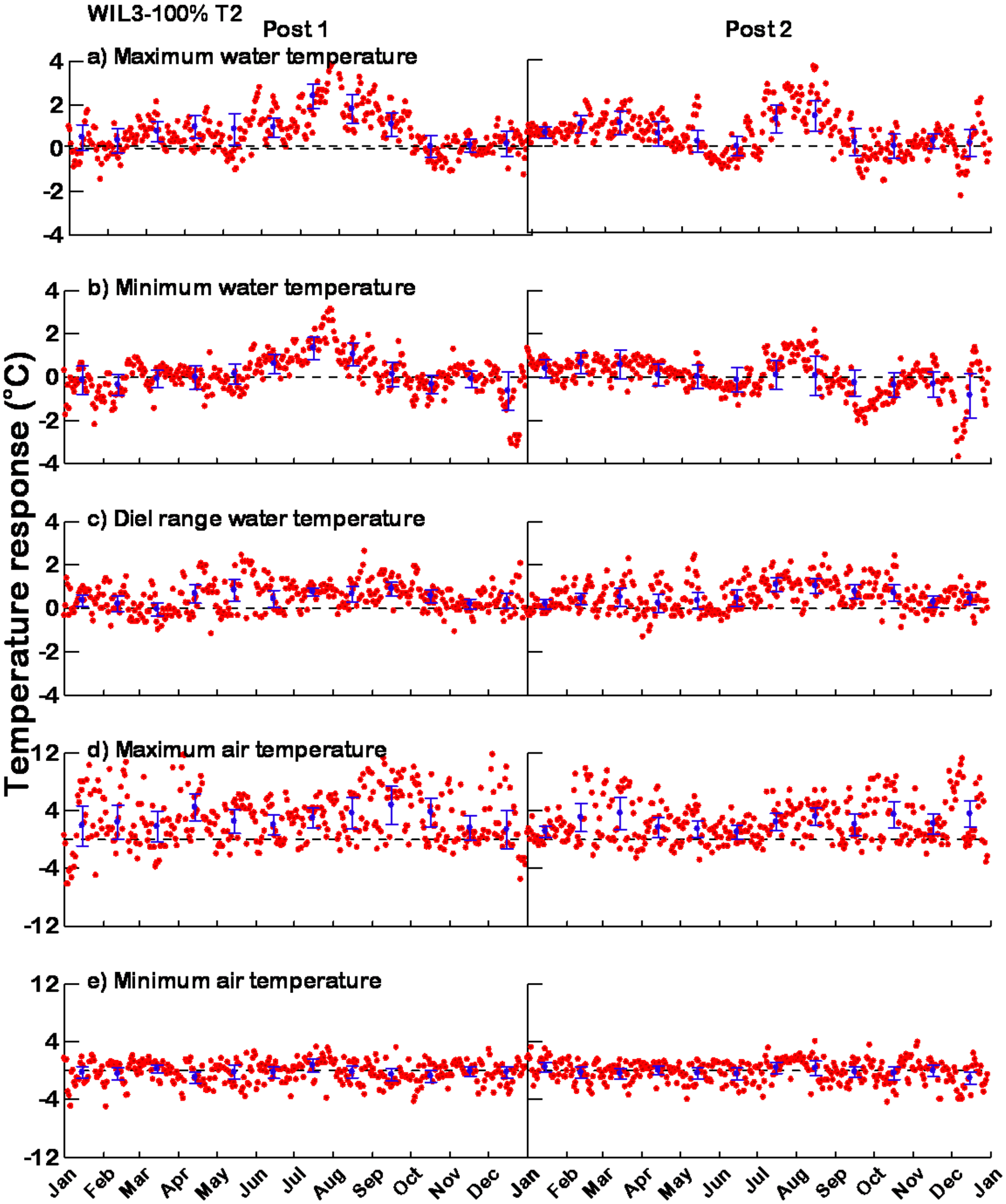
Appendix Figure 7-C-1. Water temperature and air temperature response at site OLYM-100%, location RB1. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).



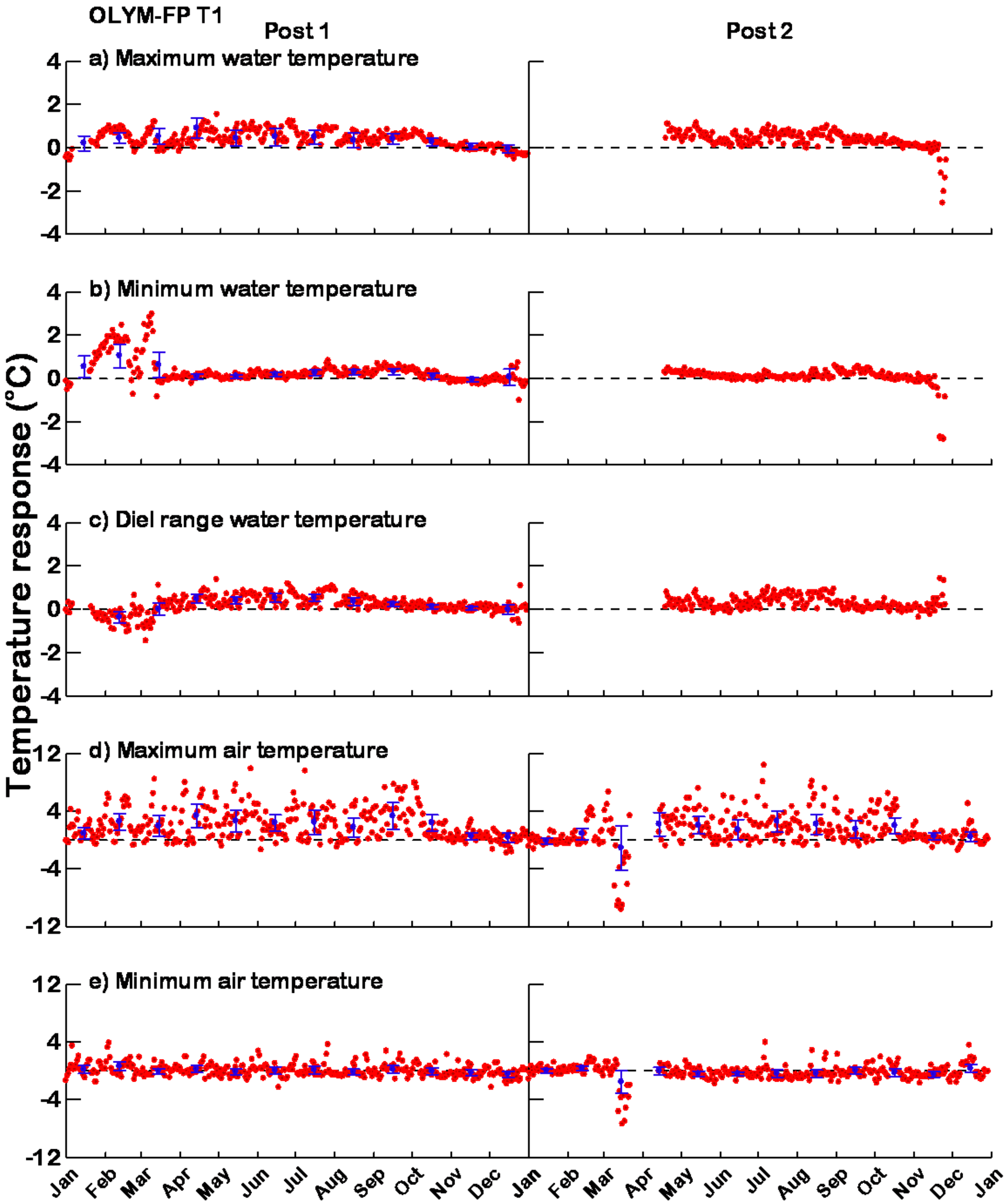
Appendix Figure 7-C-2. Water temperature and air temperature response at site WIL1-100%, location T2. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).



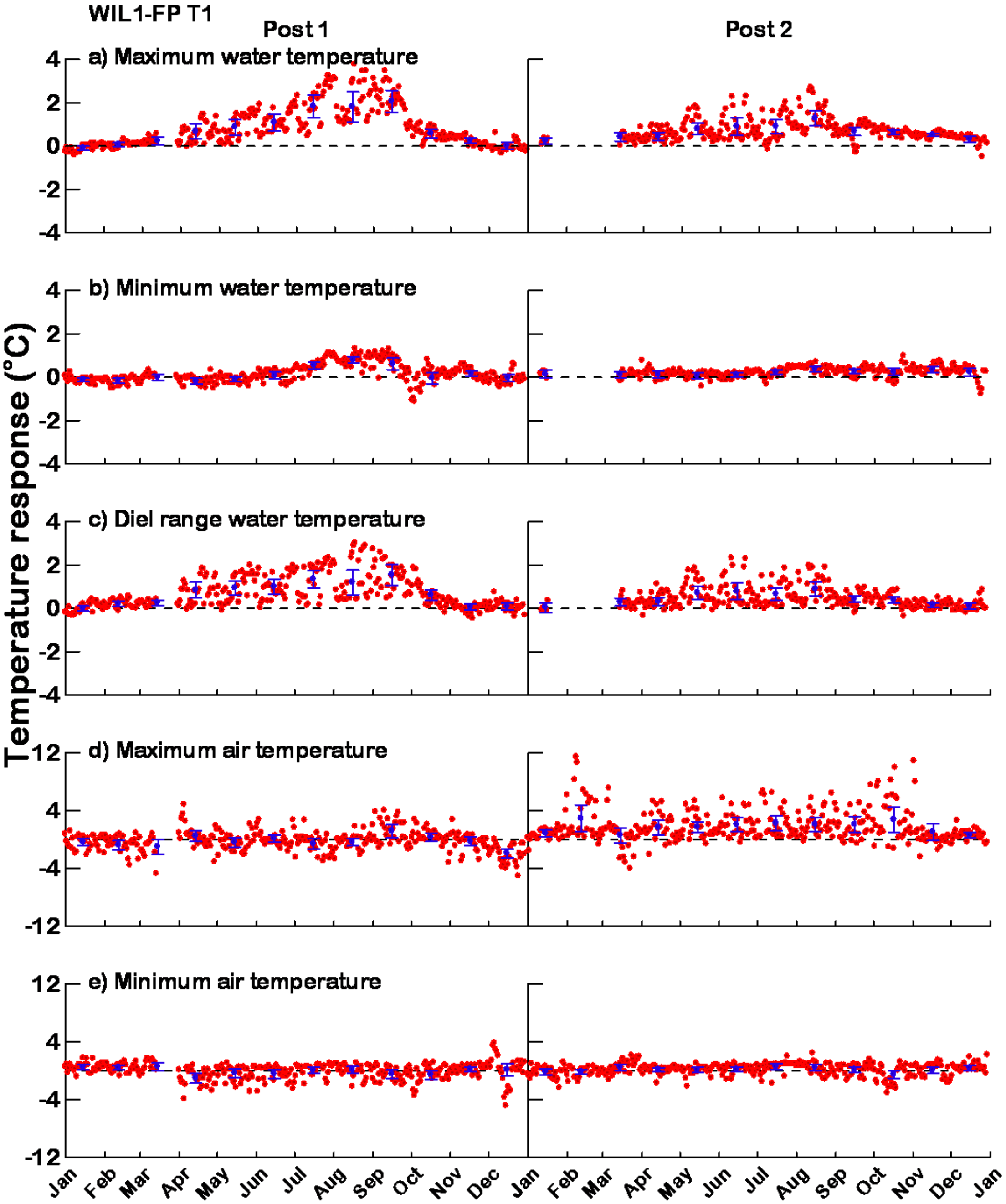
Appendix Figure 7-C-3. Water temperature and air temperature response at site WIL2-100%, location LB3. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).



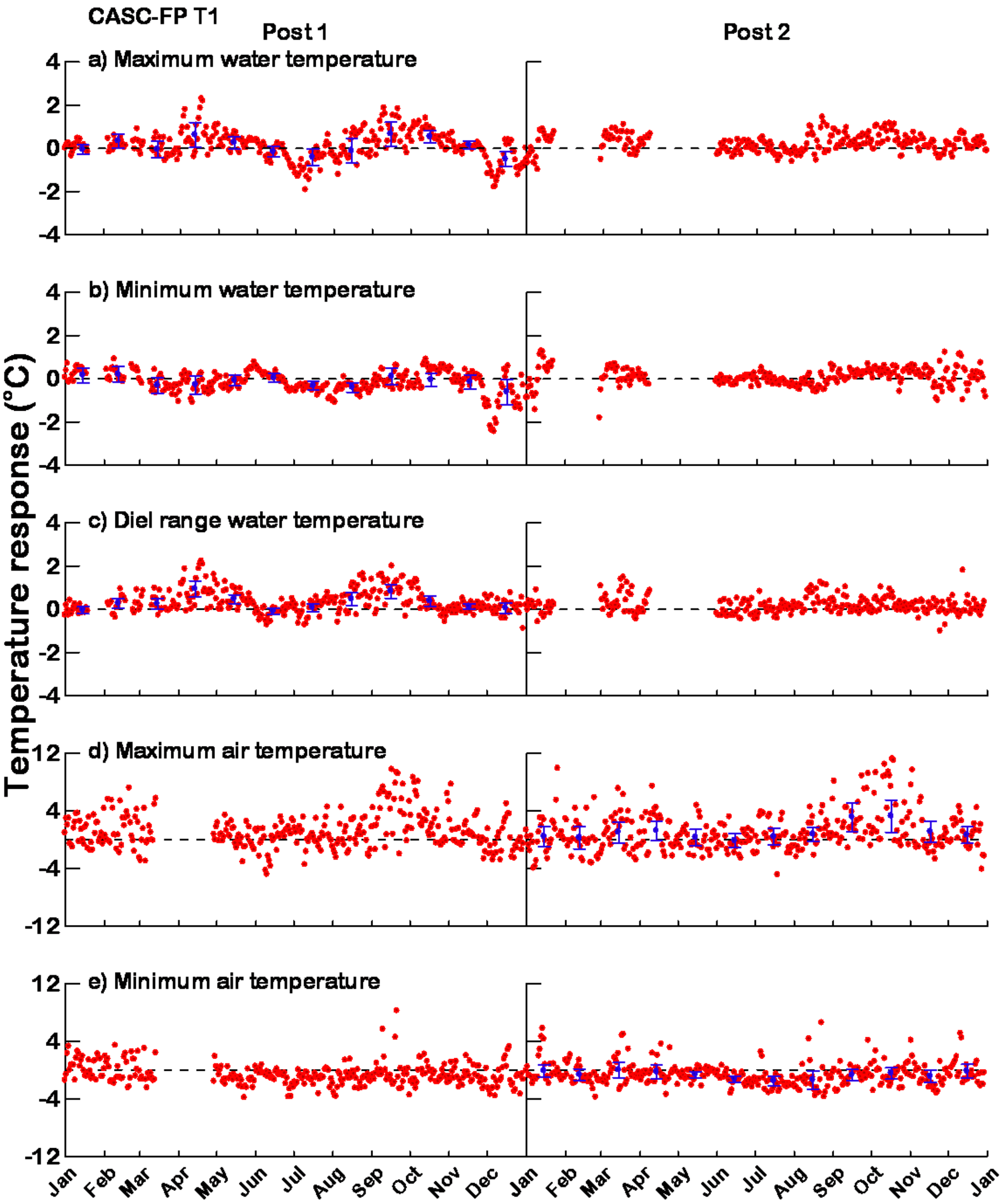
Appendix Figure 7-C-4. Water temperature and air temperature response at site WIL3-100%, location T2. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).



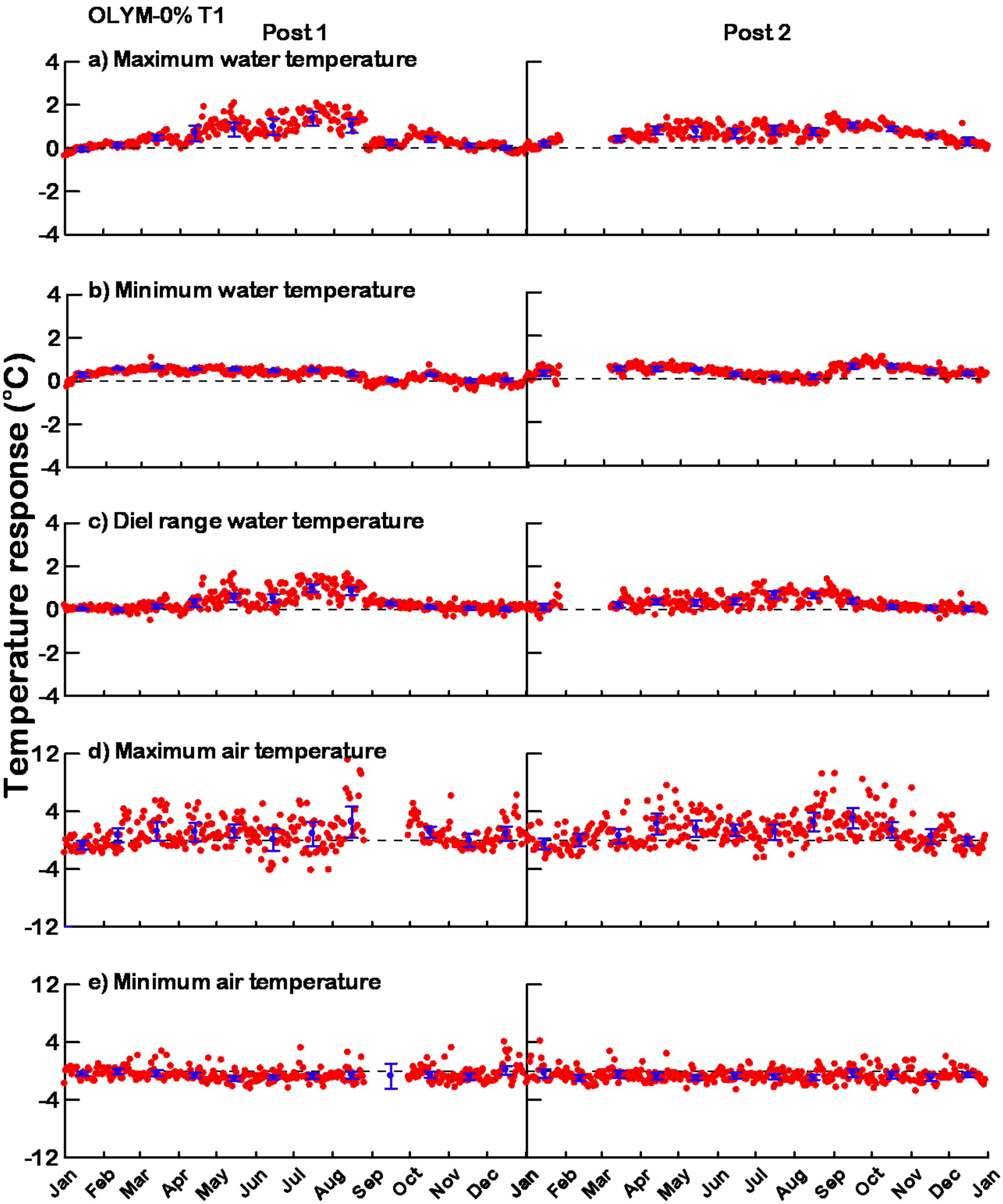
Appendix Figure 7-C-5. Water temperature and air temperature response at site OLYM-FP, location T1. Red dots are the daily temperature response (TR), Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).



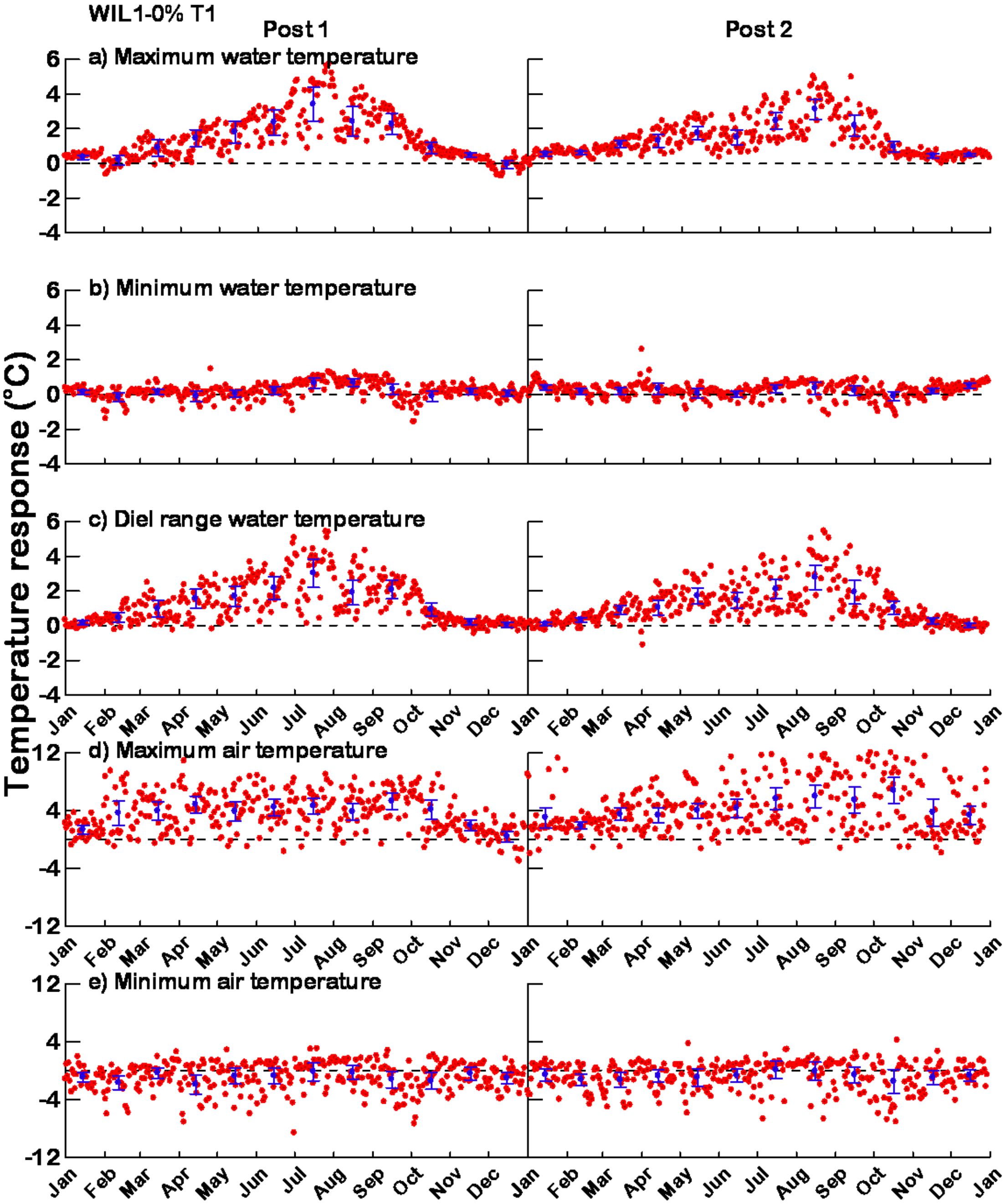
Appendix Figure 7-C-6. Water temperature and air temperature response at site WIL1-FP, location T1. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).



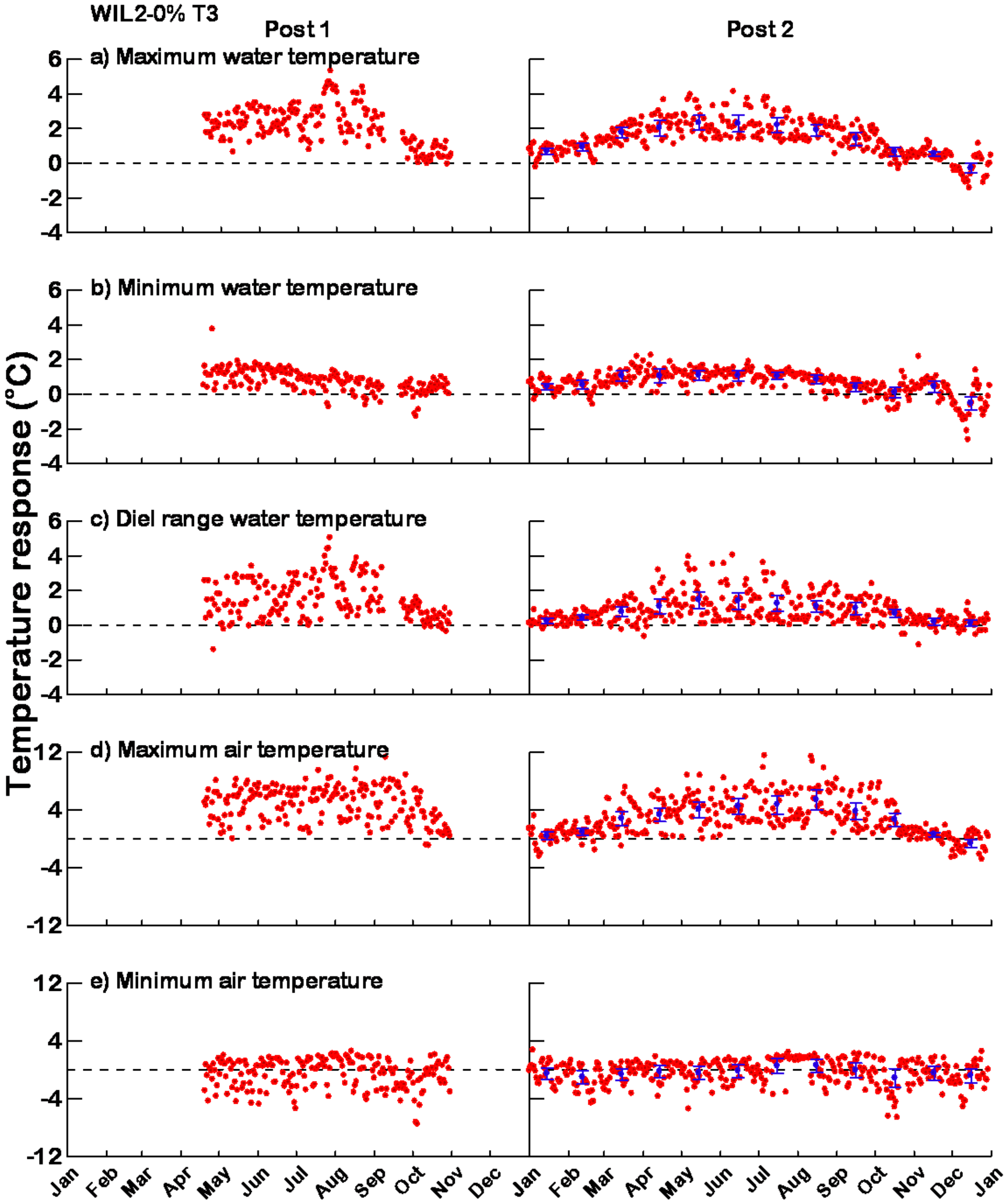
Appendix Figure 7-C-7. Water temperature and air temperature response at site CASC-FP, location T1. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).



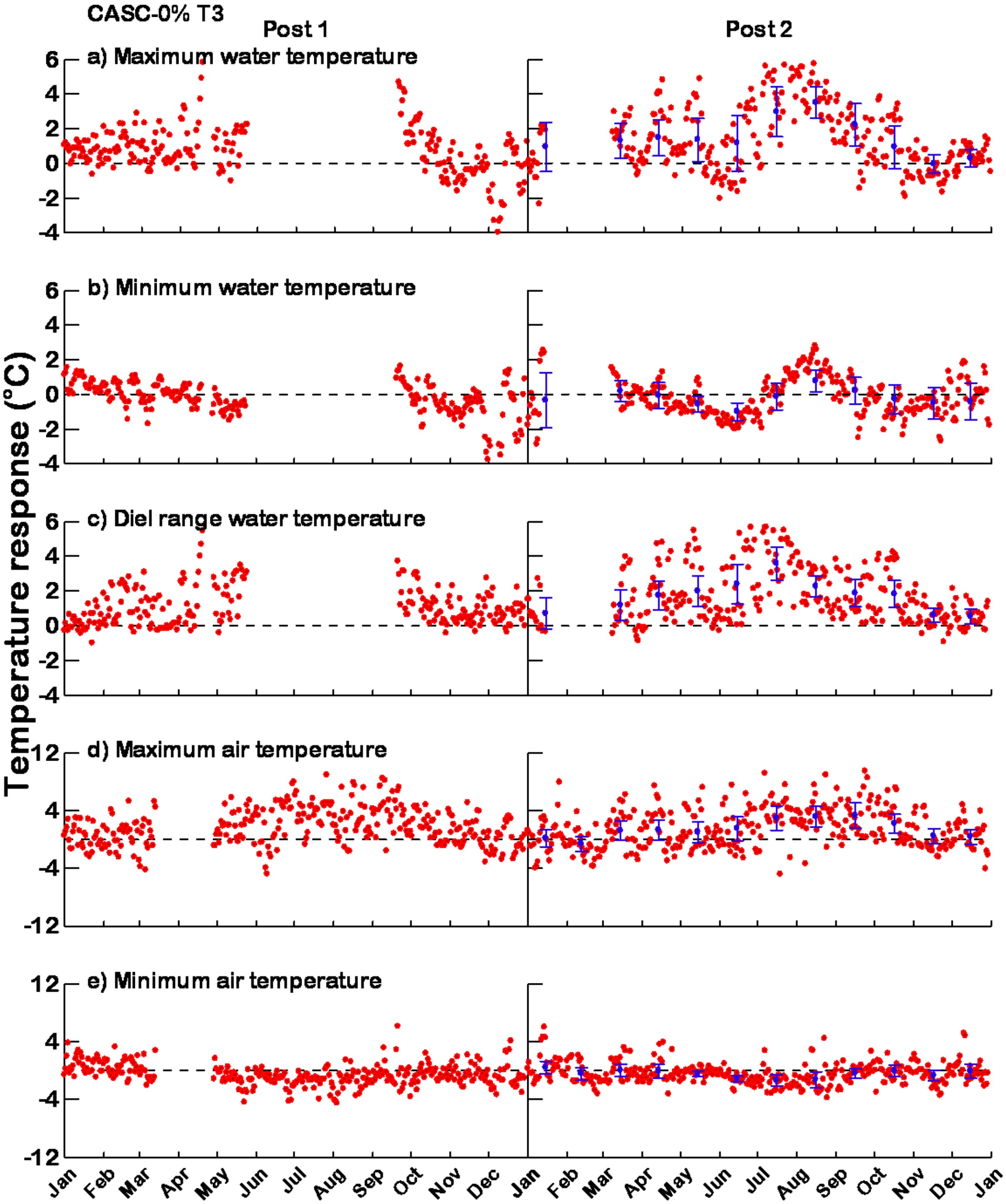
Appendix Figure 7-C-8. Water temperature and air temperature response at site OLYM-0%, location T1. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).



Appendix Figure 7-C-9. Water temperature and air temperature response at site WIL1-0%, location T1. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).

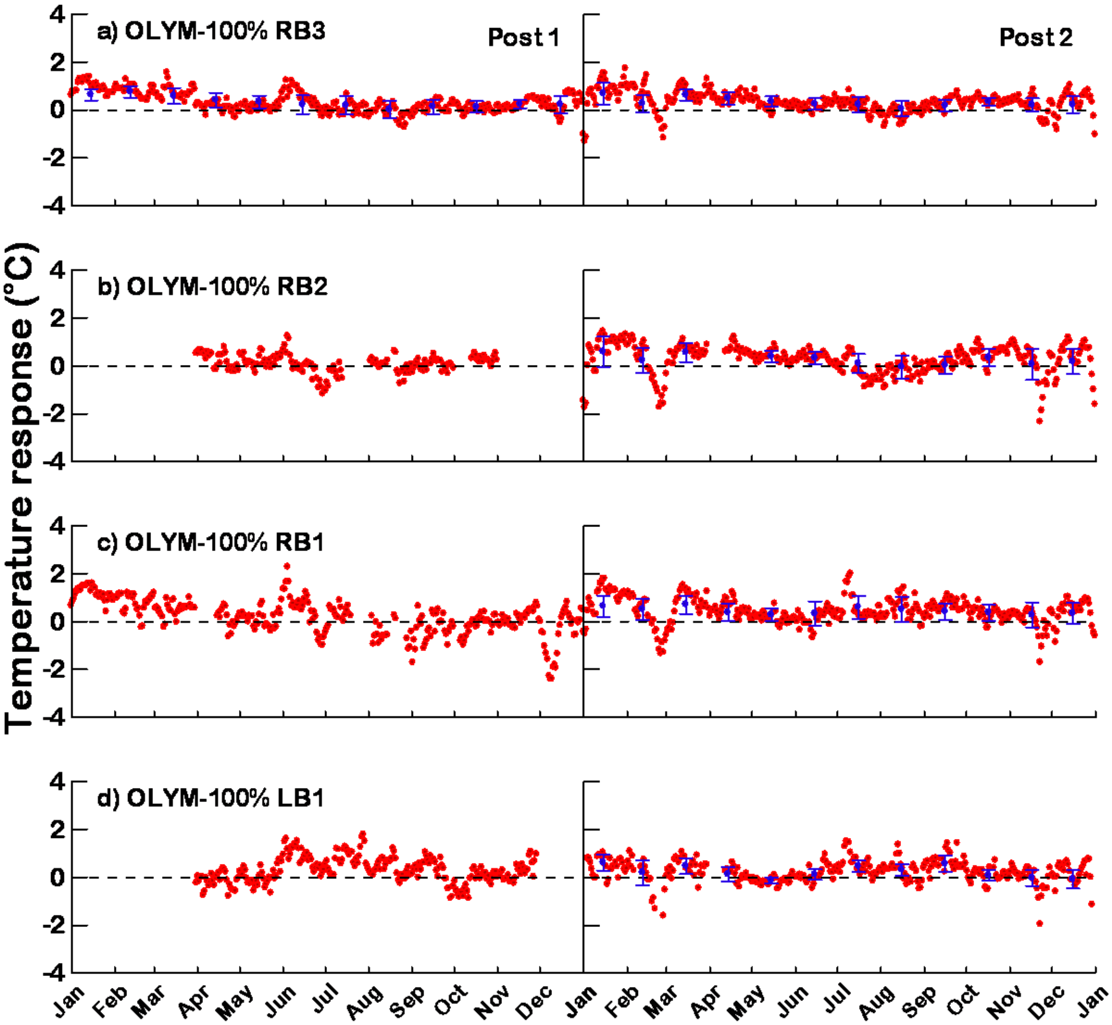


Appendix Figure 7-C-10. Water temperature and air temperature response at site WIL2-0%, location T3. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).

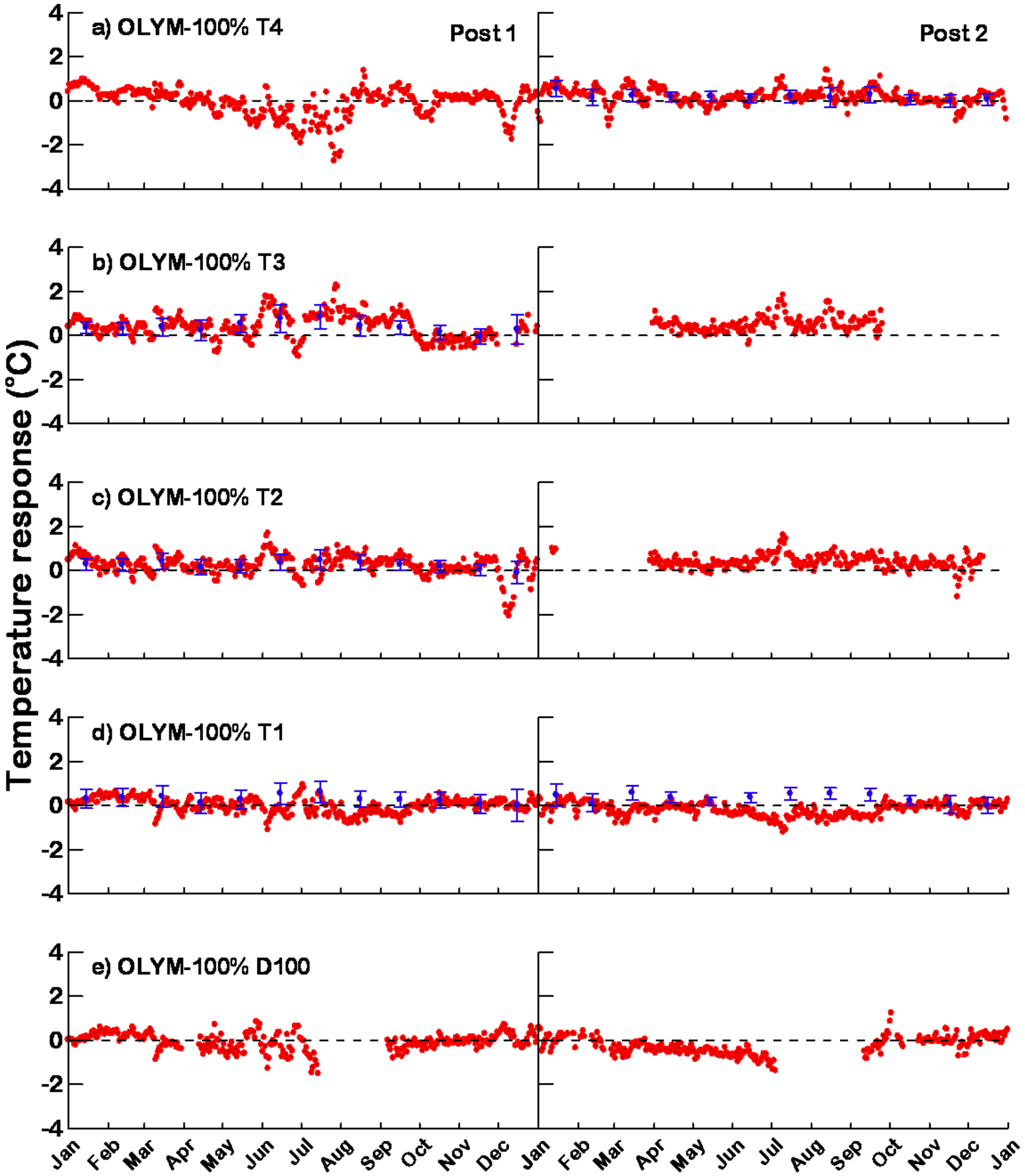


Appendix Figure 7-C-11. Water temperature and air temperature response at site CASC-0%, location T3. Red dots are the daily temperature response (TR). Blue dots and error bars are the mean monthly temperature response (MMTR) with 95% confidence intervals (CI).

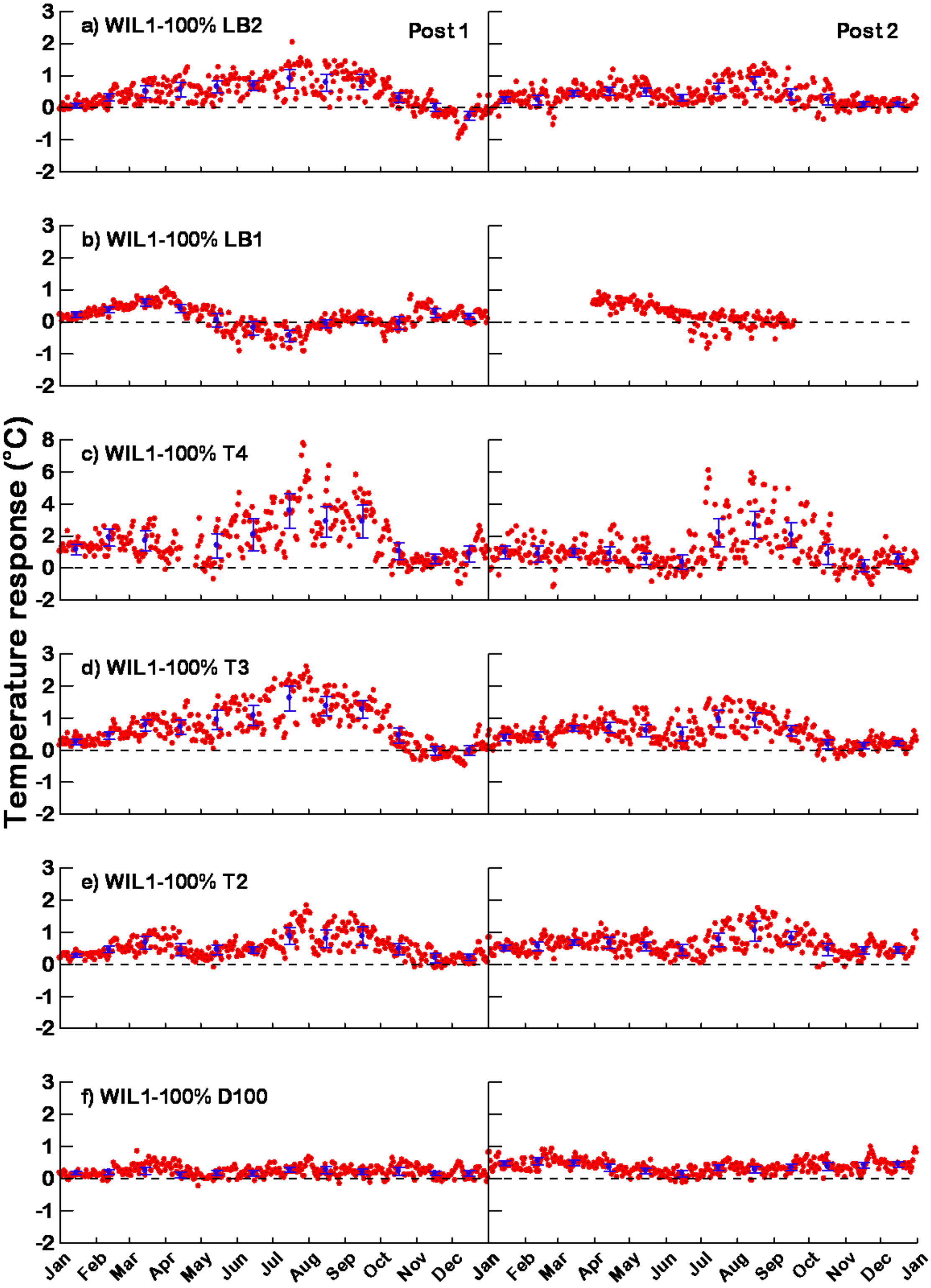
APPENDIX 7-D. LONGITUDINAL PATTERN IN MAXIMUM DAILY TEMPERATURE CHANGE



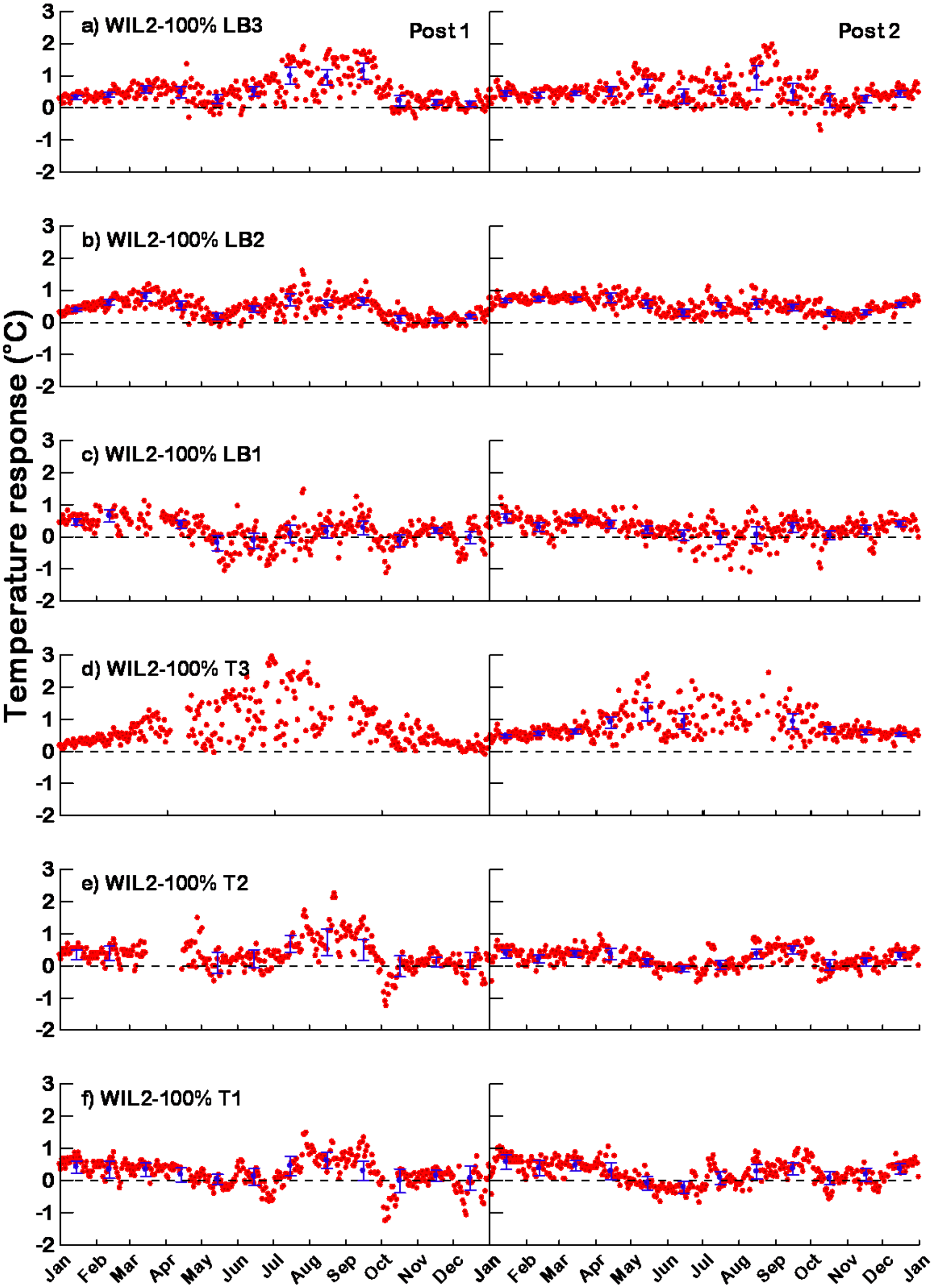
Appendix Figure 7-D-1. Daily temperature response (TR) values (in red) for all tributary locations in OLYM-100%. Mean monthly temperature response (MMTR) (in blue) with 95% confidence intervals (CI).



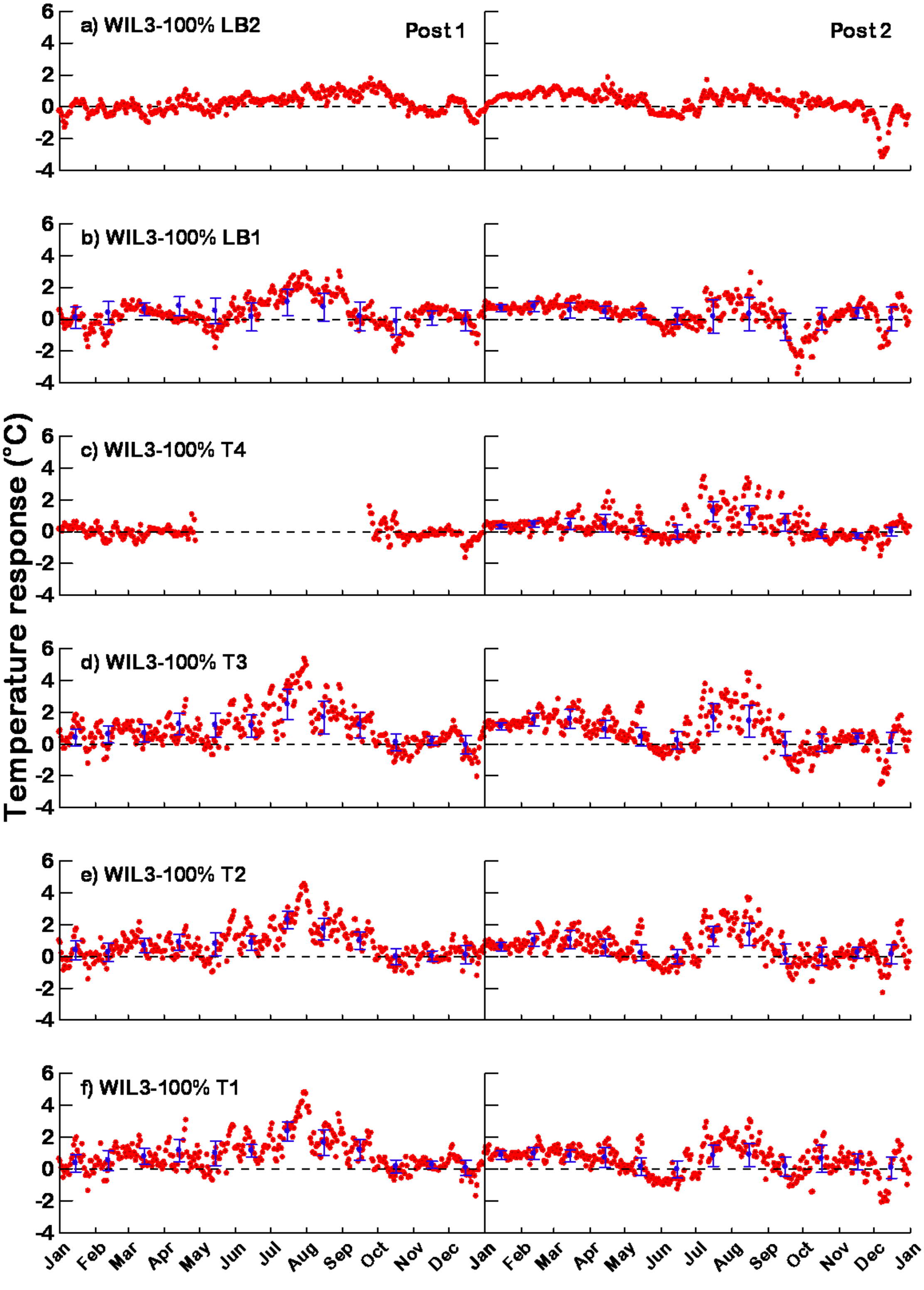
Appendix Figure 7-D-1 (continued). Daily TR (in red) and MMTR values (in blue) with 95% CI for locations in OLYM-100%.



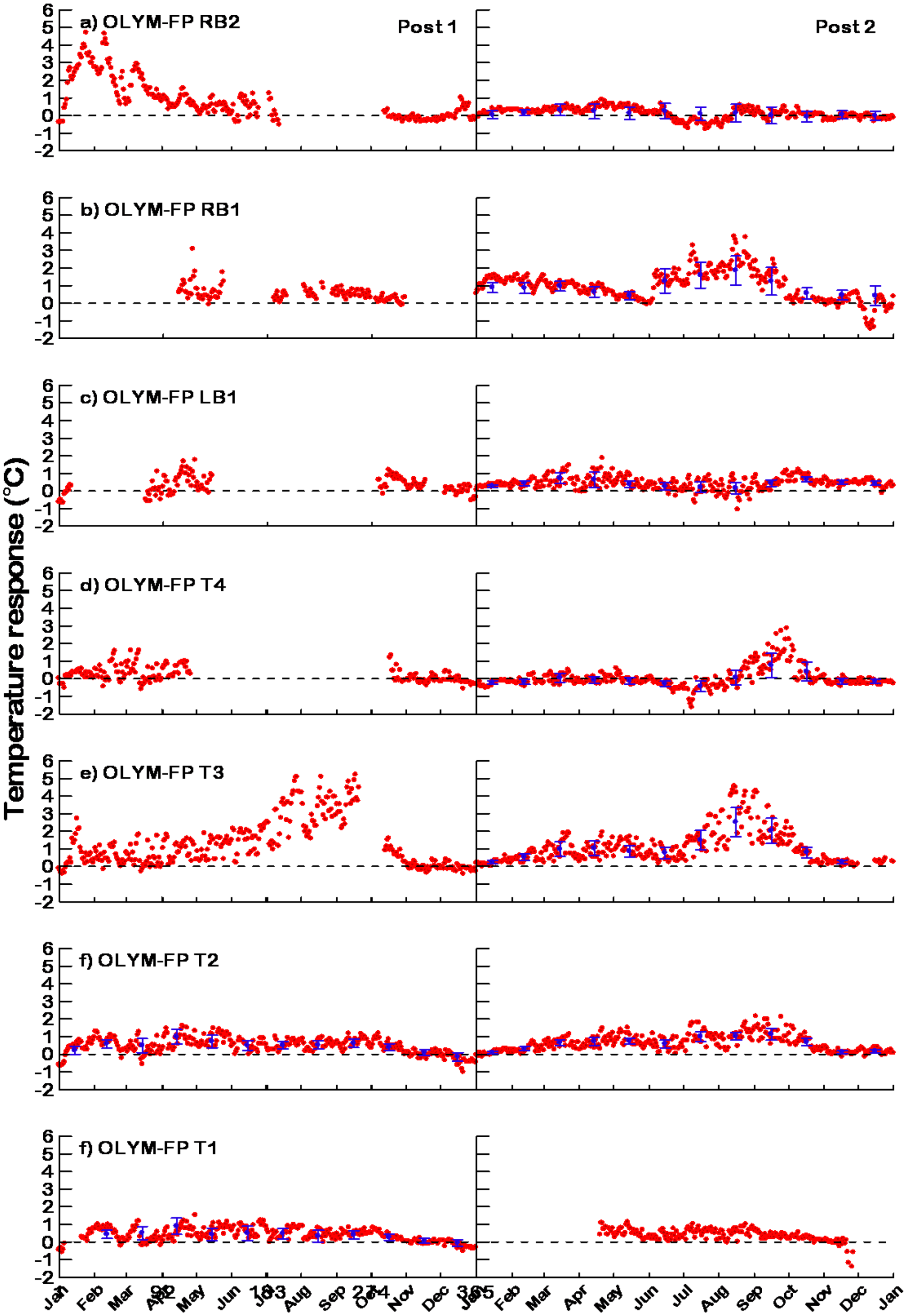
Appendix Figure 7-D-2. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in WIL1-100%.



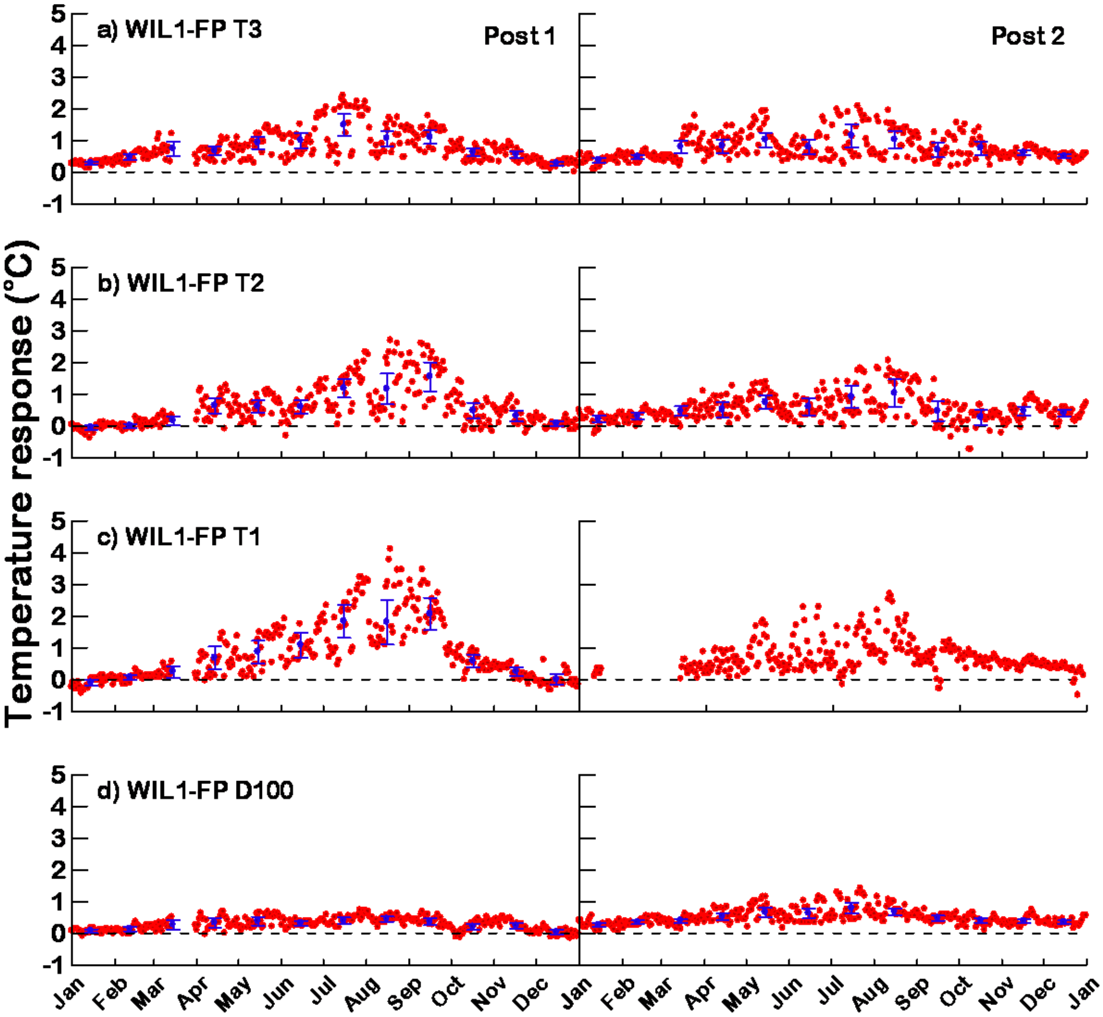
Appendix Figure 7-D-3. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in WIL2-100%.



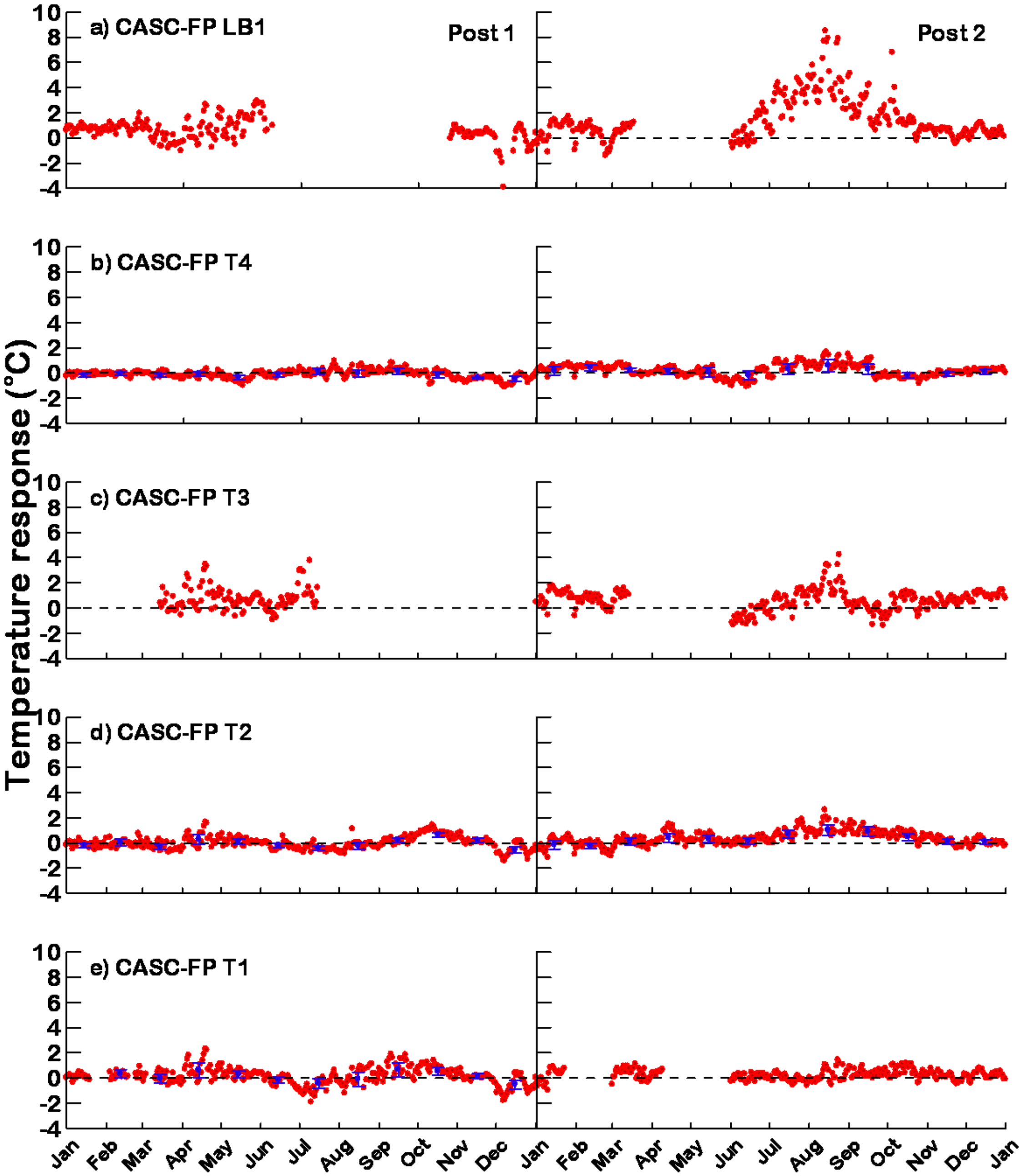
Appendix Figure 7-D-4. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in WIL3-100%.



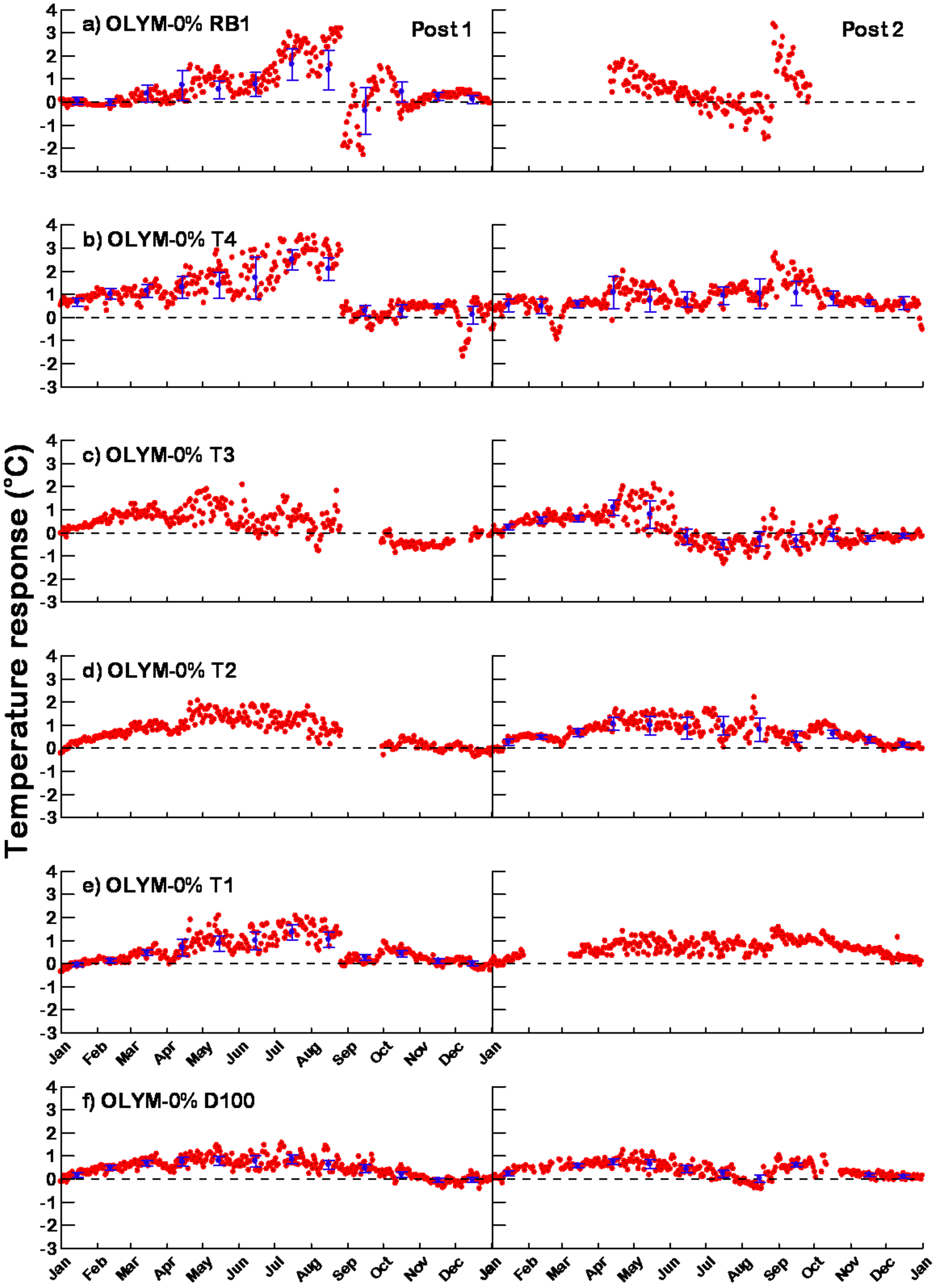
Appendix Figure 7-D-5. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in OLYM-FP.



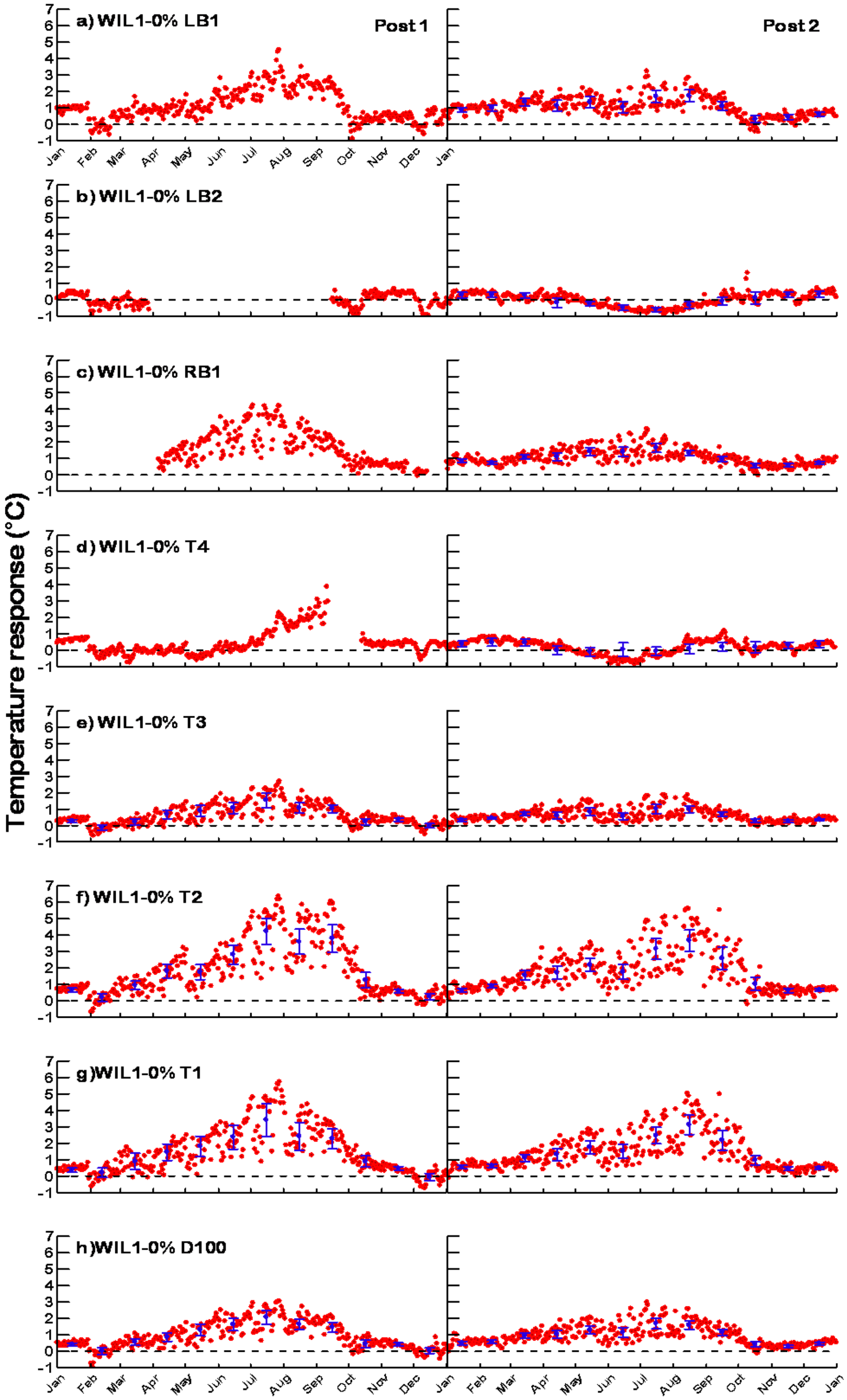
Appendix Figure 7-D-6. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in WIL1-FP treatment.



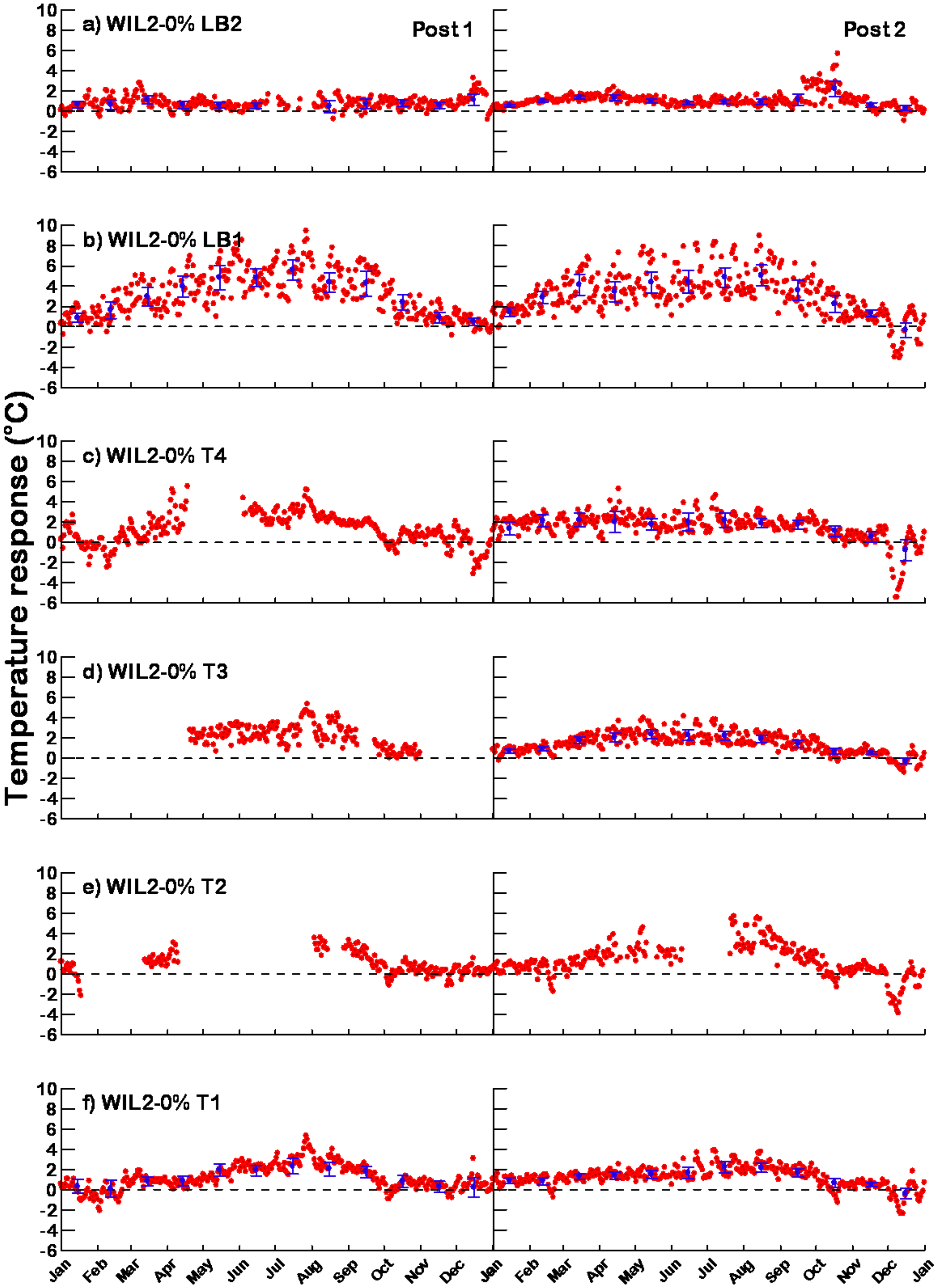
Appendix Figure 7-D-7. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in CASC-FP treatment.



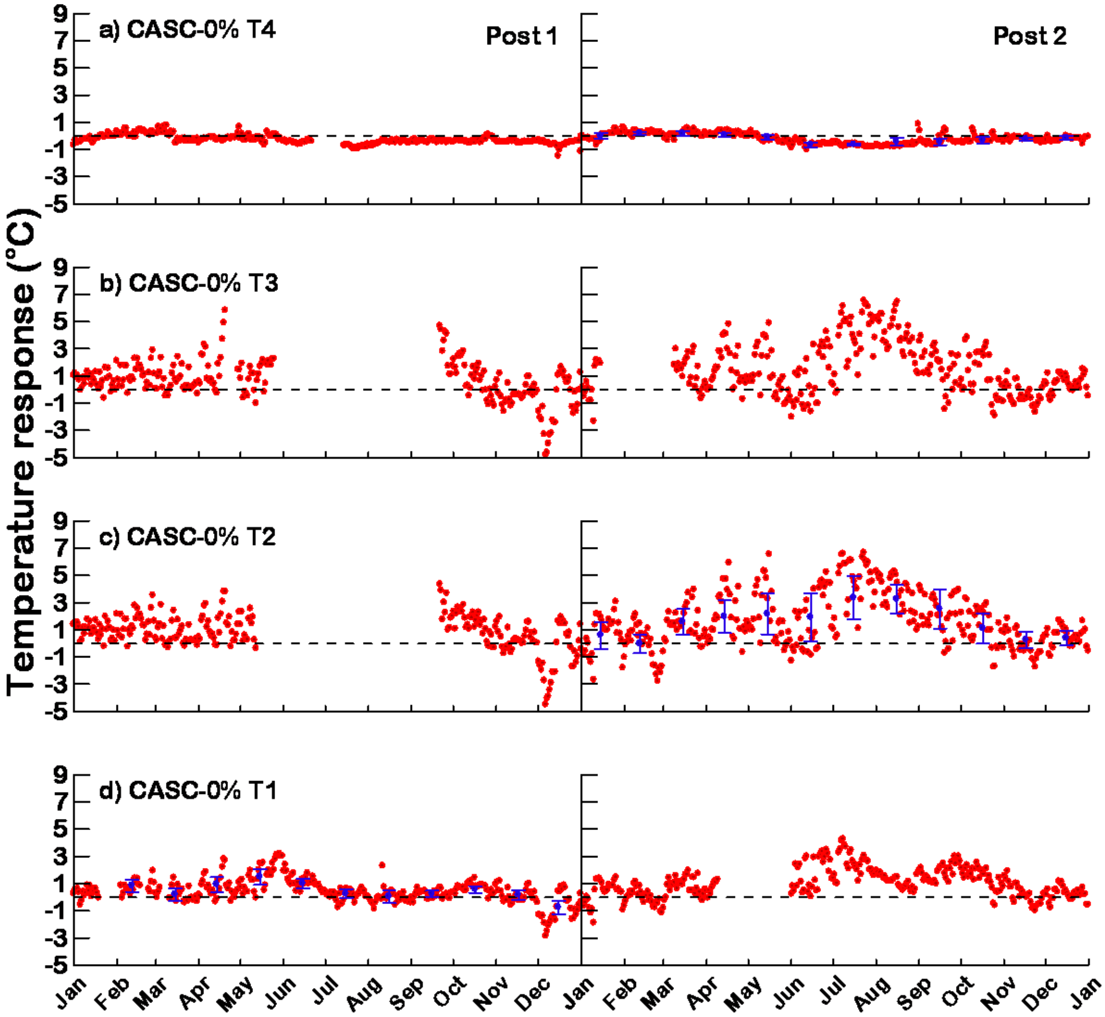
Appendix Figure 7-D-8. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in OLYM-0%.



Appendix Figure 7-D-9. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in WIL1-0%.



Appendix Figure 7-D-10. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in WIL2-0%.



Appendix Figure 7-D-11. Daily temperature response (TR) values (in red) and mean monthly temperature response (MMTR) values (in blue), with 95% confidence intervals (CI) for locations in CASC-0%.

APPENDIX 7-E. LONGITUDINAL PATTERNS IN WATER TEMPERATURE, RIPARIAN COVER, AND FLOW

Figures 7-E-1 through 7-E-11 illustrate the pattern of maximum summer stream temperature (pre- and post-harvest), post-harvest canopy closure at 0m and effective shade, post-harvest surface flow, riparian buffer length and width, and July mean monthly temperature response in the main stream channel. All variables are plotted against distance to the channel head (uppermost point of defined channel) on the x-axis.

For each figure, the upper panel includes:

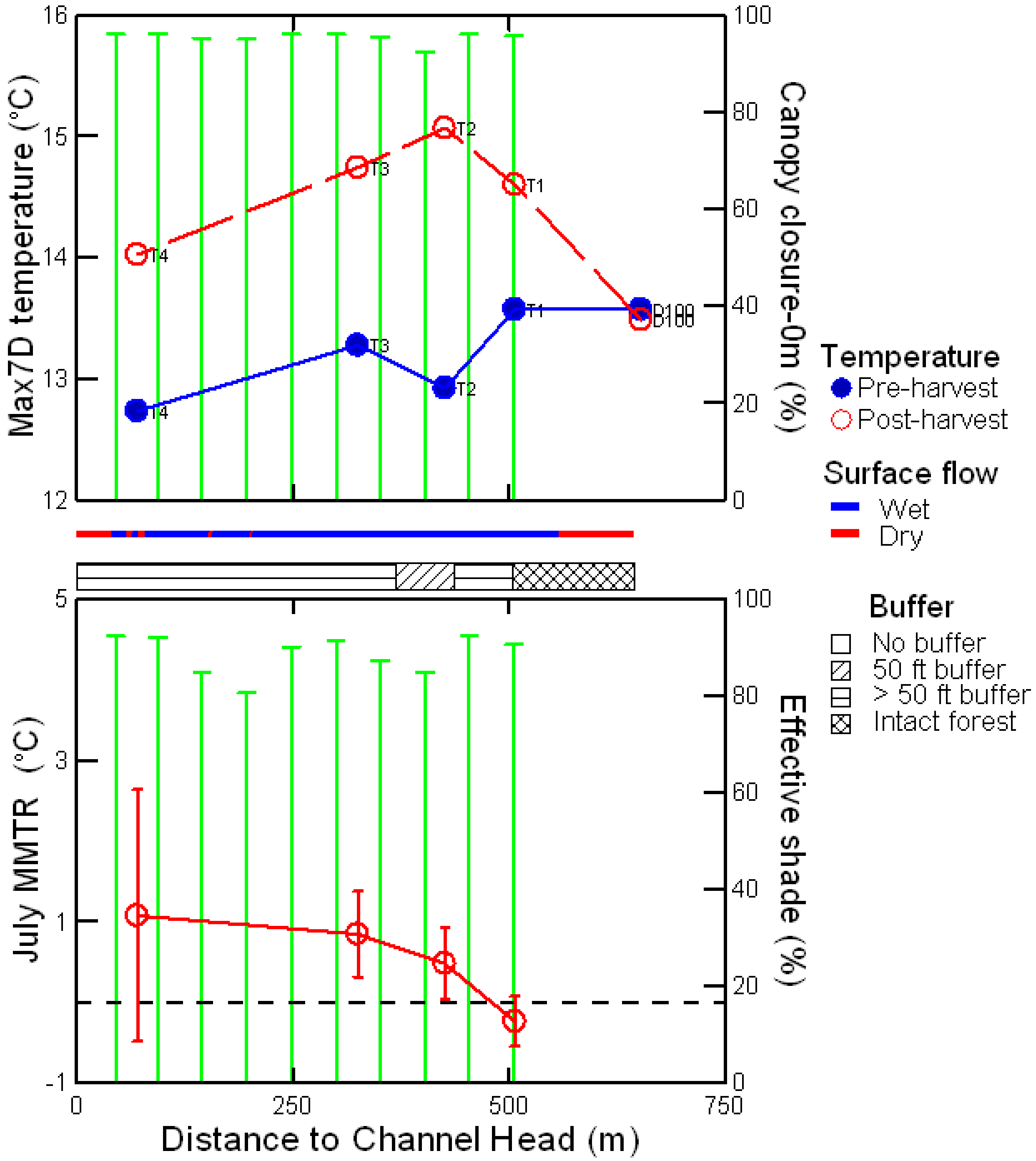
- The maximum seven-day average July–August water temperature (Max7D) in the summer immediately prior to harvest and the first year post-harvest for each main channel monitoring location. The points are labeled with the location names.
- Canopy closure measured at the water surface (CC-0m) in the first year post-harvest (vertical green lines). Units are labeled on the right side y-axis.

Surface flow (wet or dry) is shown as a colored horizontal bar below the top panel. Wetted channel reaches are shown in blue; dry reaches are in red. Surveys were conducted in summer 2010.

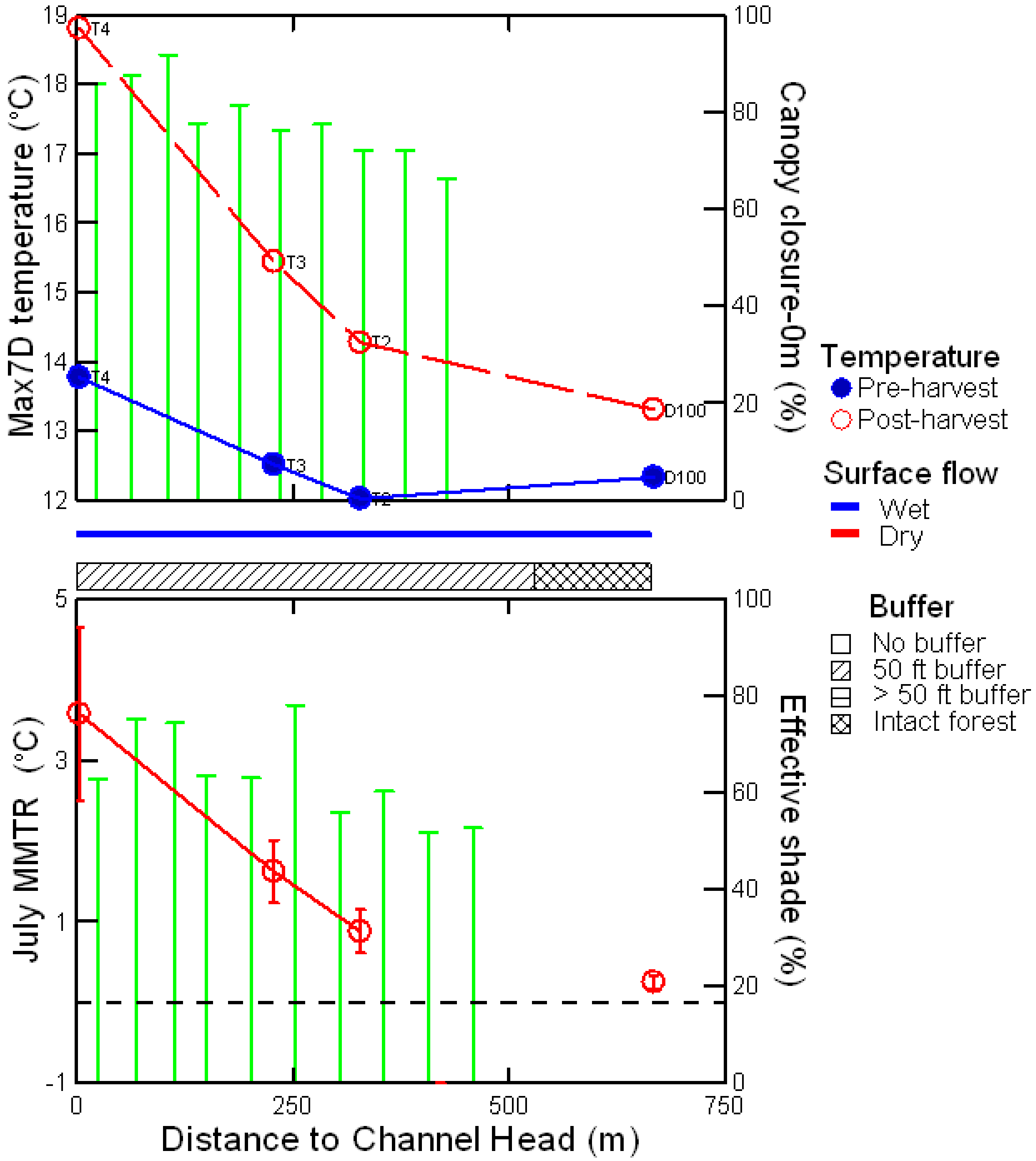
Buffer width is indicated by the black and white textured bar below the surface flow line with categories for no buffer (no bar), 50-ft (15.2-m) buffer width (diagonal lines), much greater than 50 ft (15.2 m) (horizontal lines), or intact forest (crosshatched lines).

For each figure, the lower panel includes:

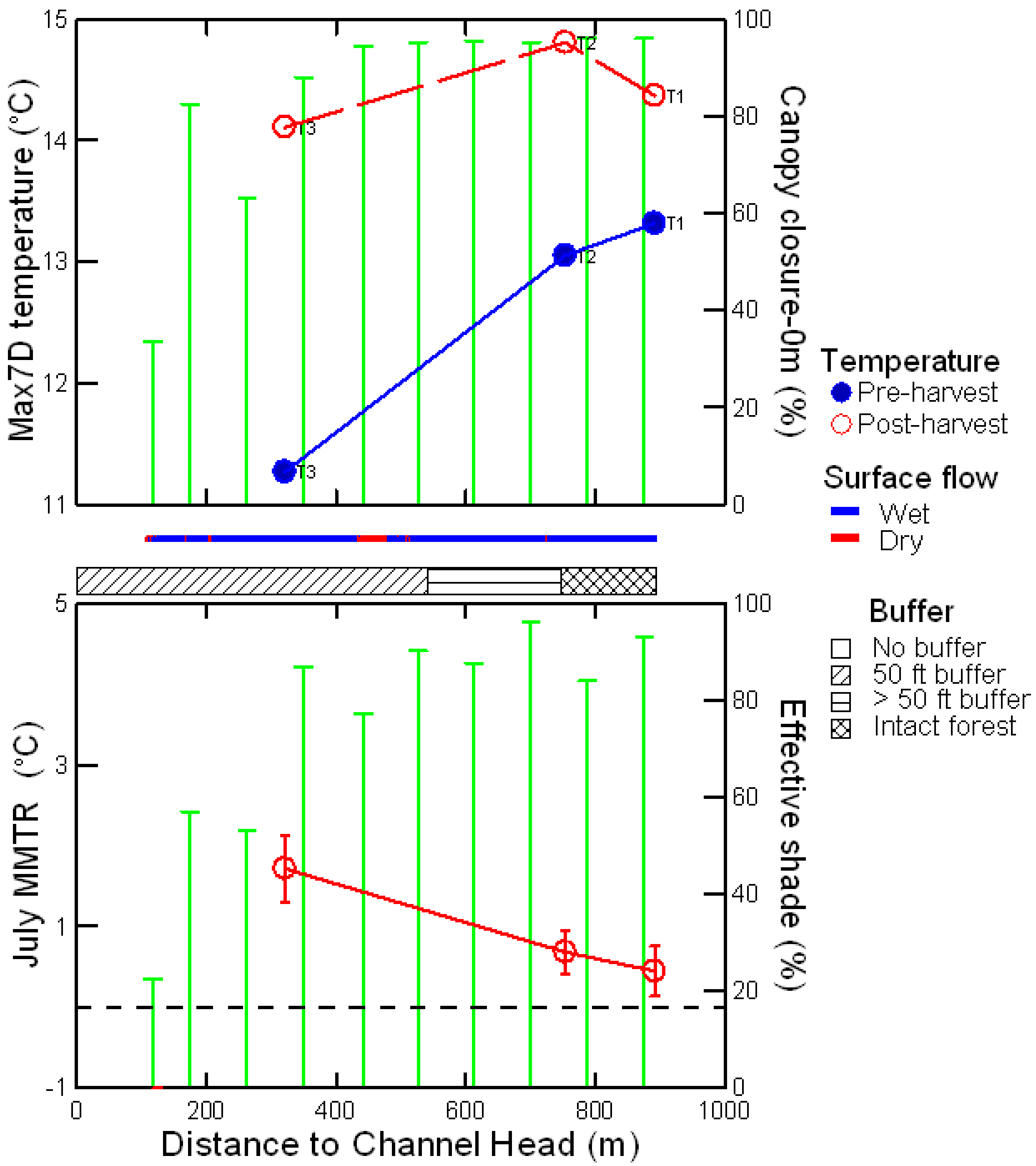
- Mean monthly temperature response in July (MMTR with 95% confidence interval [CI]) in the first year post-harvest. Temperature units on the left axis.
- Effective shade (vertical green line) in the first year post-harvest on the right side y-axis.



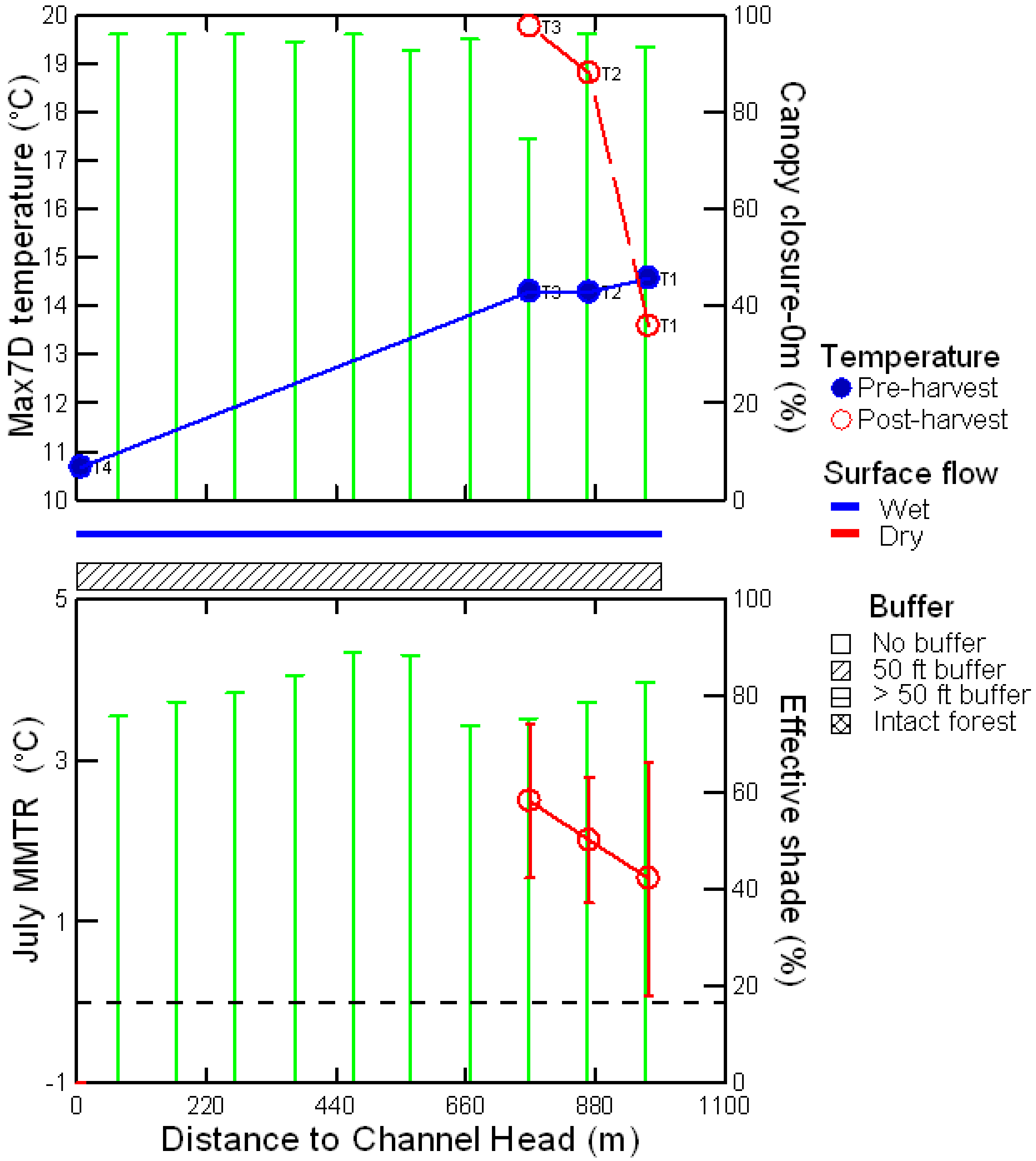
Appendix Figure 7-E-1. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at OLYM-100%. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



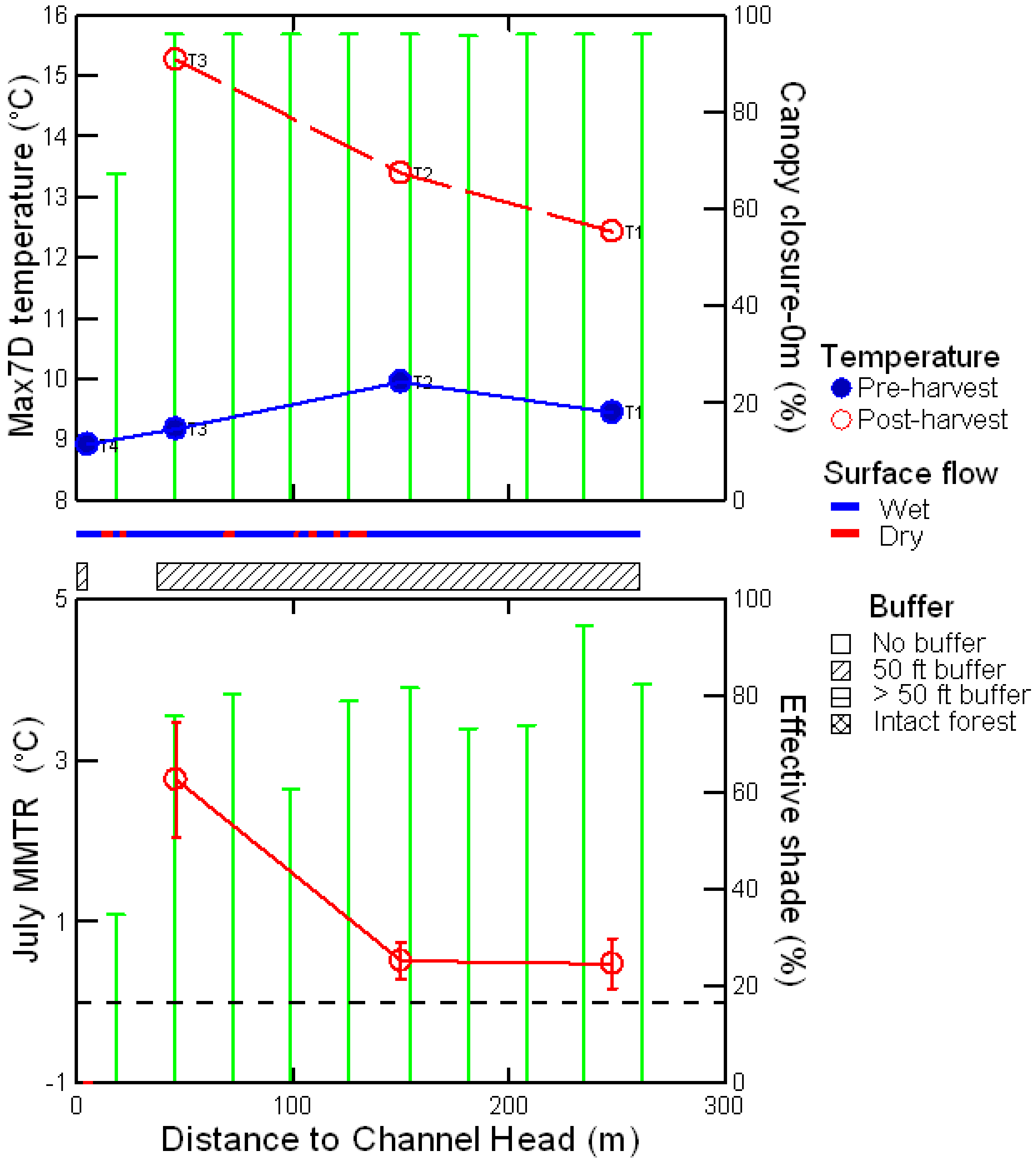
Appendix Figure 7-E-2. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at WIL1-100%. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



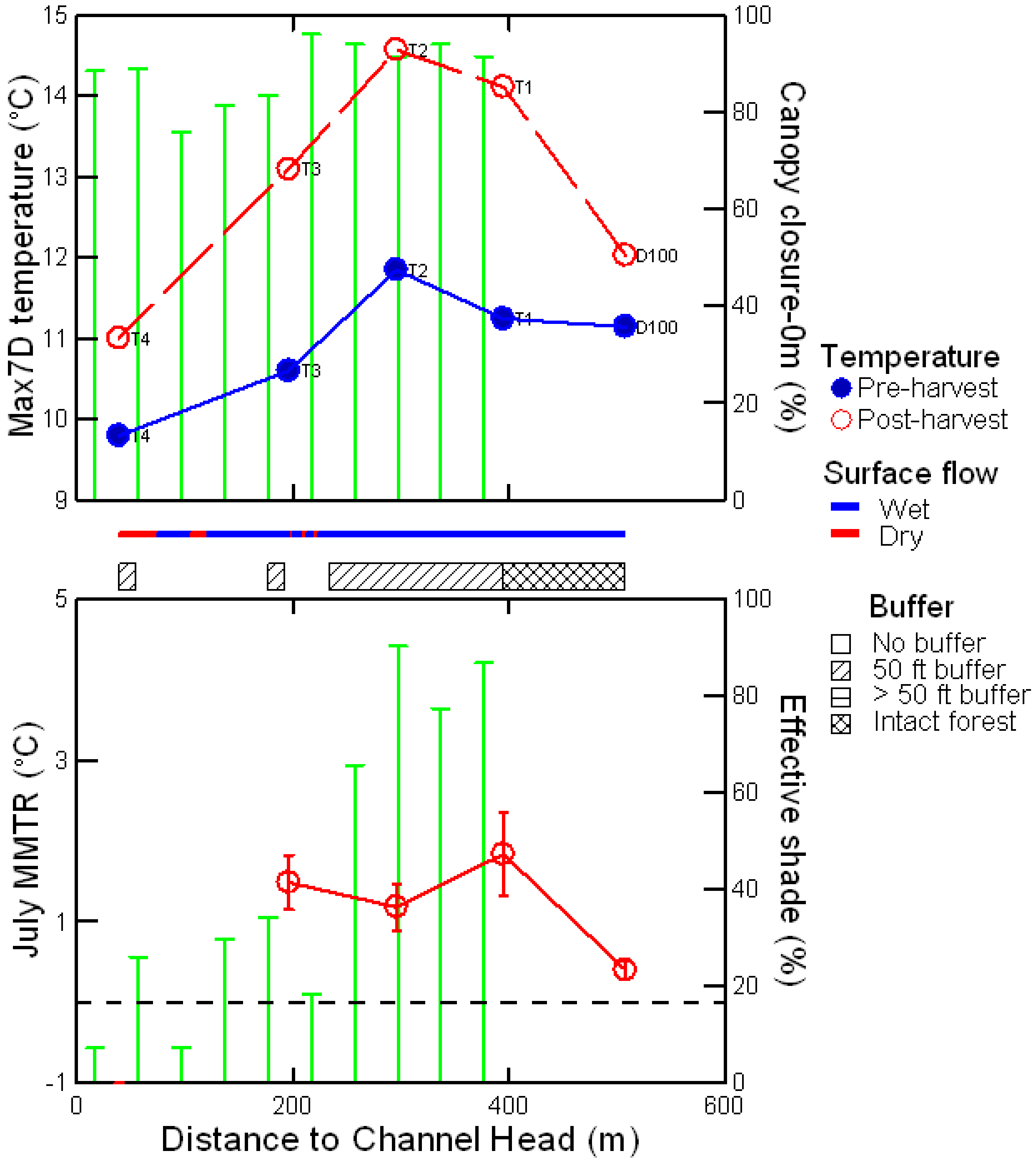
Appendix Figure 7-E-3. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at WIL2-100%. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



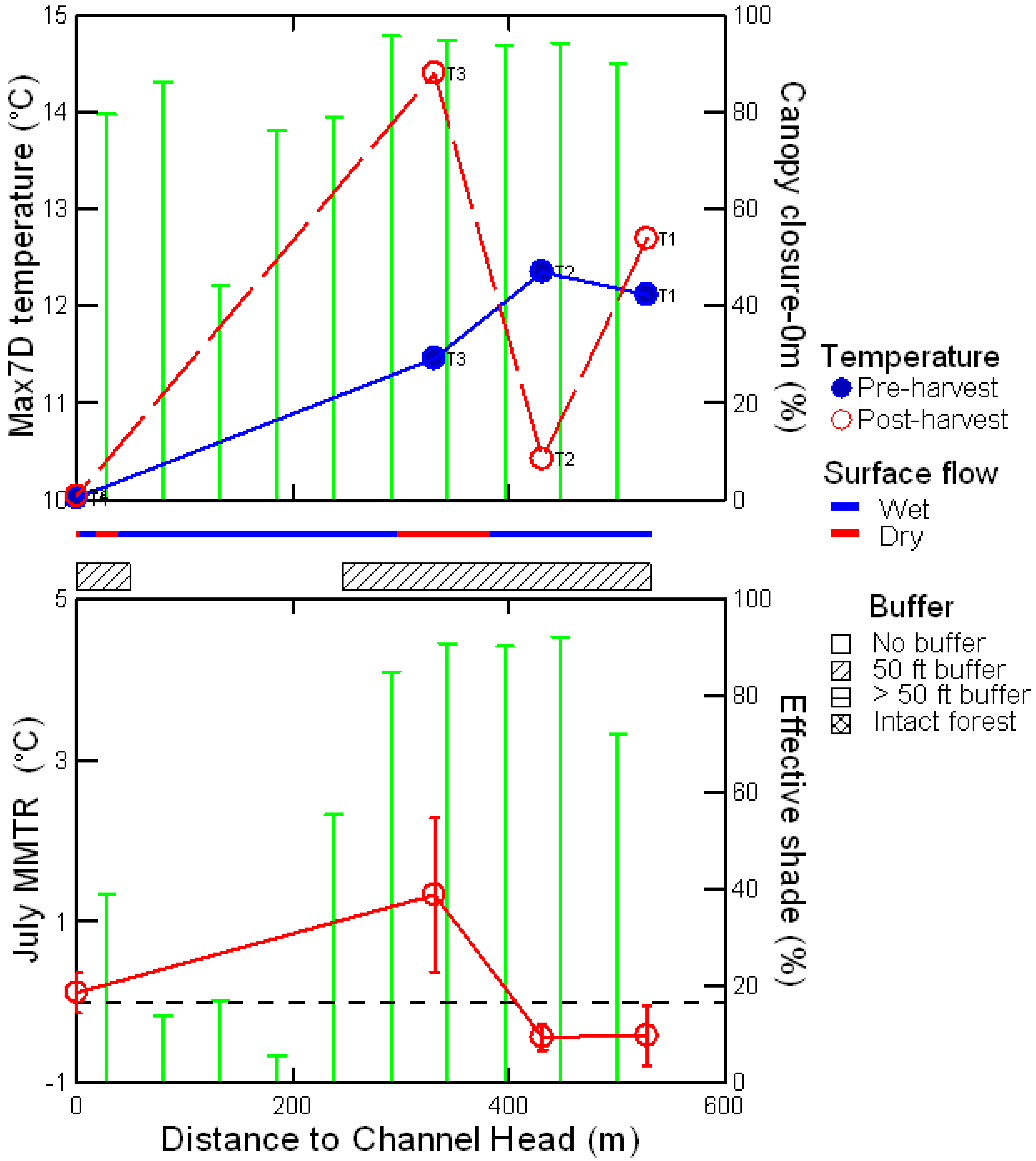
Appendix Figure 7-E-4. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at WIL3-100%. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



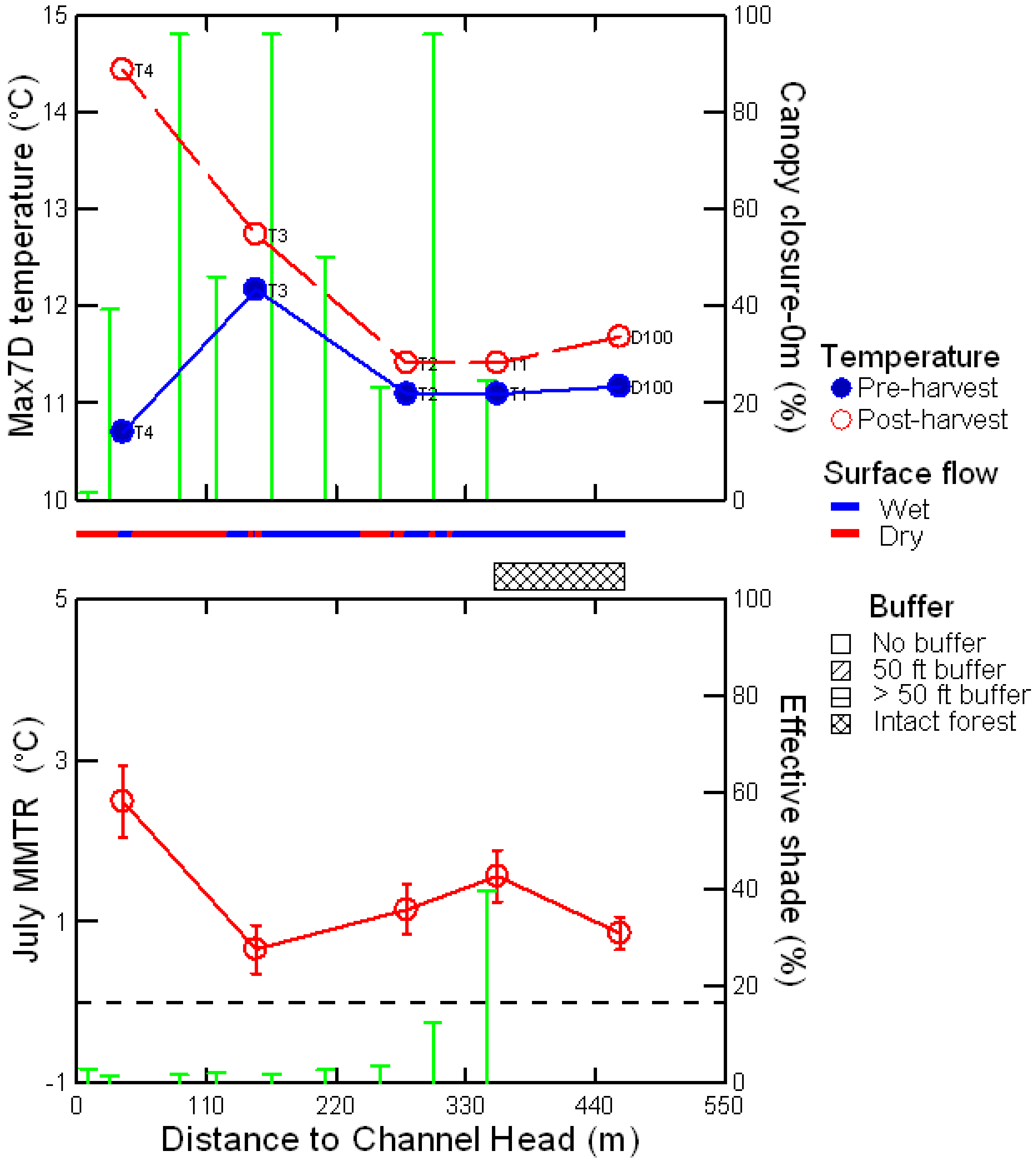
Appendix Figure 7-E-5. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at OLYM-FP. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



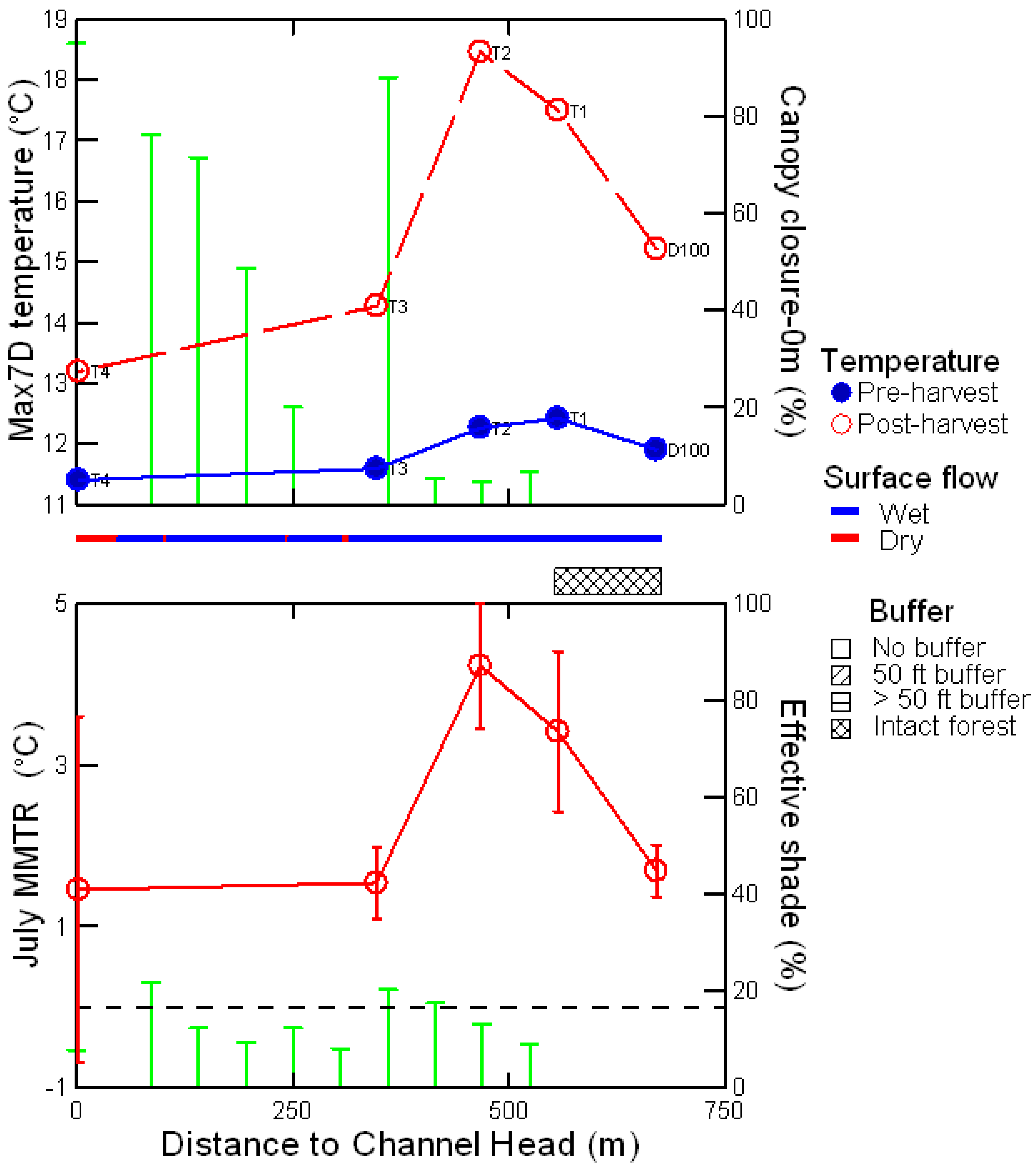
Appendix Figure 7-E-6. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at WIL1-FP. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



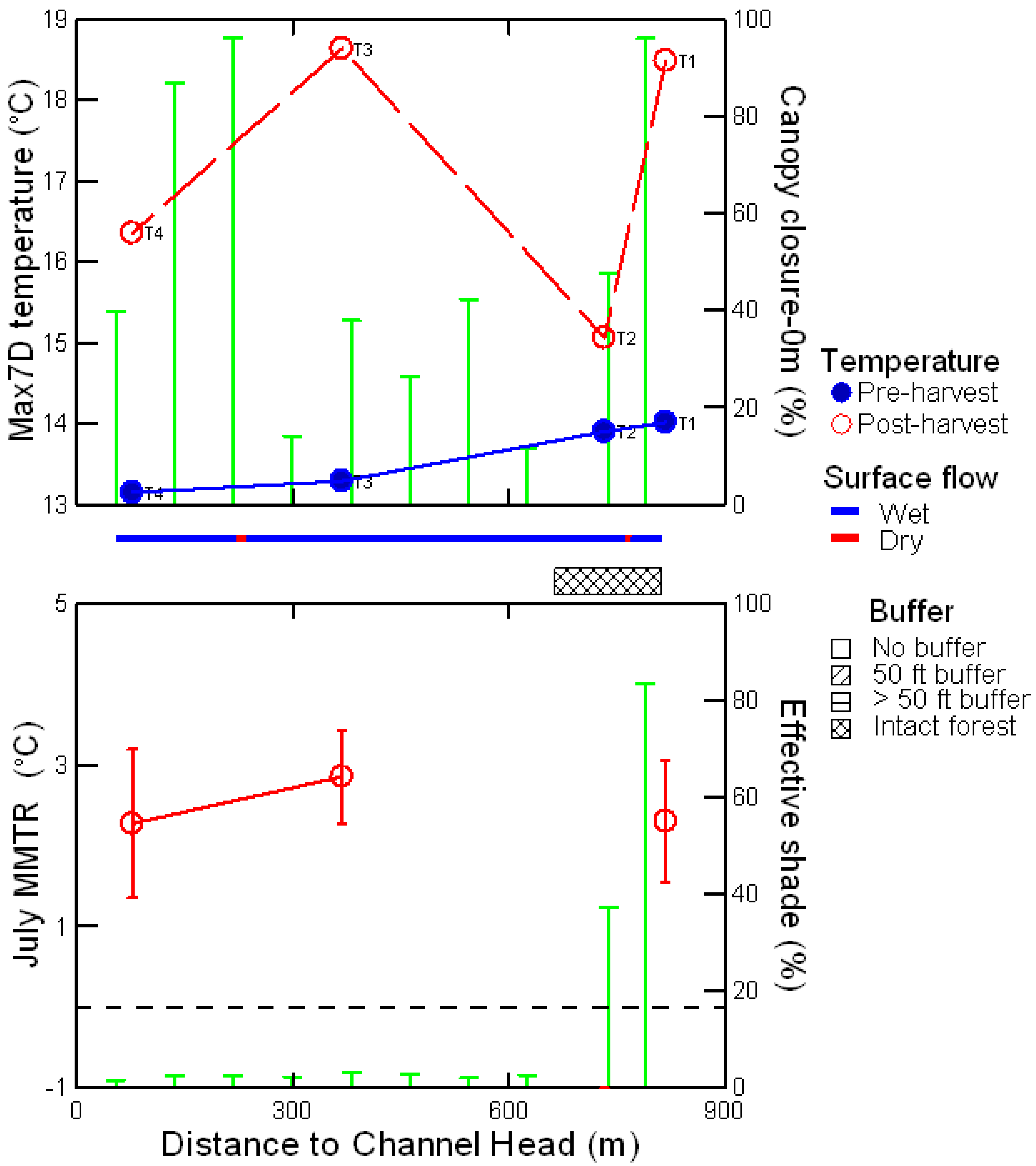
Appendix Figure 7-E-7. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at CASC-FP. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



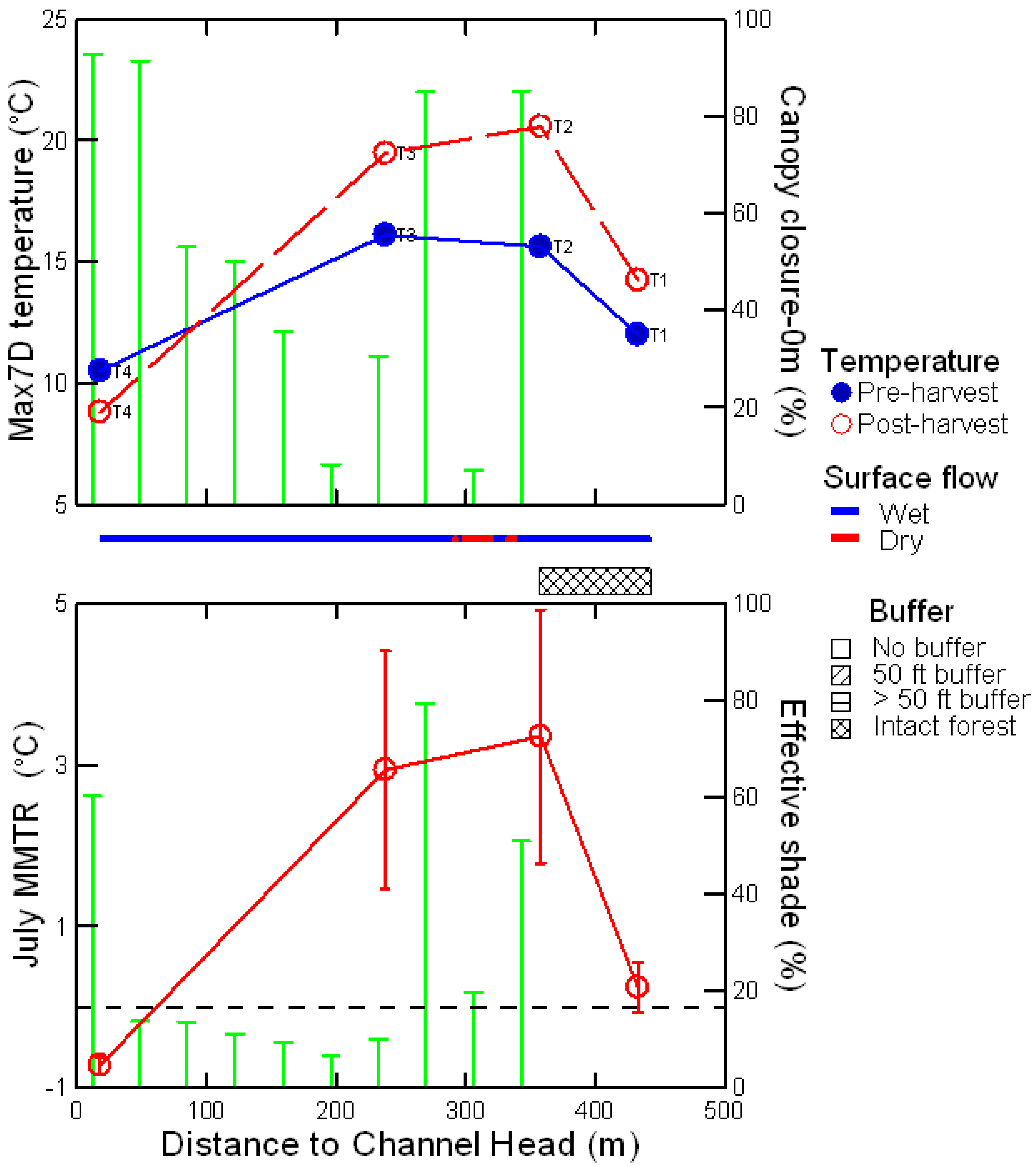
Appendix Figure 7-E-8. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at OLYM-0%. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



Appendix Figure 7-E-9. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at WIL1-0%. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



Appendix Figure 7-E-10. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at WIL2-0%. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).



Appendix Figure 7-E-11. Seven-day average maximum daily temperature and post-harvest canopy closure-0m (upper plot), wet vs. dry channel (red/blue horizontal bar), buffer type (patterned bar), and July mean monthly temperature response (MMTR) and effective shade (lower plot) at CASC-0%. Green bars are canopy closure at the water surface (top panel) or effective shade (bottom panel).

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CHAPTER 8 - DISCHARGE

Greg Stewart, William Ehinger, and Stephanie Estrella

LIST OF FIGURES

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8-1. ABSTRACT

We conducted a replicated Before-After Control-Impact (BACI) study to test the effects of timber harvest and buffer effectiveness on discharge in non-fish-bearing watersheds in western Washington as part of a much larger study. Eight of the watersheds were instrumented to measure discharge: one block of four sites in the Willapa Hills region and another block of four sites in the Olympic region. Within each block, three treatment sites were clearcut harvested and one of three riparian buffer treatments was applied to the Riparian Management Zone: (1) 50-ft (15.2-m) no-harvest buffer along 100% of the perennial stream network (100% treatment), (2) 50-ft (15.2-m) no-harvest riparian buffer along at least 50% of the perennial stream network (as well as unstable slope buffers and sensitive site buffers) as required by Washington State Forest Practices Rules (FP treatment), and (3) no buffer but with a 30-ft (9.1-m) wide equipment exclusion zone along the entire perennial channel (0% treatment). One site in each regional block was left unharvested and served as a reference. Generalized least squares regression was used to relate water discharge in treatment and reference sites in the pre-harvest period, and the regression equation and reference site discharge was used to estimate what would have been observed in the absence of the treatment. Daily treatment effects were obtained by comparing observed and predicted discharge in the treatment sites following forest harvest. The change in discharge was analyzed with a means test, as a cumulative time series, and in terms of discharge frequency. In the first two years following harvest, annual runoff increased in all treatment sites as a result of harvest but the magnitude of change varied by season and return interval. As expected, total water yield increased as a function of the proportion of basin harvested, with very little change seen in one 100% treatment site where only 46% of the basin was harvested. Mean discharge increased in the FP and 0% treatments but not the 100% treatments. A frequency analysis showed that all treatments exhibited significant changes in magnitude/frequency of events over at least part of the daily time-series of flow. Base flows decreased in the 100% treatment, were largely unchanged in the FP harvest, and increased in the 0% treatments. Changes in annual peak flows were not statistically significant, with the exception of the two highest elevation sites that may have been influenced by rain on snow, though most treatment basins did exhibit a significant increase in the frequency and magnitude of the 30-day event. The frequency analysis findings are consistent with the analyses of mean discharge, but highlight the importance of examining the entire distribution of hydrologic change as opposed to focusing on a single flow metric such as the mean.

8-2. INTRODUCTION

Forest management can affect headwater stream hydrology (Moore and Wondzell 2005). The removal of forest canopy reduces interception and evapotranspiration which changes 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 thereby increasing the water volume and the speed at which it enters the channel (Wemple *et al.* 1996; Wemple and Jones 2003). As a result, watershed studies from the Pacific Northwest and elsewhere have generally found that water yield increases following timber harvest, though the magnitude and timing of change are affected by a large number of factors, including the amount and type of harvest, road building activity, precipitation and snowmelt regimes, and type of local vegetation (Stednick 1996; MacDonald *et al.* 2003; Jones and Post 2004; Brown *et al.* 2005; Moore and Wondzell 2005). Given these factors, the exact magnitude of change due to timber harvest is hard to predict.

8-3. OBJECTIVES

This chapter examines changes in discharge magnitude and frequency following clearcut harvest outside of a Riparian Management Zone (RMZ) in the six buffer treatment basins of the Willapa 1 (WIL1) and Olympic (OLYM) blocks. The primary reason for measuring discharge was to calculate nutrient and suspended sediment loads and a secondary objective was to compare changes in discharge among treatments. The research questions we addressed were:

> 1) What is the magnitude of change in mean daily discharge that can be attributed to the buffer treatments in the first two years after harvest?
>
> 2) How does daily discharge magnitude and frequency change in the two-year period following harvest?

The information presented in this chapter is referenced in Chapter 9 – *Nutrient Export* and Chapter 10 – *Sediment Processes*, since discharge magnitude has a direct effect on constituent concentration and loads.

8-4. METHODS

8-4.1. SITE DESCRIPTION

We measured discharge in the eight basins of the Olympic and Willapa 1 blocks in western Washington State. These sites were chosen because the Olympic and Willapa 1 blocks were complete (i.e., all four treatments were represented) and readily accessible. Spur roads were constructed, but no new road crossings were installed. Basin areas above the flow measurement point varied from 11.8 to 44.3 ha (29.1 to 109.4 ac) and basin elevations range from 86 to 480 m (282 to 1574 ft; **Table 8-1**). The western Olympic coastal region is exposed to storms coming in from the Pacific Ocean and average annual precipitation ranges from 2,241 mm/yr (88 in/yr) in the WIL1-0% to 3,746 mm/yr (147 in/yr) in the OLYM-REF, OLYM-FP, and OLYM-0% based on 1981−2010 PRISM climate estimates (PRISM Climate Group 2013). Most of the precipitation falls as rain, but snow and rain-on-snow events are possible in the winter. We did not collect local meteorological information, although ground snow cover was observed for brief periods in one or more of the Olympic block sites in 2008, 2009, 2010, and 2011. In these sites, snow melts rather quickly and snow depths seldom exceeded 152 mm (6 in).

Table 8-1. Basin area and harvest amounts above the flow gauge.

Block	Treatment	Min Elev (m)	Max Elev (m)	Area (ha)	% Clearcut	% Channel Buffered
WIL1	REF	195	388	11.8	0	-
	100%	217	418	26.2	89	100
	FP	185	407	14.4	94	73
	0%	87	225	27.7	100	0
OLYM	REF	214	481	44.3	0	-
	100%	103	297	22.1	45	100
	FP	277	445	17.3	88	62
	0%	243	481	13.1	100	0

Not all of the blocks included in the study were adjacent to one another. In the Olympic block, the reference, 0% and FP treatments were within 4 km of one-another while the 100% treatment was approximately 50 km away. In the Willapa 1 block, the reference, 100% and FP treatment were adjacent, but the 0% treatment was about 27.5 km away (**Figure 8-1**). Study design and complete basin descriptions are found in Chapter 2 – *Study Design* and Chapter 3– *Management Prescriptions* of this report.

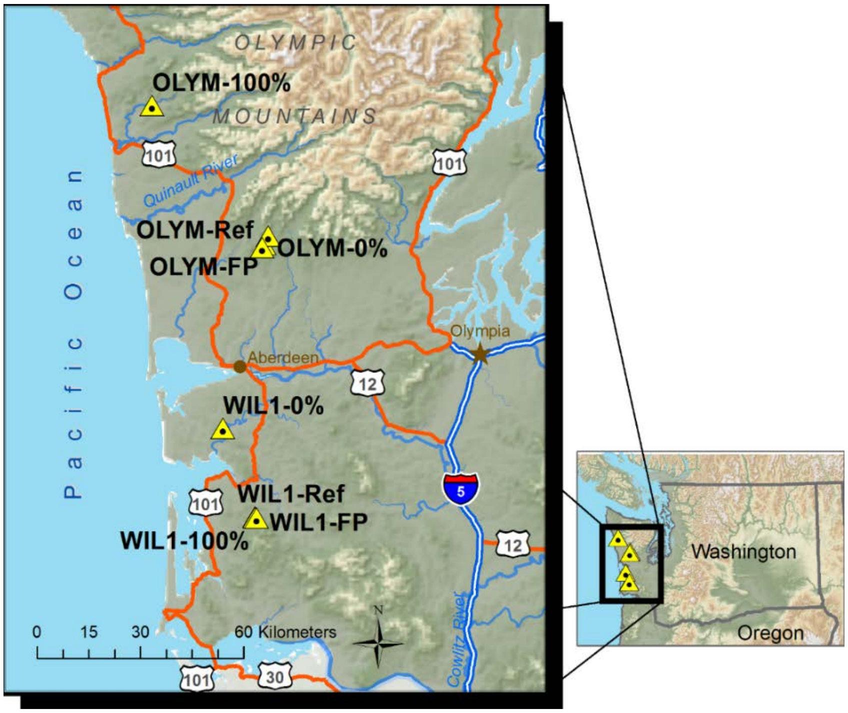

Figure 8-1. Location of the eight basins where discharge was measured. The WIL1-REF, WIL1-100%, and WIL-FP are adjacent and shown by one symbol.

We monitored water discharge continuously in five of the six treatments and the WIL1-REF from October 2006 through September 2011. The OLYM-REF was not instrumented until January 2007, and road maintenance in the WIL1-100% blocked access through the fall 2006 work window, so that the flume was not installed until August 2007.

At each site, stage height was recorded at 10-minute intervals using a system from Forest Technology Systems (www.FTSenvironmental.com) consisting of an Ott PS 1 pressure transducer, HDL1 datalogger and Forest Technology Systems StreamTrac system.

Stage height was measured within a stilling well in an 18 inch or 24 inch Montana-style Parshall flume at six sites, but no suitable location for a flume existed in two sites (OLYM-REF and OLYM-100%) so stage height was measured at the upstream end of a road culvert in each. We measured stage height with a staff gauge on each site visit (6- to 8-week intervals), and used this measurement to correct for drift in the pressure transducer’s stage height measurements prior to calculating flow. We calculated flow using the flow versus stage height relationship provided by the flume manufacturer (AccuraFlow). At the culvert sites, we measured discrete flows using a Swoffer flow meter or, at very low flows, by measuring the time to fill a bucket from the culvert. We then calculated continuous flow measurements based on the discharge versus stage height relationship developed for each location. Estimating discharge values greater than those used to develop the discharge-stage relationship was a concern so we made an effort to sample high discharge events. Only 1% of the estimated discharge values were greater than the maximum measured discharge at OLYM-REF and 0.4% at OLYM-100%.

Due to a poor flow versus stage height relationship in OLYM-100% in the first year of the study, we used only those OLYM-100% flow data collected after 1 January 2008. Also, at the OLYM-REF and OLYM-100%, where stage was measured at a culvert rather than a flume, the flow versus stage height relationship did not hold for very low stage height values (<~3 L s^{-1}), so discharge estimates below 3 L/s were not used to develop regression equations.

Given differences in basin size and location with respect to storm paths, short-term lags in discharge response were commonly observed. For this reason and others, all the analyses were conducted on total daily discharge, which was then normalized to total specific daily discharge (i.e., discharge per unit area) to facilitate direct comparison among basins in units of m^3/ha/day or mm/year. We obtained specific daily discharge by dividing total daily discharge (m^3/day) by the topographic drainage area above the flume (ha). While the dataset was largely intact, a few 10-minute records were lost so we calculated total daily discharge as the average of the 10-minute discharge measurements on a given date multiplied by 144. The annual rate is the sum of daily discharge for 365 days or the sum of daily discharge over 730 days divided by 2, depending on how it is presented.

We calculated the change in specific daily discharge resulting from the treatments in a three-step process: (1) regression between reference and treatment basin discharge in the pre-harvest period to develop a regression equation; (2) use of the regression equation and reference site flow data to estimate what would have been observed in each buffer treatment site in the absence of harvest; and (3) comparison between observed and predicted discharge. This methodology has been used for analyzing discharge in other studies including Gomi and colleagues (2006) and Alila and colleagues (2009). In this study, the difference between observed and predicted discharge (i.e., effect size) was analyzed with a means test, in terms of a cumulative discharge time series, and in terms of changes in flow frequency. All three analyses rely on chronologically paired Generalized Least Squares (GLS) regression of pre-harvest flow data and post-harvest observations in the reference basin.

8-4.2. PRE-HARVEST PERIOD GLS REGRESSION BETWEEN REFERENCE AND TREATMENT

To determine how discharge changed during the post-harvest period, we estimated the discharge that would have been observed had the buffer treatment not occurred. To derive this estimate, we modeled the relationship in daily discharge between our reference and treatment basins in the pre-harvest period, and used the equation and post-harvest reference basin flow observations to derive the expected discharge.

In this study, we used a regression model of the form:

$$y_t = \beta_0 + \beta_1 x_t + \beta_2 {x_t}^2 + A\cos(2\pi t/365.25 + \psi) + \varepsilon_t \tag{8-1}$$

where: y_t is the observed daily discharge in a buffer treatment site on pre-harvest day t,
x_t is the observed daily discharge in the reference site 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 a seasonal shift in the response between reference and treatment,
ψ is a shift in the seasonal response time, and
ε_t is an error term.

The seasonal harmonic was linearized using the trigonometric identity:

$$Acos(\lambda t + \psi) = \beta_3 \sin(\lambda t) + \beta_4 \cos(\lambda t) \qquad (8\text{-}2)$$

where: A is the amplitude of the seasonal variation ($A = \sqrt{\beta_3 + \beta_4}$),
λ is the period expressed in radians (e.g., $2\pi/365.25$),
t is time (e.g., Julian day), and
ψ is the phase shift in the response ($\psi = \arctan(\beta_4/\beta_3)/\lambda$).

Combining **Equations 8-1** and **8-2** creates a linear equation whose parameters can be estimated through regression analysis:

$$y_t = \beta_0 + \beta_1 x_t + \beta_2 {x_t}^2 + \beta_3 \sin\frac{2\pi t}{365.25} + \beta_4 \cos\frac{2\pi t}{365.25} + \varepsilon_t \qquad (8\text{-}3)$$

We used the model described by **Equation 8-3** to determine the pre-harvest relationship between daily discharge in the treatment and reference basins.

Initial regression diagnostic plots showed that the data and residuals were heteroscedastic, so the analysis was conducted on $\log_{10}$ transformed values to meet the assumption of constant error variance (**Figure 8-2**).

A key assumption in regression analysis is that the models errors (ε_t) are independent and identically distributed, but daily discharge is likely to be temporally autocorrelated (autocorrelation occurs when a measurement at one point in time or space is correlated with nearby measurements in time or space, resulting in decreased estimates of population variance and improper confidence intervals). A time series analysis showed that the regression model errors were serially autocorrelated (**Figure 8-3**).

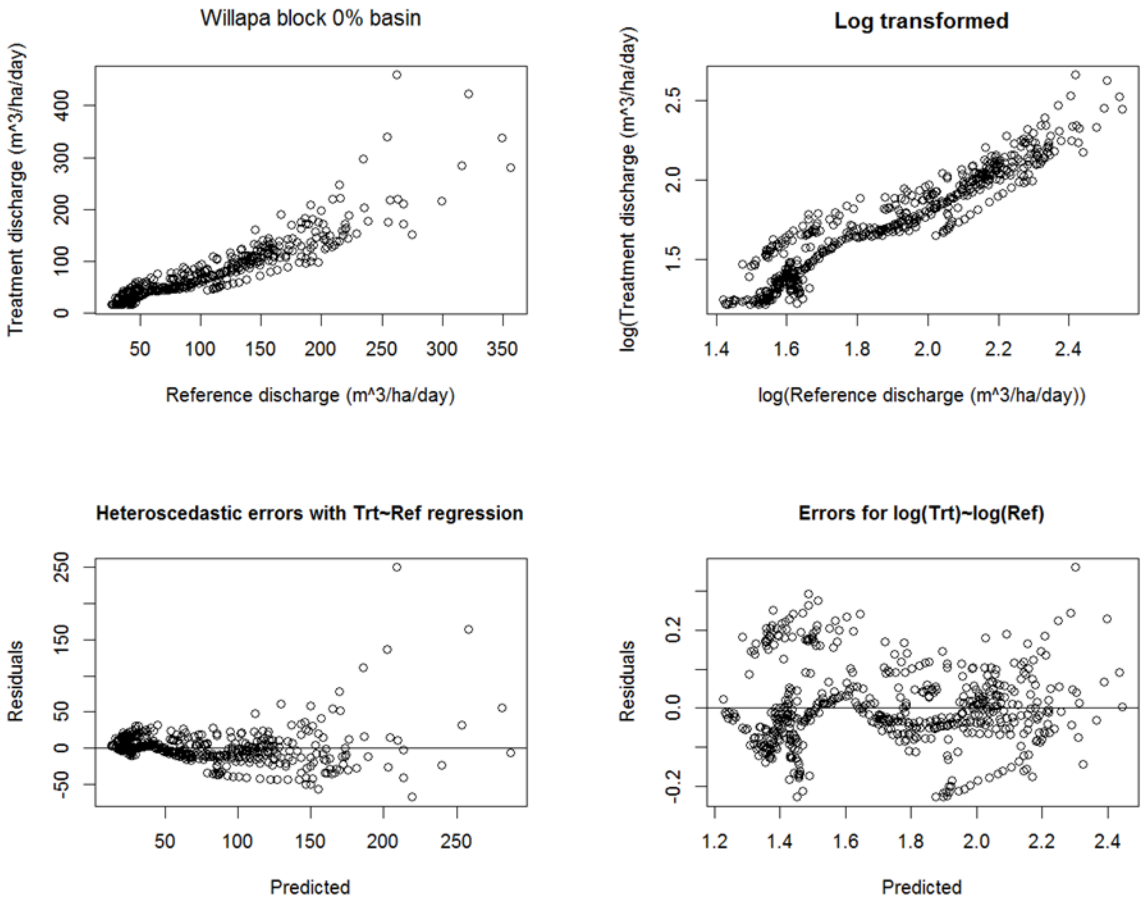

Figure 8-2. Regression between WIL1-REF and WIL-0% discharge in the pre-harvest period. Plots on the left show heteroscedasticity in the data (top left) and errors (bottom left), which were corrected by log-transforming the data (top right). The bottom right illustrates the relatively constant error variance obtained using log-transformed discharge.

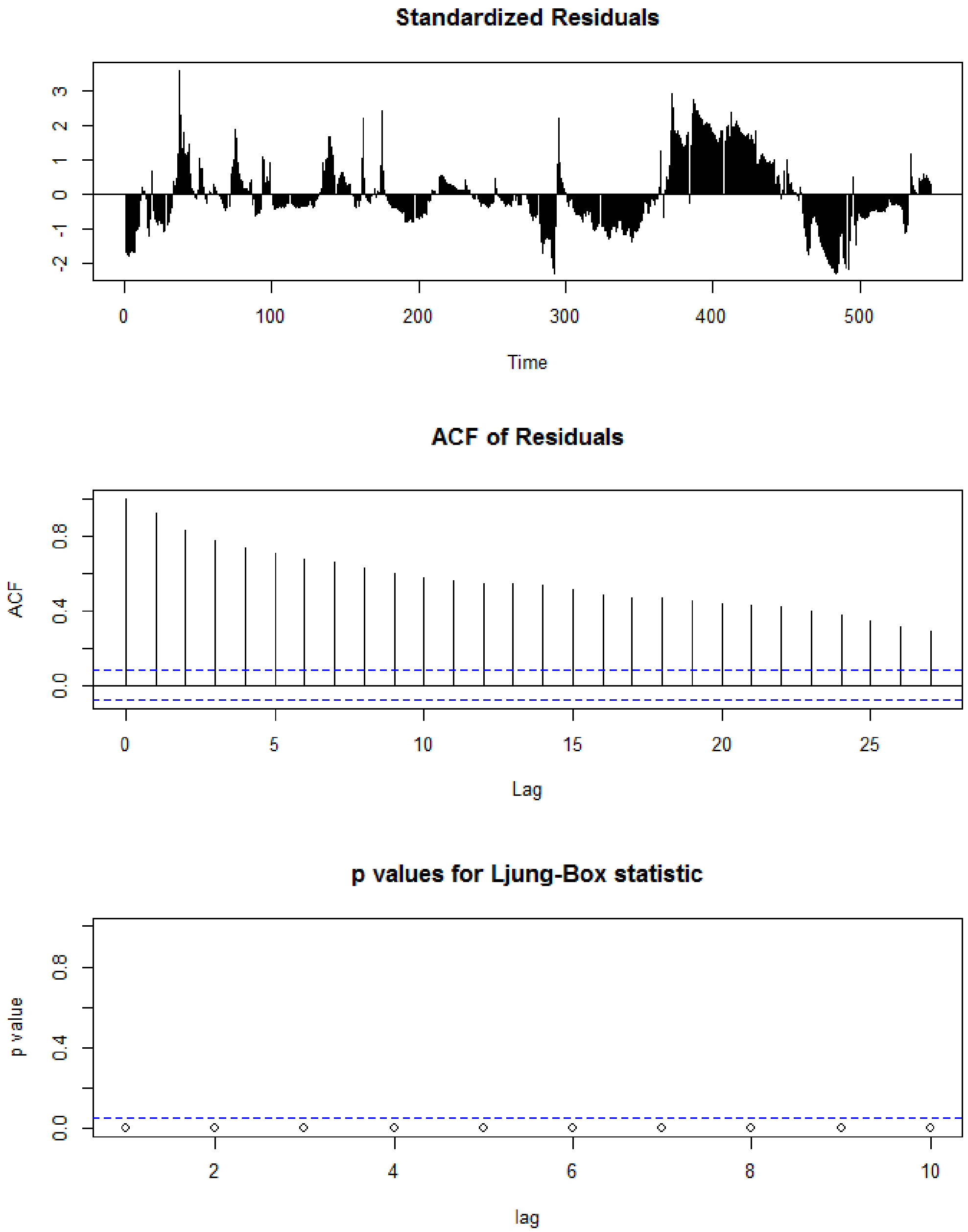

Figure 8-3. Time series diagnostics showing significant serial autocorrelation in the residuals from an OLS regression between reference and buffer treatment site discharge in the WIL1-0%. The top plot shows that the residuals (errors) are serially correlated (e.g., positive errors are followed by positive errors). The middle plot shows that there is high correlation not just to the preceding day (lag 1) but out to 25 days prior. The bottom plot confirms that the autocorrelation in lags 1–10 is significant at α=0.05.

To account for the serial autocorrelation in daily discharge, we used GLS for all our regression analyses. GLS allows for specified correlation structures that model the dependence among observations, which create unbiased estimates based on independent errors (Pinheiro and Bates 2000). Model parameters estimated by GLS are generally similar to those produced through ordinary least squares, but GLS estimators are preferred because they have smaller standard errors (Cowpertwait and Metcalfe 2009). The correlation structure we used was an autoregressive moving average (ARMA) model. ARMA models combine an autoregressive (AR) model where the current observation is expressed as a linear function of previous observations plus a homoscedastic white noise (i.e., stationary random noise) term:

$$\varepsilon_t = \phi_1 \varepsilon_{t-1} + \cdots + \phi_p \varepsilon_{t-p} + a_t \tag{8-4}$$

where: 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;

with a moving average (MA) model where the error in the current observation is expressed as a series of correlated noise terms:

$$\varepsilon_t = \theta_1 a_{t-1} + \cdots + \theta_q a_{t-q} + a_t \tag{8-5}$$

where: a_{t-q} is the noise term q days before, and

θ_q is the correlation coefficient at lag q.

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 as:

$$\varepsilon_t = \sum_{i=1}^{p} \phi_i \varepsilon_{t-1} + \sum_{j=1}^{q} \theta_j a_{t-j} + a_t \tag{8-6}$$

We used guidance from Venables and Ripley (2002) and Zurr and colleagues (2009) for determining the number of lags to include in our ARMA model. We tested a series of models using combinations of lags ranging from 0 to 6 for p and 0 to 3 for q, and chose the model with the lowest Akaike's information criterion (AIC) scores. AIC is a measure of the relative quality of a statistical model to a dataset and provides an unbiased means for choosing the best statistical model. The analysis was performed 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).

Given the regression model described by **Equations 8-3** and **8-6**, we evaluated whether the non-linear (${x_t}^2$) and seasonal terms were needed in the model using AIC scores. We chose the model that was most parsimonious with the data (i.e., the one with the lowest AIC score). We used GLS diagnostic plots to verify that the GLS errors in the final model were approximately independent and normally distributed ($\varepsilon \sim \mathrm{N}(0, \sigma^2, \mathrm{I})$; **Figure 8-4**).

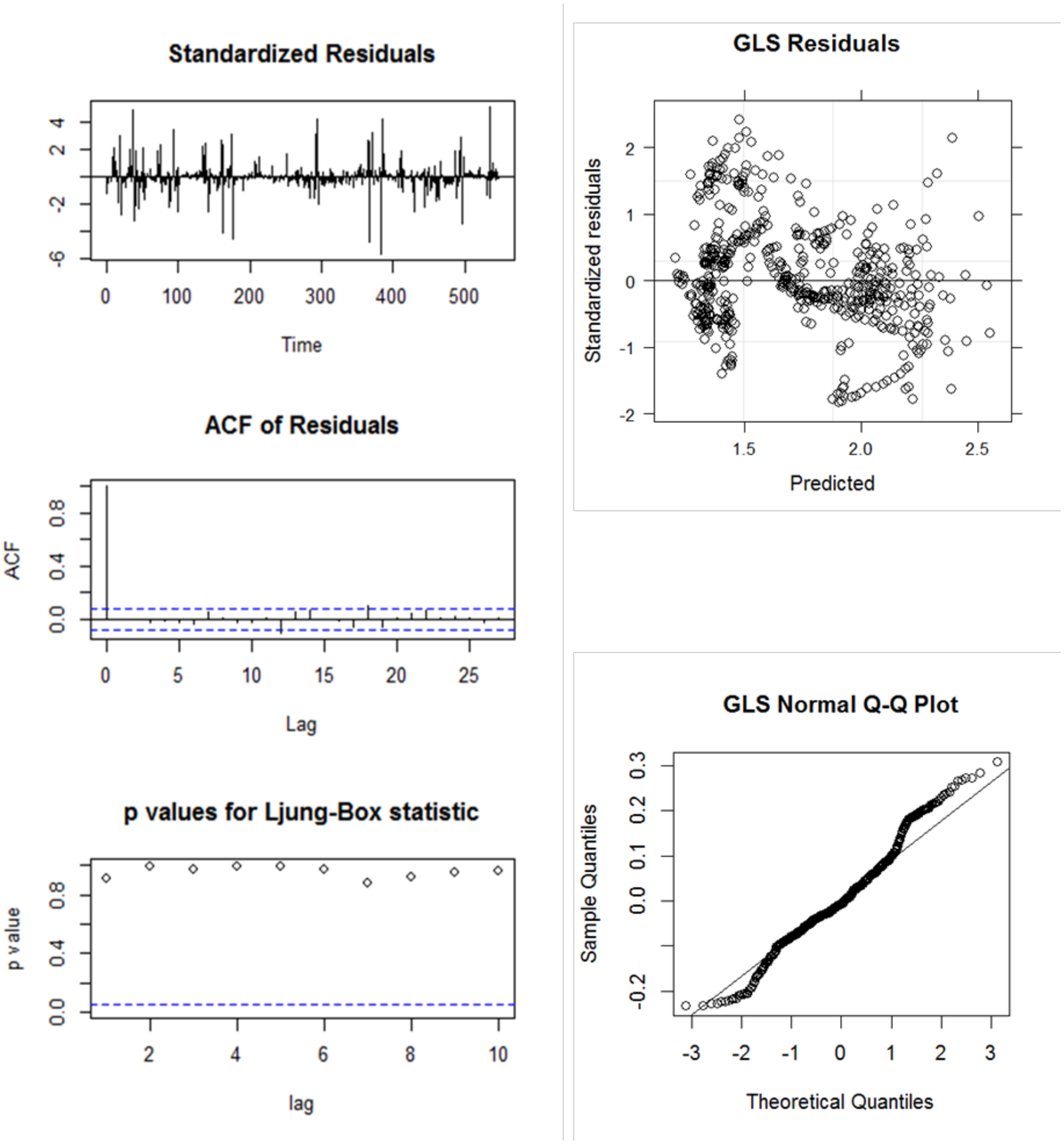

Figure 8-4. GLS regression diagnostic plot for the analysis of discharge in the WIL1-0% using the combined ARMA model. (**Equation 8-6**). The plots on the left are the same as shown in **Figure 8-2** and confirm that the residual autocorrelation has been removed, the top right shows that there is no pattern in the residuals (e.g., no heteroscedasticity), and the bottom right shows that the residuals are approximately normally distributed (e.g., $\varepsilon \sim \mathrm{N}(0, \sigma^2, \mathrm{I})$).

The square of the correlation coefficient (r^2) describes the proportion of the treatment variance that is explained by an OLS regression model. Since the standard calculation of r^2 is not appropriate to GLS regression, we estimated a coefficient of determination (R^2) based on likelihood-ratios:

$$R^2{}_{LR} = 1 - \exp({}^{-2}/_{n} * (\log Lik(x) - \log Lik(0))) \tag{8-7}$$

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 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 reflects the adequacy of the prediction model.[1] The R^2 averaged 0.8 (0.72–0.92) suggesting there was a fairly good fit between reference and treatment discharge in the pre-harvest period.

8-4.3. PREDICTING TREATMENT BASIN DISCHARGE IN THE POST-HARVEST PERIOD

The *predict* function in R was used to create unbiased estimates of what would have occurred in the buffer treatment site in the absence of harvest ($\hat{y}_t$) based on pre-harvest regression model parameters for each site and the two years of post-harvest daily discharge measurements at the block reference site (x_t).

8-4.4. DETERMINING EFFECT SIZE

The buffer treatment effect size was determined through a comparison of predictions and observations in the post-harvest period. Effect size was evaluated in terms of the average change in discharge (i.e., ANOVA framework as prescribed by the study design) for the two-year post-harvest period, a cumulative time series over the duration of the study, and change in flow frequency for a two-year post-harvest period.

8-4.4.1. Estimating the mean treatment effect

Treatment effects were calculated as the difference between the log-transformed observations from the treatment sites (log(y)) and what was expected to have occurred based on the pre-harvest relationship between reference and buffer treatment site ($\hat{y}_t$). Under the null hypothesis, the expected value of the difference is approximately zero [i.e., $\log(y) - \hat{y} \cong 0$].

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[1] 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 (a_t; Watson *et al.* 2001, Gomi *et al.* 2006).

A one-sided t-test was used to determine the statistical significance of the mean treatment effect (β_0). The t-test was conducted with GLS using an ARMA structure determined through AIC-based model selection with values of p ranging from 0 to 6 and q from 0 to 3[2]:

$$\log(y_t) - \hat{y}_t = \beta_0 + \varepsilon_t \tag{8-8}$$

where: log(y_t) is the log-transformed observed value,

$\hat{y}_t$ is the log-scale predicted value based on the pre-harvest regression,

β_0 is the intercept describing the mean treatment effect, and

ε_t are the errors modeled using **Equation 8-6**.

The mean buffer treatment effects (relative to the reference site) were calculated by back-transforming the intercept to the original scale:

$$\%\Delta = (10^{\beta_0} - 1) * 100 \tag{8-9}$$

where: $\%\Delta$ is the percent change[3].

We used time series analysis to verify that the model errors (ε_t) were approximately independent. In almost all cases, histograms and normal Q-Q plots of the residuals suggested that the errors were symmetric around zero but had heavier tails than would be found in a normal distribution (**Figure 8-5**). Heavy tails would tend to inflate standard errors making the P-values conservative (Pinheiro and Bates 2000).

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[2] This second fitting of the ARMA structure is the primary advantage of analyzing in a post-harvest t-test as opposed to looking for a significant treatment * period interaction in traditional ANCOVA and conducting post-hoc tests. Under the null hypothesis, the ARMA coefficients would be expected to remain the same, but under the alternative hypothesis where there is a change, the ARMA coefficients would be expected to change because of differences in scale. The second ARMA fit assures that the most appropriate ARMA coefficients are used in modeling the errors in each time period.

[3] Treatment effects are reported in terms of relative change because a constant difference on the log-scale is a constant ratio on the original scale (i.e., log(a) − log(b) = log(a/b)).

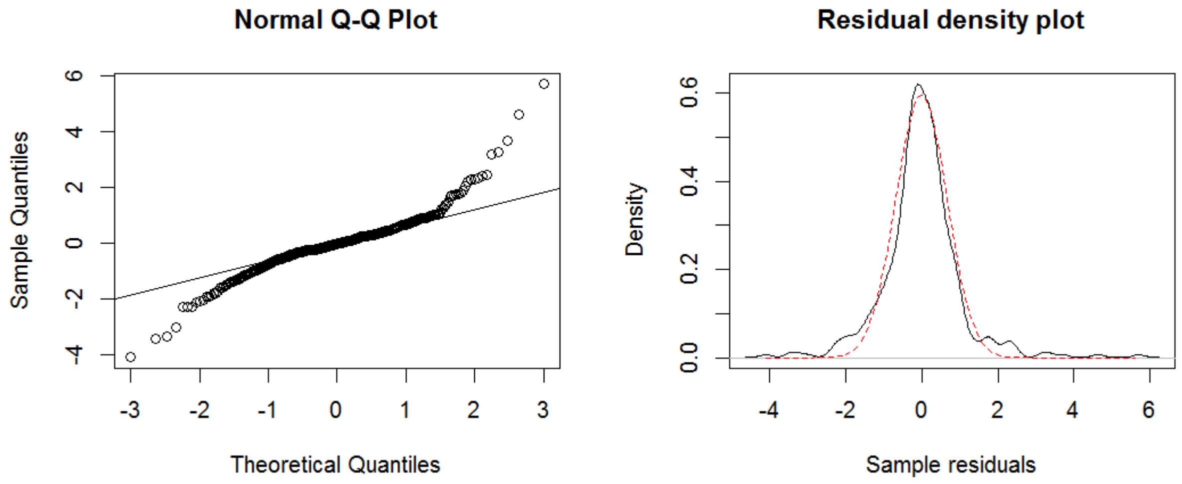

Figure 8-5. Residuals from the one-sample t-test for the first year post-harvest at WIL1-0% in a normal Q-Q plot and compared against a normal distribution (dashed red line) in a residual density plot. Heavier than normal tails tend to inflate estimated standard errors leading to more conservative (i.e., wider and bigger) confidence intervals and P-values.

8-4.4.2. Cumulative Water Yield

For the cumulative discharge analysis, we applied the regression equation to the entire set of reference basin observations to create a complete time series of expected values for a site. We then cumulatively summed the expected and observed time series so that the cumulative discharge on day *i* is the sum of the discharge on the preceding days. The discharge is normalized to basin area (i.e., m^3 ha^{-1}), the data are presented as a complete time series, and the units are in millimeters of water yield.

8-4.4.3. Change in Return Interval / Frequency

Statistical tests based on a single annual flow metric (e.g., mean) can be misleading because they ignore seasonal changes as well as the relationship between flow magnitude and frequency (Brown *et al.* 2005; Alila *et al.* 2009). Another framework for making statistical comparisons between observed and expected discharge is frequency pairing. Others have noted that frequency pairing is the only way to properly, or fully, isolate the effects of harvest on the magnitude of the flow (Alila *et al.* 2009, 2010; Green and Alila 2012; Kuras *et al.* 2012). For this analysis, we used the Frequency Pairing approach as described by Alila and colleagues (2009) and Green and Alila (2012). This framework is still dependent on chronological pairing for deriving the expected condition, but it allows us to evaluate the entire post-harvest time-series and evaluate the magnitude of change as a function of recurrence interval and to estimate changes in event frequency, both of which are important for evaluating ecological impacts.

The Frequency Pairing approach is based on the regression analysis of chronologically paired pre-harvest data and use of that regression equation and post-harvest discharge observations in the reference basin. Where this approach differs from the traditional ANOVA/ANCOVA is that the observed and expected values are paired based on their historic return period (frequency pairing) as opposed to comparing expected versus observed values at a given point in time (chronological pairing).

In frequency pairing, 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):

$$T_{days} = \frac{n+0.2}{m_i-0.4} \tag{8-10}$$

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.

Equation 8-10 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).

As explained by Alila and colleagues (2009; Kuras *et al.* 2012), the predicted values ($\hat{y}_t$) are corrected for loss of variance resulting from the regression analysis. If unaccounted for, the loss of variance could inflate estimates of treatment effects in the tails of the frequency distribution. The correction is performed in a three step Monte Carlo simulation. The first step is the addition of a random error (e_t) sampled from a t-distribution 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:

$$e_t = \sigma_{pred_t} * rt(\mathrm{df}_{resid} - 1) \tag{8-11}$$

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:

$$\sigma_{pred_t} = rse\sqrt{1 + \frac{1}{n_{pre}} + \frac{(x_t - \bar{x})^2}{\sum(x_t - \bar{x})^2}} \qquad (8\text{-}12)$$

where: rse is the residual standard error from the initial regression,
n_{pre} is pre-harvest sample size,
x_t is the reference site value on post-harvest day t, and
$\bar{x}$ is the mean of the post-harvest reference site 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:

$$95\%CI = qnorm((0.025, 0.975), \bar{\tilde{y}}_i, \sqrt{var(\tilde{y})_i + var(GEV)_i}) \qquad (8\text{-}13)$$

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.

8-5. RESULTS

8-5.1. MEAN CHANGE IN DISCHARGE

Mean daily specific discharge (m^3/ha/day) increased by an average of 59% (27–108%) and 56% (38–66%) in the FP and 0% treatments, respectively (**Table 8-2**; **Figure 8-6**). The average discharge in the 100% treatments decreased by approximately 2% (−20 to 17%) though the difference in the 100% treatment was not statistically significant at $\alpha = 0.05$ over the two-year post-harvest period.[4]

The factors that best explain mean change in relative discharge (%) are the proportion of the basin that was harvested (**Table 8-1**) and the specific discharge (m^3/ha/yr) during the pre-harvest period. As expected, the basins that were harvested entirely exhibited the greatest changes in specific discharge, and there appears to be a threshold for the amount of harvest that is required to produce measurable changes in mean discharge. Pre-harvest discharge also affects

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[4] Interpretations based on P-values and confidence intervals are consistent with significance for treatment × period interactions in traditional ANCOVA, but are expected to provide more accurate estimates of effect size as it increases because the post-harvest auto-regressive correlation structure is based on post-harvest data only.

proportional change because a fixed change in volume has a greater proportional effect in sites with low pre-harvest water yield.

In the case of this study, the Olympic block sites had higher specific discharges than sites in the Willapa 1 block. In the Olympic block, pre-harvest runoff (mm/yr) was 4,630, 2,684, and 3,944 mm/yr in the 100%, FP, and 0% treatments, respectively. In the Willapa 1 block, pre-harvest runoff was 1,937, 2,025, and 1,863 mm/yr in the 100%, FP, and 0% treatments, respectively.

Table 8-2. Change in mean daily specific discharge (%ΔQ) with 95% confidence intervals by post-harvest year, treatment and block, along with the median expected and observed annual specific discharge.

Period	Treatment	Block	P-value	$\overline{\%\Delta Q}$	%ΔQ 95% CI	Median Expected (mm/yr)	Median Observed (mm/yr)
Year 1	100%	OLYM	0.07	−20%	(−37 to 2%)	1,741	1,098
		WIL1	0.73	+12%	(−40 to 108%)	1,179	1,433
	FP	OLYM	0.01	+27%	(6–52%)	1,288	1,693
		WIL1	<0.001	+65%	(26–118%)	1,060	2,202
	0%	OLYM	0.002	+38%	(12–70%)	3,260	4,431
		WIL1	<0.001	+61%	(24–108%)	992	1,770
Year 2	100%	OLYM	0.18	−11%	(−26 to 6%)	2,428	1,960
		WIL1	0.06	+17%	(−1 to 37%)	1,632	2,031
	FP	OLYM	0.001	+37%	(13–66%)	2,129	2,485
		WIL1	<0.001	+108%	(67–158%)	1,209	2,557
	0%	OLYM	<0.001	+60%	(29–98%)	3,349	5,388
		WIL1	<0.001	+66%	(29–112%)	1,185	2,292

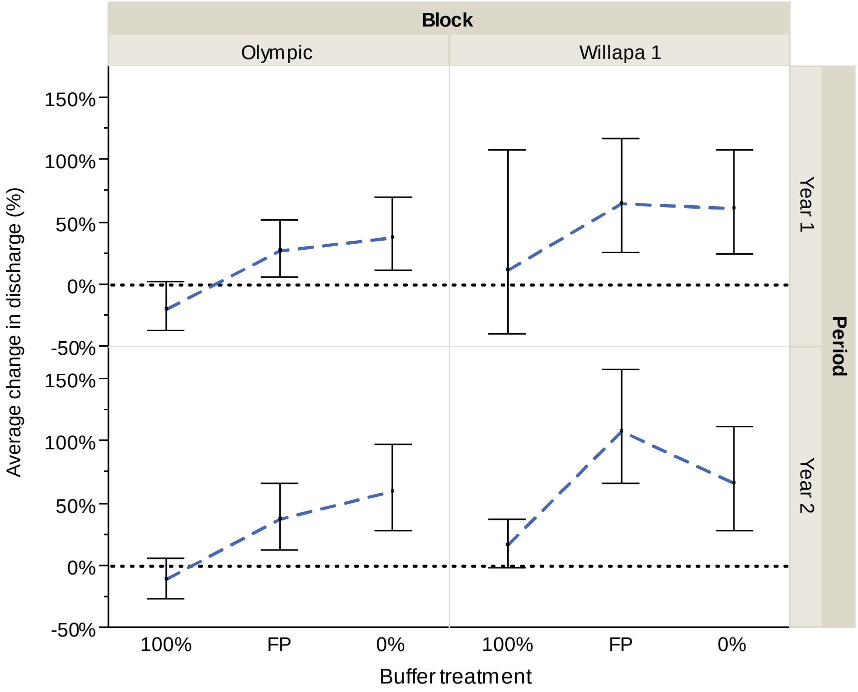

Figure 8-6. Relative change in mean discharge (%ΔQ) with 95% confidence intervals (CI) for the first (Year 1) and second (Year 2) years post-harvest, by treatment and block.

8-5.2. CHANGE IN CUMMULATIVE WATER YIELD

Over the course of the study, annual water yield in WIL1-REF ranged from 2,372 to 2,997 mm/year while the OLYM-REF ranged from 3,118 to 4,323 mm/yr. Water yield in the post-harvest was greater in all buffer treatment sites. In the post-harvest period, annual runoff was greater by 228 mm (12%), 725 mm (36%), and 690 mm (37%) in the WIL1-100%, WIL1-FP, and the WIL1-0%, respectively. In the OLYM block, annual runoff was greater by 702 mm (15%), 491 mm (18%), and 2,113 mm (50%) in the 100%, FP, and 0% sites in the post-harvest period.

Differences in observed water yield between the pre- and post-harvest period combine treatment effects and natural variability in hydrologic conditions through time. To assess the treatment effect alone, we compared the difference between observed and expected discharge in the post-harvest period (**Table 8-3**). Over the two-year post-harvest period, all treatment basins had higher water yields, with the OLYM-100% treatment basin having the smallest absolute and relative increase (42 mm/yr or +1%) and the OLYM-0% basin having the greatest absolute change in yield (2,138 mm/yr or +55%; **Table 8-4**). The FP treatments exhibited increases in total yield that were closer to the 0% treatment than the 100% treatment (**Figures 8-7** and **8-8**).

Table 8-3. Change in water yield for the first two years following harvest.

Period	Treatment	Block	Total Expected (mm/yr)	Total Observed (mm/yr)	Treatment Effect (mm/yr)	Percent Change
Year 1	100%	OLYM	3,100	2,863	−237	−8%
		WIL1	1,517	1,919	402	26%
	FP	OLYM	1,899	2,799	900	47%
		WIL1	1,494	2,383	889	60%
	0%	OLYM	3,989	5,634	1,645	41%
		WIL1	1,491	2,381	890	60%
Year 2	100%	OLYM	3,111	3,431	320	10%
		WIL1	1,991	2,489	498	25%
	FP	OLYM	2,514	3,545	1,031	41%
		WIL1	1,840	3,106	1,266	69%
	0%	OLYM	3,847	6,479	2,632	68%
		WIL1	1,828	2,723	895	49%

Table 8-4. Average change in water yield for the first two years following harvest.

Treatment	Block	Average Expected Annual Yield (mm/yr)	Average Observed Annual Yield (mm/yr)	Two-Year Treatment Effect (mm/yr)	Average Change (%)	Increase Per % Basin Harvested (mm/yr)[1]
100%	OLYM	3,105	3,147	42	1%	1.0
	WIL1	1,754	2,204	450	26%	5.1
FP	OLYM	2,206	3,172	966	44%	11.0
	WIL1	1,667	2,745	1,078	65%	11.5
0%	OLYM	3,918	6,056	2,138	55%	21.4
	WIL1	1,660	2,552	892	54%	8.9

[1] Percent basin harvested shown in **Table 8-1**.

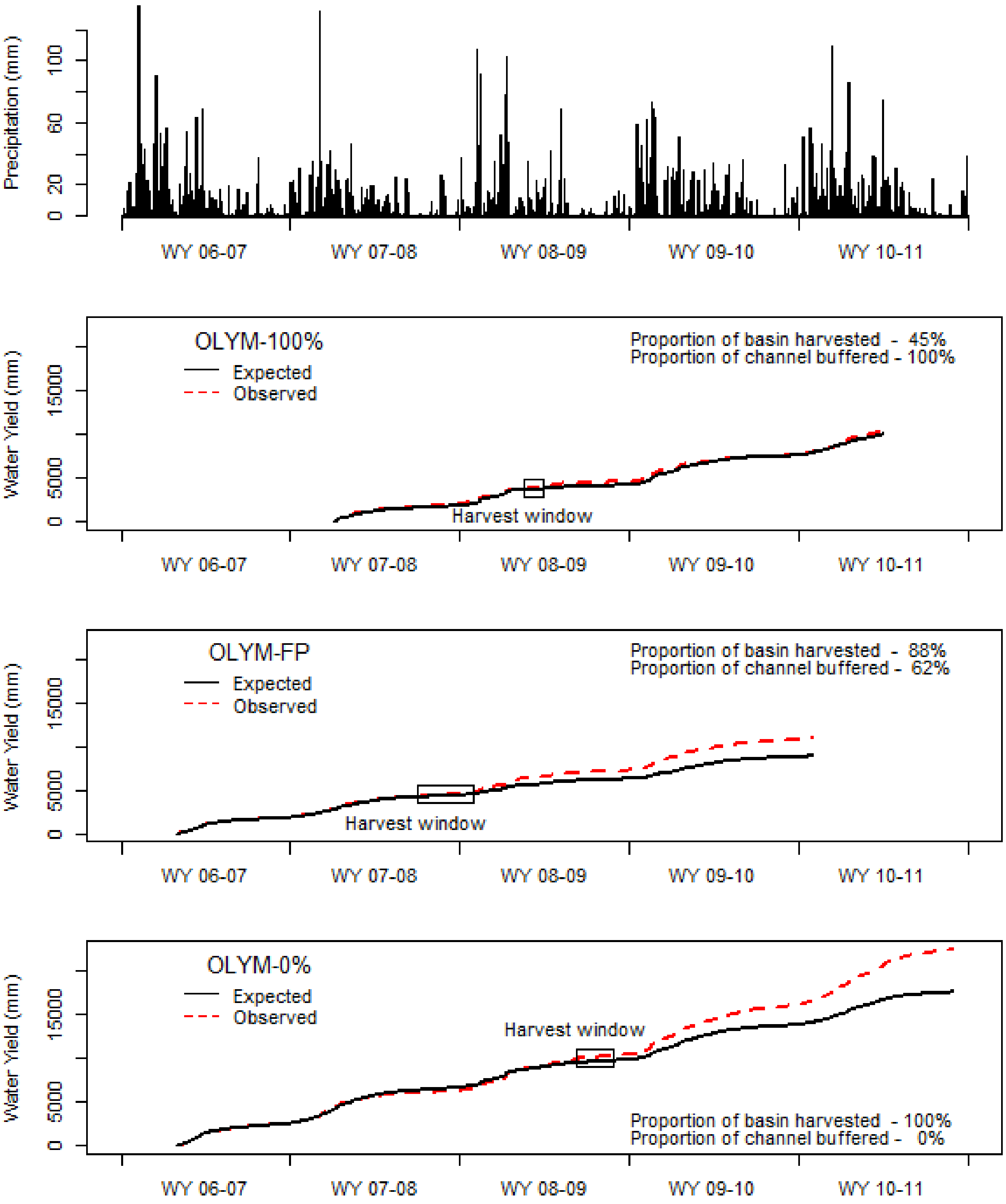

Figure 8-7. Precipitation at Aberdeen, Washington from 1 October 2006 to 30 September 2011 and cumulative water yield for the Olympic block sites by water year (WY).

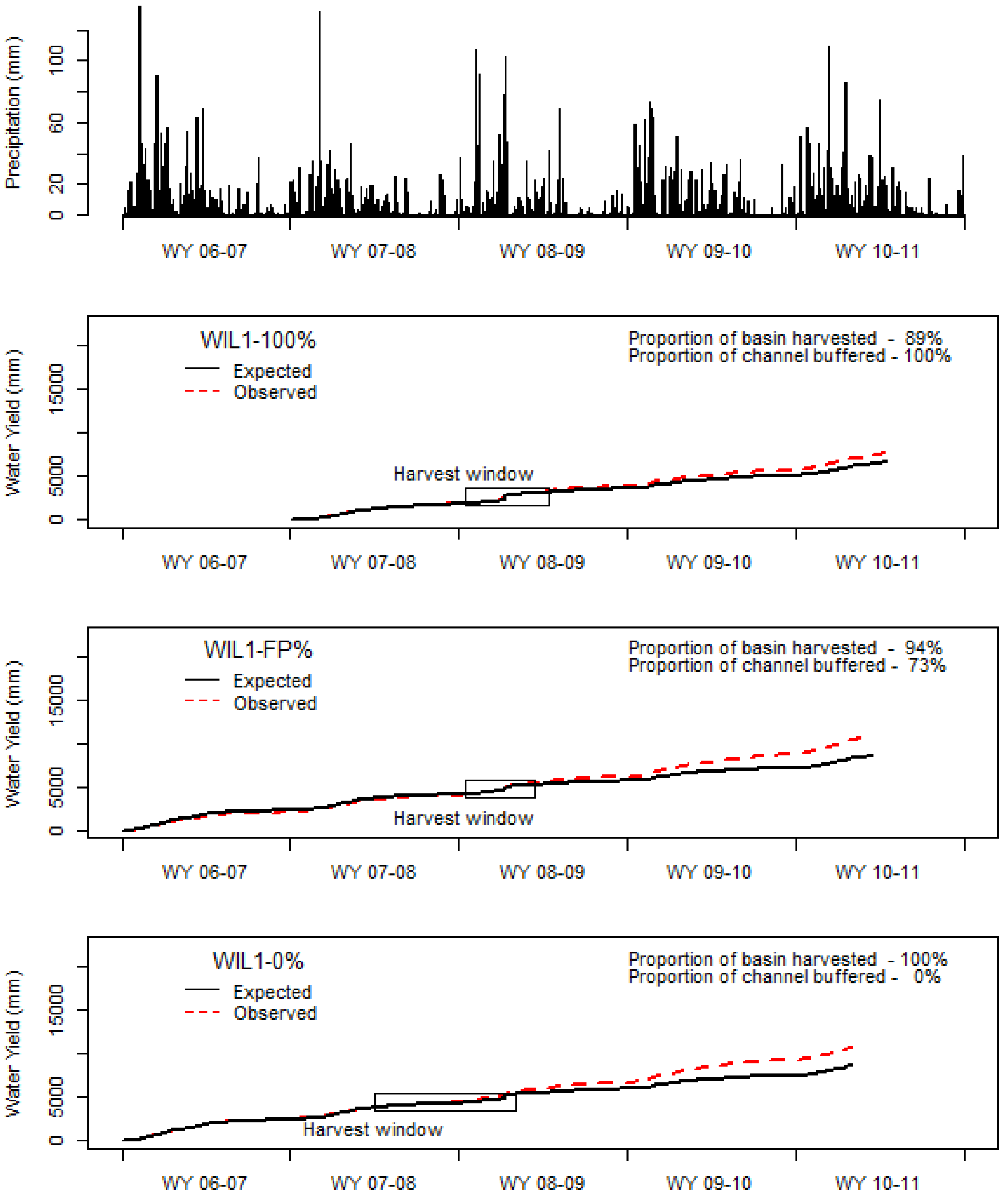

Figure 8-8. Precipitation at Aberdeen, Washington from October 1, 2006 to September 30, 2011 and cumulative water yield for the Willapa-1 block sites by water year (WY).

8-5.3. CHANGE IN FREQUENCY AND MAGNITUDE AS A FUNCTION OF RECURRENCE INTERVAL

Most basins exhibited large shifts in discharge frequency and magnitude for at least some portion of the two-year post-harvest period (**Figure 8-9**). For events with frequencies greater than the 2-day (i.e., median) but less than the 30-day recurrence interval (RI), there were significant shifts in the frequency at which a given discharge occurred. Shifts in frequency were large in all sites other than the OLYM-100%, with event frequency doubling in many cases (i.e., 7-day event occurs twice as often in the post-harvest period and becomes a 3.5-day event). Changes in frequency increased with the amount of harvest. The OLYM-100%, which had only 45% of the basin harvested, exhibited the least change in event frequency while basins that were 95–100% harvested (OLYM-0%, WIL1-FP, and WIL1-0%) exhibited large changes in event frequency.

In addition to the large changes in recurrence, frequency pairing also highlighted large changes in event magnitude. **Figure 8-9** shows that the entire distribution of discharge measurements is shifted towards higher flows in the 0% sites, while 100% sites exhibit increased flow variance but very little change in the mean. In the 0% sites, baseflow (i.e., RI <~2 day) increased by 1 to 2 mm, while the 100% sites experienced baseflow decreases of 0.5 to 2 mm (**Figure 8-10**).[5] For events with a recurrence interval between 2 and 7 days, all four FP and 0% sites exhibited statistically significant increases in specific discharge on the order of 1.5 to 7 mm, with greater changes in the sites where a higher proportion of the basin was harvested. All sites exhibited increased storm flow (~7–30 day RI), but plots of specific discharge in mm/day (**Figure 8-10**) highlight expanding confidence bounds for larger events and shows that stormflow increases were statistically significant in only half of the sites.

Interestingly, the 1.5 year recurrence interval (e.g., peak flow) response, which is responsible for most of the geomorphic work, differed between the Olympic and Willapa 1 block sites. Peak flows (e.g., >30 day RI) increased only in the two Olympic block sites (OLYM-FP and OLYM-0%) where a large proportion (88% and 100%, respectively) of the basin was harvested. In the OLYM-FP treatment, peak flow increased by a statistically significant 61%. In the OLYM-0%, peak flow increased by 23% but was within 95% confidence intervals. In the Willapa 1 block, the 1.5 year recurrence interval peak flows decreased from −10% and −3.6% (both within 95% CI) in the FP and 0% treatments, respectively.

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[5] In the OLYM-100%, stage was measured at a culvert rather than a flume; the flow vs. stage height relationship did not hold for very low stage height values (<~3 L s^{-1}), so discharge estimates below 3 L/s were not included in the dataset for that basin.

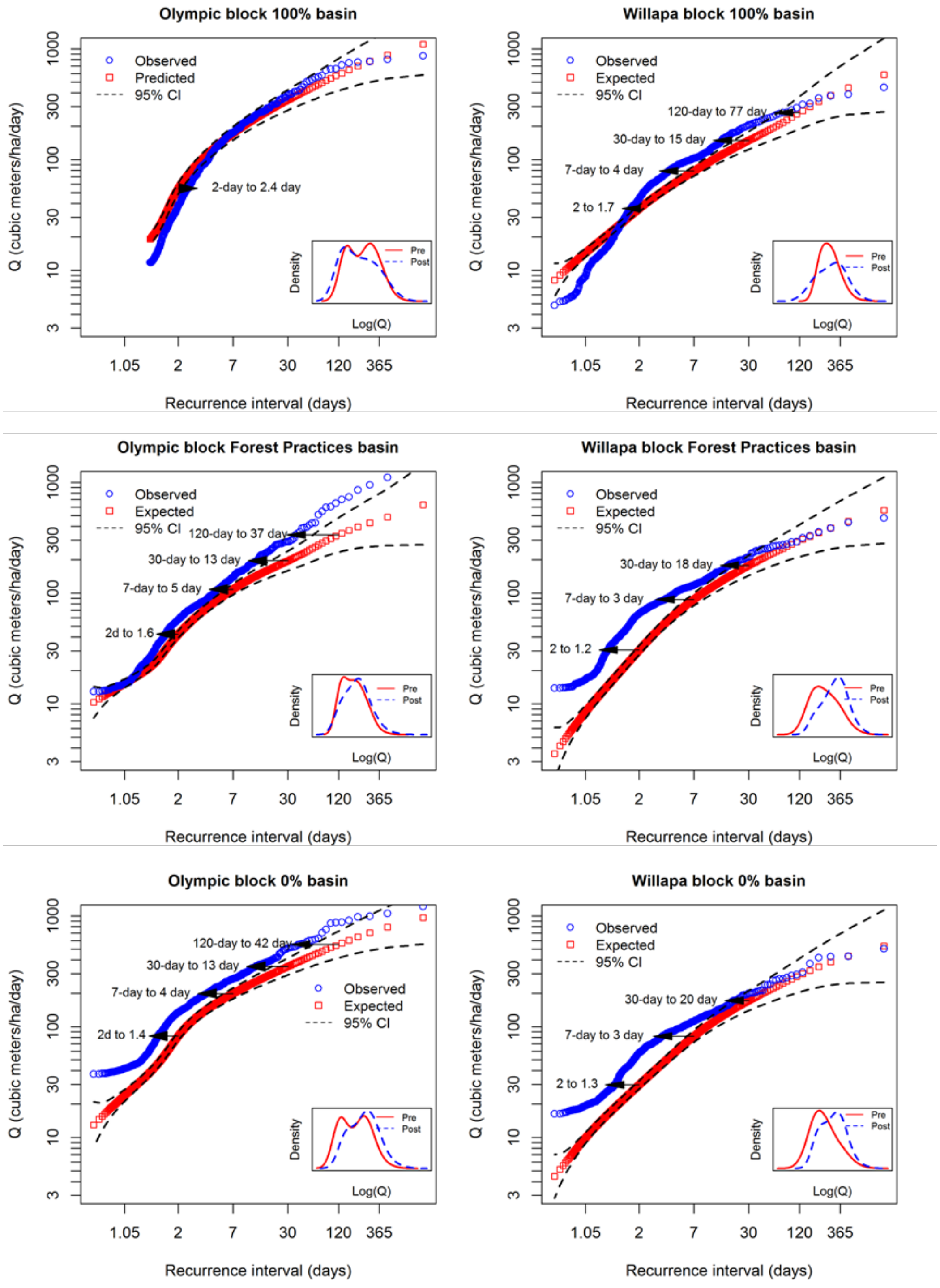

Figure 8-9. Discharge frequency curves for the two years post-harvest (730 days). Inset graphs show the distribution of log-transformed discharge, while the larger graphs shows discharge magnitude on a logarithmic axis as a function of recurrence interval (RI). Shifts in the frequency distribution that extend beyond the 95% confidence interval (CI) are displayed with arrows for 2, 7, 30, and 120-day events.

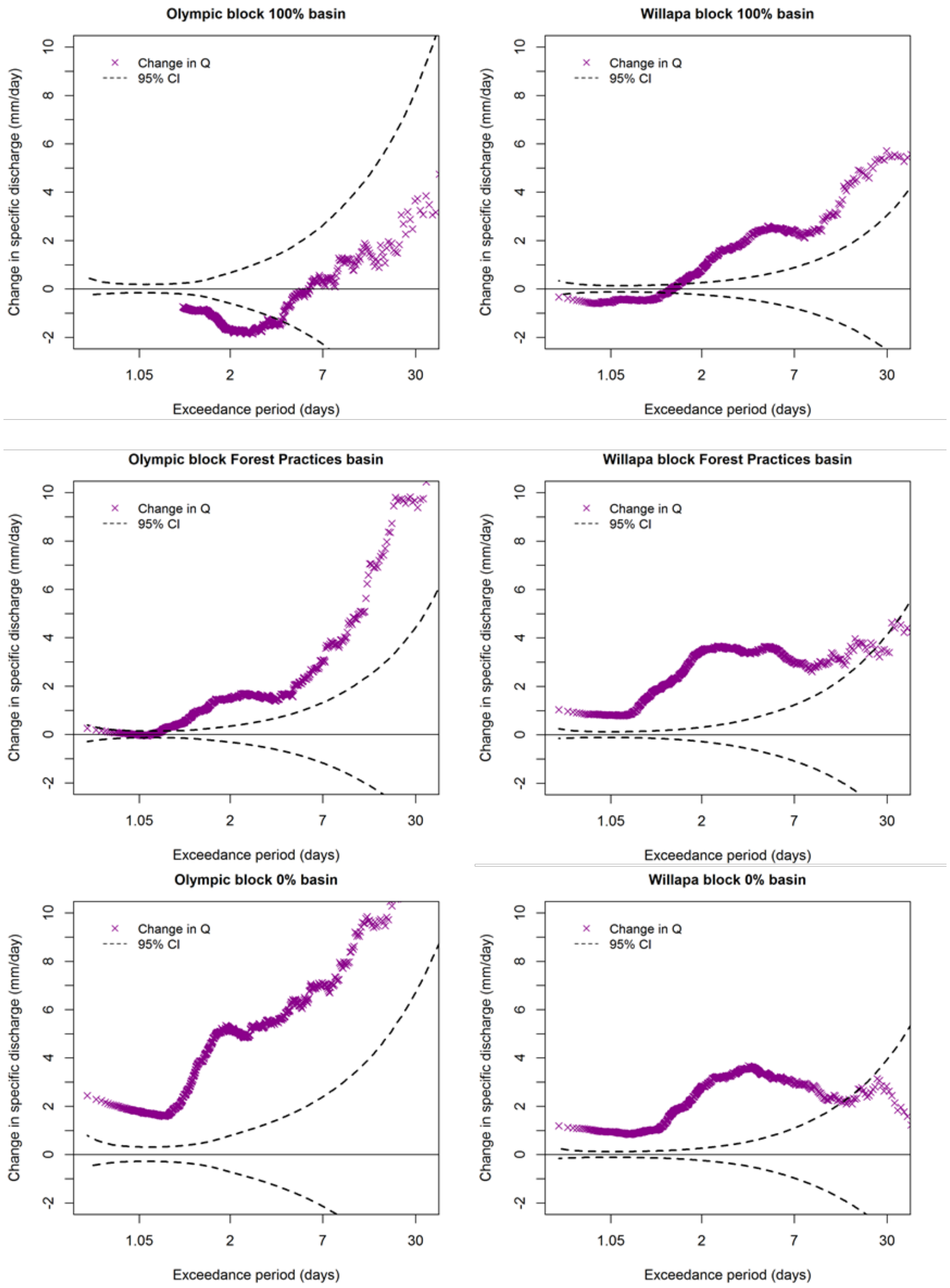

Figure 8-10. Absolute change in specific discharge as a function of the number of days that a discharge is exceeded. Confidence intervals (CI) become very large for events with an exceedance period of greater than 7 days so we restricted the analysis to recurrence intervals (RI) <30. In the 100% sites, discharge decreases for RIs <2 days and in most sites significant (i.e., beyond 95% CI) change occurs between the 2-day and 7-day RI.

8-6. DISCUSSION

Water yield typically increases following forest harvest, although the exact magnitude and timing of change is affected by a number of 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 *et al.* 1982; Harr 1983; Harr 1986; Harr and Fredriksen 1988), and 360 mm on Vancouver Island (Heatherington 1982). After 100% forest removal, paired watershed studies have reported changes of −2 to 8 mm/day with strong seasonal variation in the response (Jones and Post 2004), though it is generally accepted that in rain-dominated areas, annual runoff can increase by as much as 6 mm/yr for each percent of the basin harvested (Moore and Wondzell 2005). Results of this study are generally consistent with previous research. In the rainforests of the Olympic mountains, water yields increased from 42 to 2,138 mm/yr in basins that were 45% to 100% clearcut, respectively. In the Willapa Hills block, water yields increased by 450 to 1,078 mm/yr in basins that were 89% to 100% clearcut. Annual runoff in the two year post-harvest period increased from 1 to 21 mm/yr for each percent of the watershed that was harvested, although there was strong variation in response as a function of buffer treatment and recurrence interval. Means tests indicated that complete basin clearcutting (0% buffer) and clearcutting with Forest Practices buffers increased mean daily specific discharge (m^3/ha/day) by an average of 56% and 59%, respectively ($P < 0.002$). Total water yield changed very little in response to 45% basin harvest (OLYM-100%), and neither of the basins with 100% buffering of the perennial channel exhibited statistically significant changes in mean annual discharge at alpha=0.05.

Although neither of the 100% treatment sites exhibited a statistically significant change in mean discharge, all treatments exhibited statistically significant changes in magnitude and frequency of daily discharge events in the first two years following harvest. In the 100% treatment sites, the magnitude of moderately high flows (e.g., 7–120 day RI) increased following harvest while the magnitude of base flows (RI <2) decreased. Changes in the tails of the distribution are important because increased high magnitude events have the potential to increase sediment transport and yield, while decreases in the magnitude or frequency of summer base flow alters aquatic habitat availability and may result in increased summer stream temperatures (Johnson and Jones 2000; Gomi *et al.* 2005).

In the FP and 0% buffer treatment sites, the most consistent change was a statistically significant increase (1.5–7 mm) in specific discharge for events with a recurrence interval between 2 and 7 days. These frequencies are most likely to be associated with moderate intensity winter rainfall events when the combined effects of changes in evapotranspiration and interception are likely to be at their greatest. Base flow (RI <2 days) in the three sites where greater than 94% of the basin was harvested also increased (1–3 mm), but the OLYM-FP where only 88% of the basin was harvested exhibited only slight increases. Interestingly, the 1.5 year recurrence interval (e.g., peak flow) response, which is responsible for most of the geomorphic work, differed between the Olympic and Willapa block sites with the OLYM-FP and OLYM-100% showing peak flow increases and WIL1-FP and WIL1-100% showing slight decreases. Changes in event magnitude significantly affected event return frequency across all sites, with frequencies doubling in many cases (i.e., 7-day event becoming a 3.5-day event) for events ranging from the 2- to 120-day RI.

The discharge monitoring reported here was performed as part of a much larger study designed to examine a wide range of changes associated with forest practices and discharge was not a focal aspect of this study. For this and other reasons, local meteorology and other hydrologic data were not collected, and the lack of ancillary information severely restricted our ability to infer causal relationships for the changes that were observed. While most changes were consistent with our expectations, the observed base flow decreases in the 100% treatment sites and differential basins were not expected and therefore merit a brief discussion.

In the two 100% treatment basins with complete buffering of the perennial channel, base flows (RI <2 day) decreased following harvest even though 45% and 89% of the basin area was clearcut harvested. While the true cause remains unknown, 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 (ET) can be spatially restricted to riparian areas accounting for 6–18% of the total ET in a headwater basin (Tsang *et al.* 2014). Thus, it is possible that riparian plants were light-limited prior to harvest and that the increased light availability associated with adjacent harvest increased ET enough to decrease streamflow during relatively dry periods, when soil water is depleted and stream discharge is low.

The peak flow response differed significantly between blocks, with the OLYM-FP and OLYM-0% showing increases while the sites in the Willapa 1 block did not. One possible explanation for the difference in peak flow response among sites is elevation and the associated likelihood of rain-on-snow. The OLYM-FP lies between 277 to 445 m while the OLYM-0% lies between 243 and 481 m. In contrast, the WIL1-FP lies from 185 to 407 m and the WIL1-0% lies from only 87 to 225 m. Rain-on-snow is most likely to affect runoff in the elevation band known as the transient snow zone (roughly 300–900 m) given the greater frequency of both snowfall and warm rainstorms that occur in that zone (Harr 1981; Berris and Harr 1987). Clearcuts accumulate more snow due to the decreased canopy interception (Storck *et al.* 2002) and peak discharge in the transient snow zone is likely to be a function of both precipitation and snowmelt (Harr 1986; Marks *et al.* 1998; Jennings and Jones 2015). Thus differences in peak flow runoff response between Olympic and Willapa sites may be explained by a greater frequency of rain-on-snow events in the Olympic block sites.

8-7. CONCLUSIONS

As has been shown previously, water yield increased following clearcut harvest of a large proportion of the basin. The magnitude of change varied with several factors including the proportion of the basin harvested above some threshold (Stednick 1996). Basins that received the 100% treatment, with buffering of the entire perennial network, had lower baseflows, which partially offset increases in stormflow. In most treatment sites, water yield increased significantly for moderate storm events (2- to 7-day RI) whereas the change in yield for large storm events (e.g., 30-day RI) was more variable. Because discharge was not a focus of the study, data allowing detailed analysis of causal mechanisms were not collected. Limitations include a lack of information on local meteorology, which would have allowed for a more detailed analysis, including hydrologic mass balance. However, the literature suggests that the base flow reduction in the 100% treatment sites may have been the result of increased evapotranspiration in the riparian corridor, while peak flow responses in the OLYM-FP and OLYM-0% may be associated with a greater likelihood of rain-on-snow events.

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CHAPTER 9 - NUTRIENT EXPORT

William Ehinger and Stephanie Estrella

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9-1. ABSTRACT

The purpose of this study was to estimate the effect of timber harvest on the quantity of instream nitrogen and phosphorus exported from headwater streams. We sampled eight non-fish-bearing stream catchments, ranging in size from 11.8–44.3 ha, distributed in two blocks of four streams each: 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, each with one of three different riparian buffer treatments: 100% of the stream length buffered with a 50-ft (15.2-m) wide, two-sided buffer (100% treatment); at least 50% of the stream length buffered with a 50-ft (15.2-m) wide, two-sided buffer (Forest Practices or FP treatment), or no buffer (0% treatment). We measured stream discharge and collected water samples for the analysis of nitrogen (N) and phosphorus (P) concentration from October 2006 to September 2011. Nitrogen and P concentrations were determined from unfiltered water samples so that our nutrient export estimates included both particulate and dissolved fractions.

We found greater variability in pre-harvest total N (total-N) and nitrate nitrogen (nitrate-N) concentration among the study sites than expected. Pre-harvest N exports ranged from 1.78 to 14.42 kg ha^{-1} yr^{-1} for total-N and 1.70 to 13.32 kg ha^{-1} yr^{-1} 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 8.16 to 32.86 kg ha^{-1} yr^{-1} (7 to 358% increase) for total-N and 6.26 to 29.97 kg ha^{-1} yr^{-1} (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. Although the difference between the 100% and FP treatments and the difference between the FP and 0% treatments were not significant ($P < 0.05$), the relative magnitude of the changes was consistent with our expectations of increased N export with an increase in the proportion of the watershed harvested. Nitrogen export was also correlated with the increase in annual runoff, which was correlated with the proportion of the watershed harvested. In contrast to N, total-P concentration did not change post-harvest. Pre-harvest total-P export ranged from 0.08 to 0.35 kg ha^{-1} yr^{-1} and increased to 0.11 to 0.46 kg ha^{-1} yr^{-1} (21 to 50% increase), but did not vary significantly ($P > 0.05$) among treatments. The increase was likely a function of the low pre-harvest exports and increased runoff post-harvest because very little bank disturbance was observed at any of the sites, and there was little evidence of sediment delivery to the channel.

9-2. INTRODUCTION

Nutrient export from streams draining into Puget Sound and coastal estuaries is of special interest to state environmental regulatory authorities because excess nutrient loads can encourage high primary production in receiving waters, which accumulates as algae biomass. When this biomass dies, its decomposition may depress dissolved oxygen concentration in the bottom waters of Puget Sound (Roberts *et al*. 2008). Mohamedali and colleagues (2011) estimated that although anthropogenic, non-point source dissolved inorganic nitrogen (N) loads account for only 18% of the total loading from rivers into Puget Sound, it can account for up to 65% of the load in some subbasins. Much of the land draining into Puget Sound, Willapa Bay, and Grays Harbor is forested and managed for timber production. Better estimates of the effect of contemporary forest harvest practices on N and phosphorus (P) loads will be useful for managing the quality of Washington's coastal waters.

Feller (2005) suggests that five factors can explain most of the effects of forest harvest on stream chemistry: (1) geological weathering; (2) precipitation/climate; (3) terrestrial biological and physical processes; (4) physical/chemical reactions in soils; and (5) processes within the aquatic ecosystem. Of these, the last three are the most important with respect to the effects on N and P concentrations in streams and subsequent export from the watershed.

Clearcut harvest and vegetation control can reduce canopy interception and evapotranspiration, and increase runoff and the capacity for leaching soluble nutrients from the soil (Likens *et al.* 1970; Bosch and Hewlett 1982; Harr 1983; Stednick 1996; Moore and Wondzell 2005). Vegetation removal can decrease dissolved inorganic N (DIN) uptake, resulting in higher nitrate-N concentrations in soil water and in the stream (Dahlgren 1998; Feller *et al.* 2000). Clearcutting can increase the growing season soil temperature and may increase the rate of microbial nitrification in the soil after harvest (Feller 2005). In addition, forest harvest can adversely affect soil mycorrhizas, at least temporarily (Harvey *et al.* 1980; Hagerman *et al.* 1999) potentially further decreasing the rate of DIN uptake and increasing the amount of DIN in the soil and available for leaching.

Increases in stream concentration of nitrate-N, especially during the first fall freshets, and increases in nitrate-N export have been reported post-harvest in numerous studies (Likens *et al.* 1970; Brown *et al.* 1973; Feller and Kimmins 1984; Harr and Fredricksen 1988). Generally, the higher the proportion of a watershed harvested, the greater the increase in soil nitrate-N (Feller *et al.* 2000), in concentrations of stream N (Stark 1979; Martin *et al.* 1984; Fowler *et al.* 1988; Tiedemann *et al.* 1988), and in concentrations of nitrate-N in soil water (Feller *et al.* 2000). Also, slash burning after harvest seems to increase nitrate-N concentration in streams more than does leaving slash on site to decompose naturally (Fredricksen 1971; Stark 1979; Feller and Kimmins 1984).

Unlike nitrate-N, P is readily adsorbed onto organic material and clay particles and is generally much less mobile in the soil than nitrate-N. Increases in P export after forest harvest are more likely to be the result of soil disturbance or erosion leading to sediment input to the channel.

Instream processing of both N and P can ameliorate the effects of higher instream concentrations due to disturbance. Higher instream primary productivity after canopy removal was suggested as the cause of lower stream concentrations of nitrate-N in headwater streams of southwest British Columbia, Canada (Kiffney *et al.* 2003) and in artificial stream channels (Triska *et al.* 1983). Artigas-Alejo (2008) suggested that heterotrophic nutrient uptake by bacteria and fungi, which can exceed algal biomass in shaded streams, could provide the same function. Bernhardt and colleagues (2003) estimated that nitrate-N export after a severe wind disturbance at Hubbard Brook Experimental Watershed was substantially less than expected due to instream processing and retention. Warren and colleagues (2007) found that phosphate uptake velocity in a stream, also at Hubbard Brook, was positively correlated with both stand age and instream large wood volume, and attributed that to abiotic adsorption of phosphate by inorganic sediments retained by the wood.

9-3. OBJECTIVES

The objective of this study was to provide the Washington Forest Practices Board (WFPB) with information on the magnitude and direction of change in N and P export after timber harvest following one of three different riparian buffer treatments, including treatments more restrictive and less restrictive than allowed by current Forest Practices rules. The research questions were:

1) What is the magnitude of change in N and P concentration and export from watersheds relative to unharvested reference sites following timber harvest in each of the three buffer treatments?

2) What are the differences in the magnitude of the change in concentration and export among the three buffer treatments?

9-4. METHODS

9-4.1. *EXPERIMENTAL DESIGN*

9-4.1.1. Study Design

We used a Before-After Control-Impact (BACI) design whereby we could evaluate post-harvest changes in sites relative to the unharvested control (referred to here as reference) sites. An advantage of this design is that it controls for the effect of large-scale temporal variability (e.g., inter-annual differences in precipitation) affecting all sites by establishing relationships between the control and impact (i.e., harvested) sites in the pre- and post-harvest periods (Smith 2002).

9-4.1.2. Site Description

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 9-1**). Catchment area above the stream discharge monitoring location ranged from 11.8 to 44.3 ha (**Table 9-2**). Areas of some sites differ from those presented in **Table 2-5** (Chapter 2 **–** *Study Design*) because we could not always measure discharge at the regulatory break between fish-bearing and non-fish-bearing stream segments (F/N break).

Table 9-1. Elevation, lithology, stream gradient, and stream order (Strahler 1952) for the OLYM and 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 RMZ: two-sided 50-ft (15.2-m) riparian buffers of 100%, Forest Practice (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 (DEM) 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 9-2. Catchment area above the flow gauge, percent of catchment clearcut harvested, and percent hardwood trees in riparian stand.

Block	Treatment	Area (ha)	% Clearcut	% Hardwood in Riparian Stand
WIL1	REF	11.8	0	4
	100%	26.2	89	1
	FP	14.4	94	0
	0%	27.7	100	9
OLYM	REF	44.3	0	1
	100%	22.1	43	16
	FP	17.3	88	12
	0%	13.1	100	22

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 land. 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 (non-fish-bearing perennial) channel gradients ranging from 16 to 31%. We present site-selection criteria for each study site in **Table 2-5** (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 1,800 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°C 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).

9-4.1.3. Experimental Treatments

The riparian management zone (RMZ) for non-fish-bearing perennial and seasonal streams, Type Np and Ns, respectively, in western Washington includes the following requirements:

1) **Equipment limitation zone:** A two-sided 30-ft (9.1-m) wide zone measured horizontally from the outer edge of the bankfull width of Type Np or Ns water where equipment use and other forest practices are specifically limited. On-site mitigation is required if ground-based equipment, skid trails, stream crossings (other than existing roads), or partially suspended cabled logs expose the soil on more than 10% of the surface area of the zone. Mitigation measures (e.g., water bars, grass seeding, mulching) must be designed to replace the equivalent of lost functions, especially prevention of sediment delivery to the channel.

2) **Riparian protection:** A two-sided 50-ft (15.2-m) wide no-harvest riparian buffer along at least 50% of the Type Np stream length, including:

    a. **Stream buffer:** Required two-sided buffers must start at the F/N break and continue upstream for: (a) a minimum of 500 ft (152.4 m) for Type Np Waters longer than 1,000 ft (305 m); (b) at least equal to the greater of 300 ft (91 m) or 50% of the entire length for Type Np Waters greater than 300 ft (91 m) but less than 1,000 ft (305 m); or (c) buffered in their entirety for Type Np Waters less than or equal to 300 ft (91 m).

    b. **Sensitive site buffers:** No-harvest buffers specific to each sensitive site category (see **Table 2-2**; Chapter 2 – *Study Design*).

The four experimental treatments included (**Figure 9-1**):

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 the entire perennial stream length buffered with a two-sided 50-ft (15.2-m) riparian buffer,
3) **Forest Practices treatment (FP):** clearcut harvest with a current Forest Practices' two-sided 50-ft (15.2-m) riparian leave tree buffer along at least 50% of the RMZ, including buffers prescribed for sensitive sites (side-slope and headwall seeps, headwater springs, Type Np intersections and alluvial fans), and
4) **0% treatment (0%):** clearcut harvest with no riparian buffer retained within the RMZ.

A 30 ft (9.1 m) equipment limitation zone was maintained along all Type Np and Ns Waters, regardless of buffer treatment.

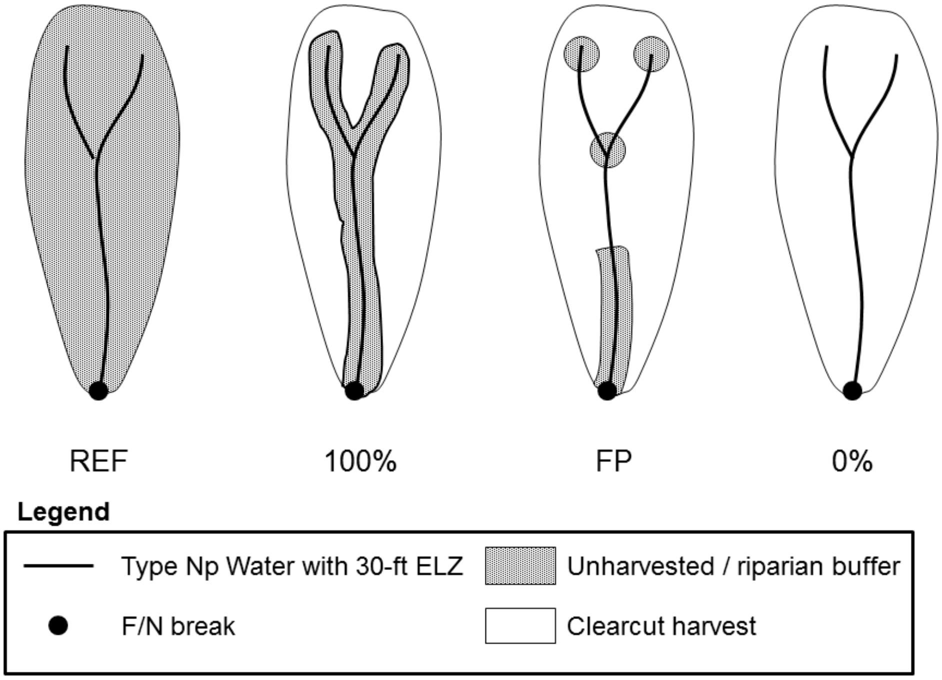

Figure 9-1. Schematic of the four experimental treatments included in the Type N Study. Treatments include unharvested reference sites (REF) and sites receiving a clearcut harvest with one of three riparian buffer treatments along the Type Np Water RMZ: two-sided 50 ft (15.2 m) riparian buffers of 100%, Forest Practice (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).

9-4.2. *STREAMFLOW AND TURBIDITY*

We monitored stream stage and turbidity in the eight sites of the Olympic and Willapa 1 blocks from September 2006 to September 2011. Data collection was made using a system from Forest Technology Systems (www.FTSenvironmental.com) consisting of:

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

We installed Montana-style Parshall flumes in six of the eight study sites, and a pressure transducer (Ott Messtechnik, Kempten, Germany) in each of the stilling wells to measure stage height.[1] We calculated discharge using the appropriate equations for each flume. There were no suitable locations for a flume in the OLYM-REF and OLYM-100%, so we measured stage height with a pressure transducer installed at the upstream end of a culvert. Discrete flows were measured using a Swoffer flow meter or, at very low flows, by measuring the time to fill a 20-L bucket from the culvert. We based discharge estimates on the flow-versus-stage height rating curve developed at each location.

Our intent was to collect two complete years of flow data for the pre-harvest and post-harvest period; 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 N and P export estimates.

9-4.3. *WATER SAMPLE COLLECTION AND CHEMICAL ANALYSIS*

We manually collected water samples at six to eight week intervals from October 2006 to September 2011, unless the site was inaccessible due to weather, road maintenance, or harvest activities. Water was collected at the flow gauging location in acid-washed Nalgene bottles containing concentrated hydrochloric acid (P samples) or sulfuric acid (total-N and nitrate-N samples) 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 2008) to collect water samples during high flow events across the range of turbidity and flow values. Twelve turbidity thresholds, ranging from 10 to 1,600 nephelometric turbidity units (NTU), were set for both the rising and falling limbs of the turbidity graph. Samples were

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[1] 18-inch flumes were installed in the OLYM-FP and WIL1-FP. 24-inch flumes were installed in the OLYM-0%, WIL1-REF, WIL1-100%, and WIL1-0%.

collected in 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^{-1}.

Turbidity threshold sampling was designed to collect samples for analysis of suspended sediment concentration (SSC). The SSC and turbidity values can be 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, nitrate-N, and total-P concentrations using the continuous flow and, in some cases, turbidity data.

We were concerned about the effect of biological activity (uptake and transformation of N) and adsorption of P onto the container wall 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 (**Appendix 9-A**). We collected four replicate water samples (four sample bottles filled in sequence from a single stream on a single visit) from four forested headwater streams in southwest Washington, near the Willapa block in this study. There was no difference between replicate samples collected, preserved, and cooled on the same day and replicate samples where one was processed as above and the other was stored at ambient temperatures for one to four weeks prior to preserving and cooling ($P > 0.05$). In addition, when expressed as the relative percent difference, the values were within the laboratory guidelines. Therefore, the delay in preserving and cooling the samples had no measurable effect on the results of the chemical analyses. Additionally, our results were consistent with Martin and Harr (1988) and Vanderbilt and colleagues (2003) who found no effect of sample storage for up to three weeks on nitrate-N concentration in forested western Oregon streams. Similarly, Burke and colleagues (2002) found no detectable difference in total-N, nitrate-N, or total-P concentrations in water samples from south Florida that were (1) processed immediately; (2) refrigerated then processed seven days later; or (3) not refrigerated, then processed seven days later.

All manually collected samples were analyzed for total-N, nitrate-N, ammonia-N, total-P, and soluble reactive P (SRP; **Table 9-3**). Samples collected using the pump sampler were analyzed for total-N, nitrate-N, and total-P only. Total-N, nitrate-N, ammonia-N, and total-P concentrations were determined from unfiltered water samples and represent the sum of particulate and dissolved forms. However, nitrate-N is very soluble. In one study in western Washington nitrate-N concentration in filtered samples was only 0.85% less than that in unfiltered water samples collected at the same time (Sackmann 2011). This was based on 71 sampling events uniformly spaced throughout an entire year and across a range of nitrate-N concentrations from 300 to 1000 μg L^{-1}. We believe our nitrate-N concentration and export estimates are comparable to estimates based on dissolved nitrate-N. Samples for soluble reactive P analysis were filtered through a 0.45μm membrane filter before analysis. 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, the Hoh River Ranger station Olympic National Park (47.8597°, −123.9325°, elevation 182 m), and site WA21, near La Grande, WA (46.8353°, −122.2867°, elevation 617 m; NADP 2013).

Table 9-3. Nitrogen and phosphorus analytical methods.

Analyte	Method	Reporting Limit (μg L^{-1})
Ammonia-N[1]	4500-NH3-H	10
Nitrate-N[1]	4500-NO3-I	10
Total-N[1]	4500-N B	25
Soluble Reactive P[1]	4500-P G	3
Total-P[2]	EPA 200.8	5

[1] APHA (2016) *Standard Methods for the Examination of Water and Wastewater*, 22nd Editions.
[2] EPA (1983) *Method 200.8, Modified for Phosphorus, EPA Methods for Chemical Analysis of Water and Wastes*, EPA 600/4-79-020 Cincinnati, OH

9-4.4. *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, nitrate-N, and total-P 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 concentrations changed rapidly over a given event. The predictive equations using the 10-minute data provided better temporal resolution and were more robust than with daily mean data. We used **Equation 9-1** to calculate total-N and nitrate-N:

$$\mathrm{Log}[N]_\mathrm{i} = \beta_0 + \beta_1 LogQ_i + \beta_2\, LogQ_i{}^2 + \beta_3 \sin\frac{c\pi t}{365.25} + \beta_4 \cos\frac{c\pi t}{365.25} + \beta_5 LogT_i + \varepsilon_i \qquad (9\text{-}1)$$

where: $\mathrm{Log}[\mathrm{N}]_\mathrm{i}$ is base 10 logarithm of total-N or nitrate-N concentration of the i^{th} sample,
$\beta_0 - \beta_5$ are regression coefficients,
$\mathrm{Log}(Q)$ is base 10 logarithm of flow,
sin and cos functions are seasonal terms,
c is 2 or 4 depending on whether the seasonal term is one or two cycles per year,
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- and post-harvest periods at all six treatment sites (**Table 9-4**) because there was a substantial and significant difference in the regression relationship between pre- and post-harvest periods. One model was used for the entire 2006 to 2011 period at each of the reference sites. Flow and the seasonal terms were used in the total-N and nitrate-N models for all sites and all 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. Typically, turbidity improved the model fit for samples collected during higher discharge, higher turbidity (i.e., high particulate load) events in the fall. A similar model was used to estimate total-P concentration:

$$\mathrm{Log}[P]_\mathrm{i} = \beta_0 + \beta_1 LogT_i + \beta_2 LogT_i^{\,2} + \beta_3 \sin\frac{c\pi t}{365.25} + \beta_4 \cos\frac{c\pi t}{365.25} + \beta_5 LogQ_i + \varepsilon_i \qquad (9\text{-}2)$$

where: $\mathrm{Log}[P]_\mathrm{i}$ is base 10 logarithm of total-P concentration of the i^{th} sample and other terms are the same as **Equation 9-1**.

We used the turbidity and seasonal terms in the total-P models for all sites and all periods. The flow 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. In contrast to the N models, a single regression model was used for both periods for total-P in all six riparian buffer treatment sites as well as both reference sites (**Table 9-4**) because there was no substantive difference between the pre- and post-harvest models. Sample size for the regression models ranged from 14 to 141 and was dependent upon the length of the pre- and post-harvest periods (some sites did not have two full years of pre-harvest data) and the number of samples collected via the automated pump sampler (e.g., fewer sampling events at some sites, occasional equipment malfunctions, and some samples could not be collected within seven days). The adjusted R^2 of the regression models ranged from 0.453 to 0.840 for total-N, 0.488 to 0.924 for nitrate-N, and 0.394 to 0.924 for total-P. 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 adjust for bias introduced when transforming from log-scale to untransformed scale. Instantaneous N and P 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 were calculated as 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^{-1} yr^{-1}). We did not estimate loads for ammonia-N because 99% of the reported concentrations were below the lab's reporting limit (10 μg N L^{-1}) or for SRP because the values were near the reporting limit and samples were not collected during high discharge events.

All regressions and load calculations were done using SYSTAT 13 statistical software (SYSTAT Software, Inc. 2009).

Table 9-4. Regression models for estimating nutrient concentration. Separate models were developed for pre- and post-harvest periods for total-N and nitrate-N in all buffer treatment sites because the relationship between concentration and discharge changed post-harvest. In contrast, a single regression model was used for both periods for total-P. A single regression model was used in both reference sites for total-P, total-N, and nitrate-N. SE = standard error, Var = variables used in regression (Q-flow, T-turbidity).

Block	**Treatment**	**Period**	**Total-N**				**Nitrate-N**				**Total-P**			
			N	**r^2**	**SE**	**Var**	**N**	**r^2**	**SE**	**Var**	**N**	**r^2**	**SE**	**Var**
WIL1	REF	All	76	0.532	0.153	Q	77	0.488	0.115	Q	52	0.728	0.124	T
	100%	Pre-	41	0.549	0.108	Q, T	35	0.499	0.104	Q				
		Post-	38	0.458	0.248	Q, T	38	0.563	0.221	Q, T	81	0.759	0.150	T, Q
	FP	Pre-	26	0.809	0.134	Q, T	27	0.680	0.184	Q, T				
		Post-	19	0.777	0.220	Q, T	19	0.565	0.250	Q	44	0.867	0.343	T
	0%	Pre-	14	0.818	0.039	Q, T	14	0.924	0.028	Q				
		Post-	45	0.630	0.125	Q,T	45	0.639	0.107	Q, T	69	0.718	0.131	T, Q
OLYM	REF	All	102	0.613	0.154	Q, T	103	0.624	0.117	Q	111	0.684	0.150	T, Q
	100%	Pre-	35	0.840	0.088	Q	35	0.818	0.097	Q				
		Post-	141	0.522	0.130	Q, T	141	0.501	0.127	Q	135	0.657	0.179	T, Q
	FP	Pre-	20	0.620	0.103	Q, T	20	0.804	0.066	Q, T				
		Post-	30	0.453	0.102	Q	49	0.511	0.128	Q	77	0.394	0.119	T
	0%	Pre-	44	0.780	0.115	Q, T	44	0.767	0.103	Q, T				
		Post-	54	0.796	0.098	Q, T	0	0.773	0.096	Q, T	86	0.830	0.096	T

9-4.5. *STATISTICAL ANALYSIS*

The timing of the timber harvest could not be synchronized across all harvest treatment sites. The start date, end date, and duration of harvest were determined by the landowner and varied among sites (**Table 9-5**). Export from each watershed was highly dependent upon flow, which varied across years, both in quantity and timing. The variables analyzed below were the difference in annual export or mean concentration (harvested site minus reference) between each buffer treatment site and its reference site over the same period. There were two pre-harvest years for each site except for the WIL1-0%, OLYM-100%, and OLYM-FP, where there was only one pre-harvest year. There were two post-harvest years for each site (**Appendix Table 9-B-1**). We included annual export estimates and annual, flow-weighted nutrient concentrations in the analysis described below.

Table 9-5. Pre-harvest and post-harvest periods and number of annual nutrient export estimates for the six treatment sites (modified from Chapter 3 – *Management Prescriptions*).

Block	Treatment	Treatment Periods		No. of Years
		Pre	Post	Pre-/Post- harvest
OLYM	100%	02/2008 to 02/2009	03/2009 to 03/2011	1/2
	FP	07/2007 to 7/2008	10/2008 to 10/2010	1/2
	0%	06/2007 to 06/2009	08/2007 to 08/2009	2/2
WIL1	100%	10/2006 to 10/2008	04/2009 to 04/2011	2/2
	FP	10/2006 to 10/2008	03/2009 to 03/2011	2/2
	0%	04/2007 to 04/2008	01/2009 to 01/2011	1/2

We used a generalized linear mixed effects model (GLMM) with site as a random effect and buffer treatment (100%, FP, and 0%), period (pre- versus post-harvest), 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 2013).

We evaluated six hypotheses grouped under the two research questions presented earlier:

1) What is the magnitude of change in nutrient (total-N, nitrate-N, and total-P) concentration and annual export relative to an unharvested reference site following timber harvest in each of the three buffer treatments? This was addressed with three post-hoc comparisons testing the following hypothesis for each harvest treatment:

a) H_0: $(\mathrm{Trmt}_{100\%,\mathrm{pre}} - \mathrm{Ref}_{\mathrm{pre}}) = (\mathrm{Trmt}_{100\%,\mathrm{post}} - \mathrm{Ref}_{\mathrm{post}})$ (9-3)

b) H_0: $(\mathrm{Trmt}_{FP,\mathrm{pre}} - \mathrm{Ref}_{\mathrm{pre}}) = (\mathrm{Trmt}_{FP,\mathrm{post}} - \mathrm{Ref}_{\mathrm{post}})$ (9-4)

c) H_0: $(\mathrm{Trmt}_{0\%,\mathrm{pre}} - \mathrm{Ref}_{\mathrm{pre}}) = (\mathrm{Trmt}_{0\%,\mathrm{post}} - \mathrm{Ref}_{\mathrm{post}})$ (9-5)

where: Trmt is export from treatment site,
Ref is export from the reference site over the same period, and
Pre, post denote pre- and post-harvest 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 testing the hypotheses:

d) H_0: $(\mathrm{Trmt}_{100\%,\mathrm{pre}} - \mathrm{Ref}_{\mathrm{pre}}) - (\mathrm{Trmt}_{100\%,\mathrm{post}} - \mathrm{Ref}_{\mathrm{post}}) = (\mathrm{Trmt}_{\mathrm{FP},\mathrm{pre}} - \mathrm{Ref}_{\mathrm{pre}}) - (\mathrm{Trmt}_{\mathrm{FP},\mathrm{post}} - \mathrm{Ref}_{\mathrm{post}})$ (9-6)

e) H_0: $(\mathrm{Trmt}_{100\%,\mathrm{pre}} - \mathrm{Ref}_{\mathrm{pre}}) - (\mathrm{Trmt}_{100\%,\mathrm{post}} - \mathrm{Ref}_{\mathrm{post}}) = (\mathrm{Trmt}_{0\%,\mathrm{pre}} - \mathrm{Ref}_{\mathrm{pre}}) - (\mathrm{Trmt}_{0\%,\mathrm{post}} - \mathrm{Ref}_{\mathrm{post}})$ (9-7)

f) H_0: $(\mathrm{Trmt}_{\mathrm{FP},\mathrm{pre}} - \mathrm{Ref}_{\mathrm{pre}}) - (\mathrm{Trmt}_{\mathrm{FP},\mathrm{post}} - \mathrm{Ref}_{\mathrm{post}}) = (\mathrm{Trmt}_{0\%,\mathrm{pre}} - \mathrm{Ref}_{\mathrm{pre}}) - (\mathrm{Trmt}_{0\%,\mathrm{post}} - \mathrm{Ref}_{\mathrm{post}})$ (9-8)

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 a Type II error and can mask subtle treatment effects. Instead, we considered the P-value, effect size, patterns of the effect size across the buffer treatments, and sample size when interpreting the results.

9-5. RESULTS

9-5.1. *NITROGEN AND PHOSPHORUS CONCENTRATIONS*

We did not calculate loads for ammonia-N because ammonia-N concentration was low in all streams over the entire study. Of the 295 analyses done across all eight sites, only three results (from different sites) were at or above the laboratory reporting limit of 10 μg L^{-1}.

Mean, flow-weighted, pre-harvest total-N concentration ranged from 120 to 841 μg L^{-1} with the lowest and highest concentrations in adjacent sites, WIL1-100% and WIL-FP, respectively (**Table 9-6**). Mean post-harvest concentration increased at all buffer treatment sites and ranged from 20 to 398 μg L^{-1} higher. However, relative to the reference sites flow-weighted total-N concentration increased 17, 15, and 150 μg L^{-1} in the 100%, FP, and 0% treatments, respectively (**Table 9-7**). The post-harvest changes were not significant ($P > 0.05$) for any of the buffer treatments nor did the changes differ among the buffer treatments.

Table 9-6. Mean flow-weighted concentrations in μg L^{-1} for pre- and post-harvest periods and the difference between periods for total-N, nitrate-N, total-P, and SRP. Ammonia-N is not presented because concentration in all but three samples was below reporting limits.

Block	Treatment	Period	Total-N	NO_3-N	Total-P	SRP
WIL1	REF	NA	621	446	22	11
	100%	Pre-	120	98	18	9
		Post-	437	263	17	8
		Difference	317	165	−1	−1
	FP	Pre-	841	800	36	17
		Post-	887	906	23	17
		Difference	46	106	−13	0
	0%	Pre-	463	450	5	6
		Post-	861	727	10	6
		Difference	398	277	5	0
OLYM	REF	NA	275	167	12	8
	100%	Pre-	396	319	21	6
		Post-	416	340	35	7
		Difference	20	21	14	1
	FP%	Pre-	235	225	8	10
		Post-	546	473	10	10
		Difference	311	248	2	0
	0%	Pre-	570	460	12	6
		Post-	668	604	8	6
		Difference	98	144	−4	0

Table 9-7. Results of hypothesis tests for nutrient concentration described in section 9-4.5. Hypotheses a), b), and c) compare post-harvest concentrations with pre-harvest in each buffer treatment and d), e), and f) compare the post-harvest changes among the three buffer treatments. Comparisons in **bold** indicate the hypothesis was rejected at P <0.05. P-values were not adjusted for multiple comparisons.

Comparison	Change	P-value	95% C.I.	
			Lower	Upper
Total-N (μg L^{-1})				
a) Pre vs. Post-100%	17	0.86	−185	218
b) Pre vs. Post-FP	15	0.88	−262	842
c) Pre vs. Post-0%	150	0.13	−53	862
d) 100% vs. FP	−2	0.99	−287	283
e) 100% vs. 0%	133	0.33	−152	418
f) FP vs. 0%	135	0.32	−150	420
Nitrate-N (μg L^{-1})				
a) Pre vs. Post-100%	77	0.27	−67	220
b) Pre vs. Post-FP	87	0.21	−57	230
c) Pre vs. Post-0%	**177**	**0.02**	**33**	**321**
d) 100% vs. FP	10	0.92	−193	213
e) 100% vs. 0%	101	0.30	−103	304
f) FP vs. 0%	91	0.35	−113	294
Total-P (μg L^{-1})				
a) Pre vs. Post-100%	2	0.83	−14	17
b) Pre vs. Post-FP	−12	0.10	−28	3
c) Pre vs. Post-0%	−3	0.66	−19	12
d) 100% vs. FP	−14	0.19	−36	8
e) 100% vs. 0%	−5	0.64	−26	17
f) FP vs. 0%	9	0.37	−12	31

Total-N was comprised largely of nitrate-N at all sites. Mean, flow-weighted nitrate-N concentration ranged from 98 to 800 μg L^{-1} pre-harvest with the minimum and maximum values also occurring in the WIL1-100% and WIL1-FP, respectively (**Table 9-6**). Post-harvest nitrate-N concentration was 21 to 277 μg L^{-1} higher in the buffer treatment sites. Relative to the reference site, nitrate-N concentration increased 77, 87, and 177 μg L^{-1} post-harvest in the 100%, FP, and 0% buffer treatments, respectively (**Table 9-7**). Only the 0% treatment increased significantly (P <0.05). There was a pattern of nitrate-N concentration increasing in proportion to the amount of hardwood vegetation in the riparian zone in the Olympic block, but not the Willapa 1 block (**Tables 9-2** and **9-6**).

A seasonal pattern of low total-N concentrations during the summer low-flow period and highest concentrations during the fall freshets was seen at all sites pre-harvest, with the least intra-annual variability seen in the WIL1-100%, which had the lowest concentration (**Figure 9-2**). Nitrate-N concentration showed a very similar pattern (**Figure 9-3**). The seasonal variability increased post-harvest in all buffer treatment sites.

Pre-harvest total-P concentrations were low (5 to 36 μg L^{-1}). Post-harvest changes were very small and variable, ranging from a 13 μg L^{-1} decrease to 14 μg L^{-1} increase (**Table 9-6**). Relative to the reference sites, there were no significant ($P > 0.05$) changes in total-P concentration and no apparent relationship with buffer treatment (**Table 9-7**). Although total-P concentration varied over the year, it was more closely related to high flow/high turbidity events than seasonal effects (**Figure 9-4**).

Flow-weighted mean SRP concentrations were very low and changed by 1 μg L^{-1} or less post-harvest at all sites (**Table 9-6**). We did not attempt to calculate SRP loads because samples were not collected during high flow events.

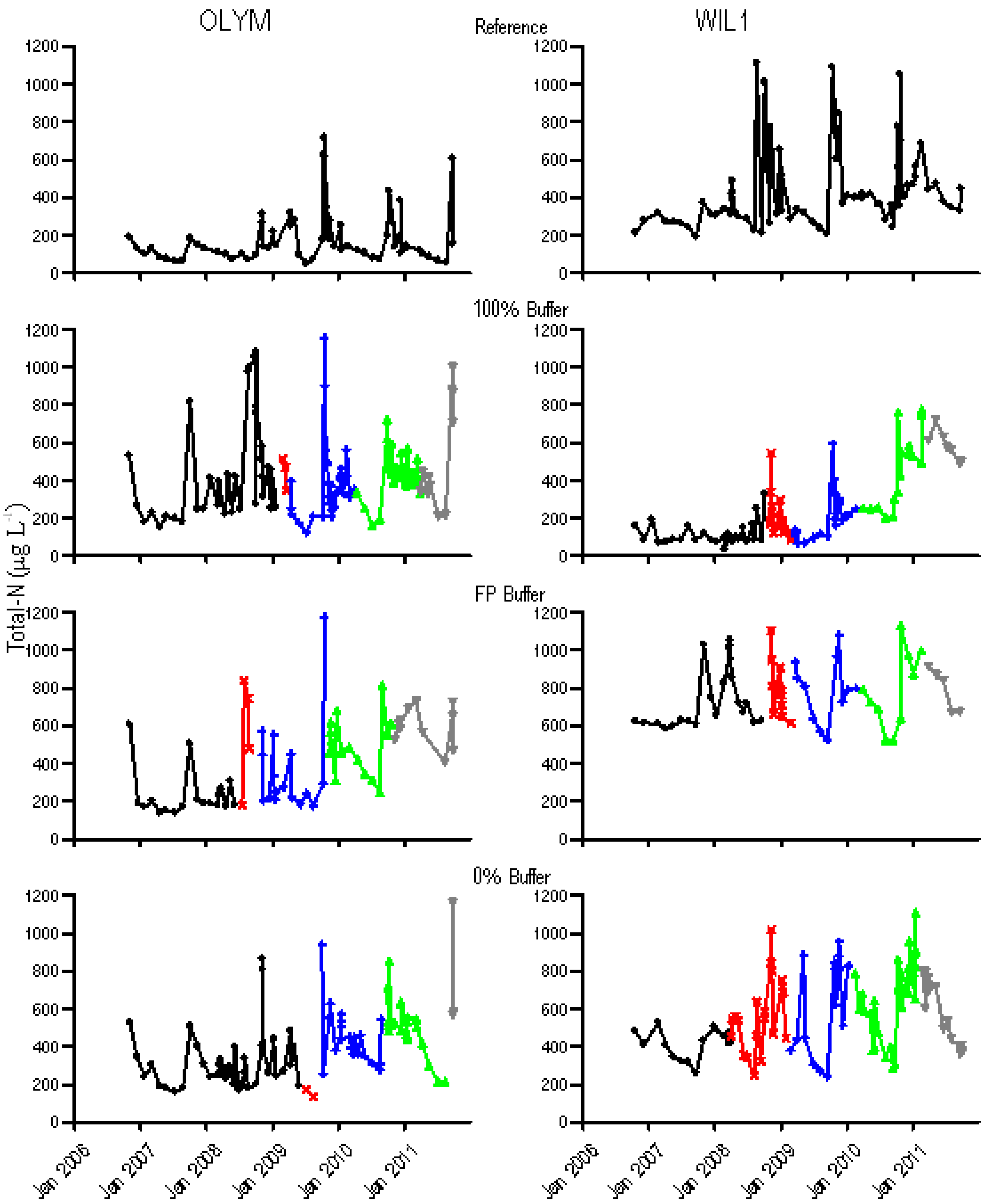

Figure 9-2. Total-N concentration from October 2006 to September 2011. Black lines are pre-harvest, red lines are during harvest, and blue, green, and gray lines are the first, second, and third year post-harvest, respectively.

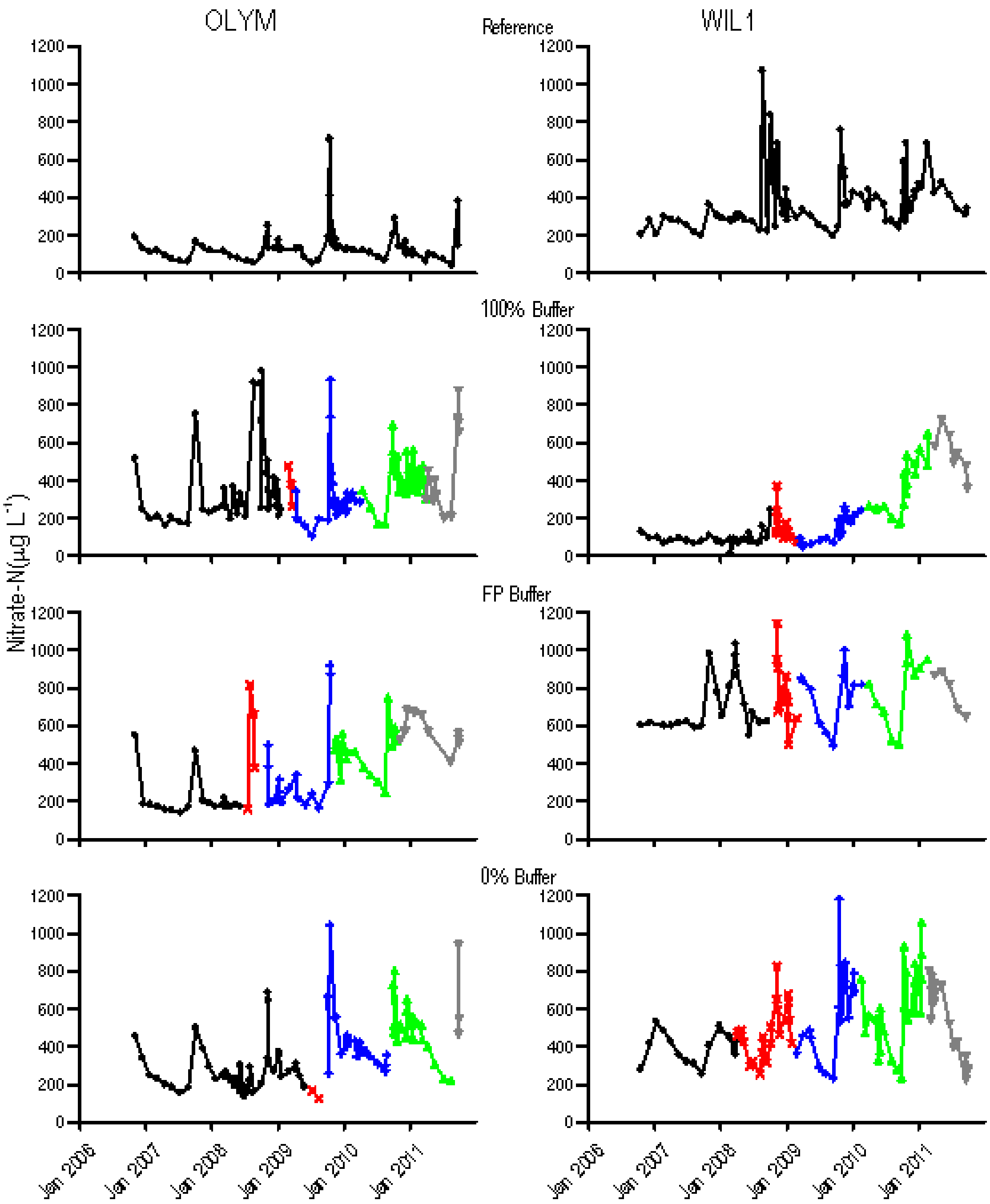

Figure 9-3. Nitrate-N concentration from October 2006 to September 2011. Black lines are pre-harvest, red lines are during harvest, and blue, green, and gray lines are the first, second, and third year post-harvest, respectively.

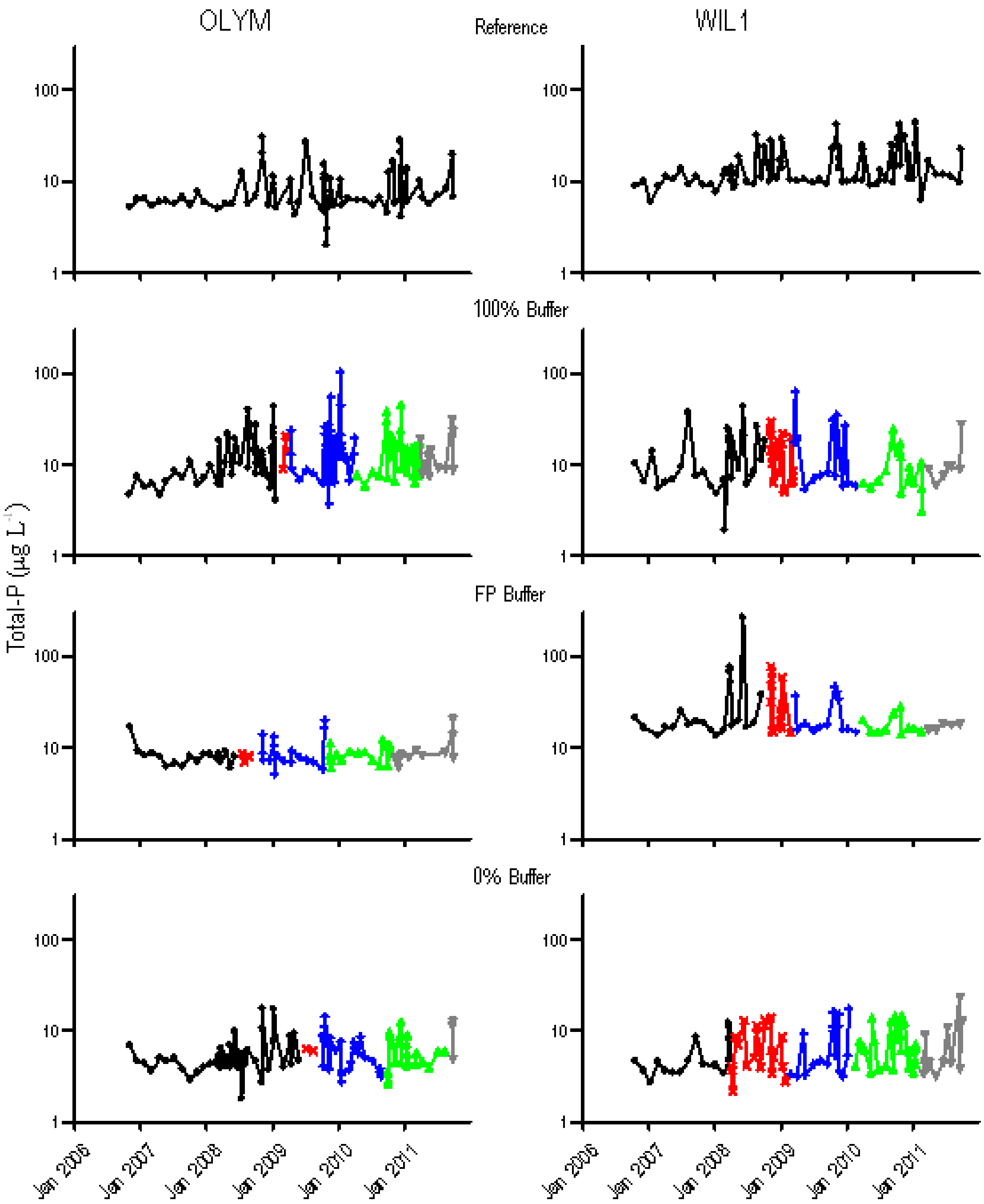

Figure 9-4. Total-P concentration from October 2006 to September 2011. Black lines are pre-harvest, red lines are during harvest, and blue, green, and gray lines are the first, second, and third year post-harvest, respectively. Note the logarithmic y-axis.

9-5.2. *NITROGEN AND PHOSPHORUS EXPORT*

9-5.2.1. Seasonal Patterns

All sites showed a spike in total-N and nitrate-N export during the first fall freshets even pre-harvest, likely due to litter entrainment and leeching. The effect of harvest is apparent in **Figure 9-5** and **Figure 9-6**, where, as a result of higher concentrations and higher flows, N export increased relative to the reference sites. This increase in N export was highest during the first high flow months in the fall and early winter.

Total-P export typically coincided with high flows, regardless of the season (**Figure 9-7**). This was likely due to entrainment of periphyton, organic matter, and sediment. A post-harvest increase in the magnitude of P export was less obvious than for N export but was apparent in all treatment sites except the OLYM-100%.

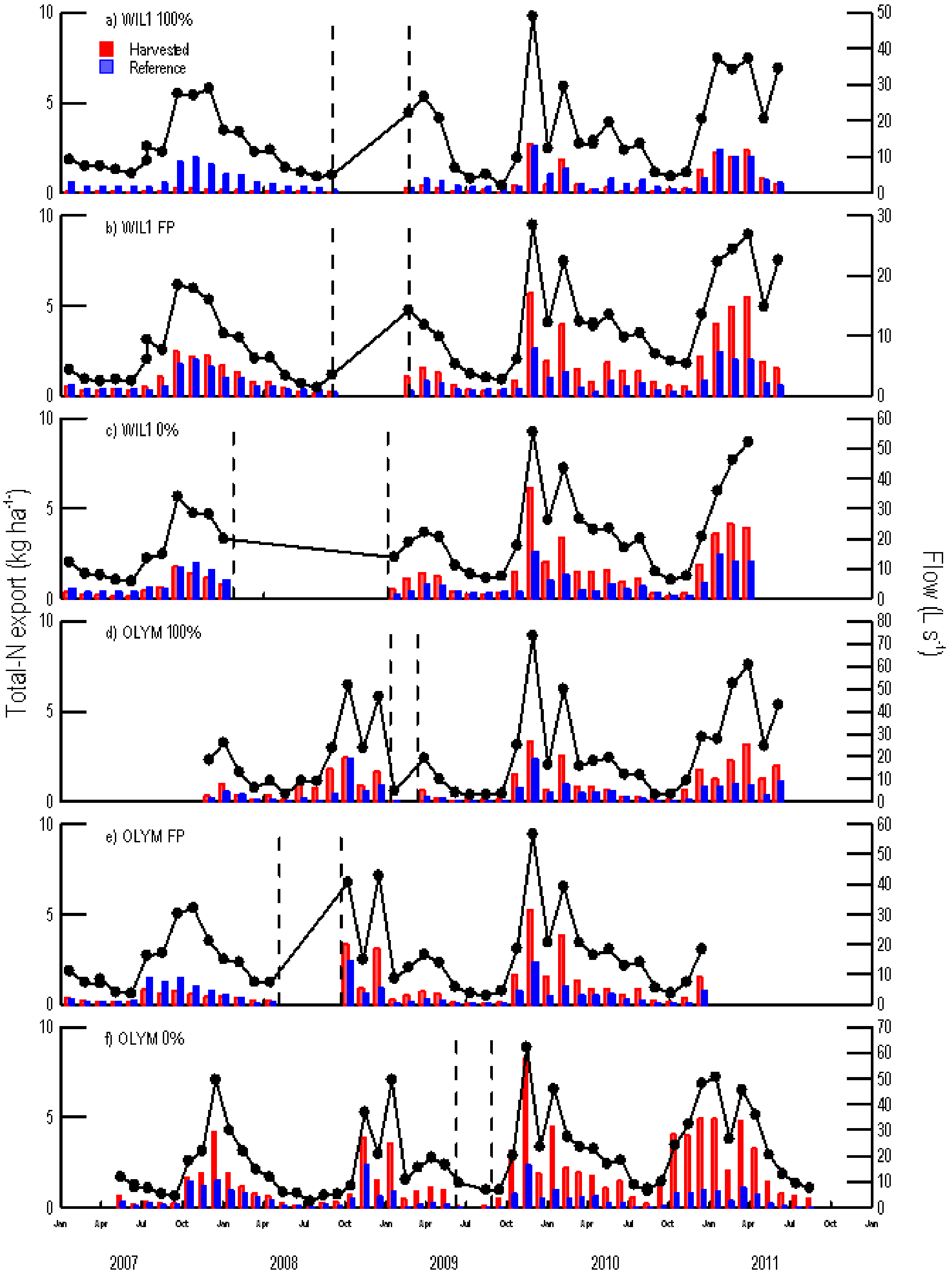

Figure 9-5. Monthly total-N export (kg ha^{-1}) from treatment (red bars) and reference (blue bars) sites are shown (left axis) with mean monthly stream flow (black line) overlaid (right axis). The vertical dashed lines bracket the active harvest period.

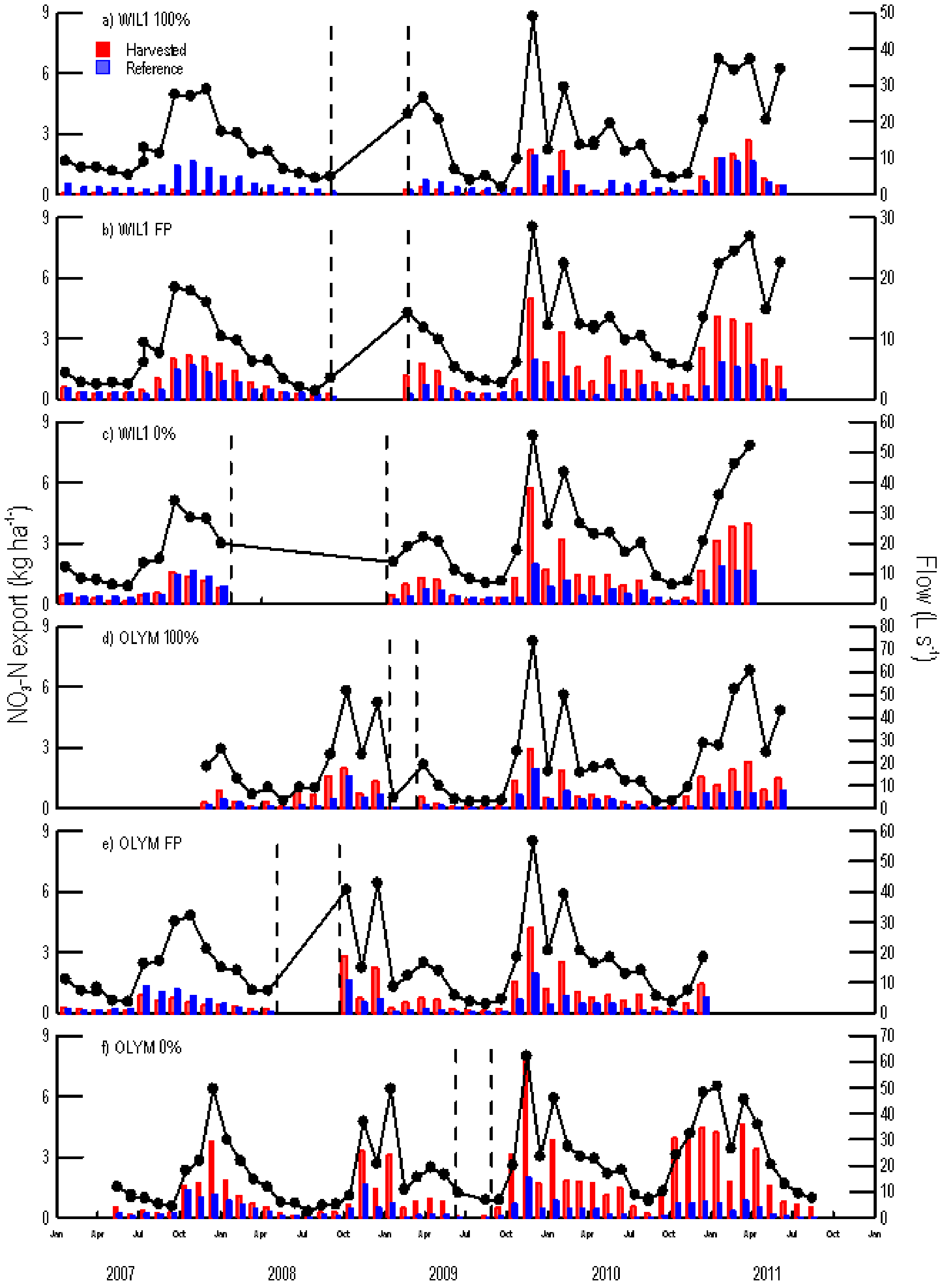

Figure 9-6. Monthly nitrate-N export (kg ha^{-1}) from treatment (red bars) and reference (blue bars) sites are shown (left axis) with mean monthly stream flow (black line) overlaid (right axis). The vertical dashed lines bracket the active harvest period.

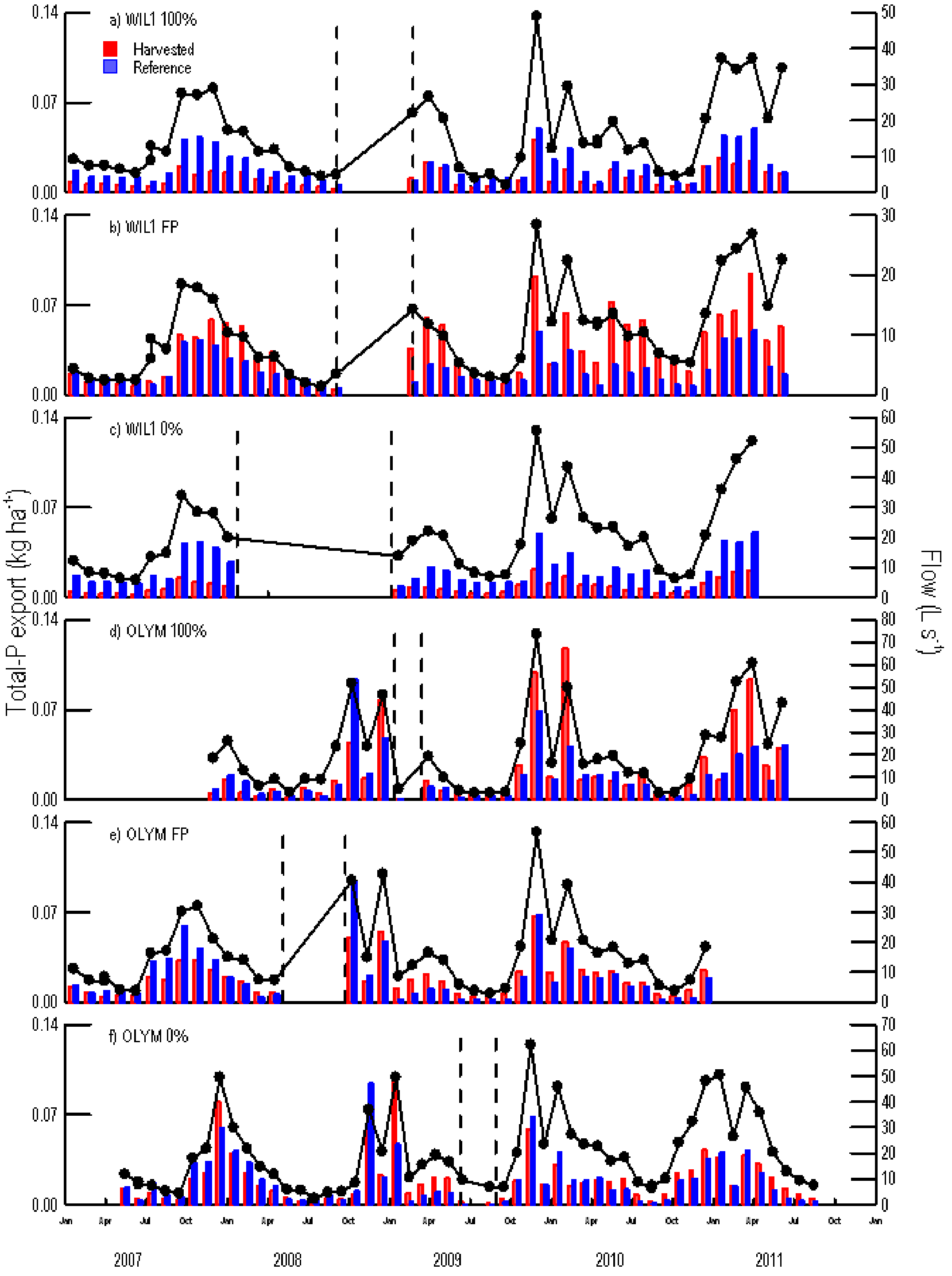

Figure 9-7. Monthly total-P export (kg ha^{-1}) from treatment (red bars) and reference (blue bars) sites are shown (left axis) with mean monthly stream flow (black line) overlaid (right axis). The vertical dashed lines bracket the active harvest period.

9-5.2.2. Annual Export

As expected, total-N and nitrate-N export increased post-harvest at all sites, with the smallest increase in the 100% treatment and the largest in the 0% treatment. Mean, pre-harvest total-N export ranged from 1.78 kg ha^{-1} yr^{-1} in the WIL1-100% to 14.42 kg ha^{-1} yr^{-1} in the WIL1-FP (**Table 9-8**). These sites are adjacent to each other (and WIL1-REF), highlighting the variability in export among sites even pre-harvest. Mean pre-harvest export from the Olympic buffer treatment sites ranged from 5.19 to 11.95 kg ha^{-1} yr^{-1}. Export from the reference basins over the course of the study ranged from 9.48 to 12.16 kg ha^{-1} yr^{-1} in the WIL1-REF and from 5.79 to 7.28 kg ha^{-1} yr^{-1} in the OLYM-REF. Pre-harvest nitrate-N export in the Olympic block increased with increasing proportion of hardwood vegetation in the riparian zone but not in the Willapa 1 block.

Post-harvest mean annual total-N export ranged from 8.16 kg ha^{-1} yr^{-1} (6.26 kg ha^{-1} yr^{-1} increase) in the WIL1-100% to 32.86 kg ha^{-1} yr^{-1} (20.91 kg ha^{-1} yr^{-1} increase) in the OLYM-0%. The post-harvest mean annual total-N export at the OLYM-100% increased the least with 0.71 kg ha^{-1} yr^{-1}. On a percentage basis, increases ranged from 7% in the OLYM-100% to 358% in the WIL1-100%.

Compared to the reference sites, the GLMM analysis showed a relative increase in total-N export post-harvest of 5.52 ($P = 0.051$), 11.52 ($P = 0.0007$), and 17.16 ($P < 0.0001$) kg ha^{-1} yr^{-1} in the 100%, FP, and 0% treatments (**Figure 9-8**; **Table 9-9**). Export from the FP treatment was not significantly different from either the 100% or 0% treatments while the 0% was 11.64 kg ha^{-1} yr^{-1} ($P = 0.007$) more than the 100% treatment.

The overall pattern of nitrate-N export is very similar to total-N export because nitrate-N comprised from 79 to more than 99% of the estimated total-N load across all sites and years. Mean pre-harvest nitrate-N export ranged from 1.70 kg ha^{-1} yr^{-1} in the WIL1-100% to 13.32 kg ha^{-1} yr^{-1} in the WIL1-FP (**Table 9-8**). Export from the Olympic block ranged from 4.85 to 11.15 kg ha^{-1} yr^{-1}. Post-harvest, nitrate-N export increased by 1.22 to 18.82 kg ha^{-1} yr^{-1}, a 13–327% increase.

The GLMM analysis shows a relative increase in nitrate-N export post-harvest of 4.83 ($P = 0.048$), 10.24 ($P = 0.001$), and 15.35 ($P < 0.0001$) kg ha^{-1} yr^{-1} in the 100%, FP, and 0% treatments, respectively (**Table 9-9**; **Figure 9-8**), only slightly less than the changes in total-N. Export from the FP treatment was not significantly different than either the 100% or 0% treatments, but the 0% was significantly ($P < 0.05$) greater than the 100% treatment.

Mean, annual, pre-harvest total-P loads ranged from 0.08 kg ha^{-1} yr^{-1} in the WIL1-0% to 0.35 kg ha^{-1} yr^{-1} in the WIL1-FP (**Table 9-8**). Post-harvest loads increased in all of the six treatment sites by 0.03 to 0.12 kg ha^{-1} yr^{-1}, a 21 to 50% increase over pre-harvest. Total-P export from the reference sites decreased by 0.02 to 0.005 kg ha^{-1} yr^{-1}over the same period.

Total-P export increased post-harvest by a similar magnitude in all treatments: 0.10 ($P = 0.006$), 0.13 ($P = 0.001$), and 0.09 ($P = 0.010$) kg ha^{-1} yr^{-1} in the 100%, FP, and 0% treatments, respectively (**Table 9-9**; **Figure 9-9**).

The increase in N, total-N and nitrate-N, from the treatment watersheds post-harvest was strongly correlated with the increase in annual runoff (R^2 = 0.970 and 0.971; P = 0.001 and 0.001, respectively) and with the proportion of the basin harvested (R^2 = 0.854 and 0.852; P = 0.031 and 0.031, respectively; **Table 9-10**). The correlation with the proportion of stream length buffered was weaker (R^2 = 0.761 and 0.772; P <0.079 and 0.072, respectively) and may be a result of the correlation between the proportion of the site harvested and the proportion of stream buffered. In contrast, total-P export was uncorrelated with all three variables.

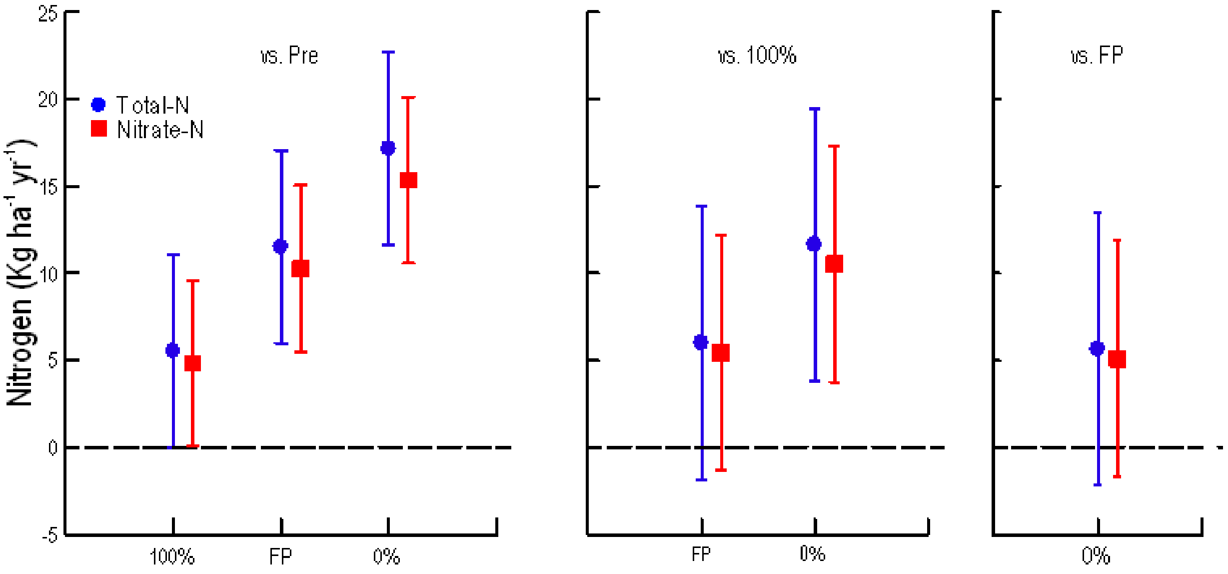

Figure 9-8. Results of the test of the six hypotheses for total-N and nitrate-N export. The left panel shows the mean change in export (and 95% confidence intervals) post-harvest in each treatment, hypotheses a), b), and c). The right panel compare the change in export among the three treatments, hypotheses d), e), and f). The horizontal dashed line equals no change. Confidence intervals that do not cross the dashed line indicate the hypothesis is rejected at P <0.05).

Table 9-8. Mean annual nutrient export (kg ha^{-1} yr^{-1}) and mean discharge for pre- and post-harvest periods for each treatment site and the corresponding period in the unharvested reference site.

Block	Treatment	Period	Treatment				Reference			
			Total-N	NO_3-N	Total-P	Flow (Ls^{-1})	Total-N	NO_3-N	Total-P	Flow (Ls^{-1})
WIL1	100%	Pre	1.78	1.70	0.14	16	12.16	10.37	0.31	11
		Post	8.16	7.26	0.17	18	9.91	8.42	0.27	9
	FP	Pre	14.42	13.32	0.35	9	12.16	10.37	0.31	11
		Post	24.67	22.48	0.46	13	9.91	8.42	0.27	9
	0%	Pre	7.71	7.70	0.08	16	10.14	8.85	0.28	10
		Post	18.85	17.54	0.11	22	9.48	8.04	0.26	9
OLYM	100%	Pre	10.64	9.23	0.28	19	5.79	4.61	0.25	44
		Post	11.35	10.45	0.40	22	5.91	5.23	0.22	51
	FP	Pre	5.19	4.85	0.20	15	7.28	6.48	0.28	61
		Post	13.97	12.42	0.26	17	5.87	4.96	0.23	47
	0%	Pre	11.95	11.15	0.18	16	6.21	5.27	0.25	50
		Post	32.86	29.97	0.27	25	6.16	5.43	0.23	53

Table 9-9. Results of hypothesis tests for nutrient export, described in section 9-4.5. Hypotheses a), b), and c) compare post-harvest exports with pre-harvest in each treatment and d), e), and f) compare the post-harvest changes among the three buffer treatments. Comparisons in **bold** indicate the hypothesis was rejected at P <0.05. P-values were not adjusted for multiple comparisons.

			95% C.I.	
Hypothesis	**Change**	**P-value**	**Lower**	**Upper**
Total-N (kg ha^{-1} yr^{-1})				
a) Pre vs. Post-100%	5.52	0.051	-0.02	11.05
b) Pre vs. Post-FP	**11.52**	**0.0007**	**5.98**	**17.05**
c) Pre vs. Post-0%	**17.16**	**<0.0001**	**11.62**	**22.70**
d) 100% vs. FP	6.00	0.121	-1.83	13.83
e) 100% vs. 0%	**11.64**	**0.007**	**3.81**	**19.47**
f) FP vs. 0%	5.65	0.142	-2.18	13.48
Nitrate-N (kg ha^{-1} yr^{-1})				
a) Pre vs. Post-100%	**4.83**	**0.048**	**0.04**	**9.61**
b) Pre vs. Post-FP	**10.24**	**0.001**	**5.46**	**15.03**
c) Pre vs. Post-0%	**15.35**	**<0.0001**	**10.56**	**20.13**
d) 100% vs. FP	5.42	0.107	-1.35	12.18
e) 100% vs. 0%	**10.52**	**0.005**	**3.75**	**17.29**
f) FP vs. 0%	5.11	0.126	-1.66	11.87
Total-P (kg ha^{-1} yr$_{-1}$)				
a) Pre vs. Post-100%	**0.10**	**0.006**	**0.03**	**0.16**
b) Pre vs. Post-FP	**0.13**	**0.001**	**0.07**	**0.20**
c) Pre vs. Post-0%	**0.09**	**0.010**	**0.03**	**0.16**
d) 100% vs. FP	0.03	0.460	-0.06	0.12
e) 100% vs. 0%	-0.01	0.857	-0.10	0.08
f) FP vs. 0%	-0.04	0.362	-0.13	0.05

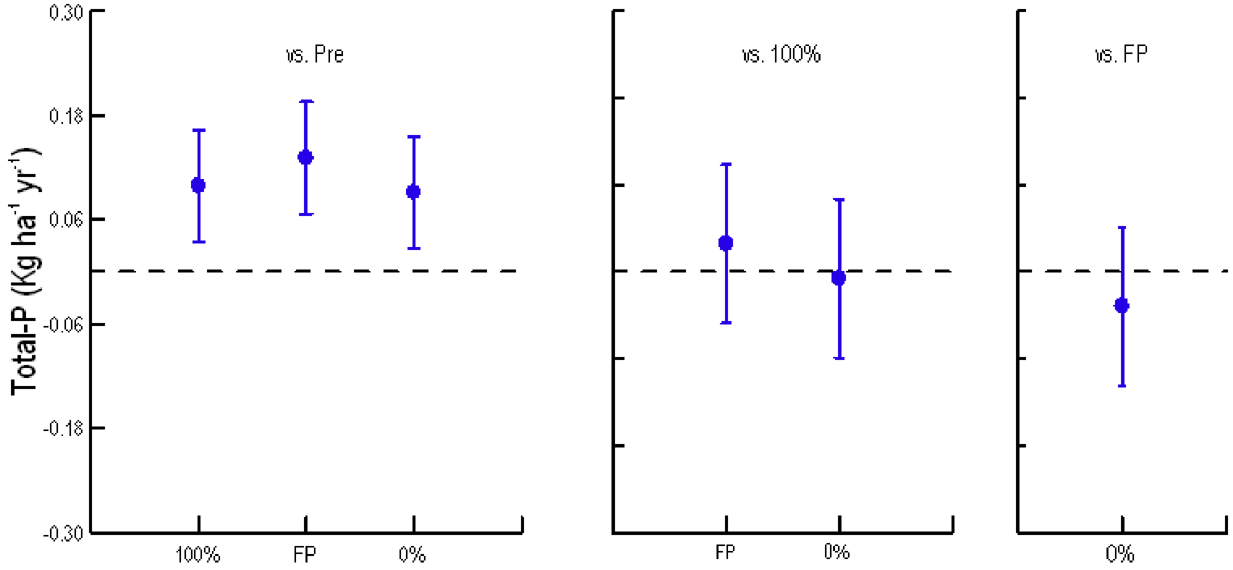

Figure 9-9. Results of the test of the six hypotheses for total-P export. The left panel shows the mean change in export (and 95% confidence intervals) post-harvest in each treatment, hypotheses a), b), and c). The right panel compares the change in export among the three treatments, hypotheses d), e), and f). The horizontal dashed line equals no change. Confidence intervals that do not cross the dashed line indicate the hypothesis is rejected at P <0.05).

Table 9-10. Pearson correlation coefficients between the change in the export of total-N, nitrate-N, total-P, and runoff by % of basin harvested, % stream buffered, and the change in runoff. Comparisons in **bold** indicate the hypothesis was rejected at P <0.05. P-values were not adjusted for multiple comparisons.

Change in	% Basin Harvested	% Stream Buffered	Change in Runoff
Total-N	**0.854 (0.031)**	–0.761 (0.079)	**0.970 (0.001)**
Nitrate-N	**0.852 (0.031)**	–0.772 (0.072)	**0.971 (0.001)**
Total-P	–0.456 (0.363)	0.335 (0.517)	0.060 (0.909)
Runoff	0.715 (0.110)	–0.760 (0.080)	NA

9-6. DISCUSSION

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 Fredricksen 1988; Cairns and Lajtha 2005; Meininger 2011), and British Columbia (Feller and Kimmins 1984). Post-harvest, mean, flow-weighted N (total-N and nitrate-N) concentrations were consistently higher and, coupled with the higher runoff, led to significantly ($P < 0.05$) greater export of total-N and nitrate-N post-harvest from all three buffer treatments relative to the reference sites. Higher N concentrations and higher flows (see Chapter 8 – *Discharge*) were both a result of the buffer treatments.

We observed seasonal patterns of low summer nitrate-N concentration followed by higher concentration during fall high flows similar to those seen in western, conifer-forested watersheds with elevated N-inputs (Lajtha *et al.* 1995; Pardo *et al.* 1995; Williams *et al.* 1996; Stottlemyer and Toczydlowski 1999) and in watersheds draining young forest stands with elevated stream nitrate-N concentrations (Cairns and Lajtha 2005). In studies of watersheds with low N-exports (<1 to 2 kg ha^{-1} yr^{-1}), the opposite pattern (i.e., higher summer concentrations) was observed (Swank and Vose 1997; Edmonds *et al.* 1998). Unlike the streams observed by Compton and colleagues (2003), our streams, which also had low atmospheric N inputs (NADP 2013) and moderate concentrations of nitrate-N, displayed seasonality in nitrate-N concentrations even pre-harvest. The general increase in N concentration in the fall is likely a result of leeching of accumulated soluble N from the forest floor as well as increased litter input and entrainment of instream organic matter.

Pre-harvest export of total-N and nitrate-N varied markedly among our eight sites. Nitrate-N concentration and export can be influenced by atmospheric deposition (Feller 2005) and the proportion of the watershed in red alder or mixed hardwood-conifer forests (Wigington *et al.* 1998; Compton *et al.* 2003). Mean annual atmospheric nitrogen deposition measured at the Hoh River Ranger Station (WA14) and near La Grande, Washington (WA21) was low, averaging 0.95 and 0.88 kg N ha^{-1} yr^{-1}, respectively, from 2006 to 2011 (NADP 2013). This is much less than is typical for eastern states or downwind of urban or industrial centers and it is unlikely that it varied enough among our study sites to be a major factor in explaining the variability in pre- or post-harvest export.

We did not have basin-wide estimates of vegetation composition; however, the riparian vegetation in our study sites was dominated by conifers, ranging from 78 to 100% of total basal area within the 50-ft (15.2-m) riparian zone (see Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers*). Although nitrate-N concentration (Wigington *et al.* 1998; Compton *et al.* 2003) and nitrate-N export (Compton *et al.* 2003) from western Oregon forests was observed to increase with the proportion of the watershed or riparian stand in alder or mixed hardwood-conifer stands, it is unlikely that N-fixation by alders was a driving factor in the wide pre-harvest range of nitrate-N export from the study basins. There was a pre-harvest pattern in the Olympic block of increasing nitrate-N concentration and export with increasing proportion of hardwood riparian vegetation; however, with all sites combined we found no significant correlation between pre-harvest nitrate-N export or the relative increase in nitrate-N export and the proportion of the riparian stand in hardwoods ($P > 0.05$), as was seen by Compton and colleagues (2003), probably because conifers dominated all of our sites. It is interesting to note that post-harvest nitrate-N export in the WIL1-FP and OLYM-0% exceeded 20 kg ha^{-1} yr^{-1}. This is near the levels reported by Compton and colleagues (2003) for watersheds with 50% or more in broadleaf and mixed forest cover.

Mean post-harvest export values of nitrate-N among the treatment watersheds were as variable as the pre-harvest values, ranging from 7.26 kg ha^{-1} yr^{-1} in the WIL1-100% to 29.97 kg ha^{-1} yr^{-1} in the OLYM 0%. Brown and colleagues (1973) estimated nitrate-N loss from Flynn Creek (unharvested) and Deer Creek (25% harvested), both in the Oregon Coast range and dominated by alder, at approximately 25 to 35 kg ha^{-1} yr^{-1}; while nearby Needle Branch, which was more similar to our sites in forest cover (80% conifer dominated), increased from less than five to more than 15 kg ha^{-1} yr^{-1} immediately after harvest. By 2006, nitrate-N export from Needle Branch had increased to 18 kg ha^{-1} yr^{-1}, which was attributed to the increase in red alder (*Alnus rubra*) forest cover over time (Hale 2007). Dahlgen (1998) estimated that nitrate-N export increased after clearcut harvest of a Douglas fir/redwood (*Pseudotsuga menziesii*/*Sequoia sempervirens*) forest in northern California from 0.4 to 1.8 kg ha^{-1} yr^{-1}. He attributed the increased N flux to higher flows and increased stream water concentrations due to mineralization and leaching of nitrate from the soil. Dahlgren (1998) suggested that the relatively low export and quick recovery may have reflected the rapid growth of redwood stump sprouts and recovery of plant uptake. Feller and Kimmins (1984) reported a doubling of nitrate-N export from approximately 3.7 to 7.0 kg ha^{-1} yr^{-1} following a clearcut harvest and an increase from 0.7 to 4.4 kg ha^{-1} yr^{-1} following a clearcut with slash burning. Sollins and McCorison (1981) reported an increase in N export of less than 2 kg ha^{-1} yr^{-1} after clearcutting old-growth conifer forest in the Oregon Cascade Mountains.

In this study nitrate-N comprised 87 to 99% of total-N exported from the treatment sites and 79 to 90% exported from the reference sites (**Appendix Table 9-B-1**) even though atmospheric N deposition at the two nearest NADP monitoring sites during the study was low (NADP 2013) and the riparian forests were conifer dominated (78 to 100% conifer; **Table 9-2**). Scott and colleagues (2007) found dissolved organic N dominated in rivers of all sizes across the U.S. and there are similar findings from temperate forests with low atmospheric inputs (Sollins *et al.* 1980; Hedin *et al.* 1995). However, Cairns and Lajtha (2005) found that dissolved organic N comprised 24, 52, and 51% of total dissolved N export in young, middle-aged, and old-growth watersheds, respectively. In areas with high N deposition, N export is generally dominated by nitrate-N (Ohrui and Mitchell 1977). Nitrate-N dominated export in many hardwood-dominated streams in the Salmon River watershed of western Oregon (Compton *et al.* 2003). Compton and colleagues (2003) suggested that high nitrate-N concentrations and lack of seasonality indicated that many of their watersheds were nitrate saturated (Stoddard 1994), probably from N-fixation by red alders. Liles (2005) and Taylor (2008) observed that total-N concentration in headwater streams in Capitol Forest near Olympia, Washington was dominated by nitrate-N, ranging from 40 to 70%.

Neither SRP nor total-P concentration changed post- harvest (**Tables 9-6** and **9-7**). This was not unexpected because the riparian buffers and equipment limitation zones along the harvested stream reaches (see Chapter 3 – *Management Prescriptions*) minimized ground disturbance near the stream and largely prevented direct input of sediment to the stream (see Chapter 10 – *Sediment Processes*). In addition, phosphate movement through the soil may be inhibited if soil pH decreases after harvest (Fredricksen 1971; Harr and Fredricksen 1988; Tiedemann *et al*. 1988; Martin and Harr 1989). An increase in the instream uptake of P may have had a role in keeping SRP concentration low, especially in streams with substantial input of woody debris after harvest (Warren *et al.* 2007). Total-P exports did increase post-harvest in all treatments; however, this was a function of higher discharge.

9-7. CONCLUSIONS

In all streams, nitrate-N was the dominant form of N, comprising at least 78% of annual total-N export. Clearcut harvest of the trees reduced the uptake of nitrate-N from the soil, increased stream discharge, and left a large amount of slash on the forest floor. As a result, mean flow-weighted concentration of total-N and nitrate-N increased at all buffer treatment sites post-harvest, however the magnitude was variable and significant only for the 0% treatment. The export of total-N increased in the FP and 0% treatments and nitrate-N increased in all buffer treatments. Increases in N export were correlated with increased stream discharge and the proportion of the site that was harvested. Pre-harvest total-P concentration was low and remained so post- harvest, although P export increased slightly post- harvest in all treatments due to the increase in discharge.

9-8. REFERENCES

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APPENDICES

APPENDIX 9-A. EFFECTS OF STORING WATER SAMPLES IN THE ISCO PUMP SAMPLER FOR EXTENDED PERIODS PRIOR TO PRESERVATION AND LABORATORY ANALYSIS

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Introduction

We have conducted several water quality studies that employ remotely triggered ISCO pump samplers to collect water samples during high flow/high turbidity events. The study sites were remote and distributed across industrial forestlands in western Washington. The use of remote sampling devices was necessary to sample during these short-lived storm events.

Sampling protocols for nutrients (total nitrogen (N), nitrate-N, and total phosphorus (P)) include acidifying the sample to a pH less than 2 (using concentrated sulfuric acid for total-N and nitrate-N and hydrochloric acid for total-P) within 15 minutes of collection, storing the bottle at ≤6°C, then conducting the chemical analysis within 28 days of collection. We were often unable to visit the field sites until days to weeks after samples were collected because of long travel times and limited staff availability. As a result, samples could not be acidified immediately after collection. Instead, they were stored at ambient temperatures in the sampler until we could retrieve them. When retrieved, the sampling protocols were followed (preservative was added, the samples were stored at ≤6°C, and the analysis was conducted within 28 days of collection).

The objective of this study was to quantify the effects of this modification to the sampling protocols on the measured concentration of total-N, nitrate-N, and total-P.

Methods

We collected four replicate water samples for the analysis of total-N, nitrate-N, and total-P from each of four Type N streams located on industrial forestland in southwestern Washington State. Each of the four samples from each stream was filled sequentially from the stream. We followed the sampling protocol on one sample (acidified immediately with the appropriate preservative, stored at ≤6°C, and analyzed within 28 days). The second through fourth samples from each stream were held for approximately one week (six or seven days), three weeks (20 or 21 days), or four weeks (27 or 28 days) at ambient temperatures (mean daily air temperature of 9.9°C to 15.1°C), respectively, then preserved and analyzed within 28 days of collection.

We calculated two variables for presentation and analysis. The first is the difference (DIFF) between each of the three samples held for analysis and the sample acidified and cooled immediately after collection.

$$DIFF_i = C_i - C_0 \tag{9-A-1}$$

where: C_i = concentration at hold time *i* (one, three, or four weeks) and
C_0 = concentration of sample processed with no hold time.

The second is the percent difference (%DIFF) between the held samples and the sample acidified and cooled immediately after collection.

$$\%DIFF_i = \left[100 * \frac{DIFF}{C_0}\right] \tag{9-A-2}$$

We also used the results of 22 replicate sample analyses of water collected from these same streams where both replicate samples were processed following the protocol (with no holding time). These results quantify the natural variability in the stream, variability added by collecting and processing the sample, and laboratory variability. We compared the replicate analyses results with the results from samples held prior to processing, to estimate the effect of holding the samples on the measured concentration of total-N, nitrate-N, and total-P. We calculated DIFF and %DIFF for each replicate pair.

The results are presented graphically by plotting:

1) The measured concentration of the four replicate samples vs. the number of days that the samples were held prior to processing;
2) DIFF vs. the number of days the samples were held; and
3) %DIFF vs. the number of days the samples were held and comparing with laboratory QA/QC standards.

We used a one-way analysis of variance with four levels of Holding Time (none, one week, three weeks, and four weeks) to evaluate differences in DIFF and %DIFF. The null hypotheses are:

$$\mathrm{DIFF}_t = \mathrm{DIFF}_{rep} \tag{9-A-3}$$

and

$$\%\mathrm{DIFF}_t = \%\mathrm{DIFF}_{rep} \tag{9-A-4}$$

where: DIFF_{rep} and $\%\mathrm{DIFF}_{rep}$ are the difference and percent difference, respectively, between replicate samples with no holding time and
DIFF_t and $\%\mathrm{DIFF}_t$ are the difference and percent difference, respectively, between replicate samples held for t = 1, 3 or 4 weeks.

Means and 95% confidence intervals (CI) are presented.

Results

Mean total-N concentration in the four streams ranged from 198 to 696 µg L^{-1}. Mean DIFF ranged from −13 to 33 µg L^{-1} and %DIFF ranged from −3% to 10% (**Figure 9-A-1**). Mean nitrate-N concentration was only slightly less than total-N and ranged from 168 to 668 µg L^{-1}. DIFF ranged from −42 to 5 µg L^{-1} and %DIFF ranged from −6% to 2%. Mean total-P concentration was 11 to 23 µg L-1. DIFF ranged from −7 to 5 µg L^{-1} and %DIFF ranged from −29% to 30%.

There was no consistent relationship in DIFF or %DIFF and the time the sample was held (**Figure 9-A-1**). For all three analytes, DIFF and %DIFF were distributed relatively evenly between positive and negative values and there was no consistent pattern of increasing or decreasing values as a function of the time the sample was held. For example, measured concentration at some streams decreased then increased as holding time increased, while others responded in the opposite direction.

Laboratory guidelines at the Manchester Environmental Lab state that the %DIFF for laboratory replicates (samples that were split in the lab, then analyzed separately) should be between −20% and 20% for samples where the measured concentration is greater than five times the reporting limit. All total-N and nitrate-N concentrations were greater than five times the reporting limit of 10 and 25 µg N L^{-1}, respectively, and the %DIFF values calculated for samples processed immediately and held for one to four weeks met the guidelines for replicate samples. The initial total-P concentration from each of the four streams was less than five times the reporting limit of 5 mg L^{-1} (**Figure 9-A-1**, bottom left) so the guidelines do not apply. In spite of the low P concentrations, the absolute value of %DIFF was less than 30% in all cases and half of the values were less than 20%.

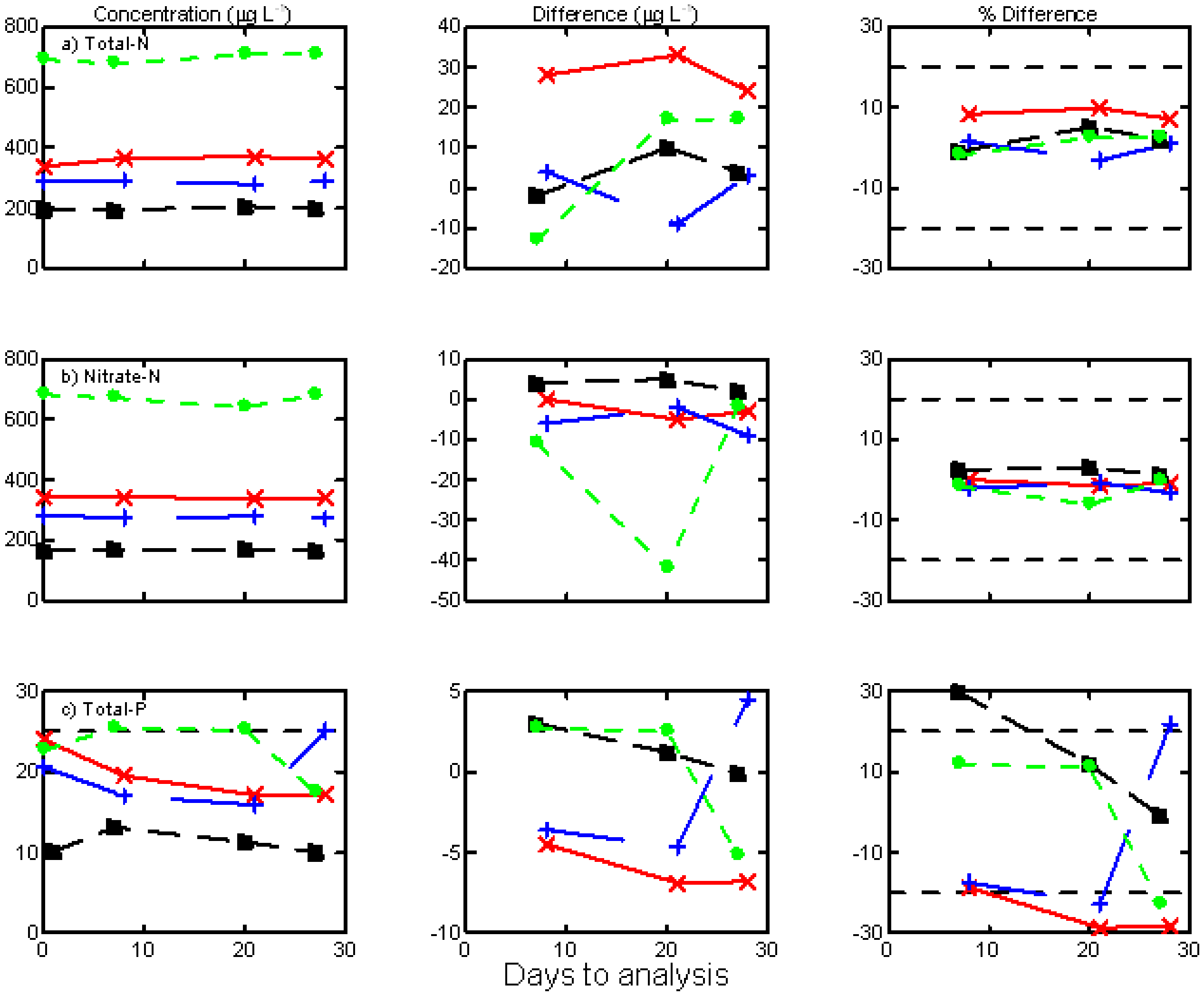
The one-way analysis of variance revealed no significant difference between the replicate samples and the analysis of samples held for one, three or four weeks for either DIFF or %DIFF (**Figure 9-A-2**).

Conclusions

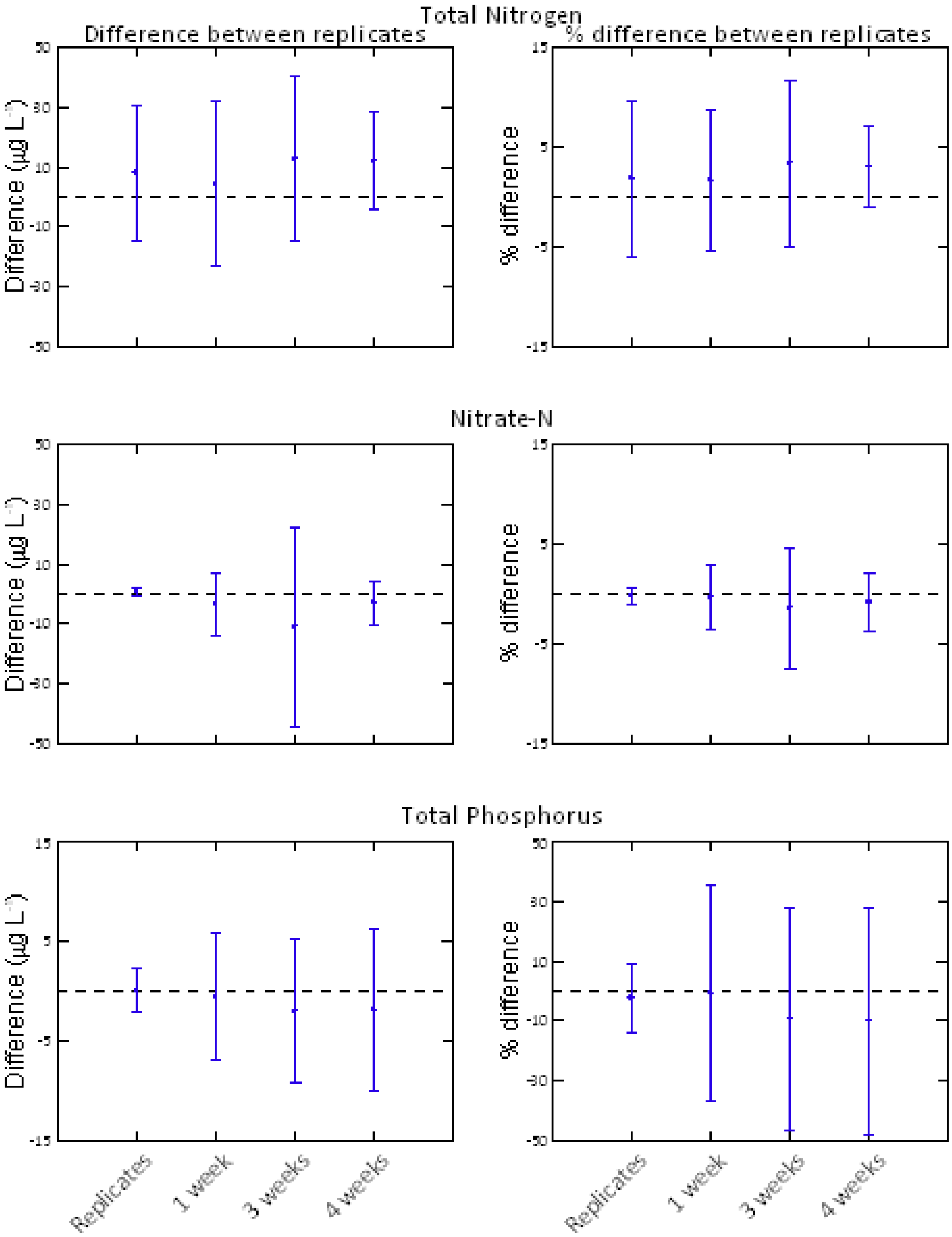
Our data show no systematic bias introduced by holding samples for up to four weeks before processing. The differences were similar in magnitude to laboratory replicate samples and, when expressed as a percentage of the initial concentration, the results met the guidelines set for laboratory replicate analyses.

Figure 9-A-2 shows wider 95% confidence intervals for nitrate-N and total-P samples held before processing. This is due to the much larger number of replicate samples (n = 22) compared to the held samples (n = 4 for each period).

Overall, any effect of holding our samples in the field before processing was undetectable and is unlikely to measurably affect nutrient export calculations.



Appendix Figure 9-A-1. Plots of concentration, difference between analyses, and %DIFF between analyses vs. number of days between sample collection and processing shown in columns one through three, respectively. Rows one through three show total-N, nitrate-N, and total-P. The different colored lines represent the four streams sampled. The dashed horizontal line in the bottom left plot shows the concentration equal to five times the reporting limit for total-P. Dashed lines in the right column are the laboratory guidelines for acceptable range in %DIFF.



Appendix Figure 9-A-2. Mean and 95% confidence intervals (CI) of replicate analyses (n = 22) and the difference between samples processed immediately and those held for one, three, or four weeks before processing (n = 4 for each category). Confidence intervals that include zero (dashed horizontal line) indicate no difference in mean value. The smaller confidence intervals in the replicate samples is a function of the much larger sample sizes compared to the ones held before processing .

APPENDIX 9-B. ESTIMATED ANNUAL NUTRIENT EXPORT

Appendix Table 9-B-1. Estimated annual nutrient export (kg ha^{-1} yr^{-1}) for each pre- and post-harvest year at all harvested sites. Export from the reference site is for the corresponding period. Nitrate-N as a percentage of total-N export is shown in the right two columns.

Block	**Treatment**	**Period**	**Treatment**			**Reference**			**% N as Nitrate-N**	
			Total-N	**NO3-N**	**Total-P**	**Total-N**	**NO3-N**	**Total-P**	**Treatment**	**Reference**
WIL1	100%	Pre 2	1.83	1.86	0.14	14.39	12.06	0.34	102%	84%
		Pre 1	1.74	1.54	0.13	9.93	8.68	0.28	89%	87%
		Post 1	6.94	6.14	0.15	8.86	7.58	0.25	88%	85%
		Post 2	9.38	8.38	0.18	10.95	9.27	0.30	89%	85%
	FP	Pre 2	14.33	13.15	0.34	14.39	12.06	0.34	92%	84%
		Pre 1	14.51	13.48	0.36	9.93	8.68	0.28	93%	87%
		Post 1	20.80	19.10	0.41	8.86	7.58	0.25	92%	85%
		Post 2	28.53	25.86	0.51	10.95	9.27	0.30	91%	85%
	0%	Pre 1	7.71	7.70	0.08	10.14	8.85	0.28	100%	87%
		Post 1	17.19	16.13	0.11	8.55	7.29	0.24	94%	85%
		Post 2	20.51	18.96	0.12	10.41	8.79	0.28	92%	84%
OLYM	100%	Pre 1	10.64	9.23	0.28	5.79	4.61	0.25	87%	79%
		Post 1	10.05	9.19	0.37	5.90	5.29	0.22	91%	90%
		Post 2	12.65	11.71	0.43	5.93	5.16	0.23	93%	87%
	FP	Pre 1	5.19	4.85	0.20	7.28	6.48	0.28	93%	89%
		Post 1	11.52	10.32	0.23	5.23	4.09	0.22	90%	78%
		Post 2	16.43	14.52	0.29	6.51	5.84	0.24	88%	90%
	0%	Pre 2	11.91	11.30	0.17	7.29	6.49	0.28	95%	89%
		Pre 1	11.99	11.00	0.19	5.13	4.05	0.23	92%	79%
		Post 1	30.13	27.43	0.25	6.42	5.73	0.24	91%	89%
		Post 2	35.59	32.50	0.30	5.90	5.14	0.23	91%	87%

CHAPTER 10 - SEDIMENT PROCESSES

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Stephanie Estrella, Frithiof T. Waterstrat, and Reed Ojala-Barbour*

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10-1. ABSTRACT

We monitored suspended sediment export (SSE) as part of a larger study evaluating the effects of alternative length riparian buffers 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 (BACI) design at the watershed scale, and was restricted to headwater (<45 ha) basins with relatively competent lithologies. Blocks consisted of at least one unharvested site that served as a reference, and sites that were clearcut harvested outside of a riparian management zone (RMZ) with 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 of 50% of the perennial stream network buffered); or (3) a clearcut RMZ (i.e., 0% buffer). In addition to RMZ buffers, all sites had unstable slope buffers where necessary and a two-sided 30-ft (9.1-m) equipment limitation zone along all stream channels. SSE was calculated from continuous discharge and turbidity measurements. Suspended sediment concentration (SSC) was estimated via linear regression from the turbidity data. SSE was estimated near the perennial non-fish-bearing (Type Np) basin outlet in two complete study blocks (n = 8). We monitored all sites for one or two years pre-harvest and two years post-harvest. Though our primary objective was to evaluate the potential differences in SSE among treatments, we also measured three indicators of sediment supply (road surface erosion, stream-delivering surface erosion, and uprooted trees) with the intent of using these data to identify, or eliminate from consideration, potential sources of sediment if changes were seen in our SSE estimates. Sediment supply indicators were measured across all study sites (n = 17) and blocks (n = 5).

The sites monitored for SSE appeared to be supply limited with respect to suspended sediment, both pre- and post-harvest. Most of the sediment export occurred during eight discrete late fall or early winter storm events, and the relative magnitude of export was stochastic across sites and treatments. By buffer treatment, annual export range ranged from a 15 t ha^{-1} *decrease (P = 0.577) in the FP treatment to a 56 t ha*$^{-1}$ *increase (P = 0.051) in the 100% treatment. Overall annual SSE was 24.5 t ha*$^{-1}$ *greater (P = 0.127) in the post-harvest period, but no significant (P <0.05) effects of buffer treatment were detected. In four of the six buffer treatment sites, SSE was greater during the harvest or in the post-harvest period, but spikes in sediment export were of similar magnitude to those observed in one of the two reference sites during the same period. Given the limited number of sites monitored for SSE and the limited number of sediment-generating storms that occurred during the study period, we were unable to separate treatment effects from natural variability to draw strong conclusions about the relative effectiveness of the buffer treatments. Finally, our sediment supply indicators lacked the temporal resolution and accuracy needed to draw strong conclusions about changes in sediment supply or links to SSE, though the data do suggest that windthrow is unlikely to be the primary source for treatment period or post-harvest spikes in SSE.*

10-2. INTRODUCTION

Sediment transport is a function of both sediment supply and transport capacity (Schumm 1971), and forest practices have the potential to affect both (Gomi *et al*. 2005). Forest practices may increase headwater sediment supply through a range of processes including road surface erosion, windthrow, bank erosion and other mass wasting processes (Roberts and Church 1986; Grizzel and Wolff 1998; Araujo *et al*. 2013), and can change transport through alterations in the frequency or magnitude of sediment transporting events (Gomi *et al*. 2005; Alila *et al*. 2009; Kaufmann *et al*. 2009).

Historically, forest practices have been shown to increase suspended sediment loads and 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 (Wilkins and Peterson 2000; Stoddard and Hayes 2005). Forest roads have long been implicated in increasing sediment yield to headwater streams (Megahan and Kidd 1972; Beschta 1978; Reid and Dunne 1984) through extensions of the drainage network and routing of road surface sediment to streams (Wemple *et al*. 1996; Wemple *et al*. 2001), as well as changes in the frequency and magnitude of road-related mass wasting. On existing unpaved road networks, changes in sediment delivery are most likely to be associated with changes in either the road surface condition or traffic levels (Ramos-Scharrón and MacDonald 2005; Sheridan *et al*. 2006; Araujo *et al*. 2013).

Given that riparian zones are adjacent to streams, disturbance within the riparian zone is also likely to affect sediment supply. Tree windthrow is commonly observed following harvest, and numerous studies have suggested that windthrow could be a source of increased stream sediment (Grizzel and Wolff 1998; Liquori 2006). However, riparian vegetation is generally very effective in trapping sediment (Lakel *et al*. 2010; Sweeney and Newbold 2014), and since windthrow preferentially occurs along the edges of the buffer away from the stream, only a small number of windthrown trees are likely to deliver sediment to the stream channel (Stewart *et al*. 2006; Bahuguna *et al*. 2010; Schuett-Hames *et al*. 2012). Bank erosion both delivers sediment directly to the channel and is an expected geomorphic response to increased discharge (Prosser *et al.* 2001).

10-3. OBJECTIVES

Washington State law (Chapter 90.48 Revised Code of Washington) requires that Forest Practices rules are effective in creating conditions that comply with the Clean Water Act and state water quality standards (Washington Department of Ecology 2010), and one of the resource objectives in the Forest Practices Habitat Conservation Plan is to:

> “[minimize] to the maximum extent practicable, the delivery of management-induced coarse and fine sediment to streams by protecting stream bank integrity, providing vegetative filtering, protecting unstable slopes, and preventing the routing of sediment to streams.”[1]

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[1] FPHCP Schedule L-1 Resource Objective for Sediment.

In this study, suspended sediment export (SSE) was measured at a subset of sites to determine whether there was a change in sediment export that could be attributed to forest practices, and whether the magnitude of change varied among buffer treatments. A secondary objective was to identify the source (e.g., road surface erosion) of any observed changes in SSE so we also monitored several sediment supply indicators at a coarse scale to help explain any unusual patterns in suspended sediment concentration (SSC) or SSE.

The research questions were:

1) Were there significant changes in SSE in the two years following harvest?

2) Does the Washington Road Sediment Erosion Model (WARSEM) predict a large change in road sediment delivery to streams following harvest?

3) Was there visual evidence of sediment delivery from windthrow following harvest?

4) Was there a visible change in the length of channel affected by seasonal bank erosion after harvest?

10-4. METHODS

10-4.1. STUDY DESIGN AND SITE DESCRIPTION

As described in Chapter 2 – *Study Design* and Chapter 3 – *Management Prescriptions*, the overall study used a replicated Before-After Control-Impact (BACI) design with five blocks consisting of four treatments including a reference. All sites were located in competent lithologies. The three buffer treatments consisted of a watershed-scale clearcut (minus buffers on unstable slopes) and a Riparian Management Zone (RMZ) buffer treatment that was: (1) clearcut (0% buffer); (2) a Washington State Forest Practices (FP) two-sided 50-ft (15.2-m) wide riparian buffer along a minimum of 50% of the perennial stream (Np) network; or (3) a two-sided 50-ft (15.2-m) wide buffer retained along the entire Np network (100% buffer; see Chapter 2 – *Study Design*). All buffer treatments maintained a two-sided 30-ft (9.1-m) wide equipment limitation zone (ELZ) that extended horizontally from the outer edge of the bankfull width of the channel, as required by Forest Practices rules.

For cost and logistical reasons, we limited SSE monitoring to the eight sites in the Olympic and Willapa 1 blocks, which each contained all three buffer treatments and a reference (**Figure 10-1**). We analyzed SSE using the BACI design. Several metrics related to indicators of sediment supply were monitored at a coarse scale in case they might be used to explain unusual patterns in SSC or SSE, if they were detected. We used three different designs (i.e., BACI, Before-After, and After-Only) in our analyses of sediment supply indicators: we looked for visual evidence of sediment delivery from windthrow in all 17 study sites and analyzed the data under the original BACI design; we examined stream-delivering surface erosion in all 17 sites in only the post-harvest period (After-Only); and we conducted road surface erosion modeling in the 12 sites with roads crossing the Type Np stream (Before-After). Catchment areas ranged from 11.8 to 44.3 ha.

Average Np channel gradients ranged from 16% to 31%, and the sites were underlain primarily by basalt flows and flow breccias (**Table 10-1**).

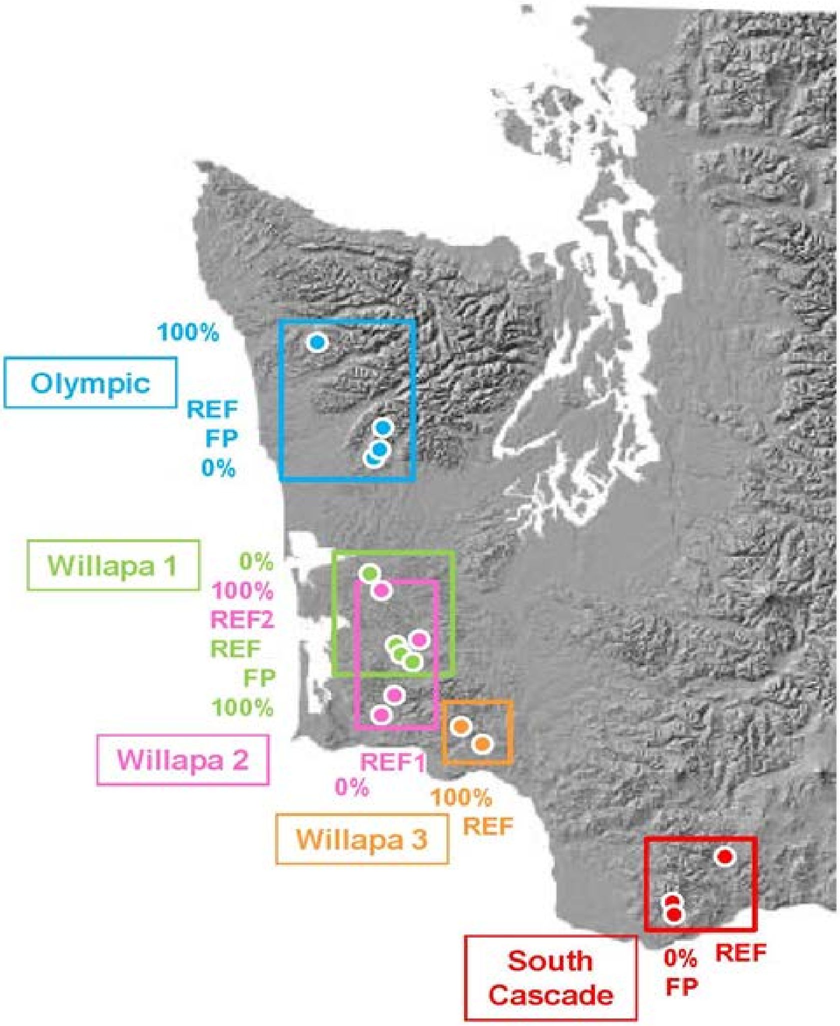

Figure 10-1. The Type N study sites and treatment type (i.e., REF, 100%, FP, 0%) in the five study blocks(color-coded). Olympic and Willapa 1 are the only complete blocks and the only ones gauged for discharge, turbidity and SSC. Only 12 of the 17 sites contained roads that crossed Type Np streams.

Table 10-1. Elevation, lithology, stream gradient and stream order (Strahler 1952) for the sites in the Olympic (OLYM) and Willapa 1 (WIL1) blocks. Elevation is the elevation at the flume and stream gradient is the average stream gradient for the entire Type Np stream network as calculated using a 10-m digital elevation model in ArcMap (ESRI 2004).

Block	Treatment	Area (ha)	Elevation (m)	Lithology	Stream Gradient (%)	Stream Order
OLYM	REF	44.3	163	Basalt flows and flow breccias	18	3
	100%	22.1	72	Tectonic breccia	27	3
	FP	17.3	277	Basalt flows and flow breccias	25	3
	0%	13.1	233	Basalt flows and flow breccias	31	2
WIL1	REF	11.8	200	Basalt flows and flow breccias	19	2
	100%	26.2	198	Basalt flows and flow breccias	18	2
	FP	14.4	197	Basalt flows and flow breccias	19	1
	0%	27.7	87	Terraced deposits	16	3

10-4.2. DATA COLLECTION AND PROCESSING

10-4.2.1. Turbidity and SSC

We monitored water discharge and turbidity in the eight sites of the Olympic and Willapa 1 blocks using a system from Forest Technology Systems (FTS; www.FTSenvironmental.com) consisting of:

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

Methods for measuring discharge are described in Chapter 8 – *Discharge*. Due to a poor flow-versus-stage height relationship in the OLYM-100% in the first year of the study, we estimated flows in the OLYM-100% from September to December 2007 by linear regression with the OLYM-0%. We used pre-harvest data collected after 1 January 2008 to calibrate the regression model.

We programmed the FTS system to conduct Turbidity Threshold Sampling (TTS; Lewis and Eads 2009) so that a water sample was collected by the ISCO pump sampler when stage height and turbidity exceeded specified thresholds for two consecutive measurements. This ensured that water samples were collected across the range of turbidity values (turbidity 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). The measured SSC values and the corresponding turbidity data were used to build a regression model to predict SSC for the entire data record. We calculated instantaneous SSE as the product of the estimated SSC and flow.

We recorded turbidity at 10-minute intervals. At each interval the DTS-12 turbidity sensor took 100 readings over a five second period and reported the summary statistics of these readings. The turbidity statistics included minimum, median, mean, maximum, and variance. We observed that variance was very low (<2) in the absence of air bubbles or stream bottom fine sediments. Air bubble interference increased the number of high readings during the five-second window. This appeared first as higher variance and maximum turbidity values and slightly higher mean values. As the frequency of high readings increased, the median and finally the minimum turbidity values were affected. We elected to use minimum turbidity to estimate SSC because the minimum turbidity value was more stable (less influenced by the interference described) than the mean or median value. We used the other turbidity statistics and stage height data to QA/QC the minimum turbidity data.

We followed guidelines in Lewis and Eads (2009) to identify data that were influenced by progressive fouling (biofilm), debris fouling, direct sunlight on the sensor, non- or partial submergence of the sensor, burial of sensor or interference from the stream bottom, and air bubbles entrained in the water.

- Progressive fouling: The DTS-12's wiper mechanism and regular cleaning of the sensor surface prevented discernible biofilm buildup. We did not observe a noticeable step change in turbidity values after cleaning the sensor surface.

- Debris fouling: We did not observe debris on the sensors because the mechanism housing the sensors allowed them to swing downstream in high flows, thereby shedding branches and leaves, and did not present any protrusions to catch debris carried in the flow.

- Direct sunlight: The sensors were recessed slightly in the housing to prevent sunlight from hitting the sensor directly.

- Non- or partial submergence of the sensor: This occurred in two different scenarios, when: (1) water level dropped during summer low flows and the sensor was exposed; and (2) a high-flow event altered the stream channel so that as the water receded and the sensor was stranded. In both cases, the turbidity values were set to zero.

- Burial of sensor or interference from the stream bottom: This sometimes occurred after a flow event when fine sediment was deposited to the extent that it interfered with the measurements and resulted in a continuous high turbidity reading over time.

- Air bubbles: Air bubbles entrained in the water cause high, erratic turbidity readings. When these were observed, we identified 'good' quality records (i.e., with low variance) and interpolated the values between them.

The DTS-12 turbidity sensors were calibrated prior to first use in 2006 then recalibrated at least three times during the study at 13- to 33-month intervals. In eight of the 30 recalibrations, the pre-recalibration turbidity value of at least one of the 10 formazin check standards, ranging from 2 to 1,600 NTUs, was outside the manufacturer's recommended range. Of these eight, four calibration checks were less than 1 NTU outside the recommended range. Three were <2 NTUs outside the recommended range. The final calibration check was <1 NTU outside the range for standards less than 100 NTUs but was 40 NTUs (5%) outside the range of the 800 NTU standard.

We analyzed water samples from the ISCO pump sampler to determine SSC (ASTM Method D 3977 B). We did not analyze samples from sample bottles that had been overfilled by a malfunctioning pump sampler or samples where the pump sample tubing was in contact with the stream bottom. The latter occurred when fine sediments accumulated in the pool to an extent where they reached the tube orifice and sediment was pumped into the sampler. SSC exceeded several thousand mg/L in these samples and were easily identified.

We assigned each day to one of four periods: (1) pre-harvest; (2) during harvest; (3) first year post-harvest; and (4) second year post-harvest. Periods were based on the timing of harvest, which varied from site to site (see Chapter 3 – *Management Prescriptions* for harvest dates and duration for each site), but we collected at least two full years' of post-harvest data for each site.

10-4.3. ESTIMATING SSC AND SSE

We used ordinary least squares regression to estimate SSC for the entire period of record. For turbidity values greater than 12 NTUs, a log-transformation was necessary in order to meet the homogeneity of variance assumption, but logarithmic models resulted in non-zero SSC prediction even when SSC was below detection limits. To eliminate this bias we performed two regression analyses to relate SSC and turbidity. For turbidities less than 12 NTU, we combined data from all sites and performed simple linear regression. Log transformation was not necessary over this range of values. The sample size was 169, the adjusted coefficient of determination (r^2) was 0.773, the residuals were approximately normally distributed and homoscedastic, and the y-intercept was not different from zero ($P < 0.05$). For turbidity values greater than 12 NTU, we initially tested a log-transformed quadratic model:

$$SSC_i = \beta_0 + \beta_1 Turb_i + \beta_2 Turb_i^{\ 2} + \varepsilon_i \tag{10-1}$$

where: *SSC* is the $\log_{10}$-transformed suspended sediment concentration for sampling event *i*,
Turb is the corresponding $\log_{10}$-transformed minimum turbidity value,
β_0, β_1, and β_2 are regression coefficients, and
ε_i is the error term.

The quadratic term was included only for the reference and 100% treatments in the Olympic block (OLYM-REF and OLYM-100%, respectively), and dropped for the other six sites because its inclusion did not improve the adjusted coefficient of determination (r^2), the distribution of the residuals, or the homoscedasticity of the residuals. Sample sizes ranged from 92 to 343 observations. P-values for all regressions were <0.0005. The adjusted r^2 ranged from 0.584 to 0.824. The residuals were examined to ensure that they were approximately normally distributed and homoscedastic. Only 0.02% of the turbidity values were outside the range used in the regressions, therefore we do not believe extrapolating beyond the range of the data used in the regressions introduced substantial error.

Transforming estimates from a regression with a log-transformed dependent variable provides an estimate of the median concentration, rather than the mean, thereby providing an underestimate of the long-term concentration. The smearing estimator (Duan 1983), was applied to the concentration estimates to account for this (after Helsel and Hirsch 1992).

We calculated SSE as the product of the SSC and discharge for each 10-minute period, and average daily SSE as the average of the daily 10-minute periods on a given date multiplied by 144 (the number of 10-minute periods in a day). We then normalized SSE by basin area to calculate export in metric tons/km^2/day. Annual SSE is the sum of normalized daily SSE over 365 days.

The timing of the timber harvest was not synchronized across all buffer treatment sites. The start and end dates and duration of harvest were determined by the landowner and varied among sites within an approximate one-year harvest window. Export from each watershed was highly dependent upon flow, which varied across years in quantity and timing. We used the difference in annual SSE between each buffer treatment site and its reference in the same block (buffer treatment minus reference) over the same time period as a measure of SSE export relative to the reference site. There were two years of pre-harvest data for each site except the WIL1-0%, OLYM-100%, and OLYM-FP, where there was only one year of pre-harvest data. All sites had data collected for two years post-harvest.

10-4.3.1. Sediment Supply Indicators

10-4.3.1.a. Road surface erosion

Since road crossing locations can be an entry point for road sediment into the stream channel, and contribute to SSE load, we used the Washington Road Surface Erosion Model (WARSEM; Dubé *et al.* 2004) to estimate sediment delivery from roads pre- and post-harvest. This model provides a standardized tool for estimating annual road surface erosion and sediment delivery to channels, although data collected following the WARSEM methods cannot be used to estimate road sediment delivery during a period of less than one year and does not have a process for incorporating the effects of observed weather. The model is intended for use on forest roads in Washington State and can be applied on a variety of spatial scales ranging from a single road segment to all roads within a watershed. We used the level 4 analysis for site/segment level monitoring described in the WARSEM manual to conduct basin-scale monitoring. The level 4 analysis is the most detailed and requires field-derived information on road conditions as well as information from landowners about road building, traffic, and road improvements.

We followed the standard field data collection methods outlined in the WARSEM manual to collect road segment data once in the pre-harvest (10 April through 9 July 2007) and once in the post-harvest (9 February through 13 December 2010) period. Twelve of the 17 study sites (four references, four 100% treatments, one FP treatment, and three 0% treatments) had road segments that delivered to the stream network within a study site, including portions of the road network that drained to a stream crossing, drained to a gully connected to the stream, or that drained to a point within 200 feet of a stream. During site visits we determined the segment length; road slope; road configuration; road tread surfacing and average tread width; cutslope ground cover, density and average height; and ditch width, delivery and condition (**Table 10-2**). Since all segments were underlain by a competent lithology type (e.g., basalt, andesite) or by a type that was competent based on weathering and age (e.g., tuff), each segment met the criteria for a low Geologic Erosion Factor rating.

We requested information on the year of road construction, approximate amount of road use during study period, and implementation of Best Management Practices (BMP) from landowners. If roads were very old and exact construction year was not known, then the landowner estimated road construction to the nearest decade. Road use was estimated on an annual basis, including non-use periods, according to the definitions outlined in the WARSEM manual (**Table 10-3**). The BMPs are defined in the Stormwater Management Manual for Western Washington (Washington Department of Ecology 2012) and include road maintenance or improvements intended to reduce sediment production (erosion) or delivery to streams. We asked landowners to identify all BMPs implemented for each road and year, specifying the BMP category from **Table 10-4**, the activity, and timing.

Most road surface erosion variables are not expected to change annually in the absence of maintenance or improvements (Dubé *et al.* 2004), so data collected in 2007 were applied to both 2006 and 2007 modeling. Likewise, since the majority of road construction, reconstruction and decommissioning occurred in 2008 (in preparation for, during and immediately after timber harvest), data collected in 2010 was assumed to be representative of road conditions 2008–2010. Therefore, the only opportunities for differences between the 2006–2007 data and 2008–2010 data were those associated with road building/decommissioning, maintenance, and use. Differences in road sediment production caused by weather cannot be assessed in WARSEM, which assigns a rainfall factor to each road segment based on road location (i.e., Township/Range/Section) and 1961–1990 annual rainfall normals from the PRISM climate model (Dubé *et al.* 2004).

Table 10-2. Road surface erosion variable definitions and values from WARSEM (Dubé *et al.* 2004).

Attribute	Value
Segment Length	Length (ft)
Average Road Slope	Flat or gently sloped (<5%) Moderate (5–10%) Steep (>10%)
Road Configuration	Insloped (or outsloped with wheel tracks) Outsloped Crowned

Table 10-2. (continued)

Attribute	Value
Surfacing	Asphalt Gravel Native Pitrun (indicate if any surface includes ruts or grass)
Average Tread Width	Feet
Cutslope Cover Density	Average % slope covered with vegetation, rock, leaf litter, or other non-erodible material: 0–10%, 10–30%, 30–50%, 50–70%, 70–90%, 90–100%
Cutslope Average Height	Average height of cutslope (slope length): 25, 10, 5, 2,.or 5 ft, or no cutlsope
Ditch Width	Width (ft)
Ditch Delivery	None (no signs of connectivity below drainage point outfall with or without evidence of sediment transport below outfall) Direct (drains directly to stream channel) Within 100 ft (drains to forest floor) Within 200 ft (drains to forest floor) Connected (connects directly to stream via gully)
Ditch Condition	Rock/vegetation (ditch has been rocked or is vegetated) Stable (ditch appears stable; no eroding) Eroding (ditch is eroding/incising)

Table 10-3. Road use categories defined in WARSEM (Dubé *et al.* 2004).

Traffic Category	Road Class	Description
Very Heavy	Highway	Very heavy use by truck and car traffic throughout the year.
Heavy	Main Haul	Heavily used by log truck traffic throughout the year; usually the main access road in a watershed that is being actively logged.
Moderately Heavy	County Road	Wide, county-maintained road that receives heavy residential and/or log truck use.
Moderate	Primary Road	Receives moderate use by log trucks throughout all or most of the year. Usually roads branching off main haul road that head up tributaries or that access large portions of the watershed.
Light	Secondary Road	Receives light log truck use during the year. May occasionally be heavily used to access a timber sale. Receives car/pickup or recreational use.
Occasional	Spur Road	Short road used to access a single logging unit. Used to haul logs for a brief time while unit is logged. On average receives little use.
Abandoned (None)	Abandoned/ Blocked/Orphaned	Road is blocked by a tank trap, boulders, etc. or is no longer used by traffic.

Table 10-4. Best Management Practices (BMP) categories for use in WARSEM (Dubé *et al.* 2004).

Category	Definition/Examples
None	No BMPs implemented.
Minor	Grading, tread reshaping, ditch pulling.
Major	Repair activities such as ditch seeding or armoring, new surfacing or surface treatment, brushing, grass seeding, culvert pulling, culvert inlet or outlet armoring, installation of other drainage structures, road decommissioning, fillslope pullback, new road construction or reconstruction.

To compare differences in the change of road sediment delivery by treatment, we calculated the difference in the average pre- and post-harvest sediment delivery (post − pre) for each site, and present the average differences by treatment. We also compared average pre-harvest delivery to the delivery estimated during the harvest implementation years following this same method. The exact years and number of years included in each site average (pre-harvest, during harvest, post-harvest) varied depending on the timing of timber harvest for the site (**Table 10-5**). Pre-, during, and post-harvest periods for reference sites were identified as follows: Pre-harvest included all years in which every other site in the corresponding block was in the pre-harvest state, harvest included all years in which any site in the corresponding block was being treated (even if all sites were not treated in the same year), and post-harvest included all years in which every other site in the corresponding block was in the post-harvest state.

Table 10-5. The timing of harvest for study sites included in the analysis of road sediment delivery, by calendar year. Highlighted cells identify calendar years within the pre-harvest (light gray), harvest (medium gray), and post-harvest years (dark gray) for each study site.

Block	Treatment	Year				
		2006	2007	2008	2009	2010
OLYM	REF	Pre-			Harvest	Post-
	100%	Pre-			Harvest	Post-
	0%	Pre-			Harvest	Post-
WIL1	REF	Pre-		Harvest		Post-
	100%	Pre-		Harvest		Post-
	FP	Pre-		Harvest		Post-
	0%	Pre-		Harvest		Post-
WIL2	REF1	Pre-		Harvest		Post-
	100%	Pre-			Harvest	Post-
	0%	Pre-		Harvest	Post-	
WIL3	REF	Pre-		Harvest	Post-	
	100%	Pre-		Harvest	Post-	

10-4.3.1.b. Uprooted trees

We collected data on evidence of sediment delivery to the stream associated with uprooted trees that fell between the summer of 2007 and 2008 (one year pre-harvest) and trees that fell between the summer of 2008 and 2010 (two years post-harvest). Data were collected in plots established for the riparian vegetation sampling (see Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers).* We evaluated two riparian management strategies defined by state Forest Practices rules (Washington Forest Practices Board 2001): RMZs (50-ft [15.2-m]) wide bands adjacent to both sides of the perennial Type N stream) and PIPs (sensitive sites located at the uppermost points of perennial flow, surrounded by a 56-ft [17.1-m] radius management zone). Sampling in the RMZ consisted of a series of permanent strip plots established systematically along the mainstem and all tributaries long enough to accommodate a plot (i.e., ≥30.5 m). Each plot was a 15.2 × 30.4 m rectangle, with the long axis perpendicular to, and bisected by, the stream channel (15.2 m on each side), for a total area of 0.05 ha. Sampling in the PIP consisted of permanent plots established at the uppermost point of perennial flow for each tributary. Each plot had a radius of 17.07 m horizontal distance, for a total area of 0.09 ha. We did not sample plots that intersected roads or that partially overlapped with plots from adjacent stream tributaries.

We recorded whether there was sediment delivery to the stream for each fallen tree (e.g., a rootpit, exposed root-wad or associated mound). We did not attempt to calculate the volume of the root-pit or sediment delivered to the stream since it has been our experience from past studies that volumetric estimates of sediment delivery require a large number of assumptions that may not be realistically met in the forested environment in which this study was conducted.

10-4.3.1.c. Stream-delivering surface erosion

We conducted visual surveys to identify the amount of recent erosion in all study sites during the post-harvest period. The surveys covered the entire stream channel network, including the mainstem and all tributaries from the F/N break up to the channel head in both post-harvest years (2009 and 2010). We identified recently eroded areas that were 10 m^2 or larger, with a surface connection to the bankfull channel. Erosion was considered recent when the bare soil lacked established vegetation or bryophytes. Sources of erosion included uprooted trees, timber harvest activities or machinery, and slope failures, but we did not attempt to discern the specific cause of erosion. For each eroded area, we recorded the average length and the slope distance to the top of the eroded area. Stream-delivering surface erosion estimates are not a precise quantitative estimate of the amount of sediment that entered the channel, but rather an estimate of the proportion of the channel network that delivered sediment directly to the channel.

10-4.4. DATA ANALYSIS

10-4.4.1. Changes in SSE

To determine if there was a pre- to post-harvest change in SSE, and whether the response differed among experimental treatments, we used a generalized linear mixed effects model (GLMM) with site as a random effect and buffer treatment (100%, FP, and 0%), period (pre- versus post-harvest), 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 2013).

We evaluated six post hoc, pairwise comparisons to estimate the post-harvest change in SSE in each buffer treatment relative to the unharvested reference sites and differences among 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 considered the P-value, effect size, patterns of the effect size across the buffer treatments, and sample size when interpreting the results.

10-4.4.2. Changes in SSE Related to Sediment Supply Indicators

We collected sediment supply indicators with the intent of using them to help identify sources of observed post-harvest changes in SSE or differences among treatments; however, due to differences in spatial and temporal resolution and extent, there was no formal method for processing the export and supply data together. Rather, we considered the different data sets jointly in the discussion.

10-5. RESULTS

10-5.1. TURBIDITY AND SSE

Turbidity was below 3.1 NTU 95% of the time across all study sites sampled, and only three sites had median turbidities greater than zero (**Table 10-6**). Both turbidity and SSC increased with increasing discharge during storm events but then rapidly fell off, with all sites 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. The generally low SSC and strong hysteresis loops suggest sites were supply limited with respect to sediment that could travel in suspension.

Table 10-6. Median, 90th, 95th, 99th percentiles and maximum recorded turbidity and SSC by site and period.

Block	Treatment	Period	50%	90%	95%	99.5%	Maximum
			Turbidity (NTU)				
OLYM	REF	All	0.1	2.1	3.6	85	2,193
	100%	Pre	0.0	1.8	4.1	49	396
		Post	0.9	3.8	7.9	74	2,055
	FP	Pre	0.0	0.0	0.0	5.6	149
		Post	0.1	0.7	1.8	14	264
	0%	Pre	0.0	0.1	0.4	10	105
		Post	0.3	1.8	3.5	19	264
WIL1	REF	All	0.0	0.2	1.2	16	310
	100%	Pre	0.0	0.5	1.7	18	209
		Post	0.2	1.5	3.3	14	115
	FP	Pre	0.0	0.4	2.3	35	1,132
		Post	0.3	0.9	1.5	10	171
	0%	Pre	0.0	0.7	1.6	31	714
		Post	0.1	1.2	2.4	19	538
			SSC (mg L^{-1})				
OLYM	REF	All	0	0	7	305	1,200
	100%	Pre	0	0	8	120	750
		Post	0	7	15	174	2,580
	FP	Pre	0	0	0	10	373
		Post	0	0	0	43	656
	0%	Pre	0	0	0	19	265
		Post	0	0	6	55	656
WIL1	REF	All	0	0	0	46	489
	100%	Pre	0	0	0	39	193
		Post	0	0	6	34	130
	FP	Pre	0	0	0	63	1,898
		Post	0	0	0	19	293
	0%	Pre	0	0	0	74	1,234
		Post	0	0	0	50	954

Graphs of cumulative discharge and SSE show that eight storm events dominated the suspended sediment budgets of the study basins from October 2006 to September 2011 (**Figures 10-2** and **10-3**). Discharge accumulated steadily but cumulative SSE was dominated by periods of high sediment export during discrete storm events in late fall and early winter. At two of the sites, OLYM-0% and WIL1-0%, a relatively high proportion of the suspended sediment load was exported during the 6 November 2006 storm event. Two sites, OLYM-REF and WIL1-REF, exported a relatively high proportion of their suspended sediment load in the 3 December 2007 storm, which was preceded by high winds and was a major storm event for the region, causing significant mass-wasting across the region, though not in our study sites (Turner *et al*. 2010; Stewart *et al.* 2013). Several of the sites (OLYM-REF, OLYM-FP, WIL1-REF, WIL1-100%, WIL1-FP) exported relatively large proportions of their suspended sediment budgets during the 9 November 2008 and 7 January 2009 storms. These two storm events corresponded with the pre-harvest period in OLYM-100%, the period of harvest in WIL1-100% and WIL1-FP, and post-harvest period in OLYM-FP. Three of the basins responded to post-harvest storms (20 November 2009; 14 January 2010; 12 December 2010) including the WIL1-REF, OLYM-100% and OLYM-FP.

Because storm events and SSE were stochastic, the pre-planned comparisons by period (i.e., pre- vs. post-harvest) revealed no clear patterns in SSE following harvest (**Table 10-7**). At some buffer treatment sites (e.g., OLYM-100%, OLYM-0%, WIL1-0%), SSE was lower than reference SSE in the pre-harvest period but became greater than reference SSE in the post-harvest period. In other cases, the treatment/reference relationship remained the same in the post-harvest period (e.g., OLYM-FP) or exhibited inconsistent change. In addition, the differences in SSE between treatment and reference were generally small relative to the overall variability in SSE through time. Over the period of study, annual SSE ranged from 2.9 to 108 tons/ km^2 while the difference between reference and treatment (TRT-REF) SSE ranged from −5.2 to 71.3 ($n = 8$) in the pre-harvest period and −77 to 96.3 ($n = 13$) in the post-harvest period. Statistical comparisons of SSE by period and treatment yielded no consistent pattern that could be attributed to buffer treatments (**Table 10-8).** In addition, there was no significant harvest effect ($P = 0.13$) when we grouped all buffer treatments together in the analysis. Relative to the reference sites, SSE increased after harvest in the 100% ($P = 0.05$) and 0% ($P = 0.24$) buffer treatments (**Table 10-8** comparisons a) and c), but decreased in the FP ($P = 0.58$) buffer treatment (**Table 10-8** comparison b). Although the 100% treatment is close, none of the comparisons were statistically significant at $\alpha = 0.05$ and the trend in estimated effect size did not vary with treatment severity as expected (e.g., 0% > FP > 100%). The relatively large estimated effect size (+56.6 tons/km^2/yr; $P = 0.051$) in the 100% treatment was driven almost entirely by timing differences between export in the OLYM-REF and OLYM-100%. The largest magnitude event in OLYM-REF occurred on 9 November 2008, which corresponded with the pre-harvest period in the OLYM-100%. The three largest events in OLYM-100% occurred on 20 November 2009, 14 January 2009, and 12 December 2010, all of which happened to correspond with the two year post-harvest period.

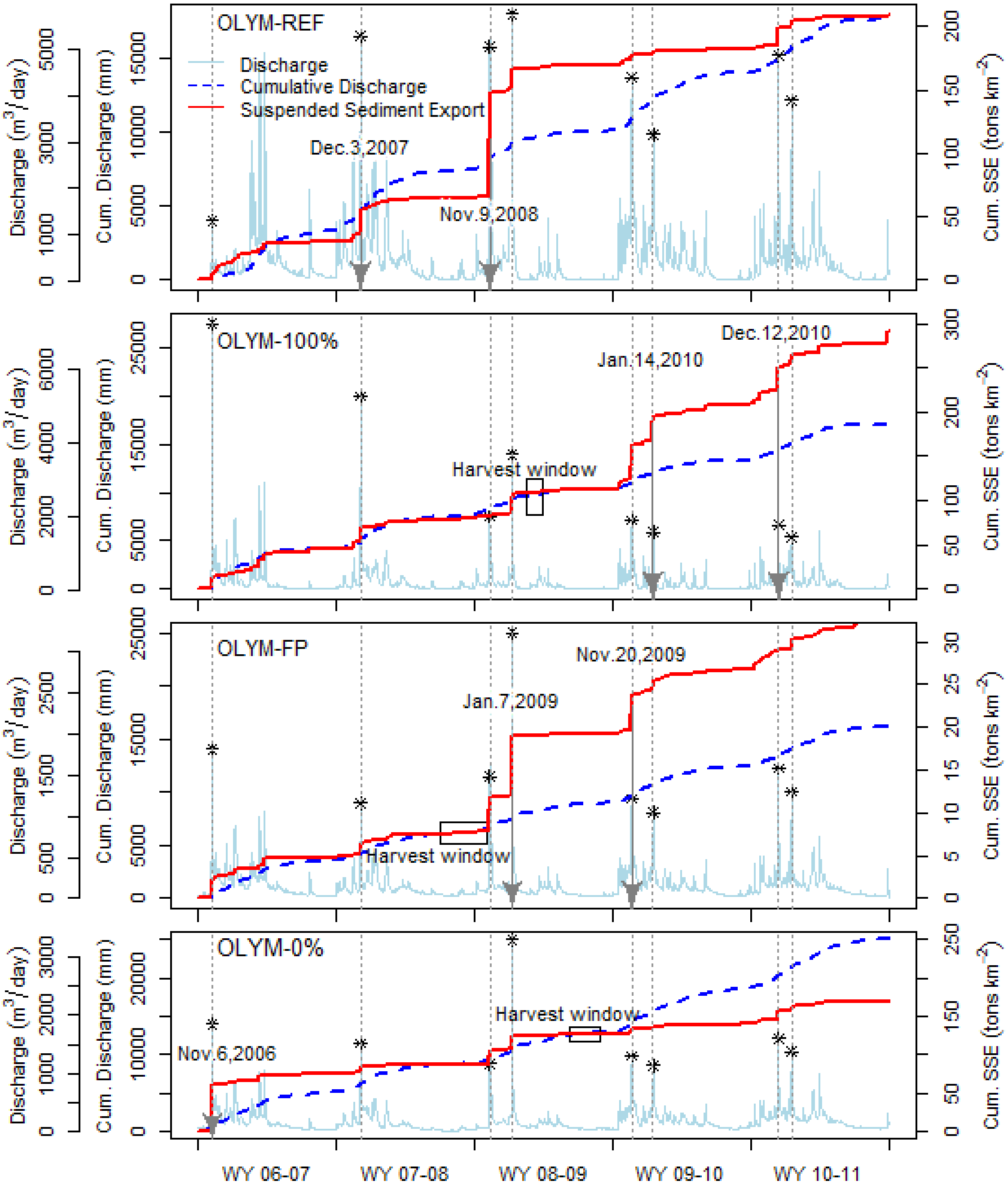

Figure 10-2. Discharge, cumulative discharge and cumulative SSE in the four sites in the Olympic block. Peak daily discharge denoted by asterisk. Vertical dashed lines correspond with storms that resulted in major sediment export at one or more sites, and arrows are associated with dates for an event.

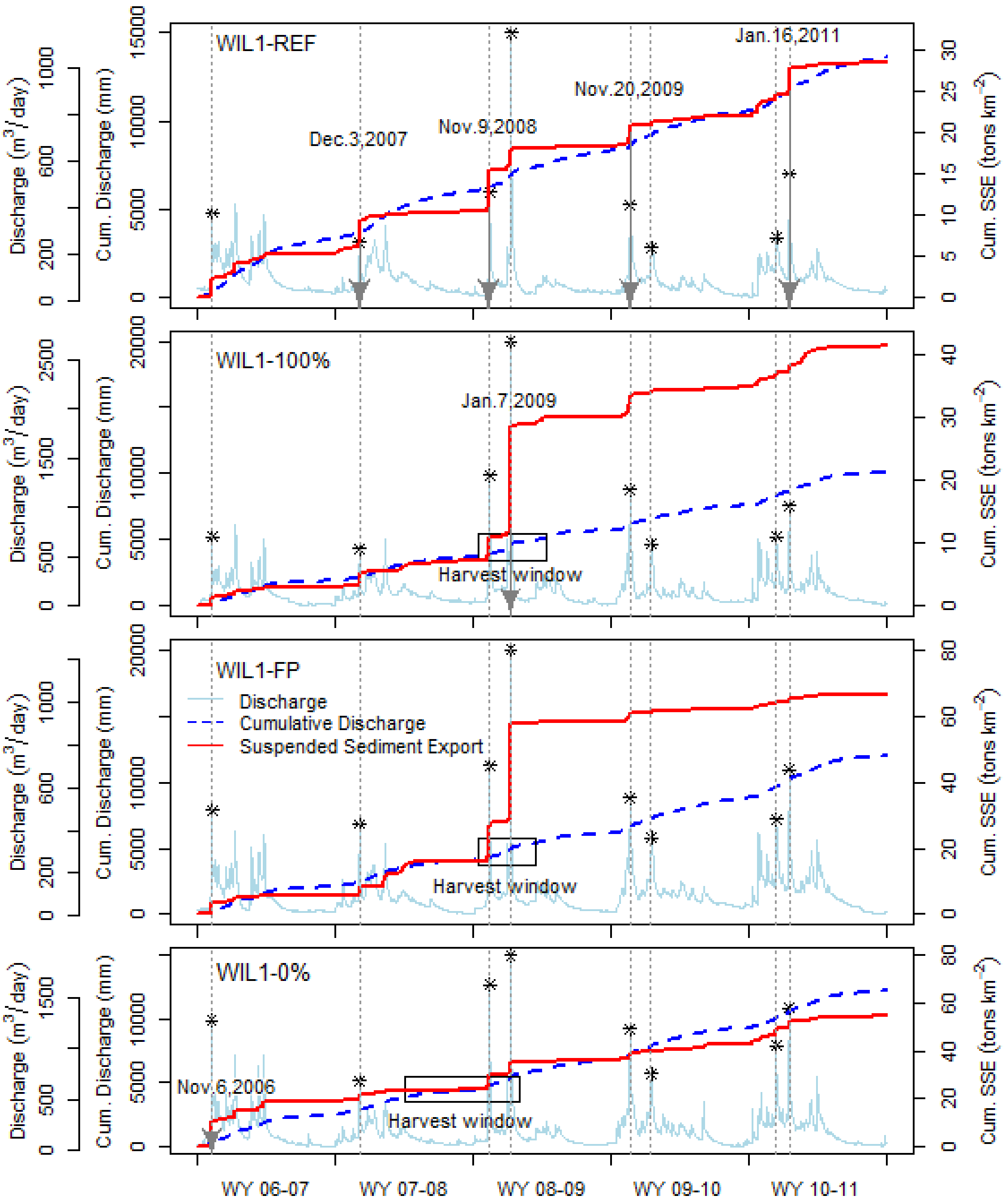

Figure 10-3. Discharge, cumulative discharge and cumulative SSE in the four sites in the Willapa 1 block. Peak daily discharge denoted by asterisk. Vertical dashed lines correspond with storms that resulted in major sediment export at one or more sites, and arrows are associated with dates for an event.

Table 10-7. SSE (tons/km^2) by period, buffer treatment, and block. Bold values show the site (treatment vs. reference) that had more export in each period. Post-harvest years are shaded. Reference basin export varied by period and export was calculated using the start and end dates of harvest activity, which varied for each buffer treatment site.

Block	Treatment	Period	SSE (tons/km^2)	
			Treatment	Reference
OLYM	100%	Pre 1	33.3	**104.6**
		Post 1	**90.9**	13.3
		Post 2	**74.4**	25.8
	FP	Pre 1	2.9	**35.9**
		Post 1	11.7	**108.0**
		Post 2	8.3	**10.7**
	0%	Pre 2	12.2	**35.8**
		Pre 1	39.6	**105.3**
		Post 1	**13.7**	12.3
		Post 2	**28.7**	25.7
WIL1	100%	Pre 2	3.0	**5.5**
		Pre 1	4.1	**4.9**
		Post 1	**5.2**	3.4
		Post 2	6.3	**6.6**
	FP	Pre 2	**5.8**	5.5
		Pre 1	**10.1**	4.9
		Post 1	**3.9**	3.4
		Post 2	4.3	**6.6**
	0%	Pre 1	4.6	**5.0**
		Post 1	**4.8**	3.3
		Post 2	**12.6**	6.6

Table 10-8. Results of the post-hoc pairwise comparisons of SSE from the GLMM. Comparisons a), b), and c) are estimates of the post-harvest change in SSE relative to the reference treatment (e.g., Pre [TRT − REF] vs. Post [TRT − REF]); comparisons d), e), and f) are differences among buffer treatments. Positive estimates indicate a post-harvest increase in SSE (a-c) or higher SSE in the second buffer treatment listed.

Comparison	Change in SSE (tons/km^2)		95% CI	
	Estimate	P-value	Lower	Upper
a) Pre vs. Post-100%	56.6	0.051	−0.2	113.4
b) Pre vs. Post-FP	−15.0	0.577	−71.8	41.8
c) Pre vs. Post-0%	32.5	0.238	−24.3	89.3
d) 100% vs. FP	−71.6	0.076	−151.9	8.7
e) 100% vs. 0%	−24.1	0.527	−104.4	56.2
f) FP vs. 0%	47.5	0.223	−32.8	127.8

10-5.2. SEDIMENT SUPPLY INDICATORS

10-5.2.1. Road Surface Erosion

This section presents results of modeled sediment delivery from road surface erosion. Road surface erosion was not directly related to the riparian treatments, but was considered a potential confounding factor in determining the effect of the riparian prescriptions on SSE. Twelve of 17 study sites contained road segments that had a sediment delivery pathway to the stream based on WARSEM definitions, including seven of the eight sites included in our SSE monitoring. The OLYM-FP, WIL2-REF2 and all sites in the South Cascade block lacked road segments with delivery to the streams. Modeled road types included primary, secondary, spur and abandoned roads. A summary of road segment values from the two years post-harvest are presented in **Appendix Table 10-A-2**. Numbers of road segments and total road length by site and year are presented in **Appendix Table 10-A-3**. Construction year ranged from an estimated 1945 to 2008, with the exact year of construction being reported only for newer roads.

Road traffic use ranged from none to moderate, with a shift to greater occasional use and less light use in the post-harvest year (**Figure 10-4**). Over the course of the study period BMPs implemented ranged from none to major, and included activities such as grading, adding rock surfaces, widening, cleaning ditches and culverts, culvert replacement, installation of cross drains and rock caps, and road construction and abandonment

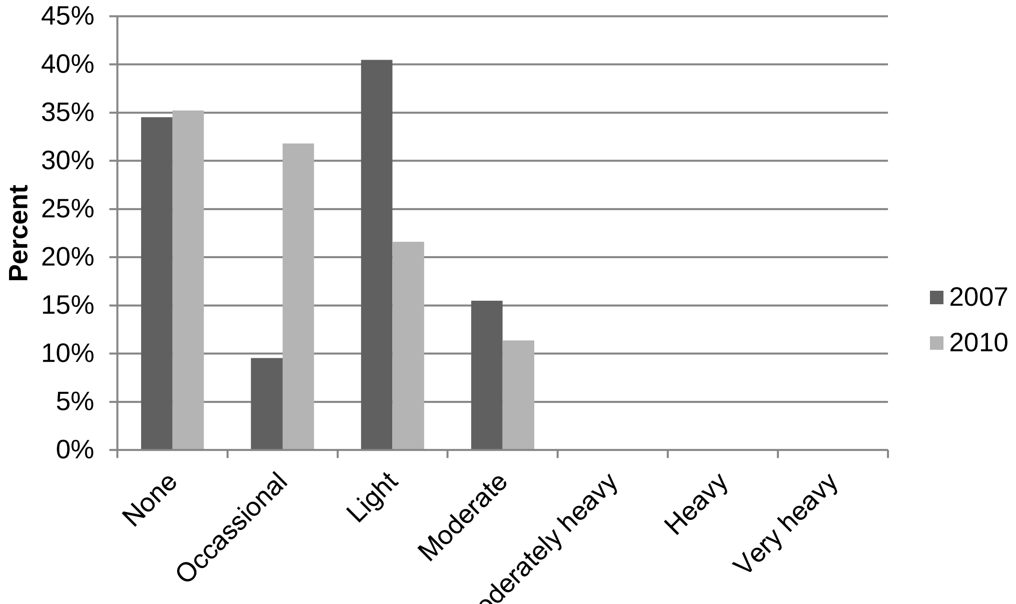

Figure 10-4. Proportion of road traffic category reported by landowners in the pre- (2007) and post-harvest (2010) periods.

Modeled sediment delivery input by roads for each study site and year ranged from 0.0 to 17.5 tons/year (**Table 10-9)** and, absent changes in traffic, road construction, or BMPs, the modeled export remained constant. Construction, reconstruction, and decommissioning of roads was not extensive, and reported traffic levels (e.g., heavy vs. light) typically remained the same or were lower in the post-harvest period, so road surface erosion estimates did not vary much from year to year, with three exceptions (**Figure 10-5**). In the WIL3-100%, previously decommissioned roads were reconstructed and used during harvest in 2008, and then decommissioned again within the same year leading to a large spike in sediment input in 2008. In the WIL2-0%, reduced road traffic led to reduced sediment inputs. The OLYM-0%, which was the only one monitored for SSE, showed increased traffic and modeled sediment inputs during the harvest period, declining again after harvest was completed.

We designed the study to evaluate change by buffer treatment between the pre- and post-harvest periods. Within that framework, we estimated an average post-harvest increase of 0.39 tons/year (SE = 0.37) for the reference, 0.88 tons/year (SE = 0.51) for the 100%, and 0.30 tons/year for the FP treatment (note that this is based off of data for one study site). Conversely, we estimated an average decrease of 1.72 tons/year in the 0% treatment (**Figure 10-6**).

Table 10-9. Annual net tonnage of sediment input by roads for sites with roads delivering to Type Np waters, estimated using the WARSEM, 2006–2010. Highlighted cells indicate calendar years within which buffer treatments were applied.

Block	Treatment	Year				
		2006	2007	2008	2009	2010
OLYM	REF	6.1	6.1	8.3	8.3	8.3
	100%	1.2	1.2	3.3	2.5	2.5
	0%	2.8	3.8	3.8	3.8	2.8
WIL1	REF	2.1	2.1	2.4	2.4	2.4
	100%	15.1	15.1	17.5	17.5	17.5
	FP	6.6	6.6	6.9	6.9	6.9
	0%	0.9	0.9	0.5	0.5	0.5
WIL2	REF1	5.7	5.7	5.6	5.6	5.6
	100%	0.3	0.3	0.8	0.8	0.8
	0%	6.2	6.2	5.6	2.4	1.8
WIL3	REF	0.5	0.5	0.4	0.4	0.4
	100%	0.4	0.4	14.6	0.6	0.6

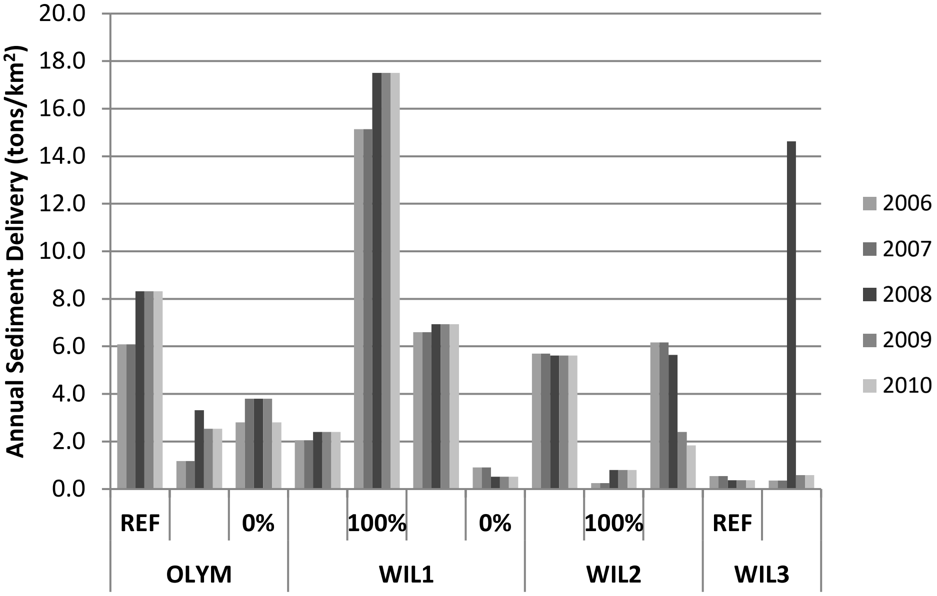

Figure 10-5. Estimated annual road sediment input (tons/km^2) for sites with roads delivering to streams, by block and treatment.[2]

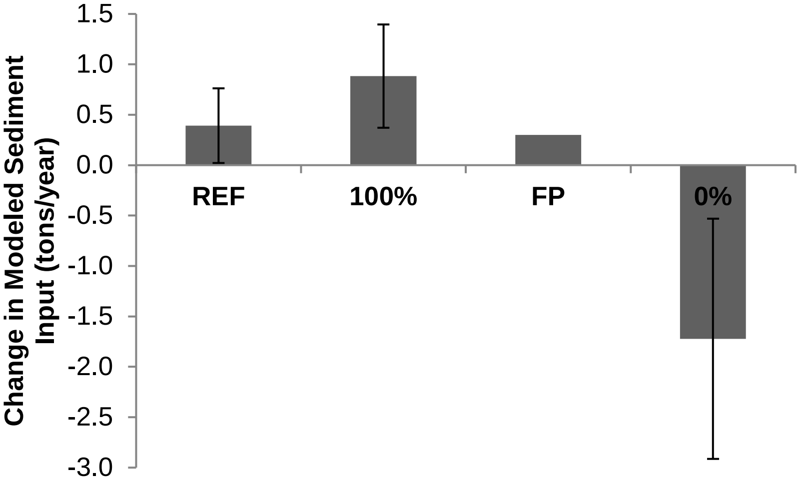

Figure 10-6. The average pre- to post-harvest change (post − pre) in modeled sediment delivery (tons/year) by roads, averaged by treatment. Error bars are the SE. Note that the estimate for the FP treatment is based on data from one study site.

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[2] WARSEM estimates the average annual amount of road surface erosion delivered to streams from each road segment. We ran the model for each year and segment and summed the segments to estimate the total annual tonnage for a site and year. We then adjusted these values by basin area (km^2). According to the WARSEM manual it is appropriate to examine relative differences in erosion estimates to compare watersheds or road segments, but the sediment values in tons/year should be regarded as estimates, not absolute values.

10-5.2.2. Uprooted Trees

Uprooted trees in the RMZs adjacent to the channel (within 13.2 m) are potential sources of sediment, and the magnitude of those sources could be expected to differ by riparian treatment. We recorded a large number of uprooted trees and noted extensive variation in soil disturbance associated with uprooted trees in both the pre- and post-harvest periods. During the pre-harvest period, the highest rates of tree uprooting (e.g., windthrow) occurred in the WIL1 and WIL2 blocks, with much lower rates in the other blocks. This appeared to be associated with regional differences in the frequency and magnitude of wind storms, as was the case for tree mortality (see Chapter 5 – *Stand Structure and Tree Mortality Rates in Riparian Buffers*). Although there was a large amount of windthrow, in most cases only a small percentage of the uprooted trees exhibited visual evidence of sediment delivery to the stream (**Table 10-10**). In the post-harvest period, the number of total uprooted trees was highly variable, with the highest rates in the 100% and FP treatments (**Table 10-11**). However, the number of uprooted trees delivering sediment was relatively similar for all treatments.

Table 10-10. Number/100m of stream length/yr of total uprooted trees with soil disturbance and the subset of uprooted trees with sediment delivery within RMZs pre- and post-harvest.

Block	Treatment	Total Uprooted Trees		Trees with Sediment Delivery		Proportion with Sediment Delivery	
		Pre-	Post-	Pre-	Post-	Pre-	Post-
OLYM	REF	1.5	6.0	0.4	1.0	30.0%	17.3%
	100%	3.9	1.6	1.0	0.2	25.0%	10.0%
	FP	4.1	5.2	1.2	0.3	30.8%	6.1%
	0%	2.0	0.0	0.7	0.0	33.3%	-
WIL1	REF	51.1	3.7	13.1	0.7	25.7%	18.8%
	100%	34.4	9.2	12.1	0.7	35.2%	7.1%
	FP	4.7	19.7	0.0	1.4	0.0%	7.1%
	0%	12.2	0.2	3.4	0.0	28.2%	0.0%
WIL2	REF	16.0	0.4	6.6	0.2	41.0%	50.0%
	REF	18.8	2.0	5.7	1.1	30.2%	55.6%
	100%	14.5	14.4	5.0	1.8	34.4%	12.6%
	0%	13.5	0.0	6.9	0.0	51.2%	-
WIL3	REF	1.1	0.4	0.2	0.1	16.7%	20.0%
	100%	2.4	1.8	0.3	0.3	11.1%	14.3%
CASC	REF	0.5	0.5	0.0	0.0	0.0%	0.0%
	FP	0.0	0.0	0.0	0.0	-	-
	0%	0.0	0.0	0.0	0.0	-	-

In the OLYM and WIL1 blocks, where suspended sediment was monitored, there was extensive variation in both pre- and post-harvest rates with site average values ranging from 0 to 13.1 uprooted trees per 100 m in these blocks. Sites with the greatest amount of windthrow, and largest number of uprooted trees with sediment delivery, were the WIL1-REF and WIL1-100%, measured in the summer 2007 pre-harvest period (**Figure 10-7**). In the post-harvest period, the number of uprooted trees with sediment delivery decreased at six of eight sites. The sites with increases were the OLYM-REF (0.4 to 1.0/100 m) and WIL1-FP (0.0 to 1.4/100 m) sites.

Table 10-11. Average rate (number/ha/yr) of uprooted trees and the subset of uprooted trees with sediment delivery in RMZ and PIP plots by treatment in the post-harvest period.

Treatment	RMZ			PIP		
	Pits/ha/yr	With Sediment Delivery/ ha/yr	Proportion with Sediment Delivery	Pits/ ha/yr	With Sediment Delivery/ ha/yr	Proportion with Sediment Delivery
REF	1.2	0.3	24.0%	0.4	0.2	42.9%
100%	3.6	0.4	10.7%	5.8	1.5	25.6%
FP (buffered)[1]	6.6	0.4	6.9%	4.1	0.4	9.8%

[1] Post-harvest FP sites refer only to the buffered portion of the stream network, since harvest removed the trees in the unbuffered portion.

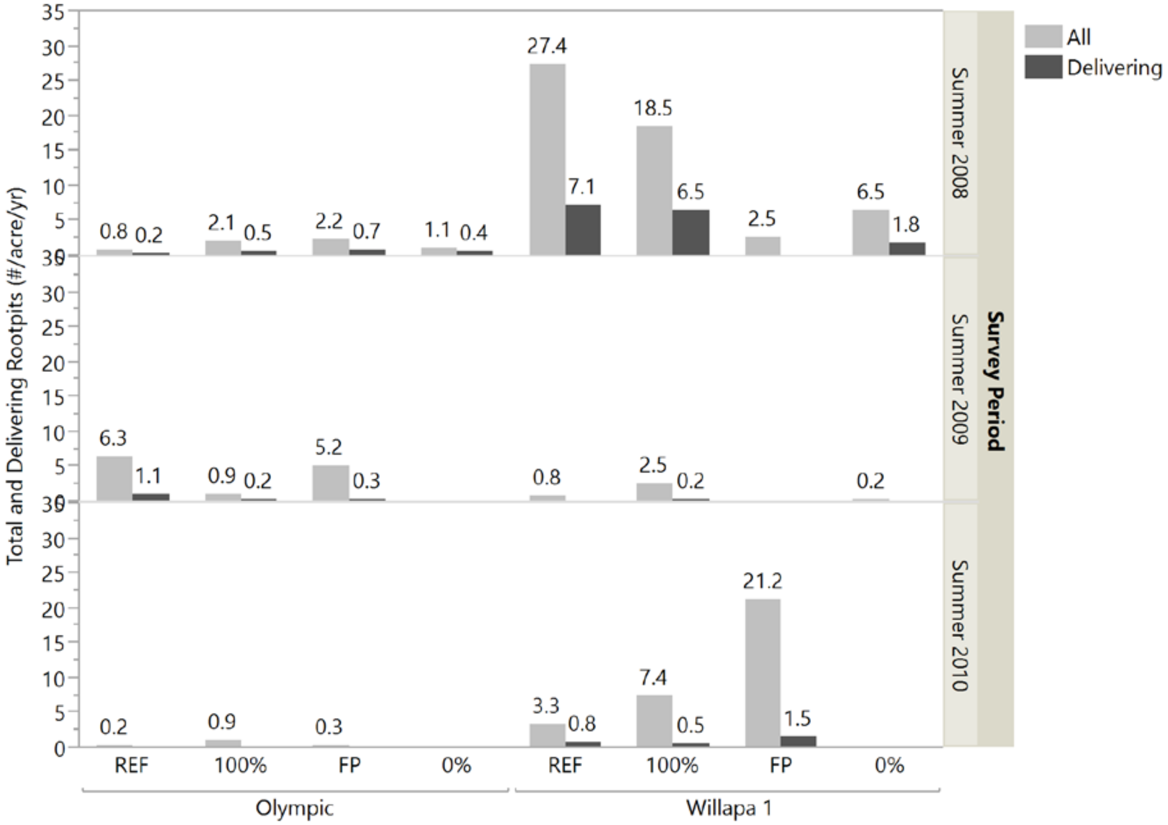

Figure 10-7. Annual rate of rootpit generation for all uprooted trees and those with visual evidence of sediment delivery by survey period.

10-5.2.3. Stream-delivering Surface Erosion

Stream-delivering surface erosion was only estimated in the post-harvest period, so we can only examine differences in post-harvest surface erosion among buffer treatments and the reference. We documented post-harvest surface erosion at 11 of the 17 sites, and the total erosion area exceeded 110 m^2 at five sites: two references, two 100% treatments, and one 0% treatment site (**Table 10-12**). At these five sites, post-harvest surface erosion was evident adjacent to only 1.5 to 4.6% (average = 2.2%) of the total stream channel length (including both mainstem and tributaries). At the remaining study sites where stream-delivering erosion events occurred, the total eroded area was 60 m^2 or less and occurred adjacent to 0.3% to 0.8% (average = 0.6%) of the stream channel length. There were no statistically significant differences in stream-delivering surface erosion among treatments ($\alpha = 0.05$), and on average, reference and buffer treatments visually exhibited a similar amount of exposed bank.

In the OLYM and WIL1 block sites where suspended sediment was monitored, we documented post-harvest stream-delivering surface erosion at six of the eight sites (**Table 10-12**). Total surface area exceeded 100 m^2 at three sites, including the OLYM-REF, OLYM-100% and the

WIL1-0%. The proportion of the channel length affected ranged from 0% to 1.7%, with highest values (>1%) at the same three sites.

Table 10-12. The number and size of stream-delivering surface erosion events observed in the post-harvest period. Number is the number of individual erosion events; Total Area (m^2) is the sum of the areas for all erosion events; Min and Max Area (m^2) are the areas for the smallest and largest eroded areas; Avg is the average area of all erosion events; Erosion length (m) is the extent of the channel length above which stream-delivering erosion occurred on either side of the channel, regardless of the location or extent of the delivery point; Proportion of channel is the proportion of the channel length (including mainstem and tributaries) that exhibited stream-delivering surface erosion. Shaded sites are those in which SSE monitoring was also conducted.

Block	Treatment	Number	Area (m^2)				Length (m)	
			Total	Min	Max	Avg	Erosion	Proportion of Channel
OLYM	REF	8	249	10	57	31	53	1.7
	100%	6	128	14	44	30	33	1.6
	FP	0	-	-	-	-	-	-
	0%	1	32	32	32	32	3	0.4
WIL1	REF	0	-	-	-	-	-	-
	100%	1	60	60	60	60	7	0.6
	FP	1	14	14	14	14	4	0.8
	0%	1	169	169	169	169	26	1.6
WIL2	REF1	5	173	19	52	35	40	4.6
	REF2	0	-	-	-	-	-	-
	100%	4	111	14	44	28	25	1.5
	0%	1	49	49	49	49	6	0.6
WIL3	REF	2	31	15	16	16	14	0.5
	100%	0	-	-	-	-	-	-
CASC	REF	1	21	21	21	21	6	0.3
	FP	0	-	-	-	-	-	-
	0%	0	-	-	-	-	-	-

10-6. DISCUSSION

Our analysis revealed no significant effects of harvest and no clear pattern regarding the relative effectiveness of buffer treatments at mitigating the effects of clearcut harvests on SSE. This is partially a function of the timing of storm events leading to SSE spikes in the OLYM-REF, which occurred during the pre-harvest period for the OLYM-100% and the post-harvest periods for the OLYM-FP and OLYM-0%. Also, the study design was restricted to headwater basins underlain by relatively competent lithologies so we expected SSC and annual estimates of SSE to be low. Indeed, turbidity and SSC were very low during most of the study, with a high proportion of sediment export in both the pre- and post-harvest periods occurring during discrete storm events in late fall or early winter. All sites exhibited strong SSC hysteresis with greater SSC on the rising limb of the hydrograph than during periods of constant or decreasing discharge. SSC hysteresis loops were observed for all sites in both pre- and post-harvest periods, suggesting that all sites were supply limited with respect to sediment that could be carried in suspension, and remained that way over the entire study period.

The total annual SSE (2 to 108 tons/km^2/yr) was within the range of reported suspended yields for unmanaged small catchments, though greater than expected for sites with competent lithologies (Table 2; Gomi *et al.* 2005). In the Olympic block, the magnitude of SSE was relatively large, with all the sites exceeding 20 tons/km^2/yr during at least one year of the study, while the Willapa 1 sites averaged only 6 tons/km^2/yr. Most of the suspended sediment was exported during eight discrete storm events and the magnitude of export was not synchronized in time across the study sites. The lack of synchronicity is likely due to distance between some of the sites, limited sediment supply, and the episodic nature of the inputs. The site with the greatest annual export was the OLYM-REF, which exported a large proportion of its suspended sediment budget during a single storm event in November 2008. The WIL1-REF exhibited only small increases in SSE across a number of events. In the treatment sites, two (OLYM-0% and WIL-0%) had SSE spikes during the November 2006 storm and didn’t spike again; two (WIL1-100% and WIL1-FP) saw large SSE spikes during the harvest period; and two (OLYM-100% and OLYM-FP) saw the largest SSE spikes in the post-harvest period.

The December 2007 windstorm caused extensive windthrow in the WIL1-REF, WIL1-100%, and WIL1-FP. Windthrow associated with sediment delivery was greatest in the WIL1-REF and WIL1-100%, yet windthrow did not play a substantive role in SSE. The SSE spikes associated with this event are unremarkable in comparison with spikes observed in fall 2006 in the same sites or with pre-harvest spikes observed in the other Willapa 1 sites. Another windthrow event occurred in the WIL1-FP during the winter of 2009/2010. When we examined SSE for these events, we found that windthrow is unlikely to account for more than 5 tons/km^2 of export in even the largest windthrow event.

The sediment supply data collected as a part of this study generally lacked the precision and temporal resolution needed to explain patterns of SSE. In the sites where SSE was monitored, modeled road surface sediment was greatest in the WIL1-100%, followed by the WIL1-FP. Predicted road sediment inputs in those two sites exceeded 40 tons/km^2/year, although measured export didn’t exceed 70 tons/km^2 over the entire five-year study period and total SSE in those two treatment sites were among the lowest measured. While WARSEM modeling is not intended to provide absolute estimates but rather to indicate relative differences, if the WARSEM results are reliable, some of the road sediment in the WIL1-100% and WIL1-FP was likely stored in the channel rather than being exported. Unfortunately, the WARSEM approach provided no other insights and is limited by its ability to predict only large changes in road sediment delivery without new road construction, major changes in road traffic, or BMP implementation. However, we had no *a priori* expectation that there would be a difference among buffer treatments since the potential need for road construction and BMPs, and increased traffic during harvest, would not necessarily be expected to vary depending on the amount or configuration of trees left in the RMZ.

The post-harvest stream-delivering surface erosion surveys failed to provide any significant insights, other than a lack of mass wasting or landslide events. The OLYM-REF exhibited both the greatest area and length of channel affected by surface erosion. In our stream-delivering surface erosion surveys in 2009 we noted that an area located between the PIP and channel head of one tributary was recently eroded. This erosion event may have contributed to the large spike in SSE observed in the OLYM-REF in November 2008. The OLYM-100% and WIL1-0% were next highest in terms of surface erosion area and length of the channel affected. In the OLYM-100% we observed newly eroded areas in August 2010 that were not present in April 2009 which may at least partially explain the spikes in SSE observed at this site in November 2009 and January 2010. However, we observed limited SSE in the WIL1-0% over the course of the entire study.

10-7. CONCLUSIONS

Our study was designed to measure SSE as function of timber harvest and buffer treatment, and to relate SSE to potential sources of sediment input, including from rootpits of windthrown trees, roads, and stream-delivering surface erosion. The focus on relatively competent lithologies was a study design choice made to ensure that sites had adequate amphibian populations for an evaluation of amphibian response to buffer treatments. Sediment supply indicators were evaluated in a cursory way, and we did not evaluate bed erosion. From the data collected, we can draw the following conclusions:

- SSE was within the ranges reported in the literature for unmanaged basins, both pre- and post-harvest.

- All sites appeared to be supply limited both pre- and post-harvest.

- Although SSE increased during or post-harvest at several sites, when evaluated relative to the unharvested reference sites, no statisticaly significant buffer treatment effect or harvest impact was detected.

- Our ability to detect a buffer treatment effect may have been hampered by the low replication (n = 2) of each treatment, the limited number of sediment generating events, and inconsitency in sediment export across sites for a given storm event owing to the limited sediment supply and stochastic nature of sediment inputs. For these sites and over these time-scales, any harvest-related change in SSE appears to have been less than the natural variability.

- Windthrow was not the major driver in SSE even in those sites that experienced severe windthrow.

- The WARSEM model does not have the temporal resolution needed to evaluate relationships between SSE and road sediment inputs in a short-term study with limited replication.

- In these sites and over the time scales observed, stream-delivering surface erosion was largely restricted to small areas of the streambank and unlikely to measureably affect SSE.

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APPENDIX 10-A. TURBIDITY OBSERVATION AND ROAD SEGMENT DATA

Appendix Table 10-A-1. Total number of turbidity observations and the number where turbidity was set equal to zero for discharge equal to or less than and greater than median.

Block	Treatment	Years	Observations	Observations with Turbidity Set = 0		
				Total	Q ≤ Median Flow	Q > Median
OLYM	REF	All	210,218	2,900	24	2,876
	100%	All	207,652	19,745	15,772	3,973
		Pre	102,536	12,500	10,781	1,719
		Post	105,116	7,245	4,991	2,254
	FP	All	197,468	6,715	1,443	5,272
		Pre	92,506	5,913	1,442	4,471
		Post	104,962	802	1	801
	0%	All	210,207	1,189	896	293
		Pre	105,088	1,146	853	293
		Post	105,119	43	43	0
WIL1	REF	All	210,233	2,436	1,294	1,142
	100%	All	207,468	614	478	136
		Pre	102,496	265	129	136
		Post	104,972	349	349	0
	FP	All	210,200	2,791	1,616	1,175
		Pre	104,976	2,790	1,616	1,174
		Post	105,225	1	0	1
	0%	All	185,842	1,743	47	1,696
		Pre	80,727	1,071	1	1,070
		Post	105,115	672	46	626

Appendix Table 10-A-2. Summary of road segment values from road surface erosion field sampling in 2007 and 2010.

Attribute	Value	2007	2010
Number of Segments		84	88
Segment Length	Feet (meters)	10 (3) to 985 (300) ($\bar{x}$ = 256 [78])	31 (9) to 680 (207) ($\bar{x}$ = 203 [62])
Average Road Slope	Flat or gently sloped (<5%)	49%	50%
	Moderate (5–10%)	46%	44%
	Steep (>10%)	5%	6%
Road Configuration	Insloped (or outsloped w/ wheel tracks)	21%	35%
	Outsloped	2%	18%
	Crowned	76%	47%
Surfacing	Asphalt	4%	2%
	Gravel	15%	22%
	Native	13%	13%
	Pitrun	68%	64%
Average Tread Width	Feet [meters)	10 (3) to 31 (9) ($\bar{x}$ = 16 [5])	9 (3) to 32 (10) ($\bar{x}$ = 17 [5])
Cutslope Cover Density	90–100%	61%	31%
	70–90%	18%	16%
	50–70%	2%	9%
	30–50%	7%	16%
	10–30%	1%	8%
	0–10%	11%	20%
Cutslope Average Height	None (0 ft)	7%	11%
	2.5 ft	31%	24%
	5 ft	19%	19%
	10 ft	30%	36%
	25 ft	13%	9%
Ditch Width	Feet (meters)	0 (0) to 13 (4) ($\bar{x}$ = 5 [2])	0 (0) to 10 (3) ($\bar{x}$ = 4 [1])
Ditch Delivery	None	21%	10%
	Direct	51%	56%
	≤100 ft	17%	20%
	≤200 ft	11%	14%
	Connected	0%	0%
Ditch Condition	Rocked or vegetated	14%	0%
	Stable (no eroding)	80%	98%
	Eroding (eroding/incising)	6%	2%

Appendix Table 10-A-3. Number of road segments and total length of segments for roads that deliver to Type Np study sites, by treatment.

Block	Treatment	Year	Segments	Total Road Length (ft/[m])
OLYM	REF	2007	14	2,965 (903)
		2010	14	3,040 (926)
	100%	2007	16	4,267 (1,301)
		2010	18	2,712 (827)
	0%	2007	13	3,171 (967)
		2010	11	1,974 (602)
WIL1	REF	2007	2	376 (115)
		2010	2	315 (96)
	100%	2007	5	1,712 (522)
		2010	5	1,610 (491)
	FP	2007	2	965 (294)
		2010	3	1,075 (326)
	0%	2007	4	723 (220)
		2010	6	864 (263)
WIL2	100%	2007	4	747 (228)
		2010	4	722 (220)
	0%	2007	4	849 (259)
		2010	6	1,149 (350)
	REF1	2007	5	1,541 (470)
		2010	5	1,726 (526)
WIL3	REF	2007	10	2,075 (632)
		2010	10	2,002 (610)
	100%	2007	5	2,001 (610)
		2010	4	512 (156)

CHAPTER 11 - STREAM CHANNEL CHARACTERISTICS

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11-1. ABSTRACT

We compared the response of headwater stream channels to clearcut timber harvest with three alternative riparian buffer treatments and reference sites that were not harvested during a five-year study period (2006–2010). Buffer treatments varied by percent of the stream length buffered and included the current state Forest Practices buffer (FP treatment), a longer (100% treatment) and no buffer (0% treatment). We included measures of the responses of macroinvertebrates and stream-associated amphibians to alternative riparian buffer treatments, and therefore many of the channel characteristics we measured were intended to help us describe harvest effects on macroinvertebrate and amphibian habitat. We hypothesized that changes to stream channel characteristics in these small headwater streams during the first two years post-harvest would be largely influenced by the input of wood into the stream channel as a result of timber harvest activities. We evaluated the response of headwater streams to timber harvest using common stream channel metrics (e.g., stream wetted width, stream substrate, and channel unit composition and characteristics). Pool length increased between the pre- and post-harvest periods by 9% in the three riparian buffer treatments, an increase that differed from the change we observed for the reference. The pre- to post-harvest change in stream bankfull width in the 0% treatment was 0.4 to 0.5 m less than the change observed in the remaining riparian buffer treatments and the reference, and we observed a similar decrease of 0.3 m in stream wetted width in the 0% treatment. The proportion of channel rise attributed to steps was 16% to 17% less in the 0% treatment relative to the changes observed in the remaining treatments. Finally, though not statistically significant, we noted a post-harvest pattern in the proportion of fines and sand substrates, with an increase of 4% in the 100% treatment, 9% in the FP treatment, and 16% in the 0% treatment, compared to no pre- to post-harvest difference in the reference. Changes to the quantity and quality of certain instream characteristics may have consequences for instream biota, including macroinvertebrates and amphibians.

11-2. INTRODUCTION

Streams can be characterized by the general form or structure of the channel (i.e., morphology), the size and distribution of substrates, and the types of channel unit (e.g., pools and riffles). The effect of forestry practices on the physical characteristics of headwater streams has been the focus of numerous studies in the Pacific Northwest. Though research has been predominantly retrospective (e.g., Corn and Bury 1989; Dupuis and Steventon 1999), a few experimental studies have evaluated headwater stream responses to timber harvest. O'Connell and colleagues (2000) studied the impacts of alternative riparian buffers on riparian vegetation, stream characteristics and stream-associated amphibians in Washington before and after harvest. Cissel and colleagues (2006) evaluated the impact of alternative riparian buffers on channel characteristics, stream-associated amphibians and fish in western Oregon. However, neither of these studies included harvest to the stream edge (i.e., timber removal in the Riparian Management Zone, or RMZ). Jackson and colleagues (2001) conducted an experimental study evaluating the impacts of clearcut timber harvest on geomorphic and biotic responses in headwater streams located in the Coast Range of Washington. Their study examined the effectiveness of riparian buffers of differing lengths, including a treatment with complete tree removal in the RMZ; unfortunately, the lack of a riparian buffer in this treatment resulted in nearly complete coverage of the stream channel with wood from logging, making it impossible to include this treatment in many of their statistical comparisons. Conversely, while the proportion of the stream length covered by wood increased in our riparian buffer treatments following harvest, we were able to include the 0% treatment in all of our statistical comparisons; a maximum of 17% of our stream length was covered and/or buried by slash in the post-harvest period (see Chapter 6 – *Wood Recruitment and Loading*).

Many studies that include the response of stream channel characteristics to timber harvest in small headwater streams are focused on aquatic or other stream-associated taxa, including fish, amphibians and macroinvertebrates (e.g., Murphy and Hall 1981; Lisle 1986; Carlson *et al.* 1990; Bull and Carter 1996; O'Connell *et al.* 2000; Jackson *et al.* 2007, among those already mentioned). Our study included the response of both macroinvertebrates and stream-associated amphibians to timber harvest, and therefore many of the channel characteristics we measured were intended to help us refine our description of harvest effects on habitat availability and quality for these taxa.

The role of wood in shaping the geomorphology and influencing the biota of lotic systems is well documented. Wood pieces trap and store sediment and organic material (Bilby and Ward 1989; Gomi *et al.* 2002; Hassan *et al.* 2005a), increase the frequency of steps (Gomi *et al.* 2002), create pool habitat (Andrus *et al.* 1988; Bilby and Ward 1989; Beechie and Sibley 1997), stabilize streambeds and banks, and dissipate energy (Curran and Wohl 2003). Riparian timber management activities affect short-term wood loading through harvest practices as well as longer-term recruitment potential by altering riparian stand conditions (Bilby and Ward 1991; Ralph *et al.* 1994; Grizzel and Wolff 1998; Jackson and Sturm 2002). Many studies have identified the role of wood in influencing channel response to timber management activities. For example, changes in pool characteristics including length (Lisle 1986), frequency (Montgomery *et al.* 1995) and proportion of total habitat (Jackson *et al.* 2001) have been associated with an increase in instream wood. Likewise, many researchers have observed an increase in the amount of fines, sand and gravel in streams adjacent to clearcuts (Corn and Bury 1989; Dupuis and

Steventon 1999; Jackson *et al.* 2001), which in some cases has been linked to the added storage capacity provided by increased wood loading in streams after timber harvest. While most studies have focused on large wood (LW), small wood (SW) is frequently abundant in smaller channels, where stream power is typically too low to transport wood downstream (Bilby and Ward 1989; Maxa 2009), and may provide functional roles (e.g., sediment storage) and influence channel morphology (Gomi *et al.* 2001; Maxa 2009). Our study is among the few that can address the role of small wood in biotic responses.

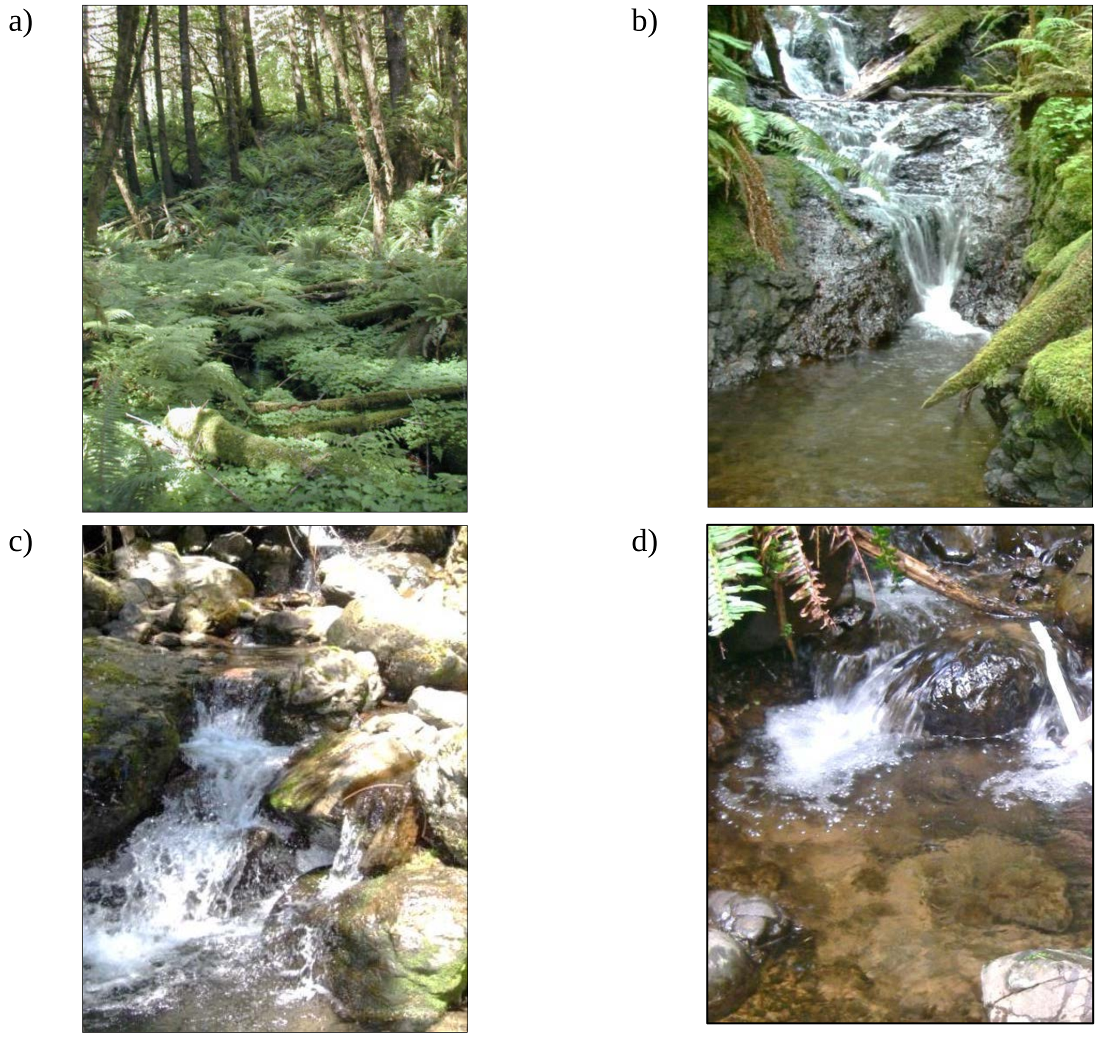
We examined headwater stream hydrology, channel characteristics and channel units among riparian buffer treatments and reference sites. In particular, we examined stream response to clearcut harvest with alternative riparian buffer treatments, including a treatment with no riparian buffer (e.g., clearcut to the stream edge). Our study design, which includes a multi-year data collection period both before and after timber harvest implementation, allowed us to distinguish between annual variability and treatment effects. Our research was unique for four primary reasons: (1) it was an experimental study that included multiple years of pre- and post-harvest data collection; (2) the geographic extent included much of western Washington; (3) it included the response of both LW and SW; and (4) it included complete removal of trees throughout the entire RMZ (clearcut to the stream edge), which was included in all statistical analyses.

11-3. OBJECTIVES

Our objective was to describe changes in headwater channels of non-fish-bearing (i.e., Type N) headwater basins associated with three different buffer treatments compared with reference conditions. Specifically, we quantified changes in headwater stream hydrology (e.g., wetted width and depth), channel characteristics (e.g., bankfull width and stream substrates) and channel units (e.g., pools and riffles) across 17 non-fish-bearing basins. We focused our analysis on basin-scale responses, that is, we did not investigate responses at the channel reach scale (sensu Montgomery and Buffington 1997).

11-4. STUDY SITE DESCRIPTION

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 11-1a**) 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 11-1b). 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 11-1c). 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 11-1d). Forced step-pool reaches are created when wood forms most of the channel-spanning steps (**Figure 11-1e, 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 suspect that our study sites have been subjected to a similar frequency of debris flow scour events (Montgomery and Buffington 1998). We also 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).



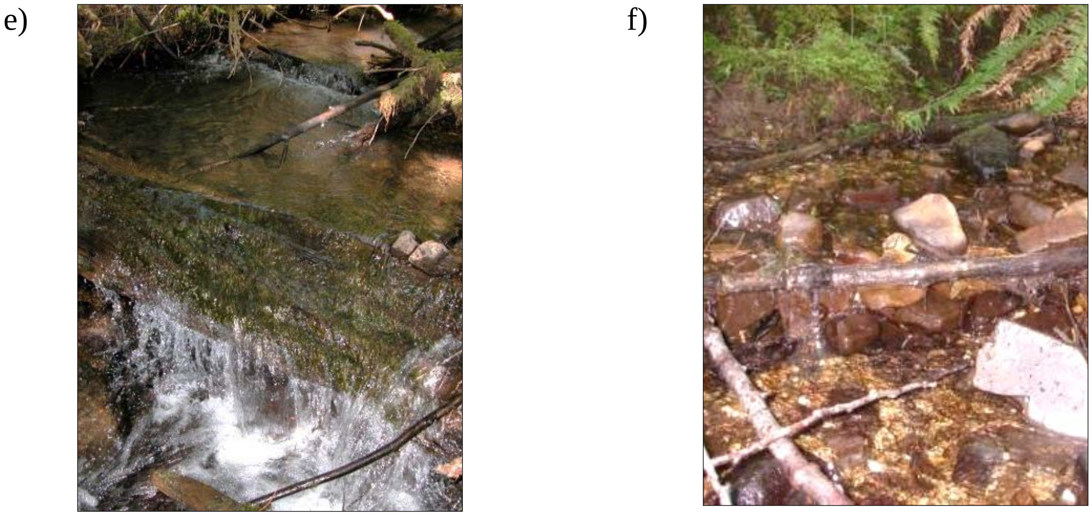


Figure 11-1. Study stream reaches exhibiting (a) colluvial, (b) bedrock, (c) cascade, and (d) step-pool channel-reach morphologies; (e) forced step-pool channel-reach morphology composed of large wood (LW; i.e., >10 cm diameter) and (f) a step composed of small wood (SW; i.e., ≤10 cm diameter).

11-5. METHODS

11-5.1. SAMPLING STRATEGY

We evaluated stream channel characteristics along the mainstem channel of each study site in three pre-harvest (2006–2008) and two post-harvest (2009−2010) years, or the pre- and post-harvest periods, respectively. For most metrics, sampling occurred between 6 May and 2 August of each year, with some exceptions. We sampled the WIL1-0% 18 through 31 March 2008 and the OLYM-0% 28 September through 12 October 2009. We did not sample the OLYM-REF in 2006 because approval for use of this site in the study had not yet been granted. We did not sample the WIL2-REF1 in 2009 because this site was intended to be a buffer treatment and the harvest had not been applied. Later, when the latter site was not harvested, we included it as a second reference in the Willapa 2 block (see Chapter 2 – *Study Design*).

We delineated 10-m intervals along the mainstem channel from the fish end point (F/N break) upstream to, but not including, the uppermost point of perennial flow (PIP), which we sampled separately. We sampled 20 contiguous 10-m intervals (200 m) 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; **Figure 11-2a**),

2) A minimum of 50% of the remaining mainstem channel length for moderately sized sites (300–800 m mainstem length; n = 10; **Figure 11-2b**), or

3) A minimum of 25% of the remaining mainstem channel length for the largest sites (>800 m mainstem length; n = 6; **Figure 11-2c**).

The systematic sample for (2) and (3) occurred above the 200-m contiguous reach sampled in two consecutive 10-m stream reaches 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 m (56 ft) in length, or equal to the riparian buffer radius required for PIPs under Forest Practices Rules (WFPB 2001). The length of the interval located upstream of the PIP was either 17 m or equal to the length of the non-perennial Type Ns Water[1] 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.

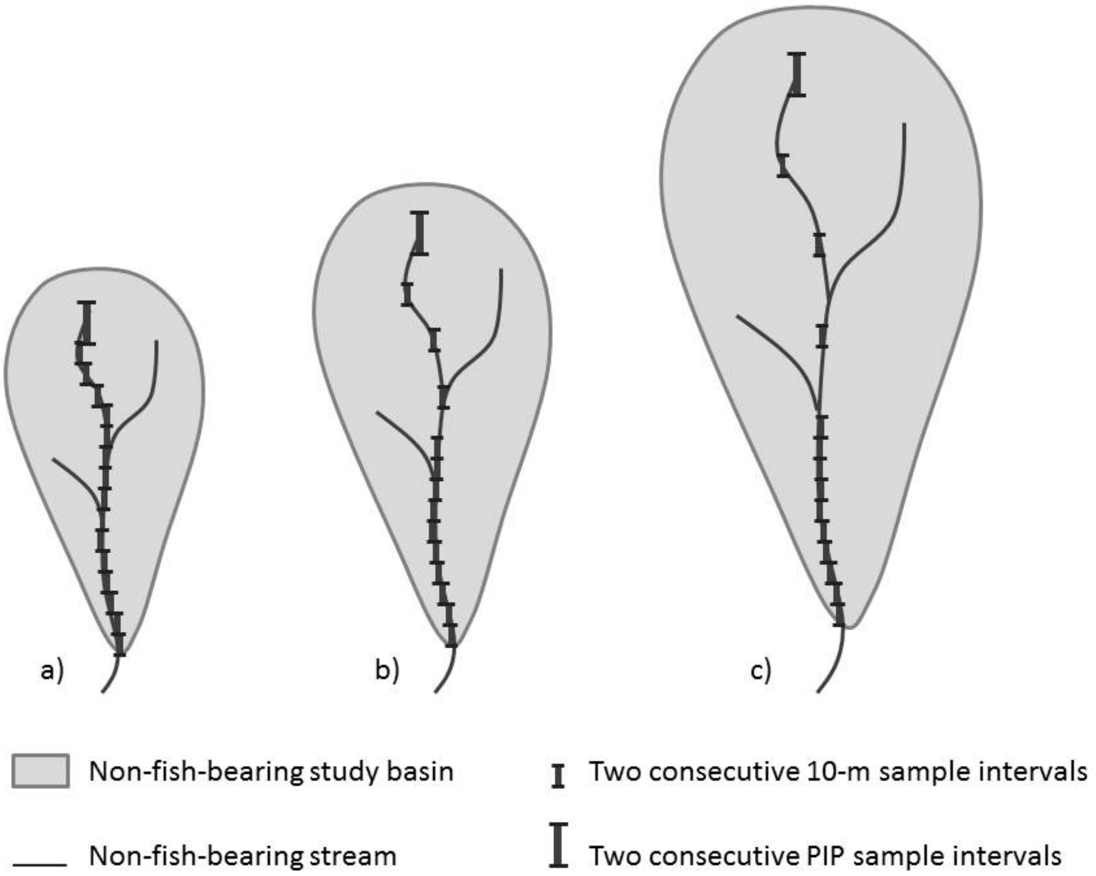


Figure 11-2. Sampling schematic for the (a) smallest (<300 m mainstem length), (b) moderate (300–800 m mainstem length), and (c) largest (>800 m mainstem length) sites.

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[1] 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).

11-5.2. DATA COLLECTION AND VARIABLE SUMMARY

We measured stream channel characteristics in two-person teams and reassigned teams daily to reduce bias. Crewmembers received extensive training over the course of a week, calibration between staff occurred periodically throughout the sampling season and at least three returning field staff members were present during each sampling season.

11-5.2.1. Stream Hydrology and Channel Metrics

11-5.2.1.a. Point measurements

We measured wetted width, stream depth, bankfull width (**Table 11-1**), and dominant stream substrate (**Table 11-2**) at a point 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 dominant substrate as the most commonly encountered inorganic substrate along a visual line perpendicular to the stream axis and within the wetted stream channel. We defined substrates according to a modified Wentworth classification (Wentworth 1922). If the stream was dry, we evaluated dominant substrate within the bankfull width. For wetted width, stream depth and bankfull width, we calculated averages for each study site for a single site-wide estimate by year. For substrate, we calculated annual site-wide estimates of the proportion of sample intervals dominated by: (1) fines and sand; (2) gravel and cobble; and (3) boulder and bedrock.

Table 11-1. Stream channel metrics measured (modified from Platts *et al.* 1983). Unit specifies the unit of measure. Precision specifies the resolution to which a value was measured.

Response Metric	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 was braided, wetted width was the sum of the individual widths for each braid.(modified from Platt *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 was braided, depth was measured for the dominant thread. (modified from Platt *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 was braided, bankfull width was the width as measured across all braids. (WAC 222-16-010)	m	0.1

Table 11-2. Substrate types and definitions for evaluation of dominant inorganic substrate.

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

Some research has focused on static measures such as residual water depth (i.e., the depth that would exist if there was no surface flow and pools were filled only to their lips; Bathurst 1981) that are independent of current discharge rates and remain constant regardless of the timing of measurement. The advantage of including static measures in evaluations of stream conditions is that they are independent of flow and can be measured consistently at any time of year. Conversely, while non-static measures, such as wetted water width, are flow dependent, including them allows the researcher to evaluate current conditions during a time identified as biologically important for biota of interest. In this case, we were interested in the stream conditions during the period in which stream-associated amphibians become active for the purpose of foraging and breeding. 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 (see section 11-5.3. *Analysis*). Consistent changes observed across all treatments, including the reference, would be an indication of annual variability (e.g., changes in annual precipitation), which is independent of a treatment effect.

11-5.2.1.b. Hydrology and channel hypotheses

We suspected that post-harvest hydrologic and channel characteristics would be greatly influenced by the large amounts of wood that were expected to enter the stream channel as a result of clearcut logging. Small wood is more abundant in headwater streams than in areas downstream because stream power is typically too low to transport the wood downstream (Bilby and Ward 1989; Maxa 2009). As such, we anticipated that both LW (in this study, pieces >10 cm diameter) and SW (pieces ≤10 cm diameter) would play functional roles, contributing to step formation and changes in channel morphology (Gomi *et al.* 2001; Maxa 2009). We expected a post-harvest increase in both wetted width and depth in harvested sites with an increase in channel roughness, which we anticipated would increase stream width (Keller and Swanson 1979; Trotter 1990) and slow the rate of water discharge (Lisle 1986). We also expected changes in the magnitude and timing of water delivery to the soil as a result of forest canopy removal and reductions in interception and evapotranspiration (Lewis *et al.* 2001; Keim and Skaugset 2003; Johnson *et al.* 2007). Consistent with Burton (1997) and Keppeler and Ziemer (1990), we expected a post-harvest increase in stream flows and, conversely, a decrease in dry stream length. Consistent with O'Connell and colleagues (2000) we expected no difference in the pre- to post-harvest change in bankfull width by treatment. Finally, consistent with Jackson and colleagues (2001) and Bilby and Bisson (1998), among others, we expected to find increasing amounts of fine and sand substrates and less gravel and cobble associated with a decreasing length of riparian buffer retained in the RMZ. We did not expect a change in the proportion of the stream dominated by boulder and bedrock as it was unlikely that the increase in fines and sand would be substantial enough to bury what in these systems are large substrates relative to the overall size of the stream channel. Hypotheses for each response variable are outlined in **Table 11-3**.

Table 11-3. Predicted responses (pre- to post-harvest change in buffer treatment sites compared to the reference) for each hydrology and stream channel metric.

Category	Response Metric	Predicted Response
Stream hydrology	Wetted width	Increase
	Depth	Increase
	Dry length	Decrease
Stream channel	Bankfull width	No change
	Fines and sand	Increase
	Gravel and cobble	Decrease
	Boulder and bedrock	No change

11-5.2.2. Stream Channel Units

Channel units are morphologically distinct areas that extend up to several channel widths in length (Montgomery and Buffington 1998). Channel units differ from channel reaches in that units are spatially embedded within a channel reach, which exhibits similar bedforms over stretches of stream that are many channel widths in length (Montgomery and Buffington 1998). Physical parameters used to separate channel units include channel slope, depth, bed material, roughness and flow velocity (MacDonald *et al.* 1991). Both the size and the classification of individual channel units are flow dependent, increasing or decreasing in area and volume with changes in discharge (MacDonald *et al.* 1991). In order to make our measurements comparable and reduce variability, we carried out our unit surveys during similar flow conditions (i.e., similar time each year; Platts *et al.* 1983) and categorized units into only three types: pool, riffle and step (**Table 11-4**). Due to the difficulty in differentiating between them, especially in headwater streams, our riffle category included both the riffle and cascade channel units described by Bisson and colleagues (1982). We measured unit length along the longest linear length of the unit. Pool maximum depth was recorded for the deepest location of the pool as the height of the water column from the existing water surface level to the channel bottom (modified from Platts *et al.* 1983).

In the post-harvest period, we recorded the length of each sample interval obstructed by newly recruited wood in the form of logging slash and windthrow from the RMZ. We were unable to quantify channel unit-specific measures in obstructed reaches due to the limited plot length and the possibility that the removal of wood required to access the stream would modify unit characteristics.

Table 11-4. Stream channel unit types and definitions. Channel unit types were classified according to the current hydrology of the stream at the time of sampling.

Channel Unit	Definition
Pool	Slow water reach: 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 reach: shallow with rapid and turbulent water, includes high and low gradient cascades, riffles and runs. Dominant substrates include boulders, cobble, gravel, and bedrock.
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.
Dry	Lacks surface water; typically characterized by the presence of mineral substrates and evidence of scour, but may include areas with long-term subterranean flow where the channel is no longer apparent (i.e., not merely temporarily obscured by downed wood).

11-5.2.2.a. Unit-specific measures

We evaluated channel units in their entirety, including any portion that extended upstream beyond the sample interval. We recorded the wetted surface length of each pool and riffle to the nearest 0.1 m (3.9 in). We recorded the maximum pool depth to the nearest centimeter. We recorded the step key piece (i.e., the piece responsible for the formation of the step) by type (**Table 11-5**), the diameter of the key piece (cm) for wood pieces, and the step height from the channel bed to the top of the step (cm). To calculate site-wide estimates of length, height and depth for riffles and pools, we averaged each by site and year. To calculate the proportion of steps keyed by wood, we divided the number of instances that the step key piece was wood by the total number of steps by site and year. We averaged the diameters of wood key pieces by site and year.

Table 11-5. Step key piece types and definitions.

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

11-5.2.2.a.(i). Unit density

To compare the frequency of each channel unit type in flowing, unobstructed reaches, we calculated an adjusted sample length by subtracting the dry and obstructed lengths from the target sample length for each site and year (**Appendix Table 11-A-1**):

$$\text{Sample length (m)} = \text{target sample} - (\text{dry} + \text{obstructed}) \tag{11-1}$$

We calculated unit density as the sum of the number for each unit type (e.g., pool, riffle) 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.

11-5.2.2.a.(ii). Channel rise attributed to steps

We calculated the channel rise attributed to steps from the interval slope length and the channel gradient. Interval slope length was typically 10 m, but differed in PIP intervals and sometimes due to other factors that resulted in intervals less than 10 m (e.g., road crossings). We measured the channel gradient from the start to end point of each interval to the nearest degree using a Suunto® clinometer. Channel gradient was recorded in only two pre- (2006 and 2007) and two post-harvest (2009 and 2010) years. We averaged pre-harvest (2006 and 2007) gradients for use with all pre-harvest calculations and post-harvest (2009 and 2010) gradients for use with all post-harvest calculations. We did this for two reasons: (1) we did not record gradient in 2008; and (2) some gradient values were missing for an interval in one pre- or post-harvest year (especially in the post-harvest period, when we could not measure gradient in some stream reaches due to instream wood obstructions).

We converted each gradient to radians (g_{rad}) and, assuming a right angle (**Figure 11-3**), calculated the channel rise, or height (h), of each interval in meters as:

$$h = sin(g_{rad}) * l \tag{11-2}$$

where: $sin(g_{rad})$ is the sine of the gradient in radians, and
l is the slope distance of the sample interval in meters.

We calculated the proportion of the channel rise attributed to steps for each interval as the sum of all step heights divided by the channel rise. We calculated the average proportion of the channel rise attributed to steps by site and year.

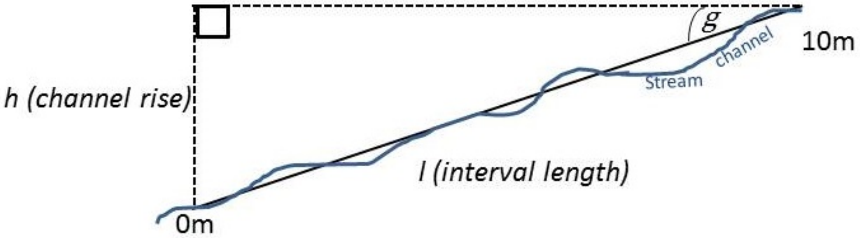


Figure 11-3. Sample interval, with gradient (g) and interval slope length (l) used to calculate channel rise (h).

11-5.2.2.b. Channel unit distributions

Though we did not classify channel unit types in dry reaches, we did compare the pre- and post-harvest distribution of channel units, including dry reaches, in unobstructed channels as the sum of the length of each divided by the sum of all unit lengths (i.e., total sample length) by site and year. We averaged results by treatment and period and displayed them as a percent.

Since we only encountered obstructed reaches in the post-harvest period, we compared the distribution of units in obstructed with those in unobstructed reaches for this period only. To do this, we sub-sampled obstructed areas according to the methods outlined in Chapter 6 – *Wood Recruitment and Loading*. We recorded the dominant channel unit by length for each 3-m sample plot. We calculated the proportion of plots dominated by each unit type by site and year. We identified the dominant channel unit in unobstructed reaches as the type with the highest proportion by length (height for steps) in each sample interval, and calculated the proportion of intervals classified as each unit type by site and year. We averaged results for unobstructed and obstructed reaches and displayed them as a percent.

11-5.2.2.c. Channel unit hypotheses

Like hydrologic and channel characteristics, we suspected channel units would be greatly influenced by the large amounts of wood in buffer treatments post-harvest. Because wood can increase pool length by extending the upstream limit of pools (Lisle 1986), we expected a pre- to post-harvest increase in pool length. Consistent with the findings of Lisle (1986), we did not anticipate a post-harvest increase in pool depth or density in harvested sites. We expected a post-harvest decrease in riffle length in harvested sites concurrent with our predicted increase in pool length. However, since we predicted no change in pool density we did not expect a change in riffle density. We anticipated a relationship between step height, step density and the proportion of steps composed of wood for harvested sites in the post-harvest period. We expected that step height would decrease as the number of wood pieces contributing to step formation (i.e., step density and proportion of wood steps) increased. Jackson and Sturm (2002) noted an increased incidence of wood steps in areas managed for timber compared with streams in unharvested forests. For harvested sites in the post-harvest period, we anticipated an increase in the proportion of SW, and a subsequent decrease in the mean diameter of wood pieces keying steps. Finally, our prediction for the change in the percent channel rise attributed to steps is consistent with the findings of MacFarlane and Wohl (2003), who found a decrease in the percentage of water surface drop created by steps in stream reaches with LW. Hypotheses for each response variable are outlined in (**Table 11-6**).

Table 11-6. Predicted response (pre- to post-harvest change in buffer treatment sites compared to the reference) for each channel unit type.

Channel Unit	Response Metric	Predicted Response
Pool	Pool length	Increase
	Pool max depth	No change
	Pool density	No change
Riffle	Riffle length	Decrease
	Riffle density	No change
Step	Step height	Decrease
	Step density	Increase
	Proportion steps-wood	Increase
	Step key piece diameter	Decrease
	Channel rise attributed to steps	Decrease

11-5.3. ANALYSIS

We designed this study to evaluate response differences among treatments at the site scale, not to investigate within-site variability. Though some data may lend themselves to evaluation within sites, those comparisons were not intended to be formally addressed in this report. Analyses following the Before-After Control-Impact (BACI) design evaluated the generalized null hypothesis:

$$\Delta T_{\mathrm{REF}} = \Delta T_{100\%} = \Delta T_{\mathrm{FP}} = \Delta T_{0\%} \tag{11-3}$$

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 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-harvest 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. 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 ratio and proportion response data (i.e., fines and sand, gravel and cobble, boulder and bedrock, step key piece, and channel rise attributed to steps) prior to analysis, in order to constrain all estimates to the (0,1) interval. Due to the presence of exact zeroes and exact 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). We used the value of 0.02 for all transformations, so 0.02 is the smallest non-zero value among all ratio or proportion response data.

We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals. We found evidence of heteroscedasticity for pool and riffle lengths, and step height and used a natural log transformation to stabilize the variance. Additionally, the GLIMMIX Procedure for analyses of channel unit density reports results on the natural log (ln) scale. In both cases, 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 −1.5 and 1.5 equate to a 50% decrease and a 50% increase from pre- to post-harvest, respectively. We will present results on a natural log scale. For results that were statistically significant, we will present the back-transformed proportional differences in the discussion.

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 = 17), and especially with an alpha of 0.1, we could expect to have a number of “statistically significant” results based on chance alone. We caution the reader to interpret our results with this in mind.

11-6. RESULTS

11-6.1. STREAM HYDROLOGY

11-6.1.1. Wetted Width

Mean stream wetted width ranged from 0.4 to 2.1 m across sites in both the pre- and post-harvest periods (**Appendix Table 11-A-2**). The pre- to post-harvest change within treatment ranged from 0.1 to 0.4 m (**Table 11-7** and **Figure 11-4**). The P-value for the period × treatment contrast was 0.03, indicating that the amount of change in stream width depended on treatment. We estimated the change in the 0% treatment to be 0.3 m less than the change in the reference (P = 0.01), 100% (P = 0.01) and FP (P = 0.04) treatments (**Table 11-8**).

Table 11-7. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for stream wetted width (m), i.e., (post − pre).

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	0.4	0.07	0.2	0.5
100%	0.4	0.08	0.2	0.5
FP	0.4	0.10	0.2	0.5
0%	0.1	0.08	−0.1	0.2

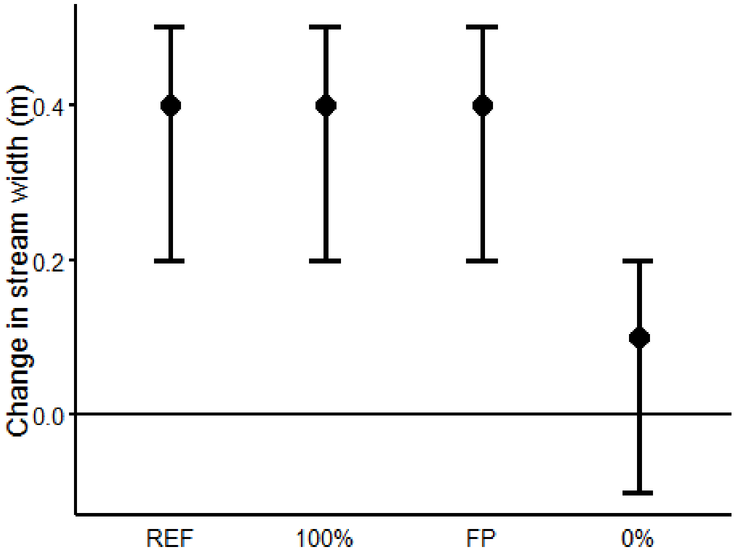


Figure 11-4. The within-treatment change and 95% confidence intervals (CI) for stream wetted width (m), i.e., (post − pre).

Table 11-8. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for stream wetted width (m), i.e., [TrtA(post − pre) − TrtB(post − pre)]. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	P-value	95% CI	
				Lower	Upper
100% vs. REF	0.0	0.11	0.89	−0.2	0.2
FP vs. REF	0.0	0.12	0.74	−0.3	0.2
0% vs. REF	−0.3	0.11	0.01	−0.5	−0.1
0% vs. FP	−0.3	0.13	0.04	−0.5	0.0
0% vs. 100%	−0.3	0.12	0.01	−0.5	−0.1
FP vs. 100%	0.0	0.13	0.85	−0.3	0.2

11-6.1.2. Depth

Mean stream depth ranged from 3 to 14 cm in the pre- and 3 to 13 cm in the post-harvest period (**Appendix Table 11-A-2**). The pre- to post-harvest change within treatment ranged from 1 to 3 cm (**Table 11-9** and **Figure 11-5**). The P-value for the period × treatment contrast was 0.11, indicating that the amount of change in stream depth did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-10**).

Table 11-9. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for stream depth (cm), i.e., (post − pre).

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	2	0.5	1	3
100%	3	0.6	2	4
FP	3	0.7	2	4
0%	1	0.6	0	3

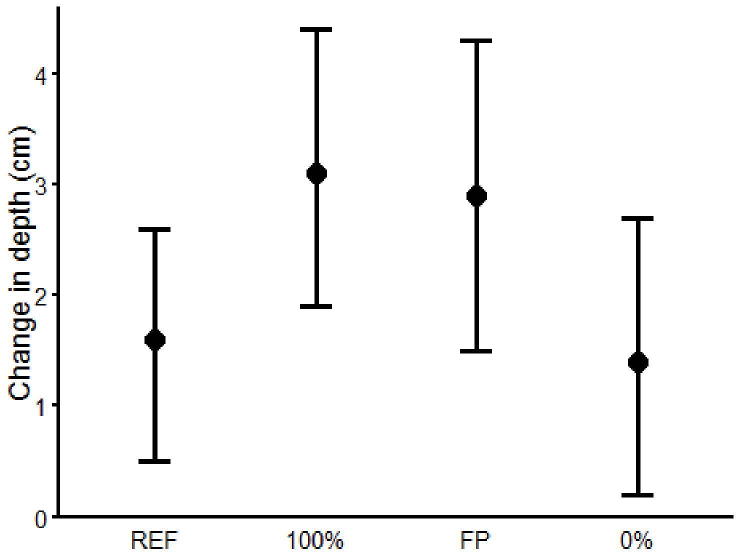


Figure 11-5. The within-treatment change and 95% confidence intervals (CI) for stream depth (cm), i.e., (post − pre).

Table 11-10. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence interval (CI) for stream depth (cm), i.e., [TrtA(post − pre) − TrtB(post − pre)]. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	**Difference**	**SE**	**95% CI**	
			Lower	**Upper**
100% vs. REF	2	0.8	0	3
FP vs. REF	1	0.9	0	3
0% vs. REF	0	0.8	−2	2
0% vs. FP	−2	0.9	−3	0
0% vs. 100%	−2	0.9	−3	0
FP vs. 100%	0	0.9	−2	2

11-6.1.3. Dry Length

The length of dry stream ranged from 0.0 to 188.6 m in the pre- and 3.7 to 152.8 m in the post-harvest period (**Appendix Table 11-A-1**). The pre- to post-harvest change in dry length was estimated for the logit-transformed proportion of the stream characterized as dry (**Table 11-11** and **Figure 11-6**). The P-value for the period × treatment contrast was 0.12, indicating that the amount of change in dry stream length did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-12**).

Table 11-11. The within-treatment change, standard error (SE) of the estimate, and 95% confidence interval (CI) for the logit-transformed proportion of the stream characterized as dry, i.e., [logit(post) − logit(pre)].

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	0.03	0.23	−0.44	0.50
100%	0.83	0.27	0.28	1.38
FP	0.11	0.32	−0.52	0.75
0%	0.58	0.27	0.03	1.13

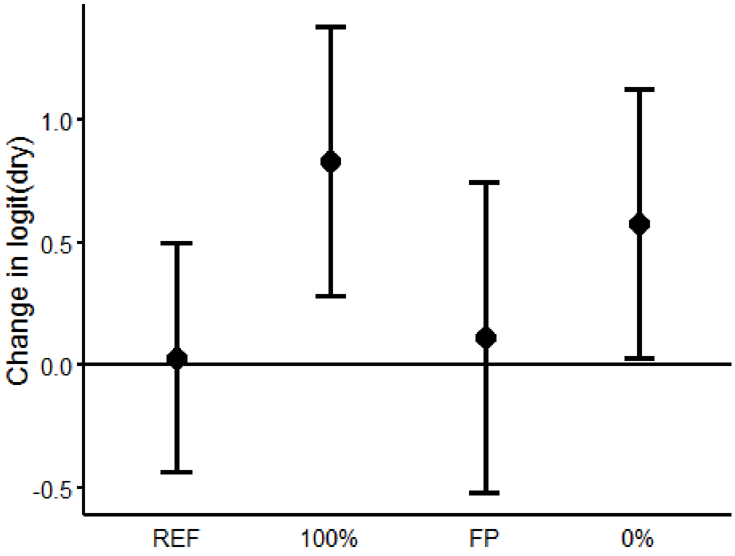


Figure 11-6. The within-treatment change and 95% confidence interval (CI) for the logit-transformed proportion of the stream characterized as dry, i.e., [logit(post) − logit(pre)].

Table 11-12. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence interval (CI) for the logit-transformed proportion of the stream characterized as dry, i.e., (logit[TrtA(post)] − logit[TrtA(pre)]) − (logit[TrtB(post)] − logit[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	0.80	0.36	0.08	1.52
FP vs. REF	0.08	0.39	−0.71	0.87
0% vs. REF	0.55	0.36	−0.18	1.27
0% vs. FP	0.47	0.42	−0.37	1.30
0% vs. 100%	−0.25	0.39	−1.03	0.52
FP vs. 100%	−0.72	0.42	−1.56	0.12

11-6.2. STREAM CHANNEL

11-6.2.1. Bankfull Width

Mean bankfull width ranged from 0.8 to 3.1 m in the pre- and 1.0 to 2.6 m in the post-harvest period (**Appendix Table 11-A-2**). The pre- to post-harvest change within treatment ranged from −0.3 to 0.2 m among treatments (**Table 11-13** and **Figure 11-7**). The P-value for the period × treatment contrast was <0.001, indicating that the amount of change in the bankfull width depended on treatment. We estimated the change in the 0% treatment to be 0.4 m less than the change in the reference (P <0.001) and 0.5 m less than the change in the 100% (P <0.0001) and FP (P <0.001) treatments, respectively (**Table 11-14**).

Table 11-13. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for bankfull width (m), i.e., (post − pre).

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	0.1	0.07	0.0	0.3
100%	0.2	0.08	0.1	0.4
FP	0.2	0.10	0.0	0.4
0%	−0.3	0.08	−0.5	−0.1

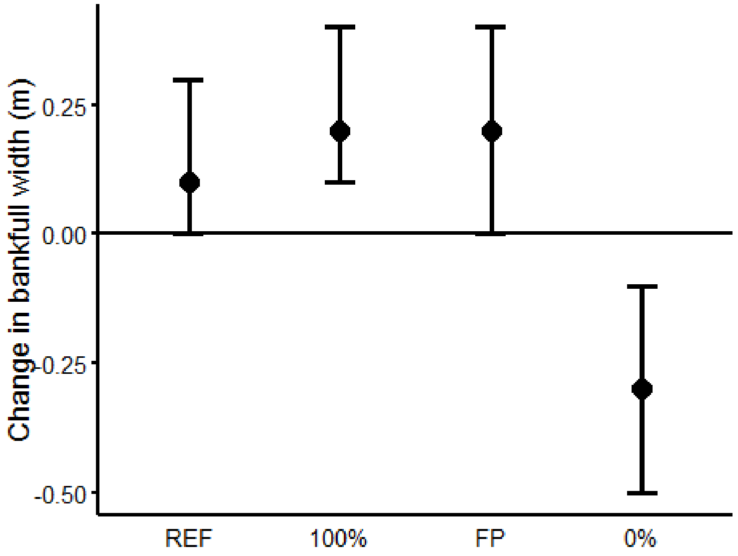


Figure 11-7. The within-treatment change and 95% confidence intervals (CI) for bankfull width (m), i.e., (post − pre).

Table 11-14. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the bankfull width (m), i.e., [TrtA(post − pre) – TrtB(post − pre)]. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	**Difference**	**SE**	**P-value**	**95% CI**	
				Lower	**Upper**
100% vs. REF	0.1	0.11	0.27	−0.1	0.3
FP vs. REF	0.1	0.12	0.53	−0.2	0.3
0% vs. REF	−0.4	0.11	<0.001	−0.6	−0.2
0% vs. FP	−0.5	0.13	<0.001	−0.7	−0.2
0% vs. 100%	−0.5	0.12	<0.0001	−0.8	−0.3
FP vs. 100%	0.0	0.13	0.72	−0.3	0.2

11-6.2.2. Stream Substrate

11-6.2.2.a. Fines and Sand

The proportion of stream dominated by fines and sand ranged from 0.00 to 0.86 in the pre- and 0.02 to 0.70 in the post-harvest period **Appendix Table 11-A-2**). We estimated the pre- to post-harvest change within treatment on the logit-transformed proportion of stream dominated by fines and sand (**Table 11-15** and **Figure 11-8**). The P-value for the period × treatment contrast was 0.20, indicating that the amount of changes in the proportion of the stream dominated by fines and sand did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-16**).

Table 11-15. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the logit-transformed proportion of stream dominated by fines and sand, i.e., [logit(post) − logit(pre)].

Treatment	**Change**	**SE**	**95% CI**	
			Lower	**Upper**
REF	0.00	0.29	−0.59	0.59
100%	0.21	0.34	−0.48	0.90
FP	0.67	0.39	−0.12	1.47
0%	0.90	0.34	0.22	1.59

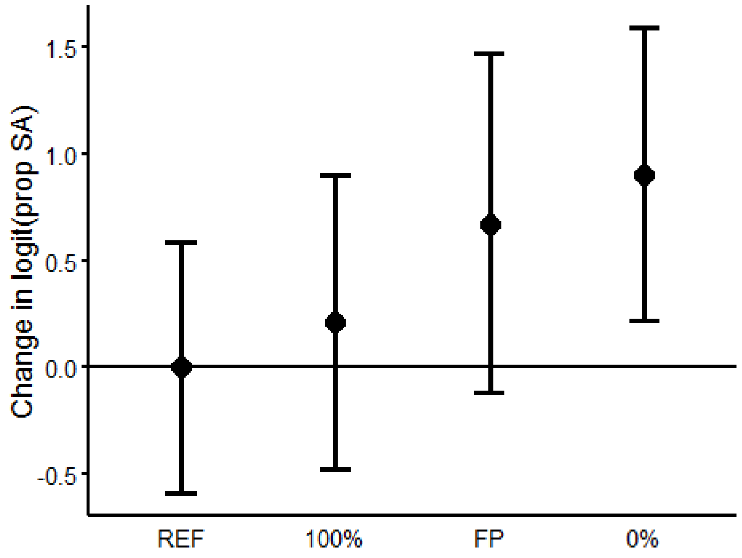


Figure 11-8. The within-treatment change and 95% confidence intervals (CI) for the logit-transformed proportion of stream dominated by fines and sand, i.e., [logit(post) − logit(pre)].

Table 11-16. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the logit-transformed proportion of stream dominated by fines and sand, i.e., (logit[TrtA(post)] − logit[TrtA(pre)]) − (logit[TrtB(post)] − logit[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	0.21	0.45	−0.70	1.11
FP vs. REF	0.67	0.49	−0.32	1.66
0% vs. REF	0.90	0.45	0.00	1.81
0% vs. FP	0.23	0.52	−0.82	1.28
0% vs. 100%	0.69	0.48	−0.28	1.66
FP vs. 100%	0.46	0.52	−0.59	1.51

11-6.2.2.b. Gravel and Cobble

The proportion of stream dominated by gravel and cobble ranged from 0.14 to 1.00 in the pre- and 0.30 to 0.85 in the post-harvest period (**Appendix Table 11-A-2**). We estimated the pre- to post-harvest change within treatment on the logit-transformed proportion of stream dominated by gravel and cobble (**Table 11-17**and **Figure 11-9**). The P-value for the period × treatment contrast was 0.46, indicating that the amount of change in the proportion of stream dominated by gravel and cobble did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-18**).

Table 11-17. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the logit-transformed proportion of stream dominated by gravel and cobble, i.e., [logit(post) − logit(pre)].

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	−0.11	0.24	−0.60	0.38
100%	−0.22	0.28	−0.79	0.35
FP	−0.61	0.33	−1.27	0.04
0%	−0.61	0.28	−1.18	−0.04

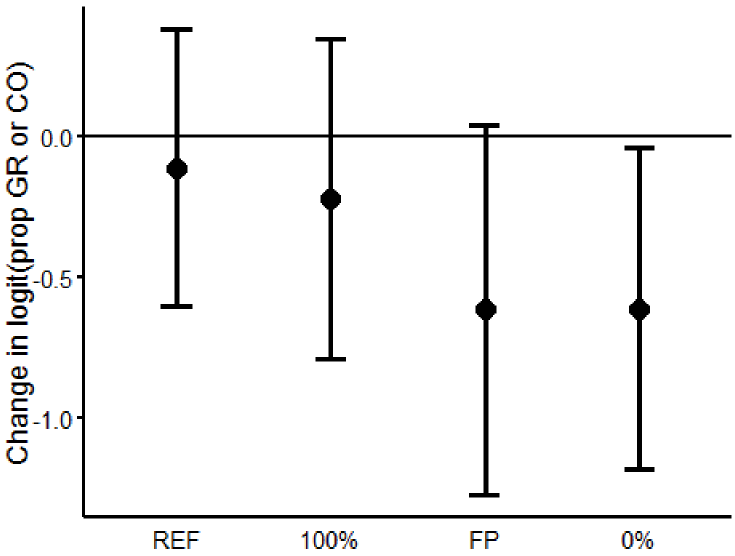


Figure 11-9. The within-treatment change and 95% confidence intervals (CI) for the logit-transformed proportion of stream dominated by gravel and cobble, i.e., [logit(post) − logit(pre)].

Table 11-18. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence interval (CI) for the logit-transformed proportion of stream dominated by gravel and cobble, i.e., (logit[TrtA(post)] − logit[TrtA(pre)]) − (logit[TrtB(post)] − logit[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	−0.11	0.37	−0.86	0.65
FP vs. REF	−0.50	0.41	−1.32	0.32
0% vs. REF	−0.50	0.37	−1.25	0.26
0% vs. FP	0.01	0.43	−0.86	0.88
0% vs. 100%	−0.39	0.40	−1.20	0.42
FP vs. 100%	−0.40	0.43	−1.27	0.47

11-6.2.2.c. Boulder and Bedrock

The proportion of stream dominated by boulder and bedrock ranged from 0.00 to 0.32 in the pre- and 0.00 to 0.39 in the post-harvest period (**Appendix Table 11-A-2**). We estimated the pre- to post-harvest change within-treatment on the logit-transformed proportion of stream dominated by boulder and bedrock (**Table 11-19** and **Figure 11-10**). The P-value for the period × treatment contrast was 0.50, indicating that the amount of change in the proportion of the stream dominated by boulder and bedrock did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-20**).

Table 11-19. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the logit-transformed proportion of stream dominated by boulder and bedrock, i.e., [logit(post) − logit(pre)].

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	0.24	0.22	−0.20	0.68
100%	0.04	0.26	−0.48	0.55
FP	0.18	0.30	−0.41	0.77
0%	−0.26	0.26	−0.77	0.26

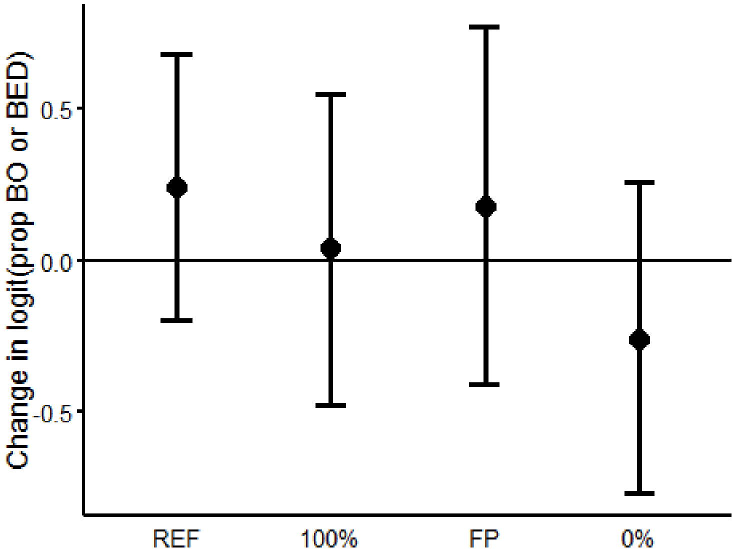


Figure 11-10. The within-treatment change and 95% confidence intervals (CI) for the logit-transformed proportion of stream dominated by boulder and bedrock, i.e., [logit(post) − logit(pre)].

Table 11-20. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the logit-transformed proportion of stream dominated by boulder and bedrock, i.e., (logit[TrtA(post)] − logit[TrtA(pre)]) − (logit[TrtB(post)] − logit[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	−0.20	0.34	−0.88	0.47
FP vs. REF	−0.06	0.37	−0.80	0.68
0% vs. REF	−0.50	0.34	−1.18	0.18
0% vs. FP	−0.44	0.39	−1.22	0.35
0% vs. 100%	−0.29	0.36	−1.02	0.43
FP vs. 100%	0.14	0.39	−0.64	0.93

11-6.3. STREAM CHANNEL UNITS

11-6.3.1. Pools

11-6.3.1.a. Pool length

Mean pool length ranged from 0.7 to 1.7 m in the pre- and 0.7 to 1.5 m in the post-harvest period (**Appendix Table 11-A-3**). We estimated the pre- to post-harvest change within treatment on the natural log (ln) of pool length (**Table 11-21** and **Figure 11-11**). The P-value for the period × treatment contrast was 0.03, indicating that the amount of change in pool length depended on treatment. We estimated the change in the reference to be less than the change in the 100% (P = 0.05), FP (P <0.01) and 0% treatments (P = 0.04; **Table 11-22**).

Table 11-21. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the natural log (ln) of pool length (m), i.e., [ln(post) − ln(pre)].

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	−0.09	0.04	−0.16	−0.01
100%	0.03	0.04	−0.06	0.11
FP	0.09	0.05	−0.01	0.19
0%	0.03	0.04	−0.05	0.12

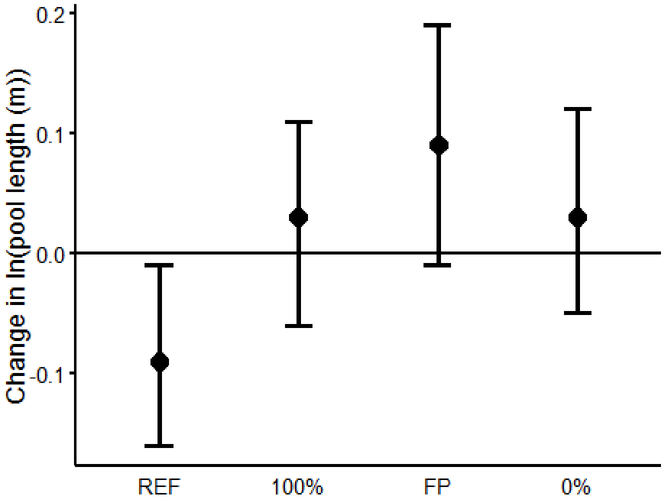


Figure 11-11. The within-treatment change and 95% confidence intervals (CI) for the natural log (ln) of pool length (m), i.e., [ln(post) − ln(pre)].

Table 11-22. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the natural log (ln) of pool length (m), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	P-value	95% CI	
				Lower	Upper
100% vs. REF	0.12	0.06	0.05	0.00	0.23
FP vs. REF	0.18	0.06	0.01	0.06	0.31
0% vs. REF	0.12	0.06	0.04	0.01	0.24
0% vs. FP	−0.06	0.07	0.37	−0.19	0.07
0% vs. 100%	0.01	0.06	0.91	−0.12	0.13
FP vs. 100%	0.07	0.07	0.32	−0.07	0.20

11-6.3.1.b. Pool maximum depth

The pool maximum depth ranged from 11 to 28 cm in the pre- and 11 to 26 cm in the post-harvest period (**Appendix Table 11-A-3**). The pre- to post-harvest change within treatment ranged from 1 to 3 cm among treatments (**Table 11-23** and **Figure 11-12**). The P-value for the period × treatment contrast 0.70, indicating that the amount of change in pool maximum depth did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-24**).

Table 11-23. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the pool maximum depth (cm), i.e., (post − pre).

Treatment	Change	SE	95% CI Lower	95% CI Upper
REF	1.2	0.8	−0.4	2.8
100%	2.4	0.9	0.5	4.2
FP	2.6	1.1	0.4	4.7
0%	1.8	0.9	−0.0	3.6

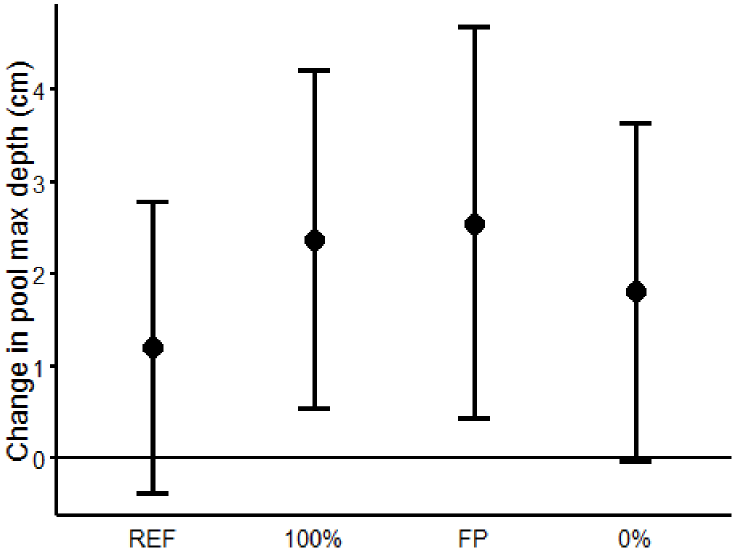


Figure 11-12. The within-treatment change and 95% confidence intervals (CI) for the pool maximum depth (cm), i.e., (post − pre)]

Table 11-24. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence interval (CI) for the pool maximum depth (cm), i.e., [TrtA(post – pre) – TrtB(post – pre)]. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Difference	SE	95% CI Lower	95% CI Upper
100% vs. REF	1.2	1.2	−1.3	3.6
FP vs. REF	1.4	1.3	−1.3	4.0
0% vs. REF	0.6	1.2	−1.8	3.0
0% vs. FP	−0.7	1.4	−3.5	2.1
0% vs. 100%	−0.6	1.3	−3.2	2.0
FP vs. 100%	0.2	1.4	−2.6	3.0

11-6.3.1.c. Pool density

Pool density ranged from 0.1 to 0.4 pools/m in the pre- and 0.1 to 0.6 pools/m in the post-harvest period (**Appendix Table 11-A-3**). We estimated the pre- to post-harvest change within treatment for the natural log (ln) of pool density (number/m; **Table 11-25** and **Figure 11-13**). The P-value for the period × treatment contrast was 0.55, indicating that the amount of change in pool density did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-26**).

Table 11-25. The within-treatment change, standard error (SE) of the estimate, and 95% confidence interval (CI) for the natural log (ln) of pool density (number/m), i.e., [ln(post) − ln(pre)].

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	0.20	0.09	0.02	0.39
100%	0.24	0.11	0.03	0.46
FP	0.26	0.15	−0.04	0.57
0%	0.01	0.14	−0.28	0.29

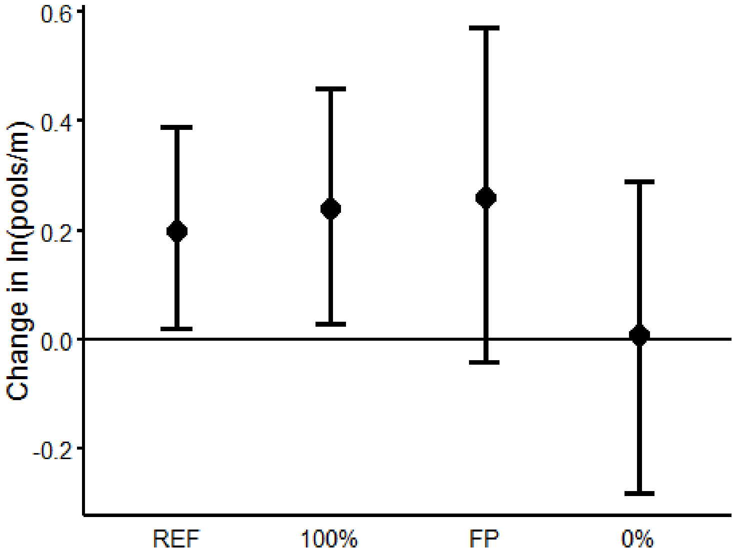


Figure 11-13. The within-treatment change and 95% confidence interval (CI) for the natural log (ln) of pool density (number/m), i.e., [ln(post) − ln(pre)].

Table 11-26. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence interval (CI) for the natural log (ln) of pool density (number/m), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	0.04	0.14	−0.24	0.32
FP vs. REF	0.06	0.18	−0.30	0.42
0% vs. REF	−0.19	0.17	−0.53	0.15
0% vs. FP	−0.26	0.21	−0.68	0.16
0% vs. 100%	−0.23	0.18	−0.59	0.12
FP vs. 100%	0.02	0.19	−0.35	0.40

11-6.3.2. Riffles

11-6.3.2.a. Riffle length

Mean riffle length ranged from 1.2 to 3.6 m in the pre- and 1.2 to 2.8 m in the post-harvest period (**Appendix Table 11-A-3**). We estimated the pre- to post-harvest change within treatment for the natural log (ln) of riffle length (**Table 11-27** and **Figure 11-14**). The P-value for the period × treatment contrast was 0.72, indicating that the amount of change in riffle length did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-28**).

Table 11-27. The within-treatment change, standard error (SE) of the estimate, and 95% confidence interval (CI) for the natural log (ln) of riffle length (m), i.e., [ln(post) − ln(pre)].

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	0.02	0.06	−0.11	0.14
100%	−0.05	0.07	−0.20	0.10
FP	−0.08	0.08	−0.25	0.09
0%	0.02	0.07	−0.13	0.17

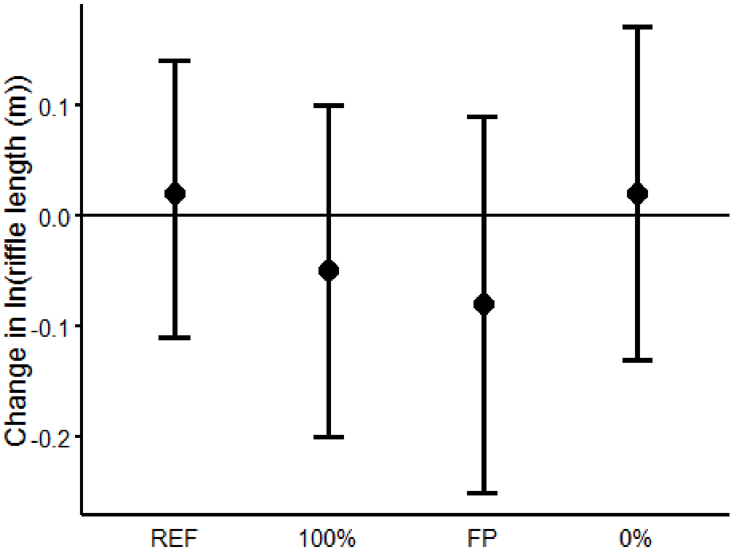


Figure 11-14. The within-treatment change and 95% confidence interval (CI) for the natural log (ln) of riffle length (m), i.e., [ln(post) − ln(pre)].

Table 11-28. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence interval (CI) for the natural log (ln) of riffle length (m), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	−0.07	0.10	−0.26	0.13
FP vs. REF	−0.10	0.11	−0.31	0.11
0% vs. REF	<0.01	0.10	−0.19	0.20
0% vs. FP	0.10	0.11	−0.12	0.33
0% vs. 100%	0.07	0.10	−0.14	0.28
FP vs. 100%	−0.03	0.11	−0.26	0.19

11-6.3.2.b. Riffle density

Riffle density ranged from 0.3 to 0.9 riffles/m in the pre- and 0.4 to 1.3 riffles/m in the post-harvest period (**Appendix Table 11-A-3**). We estimated the pre- to post-harvest change within treatment for the natural log (ln) of riffle density (**Table 11-29** and **Figure 11-15**). The P-value for the period × treatment contrast was 0.58, indicating that the amount of change in riffle density did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-30**).

Table 11-29. The within treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the natural log (ln) of riffle density (number/m), i.e., [ln(post) − ln(pre)].

Treatment	**Change**	**SE**	**95% CI**	
			Lower	**Upper**
REF	0.13	0.07	−0.02	0.27
100%	0.27	0.08	0.12	0.43
FP	0.17	0.11	−0.05	0.39
0%	0.16	0.09	−0.02	0.34

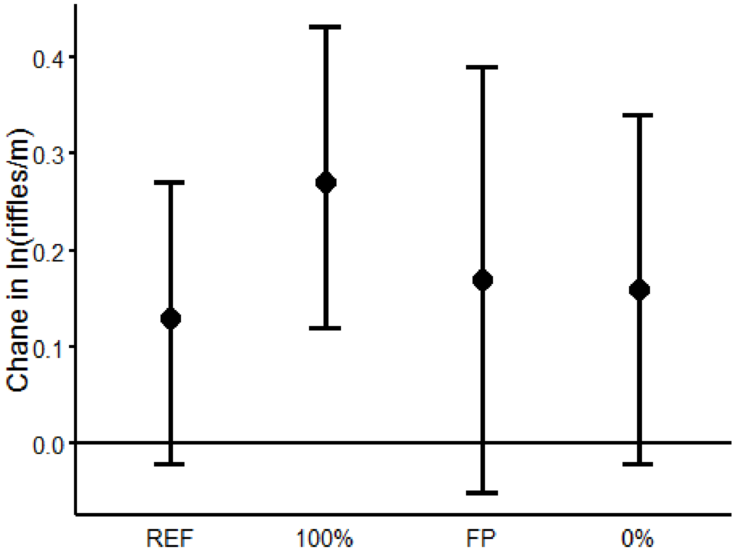


Figure 11-15. The within-treatment change and 95% confidence intervals (CI) for the natural log (ln) of riffle density (number/m), i.e., [ln(post) − ln(pre)].

Table 11-30. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the natural log (ln) of riffle density (number/m), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	0.15	0.11	−0.07	0.36
FP vs. REF	0.04	0.13	−0.22	0.30
0% vs. REF	0.03	0.12	−0.20	0.27
0% vs. FP	−0.01	0.14	−0.29	0.27
0% vs. 100%	−0.11	0.12	−0.35	0.13
FP vs. 100%	−0.10	0.13	−0.37	0.17

11-6.3.3. Steps

11-6.3.3.a. Step height

Mean step height ranged from 22 to 46 cm in the pre- and 26 to 53 cm in the post-harvest period (**Appendix Table 11-A-3**). We estimated the pre- to post-harvest change within-treatment for the natural log (ln) of step height (**Table 11-31** and **Figure 11-16**). The P-value for the period × treatment contrast was 0.65, indicating that the amount of change in step height did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-32**).

Table 11-31. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the natural log (ln) of step height (cm), i.e., [ln(post) − ln(pre)].

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	0.06	0.04	−0.02	0.14
100%	0.11	0.04	0.02	0.19
FP	0.11	0.05	0.01	0.21
0%	0.04	0.04	−0.04	0.13

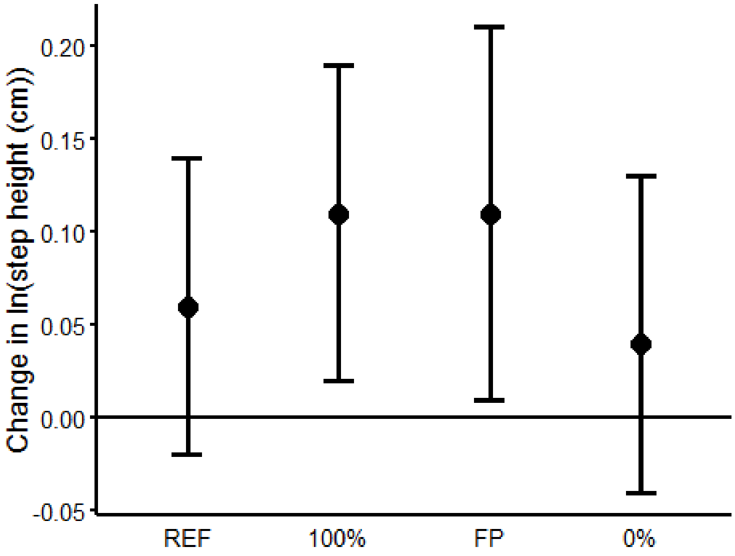


Figure 11-16. The within-treatment change and 95% confidence intervals (CI) for the natural log (ln) of step height (cm), i.e., [ln(post) − ln(pre)].

Table 11-32. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the natural log (ln) of step height (cm), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	0.05	0.06	−0.07	0.16
FP vs. REF	0.05	0.06	−0.08	0.18
0% vs. REF	−0.02	0.06	−0.13	0.10
0% vs. FP	−0.07	0.07	−0.20	0.07
0% vs. 100%	−0.06	0.06	−0.19	0.06
FP vs. 100%	0.00	0.07	−0.13	0.14

11-6.3.3.b. Step density

Step density ranged from 0.20 to 0.66 steps/m in the pre- and 0.22 to 0.52 steps/m in the post-harvest period (**Appendix Table 11-A-3**). We estimated the pre- to post-harvest change within-treatment for the natural log (ln) of step density (numbers/m; **Table 11-33** and **Figure 11-17**). The P-value for the period × treatment contrast was 0.45, indicating that the amount of change in step density did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-34**).

Table 11-33. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the natural log (ln) of step density (number/m), i.e., [ln(post) − ln(pre)].

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	0.05	0.07	−0.10	0.20
100%	0.11	0.08	−0.05	0.26
FP	−0.04	0.11	−0.27	0.18
0%	−0.08	0.10	−0.28	0.12

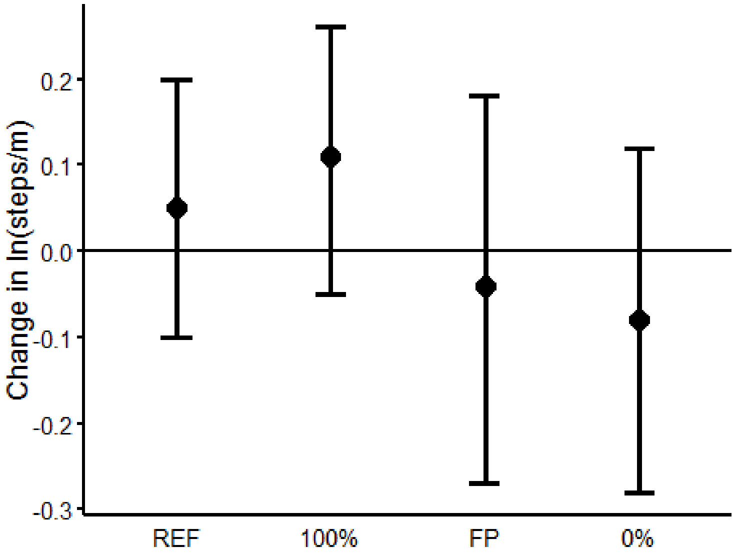


Figure 11-17. The within-treatment change and 95% confidence intervals (CI) for the natural log (ln) of step density (number/m), i.e., [ln(post) − ln(pre)].

Table 11-34. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the natural log (ln) of step density (number/m), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	**Difference**	**SE**	**95% CI**	
			Lower	**Upper**
100% vs. REF	0.06	0.11	−0.16	0.28
FP vs. REF	−0.09	0.13	−0.36	0.18
0% vs. REF	−0.13	0.12	−0.38	0.12
0% vs. FP	−0.04	0.15	−0.34	0.26
0% vs. 100%	−0.19	0.13	−0.44	0.07
FP vs. 100%	−0.15	0.14	−0.42	0.13

11-6.3.3.c. Step key piece

The proportion of steps keyed by wood ranged from 0.25 to 0.91 in the pre- and 0.32 to 0.92 in the post-harvest period (**Appendix Table 11-A-3**). We estimated the pre- to post-harvest change within-treatment for the logit-transformed proportion of steps keyed by wood (**Table 11-35** and **Figure 11-18**). The P-value for the period × treatment contrast was 0.59, indicating that the amount of change in the proportion of steps keyed by wood did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 11-36**).

Table 11-35. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the logit-transformed proportion of steps keyed by wood, i.e., [logit(post) − logit(pre)].

Treatment	**Change**	**SE**	**95% CI**	
			Lower	**Upper**
REF	0.16	0.12	−0.08	0.41
100%	−0.09	0.14	−0.38	0.20
FP	0.12	0.17	−0.21	0.46
0%	0.04	0.14	−0.25	0.33

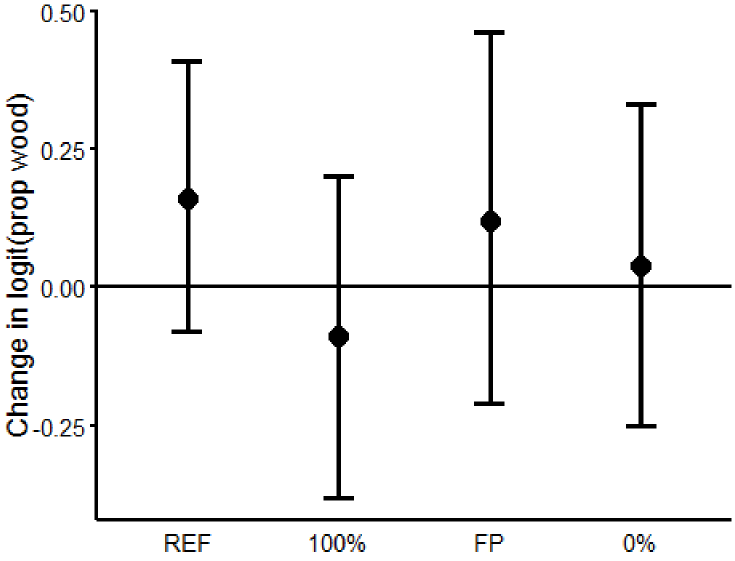


Figure 11-18. The within-treatment change and 95% confidence intervals (CI) for the logit-transformed proportion of steps keyed by wood, i.e., [logit(post) − logit(pre)].

Table 11-36. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the logit-transformed proportion of steps keyed by wood, i.e., (logit[TrtA(post)] − logit[TrtA(pre)]) − (logit[TrtB(post)] − logit[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	−0.25	0.19	−0.63	0.13
FP vs. REF	−0.04	0.21	−0.46	0.38
0% vs. REF	−0.12	0.19	−0.51	0.26
0% vs. FP	−0.08	0.22	−0.53	0.36
0% vs. 100%	0.13	0.20	−0.28	0.54
FP vs. 100%	0.21	0.22	−0.23	0.65

11-6.3.3.d. Diameter of key pieces in steps formed by wood

Mean diameter of key pieces for steps formed by wood ranged from 10 to 26 cm in the pre- and 10 to 38 cm in the post-harvest period (**Appendix Table 11-A-3**). The pre- to post-harvest change within-treatment ranged from 0 to 4 cm among treatments (**Table 11-37** and **Figure 11-19**). The P-value for the period × treatment contrast was 0.52, indicating that the amount of change in mean diameter of wood pieces keying steps did not depend on treatment. We estimated the difference in the change for each of the pairwise comparisons among the six combinations among references and treatments (**Table 11-38**).

Table 11-37. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the diameter (cm) of key pieces for steps formed by wood, i.e., [(post) – (pre)].

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	1	1	−2	4
100%	2	2	−2	5
FP	0	2	−4	4
0%	4	2	0	7

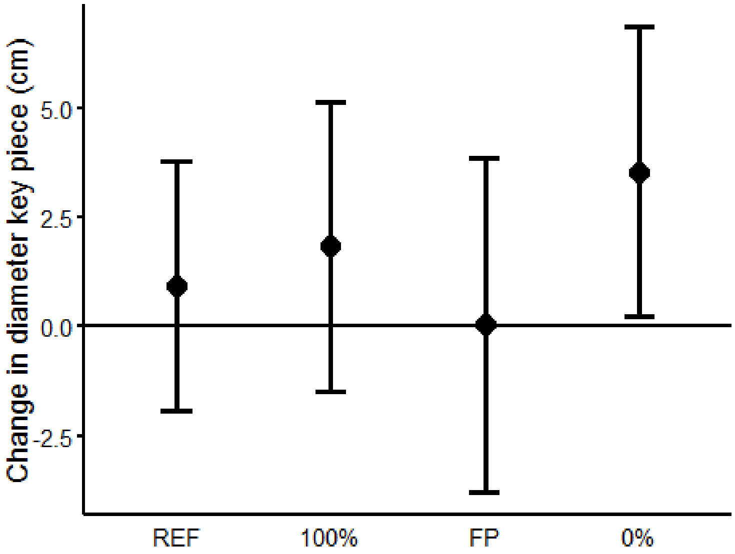


Figure 11-19. The within-treatment change and 95% confidence intervals (CI) for the diameter of key pieces for steps formed by wood, i.e., (post − pre).

Table 11-38. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence interval (CI) for the diameter of key pieces for steps formed by wood, i.e., [TrtA(post − pre) − TrtB(post − pre)]. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	95% CI	
			Lower	Upper
100% vs. REF	0.9	2.2	−3.5	5.3
FP vs. REF	−0.9	2.4	−5.7	3.9
0% vs. REF	2.6	2.2	−1.8	7.0
0% vs. FP	3.5	2.5	−1.6	8.6
0% vs. 100%	1.7	2.3	−3.0	6.4
FP vs. 100%	−1.8	2.5	−6.9	3.3

11-6.3.3.e. Channel rise attributed to steps

The proportion of channel rise attributed to steps ranged from 0.31 to 0.79 (i.e., 31% to 79%) in the pre- and 0.15 to 0.74 (i.e., 15% to 74%) in the post-harvest period (**Appendix Table 11-A-3**). We estimated the pre- to post-harvest change within-treatment for the logit-transformed channel rise (%) attributed to steps (**Table 11-39** and **Figure 11-20**). The P-value for the period × treatment contrast was <0.01, indicating that the amount of change in the channel rise attributed to steps depended on treatment. We estimated the change in the 0% treatment to be less than the change in the reference (P <0.01), 100% (P <0.01) and FP (P <0.01) treatments, respectively (**Table 11-40**).

Table 11-39. The within-treatment change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the logit-transformed channel rise (%) attributed to steps, i.e., [logit(post) − logit(pre)]. SE is the standard error.

Treatment	Change	SE	95% CI	
			Lower	Upper
REF	0.22	0.13	−0.04	0.49
100%	0.19	0.15	−0.12	0.50
FP	0.18	0.18	−0.17	0.54
0%	−0.46	0.15	−0.76	−0.15

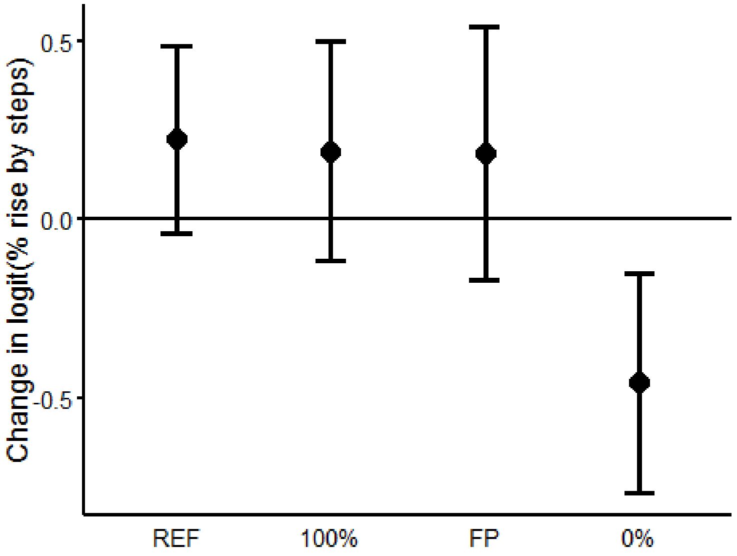


Figure 11-20. The within-treatment change and 95% confidence intervals (CI) for the logit-transformed channel rise (%) attributed to steps, i.e., [logit(post) − logit(pre)].

Table 11-40. The between-treatment difference in the change, standard error (SE) of the estimate, and 95% confidence intervals (CI) for the logit-transformed channel rise (%) attributed to steps, i.e., (logit[TrtA(post)] − logit[TrtA(pre)]) − (logit[TrtB(post)] − logit[TrtB(pre)]). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Difference	SE	P-value	95% CI	
				Lower	Upper
100% vs. REF	−0.03	0.20	0.87	−0.44	0.37
FP vs. REF	−0.04	0.22	0.85	−0.48	0.40
0% vs. REF	−0.68	0.20	<0.01	−1.08	−0.28
0% vs. FP	−0.64	0.23	<0.01	−1.11	−0.17
0% vs. 100%	−0.65	0.22	<0.01	−1.08	−0.22
FP vs. 100%	−0.01	0.23	0.97	−0.48	0.46

11-6.3.4. Channel Unit Distribution

In the pre-harvest period, riffles, pools and steps accounted for 68, 16, and 8% (based on step height) of channel length, respectively (**Figure 11-21**). We compared the distribution of channel units among treatments and noted very little change between the pre- and post-harvest periods (**Figure 11-22**). However, the distribution of channel units was based on total stream length surveyed.

Since we could not measure the length of units in reaches obstructed by instream wood, we were not able to include these reaches in our post-harvest comparisons. The length of stream obstructed by instream wood in the post-harvest period ranged from 0.0 to 44.1 m, or 0% to 17%, of the sampled stream length.

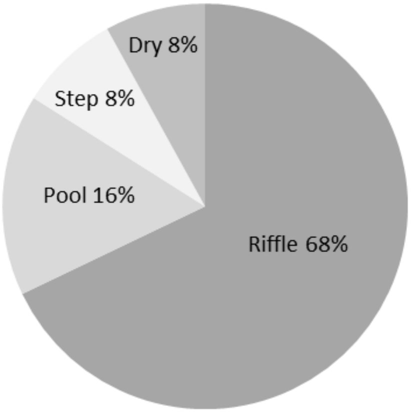
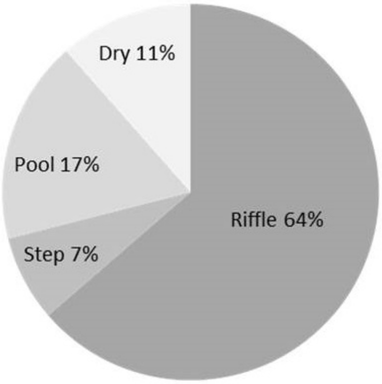
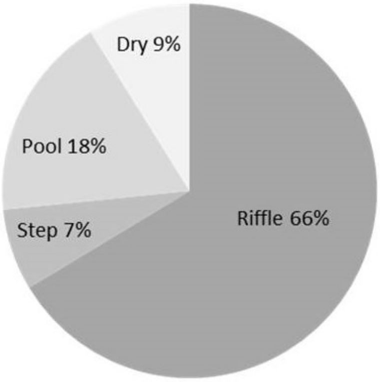


Figure 11-21. Cumulative distribution of channel units and dry stream reaches across all study sites in the pre-harvest period.

Treatment	Period	
	Pre-harvest	**Post-harvest**
REF		

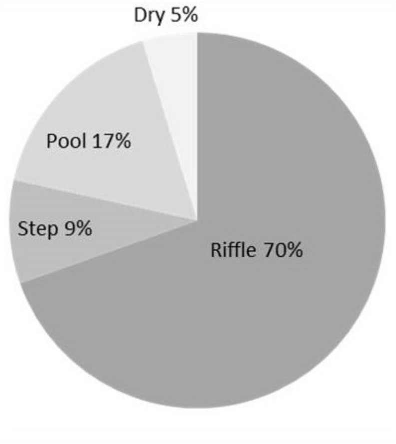
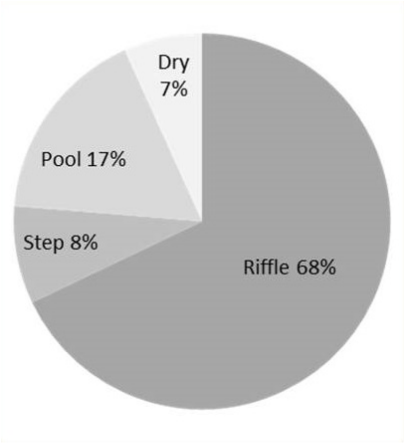
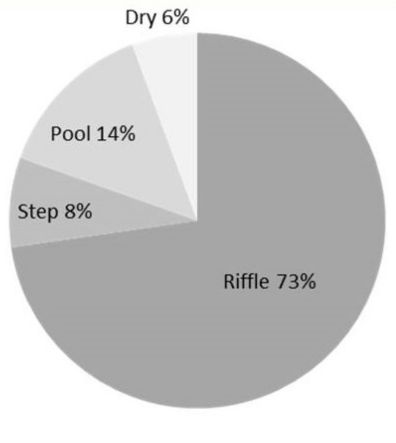
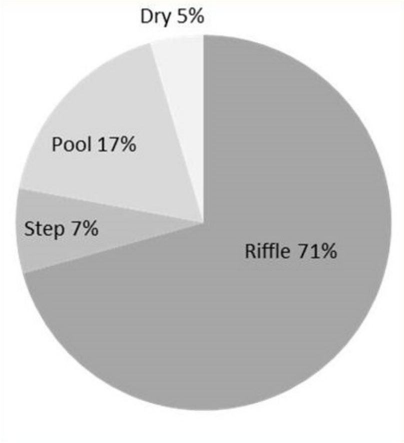
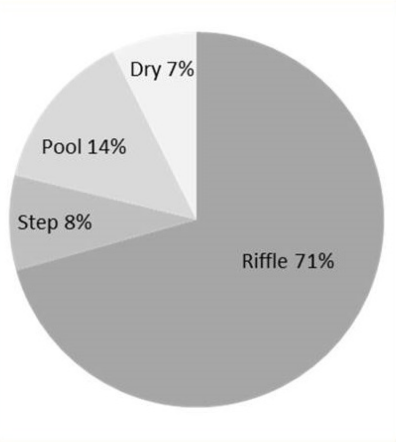
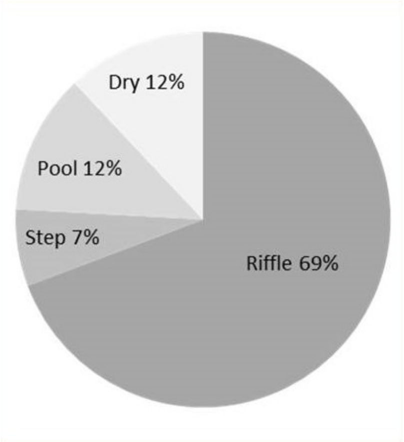
Treatment	**Period**	
	Pre-harvest	**Post-harvest**
100%		
FP		
0%		

Figure 11-22. Comparison of the distribution of channel units and dry stream reaches for all treatments in the pre- and post-harvest periods.

11-6.3.5. Dominant Units in Unobstructed and Obstructed Reaches

Though we could not evaluate total channel unit length in reaches obstructed by instream wood in the post-harvest period, we were able to categorize channel unit types in a subset of sites where we removed wood to evaluate wood loading and amphibian density (see Chapter 6 – *Wood Recruitment and Loading*). We compared the frequency of dominant channel unit types between stream reaches that were and were not obstructed by instream wood (**Figure 11-23**). Riffle was the dominant channel unit type in both unobstructed and obstructed stream reaches; however, we did observe an overall increase in other channel unit types (pool and step) in obstructed reaches.

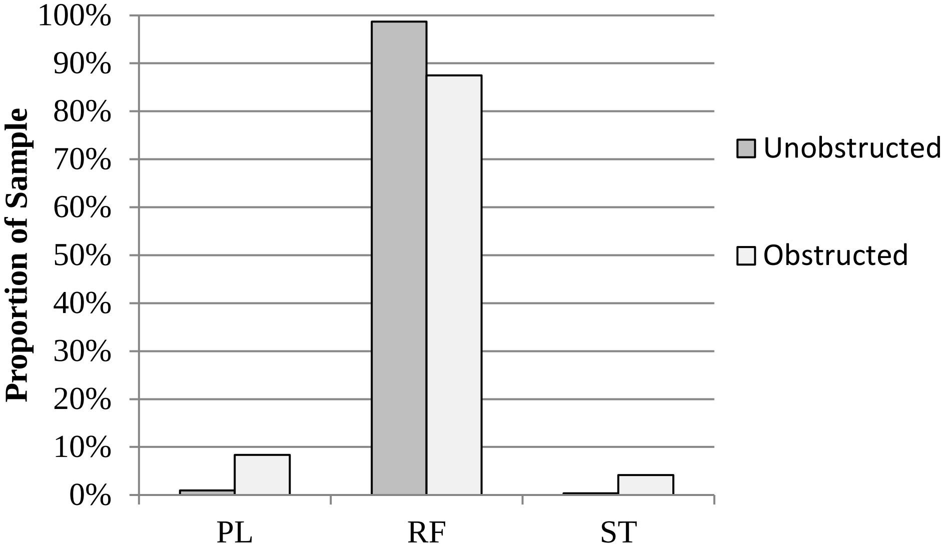


Figure 11-23. The proportion of sampled stream intervals having a pool (PL), riffle (RF), or step (ST) as the dominant channel unit by length for sample intervals in unobstructed reaches and plots in obstructed reaches in the post-harvest period (2009–2010).

11-6.4. SUMMARY OF RESULTS

Overall, we presented results for 17 statistical comparisons. The period × treatment contrast for four comparisons indicated a difference in response among treatments ($P \leq 0.1$; **Table 11-41**):

1) The 0% treatment differed from all other treatments, including the reference, in three instances: stream wetted width, bankfull width and step rise, and

2) All riparian buffer treatments differed from the reference for pool length.

Besides the channel characteristics that showed evidence of a treatment effect (i.e., $P \leq 0.1$), several variables consistently differed through time (i.e., pre- versus post-harvest). Of note, mean monthly precipitation during the sample period (May through July) was consistently greater in the post-harvest period, ranging from 2,286 to 4,316 mm*100 in the pre- and 7,135 to 12,450 mm*100 in the post-harvest period, an increase of 1,799 to 7,322 mm*100 across all study sites. (PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, accessed 21 Feb 2013; **Appendix Figure 11-A-1**).

Table 11-41. Summary results of pairwise contrasts for comparisons where the response (period × treatment contrast) had a P-value less than or equal to 0.1. Pairwise comparisons are between the shorter (listed horizontally) and longer (listed vertically) buffers, where ‘+’ indicates that the estimated value for the shorter buffer is more than that estimated for the longer buffer and ‘-’ indicates that the estimate for the shorter buffer is less than that for the longer buffer; ** indicates α ≤0.05 and * indicates α ≤0.1. Cells for comparisons already presented in the table are blocked out.

Response		100%	FP	0%	Response		100%	FP	0%
Stream wetted width P <0.001	REF			-**	Bankfull width P = 0.03	REF			-**
	100%			-**		100%			-**
	FP			-**		FP			-**
Pool length P = 0.03	REF	+**	+**	+**	Step rise (%) P <0.01	REF			-**
	100%					100%			-**
	FP					FP			-**

Our hypotheses for stream hydrology, channel and channel unit responses were consistent with our predictions in eight of 17 instances; though the pattern of response we observed for sand and fines and gravel and cobble matched our predictions, they were not statistically significant (**Table 11-42**).

Table 11-42. Predicted response (pre- to post-harvest change in buffer treatment sites compared to the reference; ↑ = increase, ↓ = decrease, − = no difference, NS = non-significant pattern that matched our prediction) for each hydrology, stream channel, and unit metric, observed response, and whether the observed response was expected based on our predicted response (Y = yes, N = no).

Category	Sub-category	Response Metric	Predicted	Observed Response	P-value	Expected?
Stream Hydrology		Wetted width	↑	↓ in 0% relative to the change in the other treatments	<0.001	N
		Depth	↑	−	>0.1	N
		Dry length	↓	−	>0.1	N
Stream channel	Bankfull width	Bankfull width	−	↓ in 0% relative to the change in the other treatments (P = 0.03)		N
	Substrate	Fines and sand	↑	NS	>0.1	Y
		Gravel and cobble	↓	NS	>0.1	Y
		Boulder and bedrock	−	−	>0.1	Y
Units	Pool	Length	↑	↑ in buffer treatments relative to reference	0.03	Y
		Max depth	−	−	>0.1	Y
		Density	−	−	>0.1	Y
	Riffle	Length	↓	−	>0.1	N
		Density	−	−	>0.1	Y
	Step	Height	↓	−	>0.1	N
		Density	↑	−	>0.1	N
		Proportion wood	↑	−	>0.1	N
		Key piece diameter	↓	−	>0.1	N
		Channel rise	↓	↓ in 0% relative to other treatments	0.04	Y

11-7. 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 input of wood (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 riparian forest harvest impacts (Richardson and Danehy 2007). Changes in 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 both short-term wood loading through harvest practices as well as the recruitment potential of wood by changing riparian forest stand conditions (Bilby and Ward 1991; Ralph *et al.* 1994; Jackson and Sturm 2002; Hassan *et al.* 2005b). 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, which affects both flow and sediment routing resulting from scour and sediment deposition (Jackson and Sturm 2002). Large wood has been the focus of most studies, but SW is more abundant than LW and plays an increasingly important role as channel size decreases (Bilby and Ward 1989; Gurnell *et al.* 2002; Maxa 2009). For example, in their evaluation of wood frequency and channel morphology in small, non-fish-bearing streams in the Coast Range of Washington State, Jackson and Sturm (2002) noted that SW, or that which they termed “organic debris” (i.e., wood pieces less than 10 cm diameter), created steps and affected channel morphology. Similarly, we found that SW as well as LW provided these important functions in our study streams (see Chapter 6 – *Wood Recruitment and Loading*).

Based on available literature, we developed a basic hypothesis about post-harvest changes linked to wood. Specifically, we hypothesized that changes to stream channel characteristics during the first two years after harvest implementation would largely reflect increased wood as a result of harvest, particularly in streams lacking riparian buffers (i.e., FP and 0% treatments). Our results for both LW and SW were consistent with this hypothesis. However, available literature had also led us to develop separate secondary hypotheses for each of LW and SW due to their divergent contributions to instream patterns, which we discuss later. We observed several significant post-harvest differences in channel characteristics among reference and riparian buffer treatment streams. In particular, percent channel rise attributed to steps, wetted width, and bankfull width all differed in the 0% treatment relative to the other treatments. Moreover, pool length in all three riparian buffer treatments differed from the reference. We attribute these collective differences to the increases in post-harvest wood inputs in the form of logging slash and windthrow from riparian buffers.

Our interest in stream channel characteristic response was also related to our desire to describe harvest effects on stream-associated amphibians. As such, we included both static measures and non-static measures of stream channel morphology that depend on current flow conditions. As a result, we were able to identify what appear to be year effects, or annual patterns, for some non-static variables. Year effects are expressed by a pre- to post-harvest change that is consistent across all treatments (i.e., including the reference). In particular, we observed a 1 (95% CI: 0 to 3) to 3 (95% CI: 1 to 4) cm post-harvest increase in stream depth across all treatments and the reference. We observed a similar pattern in pool maximum depth, which increased 1 (95% CI: 0 to 3) to 3 (95% CI: 0 to 5) cm in the post-harvest period. We believe these patterns reflect changes in flow associated with the increased annual precipitation documented across study sites during the post-harvest sample period. Due to their relatively small storage capacity and short flow paths, stream flows in headwaters respond more rapidly to rainfall than streams flows in larger basins (Gomi *et al.* 2002). In a thinning study, Olson and Rugger (2007) similarly observed parallel changes across treatments in instream conditions attributable to a year effect rather than thinning.

Changes in bankfull or wetted widths in response to harvest appear complex. Based on the findings of Trotter (1990) in a comparison of stream reaches with and without wood, we expected increased wood loading in clearcut streams would decrease stream velocity, while increasing stream wetted width and depth. However, the pre- to post-harvest change in the 0% treatment wetted width was actually 0.3 m less than the change in all other treatments (95% CI: 0.0 to 0.5 m less than FP treatment and 0.1 to 0.5 m less than reference and 100% treatment). Our findings are consistent with Carlson and colleagues (1990), who found that LW in logged streams in northeastern Oregon restricted wetted stream width during summer. Further, Ralph and colleagues (1994) found that timber harvest resulted in a concentration of LW towards the channel margins. Still others found no difference in wetted stream widths between logged and uncut stands (Corn and Bury 1989; O'Connell *et al.* 2000). Our stream depth findings are similar to O’Connell and colleagues (2000) who detected no differences in water depth between control and clearcut sites in the first two years following timber harvest with variable width riparian buffers.

Similar to the pattern we observed for stream wetted width, the pre- to post-harvest change in bankfull width in the 0% treatment was also less than the changes estimated for the other treatments. The between-treatment difference in the change in the 0% treatment was 0.4 m less than the change in the reference (95% CI: 0.2 to 0.6 m) and 0.5 m less than the FP and 100% treatments (95% CI: 0.2 to 0.7 m and 0.3 to 0.8 m for the FP and 100% treatments, respectively). In contrast, Jackson and Sturm (2002) found that bankfull channel width increased with increasing LW frequency. Our results also differed from those observed by O’Connell and colleagues (2000), who observed no difference in bankfull width between control and clearcut sites in the first two years following timber harvest. This latter study involved somewhat larger streams, at least in part, which may explain the differential patterns. Though study context is important in interpreting the differences among these studies, we emphasize that bankfull width is innately difficult to measure accurately (Platts *et al.* 1983).

We did not see changes in the extent of wetted or dry channel by treatment, though other studies have shown that forest management can alter the magnitude and timing of flow in headwater streams as a result of forest canopy removal and reductions in interception and evapotranspiration (Lewis *et al.* 2001; Keim and Skaugset 2003; Johnson *et al.* 2007). This pattern is inconsistent among studies. Jackson and Sturm (2002) found that large wood (>40 cm diameter) can retain so much stream sediment that surface water goes entirely subsurface, decreasing the extent of wetted channel. In our evaluation of discharge at some study sites (see Chapter 8 – *Discharge*), we observed increased annual runoff in harvested sites in the post-harvest period, but the magnitude of change varied by season. While baseflows decreased in the 100% treatment, were largely unchanged in the FP treatment, and increased in the 0% treatment, changes in annual peak flows generally did not differ by treatment. Ultimately, changes in hydrograph were not expressed as changes in the extent of wetted or dry channel during our sample period.

Numerous studies have noted an increase in fine sediment in previously harvested headwater streams (Corn and Bury 1989; Dupuis and Steventon 1999), and some have associated this with increased hydraulic roughness related to conifer needles, twigs and branches that trap finer sediments (Montgomery *et al.* 1996; Bilby and Bisson 1998; Jackson *et al.* 2001). This finding has been especially prevelant in smaller streams (Bilby and Ward 1989). Though our results for stream substrate were not statistically significant at an alpha level of 0.1 , we estimated that the mean within-treatment change in the proportion of the stream dominated by fines and sand was 4, 9, and 16% greater in the 100%, FP and 0% treatments than the reference, respectively. Hence, substantial variability may conceal a gradient-level change where effect size in the response was only modest. However, in our own results for suspended sediment export, we found little direct evidence of increased sediment concentrations resulting from harvest and we did not observe any obvious sources of increased sediment input from roads, windthrow or bank erosion (see Chapter 10 – *Sediment Processes*). Jackson and colleagues (2001) observed a 32% increase in fine sediment for streams that had recently undergone clearcut harvest to the stream edge (equivalent to our 0% treatment). They attributed this finding to the entrapment of fine sediment by instream logging debris; however, they did not observe a significant increase in fine sediment in clearcut streams that retained a full riparian buffer (similar to our 100% treatment). Others have found that previously clearcut headwater streams contained large accumulations of logging debris and that fine sediment volume was correlated to number of LW pieces (Lisle 1986; Gomi *et al.* 2001; Jackson and Sturm 2002). Elsewhere, presence of LW has been shown to facilitate deposition of sediment (Bilby and Bisson 1998) and create areas of low energy that slow the transport of sediment and organic material (Heede 1972; Bilby and Ward 1989).

Within-treatment change in mean pool length was 12% (95% CI: 0 to 26%) to 20% (95% CI: 6 to 36%) greater in harvested sites than in the reference. However, we found no difference in the pre- to post-harvest change in pool depth among treatments. Individual pieces of LW and debris dams dominate pool formation (Lisle 1986; Montgomery *et al.* 1995), and Lisle (1986) suggested that instream LW retards the rate of water discharge in small streams by greatly increasing channel roughness, which increases pool length by extending the upstream limit of pools. Lisle (1986) also found that while pools were longer, no difference existed in residual pool depth between forested and clearcut streams. While we do not know how pool depth or length may be correlated with pool volume, some investigators have compared pool volume between logged and undisturbed streams. Carlson and colleagues (1990) observed no difference in pool volume between paired logged and undisturbed streams, however, they did find that pool volume was greater when more instream wood was present. Likewise, Bilby (1984) found that pool area and volume decreased when logging debris was removed from small streams in western Washington. Similarly, Lisle (1995) found that average pool depth decreased following experimental removal of wood from several stream reaches in the area impacted by the 1980 eruption of Mt. St. Helens. Again, context differences among these studies may also be important in explaining differences in results.

We did not observe a response to riparian buffer treatments for any of our channel unit density measures. In particular, our results differ from those of Montgomery and colleagues (1995), who found that streams previously clearcut to the stream edge had fewer pools than similar reaches in old growth forests. Interestingly, and inverse to our own findings, the wood loading (pieces >10 cm diameter) observed by Montgomery and colleagues (1995) in the same study was less in clearcut streams. Jackson and Sturm (2002) noted that relatively small wood (10–40 cm diameter) and organic debris (<10 cm diameter) were major step-forming elements in small forested streams, results consistent with our own (see Chapter 6 – *Wood Recruitment and Loading*). Our findings for step density are also consistent with those of Jackson and colleagues (2001), who found the same amount of steps in buffered and unharvested reference streams. However, wood pieces have been shown to increase the frequency of steps in other studies (Bilby and Ward 1989; Gomi *et al.* 2002).

We observed no differences in the change of step height, the proportion of steps formed by wood, or the size of wood pieces contributing to formation of steps. We based our predictions for these metrics on the idea that the proportion of SW would increase in riparian buffer treatment sites post-harvest. However, while we did observe a pre- to post- harvest increase in wood loading that differed among treatments, the proportion of post-harvest SW and LW did not differ (see Chapter 6 – *Wood Recruitment and Loading*). Thus, it should not be surprising that we saw no response in these step-related metrics. However, we did see a difference in the pre- to post-harvest change in the proportion of the channel rise attributed to steps, which was 16% to 17% less in the 0% treatment than the other treatments. This was the opposite of our prediction. We hypothesized that an increase in wood loading in streams would increase the number of steps and subsequently that the proportion of the channel rise attributed to steps would increase. However, fluvial power in these small debris-driven streams was insufficient to sort the wood and form distinctive steps, at least over the two years immediately post-harvest. Rather than forming distinct steps, wood tended to accumulate in dense matrices of branches, twigs, conifer needles and fine sediment, essentially covering or filling the stream channel in some reaches. As a result, this matrix of organic and inorganic materials covered some steps in our streams, especially where a riparian buffer was lacking, and the percent channel rise attributed to steps decreased in the 0% treatment. This finding is consistent with those of Jackson and colleagues (2001) who observed massive amounts of organic debris input in clearcut streams lacking a riparian buffer in their evaluation of timber harvest impacts along first- and second-order streams in western Washington.

Our results support our hypothesis that instream increases in wood due to clearcut harvest alter stream channel characteristics. However, the implications for the impacts of wood on channel morphology at our study sites over the longer term are not clear. Wood entering headwater stream channels is frequently suspended over the channel (Hassan *et al.* 2005b). We found that up to 45% of SW and 37% of LW in the post-harvest period was completely suspended above the stream channel and did not contribute to in-channel hydraulics, versus 17% and 16% in the pre-harvest period, respectively (see Chapter 6 – *Wood Recruitment and Loading*). It is not certain how long newly recruited SW will persist or what roles that LW pieces suspended over the channel may have in the future.

Headwater streams are understudied given their relative frequency in the forest, in part because of their typical lack of fish (Richardson and Danehy 2007). Headwater streams differ widely in physical, chemical and biotic attributes, providing habitats for a range of unique species (Meyer *et al.* 2007; Richardson and Danehy 2007). Forest practices have a suite of characteristic influences on headwater systems; however, the particular mechanisms that are responsible for changes to stream communities as a result of forestry activities are challenging to identify (Richardson and Danehy 2007). We evaluated the short-term impacts of clearcut harvest, including a range of alternative riparian buffer treatments, on wood recruitment and loading and subsequent effects on instream channel morphology. We found that the 0% treatment, which lacked a riparian buffer throughout the entirety of the RMZ, differed most frequently from reference conditions. In fact, the other riparian buffer treatments (100% and FP treatments) only differed from the reference for pool length. These results will make an important contribution to informing the effectiveness of the current Forest Practices patch buffer prescription for timber harvest along non-fish-bearing streams in Washington State. In that context, the observed changes in channel characteristics also will help us to interpret the impacts of these same treatments on stream-associated biota, in particular amphibians.

11-8. REFERENCES

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APPENDIX 11-A. DATA TABLES

Appendix Table 11-A-1. The target stream length, length of obstructed (post-harvest years only) and dry stream, and resulting sample length (Sample Len = Target (m) – [Obstructed + Dry]) for each site (block and treatment combination) and year. NA indicates that we did not sample a site in a particular year.

Block	Treatment	Year	Target (m)	Obstructed (m)	Dry (m)	Sample Length (m)
OLYM	REF	2007	501	-	7	494
		2008	501	-	11	490
		2009	481	2	35	444
		2010	491	19	20	452
	100%	2006	451	-	121	329
		2007	433	-	74	359
		2008	384	-	30	354
		2009	418	9	55	355
		2010	423	3	41	380
	FP	2006	217	-	15	202
		2007	217	-	24	193
		2008	217	-	10	207
		2009	217	0	26	191
		2010	217	16	23	178
	0%	2006	243	-	84	158
		2007	244	-	113	131
		2008	244	-	30	214
		2009	254	34	153	68
		2010	254	43	31	180
WIL1	REF	2006	343	-	30	313
		2007	353	-	56	297
		2008	344	-	61	283
		2009	333	6	55	272
		2010	334	15	51	268
	100%	2006	387	-	0	387
		2007	325	-	0	325
		2008	331	-	0	331
		2009	396	27	4	365
		2010	394	11	20	363
	FP	2006	290	-	4	286
		2007	274	-	0	274
		2008	264	-	1	263
		2009	274	44	4	226
		2010	274	19	8	247

Appendix Table 11-A-1. (continued)

Block	Treatment	Year	Target (m)	Obstructed (m)	Dry (m)	Sample Length (m)
WIL1	0%	2006	327	-	7	320
		2007	197	-	0	197
		2008	264	-	3	261
		2009	304	16	12	276
		2010	274	32	23	219
WIL2	REF1	2006	442	-	82	360
		2007	401	-	49	352
		2008	418	-	37	381
		2010	454	4	16	434
	REF2	2006	278	-	6	272
		2007	280	-	15	265
		2008	250	-	17	233
		2009	280	0	11	269
		2010	270	6	15	249
	100%	2006	377	-	23	354
		2007	398	-	37	362
		2008	378	-	11	367
		2009	398	9	43	346
		2010	388	1	111	276
	0%	2006	350	-	15	335
		2007	334	-	0	334
		2008	328	-	14	314
		2009	344	42	36	266
		2010	274	25	4	245
WIL3	REF	2006	513	-	189	324
		2007	487	-	175	312
		2008	412	-	56	356
		2009	402	0	84	318
		2010	462	0	109	353
	100%	2006	420	-	2	418
		2007	414	-	1	413
		2008	394	-	1	393
		2009	320	0	52	268
		2010	384	1	19	364
CASC	REF	2006	407	-	83	324
		2007	454	-	135	319
		2008	384	-	17	367
		2009	444	5	146	293
		2010	394	0	35	359

Appendix Table 11-A-1. (continued)

Block	Treatment	Year	Target (m)	Obstructed (m)	Dry (m)	Sample Length (m)
CASC	FP	2006	346	-	53	293
		2007	353	-	53	301
		2008	333	-	13	321
		2009	353	0	32	321
		2010	353	4	9	340
	0%	2006	327	-	28	299
		2007	334	-	23	311
		2008	334	-	30	304
		2009	334	8	48	278
		2010	334	26	61	248

Appendix Table 11-A-2. Mean stream wetted width (Str Wid), bankfull width (BF Wid), stream depth (Depth), and the proportion of the stream dominated by fines and sand (F/SA), gravel or cobble (GR/CO) and boulder or bedrock (BO/BED) by site (block and treatment combination) and year.

Block	Treatment	Year	Str Wid	BF Wid	Depth	F/SA	GR/CO	BO/BED
OLYM	REF	2007	1.3	3.0	11	0.08	0.60	0.31
		2008	1.8	2.2	14	0.10	0.58	0.32
		2009	2.0	2.6	13	0.02	0.59	0.39
		2010	2.1	2.4	11	0.15	0.56	0.29
	100%	2006	0.8	1.8	3	0.21	0.67	0.12
		2007	0.9	2.6	4	0.18	0.70	0.11
		2008	1.3	1.7	7	0.14	0.80	0.07
		2009	1.3	2.1	7	0.09	0.82	0.09
		2010	2.0	2.2	10	0.31	0.57	0.12
	FP	2006	0.7	0.8	4	0.05	0.75	0.20
		2007	0.6	1.1	4	0.39	0.56	0.06
		2008	0.7	1.0	4	0.38	0.57	0.05
		2009	0.9	1.4	6	0.05	0.76	0.19
		2010	0.9	1.0	5	0.38	0.57	0.05
	0%	2006	0.7	1.7	3	0.18	0.77	0.05
		2007	0.4	2.0	3	0.16	0.76	0.08
		2008	0.8	1.0	7	0.21	0.67	0.13
		2009	0.4	1.2	3	0.32	0.64	0.05
		2010	1.0	1.4	6	0.41	0.55	0.05

Appendix 11-A-2. (continued)

Block	Treatment	Year	Str Wid	BF Wid	Depth	F/SA	GR/CO	BO/BED
WIL1	REF	2006	0.7	1.1	5	0.24	0.74	0.03
		2007	0.8	1.6	5	0.35	0.65	0.00
		2008	1.0	1.3	7	0.86	0.14	0.00
		2009	1.0	1.5	5	0.32	0.61	0.06
		2010	1.1	1.4	6	0.56	0.35	0.09
	100%	2006	1.6	2.1	6	0.05	0.95	0.00
		2007	1.6	1.9	5	0.14	0.80	0.06
		2008	1.6	1.8	6	0.32	0.65	0.03
		2009	1.9	2.4	11	0.21	0.76	0.03
		2010	1.8	2.1	8	0.30	0.70	0.00
	FP	2006	0.9	1.4	4	0.00	1.00	0.00
		2007	1.0	1.4	5	0.00	0.92	0.08
		2008	0.9	1.2	5	0.23	0.69	0.08
		2009	1.3	1.6	8	0.24	0.64	0.12
		2010	1.5	1.8	8	0.27	0.73	0.00
	0%	2006	1.1	1.5	5	0.28	0.66	0.07
		2007	1.1	2.4	7	0.10	0.81	0.10
		2008	1.4	1.7	7	0.28	0.66	0.07
		2009	1.4	1.8	10	0.39	0.43	0.18
		2010	1.3	1.5	9	0.59	0.34	0.07
WIL2	REF1	2006	1.0	1.8	3	0.13	0.80	0.07
		2007	1.1	2.1	3	0.07	0.73	0.20
		2008	1.3	1.7	6	0.11	0.80	0.09
		2010	1.8	2.1	8	0.13	0.79	0.09
	REF2	2006	0.9	1.1	4	0.46	0.54	0.00
		2007	0.9	1.3	6	0.48	0.52	0.00
		2008	1.0	1.1	6	0.67	0.29	0.04
		2009	1.3	1.6	7	0.57	0.43	0.00
		2010	1.3	1.4	5	0.70	0.30	0.00
	100%	2006	1.2	1.8	5	0.17	0.63	0.20
		2007	1.6	2.1	5	0.51	0.38	0.11
		2008	1.2	1.5	5	0.33	0.54	0.13
		2009	1.6	2.2	10	0.28	0.49	0.23
		2010	1.7	2.1	7	0.29	0.58	0.13

Appendix Table 11-A-2. (continued)

Block	Treatment	Year	Str Wid	BF Wid	Depth	F/SA	GR/CO	BO/BED
WIL2	0%	2006	1.5	2.2	4	0.17	0.63	0.20
		2007	2.1	3.1	6	0.18	0.58	0.24
		2008	1.7	1.9	5	0.27	0.64	0.09
		2009	1.7	2.1	7	0.30	0.67	0.03
		2010	1.7	1.9	5	0.32	0.64	0.04
WIL3	REF	2006	0.7	1.5	3	0.27	0.67	0.07
		2007	1.0	2.2	4	0.51	0.35	0.14
		2008	1.0	1.6	4	0.55	0.37	0.08
		2009	1.0	1.7	6	0.29	0.57	0.14
		2010	1.4	1.9	6	0.49	0.38	0.13
	100%	2006	1.4	2.0	5	0.26	0.47	0.26
		2007	1.3	2.5	5	0.05	0.74	0.21
		2008	1.5	1.9	5	0.22	0.59	0.20
		2009	1.4	2.1	7	0.24	0.48	0.27
		2010	2.0	2.3	6	0.19	0.56	0.25
CASC	REF	2006	0.8	1.8	5	0.11	0.70	0.20
		2007	1.1	2.4	6	0.30	0.60	0.10
		2008	1.5	1.9	8	0.16	0.70	0.14
		2009	1.2	1.9	9	0.23	0.65	0.13
		2010	1.8	1.9	9	0.18	0.66	0.16
	FP	2006	0.7	1.2	3	0.26	0.74	0.00
		2007	0.7	1.9	4	0.09	0.86	0.06
		2008	1.3	1.5	9	0.06	0.94	0.00
		2009	1.2	1.5	8	0.26	0.71	0.03
		2010	1.4	1.4	10	0.12	0.85	0.03
	0%	2006	1.2	2.0	3	0.16	0.81	0.03
		2007	0.8	1.9	5	0.00	0.91	0.09
		2008	1.1	1.3	8	0.16	0.81	0.03
		2009	1.0	1.5	6	0.31	0.59	0.09
		2010	1.3	1.4	7	0.10	0.84	0.06

Appendix Table 11-A-3. Mean length (Len) and number per meter (#/m) for pools and riffles. average maximum depth (Dep) for pools, and the height (Ht), number per meter (#/m), proportion of steps keyed by wood (WD), average diameter of wood keys (Avg Dia), and the percent of the channel rise that can be attributed to steps (% Rise), by site (block and treatment combination) and year.

			Pool			Riffle		Step				
Block	**Treatment**	**Year**	**Len (m)**	**Dep (cm)**	**#/m**	**Len (m)**	**#/m**	**Ht (cm)**	**#/m**	**WD**	**Avg Dia**	**% Rise**
OLYM	REF	2007	1.7	25	0.19	2.1	0.44	39	0.2	0.81	23	42
		2008	1.5	28	0.19	2.1	0.55	43	0.26	0.73	22	52
		2009	1.5	26	0.23	2.8	0.43	49	0.25	0.81	38	47
		2010	1.3	25	0.29	2.2	0.60	39	0.3	0.73	20	46
	100%	2006	0.9	12	0.4	1.4	0.66	37	0.47	0.55	12	51
		2007	1.3	16	0.27	1.5	0.67	35	0.43	0.66	17	55
		2008	1.1	21	0.2	1.8	0.67	39	0.43	0.46	16	66
		2009	1	17	0.25	2.0	0.69	41	0.38	0.57	23	51
		2010	1	21	0.19	1.9	0.75	38	0.41	0.68	17	59
	FP	2006	0.7	11	0.27	1.5	0.61	22	0.52	0.76	10	41
		2007	0.7	12	0.19	1.7	0.59	27	0.32	0.89	10	30
		2008	0.7	11	0.13	1.6	0.69	26	0.34	0.8	12	33
		2009	0.8	15	0.33	1.6	0.67	32	0.52	0.77	13	62
		2010	0.8	15	0.21	1.6	0.71	32	0.44	0.81	13	59
	0%	2006	1	14	0.35	1.4	0.53	35	0.49	0.52	20	45
		2007	0.9	14	0.33	1.3	0.65	35	0.34	0.76	23	36
		2008	0.9	15	0.2	1.3	0.77	39	0.36	0.58	22	50
		2009	1.1	14	0.25	1.9	0.38	39	0.31	0.43	33	13
		2010	0.9	16	0.19	1.7	0.67	39	0.37	0.52	29	38
WIL1	REF	2006	1	15	0.19	2.6	0.35	28	0.3	0.8	15	45
		2007	0.9	12	0.31	2.4	0.39	27	0.34	0.83	14	47
		2008	1.1	15	0.11	2.7	0.36	31	0.23	0.75	18	40
		2009	0.8	17	0.28	1.8	0.57	33	0.31	0.8	15	58
		2010	0.9	16	0.25	2.1	0.51	32	0.26	0.83	12	56

Appendix Table 11-A-3. (continued)

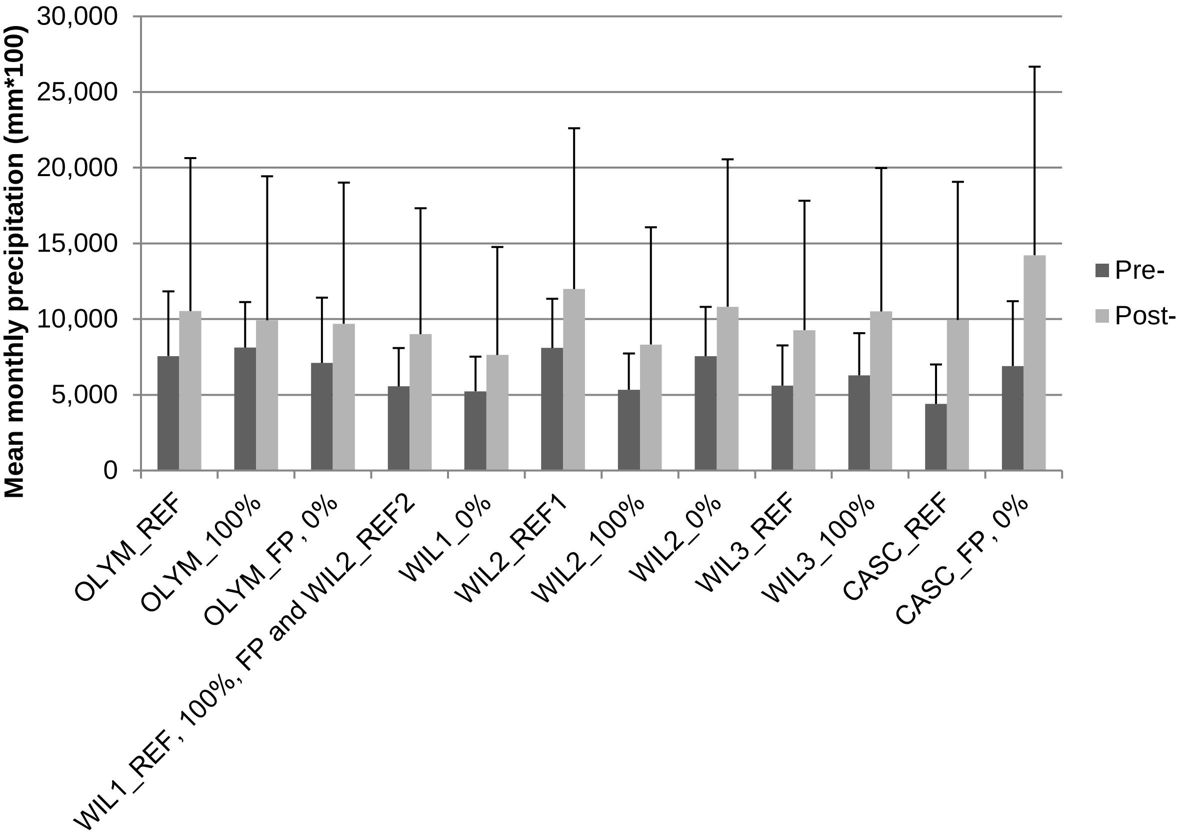
			Pool			Riffle		Step				
Block	**Treatment**	**Year**	**Len (m)**	**Dep (cm)**	**#/m**	**Len (m)**	**#/m**	**Ht (cm)**	**#/m**	**WD**	**Avg Dia**	**% Rise**
WIL1	100%	2006	1.1	19	0.13	3.0	0.31	38	0.29	0.64	20	58
		2007	0.9	14	0.27	2.3	0.51	34	0.41	0.78	17	62
		2008	1.1	17	0.12	2.7	0.36	32	0.33	0.63	16	53
		2009	1.1	18	0.37	1.8	0.54	34	0.41	0.66	15	59
		2010	1	20	0.25	2.3	0.51	34	0.38	0.75	15	69
	FP	2006	0.9	17	0.3	1.9	0.43	30	0.37	0.58	15	58
		2007	0.8	15	0.28	1.9	0.45	28	0.39	0.82	16	55
		2008	0.9	13	0.18	2.7	0.38	28	0.39	0.66	16	56
		2009	1.1	16	0.32	2.0	0.45	31	0.3	0.7	16	49
		2010	1	17	0.24	1.7	0.47	31	0.28	0.79	13	52
	0%	2006	1	16	0.2	2.5	0.34	28	0.23	0.81	12	45
		2007	1	15	0.23	2.5	0.44	27	0.31	0.74	15	60
		2008	1	17	0.15	3.6	0.26	32	0.24	0.85	15	52
		2009	1.2	20	0.14	2.7	0.38	34	0.22	0.84	12	49
		2010	1	18	0.19	2.8	0.45	31	0.27	0.92	13	60
WIL2	REF1	2006	0.9	14	0.37	1.5	0.75	40	0.66	0.84	19	59
		2007	1	15	0.33	2.4	0.40	46	0.35	0.88	26	45
		2008	0.9	16	0.27	1.7	0.66	46	0.33	0.78	26	46
		2010	1	20	0.24	2.5	0.50	53	0.39	0.86	22	64
	REF2	2006	0.8	12	0.17	1.7	0.58	23	0.45	0.66	10	64
		2007	0.8	12	0.25	2.2	0.45	27	0.47	0.58	21	70
		2008	0.8	13	0.16	2.0	0.48	26	0.43	0.57	12	63
		2009	0.7	11	0.39	1.8	0.55	26	0.51	0.7	14	66
		2010	0.9	16	0.13	2.7	0.38	29	0.32	0.72	13	52

Appendix Table 11-A-3. (continued)

			Pool			Riffle		Step				
Block	**Treatment**	**Year**	**Len (m)**	**Dep (cm)**	**#/m**	**Len (m)**	**#/m**	**Ht (cm)**	**#/m**	**WD**	**Avg Dia**	**% Rise**
WIL2	100%	2006	1	19	0.15	2.2	0.45	38	0.31	0.82	17	54
		2007	1	17	0.32	1.9	0.53	38	0.45	0.91	20	70
		2008	0.9	17	0.22	2.1	0.51	43	0.36	0.84	21	67
		2009	1.3	22	0.38	1.9	0.58	48	0.45	0.75	21	75
		2010	1.2	19	0.35	1.9	0.82	47	0.45	0.78	22	64
	0%	2006	0.9	13	0.21	1.8	0.66	25	0.51	0.59	12	59
		2007	1	13	0.23	2.5	0.52	28	0.43	0.69	16	63
		2008	1	15	0.12	2.4	0.53	28	0.45	0.47	14	60
		2009	1.1	16	0.18	1.6	0.76	28	0.47	0.57	15	54
		2010	1	15	0.21	1.9	0.67	28	0.44	0.79	11	69
WIL3	REF	2006	1	12	0.29	1.6	0.72	30	0.51	0.38	15	43
		2007	1	14	0.25	1.6	0.58	33	0.32	0.53	13	31
		2008	1.1	17	0.15	2.2	0.45	36	0.22	0.45	17	37
		2009	0.9	20	0.2	1.7	0.81	36	0.36	0.37	27	44
		2010	1	16	0.21	1.9	0.65	35	0.32	0.59	16	42
	100%	2006	1.1	15	0.22	1.7	0.56	27	0.38	0.41	10	48
		2007	1.1	15	0.3	2.0	0.45	27	0.31	0.49	13	47
		2008	1.2	16	0.19	2.2	0.42	35	0.25	0.37	13	41
		2009	1.2	17	0.29	1.7	0.80	44	0.34	0.37	20	58
		2010	1	16	0.27	2.1	0.68	30	0.44	0.4	10	58
CASC	REF	2006	1.5	22	0.29	1.7	0.40	33	0.23	0.25	22	46
		2007	1.5	21	0.3	2.3	0.37	40	0.22	0.41	24	45
		2008	1.3	24	0.27	1.8	0.59	40	0.25	0.48	24	60
		2009	1.1	18	0.57	1.5	0.77	31	0.38	0.35	19	55
		2010	1.3	21	0.39	1.8	0.53	33	0.33	0.5	15	66

Appendix 11-A-3. (continued)

			Pool			Riffle		Step				
Block	**Treatment**	**Year**	**Len (m)**	**Dep (cm)**	**#/m**	**Len (m)**	**#/m**	**Ht (cm)**	**#/m**	**WD**	**Avg Dia**	**% Rise**
CASC	FP	2006	0.9	12	0.14	2.1	0.46	31	0.21	0.29	17	37
		2007	1.1	13	0.23	1.8	0.49	23	0.34	0.32	18	47
		2008	1.1	20	0.15	1.8	0.58	28	0.49	0.4	12	79
		2009	0.9	16	0.28	1.7	0.64	27	0.32	0.4	19	52
		2010	1	19	0.23	1.6	0.70	28	0.35	0.43	10	58
	0%	2006	0.9	11	0.17	1.5	0.66	39	0.34	0.29	22	51
		2007	1	13	0.29	1.3	0.69	29	0.52	0.42	21	58
		2008	0.9	17	0.25	1.2	0.91	34	0.44	0.4	15	56
		2009	0.8	15	0.29	1.5	0.64	33	0.34	0.33	26	43
		2010	0.8	17	0.3	1.2	1.33	31	0.42	0.32	26	44



Appendix Figure 11-A-1. Mean monthly precipitation (mm*100) for the stream characterization period (May–July) for the pre- (2006–2008) and post- (2009–2010) harvest periods for study sites or groups of study sites located within 4 km of each other (PRISM Climate Group). PRISM (PRISM Climate Group, Oregon State University, http://prism.oregonstate.edu, accessed 21 Feb 2013) is an analytical model that uses point data and an underlying grid such as a digital elevation model (DEM) or a 30 year climatological average (e.g., 1981–2010 average) to generate gridded estimates of monthly and annual precipitation and temperature (as well as other climatic parameters). PRISM is well suited to regions with mountainous terrain, because it incorporates a conceptual framework that addresses the spatial scale and pattern of orographic processes. Grids were modeled on a monthly basis. Annual grids of temperature are produced by averaging the monthly grids, and summing for precipitation. The PRISM estimates of precipitation and temperature are calculated at a spatial resolution of approximately 4 km, and estimates cannot be derived from these data at a spatial resolution less than 2 km (PRISM Climate Group). The estimates in this file are based on bilinear interpolation from the centroid of a basin, or the centroid of a group of basins that are in close proximity to one another.

CHAPTER 12 - LITTERFALL INPUT AND DETRITUS EXPORT

Stephanie Estrella, William Ehinger, and Charlotte Milling

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12-1. ABSTRACT

Forested headwater streams, such as those characteristic of Washington State, depend on organic matter inputs originating from outside the stream channel for their primary source of energy. Timber harvest in non-fish-bearing stream basins may alter the quantity, composition, and timing of these litterfall inputs, which may affect the quantity and composition of instream detritus and thus food availability to the aquatic biotic community. We assessed the response of litterfall input and instream detritus export from non-fish-bearing streams pre- and post-timber harvest from study sites treated with the current Washington State Forest Practices riparian buffer (FP treatment), a more extensive buffer (100% treatment), and no buffer (0% treatment) relative to unharvested reference sites. Litterfall input was sampled continuously at each study site. Samples were collected every six weeks and quantified in grams ash-free dry mass (AFDM) per square meter per day. Detritus in transport was collected roughly every six weeks, and export quantified in grams AFDM per day. We measured a treatment x period interaction for deciduous litterfall input, and total, wood, miscellaneous, and coarse and fine particulate detritus export. Input of total, conifer, and wood litterfall input decreased in the FP and 0% treatments post-harvest, while deciduous and miscellaneous litterfall input decreased in the 0% treatment ($P < 0.05$). Export of total, wood, miscellaneous, and coarse and fine particulate detritus increased in the 100% treatment, but decreased in the 0% treatment ($P < 0.05$). Conifer and deciduous detritus export also decreased in the 0% treatment ($P < 0.05$). An increase in slash in sites receiving the buffer treatments may have enhanced retention of detritus within the stream channel. Litterfall input and detritus export, however, followed a comparable trend in that both increased in the 100% treatment and decreased in the FP and 0% treatments post-harvest, indicating that the change in input post-harvest is the primary factor controlling export.

12-2. INTRODUCTION

Allochthonous organic matter inputs, or inputs originating outside the stream channel, represent the primary energy source in small forested stream ecosystems (Cummins *et al*. 1983; Gregory *et al*. 1991; Bilby and Bisson 1992). In the Pacific Northwest, these inputs may include conifer needles, leaves from deciduous trees and shrubs, wood, and other material such as flowers, cones, and seeds. Litterfall input is variable and depends on a number of factors, including climate, canopy composition, and stand age (Abelho 2001). Litterfall input volume is inversely related to stream order, with the highest contributions in first-order streams and a decreasing contribution with an increase in stream order (Conners and Naiman 1984).

Litterfall inputs enter the stream channel vertically by falling directly from the overhead canopy or laterally by blowing or washing in from litter stocks previously deposited on the forest floor and stream bank (Benfield 1997). The relative importance of these pathways is frequently variable and depends on wind patterns, aspect, bank slope, and other site specific characteristics (Rhoades and Binkley 1992; Benfield 1997; Hart *et al.* 2013). Once in the stream channel, litterfall can be transported downstream or retained in the substrate, in wood jams, or in slow-water habitats such as pools or channel margins (Anderson *et al.* 1978; Bilby and Likens 1980; Bilby 1981; Webster *et al.* 1999). There, litterfall inputs will undergo breakdown through leaching of soluble nutrients, microbial colonization and degradation, and fragmentation through physical abrasion and invertebrate shredding (Webster *et al.* 1999; Abelho 2001). Litterfall inputs that enter the stream laterally may experience some degree of breakdown in the terrestrial environment before entering the stream channel (Abelho 2001). The breakdown process of colonization, degradation, and fragmentation makes the litterfall useable to other stream biota, which in turn are available for higher trophic levels.

The quantity, quality, and timing of litterfall inputs are largely determined by the composition of the riparian forest. Conifer trees, depending on species, can release litterfall material continuously or seasonally (Bilby and Bisson 1992; Benfield 1997). Deciduous trees and shrubs may lose some leaves during the summer (Richardson 1992; Richardson *et al.* 2004; Hart *et al.* 2013), but lose most of their biomass in a six- to eight-week pulse in the autumn (Gregory *et al.* 1991; Bilby and Bisson 1992; Hart *et al.* 2013). Despite the availability of conifer litterfall, decomposition of conifer needles is slower than that of most deciduous leaves, and conifer needles require a longer period of microbial conditioning to increase palatability before consumption by invertebrates (Triska *et al.* 1982). Deciduous litterfall, such as red alder (*Alnus rubra*), typically has a higher nitrogen content and lower carbon to nitrogen ratio than conifer needles and so decays rapidly (Triska *et al.* 1982; Richardson *et al.* 2004; Kominoski *et al.* 2011; Martinez *et al.* 2013), reducing the length of time the material is available to consumers (Kiffney and Richardson 2010). Although wood has a high carbon to nitrogen ratio and is rich in secondary compounds, which inhibit decomposition and slow breakdown (McKie and Cranston 2001), wood provides a long-term reserve of essential nutrients (Anderson *et al.* 1978) and serves as a retention structure for litterfall inputs (Hetrick *et al.* 1998).

Timber harvest can alter the volume, species composition, and timing of litterfall inputs. Research has shown that conifer litterfall inputs decrease in clearcut streams following tree removal, whereas deciduous litterfall inputs increase (Kiffney and Richardson 2010). Wood input may also decrease following harvest because even though young red alder grow rapidly the trees that die are small and may have minimal contribution to channel structure (Gomi *et al.* 2006). In Pacific Northwest streams, however, logging practices introduce large quantities of wood to streams in the form of slash (Jackson *et al.* 2001; Haggerty *et al.* 2004), which may help maintain stream channel structure and retention of litterfall inputs at least in the short term (Hetrick *et al.* 1998).

Removal of the tree canopy creates conditions favorable for early successional, fast-growing plant species such as red alder. In contrast to an intact forest stream, where the diversity of litterfall sources allows a more constant input of litter throughout the year, a clearcut stream with early successional deciduous vegetation receives the bulk of its litter over a short period of time (Bilby and Bisson 1992). In addition, an increase in flows and sediment loads could accelerate breakdown of litterfall through physical abrasion (Benfield *et al.* 2001), and an increase in temperature and nutrient concentrations could stimulate microbial colonization and degradation of litter, which may in turn stimulate activity of detritivores (Benfield *et al.* 2001; McKie and Malmqvist 2009).

Changes in litterfall inputs following timber harvest may influence the quantity of organic material stored in the streambed substrate, but research suggests that there is little change in storage with a reduction in inputs (Hetrick *et al.* 1998; Wallace *et al.* 1999; Richardson and Beraud 2014). The proportion of deciduous leaves stored in the substrate, however, is higher in clearcut streams compared with forested streams (Bilby and Bisson 1992; McKie and Malmqvist 2009; Martinez *et al.* 2013). Although deciduous leaves and other coarse particulate organic matter comprise most of the litterfall input into streams, only a small percentage of this material is exported (O’hop and Wallace 1983; Wallace *et al.* 1995). In headwater streams, much of the coarse particulate organic matter is retained within the system and processed by microbes and detritivores to fine particulate organic matter and dissolved organic matter (Wallace *et al.* 1995). Export of fine detritus can be several times higher than export of coarse detritus (O’hop and Wallace 1983) and is positively correlated with stream discharge (Wallace *et al.* 1991; Waringer 1992; Kiffney *et al.* 2000). While timber harvest may increase stream flows, research shows that there is little difference in total detritus export from a wide range of treatments following harvest (Piccolo and Wipfli 2002; Wipfli and Gregovich 2002; Musslewhite and Wipfli 2004).

The Type N Experimental Buffer Treatment Study (Type N Study) was designed to evaluate the response of a variety of riparian and instream processes to different stream buffer treatments pre- and post-harvest. This component of the study examined the changes in quantity and composition of litterfall inputs and detritus exports from Type N Waters. We hypothesized that total litterfall inputs would decrease immediately post-harvest, and that input of deciduous litterfall would increase in proportion thereafter with the establishment of early successional red alder and shrub communities. We expected that the overall magnitude of change in litterfall input post-harvest would reflect the buffer treatment, with no change in the 100% treatment, a moderate decrease in the FP treatment, and a large decrease in the 0% treatment relative to the reference. We also hypothesized that total detritus export would not change post-harvest in any treatment relative to the reference, but that the proportion of wood detritus in export would increase with the addition of slash to streams. Furthermore, we expected that the addition of slash would increase retention of coarser litter in the streams, which would decrease the proportion of conifer and deciduous material in export and increase the proportion of fine particulate organic matter in export.

12-3. OBJECTIVES

The overarching objective of this part of the study was to examine the magnitude and direction of change in litterfall input and detritus export after timber harvest in three experimental buffer treatments. The research questions were:

1) What is the magnitude of change in total annual litterfall input and composition and mean detritus export quantity and composition following timber harvest in each of three experimental buffer treatments relative to an unharvested reference site?

2) What are the differences in the magnitude of the change in total annual litterfall input and composition and mean detritus export quantity and composition among the three experimental buffer treatments?

12-4. METHODS

12-4.1. SITE DESCRIPTION

We limited sampling of litterfall input and detritus export to the Olympic and Willapa 1 blocks, which consisted of a total of eight study sites with two replicates of each treatment, because of the time and cost required to collect and process samples and the need for quantifying flows for interpreting exports (**Figure 12-1**). The 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. The sites were located in managed Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*)-dominated second-growth forests on private, state, and federal land, with stand ages ranging from 30 to 80 years old. The riparian stands consisted predominantly of conifers pre-harvest, with deciduous trees, primarily red alder, making up as little as 0% of the basal area per acre in the WIL1-FP and as much as 22% in the OLYM-0% (**Table 12-1**) (see Chapter 5 - *Stand Structure and Tree Mortality Rates in Riparian Buffers*). Sites were located in areas dominated by competent lithology types, with average Np channel gradients ranging from 16% to 31% (see **Table 2-6** in Chapter 2 - *Study Design*). These sites can be considered as representative of Type N basins located in second-growth forests on lands managed for timber production, dominated by competent lithologies, located in western Washington, and consistent with our other site selection criteria (size, gradient, etc.; see **Table 2-1** in Chapter 2 - *Study Design*).

12-4.2. EXPERIMENTAL TREATMENTS

The four sites within each block received one of the following treatments (see **Figure 2-2** in Chapter 2 - *Study Design*). A 30-ft (9.1-m) equipment limitation zone was maintained along all Type Np and Ns Waters, regardless of assigned treatment.

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 the entire perennial stream length buffered with a two-sided 50-ft (15.2-m) riparian buffer,

3) **Forest Practices treatment (FP):** clearcut harvest with a current Forest Practices two-sided 50-ft (15.2-m) riparian leave-tree buffer along at least 50% of the Riparian Management Zone (RMZ), including buffers prescribed for sensitive sites (side-slope and headwall seeps, headwater springs, Type Np intersections and alluvial fans), and

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

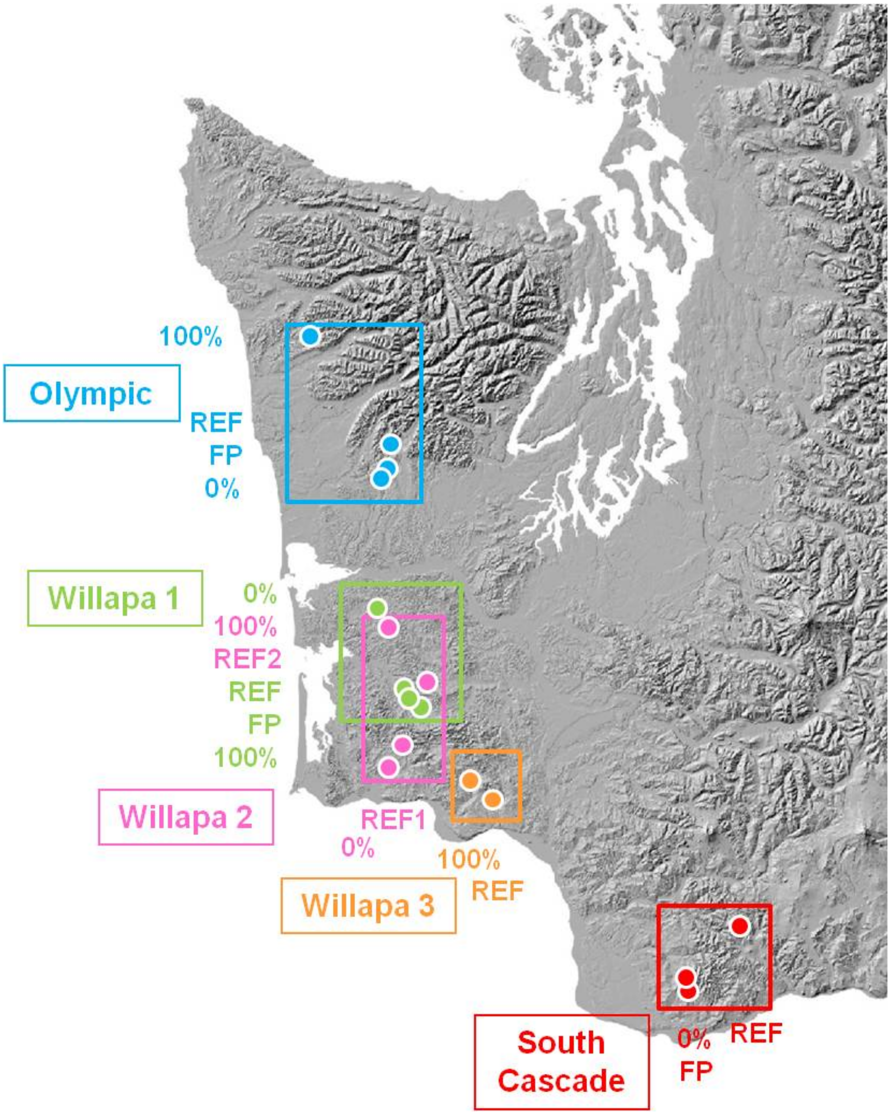


Figure 12-1. Distribution of the Type N study sites and treatments. Study sites are blocked and based on geography. Litterfall and detritus samples were collected from the Olympic (color-coded blue) and Willapa 1 (color-coded green) blocks. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment.

Table 12-1. Mean pre-harvest stand composition of the riparian management zone in percent live conifer and deciduous tree basal area per acre (% BAPA) averaged by treatment year. OLYM = Olympic Block; WIL1 = Willapa Block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment. Adapted from Chapter 5 – *Riparian Stand Conditions and Buffer Tree Mortality Rates*.

Study Site	Conifer (% BAPA)	Deciduous (% BAPA)
OLYM-REF	99.2%	0.8%
OLYM-100%	84.4%	15.7%
OLYM-FP	88.3%	11.7%
OLYM-0%	78.1%	22.0%
WIL1-REF	96.1%	3.9%
WIL1-100%	99.3%	0.7%
WIL1-FP	100.0%	0.0%
WIL-0%	91.5%	8.5%

12-4.3. LITTERFALL INPUT

12-4.3.1. *Sample Collection*

Litterfall sampling began during the summer and early fall of 2006. In 2008 and 2009, sites received one of the four treatments. Sampling ceased in all sites at the end of September 2011 when at least two years of post-harvest data collection were completed at all sites.

We installed four litterfall collection stations along the mainstem channel of each site in locations consistent with the riparian management prescriptions outlined by the Forest Practices rules. In the sites receiving the FP buffer treatment, we installed the first litterfall trap station (L1) just upstream of the F/N break, where the Type F fish-bearing stream becomes the Type Np perennial non-fish-bearing stream; the second station (L2) upstream of the first, inside the prescribed stream buffer; the third station (L3) upstream of the second station and outside the FP prescribed buffer; and the fourth station (L4) near the uppermost point of perennial flow. Buffer locations were not laid out by the landowners prior to litterfall station installation, so we approximated the location of the buffers based on the size of the FP basins. Stations were positioned in comparable locations (in terms of their general proximity to one another) in the reference sites and in the 100% and 0% treatment sites. **Table 12-2** provides the location of each litterfall station in each study site relative to the F/N break and other stations as well as the adjacent riparian buffer type.

Table 12-2. Distance of the litterfall stations to the F/N break and to the nearest downstream station, and the adjacent riparian management prescription. OLYM = Olympic Block; WIL1 = Willapa Block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; NAH = no adjacent harvest; BUF = buffered; UNB = unbuffered (no riparian buffer); PIP = perennial initiation point buffer.

Study Site	Station	Distance from F/N break (m)	Distance from Downstream Station (m)	Riparian Management Prescription
OLYM-REF	L1	373	–	NAH
	L2	420	47	NAH
	L3	544	124	NAH
	L4	1369	825	NAH
OLYM-100%	L1	237	–	BUF
	L2	318	81	BUF
	L3	419	101	BUF
	L4	674	255	BUF
OLYM-FP	L1	23	–	BUF
	L2	121	98	BUF
	L3	225	104	UNB
	L4	266	41	PIP
OLYM-0%	L1	4	–	UNB
	L2	81	77	UNB
	L3	209	128	UNB
	L4	342	133	UNB
WIL1-REF	L1	11	–	NAH
	L2	104	93	NAH
	L3	200	96	NAH
	L4	459	259	NAH
WIL1-100%	L1	147	–	BUF
	L2	240	93	BUF
	L3	340	100	BUF
	L4	615	275	BUF
WIL1-FP	L1	5	–	BUF
	L2	104	99	BUF
	L3	204	100	UNB
	L4	370	166	PIP
WIL-0%	L1	14	–	UNB
	L2	104	90	UNB
	L3	224	120	UNB
	L4	569	345	UNB

Each station was equipped with two litterfall traps, one on each bank, located just outside of the bankfull channel. The litterfall traps consisted of plastic household laundry baskets 25.4 cm high with a 0.153 m^2 opening (**Figure 12-2a**). The traps were positioned to sample overhead litterfall. We did not sample lateral ground inputs because of the time and cost required to collect and process samples. We drilled holes in the bottom of the traps to allow water to drain, and anchored the traps to the substrate with rebar to ensure that the traps were stable and level. We installed the traps in areas without overhanging obstructions that could potentially bias litterfall samples (e.g., directly beneath an overhanging log; Grady 2001). After installation, we lined the traps with screen (14 × 18 wires per inch), cut to fit the interior surface area and held in place with clothespins.

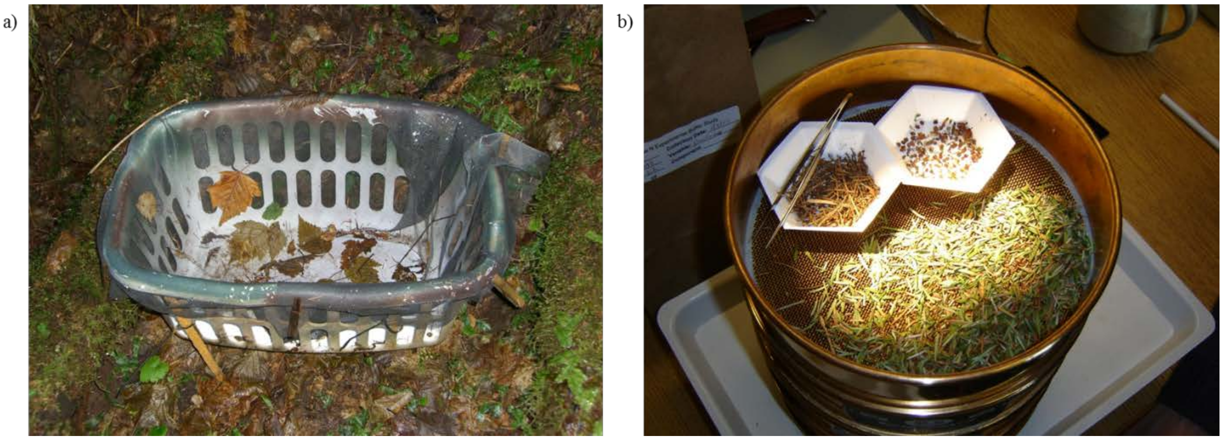


Figure 12-2. Litterfall trap deployment in the Type N Study sites (a) and litterfall sample sorting in the lab (b).

We collected the litterfall samples at roughly six-week intervals to minimize decomposition of material in the trap, although site inaccessibility due to snow and ice or harvest of adjacent timber stands delayed sample collection until a later date on a few occasions. Samples were collected by brushing material off of the screen on the basket periphery and folding the nets in on themselves to prevent loss of contents. We combined the nets from the paired traps at each station into a plastic bag that we labeled and sealed, and then immediately placed a clean net in each trap so that sampling was nearly continuous. The litterfall traps remained in the sites over the duration of the project unless they were located in a section of the site subject to harvest, in which case they were removed prior to harvest and returned when harvest was complete. Litterfall sampling may have been interrupted for several reasons, including trap removal during harvest, technician error in the collection of samples, or trap disturbance or damage from falling branches and trees (**Table 12-3**). Litterfall trap net installation and retrieval dates are listed for each site and station in **Appendix Table 12-A-1**.

Table 12-3. Timing and duration of disruptions in the litterfall sample collection record for each study site, station, and treatment year. OLYM = Olympic Block; WIL1 = Willapa Block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; L1 = litterfall station 1; L2 = litterfall station 2; L3 = litterfall station 3; L4 = litterfall station 4; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2; HARVEST = harvest period.

Study Site	Station	Treatment Year	Disruption		Duration (days)	Reason
			Start Date	End Date		
OLYM-REF	L1	Pre1	5/24/2007	7/10/2007	47	Sample missing
	L3	Pre1	1/25/2007	3/5/2007	39	Sample missing
OLYM-100%	L1	Harvest	10/1/2008	4/15/2009	196	Harvest period
	L2	Harvest	10/1/2008	4/15/2009	196	Harvest period
	L3	Harvest	10/1/2008	4/15/2009	196	Harvest period
	L4	Harvest	10/1/2008	4/15/2009	196	Harvest period
	L4	Post2	1/19/2011	4/12/2011	83	Sample labeling error
OLYM-FP	L1	Harvest	7/14/2008	12/3/2008	142	Harvest period
	L2	Harvest	7/14/2008	12/3/2008	142	Harvest period
	L3	Harvest	7/14/2008	12/3/2008	142	Harvest period
	L4	Harvest	7/14/2008	12/3/2008	142	Harvest period
OLYM-0%	L1	Harvest	5/28/2009	9/30/2009	125	Harvest period
	L2	Harvest	5/28/2009	9/30/2009	125	Harvest period
	L3	Harvest	5/28/2009	9/30/2009	125	Harvest period
	L4	Harvest	5/28/2009	9/30/2009	125	Harvest period
WIL1-100%	L1	Harvest	9/15/2008	4/27/2009	224	Harvest period
	L2	Pre1	5/15/2007	6/29/2007	45	Sample missing
	L2	Harvest	9/15/2008	4/27/2009	224	Harvest period
	L2	Post1	10/27/2009	12/8/2009	42	Trap damaged by fallen tree
	L3	Harvest	9/15/2008	4/27/2009	224	Harvest period
	L4	Harvest	9/15/2008	5/11/2009	238	Harvest period
WIL1-FP	L1	Harvest	9/15/2008	3/10/2009	176	Harvest period
	L2	Harvest	9/15/2008	3/10/2009	176	Harvest period
	L3	Harvest	9/15/2008	3/10/2009	176	Harvest period
	L4	Harvest	9/15/2008	3/10/2009	176	Harvest period
WIL-0%	L1	Harvest	3/31/2008	1/26/2009	301	Harvest period
	L2	Harvest	3/31/2008	1/26/2009	301	Harvest period
	L3	Harvest	3/31/2008	1/26/2009	301	Harvest period
	L4	Harvest	3/31/2008	1/26/2009	301	Harvest period

12-4.3.2. *Sample Processing*

After sample collection, we air-dried the litterfall samples by opening the sample bags and placing them inside a storage cabinet with a 250-Watt heat lamp to limit decomposition and mold growth. Terrestrial detritivores, if present in the samples, were rare and likely had no appreciable effect on the litterfall samples. When the sample was thoroughly dry (seven to ten days), we brushed the litterfall from the nets and stored the sample in a labeled plastic bag until processing. Every quarter, we set aside about 10 to 20 mg of litterfall for stable isotope analysis for the trophic pathways component of the study (described in Supplement 2 – *Stable Isotopes Analysis*). This was accomplished by thoroughly mixing the litterfall sample, removing a subsample, visually examining the subsample to ensure that it was representative of the total sample and that the litterfall components (listed below) were distributed proportionally, and then weighing and freezing the subsample.

We sorted the remaining litterfall sample into four components: coniferous needles and cedar scales (CONIF), deciduous tree and shrub leaves (DECID), woody material and cones (WOOD), and miscellaneous (MISC; **Figure 12-2b**). MISC consisted of litterfall such as flowers, lichens, and moss that did not fit into any of the other component categories. If a sample was too large to be sorted within a reasonable amount of time (about eight hours), we thoroughly mixed the litterfall sample, removed a subsample, visually examined the subsample to ensure that it was representative of the total sample and that the litterfall components were distributed proportionally, and then weighed and sorted the subsample.

Sorted components were stored in labeled paper bags and then dried in a drying oven at 55°C for at least 96 hours. After cooling, we weighed the component on a calibrated scale and recorded the dry weight to the nearest milligram. We then placed the dried and weighed component (about 0.23 g on average) into a crucible and ashed the component in a muffle furnace at 550°C for one hour. If the component was too large to ash within an hour, we thoroughly mixed the component, removed a subsample, visually examined the subsample to ensure that it was representative of the component, and then weighed and ashed the subsample. After ashing, we cooled and weighed the remaining material on a calibrated scale and recorded the ashed weight to the nearest milligram.

12-4.3.3. *Statistical Analysis*

For each sample component, we calculated ash-free dry mass (AFDM) by subtracting the component ashed weight from the component dry weight. To account for the mass of the subsample removed for stable isotope analysis, we added the dry weight of the subsample to that of the remaining sorted litterfall sample, and then proportionally adjusted the dry weight and AFDM of each litterfall component based on the relative proportion of the component in the sorted sample. If the litterfall sample was subsampled before sorting or a component was subsampled prior to ashing due to size, we adjusted dry weight and AFDM based on the proportion of the original sample that was processed. We then divided AFDM by 0.307 m^2, the surface area of the paired litterfall trap openings, to determine the amount of litter entering the stream per square meter (g AFDM m^{-2}). If only one of the paired litterfall traps was collected, which occurred on three occasions when one of the paired traps was destroyed, we divided AFDM by 0.153 m^2, the surface area of one litterfall trap opening.

We grouped the g AFDM m^{-2} data into designated treatment years: pre-harvest year 1 (PRE1), pre-harvest year 2 (PRE2), post-harvest year 1 (POST1), and post-harvest year 2 (POST2) (**Appendix Table 12-A-1**). For the treatment sites, the pre-harvest period (PRE) consisted of the two years preceding the beginning of harvest, and the post-harvest period (POST) the two years after the end of harvest. "Harvest" years and periods were designated for the reference sites and were determined by the time in which harvest was scheduled to occur in most of the treatment sites. The PRE period consisted of the two years before 11 December 2008 in the OLYM-REF and 9 December 2008 in the WIL1-REF, and the POST period the two years after those dates (**Appendix Table 12-A-1**).

We designated discrete sample collection dates for the beginning and end of each treatment year because although litterfall sampling (i.e., trapping) was continuous, litterfall collection was episodic. We omitted selected samples from the analysis due to mishandling, loss, or damage to traps and samples in the field (**Table 12-3**). Only one of the omitted samples (WIL1-100% L2 POST1) was collected during the critical leaf drop period (27 October 2009 to 8 December 2009) and was discarded because the litterfall trap and its contents were damaged by a fallen tree. In addition to omitted samples, differences in harvest timing, scheduling of field visits, and accessibility of sites resulted in differences in the length of each treatment year. Most treatment years were therefore not exactly 365 days, and the number of days in a treatment year varied between sites and even between stations (**Table 12-4**). Litterfall traps were deployed at each site for an average of 721 days during the PRE period and ranged from 656 to 812 days per site, depending on when traps were installed and harvest initiated. During the POST period, litterfall traps were deployed at each site for an average of 728 days with a range of 644 to 756 days per site.

Total litterfall (TOTAL) input was calculated for each sample by summing g AFDM m^{-2} of the CONIF, DECID, WOOD, and MISC litterfall components, and total leaf (LEAF) litterfall by summing g AFDM m^{-2} of the CONIF and DECID litterfall components. We then summed g AFDM m^{-2} for TOTAL, LEAF, CONIF, DECID, WOOD, and MISC litterfall by treatment year and divided the annual totals by the number of sampling days for each respective treatment year to determine grams AFDM per square meter per day (g AFDM m^{-2} day^{-1}). Finally, we averaged g AFDM m^{-2} day^{-1} for TOTAL, LEAF, CONIF, DECID, WOOD, and MISC litterfall for the four stations across each study site.

The data were not normally distributed so we calculated the base 10 log of the g AFDM m^{-2} day^{-1} values +1. We analyzed the data using a generalized linear mixed effects model (GLMM) analysis of variance (ANOVA) with buffer treatment (REF, 100%, FP, and 0%), period (PRE and POST), and the treatment × period interaction as fixed effects, and site as a random effect. 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 used the Kenward-Roger method (Kenward and Roger 1997) for estimating the denominator degrees of freedom for tests of fixed effects and the contrasts because of the unbalanced design. We used SAS software version 9.4 for the GLMM analyses (SAS 2013).

Table 12-4. Number of litterfall sampling days for each study site, station, and treatment year. OLYM = Olympic Block; WIL1 = Willapa Block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; L1 = litterfall station 1; L2 = litterfall station 2; L3 = litterfall station 3; L4 = litterfall station 4; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2.

Study Site	**Station**	**Treatment Year**			
		PRE1	**PRE2**	**POST1**	**POST2**
OLYM-REF	L1	324	358	363	369
	L2	371	358	363	369
	L3	332	358	363	369
	L4	365	358	363	369
OLYM-100%	L1	434	366	362	365
	L2	434	366	362	365
	L3	434	366	362	365
	L4	446	366	362	282
OLYM-FP	L1	360	368	376	365
	L2	360	368	376	365
	L3	360	368	376	365
	L4	360	368	376	365
OLYM-0%	L1	369	366	364	364
	L2	369	366	364	364
	L3	369	366	364	364
	L4	369	366	364	364
WIL1-REF	L1	372	363	363	365
	L2	372	363	363	365
	L3	372	363	363	365
	L4	372	363	363	365
WIL1-100%	L1	341	363	379	363
	L2	296	363	337	363
	L3	341	363	379	363
	L4	341	363	365	363
WIL1-FP	L1	342	362	386	370
	L2	342	362	386	370
	L3	342	362	386	370
	L4	342	362	386	370
WIL1-0%	L1	293	363	344	363
	L2	302	363	344	363
	L3	302	363	344	363
	L4	302	363	344	363

We evaluated six hypotheses that can be grouped under the two research questions:

1) What was the magnitude of change in litterfall input post-harvest in each treatment relative to an unharvested reference site?

This was addressed with three *post hoc* comparisons testing the following hypothesis for each treatment:

$$\text{H0: } \Delta L_{\text{REF}} = \Delta L_{100} \tag{12-1}$$

$$\text{H0: } \Delta L_{\text{REF}} = \Delta L_{\text{FP}} \tag{12-2}$$

$$\text{H0: } \Delta L_{\text{REF}} = \Delta L_{0} \tag{12-3}$$

where: ΔL_{REF} is change in litterfall input in the unharvested reference site,
ΔL_{100} is change in litterfall input in the 100% treatment,
ΔL_{FP} is change in litterfall input in the FP treatment, and
ΔL_{0} is change in litterfall input in the 0% treatment.

2) Did the magnitude of the change differ among treatments?

This was addressed with three *post hoc* comparisons testing the following hypotheses:

$$\text{H0: } \Delta L_{100} = \Delta L_{\text{FP}} \tag{12-4}$$

$$\text{H0: } \Delta L_{100} = \Delta L_{0} \tag{12-5}$$

$$\text{H0: } \Delta L_{\text{FP}} = \Delta L_{0} \tag{12-6}$$

We hypothesized that total litterfall inputs would decrease immediately post-harvest, and that input of deciduous litterfall would increase in proportion thereafter with the establishment of early successional red alder and shrub communities. We expected that the overall magnitude of change in litterfall input post-harvest would reflect the buffer treatment, with no change in the 100% treatment, a moderate decrease in the FP treatment, and a large decrease in the 0% treatment relative to the reference.

We present the results of the GLMM ANOVA (**Table 12-6**) and estimates of the effects and the associated 95% confidence intervals (CI) for each comparison (**Table 12-7**; **Figure 12-4**). The P-values were not adjusted for the multiple comparisons because the large number of comparisons relative to the limited replication of each treatment (two) increases the chance of a 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.

12-4.4. DETRITUS EXPORT

12-4.4.1. *Sample Collection*

We collected detritus from the drift in conjunction with the macroinvertebrate sampling (described in Chapter 14 - *Macroinvertebrate Export*). We did not sample benthic detritus because of the time and cost required to collect and process samples. Drift sampling began in the fall of 2006 after installation of the flumes and other hydrological monitoring equipment. In 2008 and 2009, the sites received one of four treatments. Sampling ceased in all sites at the end of September 2011 when at least two years of post-harvest data collection were completed in all sites.

We collected drift samples at the location of the hydrological monitoring equipment to enable flow quantification. The hydrological equipment was installed as close to the F/N break as logistically feasible, although stream channel morphology often restricted installation of the equipment to stream reaches well away from the F/N break (**Table 12-5**). In the OLYM-REF and OLYM-100%, we installed the hydrological equipment in culverts, which were located well upstream of the F/N break but in locations representative of the treatment units. The flumes installed in the OLYM-FP and OLYM-0%, on the other hand, were located 50 and 240 m respectively, downstream of both the F/N break and the treatment units. While the flumes in the Willapa block sites were not located exactly at the F/N break, they were still in locations representative of their respective treatments.

Table 12-5. Distance of the hydrological measuring equipment and drift net to the F/N break and treatment unit, and the adjacent riparian management prescription. A positive distance denotes an upstream direction from the F/N break or treatment unit, while a negative distance a downstream direction. OLYM = Olympic Block; WIL1 = Willapa Block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; NAH = no adjacent harvest; BUF = buffered; UNB = unbuffered (no riparian buffer).

Block	Treatment	Equipment Type	Distance from		Riparian Management Prescription
			F/N Break (m)	Treatment Unit (m)	
OLYM	REF	Culvert	357	–	NAH
	100%	Culvert	229	0	BUF
	FP	Flume	−50	−50	NAH[1]
	0%	Flume	−240	−240	NAH
WIL1	REF	Flume	−25	–	NAH
	100%	Flume	140	0	BUF
	FP	Flume	10	0	BUF
	0%	Flume	19	0	UNB

[1] Flume downstream of the F/N break. There was no adjacent harvest until May 2011, when the surrounding stand was harvested and a Type F buffer was left adjacent to the stream. The Type F buffer is visible in **Figure 14-3a**.

In the six flume sites, a 250-µm mesh drift net was wrapped around an aluminum lip installed below the mouth of the flume and secured with zip ties to eyebolts attached to the flume (**Figure 12-3a**). In the two culvert sites, the drift net was attached with zip ties to a metal rod that extended along the streambed across the stream channel perpendicular to flow (**Figure 12-3b**). We installed two other metal rods oriented vertically into the streambed to support the upper portion of the drift net to allow sampling of the water column. Although this method did not sample the entire stream cross-section, the net sampled most of the cross-section and always included the thalweg, the area of most active detrital transport. The depositional margins of the stream channel where detritus was most likely to be retained was not sampled.

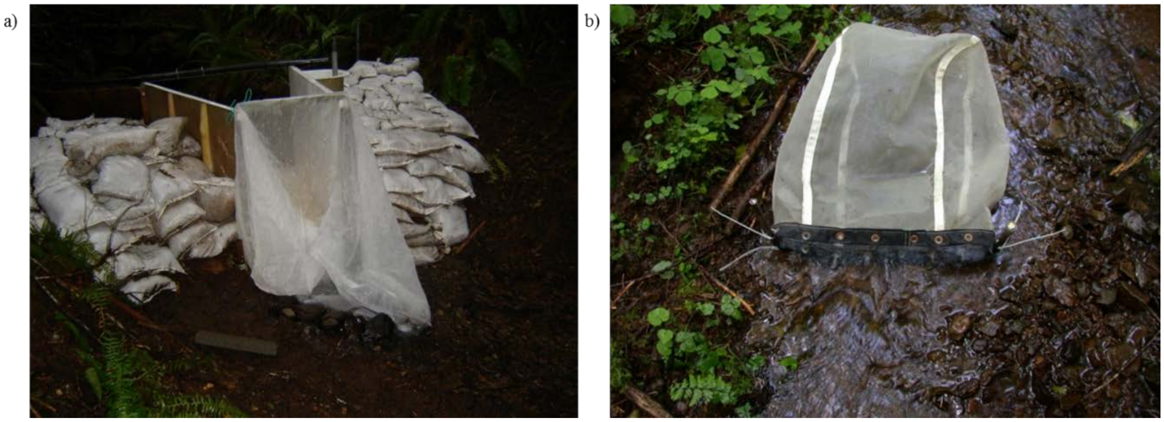


Figure 12-3. Drift net deployment in the Type N Study sites with a flume (a) and without a flume (b).

We sampled drift for approximately one continuous 24-hour period every six weeks throughout the year for a total of about nine samples per year per site. The number of samples per year varied for some of the sites, however, depending on the harvest schedule, site accessibility, and flow conditions. Field visit scheduling and high flows during some of the fall, winter, and spring sampling efforts often forced shorter deployment periods to prevent the mesh from clogging with detritus and the loss of the net (**Appendix Table 12-B-1**). While we were able to sample high flows in some cases, we were not able to target specific storm events. For each effort, we recorded the drift net deployment and retrieval date and time, and estimated the proportion of the flow volume sampled by the drift net (Estrella 2006). We used *in situ* pressure transducers to measure stage height at 10-minute intervals. At the six flume sites, discharge was calculated from the flow versus stage height curve for that size flume (described in Chapter 8 - *Discharge*). For the two culvert sites, discharge was estimated from a site-specific flow versus stage height curve developed over the course of the study (see Chapter 8 - *Discharge*).

We preserved the samples in 70% ethanol. Drift samples analyzed for the trophic pathways component of the study (see Supplement 2 – *Stable Isotopes Analysis*) were stored in stream water and kept on ice until processed.

12-4.4.2. *Sample Processing*

We rinsed the drift samples through 1-mm and 250-µm nested sieves and sorted the macroinvertebrates from the detritus. Detritus retained on the 1-mm sieve was designated coarse particulate organic matter (CPOM) and the detritus retained on the 250-µm sieve was designated fine particulate organic matter (FPOM). The CPOM and FPOM were placed into labeled paper bags and air dried to limit decomposition and mold growth. Every quarter, we set aside about 10 to 20 mg of CPOM for stable isotope analysis for the trophic pathways component of the study (see Supplement 2 – *Stable Isotopes Analysis*). This was accomplished by thoroughly mixing the CPOM sample, removing a subsample, visually examining the subsample to ensure that it was representative of the total sample and that the CPOM components were distributed proportionally, and then weighing and freezing the subsample.

We sorted the remaining CPOM sample into the same four components as litterfall (**Figure 12-2b**). If a sample was too large to be sorted within a reasonable amount of time (about eight hours), we thoroughly mixed the CPOM sample, removed a subsample, visually examined the subsample to ensure that it was representative of the total sample and that the CPOM components were distributed proportionally, and then weighed and sorted the subsample.

Sorted CPOM and FPOM components were stored in labeled paper bags and dried in a drying oven at 55°C for at least 96 hours. After cooling, we weighed the component on a calibrated scale and recorded the dry weight to the nearest milligram. We then placed the dried and weighed component (about 0.40 and 0.55 g on average for CPOM and FPOM components, respectively) into a crucible and ashed the component in a muffle furnace at 550°C for one hour. If the component was too large to ash within an hour, we thoroughly mixed the component, removed a subsample, visually examined the subsample to ensure that it was representative of the component, and then weighed and ashed the subsample. After ashing, we cooled and weighed the remaining material on a calibrated scale and recorded the ashed weight to the nearest milligram.

12-4.4.3. *Statistical Analysis*

For each sample component, we calculated AFDM by subtracting the component ashed weight from the component dry weight. To account for the mass of the subsample removed for stable isotope analysis, we added the dry weight of the subsample to that of the remaining sorted CPOM sample, and then proportionally adjusted the dry weight and AFDM of each CPOM component based on the relative proportion of the component in the sorted sample. If the CPOM sample was subsampled before sorting or a component was subsampled prior to ashing due to size, we adjusted dry weight and AFDM based on the proportion of the original sample that was processed. We then calculated g AFDM day^{-1} by dividing the AFDM of the sample component by the drift net deployment time and then extrapolating to a 24-hour sampling period and a 100% sampled stream flow volume. Total detritus (TOTAL) export was calculated for each sample by summing g AFDM day^{-1} of the CONIF, DECID, WOOD, MISC, and FPOM detritus components, total CPOM (CPOM) detritus by summing g AFDM day^{-1} of the CONIF, DECID, WOOD, and MISC detritus components, and total leaf (LEAF) detritus by summing g AFDM day^{-1} of the CONIF and DECID detritus components.

The stream volume per 10-minute interval was summed for the entire deployment period and then standardized to a 24-hour day to give flow in cubic meters per day (m^3 day^{-1}). If part or all of the drift net detached from the flume or rebar during deployment, the proportion of flow sampled when the net was installed and when the net was removed was averaged (this assumed that detritus export was uniform across the stream cross-section). Flow was then divided by the basin area above the hydrological equipment to give flow per hectare.

We designated consistent treatment years for all four sites in a block to address inconsistent timing and duration of the timber harvests between sites and that detrital drift is dependent on stream flows. The PRE period consisted of the two pre-harvest years (PRE1 and PRE2) from the time when the first sample was collected in the fall of 2006 to the time when harvest first began in any site of the block. The POST period consisted of the two post-harvest years (POST1 and POST2) following completion of harvest in the last site of the block. An average of 15 drift samples, with a range of 14 to 16 samples, was collected from each site during the PRE period. During the POST period, an average of 18 samples was collected from each site, with a range of 16 to 18 samples per site. The harvest schedule, inaccessibility of sites due to harvest, snow, and road conditions, and inability to sample because of high stream flows resulted in differences in the number of samples collected per site pre- and post-harvest (**Appendix Table 12-B-1**).

We analyzed detritus export for the TOTAL, CPOM, LEAF, CONIF, DECID, WOOD, MISC, and FPOM detritus components with each value (i.e., each drift sample) as an observation. The data were not normally distributed so we calculated the base 10 log of the calculated means +1. We analyzed the data using a GLMM ANOVA with buffer treatment (REF, 100%, FP, and 0%), period (PRE and POST), and the treatment × period interaction as fixed effects, and site and flow per area as random effects. We initially included block and season as random effects, but dropped them because the variance estimates associated with block and season were zero (i.e., block and season did not explain any additional variation in the dependent variables). We used the Kenward-Roger method (Kenward and Roger 1997) for estimating the denominator degrees of freedom for tests of fixed effects and the contrasts because of the unbalanced design. We used SAS software version 9.4 for the GLMM analyses (SAS 2013).

We evaluated six hypotheses that can be grouped under the two research questions:

1) What was the magnitude of change in mean detritus export post-harvest in each treatment relative to an unharvested reference site?

This was addressed with three *post hoc* comparisons testing the following hypothesis for each treatment:

$$\text{H0: } \Delta D_{\text{REF}} = \Delta D_{100} \qquad (12\text{-}7)$$

$$\text{H0: } \Delta D_{\text{REF}} = \Delta D_{\text{FP}} \qquad (12\text{-}8)$$

$$\text{H0: } \Delta D_{\text{REF}} = \Delta D_{0} \qquad (12\text{-}9)$$

where: ΔD_{REF} is change in detritus export from the unharvested reference site,
ΔD_{100} is change in detritus export from the 100% treatment,
ΔD_{FP} is change in detritus export from the FP treatment, and
ΔD_{0} is change in detritus export from the 0% treatment.

2) Did the magnitude of the change differ among treatments?

This was addressed with three *post hoc* comparisons testing the hypothesis:

$$H0: \Delta D_{100} = \Delta D_{FP} \tag{12-10}$$

$$H0: \Delta D_{100} = \Delta D_{0} \tag{12-11}$$

$$H0: \Delta D_{FP} = \Delta D_{0} \tag{12-12}$$

We hypothesized that total detritus export would not change post-harvest in any treatment relative to the reference, but that the proportion of wood detritus in export would increase with the addition of slash to streams. Furthermore, we expected that the addition of slash would increase retention of coarser litter in the streams, which would decrease the proportion of conifer and deciduous material in export and increase the proportion of fine particulate organic matter in export.

We present the results of the GLMM ANOVA (**Table 12-8**) and estimates of the effects and the associated 95% confidence intervals (CI) for each comparison (**Table 12-9**; **Figure 12-7**). The P-values were not adjusted for the multiple comparisons because the large number of comparisons relative to the limited replication of each treatment (two) increases the chance of a 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.

12-5. RESULTS

12-5.1. LITTERFALL INPUT

The GLMM ANOVA showed evidence of a strong treatment × period interaction for LEAF (P = 0.0045) and DECID (P = 0.0002) litterfall input (**Table 12-6**). There was a moderate interaction effect for TOTAL (P = 0.0854) litterfall input, but no interaction effect for CONIF (P = 0.2489), WOOD (P = 0.5547), and MISC (P = 0.5977) litterfall input.

Post-hoc comparisons between the PRE and POST treatment periods showed a decrease in TOTAL litterfall input in the FP (P = 0.0034) and 0% (P = 0.0001) treatments (**Table 12-7**). LEAF litterfall input decreased in the FP (P = 0.0114) and 0% (P <0.0001) treatments in the POST period. In addition, CONIF litterfall input decreased in the FP (P = 0.0437) and 0% (P <0.0001) treatments, DECID in the 0% (P <0.0001) treatment, WOOD in the FP (P = 0.0044) and 0% (P = 0.0153) treatments, and MISC in the 0% (P = 0.0422) treatment.

Post-hoc comparisons between the reference and the treatments showed a decrease in TOTAL litterfall input in the 0% treatment relative to the 100% treatment (P = 0.0205) (**Table 12-7**; **Figure 12-4**). LEAF litterfall input decreased in the 0% treatment relative to the reference (P = 0.0040), 100% (P = 0.0008), and FP (P = 0.0267) treatments. Likewise, there was a decrease in DECID litterfall input in the 0% treatment relative to the reference (P = 0.0001), 100% (P <0.0001), and FP (P = 0.0015) treatments.

Annual litterfall input in g AFDM m^{-2} day^{-1} for each Type N Study site by station, component, and treatment year is provided in **Appendix Table 12-C-1**. WOOD litterfall comprised most of the mean annual litterfall input in both the PRE and POST periods in the reference and in the buffer treatments (**Figure 12-5**). The proportion of WOOD litterfall input decreased during the POST period in the reference and in the 100% and FP treatments, while the proportion of CONIF, DECID, and MISC litterfall input increased. In the 0% treatment, the proportion of WOOD and MISC litterfall input increased in the POST period, while the proportion of CONIF and DECID litterfall input decreased.

Table 12-6. Results of the GLMM ANOVA for litterfall input in grams ash-free dry mass per square meter per day (g AFDM m^{-2} day^{-1}) by treatment (TRMT), period (TRYR), and the treatment × period interaction (TRMT × TRYR). Comparisons in **bold** print indicate P <0.05. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Effect	Num DF	Den DF	F-value	P-value
Total Litterfall				
TRMT	3	4	0.6	0.6471
TRYR	1	20	30.0	<0.0001
TRMT × TRYR	3	20	2.54	0.0854
Total Leaf Litterfall				
TRMT	3	4	1.13	0.4379
TRYR	1	20	30.65	<0.0001
TRMT × TRYR	3	20	5.97	**0.0045**
Coniferous				
TRMT	3	4	0.64	0.6269
TRYR	1	20	13.86	0.0013
TRMT × TRYR	3	20	1.49	0.2489
Deciduous				
TRMT	3	4	0.1	0.9587
TRYR	1	20	23.29	0.0001
TRMT × TRYR	3	20	11.26	**0.0002**

Table 12-6. (continued)

Effect	Num DF	Den DF	F-value	P-value
Wood				
TRMT	3	4	0.28	0.8405
TRYR	1	20	20.24	0.0002
TRMT × TRYR	3	20	0.71	0.5547
Miscellaneous				
TRMT	3	4	2.35	0.2136
TRYR	1	20	5.08	0.0356
TRMT × TRYR	3	20	0.64	0.5977

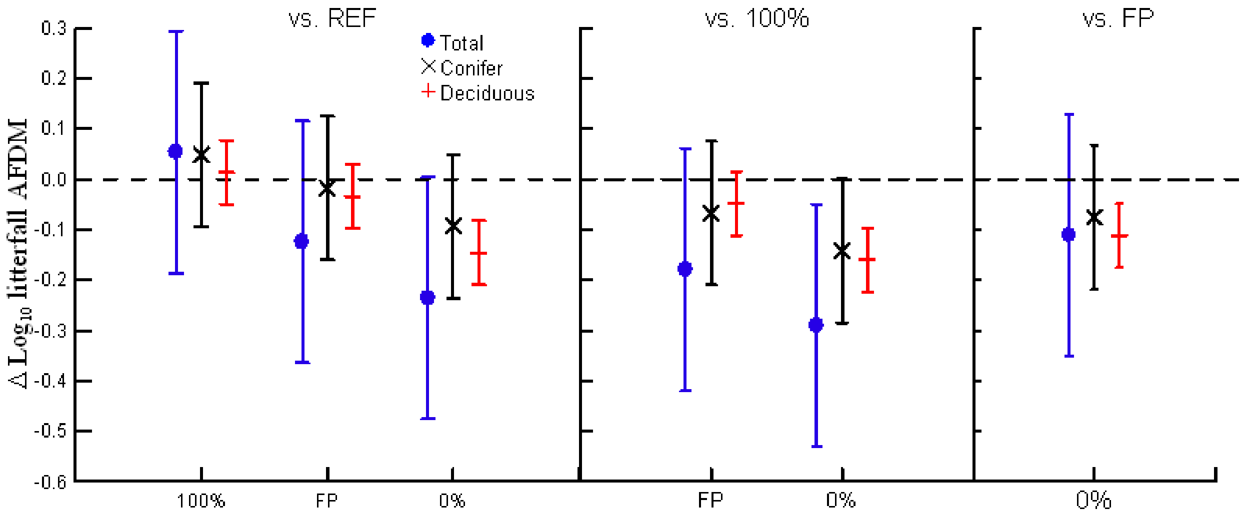


Figure 12-4. Results of the test of the six hypotheses for total, conifer, and deciduous litterfall input. The left panel shows the change in litterfall input post-harvest in each treatment relative to the reference. The right two panels compare the change in litterfall input among the three treatments. The error bars are 95% confidence intervals (CI). The horizontal dashed line equals zero (no change). Confidence intervals that do not cross the dashed line indicate the hypothesis is rejected at P <0.05.

Table 12-7. Results of hypothesis tests described in section 12-4.3.3 for litterfall input. Comparisons a, b, c, and d compare pre- and post-harvest litterfall input for each treatment, and comparisons e, f, g, h, i, and j compare post-harvest changes in litterfall input between the treatments. Comparisons in **bold** print indicate P <0.05. P-values were not adjusted for multiple comparisons.

Comparison	Change	P-value	Lower 95% C.I.	Upper 95% C.I.	Change	P-value	Lower 95% C.I.	Upper 95% C.I.	Change	P-value	Lower 95% C.I.	Upper 95% C.I.
	Total Litterfall				**Total Leaf Litterfall**				**Coniferous**			
a. REF-Pre vs. Post	−0.1467	0.0869	−0.3166	0.0233	−0.0705	0.1332	−0.1645	0.0235	−0.0558	0.1105	−0.1247	0.0130
b. 100%-Pre vs. Post	−0.0924	0.2702	−0.2623	0.0776	−0.0250	0.5852	−0.1190	0.0690	−0.0251	0.4702	−0.0940	0.0437
c. FP-Pre vs. Post	−0.2711	**0.0034**	−0.4410	−0.1011	−0.1255	**0.0114**	−0.2194	−0.0315	−0.0709	**0.0437**	−0.1397	−0.0021
d. 0%-Pre vs. Post	−0.3823	**0.0001**	−0.5523	−0.2124	−0.2779	**<0.0001**	−0.3719	−0.1839	−0.1574	**<0.0001**	−0.2263	−0.0886
e. REF vs. 100%	0.0543	0.6426	−0.1860	0.2946	0.0455	0.4831	−0.0874	0.1785	0.0491	0.4814	−0.0935	0.1916
f. REF vs. FP	−0.1244	0.2932	−0.3647	0.1160	−0.0549	0.3989	−0.1878	0.0780	−0.0176	0.7991	−0.1602	0.1250
g. REF vs. 0%	−0.2356	0.0542	−0.4760	0.0047	−0.2073	**0.0040**	−0.3403	−0.0744	−0.0929	0.1891	−0.2355	0.0497
h. 100% vs. FP	−0.1787	0.1367	−0.4190	0.0617	−0.1005	0.1306	−0.2334	0.0325	−0.0667	0.3410	−0.2093	0.0759
i. 100% vs. 0%	−0.2899	**0.0205**	−0.5303	−0.0496	−0.2529	**0.0008**	−0.3858	−0.1200	−0.1420	0.0509	−0.2846	0.0006
j. FP vs. 0%	−0.1113	0.3457	−0.3516	0.1291	−0.1524	**0.0267**	−0.2853	−0.0195	−0.0753	0.2837	−0.2179	0.0673
	Deciduous				**Wood**				**Miscellaneous**			
a. REF-Pre vs. Post	−0.0106	0.6285	−0.0556	0.0344	−0.1530	0.0806	−0.3263	0.0204	−0.0202	0.1959	−0.0517	0.0113
b. 100%-Pre vs. Post	0.0034	0.8782	−0.0416	0.0483	−0.1081	0.2081	−0.2815	0.0653	−0.0070	0.6483	−0.0385	0.0245
c. FP-Pre vs. Post	−0.0446	0.0519	−0.0896	0.0004	−0.2665	**0.0044**	−0.4399	−0.0931	−0.0081	0.5975	−0.0396	0.0234
d. 0%-Pre vs. Post	−0.1563	**<0.0001**	−0.2013	−0.1113	−0.2203	**0.0153**	−0.3937	−0.0469	−0.0327	**0.0422**	−0.0642	−0.0013
e. REF vs. 100%	0.0139	0.6524	−0.0497	0.0776	0.0448	0.7069	−0.2003	0.2900	0.0132	0.5431	−0.0313	0.0577
f. REF vs. FP	−0.0340	0.2783	−0.0976	0.0296	−0.1136	0.3455	−0.3587	0.1316	0.0121	0.5772	−0.0324	0.0566
g. REF vs. 0%	−0.1457	**0.0001**	−0.2093	−0.0821	−0.0674	0.5730	−0.3125	0.1778	−0.0126	0.5628	−0.0571	0.0320
h. 100% vs. FP	−0.0479	0.1317	−0.1115	0.0157	−0.1584	0.1929	−0.4036	0.0868	−0.0011	0.9591	−0.0456	0.0434
i. 100% vs. 0%	−0.1596	**<0.0001**	−0.2233	−0.0960	−0.1122	0.3512	−0.3574	0.1330	−0.0258	0.2415	−0.0703	0.0188
j. FP vs. 0%	−0.1117	**0.0015**	−0.1753	−0.0481	0.0462	0.6985	−0.1990	0.2914	−0.0247	0.2616	−0.0692	0.0199

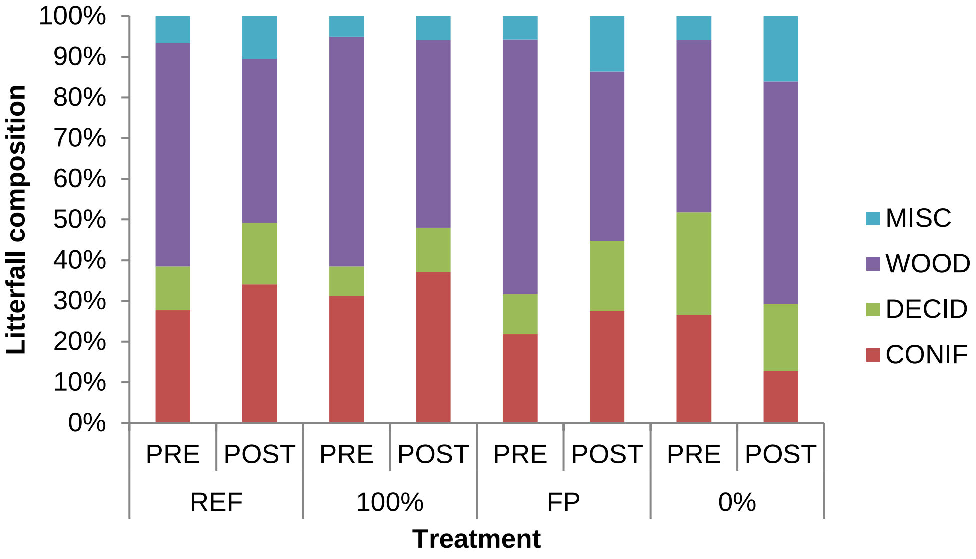
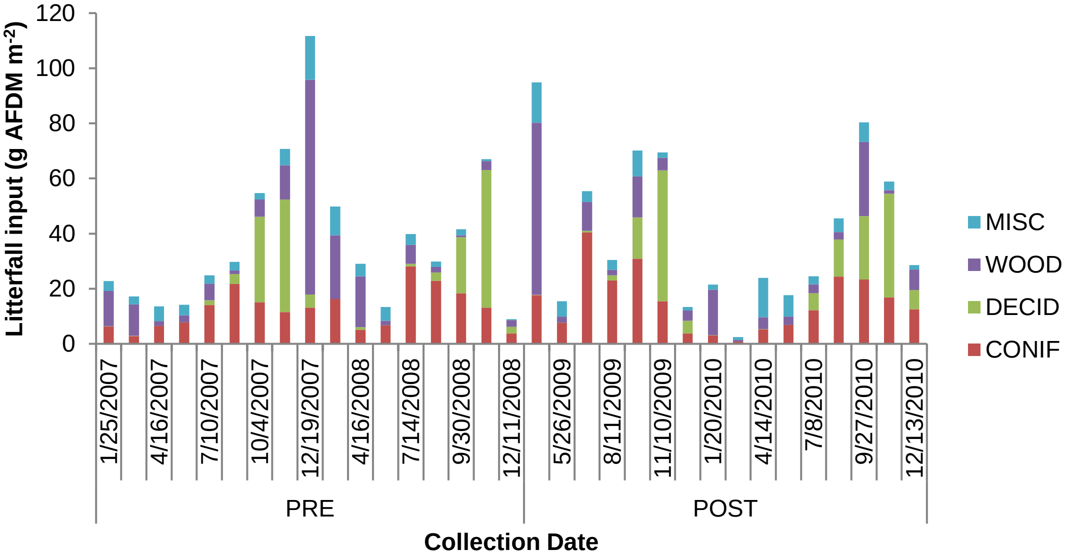


Figure 12-5. Composition of mean annual litterfall input in percent (%) for the Type N Study references and buffer treatments by component and treatment period. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period; CONIF = coniferous litterfall; DECID = deciduous litterfall; WOOD = wood litterfall; MISC = miscellaneous litterfall.

The litterfall input data are difficult to assess on a seasonal basis because litterfall sampling was continuous and the timing of sample collection varied relative to the beginning and ending of the seasons. The timing of sample collection also varied between the sites, which makes the comparison of individual samples between study sites problematic. With these caveats, we provide **Figures 12-6a** through **12-6h** as a rough sketch of litterfall inputs over the course of the study for each of the study sites. Litterfall inputs followed a predictable pattern, with the majority of input in the fall. DECID litterfall dominated most of the input in the fall, especially in the OLYM-REF, OLYM-100%, and OLYM-FP in the PRE and POST periods, and in the OLYM-0% and WIL1-0% in the PRE period. In the WIL1-REF, WIL1-100%, and WIL1-FP, CONIF and WOOD litterfall dominated more of the input in the fall. Large increases in litterfall inputs at other times of the year were dominated mostly by WOOD and sometimes CONIF litterfall. There was an apparent decrease in inputs from the PRE to POST period in the OLYM-0% and WIL1-0%, a slight decrease in the OLYM-FP and WIL1-FP, but no obvious decrease in the other sites.

(a) OLYM-REF



(b) OLYM-100%

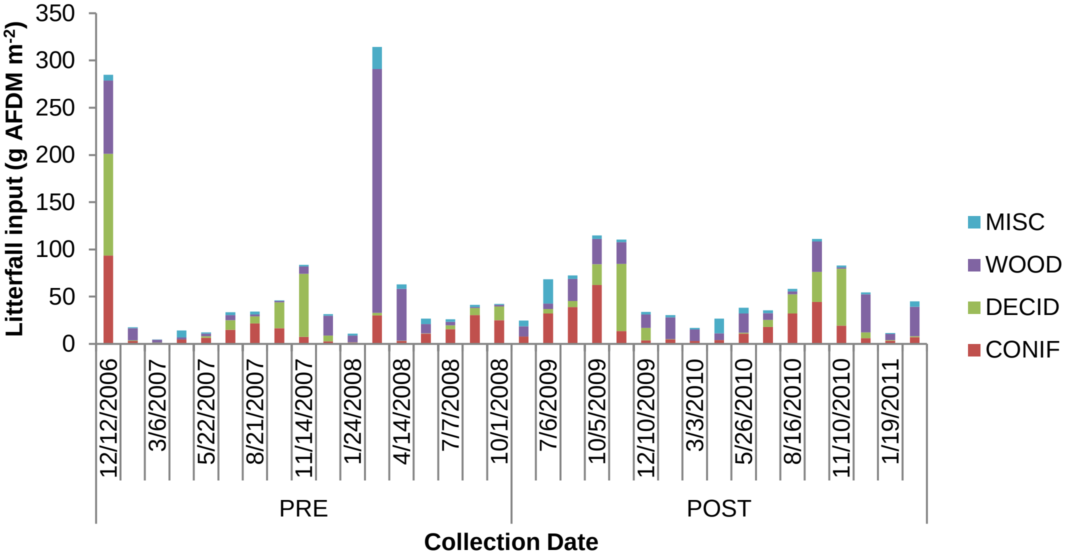
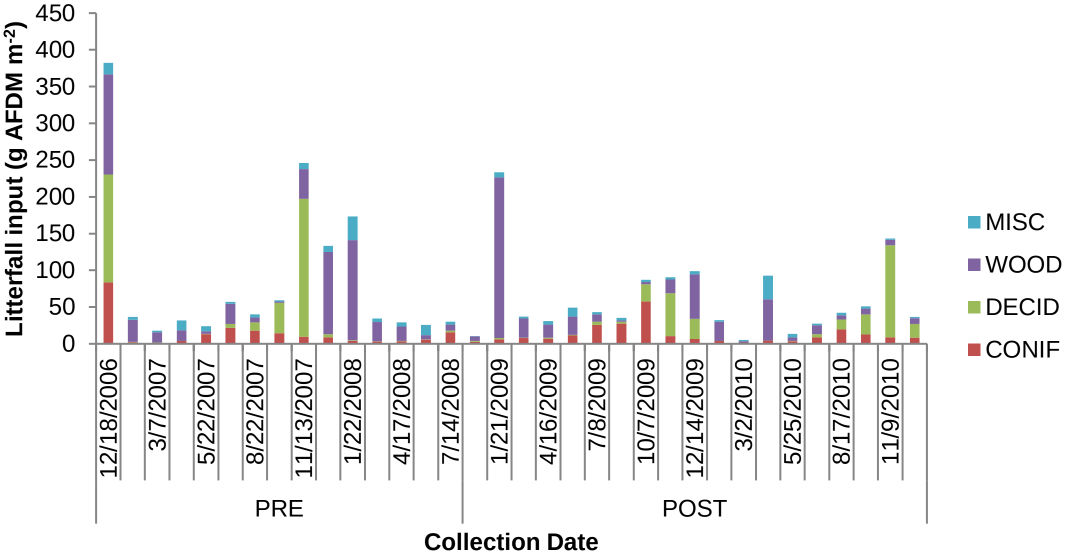


Figure 12-6. Litterfall inputs in grams ash-free dry mass per square meter (g AFDM m^{-2}) in the (a) OLYM-REF and (b) OLYM-100%. OLYM = Olympic block; WIL1 = Willapa 1 block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period; CONIF = coniferous litterfall; DECID = deciduous litterfall; WOOD = wood litterfall; MISC = miscellaneous litterfall.

(c) OLYM-FP



(d) OLYM-0%

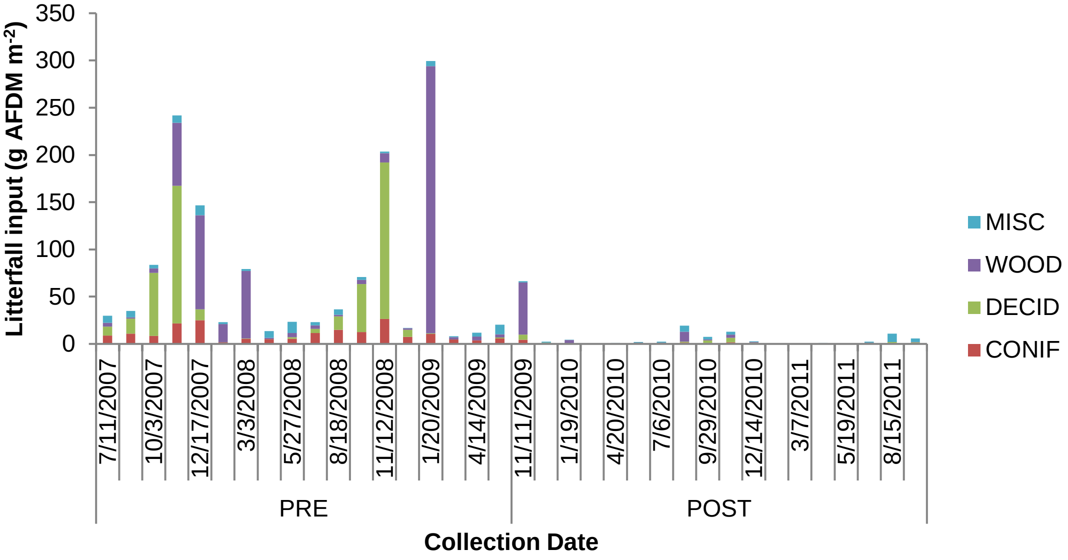
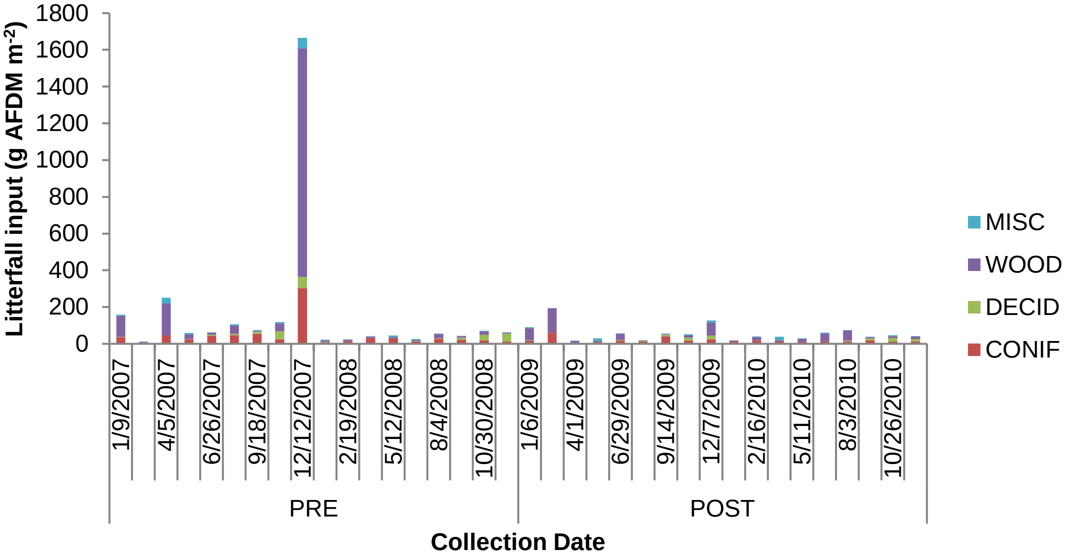


Figure 12–6. (continued) Litterfall inputs in grams ash-free dry mass per square meter (g AFDM m^{-2}) in the (c) OLYM-FP and (d) OLYM-0%.

(e) WIL1-REF



(f) WIL1-100%

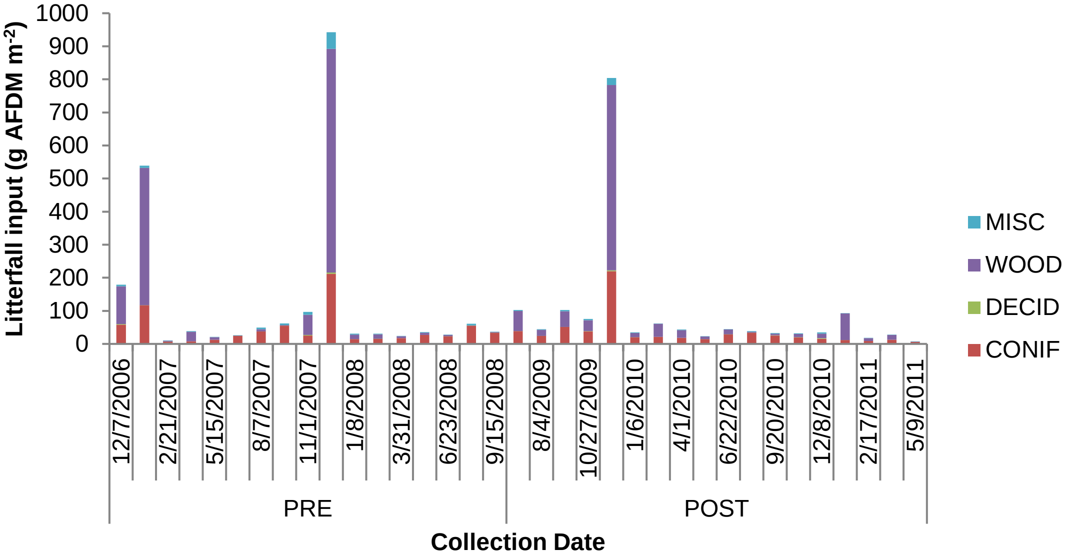
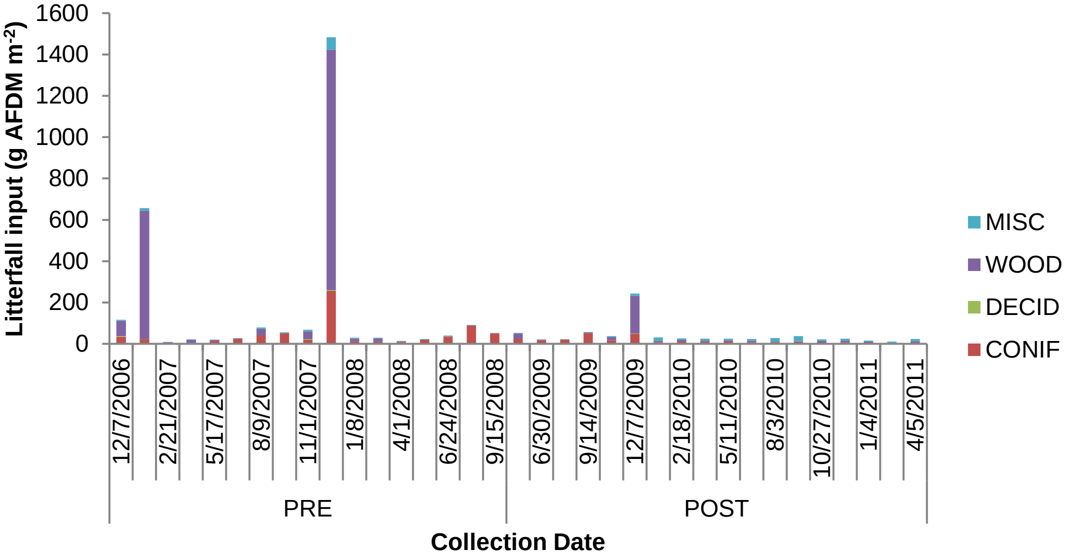


Figure 12–6. (continued) Litterfall inputs in grams ash-free dry mass per square meter (g AFDM m^{-2}) in the (e) WIL1-REF and (f) WIL1-100%.

(g) WIL1-FP



(h) WIL1-0%

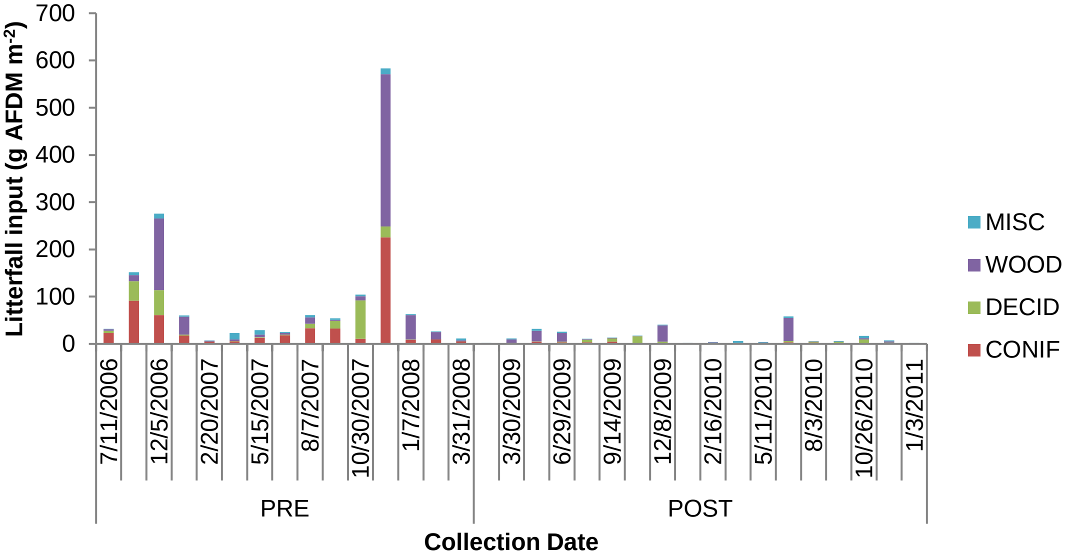


Figure 12-6. (continued) Litterfall inputs in grams ash-free dry mass per square meter (g AFDM m^{-2}) in the (g) WIL1-FP and (h) WIL1-0%.

12-5.2. DETRITUS EXPORT

The GLMM ANOVA showed evidence of a strong treatment x period interaction for TOTAL ($P = 0.0004$), CPOM ($P = 0.0018$), LEAF ($P = 0.0421$), WOOD ($P = 0.0011$), MISC ($P = 0.0119$), and FPOM ($P = 0.0046$) detritus export (**Table 12-8**). There was a moderate interaction effect for CONIF ($P = 0.0709$) detritus export, but no interaction effect for DECID ($P = 0.1021$) detritus export.

Post-hoc comparisons between the PRE and POST treatment periods showed an increase in TOTAL detritus export in the 100% treatment ($P = 0.0051$) and a decrease in the 0% treatment ($P = 0.0046$; **Table 12-9**). Likewise, there was an increase in CPOM, WOOD, MISC, and FPOM detritus export in the 100% treatment ($P <0.05$), but a decrease in the 0% treatment ($P <0.05$). LEAF, CONIF, and DECID detritus export also decreased in the 0% treatment ($P <0.05$).

Post-hoc comparisons between the reference and the buffer treatments showed a decrease in TOTAL, CPOM, WOOD, and FPOM detritus export in the 0% treatment relative to the reference and in the FP and 0% treatments relative to the 100% treatment ($P <0.05$) (**Table 12-9**; **Figure 12-7**). There was also a decrease in LEAF, CONIF, and DECID detritus export in the 0% treatment relative to the 100% treatment ($P <0.05$), and a decrease in MISC detritus export in the FP and 0% treatments relative to the 100% treatment ($P <0.05$).

Table 12-8. Results of the GLMM ANOVA for detritus export in grams ash-free dry mass per day (g AFDM day^{-1}) by treatment (TRMT), period (TRYR), and the treatment × period interaction (TRMT × TRYR).. Comparisons in **bold** print indicate P <0.05. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Effect	Num DF	Den DF	F-value	P-value
Total Detritus				
TRMT	3	3.69	0.97	0.4963
TRYR	1	240	0.26	0.6089
TRMT × TRYR	3	240	6.27	**0.0004**
Total CPOM Detritus				
TRMT	3	3.32	1.45	0.3732
TRYR	1	241	0.84	0.3604
TRMT × TRYR	3	241	5.15	**0.0018**
Total Leaf Detritus				
TRMT	3	2.69	0.9	0.5402
TRYR	1	243	0.17	0.6773
TRMT × TRYR	3	242	2.77	**0.0421**
Coniferous				
TRMT	3	3.86	0.15	0.9219
TRYR	1	243	0.11	0.7395
TRMT × TRYR	3	242	2.37	0.0709
Deciduous				
TRMT	3	3.62	0.11	0.9471
TRYR	1	241	2.4	0.1226
TRMT × TRYR	3	241	2.09	0.1021
Wood				
TRMT	3	2.9	2.09	0.2853
TRYR	1	242	1.15	0.2853
TRMT × TRYR	3	242	5.5	**0.0011**
Miscellaneous				
TRMT	3	3.65	1.69	0.3158
TRYR	1	240	0.88	0.3497
TRMT × TRYR	3	240	3.73	**0.0119**
FPOM				
TRMT	3	3.85	0.58	0.6615
TRYR	1	240	0.03	0.8697
TRMT × TRYR	3	240	4.46	**0.0046**

Table 12-9. Results of hypothesis tests described in section 12-4.4.3. for detritus export. Comparisons a, b, c, and d compare pre- and post-harvest detritus export for each treatment, and comparisons e, f, g, h, i, and j compare post-harvest changes in detritus export between the treatments. Comparisons in **bold** print indicate P <0.05. P-values were not adjusted for multiple comparison.

Comparison	Change	P-value	Lower 95% C.I.	Upper 95% C.I.	Change	P-value	Lower 95% C.I.	Upper 95% C.I.	Change	P-value	Lower 95% C.I.	Upper 95% C.I.	Change	P-value	Lower 95% C.I.	Upper 95% C.I.
	Total Detritus				**Total CPOM Detritus**				**Total Leaf Detritus**				**FPOM**			
a. REF-Pre vs. Post	0.0761	0.5464	−0.1721	0.3243	0.0105	0.9401	−0.2630	0.2839	−0.0072	0.9546	−0.2556	0.2413	0.1227	0.2508	−0.0873	0.3327
b. 100%-Pre vs. Post	0.3517	**0.0051**	0.1065	0.5969	0.3361	**0.0152**	0.0653	0.6069	0.2162	0.0890	−0.0332	0.4657	0.2603	**0.0141**	0.0530	0.4677
c. FP-Pre vs. Post	−0.2007	0.1120	−0.4485	0.0472	−0.2140	0.1238	−0.4871	0.0590	−0.0202	0.8725	−0.2684	0.2279	−0.1212	0.2562	−0.3308	0.0885
d. 0%-Pre vs. Post	−0.3554	**0.0046**	−0.6003	−0.1104	−0.3853	**0.0053**	−0.6551	−0.1156	−0.2936	**0.0189**	−0.5384	−0.0489	−0.2272	**0.0319**	−0.4344	−0.0199
e. REF vs. 100%	0.2756	0.1210	−0.0733	0.6245	0.3257	0.0968	−0.0592	0.7105	0.2234	0.2124	−0.1286	0.5755	0.1376	0.3592	−0.1575	0.4327
f. REF vs. FP	−0.2768	0.1214	−0.6275	0.0740	−0.2245	0.2536	−0.6109	0.1619	−0.0130	0.9417	−0.3642	0.3381	−0.2439	0.1068	−0.5406	0.0529
g. REF vs. 0%	−0.4315	**0.0155**	−0.7801	−0.0828	−0.3958	**0.0435**	−0.7799	−0.0117	−0.2864	0.1070	−0.6352	0.0623	−0.3499	**0.0203**	−0.6449	−0.0548
h. 100% vs. FP	−0.5524	**0.0020**	−0.9010	−0.2037	−0.5502	**0.0052**	−0.9347	−0.1656	−0.2365	0.1868	−0.5883	0.1153	−0.3815	**0.0114**	−0.6764	−0.0866
i. 100% vs. 0%	−0.7071	**<0.0001**	−1.0536	−0.3605	−0.7214	**0.0002**	−1.1037	−0.3392	−0.5099	**0.0044**	−0.8593	−0.1604	−0.4875	**0.0012**	−0.7807	−0.1943
j. FP vs. 0%	−0.1547	0.3826	−0.5031	0.1937	−0.1713	0.3802	−0.5551	0.2125	−0.2734	0.1236	−0.6219	0.0752	−0.1060	0.4795	−0.4008	0.1888
	Coniferous				**Deciduous**				**Wood**				**Miscellaneous**			
a. REF-Pre vs. Post	−0.0131	0.9078	−0.2347	0.2086	−0.0036	0.9751	−0.2326	0.2253	0.0449	0.7435	−0.2250	0.3148	−0.0027	0.9843	−0.2703	0.2650
b. 100%-Pre vs. Post	0.1701	0.1373	−0.0547	0.3949	0.0951	0.4089	−0.1314	0.3216	0.3123	**0.0227**	0.0440	0.5807	0.2774	**0.0399**	0.0129	0.5418
c. FP-Pre vs. Post	0.0164	0.8843	−0.2048	0.2375	−0.1747	0.1336	−0.4034	0.0540	−0.2354	0.0866	−0.5048	0.0341	−0.2299	0.0915	−0.4971	0.0374
d. 0%-Pre vs. Post	−0.2483	**0.0261**	−0.4668	−0.0298	−0.2746	**0.0173**	−0.5004	−0.0489	−0.4138	**0.0025**	−0.6800	-0.1476	−0.2978	**0.0273**	−0.5619	−0.0336
e. REF vs. 100%	0.1832	0.2542	−0.1325	0.4988	0.0988	0.5464	−0.2233	0.4208	0.2675	0.1676	−0.1131	0.6481	0.2800	0.1439	−0.0962	0.6563
f. REF vs. FP	0.0294	0.8534	−0.2837	0.3425	−0.1711	0.2987	−0.4947	0.1525	−0.2803	0.1491	−0.6617	0.1011	−0.2272	0.2379	−0.6054	0.1510
g. REF vs. 0%	−0.2353	0.1378	−0.5465	0.0760	−0.2710	0.0982	−0.5926	0.0506	−0.4587	**0.0179**	−0.8378	−0.0796	−0.2951	0.1234	−0.6712	0.0809
h. 100% vs. FP	−0.1538	0.3377	−0.4690	0.1615	−0.2699	0.0999	−0.5917	0.0520	−0.5477	**0.0049**	−0.9280	−0.1674	−0.5072	**0.0084**	−0.8832	−0.1313
i. 100% vs. 0%	−0.4184	**0.0091**	−0.7319	−0.1050	−0.3698	**0.0236**	−0.6896	−0.0500	−0.7261	**0.0002**	−1.1041	−0.3481	−0.5752	**0.0027**	−0.9489	−0.2014
j. FP vs. 0%	−0.2647	0.0949	−0.5756	0.0462	−0.0999	0.5409	−0.4213	0.2215	−0.1784	0.3544	−0.5572	0.2004	−0.0679	0.7221	−0.4437	0.3078

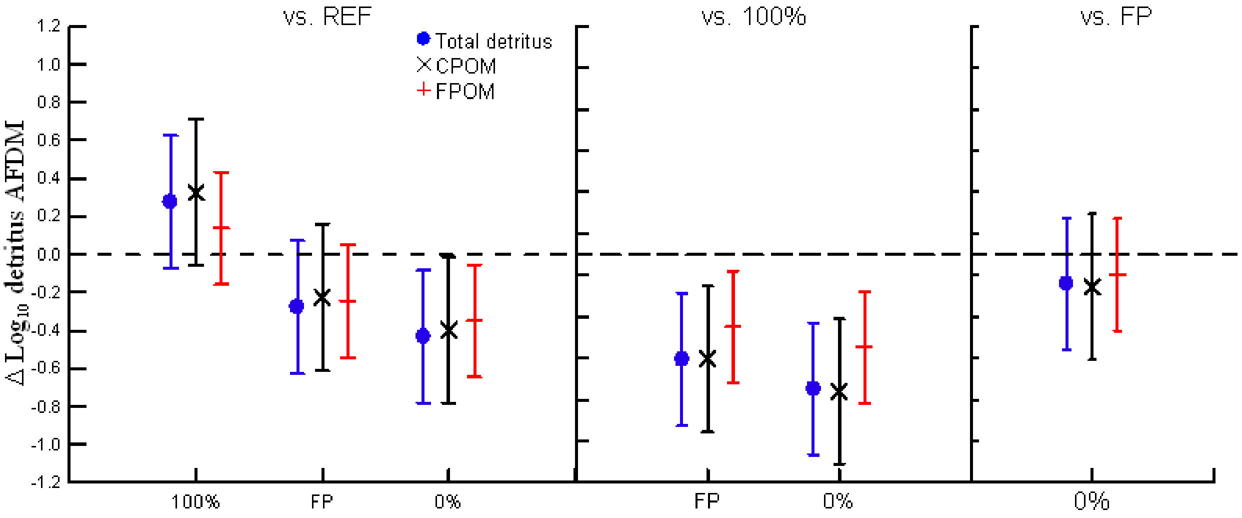


Figure 12-7. Results of the test of the six hypotheses for total, coarse particulate organic matter (CPOM; >1 mm), and fine particulate organic matter (FPOM; <1 mm) detritus export. The left panel shows the change in detritus export post-harvest in each treatment relative to the reference. The right two panels compare the change in detritus export among the three treatments. The error bars are 95% confidence intervals (CI). The horizontal dashed line equals zero (no change). Confidence intervals that do not cross the dashed line indicate the hypothesis is rejected at P <0.05.

Descriptive statistics for detritus export by component in g AFDM day^{-1} for each study site by treatment year are provided in **Appendix Table 12-D-1**. WOOD and FPOM detritus comprised most of the mean detritus export in g AFDM day^{-1} in the reference and in the 100% treatment during the PRE and POST periods (**Figure 12-8**). The proportion of WOOD detritus export increased in the reference during the POST period, but decreased in the 100% treatment. In contrast, the proportion of FPOM detritus export decreased in the reference during the POST period, but increased in the 100% treatment. MISC detritus export comprised a larger proportion of the detritus in the FP treatment during the PRE and POST periods and in the 0% treatment during the PRE period. While MISC decreased in proportion in the 0% treatment during the POST period, the proportion of FPOM increased. CONIF and DECID detritus consistently comprised a small proportion of the detritus exported from the reference and the buffer treatments. While the proportion of DECID detritus export increased in the 100% and 0% treatments and decreased in the FP treatment from the PRE to POST periods, there was no appreciable change in the proportion of CONIF detritus.

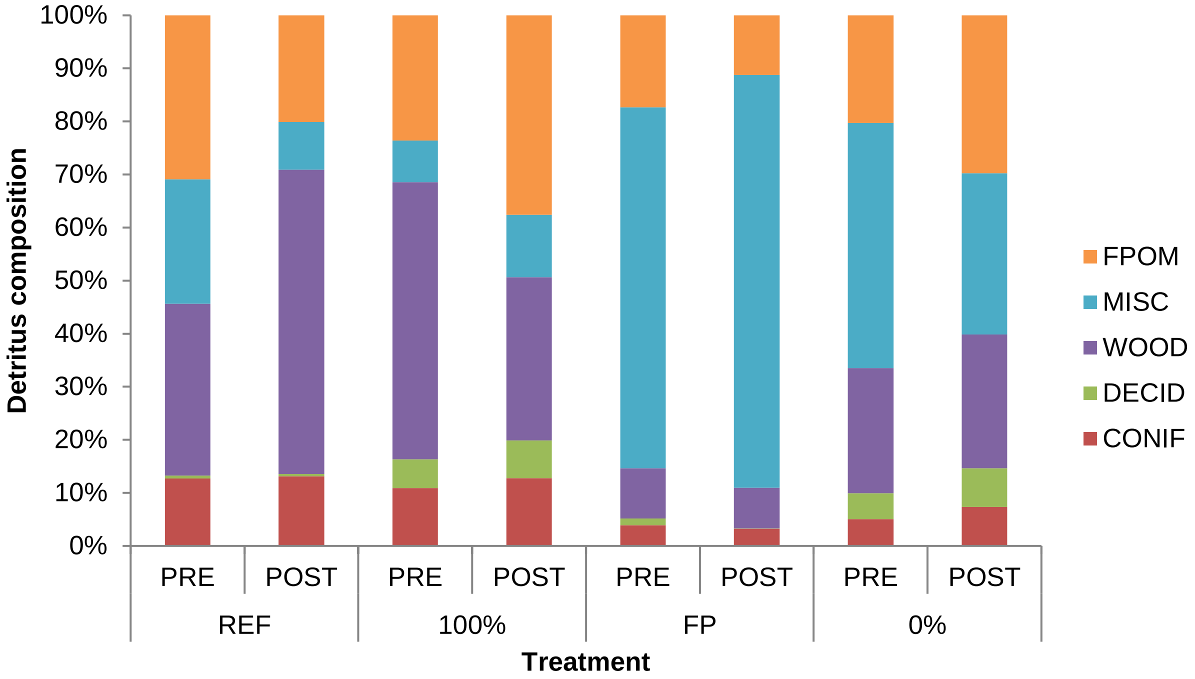
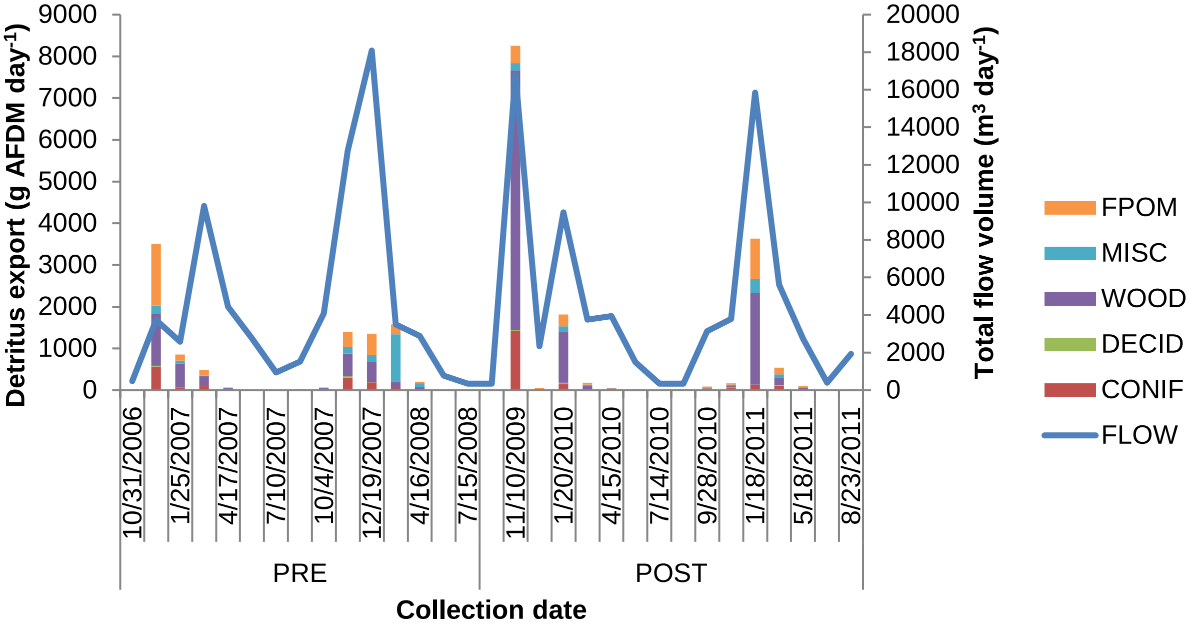


Figure 12-8. Composition of mean detritus export in percent (%) for the Type N Study references and buffer treatments by component and treatment period. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period; CONIF = coniferous detritus; DECID = deciduous detritus; WOOD = wood detritus; MISC = miscellaneous detritus; FPOM = fine particulate organic matter detritus.

Since detritus export is a function of flow we graphed detritus export in g AFDM day^{-1} relative to the total flow volume sampled per day during detritus collection for each of the study sites (**Figures 12-9a** through **12-9h**). Detritus export was highest in the fall and winter months for all eight sites when total flow volume sampled per day was also high. WOOD and FPOM detritus dominated most of the export collected during these periods in the OLYM-REF and OLYM-100% and MISC detritus in the OLYM-FP. WOOD, MISC, FPOM, and sometimes CONIF detritus dominated the export collected during the fall and winter in the OLYM-0% and in the Willapa 1 block sites. There were no consistent patterns in export between the PRE and POST periods as export responded more to the total flow volume sampled per day.

(a) OLYM-REF



(b) OLYM-100%

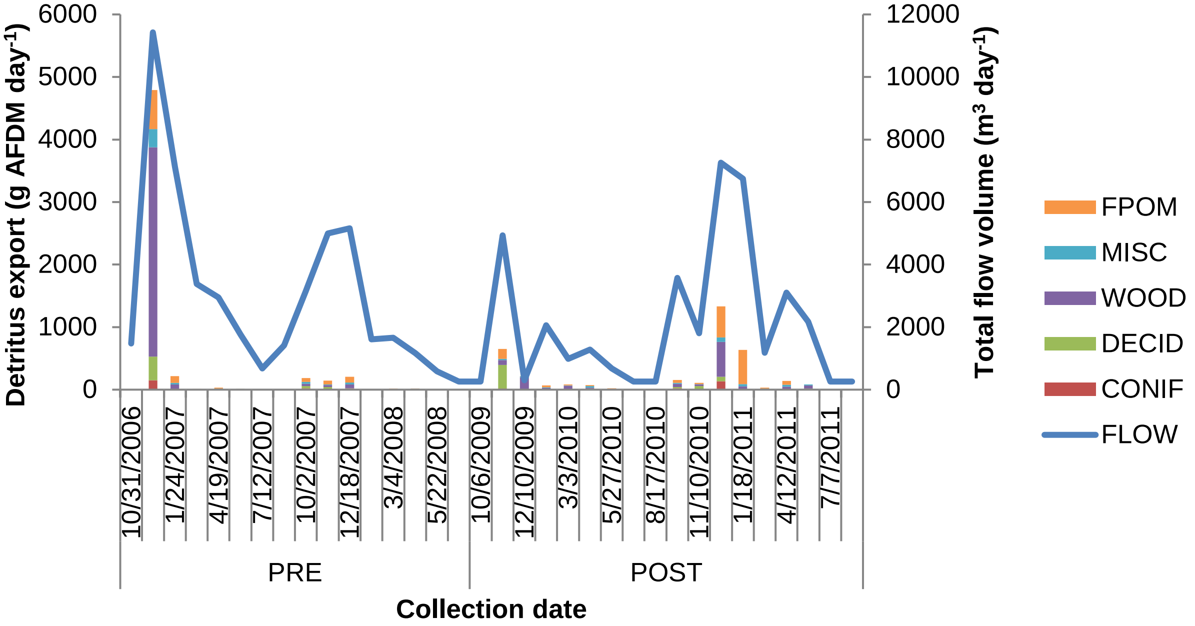
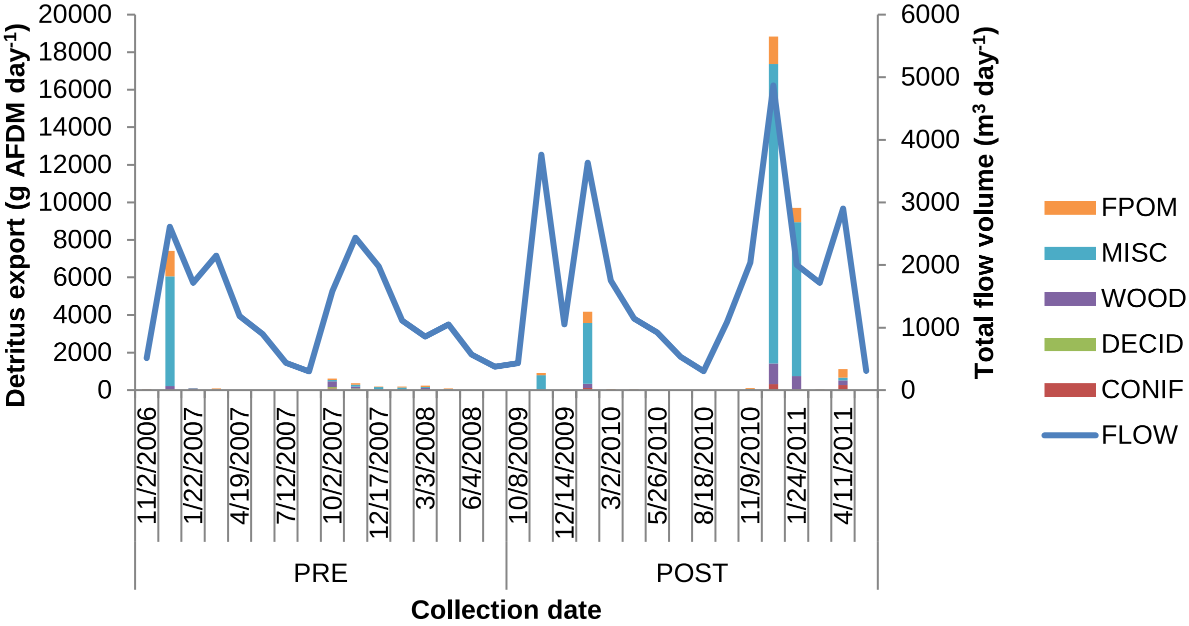


Figure 12-9. Detritus export in grams ash-free dry mass per day (g AFDM day^{-1}) and total flow volume per day in the (a) OLYM-REF and (b) OLYM-100%. OLYM = Olympic block; WIL1 = Willapa 1 block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period; CONIF = coniferous detritus; DECID = deciduous detritus; WOOD = wood detritus; MISC = miscellaneous detritus; FPOM = fine particulate organic matter detritus.

(c) OLYM-FP



(d) OLYM-0%

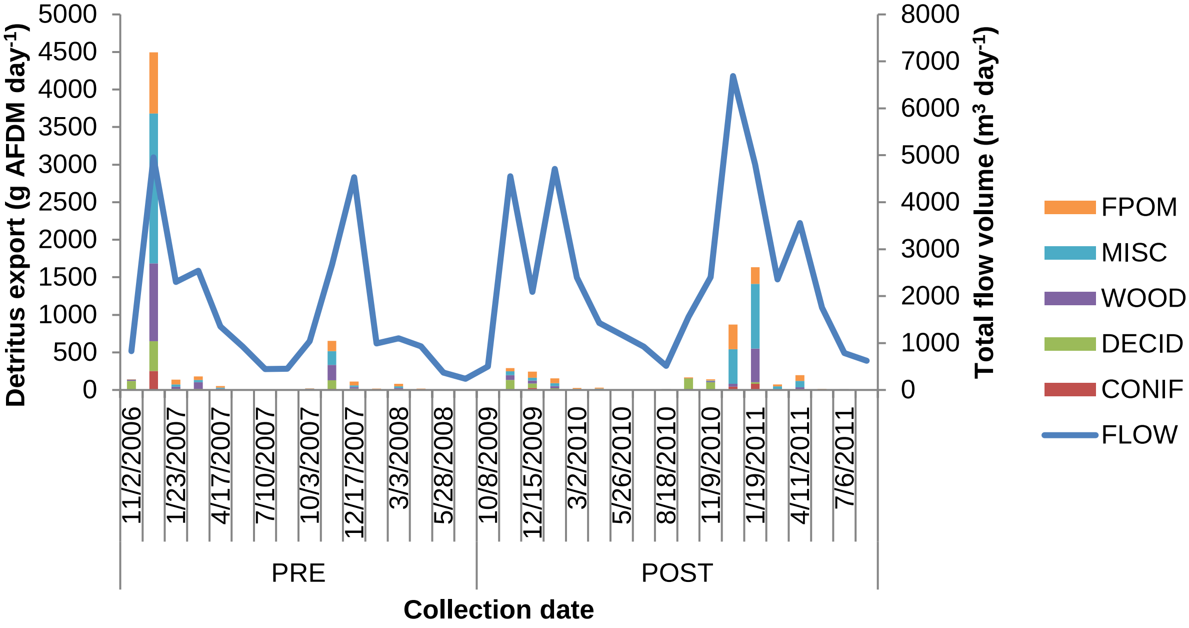
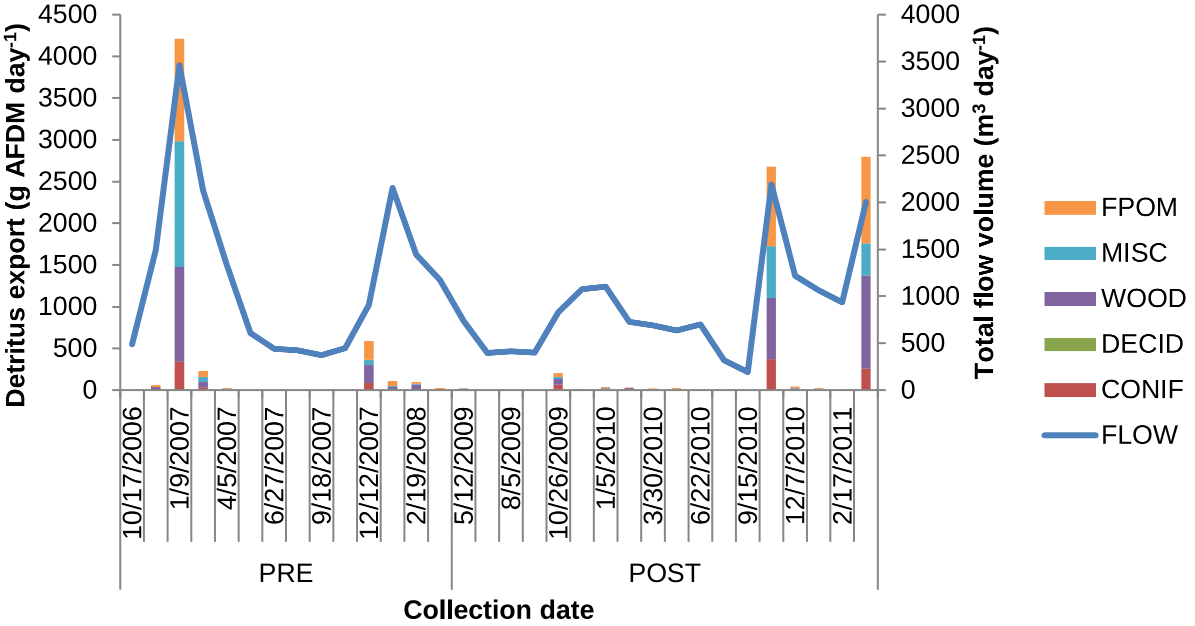


Figure 12-9. (continued) Detritus export in grams ash-free dry mass per day (g AFDM day^{-1}) and total flow volume per day in the (c) OLYM-FP and (d) OLYM-0%.

(e) WIL1-REF



(f) WIL1-100%

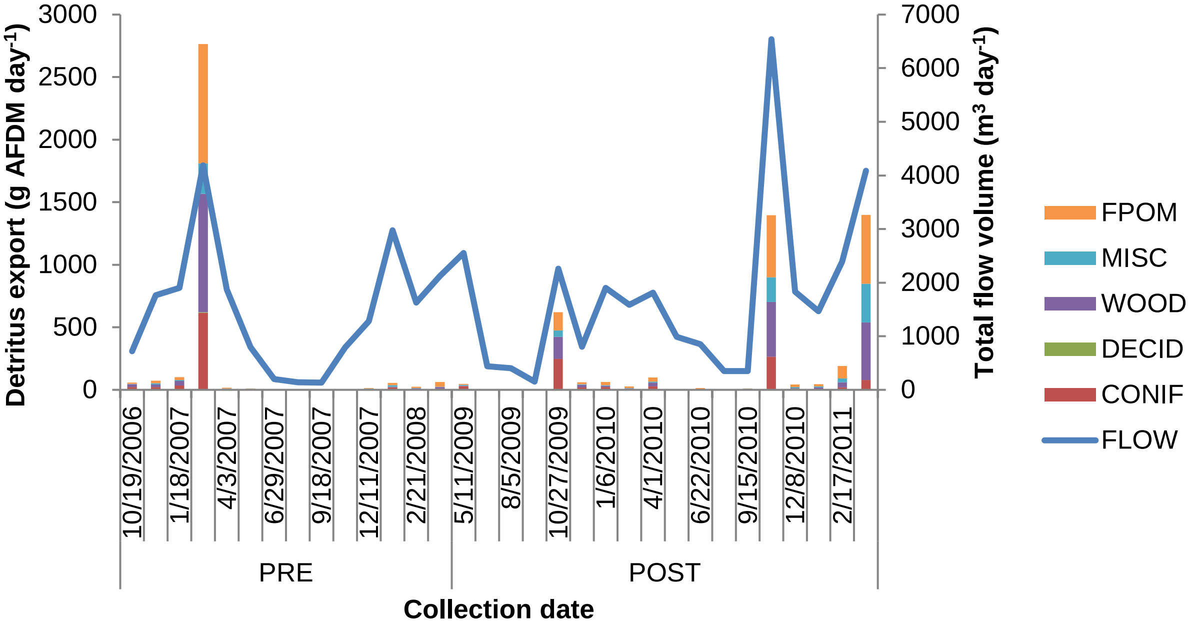
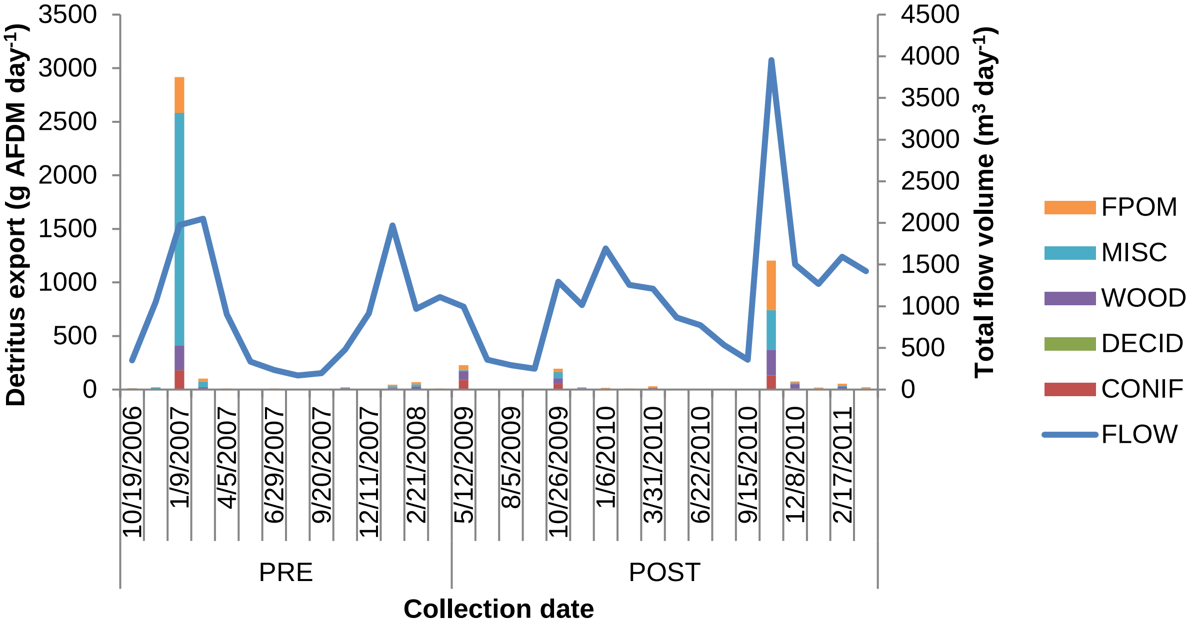


Figure 12-9. (continued) Detritus export in grams ash-free dry mass per day (g AFDM day^{-1}) and total flow volume per day in the (e) WIL1-REF and (f) WIL1-100%.

(g) WIL1-FP



(h) WIL1-0%

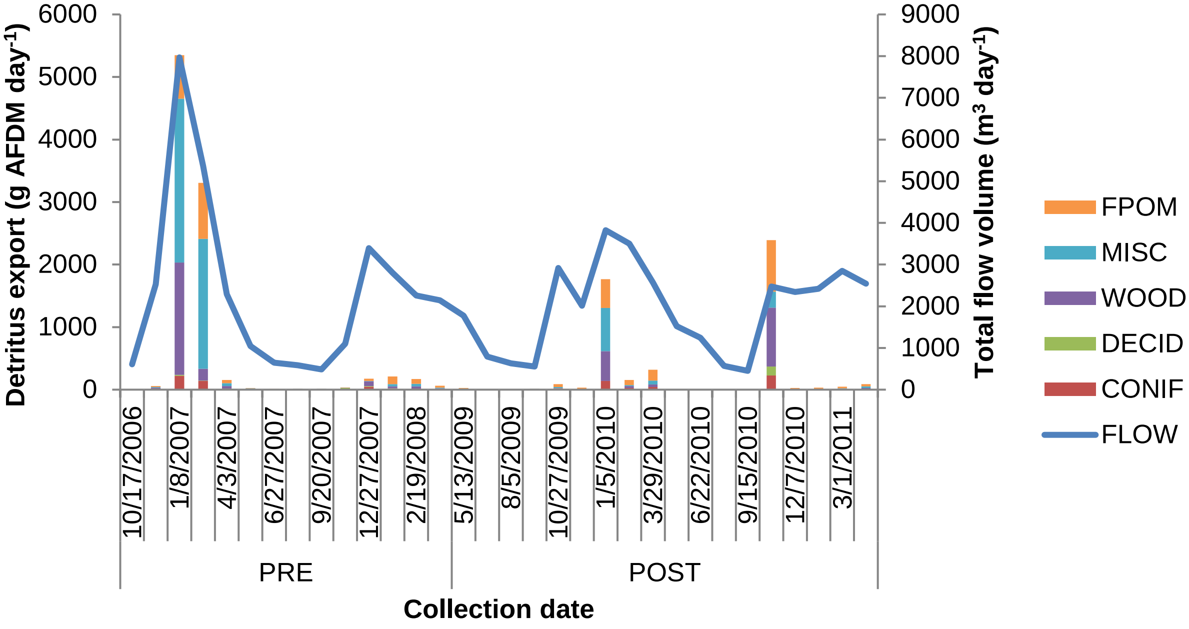


Figure 12-9. (continued) Detritus export in grams ash-free dry mass per day (g AFDM day^{-1}) and total flow volume per day in the (g) WIL1-FP and (h) WIL1-0%.

12-6. DISCUSSION

Changes in litterfall input and detritus export followed a similar pattern, with an increase in input and export in the 100% treatment, a decrease in the FP treatment, and an even larger decrease in the 0% treatment (**Figures 12-4** and **12-7**).

Litterfall input in the Type N Study sites was higher than that measured in a retrospective study comparing an unharvested reference site and a harvested site with a 30-m buffer in the Green River drainage in western Washington State (Grady 2001). In our study, POST period litterfall input averaged 1.16 g AFDM m^{-2} day^{-1} in the reference, 1.61 g AFDM m^{-2} day^{-1} in the 100% treatment, 1.22 g AFDM m^{-2} day^{-1} in the FP treatment, and 0.29 g AFDM m^{-2} day^{-1} in the 0% treatment. Grady (2001) observed post-harvest litterfall input rates of 0.403 g m^{-2} day^{-1} to 0.189 g m^{-2} day^{-1} between the first and second post-harvest year in a reference site, and from 0.164 g m^{-2} day^{-1} to 0.103 g m^{-2} day^{-1} between the first and second post-harvest years in a treatment site with an approximately 30-m buffer. While his study design was similar to ours, with paired traps located just outside the bankfull channel, he did not collect litterfall samples during the winter and early spring because of site inaccessibility.

Other regional studies of variability in litterfall inputs in relation to forest harvesting have found conflicting results. Bisson and colleagues (2013) measured a decrease ($P < 0.05$) in litterfall inputs during the spring, summer, and fall in western Washington headwater streams with no buffers and during the fall in their streams with patch buffers compared with unharvested reference streams. In a retrospective study of headwater tributaries of the Deschutes River in Washington State, Bilby and Bisson (1992) found that litterfall inputs to a stream flowing through a seven-year-old clearcut were lower ($P < 0.05$) than those to a stream flowing through an old-growth forest. An experimental study of timber harvest and riparian management zones on a tributary of the Bogachiel River in Washington State found a decrease ($P < 0.05$) in litterfall inputs in their clearcut treatment zone compared with their unharvested control zone in April, August, and October, but no difference ($P > 0.05$) in the late spring and early summer months (Martin *et al.* 1981). While Grady (2001) observed that his unharvested reference site had a greater amount of litterfall than his harvested site with an approximately 30-m buffer, he did not detect a significant difference ($P > 0.05$) between the two sites in his retrospective study of tributaries of the Green River in Washington State.

Detritus export was also higher in the Type N Study compared with a Before-After-Control-Impact (BACI) study of drift response to different harvest treatments in headwater streams of Southwest Alaska (Musslewhite and Wipfli 2004). We measured an average export rate of 489 g AFDM day^{-1}, and seasonal averages of 146 g AFDM day^{-1} in spring, 6 g AFDM day^{-1} in summer, 926 g AFDM day^{-1} in fall, and 778 g AFDM day^{-1} in winter. Musslewhite and Wipfli (2004) reported a mean detritus export rate of 18 g AFDM day^{-1}, with seasonal averages of 34 g AFDM day^{-1} in spring, 3 g AFDM day^{-1} in summer, and 17 g AFDM day^{-1} in fall. We measured higher detritus export rates in the fall and winter months, whereas Musslewhite and Wipfli (2004) measured higher export rates in the spring (they did not sample their streams in winter).

In contrast to our detritus export results, Musslewhite and Wipfli (2004) did not detect a relationship between detritus export and tree retention treatment. The streams in their study were comparable to ours with regard to mean bankfull width, but the basin size of the Type N Study sites was about the same or greater than that of their study sites. In addition, their study design differed from ours in that (a) their harvest treatments consisted of different tree retentions, such as aggregate and dispersed tree retentions, while ours consisted of different stream buffer lengths of equal widths including no buffer; (b) their sampling methodology included three samples collected over consecutive 24-hour periods (or two samples collected over consecutive 48-hour periods in their less accessible sites) only three times per year in the spring, summer, and fall, whereas we collected one sample at six-week intervals year-round; and (c) they approximated flows from a pipe embedded in a sandbag dam, while we used a continuous record of stream flow recorded by our hydrological measuring equipment.

In the Type N Study, there was a significant decrease in all of the litterfall components in the FP and/or 0% treatments in the post-harvest period, and a decrease in TOTAL, LEAF, and DECID litterfall in the 0% treatment relative to the reference, 100%, and/or FP treatments ($P < 0.05$). This was a result of the complete harvest of the riparian zone in the 0% treatment and in the unbuffered portion of the FP treatment. The decrease in TOTAL and LEAF litterfall inputs was driven mostly by the decrease in DECID litter, specifically in the 0% treatment, as indicated by the results of the GLMM and post-hoc comparisons. While this is mostly a result of the buffer treatment, it could also reflect the composition of the pre-harvest riparian stand in the 0% treatment sites, which consisted of a larger proportion of deciduous trees compared with the other sites in their respective blocks (**Table 12-1**).

Before harvest, deciduous trees and shrubs delivered large quantities of leaves to the streams in the fall, and the predominately coniferous stand delivered needles and wood to the streams in the fall and also year round (**Figure 12-6**). A decrease in litterfall input, at least in the FP and 0% treatments, was expected following timber harvest, and annual input remained low at the end of the second post-harvest year (**Appendix Table 12-C-1**). This decrease in litterfall input would reduce the amount of organic matter available for stream biota, at least in the short-term until plant communities are reestablished. The Type N Buffer Characteristics, Integrity, and Function Study found that during the first five years post-harvest, overhead cover provided by trees and shrubs increased from 12% to 37% and understory plant cover from 18% to 41% in Western Washington streams (Schuett-Hames *et al.* 2012). Kiffney and Richardson (2010) found that total litter input in their clearcut treatment was about 34% less than that of their buffer treatments eight years following harvest.

In the first few years after harvest, litterfall input may consist mostly of leaves from the growth of early successional deciduous trees and shrubs (Bilby and Bisson 1992). The leaves of deciduous trees such as red alder typically have a higher nutrient content than conifer needles and are readily consumed by microbes and invertebrates (Triska *et al.* 1982; Richardson *et al.* 2004; McKie and Malmqvist 2009; Kominoski *et al.* 2011). Riparian areas dominated by red alder have higher densities, biomass, and/or richness of both aquatic (Hernandez *et al.* 2005) and terrestrial (LeSage *et al.* 2005) invertebrates, and export more detritus and aquatic and terrestrial invertebrates than areas with little alder (Wipfli and Musslewhite 2004). A coniferous canopy would take longer to reestablish and, even with the planting of conifer trees by landowners, grow large enough to contribute needles and wood to the stream (Sweeney and Newbold 2014). Although the nutrient content of conifer needles is poor in comparison to deciduous leaves, the palatability of conifer needles is improved over several weeks to months by microbial conditioning, so that the conifer needles that are delivered in the fall are ready for consumption in the spring when temperatures have warmed and after most of the deciduous leaves have decomposed (Triska *et al.* 1982).

Wood may help to support the biota as the riparian zone is reestablished with deciduous trees and shrubs and conifers. While wood litterfall input decreased in the FP and 0% treatments in the post-harvest period, wood comprised most of the mean annual litterfall input in both the pre- and post-harvest periods in all of the treatments and dominated most of the large litterfall inputs outside of the fall leaf drop period. Substantial amounts of large and small woody debris were added to our study sites on three separate occasions: during an historic windstorm in the Coast Range, which resulted in the loss of a substantial number of trees in the Willapa Hills study sites (see Chapter 4 - *Unanticipated Disturbance*); following harvest, when logging slash was added disproportionately to all harvested stream basins (see Chapter 6 - *Wood Recruitment and Loading*); and after harvest, with the addition of windthrow in the 100% and FP treatments (see Chapter 6 - *Wood Recruitment and Loading*). We found an increase (treatment × period interaction $P < 0.05$) in wood cover and in total and functional (contributing to step formation, bank stability, and hydraulic roughness) small wood and large wood (see Chapter 6 - *Wood Recruitment and Loading*). These instream wood pieces may provide an important food source for xylophagous invertebrates and a substrate for microbial and periphyton growth for grazing invertebrates (Anderson *et al.* 1978; Triska *et al.* 1982; McKie and Cranston 2001).

Wood is also essential in establishing and maintaining channel structure and, in the process, retaining litter inputs that fall into the stream channel (Brookshire and Dwire 2003; Hoover *et al.* 2006). We measured an increase in pool length in all of the buffer treatments relative to the reference (see Chapter 11 - *Stream Channel Characteristics*). The slow water habitat may have created depositional areas (Anderson *et al.* 1978; Bilby and Likens 1980; Bilby 1981; Gregory *et al.* 1991; Wallace *et al.* 1995; Hetrick *et al.* 1998), which is evidenced by the measured increase in the proportion of sand post-harvest (see Chapter 11 - *Stream Channel Characteristics*). Litter inputs would also be retained within these depositional areas where the litter would then be consumed or fragmented by microbes and macroinvertebrates into finer particles (Bilby and Likens 1980; Swanson *et al.* 1982; Gregory *et al.* 1991; Hetrick *et al.* 1998).

In the Type N study, there was a significant post-harvest decrease in all of the detritus export components in the 0% treatment, and a decrease in all of the detritus export components in the FP and/or 0% treatments relative to the reference and/or 100% treatments. As with litterfall input, a decrease in detritus export in these two treatments may be a result of the complete harvest of the riparian zone in the 0% treatment and in the unbuffered portion of the FP treatment. Stream flow did increase post-harvest (see Chapter 8 - *Discharge*), but we cannot differentiate the effects of higher flows from changes in input. The data suggest that input is a factor because the response of litterfall input and detritus export are comparable.

In addition to the change in inputs, the decrease in detritus export could reflect the degree of retention in the stream from the increase in wood (see Chapter 6 - *Wood Recruitment and Loading*) and resultant changes in morphology (see Chapter 11 - *Stream Channel Characteristics*). Litter inputs would be retained behind wood and in slow-water habitats and available for processing by stream biota. We also observed increases in temperature (see Chapter 7 - *Stream Temperature and Cover*) and the export of total nitrogen and nitrate (see Chapter 9 - *Nutrient Export*), which may have influenced litter decomposition rates (Benfield *et al.* 2001;

Kreutzweiser *et al.* 2008; McKie and Malmqvist 2009). Depending on the degree of litter processing, most of the organic matter may have been exported as dissolved organic matter. Kiffney and colleagues (2000), for example, estimated that dissolved organic matter made up 69 to 86% of the total organic matter in stream water. Perhaps a measure of the amount of benthic detritus retained in the stream and exported as dissolved organic matter would have helped to interpret our results, but time and cost precluded such sampling.

There was an upward trend in litterfall input and detritus export in the 100% treatment following harvest, and the result was significant for TOTAL, CPOM, WOOD, MISC, and FPOM detritus export ($P < 0.05$). We expected that litterfall input would not change in the 100% treatment since input would be maintained by the remaining buffer. Kiffney and Richardson (2010), for example, found that total litter input in streams with 10- and 30-m buffers was similar to that of controls. We hypothesize that even though the entire length of the 100% treatment sites was buffered, the buffers may have been more vulnerable to wind and rain events, which may have caused an increase in the amount of litterfall input and a significant increase in the export of some of the detritus components. In addition, the increase in wetted and bankfull width in the 100% treatment (see Chapter 11 - *Stream Channel Characteristics*) may have led to the inundation, retention, and export of litter material normally stored on the floodplain.

The historic windstorm that brought hurricane-force winds and caused extensive windthrow in the Willapa Hills in early December 2007 (see Chapter 4 - *Unanticipated Disturbance*) may complicate the interpretation of the litterfall input results. Litterfall inputs followed a predictable pattern in the OLYM-REF, which was less affected by the windstorm (**Figure 12-6**). In the WIL1-REF, however, there was a large increase in litterfall input for the sample collected in the PRE period between 30 October 2007 and 12 December 2007. The windstorm may also be responsible for the large increase in litterfall input in the samples collected between 1 November 2007 and 13 December 2007 in the WIL1-100% and WIL1-FP and between 30 October 2007 and 27 December 2007 in the WIL1-0%. These increases in litterfall input may be skewing the data for the WIL1-REF and possibly the other Willapa 1 sites to show higher inputs in the PRE period and relatively lower inputs in the POST period.

Time and cost restrictions meant that we limited litterfall sampling to overhead litterfall and did not sample lateral ground inputs. This may have led to an underestimation of litter inputs into the stream channel. Benfield (1997), for example, estimated that lateral ground inputs may be as high as 40% to 55% of total litter input in conifer streams with steep slopes, while Hart and colleagues (2013) observed that lateral input relative to vertical input was highest in winter. In the Type N study sites, lateral inputs could have varied by treatment with the presence or absence of riparian shrubs and herbs, although Hart and colleagues (2013) did not find a relationship between understory density and lateral movement of litter in their study. In addition, the position of our traps on the stream bank may have overestimated litter inputs, as inputs tend to be higher in the riparian zone than over the stream channel (reviewed in Abelho 2001).

Error associated with the flow estimates for some of the study sites could have also introduced error into our detritus export estimates. We calculated flow from a flow-versus-stage height curve for the size of flume used in each flume site, or from a site-specific flow-versus-stage height curve developed for the two culvert sites (see Chapter 8 - *Discharge*). The flow-versus-stage relationships were good for the flume sites; however, those for the two culvert sites did not hold during summer low flows so we set flow at a fixed value when stage height was less than 3 L s^{-1}. This may have led to error in our detritus export estimates as flow per area was a random effect in the GLMM model, but the quantity of detritus exported during summer flows was low and the amount of error may be negligible.

During some of the fall, winter, and spring sampling efforts, stream flows were often high and forced us to shorten the duration of drift net deployment to an hour or less to avoid losing the net and its contents. We calculated AFDM day^{-1} by extrapolating the net deployment time to a 24-hour sampling period, which assumes that flow remained constant over the 24-hour period. This may have resulted in an underestimate or overestimate of detritus export if flows varied substantially over the 24 hours. In addition, we were not able to measure the larger storm events, and so may not have a full picture of the extent of detritus export from our study sites. Sample preservation in ethanol may have also led to an underestimation of detritus export through leaching and shrinking. While we could not find a study documenting changes in ash-free dry mass of detritus after preservation in ethanol, other studies have found substantial decreases in wet weight, dry weight, and/or ash-free dry mass of invertebrates preserved in ethanol (Howmiller 1972; Leuven *et al.* 1985).

Limitations of the litterfall input and detritus export methodology included a low number of replicates of each treatment and sample representativeness. We limited litterfall input and detritus export sampling to the eight study sites instrumented with hydrological monitoring equipment because of the time and cost required to collect and process samples and the need for flow quantification for interpreting detritus export. This left us with two replicates of each treatment. Litterfall stations were installed in four locations representative of buffered and unbuffered areas in the FP treatment and in comparable locations in the other treatments. While we believe that four stations were adequate coverage for each study site, we did not cover the entire stream channel and so may have missed some material. In addition, the limited size of our litterfall traps did not allow sampling of larger material such as branches and trunks from windthrown trees. Sampling of detritus export was limited to the downstream end of each study site mostly because we were interested in drift export to fish-bearing reaches and because of the need for flow quantification. That, however, does not give us a whole picture of detritus transport or storage within the Type N streams. Also, while most of our drift sampling locations were in close proximity to the treatment units, those of the OLYM-FP and OLYM-0% were not (**Table 12-5**), and the drift samples collected from those two basins may not have reflected the conditions in those treatment units. In spite of the low number of replicates and other limitations of the methodology, however, we observed a comparable response in litterfall input and detritus export and measured significant changes in both variables.

12-7. CONCLUSIONS

The Type N Study was designed to evaluate the response of a variety of riparian and instream processes to different stream buffer treatments pre- and post-harvest. We hypothesized that total litterfall inputs would decrease immediately post-harvest in the FP and 0% treatments, and that the proportion of deciduous litterfall input would increase post-harvest with the growth of early successional vegetation. We did see a significant decrease in total litterfall input post-harvest in the FP and 0% treatments. Deciduous litterfall input decreased significantly in the 0% treatment relative to the other treatments and decreased in proportion in the post-harvest period, which may indicate that the riparian vegetation community, at least in the 0% treatment, did not recover within the two years following timber harvest.

We also hypothesized that total detritus export would not change post-harvest. Total detritus export, however, increased significantly in the 100% treatment and decreased significantly in the 0% treatment. The proportion of the detritus export components was variable in response to harvest. We saw an increase in wood and changes in channel morphology that would enhance detritus retention, and retention may have played a role in the observed decrease in detritus export in some of the treatments, but we did not see the expected increase in the proportion of finer particulates with retention of coarser particulates by wood and processing by stream biota. Since the response of litterfall input and detritus export were similar, the change in input following timber harvest is the primary factor controlling export.

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APPENDICES

APPENDIX 12-A. LITTERFALL TRAP NET INSTALLATION DATA

Appendix Table 12-A-1. Litterfall trap net installation and collection date and number of sampling days for each study site, station, and treatment year. OLYM = Olympic Block; WIL1 = Willapa Block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; L1 = litterfall station 1; L2 = litterfall station 2; L3 = litterfall station 3; L4 = litterfall station 4; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2.

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-REF	L1	PRE1	12/13/2006	1/25/2007	43
			1/25/2007	3/5/2007	39
			3/5/2007	4/16/2007	42
			4/16/2007	5/24/2007	38
			7/10/2007	8/21/2007	42
			8/21/2007	10/4/2007	44
			10/4/2007	11/15/2007	42
			11/15/2007	12/19/2007	34
		PRE2	12/19/2007	3/5/2008	77
			3/5/2008	4/16/2008	42
			4/16/2008	5/21/2008	35
			5/21/2008	7/14/2008	54
			7/14/2008	8/18/2008	35
			8/18/2008	9/30/2008	43
			9/30/2008	11/12/2008	43
			11/12/2008	12/11/2008	29
		POST1	12/11/2008	4/13/2009	123
			4/13/2009	5/26/2009	43
			5/26/2009	7/7/2009	42
			7/7/2009	8/11/2009	35
			8/11/2009	10/5/2009	55
			10/5/2009	11/10/2009	36
			11/10/2009	12/9/2009	29
		POST2	12/9/2009	1/20/2010	42
			1/20/2010	2/23/2010	34
			2/23/2010	4/14/2010	50
			4/14/2010	5/25/2010	41
			5/25/2010	7/8/2010	44
			7/8/2010	8/16/2010	39
			8/16/2010	9/27/2010	42
			9/27/2010	11/8/2010	42
			11/8/2010	12/13/2010	35

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-REF	L2	PRE1	12/13/2006	1/25/2007	43
			1/25/2007	3/5/2007	39
			3/5/2007	4/16/2007	42
			4/16/2007	5/24/2007	38
			5/24/2007	7/10/2007	47
			7/10/2007	8/21/2007	42
			8/21/2007	10/4/2007	44
			10/4/2007	11/15/2007	42
			11/15/2007	12/19/2007	34
		PRE2	12/19/2007	3/5/2008	77
			3/5/2008	4/16/2008	42
			4/16/2008	5/21/2008	35
			5/21/2008	7/14/2008	54
			7/14/2008	8/18/2008	35
			8/18/2008	9/30/2008	43
			9/30/2008	11/12/2008	43
			11/12/2008	12/11/2008	29
		POST1	12/11/2008	4/13/2009	123
			4/13/2009	5/26/2009	43
			5/26/2009	7/7/2009	42
			7/7/2009	8/11/2009	35
			8/11/2009	10/5/2009	55
			10/5/2009	11/10/2009	36
			11/10/2009	12/9/2009	29
		POST2	12/9/2009	1/20/2010	42
			1/20/2010	2/23/2010	34
			2/23/2010	4/14/2010	50
			4/14/2010	5/25/2010	41
			5/25/2010	7/8/2010	44
			7/8/2010	8/16/2010	39
			8/16/2010	9/27/2010	42
			9/27/2010	11/8/2010	42
			11/8/2010	12/13/2010	35

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-REF	L3	PRE1	12/13/2006	1/25/2007	43
			3/5/2007	4/16/2007	42
			4/16/2007	5/24/2007	38
			5/24/2007	7/10/2007	47
			7/10/2007	8/21/2007	42
			8/21/2007	10/4/2007	44
			10/4/2007	11/15/2007	42
			11/15/2007	12/19/2007	34
		PRE2	12/19/2007	3/5/2008	77
			3/5/2008	4/16/2008	42
			4/16/2008	5/21/2008	35
			5/21/2008	7/14/2008	54
			7/14/2008	8/18/2008	35
			8/18/2008	9/30/2008	43
			9/30/2008	11/12/2008	43
			11/12/2008	12/11/2008	29
		POST1	12/11/2008	4/13/2009	123
			4/13/2009	5/26/2009	43
			5/26/2009	7/7/2009	42
			7/7/2009	8/11/2009	35
			8/11/2009	10/5/2009	55
			10/5/2009	11/10/2009	36
			11/10/2009	12/9/2009	29
		POST2	12/9/2009	1/20/2010	42
			1/20/2010	2/23/2010	34
			2/23/2010	4/14/2010	50
			4/14/2010	5/25/2010	41
			5/25/2010	7/8/2010	44
			7/8/2010	8/16/2010	39
			8/16/2010	9/27/2010	42
			9/27/2010	11/8/2010	42
			11/8/2010	12/13/2010	35

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-REF	L4	PRE1	12/19/2006	1/25/2007	37
			1/25/2007	3/5/2007	39
			3/5/2007	4/16/2007	42
			4/16/2007	5/24/2007	38
			5/24/2007	7/10/2007	47
			7/10/2007	8/21/2007	42
			8/21/2007	10/4/2007	44
			10/4/2007	11/15/2007	42
			11/15/2007	12/19/2007	34
		PRE2	12/19/2007	4/16/2008	119
			4/16/2008	5/21/2008	35
			5/21/2008	7/14/2008	54
			7/14/2008	8/18/2008	35
			8/18/2008	9/30/2008	43
			9/30/2008	11/12/2008	43
			11/12/2008	12/11/2008	29
		POST1	12/11/2008	4/13/2009	123
			4/13/2009	5/26/2009	43
			5/26/2009	7/7/2009	42
			7/7/2009	8/11/2009	35
			8/11/2009	10/5/2009	55
			10/5/2009	11/10/2009	36
			11/10/2009	12/9/2009	29
		POST2	12/9/2009	1/20/2010	42
			1/20/2010	2/23/2010	34
			2/23/2010	4/14/2010	50
			4/14/2010	5/25/2010	41
			5/25/2010	7/8/2010	44
			7/8/2010	8/16/2010	39
			8/16/2010	9/27/2010	42
			9/27/2010	11/8/2010	42
			11/8/2010	12/13/2010	35

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-100%	L1	PRE1	7/24/2006	12/12/2006	141
			12/12/2006	1/24/2007	43
			1/24/2007	3/6/2007	41
			3/6/2007	4/18/2007	43
			4/18/2007	5/22/2007	34
			5/22/2007	7/12/2007	51
			7/12/2007	8/21/2007	40
			8/21/2007	10/1/2007	41
		PRE2	10/1/2007	11/14/2007	44
			11/14/2007	12/18/2007	34
			12/18/2007	1/24/2008	37
			1/24/2008	3/4/2008	40
			3/4/2008	4/14/2008	41
			4/14/2008	5/21/2008	37
			5/21/2008	7/7/2008	47
			7/7/2008	8/20/2008	44
			8/20/2008	10/1/2008	42
		POST1	4/15/2009	5/26/2009	41
			5/26/2009	7/6/2009	41
			7/6/2009	8/12/2009	37
			8/12/2009	10/5/2009	54
			10/5/2009	11/10/2009	36
			11/10/2009	12/10/2009	30
			12/10/2009	1/25/2010	46
			1/25/2010	3/3/2010	37
			3/3/2010	4/12/2010	40
		POST2	4/12/2010	5/26/2010	44
			5/26/2010	7/7/2010	42
			7/7/2010	8/16/2010	40
			8/16/2010	9/27/2010	42
			9/27/2010	11/10/2010	44
			11/10/2010	12/13/2010	33
			12/13/2010	1/19/2011	37
			1/19/2011	4/12/2011	83

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-100%	L2	PRE1	7/24/2006	12/12/2006	141
			12/12/2006	1/24/2007	43
			1/24/2007	3/6/2007	41
			3/6/2007	4/18/2007	43
			4/18/2007	5/22/2007	34
			5/22/2007	7/12/2007	51
			7/12/2007	8/21/2007	40
			8/21/2007	10/1/2007	41
		PRE2	10/1/2007	11/14/2007	44
			11/14/2007	12/18/2007	34
			12/18/2007	1/24/2008	37
			1/24/2008	3/4/2008	40
			3/4/2008	4/14/2008	41
			4/14/2008	5/21/2008	37
			5/21/2008	7/7/2008	47
			7/7/2008	8/20/2008	44
			8/20/2008	10/1/2008	42
		POST1	4/15/2009	5/26/2009	41
			5/26/2009	7/6/2009	41
			7/6/2009	8/12/2009	37
			8/12/2009	10/5/2009	54
			10/5/2009	11/10/2009	36
			11/10/2009	12/10/2009	30
			12/10/2009	1/25/2010	46
			1/25/2010	3/3/2010	37
			3/3/2010	4/12/2010	40
		POST2	4/12/2010	5/26/2010	44
			5/26/2010	7/7/2010	42
			7/7/2010	8/16/2010	40
			8/16/2010	9/27/2010	42
			9/27/2010	11/10/2010	44
			11/10/2010	12/13/2010	33
			12/13/2010	1/19/2011	37
			1/19/2011	4/12/2011	83

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-100%	L3	PRE1	7/24/2006	12/12/2006	141
			12/12/2006	1/24/2007	43
			1/24/2007	3/6/2007	41
			3/6/2007	4/18/2007	43
			4/18/2007	5/22/2007	34
			5/22/2007	7/12/2007	51
			7/12/2007	8/21/2007	40
			8/21/2007	10/1/2007	41
		PRE2	10/1/2007	11/14/2007	44
			11/14/2007	12/18/2007	34
			12/18/2007	1/24/2008	37
			1/24/2008	3/4/2008	40
			3/4/2008	4/14/2008	41
			4/14/2008	5/21/2008	37
			5/21/2008	7/7/2008	47
			7/7/2008	8/20/2008	44
			8/20/2008	10/1/2008	42
		POST1	4/15/2009	5/26/2009	41
			5/26/2009	7/6/2009	41
			7/6/2009	8/12/2009	37
			8/12/2009	10/5/2009	54
			10/5/2009	11/10/2009	36
			11/10/2009	12/10/2009	30
			12/10/2009	1/25/2010	46
			1/25/2010	3/3/2010	37
			3/3/2010	4/12/2010	40
		POST2	4/12/2010	5/26/2010	44
			5/26/2010	7/7/2010	42
			7/7/2010	8/16/2010	40
			8/16/2010	9/27/2010	42
			9/27/2010	11/10/2010	44
			11/10/2010	12/13/2010	33
			12/13/2010	1/19/2011	37
			1/19/2011	4/12/2011	83

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-100%	L4	PRE1	7/12/2006	12/12/2006	153
			12/12/2006	1/24/2007	43
			1/24/2007	3/6/2007	41
			3/6/2007	4/18/2007	43
			4/18/2007	5/22/2007	34
			5/22/2007	7/12/2007	51
			7/12/2007	8/21/2007	40
			8/21/2007	10/1/2007	41
		PRE2	10/1/2007	11/14/2007	44
			11/14/2007	12/18/2007	34
			12/18/2007	1/24/2008	37
			1/24/2008	3/4/2008	40
			3/4/2008	4/14/2008	41
			4/14/2008	5/21/2008	37
			5/21/2008	7/7/2008	47
			7/7/2008	8/20/2008	44
			8/20/2008	10/1/2008	42
		POST1	4/15/2009	5/26/2009	41
			5/26/2009	7/6/2009	41
			7/6/2009	8/12/2009	37
			8/12/2009	10/5/2009	54
			10/5/2009	11/10/2009	36
			11/10/2009	12/10/2009	30
			12/10/2009	1/25/2010	46
			1/25/2010	3/3/2010	37
			3/3/2010	4/12/2010	40
		POST2	4/12/2010	5/26/2010	44
			5/26/2010	7/7/2010	42
			7/7/2010	8/16/2010	40
			8/16/2010	9/27/2010	42
			9/27/2010	11/10/2010	44
			11/10/2010	12/13/2010	33
			12/13/2010	1/19/2011	37

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-FP	L1	PRE1	7/17/2006	12/18/2006	154
			12/18/2006	1/22/2007	35
			1/22/2007	3/7/2007	44
			3/7/2007	4/18/2007	42
			4/18/2007	5/22/2007	34
			5/22/2007	7/12/2007	51
		PRE2	7/12/2007	8/22/2007	41
			8/22/2007	10/2/2007	41
			10/2/2007	11/13/2007	42
			11/13/2007	12/17/2007	34
			12/17/2007	1/22/2008	36
			1/22/2008	3/3/2008	41
			3/3/2008	4/17/2008	45
			4/17/2008	6/3/2008	47
			6/3/2008	7/14/2008	41
		POST1	12/3/2008	12/15/2008	12
			12/15/2008	3/3/2009	78
			3/3/2009	4/16/2009	44
			4/16/2009	6/1/2009	46
			6/1/2009	7/8/2009	37
			7/8/2009	8/12/2009	35
			8/12/2009	10/7/2009	56
			10/7/2009	11/11/2009	35
			11/11/2009	12/14/2009	33
		POST2	12/14/2009	1/19/2010	36
			1/19/2010	3/2/2010	42
			3/2/2010	4/19/2010	48
			4/19/2010	5/25/2010	36
			5/25/2010	7/7/2010	43
			7/7/2010	8/17/2010	41
			8/17/2010	9/29/2010	43
			9/29/2010	11/9/2010	41
			11/9/2010	12/14/2010	35

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-FP	L2	PRE1	7/17/2006	12/18/2006	154
			12/18/2006	1/22/2007	35
			1/22/2007	3/7/2007	44
			3/7/2007	4/18/2007	42
			4/18/2007	5/22/2007	34
			5/22/2007	7/12/2007	51
		PRE2	7/12/2007	8/22/2007	41
			8/22/2007	10/2/2007	41
			10/2/2007	11/13/2007	42
			11/13/2007	12/17/2007	34
			12/17/2007	1/22/2008	36
			1/22/2008	3/3/2008	41
			3/3/2008	4/17/2008	45
			4/17/2008	6/3/2008	47
			6/3/2008	7/14/2008	41
		POST1	12/3/2008	12/15/2008	12
			12/15/2008	1/21/2009	37
			1/21/2009	3/3/2009	41
			3/3/2009	4/16/2009	44
			4/16/2009	6/1/2009	46
			6/1/2009	7/8/2009	37
			7/8/2009	8/12/2009	35
			8/12/2009	10/7/2009	56
			10/7/2009	11/11/2009	35
			11/11/2009	12/14/2009	33
		POST2	12/14/2009	1/19/2010	36
			1/19/2010	3/2/2010	42
			3/2/2010	4/19/2010	48
			4/19/2010	5/25/2010	36
			5/25/2010	7/7/2010	43
			7/7/2010	8/17/2010	41
			8/17/2010	9/29/2010	43
			9/29/2010	11/9/2010	41
			11/9/2010	12/14/2010	35

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-FP	L3	PRE1	7/17/2006	12/18/2006	154
			12/18/2006	1/22/2007	35
			1/22/2007	3/7/2007	44
			3/7/2007	4/18/2007	42
			4/18/2007	5/22/2007	34
			5/22/2007	7/12/2007	51
		PRE2	7/12/2007	8/22/2007	41
			8/22/2007	10/2/2007	41
			10/2/2007	11/13/2007	42
			11/13/2007	12/17/2007	34
			12/17/2007	1/22/2008	36
			1/22/2008	3/3/2008	41
			3/3/2008	4/17/2008	45
			4/17/2008	6/3/2008	47
			6/3/2008	7/14/2008	41
		POST1	12/3/2008	12/15/2008	12
			12/15/2008	1/21/2009	37
			1/21/2009	3/3/2009	41
			3/3/2009	4/16/2009	44
			4/16/2009	6/1/2009	46
			6/1/2009	7/8/2009	37
			7/8/2009	8/12/2009	35
			8/12/2009	10/7/2009	56
			10/7/2009	11/11/2009	35
			11/11/2009	12/14/2009	33
		POST2	12/14/2009	1/19/2010	36
			1/19/2010	3/2/2010	42
			3/2/2010	4/19/2010	48
			4/19/2010	5/25/2010	36
			5/25/2010	7/7/2010	43
			7/7/2010	8/17/2010	41
			8/17/2010	9/29/2010	43
			9/29/2010	11/9/2010	41
			11/9/2010	12/14/2010	35

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-FP	L4	PRE1	7/17/2006	12/18/2006	154
			12/18/2006	1/22/2007	35
			1/22/2007	3/7/2007	44
			3/7/2007	4/18/2007	42
			4/18/2007	5/22/2007	34
			5/22/2007	7/12/2007	51
		PRE2	7/12/2007	8/22/2007	41
			8/22/2007	10/2/2007	41
			10/2/2007	11/13/2007	42
			11/13/2007	12/17/2007	34
			12/17/2007	1/22/2008	36
			1/22/2008	3/3/2008	41
			3/3/2008	4/17/2008	45
			4/17/2008	6/3/2008	47
			6/3/2008	7/14/2008	41
		POST1	12/3/2008	12/15/2008	12
			12/15/2008	1/21/2009	37
			1/21/2009	3/3/2009	41
			3/3/2009	4/16/2009	44
			4/16/2009	6/1/2009	46
			6/1/2009	7/8/2009	37
			7/8/2009	8/12/2009	35
			8/12/2009	10/7/2009	56
			10/7/2009	11/11/2009	35
			11/11/2009	12/14/2009	33
		POST2	12/14/2009	1/19/2010	36
			1/19/2010	3/2/2010	42
			3/2/2010	4/19/2010	48
			4/19/2010	5/25/2010	36
			5/25/2010	7/7/2010	43
			7/7/2010	8/17/2010	41
			8/17/2010	9/29/2010	43
			9/29/2010	11/9/2010	41
			11/9/2010	12/14/2010	35

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-0%	L1	PRE1	5/24/2007	7/11/2007	48
			7/11/2007	8/22/2007	42
			8/22/2007	10/3/2007	42
			10/3/2007	11/13/2007	41
			11/13/2007	12/17/2007	34
			12/17/2007	1/23/2008	37
			1/23/2008	3/3/2008	40
			3/3/2008	4/15/2008	43
			4/15/2008	5/27/2008	42
		PRE2	5/27/2008	7/7/2008	41
			7/7/2008	8/18/2008	42
			8/18/2008	9/30/2008	43
			9/30/2008	11/12/2008	43
			11/12/2008	12/11/2008	29
			12/11/2008	1/20/2009	40
			1/20/2009	3/4/2009	43
			3/4/2009	4/14/2009	41
			4/14/2009	5/28/2009	44
		POST1	9/30/2009	11/11/2009	42
			11/11/2009	12/15/2009	34
			12/15/2009	1/19/2010	35
			1/19/2010	3/2/2010	42
			3/2/2010	4/20/2010	49
			4/20/2010	5/25/2010	35
			5/25/2010	7/6/2010	42
			7/6/2010	8/17/2010	42
			8/17/2010	9/29/2010	43
		POST2	9/29/2010	11/9/2010	41
			11/9/2010	12/14/2010	35
			12/14/2010	1/19/2011	36
			1/19/2011	3/7/2011	47
			3/7/2011	4/11/2011	35
			4/11/2011	5/19/2011	38
			5/19/2011	7/5/2011	47
			7/5/2011	8/15/2011	41
			8/15/2011	9/28/2011	44

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-0%	L2	PRE1	5/24/2007	7/11/2007	48
			7/11/2007	8/22/2007	42
			8/22/2007	10/3/2007	42
			10/3/2007	11/13/2007	41
			11/13/2007	12/17/2007	34
			12/17/2007	1/23/2008	37
			1/23/2008	3/3/2008	40
			3/3/2008	4/15/2008	43
			4/15/2008	5/27/2008	42
		PRE2	5/27/2008	7/7/2008	41
			7/7/2008	8/18/2008	42
			8/18/2008	9/30/2008	43
			9/30/2008	11/12/2008	43
			11/12/2008	12/11/2008	29
			12/11/2008	1/20/2009	40
			1/20/2009	3/4/2009	43
			3/4/2009	4/14/2009	41
			4/14/2009	5/28/2009	44
		POST1	9/30/2009	11/11/2009	42
			11/11/2009	12/15/2009	34
			12/15/2009	1/19/2010	35
			1/19/2010	3/2/2010	42
			3/2/2010	4/20/2010	49
			4/20/2010	5/25/2010	35
			5/25/2010	7/6/2010	42
			7/6/2010	8/17/2010	42
			8/17/2010	9/29/2010	43
		POST2	9/29/2010	11/9/2010	41
			11/9/2010	12/14/2010	35
			12/14/2010	1/19/2011	36
			1/19/2011	3/7/2011	47
			3/7/2011	4/11/2011	35
			4/11/2011	5/19/2011	38
			5/19/2011	7/5/2011	47
			7/5/2011	8/15/2011	41
			8/15/2011	9/28/2011	44

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-0%	L3	PRE1	5/24/2007	7/11/2007	48
			7/11/2007	8/22/2007	42
			8/22/2007	10/3/2007	42
			10/3/2007	11/13/2007	41
			11/13/2007	12/17/2007	34
			12/17/2007	1/23/2008	37
			1/23/2008	3/3/2008	40
			3/3/2008	4/15/2008	43
			4/15/2008	5/27/2008	42
		PRE2	5/27/2008	7/7/2008	41
			7/7/2008	8/18/2008	42
			8/18/2008	9/30/2008	43
			9/30/2008	11/12/2008	43
			11/12/2008	12/11/2008	29
			12/11/2008	1/20/2009	40
			1/20/2009	3/4/2009	43
			3/4/2009	4/14/2009	41
			4/14/2009	5/28/2009	44
		POST1	9/30/2009	11/11/2009	42
			11/11/2009	12/15/2009	34
			12/15/2009	1/19/2010	35
			1/19/2010	3/2/2010	42
			3/2/2010	4/20/2010	49
			4/20/2010	5/25/2010	35
			5/25/2010	7/6/2010	42
			7/6/2010	8/17/2010	42
			8/17/2010	9/29/2010	43
		POST2	9/29/2010	11/9/2010	41
			11/9/2010	12/14/2010	35
			12/14/2010	1/19/2011	36
			1/19/2011	3/7/2011	47
			3/7/2011	4/11/2011	35
			4/11/2011	5/19/2011	38
			5/19/2011	7/5/2011	47
			7/5/2011	8/15/2011	41
			8/15/2011	9/28/2011	44

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
OLYM-0%	L4	PRE1	5/24/2007	7/11/2007	48
			7/11/2007	8/22/2007	42
			8/22/2007	10/3/2007	42
			10/3/2007	11/13/2007	41
			11/13/2007	12/17/2007	34
			12/17/2007	1/23/2008	37
			1/23/2008	3/3/2008	40
			3/3/2008	4/15/2008	43
			4/15/2008	5/27/2008	42
		PRE2	5/27/2008	7/7/2008	41
			7/7/2008	8/18/2008	42
			8/18/2008	9/30/2008	43
			9/30/2008	11/12/2008	43
			11/12/2008	12/11/2008	29
			12/11/2008	1/20/2009	40
			1/20/2009	3/4/2009	43
			3/4/2009	4/14/2009	41
			4/14/2009	5/28/2009	44
		POST1	9/30/2009	11/11/2009	42
			11/11/2009	12/15/2009	34
			12/15/2009	1/19/2010	35
			1/19/2010	3/2/2010	42
			3/2/2010	4/20/2010	49
			4/20/2010	5/25/2010	35
			5/25/2010	7/6/2010	42
			7/6/2010	8/17/2010	42
			8/17/2010	9/29/2010	43
		POST2	9/29/2010	11/9/2010	41
			11/9/2010	12/14/2010	35
			12/14/2010	1/19/2011	36
			1/19/2011	3/7/2011	47
			3/7/2011	4/11/2011	35
			4/11/2011	5/19/2011	38
			5/19/2011	7/5/2011	47
			7/5/2011	8/15/2011	41
			8/15/2011	9/28/2011	44

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-REF	L1	PRE1	12/5/2006	1/9/2007	35
			1/9/2007	2/21/2007	43
			2/21/2007	4/5/2007	43
			4/5/2007	5/17/2007	42
			5/17/2007	6/26/2007	40
			6/26/2007	8/9/2007	44
			8/9/2007	9/18/2007	40
			9/18/2007	10/30/2007	42
			10/30/2007	12/12/2007	43
		PRE2	12/12/2007	1/7/2008	26
			1/7/2008	2/19/2008	43
			2/19/2008	4/1/2008	42
			4/1/2008	5/12/2008	41
			5/12/2008	6/24/2008	43
			6/24/2008	8/4/2008	41
			8/4/2008	9/15/2008	42
			9/15/2008	10/30/2008	45
			10/30/2008	12/9/2008	40
		POST1	12/9/2008	1/6/2009	28
			1/6/2009	2/23/2009	48
			2/23/2009	4/1/2009	37
			4/1/2009	5/12/2009	41
			5/12/2009	6/29/2009	48
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/26/2009	42
			10/26/2009	12/7/2009	42
		POST2	12/7/2009	1/5/2010	29
			1/5/2010	2/16/2010	42
			2/16/2010	3/30/2010	42
			3/30/2010	5/11/2010	42
			5/11/2010	6/21/2010	41
			6/21/2010	8/3/2010	43
			8/3/2010	9/20/2010	48
			9/20/2010	10/26/2010	36
			10/26/2010	12/7/2010	42

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-REF	L2	PRE1	12/5/2006	1/9/2007	35
			1/9/2007	2/21/2007	43
			2/21/2007	4/5/2007	43
			4/5/2007	5/17/2007	42
			5/17/2007	6/26/2007	40
			6/26/2007	8/9/2007	44
			8/9/2007	9/18/2007	40
			9/18/2007	10/30/2007	42
			10/30/2007	12/12/2007	43
		PRE2	12/12/2007	1/7/2008	26
			1/7/2008	2/19/2008	43
			2/19/2008	4/1/2008	42
			4/1/2008	5/12/2008	41
			5/12/2008	6/24/2008	43
			6/24/2008	8/4/2008	41
			8/4/2008	9/15/2008	42
			9/15/2008	10/30/2008	45
			10/30/2008	12/9/2008	40
		POST1	12/9/2008	1/6/2009	28
			1/6/2009	2/23/2009	48
			2/23/2009	4/1/2009	37
			4/1/2009	5/12/2009	41
			5/12/2009	6/29/2009	48
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/26/2009	42
			10/26/2009	12/7/2009	42
		POST2	12/7/2009	1/5/2010	29
			1/5/2010	2/16/2010	42
			2/16/2010	3/30/2010	42
			3/30/2010	5/11/2010	42
			5/11/2010	6/21/2010	41
			6/21/2010	8/3/2010	43
			8/3/2010	9/20/2010	48
			9/20/2010	10/26/2010	36
			10/26/2010	12/7/2010	42

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-REF	L3	PRE1	12/5/2006	1/9/2007	35
			1/9/2007	2/21/2007	43
			2/21/2007	4/5/2007	43
			4/5/2007	5/17/2007	42
			5/17/2007	6/26/2007	40
			6/26/2007	8/9/2007	44
			8/9/2007	9/18/2007	40
			9/18/2007	10/30/2007	42
			10/30/2007	12/12/2007	43
		PRE2	12/12/2007	1/7/2008	26
			1/7/2008	2/19/2008	43
			2/19/2008	4/1/2008	42
			4/1/2008	5/12/2008	41
			5/12/2008	6/24/2008	43
			6/24/2008	8/4/2008	41
			8/4/2008	9/15/2008	42
			9/15/2008	10/30/2008	45
			10/30/2008	12/9/2008	40
		POST1	12/9/2008	1/6/2009	28
			1/6/2009	2/23/2009	48
			2/23/2009	4/1/2009	37
			4/1/2009	5/12/2009	41
			5/12/2009	6/29/2009	48
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/26/2009	42
			10/26/2009	12/7/2009	42
		POST2	12/7/2009	1/5/2010	29
			1/5/2010	2/16/2010	42
			2/16/2010	3/30/2010	42
			3/30/2010	5/11/2010	42
			5/11/2010	6/21/2010	41
			6/21/2010	8/3/2010	43
			8/3/2010	9/20/2010	48
			9/20/2010	10/26/2010	36
			10/26/2010	12/7/2010	42

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-REF	L4	PRE1	12/5/2006	1/9/2007	35
			1/9/2007	2/21/2007	43
			2/21/2007	4/5/2007	43
			4/5/2007	5/17/2007	42
			5/17/2007	6/26/2007	40
			6/26/2007	8/9/2007	44
			8/9/2007	9/18/2007	40
			9/18/2007	10/30/2007	42
			10/30/2007	12/12/2007	43
		PRE2	12/12/2007	1/7/2008	26
			1/7/2008	2/19/2008	43
			2/19/2008	4/1/2008	42
			4/1/2008	5/12/2008	41
			5/12/2008	6/24/2008	43
			6/24/2008	8/4/2008	41
			8/4/2008	9/15/2008	42
			9/15/2008	10/30/2008	45
			10/30/2008	12/9/2008	40
		POST1	12/9/2008	1/6/2009	28
			1/6/2009	2/23/2009	48
			2/23/2009	4/1/2009	37
			4/1/2009	5/12/2009	41
			5/12/2009	6/29/2009	48
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/26/2009	42
			10/26/2009	12/7/2009	42
		POST2	12/7/2009	1/5/2010	29
			1/5/2010	2/16/2010	42
			2/16/2010	3/30/2010	42
			3/30/2010	5/11/2010	42
			5/11/2010	6/21/2010	41
			6/21/2010	8/3/2010	43
			8/3/2010	9/20/2010	48
			9/20/2010	10/26/2010	36
			10/26/2010	12/7/2010	42

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-100%	L1	PRE1	10/12/2006	12/7/2006	56
			12/7/2006	1/18/2007	42
			1/18/2007	2/21/2007	34
			2/21/2007	4/3/2007	41
			4/3/2007	5/15/2007	42
			5/15/2007	6/29/2007	45
			6/29/2007	8/7/2007	39
			8/7/2007	9/18/2007	42
		PRE2	9/18/2007	11/1/2007	44
			11/1/2007	12/13/2007	42
			12/13/2007	1/8/2008	26
			1/8/2008	2/21/2008	44
			2/21/2008	3/31/2008	39
			3/31/2008	5/12/2008	42
			5/12/2008	6/23/2008	42
			6/23/2008	8/5/2008	43
			8/5/2008	9/15/2008	41
		POST1	4/27/2009	6/30/2009	64
			6/30/2009	8/4/2009	35
			8/4/2009	9/15/2009	42
			9/15/2009	10/27/2009	42
			10/27/2009	12/8/2009	42
			12/8/2009	1/6/2010	29
			1/6/2010	2/18/2010	43
			2/18/2010	4/1/2010	42
			4/1/2010	5/11/2010	40
		POST2	5/11/2010	6/22/2010	42
			6/22/2010	8/3/2010	42
			8/3/2010	9/20/2010	48
			9/20/2010	10/27/2010	37
			10/27/2010	12/8/2010	42
			12/8/2010	1/4/2011	27
			1/4/2011	2/17/2011	44
			2/17/2011	3/31/2011	42
			3/31/2011	5/9/2011	39

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-100%	L2	PRE1	10/12/2006	12/7/2006	56
			12/7/2006	1/18/2007	42
			1/18/2007	2/21/2007	34
			2/21/2007	4/3/2007	41
			4/3/2007	5/15/2007	42
			6/29/2007	8/7/2007	39
			8/7/2007	9/18/2007	42
		PRE2	9/18/2007	11/1/2007	44
			11/1/2007	12/13/2007	42
			12/13/2007	1/8/2008	26
			1/8/2008	2/21/2008	44
			2/21/2008	3/31/2008	39
			3/31/2008	5/12/2008	42
			5/12/2008	6/23/2008	42
			6/23/2008	8/5/2008	43
			8/5/2008	9/15/2008	41
		POST1	4/27/2009	6/30/2009	64
			6/30/2009	8/4/2009	35
			8/4/2009	9/15/2009	42
			9/15/2009	10/27/2009	42
			12/8/2009	1/6/2010	29
			1/6/2010	2/18/2010	43
			2/18/2010	4/1/2010	42
			4/1/2010	5/11/2010	40
		POST2	5/11/2010	6/22/2010	42
			6/22/2010	8/3/2010	42
			8/3/2010	9/20/2010	48
			9/20/2010	10/27/2010	37
			10/27/2010	12/8/2010	42
			12/8/2010	1/4/2011	27
			1/4/2011	2/17/2011	44
			2/17/2011	3/31/2011	42
			3/31/2011	5/9/2011	39

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-100%	L3	PRE1	10/12/2006	12/7/2006	56
			12/7/2006	1/18/2007	42
			1/18/2007	2/21/2007	34
			2/21/2007	4/3/2007	41
			4/3/2007	5/15/2007	42
			5/15/2007	6/29/2007	45
			6/29/2007	8/7/2007	39
			8/7/2007	9/18/2007	42
		PRE2	9/18/2007	11/1/2007	44
			11/1/2007	12/13/2007	42
			12/13/2007	1/8/2008	26
			1/8/2008	2/21/2008	44
			2/21/2008	3/31/2008	39
			3/31/2008	5/12/2008	42
			5/12/2008	6/23/2008	42
			6/23/2008	8/5/2008	43
			8/5/2008	9/15/2008	41
		POST1	4/27/2009	6/30/2009	64
			6/30/2009	8/4/2009	35
			8/4/2009	9/15/2009	42
			9/15/2009	10/27/2009	42
			10/27/2009	12/8/2009	42
			12/8/2009	1/6/2010	29
			1/6/2010	2/18/2010	43
			2/18/2010	3/31/2010	41
			3/31/2010	5/11/2010	41
		POST2	5/11/2010	6/22/2010	42
			6/22/2010	8/3/2010	42
			8/3/2010	9/20/2010	48
			9/20/2010	10/27/2010	37
			10/27/2010	12/8/2010	42
			12/8/2010	1/4/2011	27
			1/4/2011	2/17/2011	44
			2/17/2011	3/31/2011	42
			3/31/2011	5/9/2011	39

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-100%	L4	PRE1	10/12/2006	12/7/2006	56
			12/7/2006	1/18/2007	42
			1/18/2007	2/21/2007	34
			2/21/2007	4/3/2007	41
			4/3/2007	5/15/2007	42
			5/15/2007	6/29/2007	45
			6/29/2007	8/7/2007	39
			8/7/2007	9/18/2007	42
		PRE2	9/18/2007	11/1/2007	44
			11/1/2007	12/13/2007	42
			12/13/2007	1/8/2008	26
			1/8/2008	2/21/2008	44
			2/21/2008	3/31/2008	39
			3/31/2008	5/12/2008	42
			5/12/2008	6/23/2008	42
			6/23/2008	8/5/2008	43
			8/5/2008	9/15/2008	41
		POST1	5/11/2009	6/30/2009	50
			6/30/2009	8/4/2009	35
			8/4/2009	9/15/2009	42
			9/15/2009	10/27/2009	42
			10/27/2009	12/8/2009	42
			12/8/2009	1/6/2010	29
			1/6/2010	2/18/2010	43
			2/18/2010	4/1/2010	42
			4/1/2010	5/11/2010	40
		POST2	5/11/2010	6/22/2010	42
			6/22/2010	8/3/2010	42
			8/3/2010	9/20/2010	48
			9/20/2010	10/27/2010	37
			10/27/2010	12/8/2010	42
			12/8/2010	1/4/2011	27
			1/4/2011	2/17/2011	44
			2/17/2011	3/31/2011	42
			3/31/2011	5/9/2011	39

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-FP	L1	PRE1	10/12/2006	12/7/2006	56
			12/7/2006	1/18/2007	42
			1/18/2007	2/21/2007	34
			2/21/2007	4/5/2007	43
			4/5/2007	5/17/2007	42
			5/17/2007	6/29/2007	43
			6/29/2007	8/9/2007	41
			8/9/2007	9/19/2007	41
		PRE2	9/19/2007	11/1/2007	43
			11/1/2007	12/13/2007	42
			12/13/2007	1/8/2008	26
			1/8/2008	2/21/2008	44
			2/21/2008	4/1/2008	40
			4/1/2008	5/12/2008	41
			5/12/2008	6/24/2008	43
			6/24/2008	8/5/2008	42
			8/5/2008	9/15/2008	41
		POST1	3/10/2009	5/12/2009	63
			5/12/2009	6/29/2009	48
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/26/2009	42
			10/26/2009	12/7/2009	42
			12/7/2009	1/6/2010	30
			1/6/2010	2/18/2010	43
			2/18/2010	3/31/2010	41
		POST2	3/31/2010	5/11/2010	41
			5/11/2010	6/22/2010	42
			6/22/2010	8/3/2010	42
			8/3/2010	9/20/2010	48
			9/20/2010	10/27/2010	37
			10/27/2010	12/8/2010	42
			12/8/2010	1/4/2011	27
			1/4/2011	2/17/2011	44
			2/17/2011	4/5/2011	47

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-FP	L2	PRE1	10/12/2006	12/7/2006	56
			12/7/2006	1/18/2007	42
			1/18/2007	2/21/2007	34
			2/21/2007	4/5/2007	43
			4/5/2007	5/17/2007	42
			5/17/2007	6/29/2007	43
			6/29/2007	8/9/2007	41
			8/9/2007	9/19/2007	41
		PRE2	9/19/2007	11/1/2007	43
			11/1/2007	12/13/2007	42
			12/13/2007	1/8/2008	26
			1/8/2008	2/21/2008	44
			2/21/2008	4/1/2008	40
			4/1/2008	5/12/2008	41
			5/12/2008	6/24/2008	43
			6/24/2008	8/5/2008	42
			8/5/2008	9/15/2008	41
		POST1	3/10/2009	5/12/2009	63
			5/12/2009	6/29/2009	48
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/26/2009	42
			10/26/2009	12/7/2009	42
			12/7/2009	1/6/2010	30
			1/6/2010	2/18/2010	43
			2/18/2010	3/31/2010	41
		POST2	3/31/2010	5/11/2010	41
			5/11/2010	6/22/2010	42
			6/22/2010	8/3/2010	42
			8/3/2010	9/20/2010	48
			9/20/2010	10/27/2010	37
			10/27/2010	12/8/2010	42
			12/8/2010	1/4/2011	27
			1/4/2011	2/17/2011	44
			2/17/2011	4/5/2011	47

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-FP	L3	PRE1	10/12/2006	12/7/2006	56
			12/7/2006	1/18/2007	42
			1/18/2007	2/21/2007	34
			2/21/2007	4/5/2007	43
			4/5/2007	5/17/2007	42
			5/17/2007	6/29/2007	43
			6/29/2007	8/9/2007	41
			8/9/2007	9/19/2007	41
		PRE2	9/19/2007	11/1/2007	43
			11/1/2007	12/13/2007	42
			12/13/2007	1/8/2008	26
			1/8/2008	2/21/2008	44
			2/21/2008	4/1/2008	40
			4/1/2008	5/12/2008	41
			5/12/2008	6/24/2008	43
			6/24/2008	8/5/2008	42
			8/5/2008	9/15/2008	41
		POST1	3/10/2009	5/12/2009	63
			5/12/2009	6/30/2009	49
			6/30/2009	8/4/2009	35
			8/4/2009	9/14/2009	41
			9/14/2009	10/26/2009	42
			10/26/2009	12/7/2009	42
			12/7/2009	1/6/2010	30
			1/6/2010	2/18/2010	43
			2/18/2010	3/31/2010	41
		POST2	3/31/2010	5/11/2010	41
			5/11/2010	6/22/2010	42
			6/22/2010	8/3/2010	42
			8/3/2010	9/20/2010	48
			9/20/2010	10/27/2010	37
			10/27/2010	12/8/2010	42
			12/8/2010	1/4/2011	27
			1/4/2011	2/17/2011	44
			2/17/2011	4/5/2011	47

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-FP	L4	PRE1	10/12/2006	12/7/2006	56
			12/7/2006	1/18/2007	42
			1/18/2007	2/21/2007	34
			2/21/2007	4/5/2007	43
			4/5/2007	5/17/2007	42
			5/17/2007	6/29/2007	43
			6/29/2007	8/9/2007	41
			8/9/2007	9/19/2007	41
		PRE2	9/19/2007	11/1/2007	43
			11/1/2007	12/13/2007	42
			12/13/2007	1/8/2008	26
			1/8/2008	2/21/2008	44
			2/21/2008	4/1/2008	40
			4/1/2008	5/12/2008	41
			5/12/2008	6/24/2008	43
			6/24/2008	8/5/2008	42
			8/5/2008	9/15/2008	41
		POST1	3/10/2009	5/12/2009	63
			5/12/2009	6/29/2009	48
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/26/2009	42
			10/26/2009	12/7/2009	42
			12/7/2009	1/6/2010	30
			1/6/2010	2/18/2010	43
			2/18/2010	3/31/2010	41
		POST2	3/31/2010	5/11/2010	41
			5/11/2010	6/22/2010	42
			6/22/2010	8/3/2010	42
			8/3/2010	9/20/2010	48
			9/20/2010	10/27/2010	37
			10/27/2010	12/8/2010	42
			12/8/2010	1/4/2011	27
			1/4/2011	2/17/2011	44
			2/17/2011	4/5/2011	47

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-0%	L1	PRE1	6/14/2006	7/11/2006	27
			7/11/2006	10/11/2006	92
			10/11/2006	12/5/2006	55
			12/5/2006	1/8/2007	34
			1/8/2007	2/20/2007	43
			2/20/2007	4/3/2007	42
		PRE2	4/3/2007	5/15/2007	42
			5/15/2007	6/26/2007	42
			6/26/2007	8/7/2007	42
			8/7/2007	9/20/2007	44
			9/20/2007	10/30/2007	40
			10/30/2007	12/27/2007	58
			12/27/2007	1/7/2008	11
			1/7/2008	2/19/2008	43
			2/19/2008	3/31/2008	41
		POST1	1/26/2009	2/23/2009	28
			2/23/2009	3/30/2009	35
			3/30/2009	5/13/2009	44
			5/13/2009	6/29/2009	47
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/27/2009	43
			10/27/2009	12/8/2009	42
			12/8/2009	1/5/2010	28
		POST2	1/5/2010	2/16/2010	42
			2/16/2010	3/29/2010	41
			3/29/2010	5/11/2010	43
			5/11/2010	6/21/2010	41
			6/21/2010	8/3/2010	43
			8/3/2010	9/14/2010	42
			9/14/2010	10/26/2010	42
			10/26/2010	12/7/2010	42
			12/7/2010	1/3/2011	27

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-0%	L2	PRE1	6/5/2006	7/11/2006	36
			7/11/2006	10/11/2006	92
			10/11/2006	12/5/2006	55
			12/5/2006	1/8/2007	34
			1/8/2007	2/20/2007	43
			2/20/2007	4/3/2007	42
		PRE2	4/3/2007	5/15/2007	42
			5/15/2007	6/26/2007	42
			6/26/2007	8/7/2007	42
			8/7/2007	9/20/2007	44
			9/20/2007	10/30/2007	40
			10/30/2007	12/27/2007	58
			12/27/2007	1/7/2008	11
			1/7/2008	2/19/2008	43
			2/19/2008	3/31/2008	41
		POST1	1/26/2009	2/23/2009	28
			2/23/2009	3/30/2009	35
			3/30/2009	5/13/2009	44
			5/13/2009	6/29/2009	47
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/27/2009	43
			10/27/2009	12/8/2009	42
			12/8/2009	1/5/2010	28
		POST2	1/5/2010	2/16/2010	42
			2/16/2010	3/29/2010	41
			3/29/2010	5/11/2010	43
			5/11/2010	6/21/2010	41
			6/21/2010	8/3/2010	43
			8/3/2010	9/14/2010	42
			9/14/2010	10/26/2010	42
			10/26/2010	12/7/2010	42
			12/7/2010	1/3/2011	27

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-0%	L3	PRE1	6/5/2006	7/11/2006	36
			7/11/2006	10/11/2006	92
			10/11/2006	12/5/2006	55
			12/5/2006	1/8/2007	34
			1/8/2007	2/20/2007	43
			2/20/2007	4/3/2007	42
		PRE2	4/3/2007	5/15/2007	42
			5/15/2007	6/26/2007	42
			6/26/2007	8/7/2007	42
			8/7/2007	9/20/2007	44
			9/20/2007	10/30/2007	40
			10/30/2007	12/27/2007	58
			12/27/2007	1/7/2008	11
			1/7/2008	2/19/2008	43
			2/19/2008	3/31/2008	41
		POST1	1/26/2009	2/23/2009	28
			2/23/2009	3/30/2009	35
			3/30/2009	5/13/2009	44
			5/13/2009	6/29/2009	47
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/27/2009	43
			10/27/2009	12/8/2009	42
			12/8/2009	1/5/2010	28
		POST2	1/5/2010	2/16/2010	42
			2/16/2010	3/29/2010	41
			3/29/2010	5/11/2010	43
			5/11/2010	6/21/2010	41
			6/21/2010	8/3/2010	43
			8/3/2010	9/14/2010	42
			9/14/2010	10/26/2010	42
			10/26/2010	12/7/2010	42
			12/7/2010	1/3/2011	27

Appendix Table 12-A-1. (continued)

Study Site	Station	Treatment Year	Installation Date	Collection Date	Sampling Days
WIL1-0%	L4	PRE1	6/5/2006	7/11/2006	36
			7/11/2006	10/11/2006	92
			10/11/2006	12/5/2006	55
			12/5/2006	1/8/2007	34
			1/8/2007	2/20/2007	43
			2/20/2007	4/3/2007	42
		PRE2	4/3/2007	5/15/2007	42
			5/15/2007	6/26/2007	42
			6/26/2007	8/7/2007	42
			8/7/2007	9/20/2007	44
			9/20/2007	10/30/2007	40
			10/30/2007	12/27/2007	58
			12/27/2007	1/7/2008	11
			1/7/2008	2/19/2008	43
			2/19/2008	3/31/2008	41
		POST1	1/26/2009	2/23/2009	28
			2/23/2009	3/30/2009	35
			3/30/2009	5/13/2009	44
			5/13/2009	6/29/2009	47
			6/29/2009	8/4/2009	36
			8/4/2009	9/14/2009	41
			9/14/2009	10/27/2009	43
			10/27/2009	12/8/2009	42
			12/8/2009	1/5/2010	28
		POST2	1/5/2010	2/16/2010	42
			2/16/2010	3/29/2010	41
			3/29/2010	5/11/2010	43
			5/11/2010	6/21/2010	41
			6/21/2010	8/3/2010	43
			8/3/2010	9/14/2010	42
			9/14/2010	10/26/2010	42
			10/26/2010	12/7/2010	42
			12/7/2010	1/3/2011	27

APPENDIX 12-B. DRIFT COLLECTION LOGISTICS DATA

Appendix Table 12-B-1. Drift collection date, time, and deployment period for the Type N Study sites by treatment year. OLYM = Olympic Block; WIL1 = Willapa Block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2.

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
OLYM-REF	PRE1	10/30/2006	10:52	10/31/2006	11:56	1504
		12/13/2006	12:05	12/13/2006	12:35	30
		1/25/2007	9:20	1/25/2007	10:20	60
		3/5/2007	11:28	3/5/2007	12:28	60
		4/16/2007	10:18	4/17/2007	7:32	1274
		5/23/2007	9:30	5/24/2007	7:38	1328
		7/9/2007	10:50	7/10/2007	8:00	1270
	PRE2	8/20/2007	9:40	8/21/2007	11:50	1570
		10/4/2007	10:05	10/4/2007	11:05	60
		11/15/2007	10:35	11/15/2007	11:35	60
		12/19/2007	11:33	12/19/2007	12:33	60
		3/5/2008	13:50	3/5/2008	14:50	60
		4/16/2008	11:18	4/16/2008	12:18	60
		5/21/2008	9:47	5/22/2008	8:12	1345
		7/14/2008	13:34	7/15/2008	7:38	1084
	POST1	10/5/2009	11:30	10/6/2009	10:11	1361
		11/10/2009	10:29	11/10/2009	10:52	23
		12/9/2009	12:06	12/9/2009	13:10	64
		1/20/2010	11:02	1/20/2010	12:02	60
		2/23/2010	10:31	2/23/2010	11:31	60
		4/15/2010	8:27	4/15/2010	9:27	60
		5/25/2010	14:20	5/26/2010	10:47	1227
		7/13/2010	9:45	7/14/2010	9:49	1444
		8/16/2010	9:18	8/17/2010	15:26	1808
	POST2	9/27/2010	10:50	9/28/2010	10:04	1394
		11/8/2010	10:18	11/8/2010	11:18	60
		1/18/2011	11:50	1/18/2011	11:55	5
		4/13/2011	8:54	4/13/2011	9:41	47
		5/18/2011	8:49	5/18/2011	9:49	60
		7/5/2011	9:30	7/6/2011	11:58	1588
		8/22/2011	9:24	8/23/2011	8:40	1396

Appendix Table 12-B-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
OLYM-100%	PRE1	10/30/2006	13:36	10/31/2006	8:30	1134
		12/11/2006	15:19	12/11/2006	15:49	30
		1/24/2007	10:39	1/24/2007	11:39	60
		3/6/2007	9:43	3/6/2007	10:45	62
		4/18/2007	14:00	4/19/2007	7:15	1035
		5/21/2007	12:35	5/22/2007	7:30	1135
		7/11/2007	15:50	7/12/2007	7:55	965
	PRE2	8/20/2007	12:45	8/21/2007	7:20	1115
		10/1/2007	16:40	10/2/2007	7:25	885
		11/14/2007	8:45	11/14/2007	9:45	60
		12/18/2007	8:58	12/18/2007	9:58	60
		1/24/2008	9:08	1/24/2008	10:08	60
		3/4/2008	9:58	3/4/2008	10:58	60
		4/14/2008	10:27	4/14/2008	11:31	64
		5/21/2008	14:22	5/22/2008	11:10	1248
		7/7/2008	14:10	7/8/2008	7:12	1022
	POST1	10/5/2009	15:00	10/6/2009	7:22	982
		11/10/2009	15:50	11/10/2009	16:45	55
		12/10/2009	10:31	12/10/2009	11:31	60
		1/25/2010	10:15	1/25/2010	11:17	62
		3/3/2010	8:31	3/3/2010	9:41	70
		4/12/2010	9:06	4/12/2010	10:08	62
		5/26/2010	7:20	5/27/2010	6:59	1419
		7/6/2010	9:37	7/7/2010	9:14	1417
		8/16/2010	12:51	8/17/2010	7:20	1109
	POST2	9/27/2010	15:35	9/28/2010	7:33	958
		11/10/2010	10:14	11/10/2010	11:14	60
		12/13/2010	13:25	12/13/2010	14:29	64
		1/18/2011	16:53	1/18/2011	17:16	23
		2/28/2011	10:22	2/28/2011	11:22	60
		4/12/2011	9:02	4/12/2011	10:00	58
		5/18/2011	12:10	5/18/2011	13:10	60
		7/6/2011	14:37	7/7/2011	8:47	1090
		8/16/2011	14:05	8/17/2011	9:32	1167

Appendix Table 12-B-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
OLYM-FP	PRE1	11/1/2006	11:52	11/2/2006	11:55	1443
		12/18/2006	11:38	12/18/2006	12:38	60
		1/22/2007	11:11	1/22/2007	12:11	60
		3/7/2007	10:43	3/7/2007	11:45	62
		4/18/2007	10:08	4/19/2007	11:25	1517
		5/21/2007	9:20	5/22/2007	11:33	1573
		7/11/2007	12:50	7/12/2007	12:05	1395
	PRE2	8/22/2007	13:01	8/23/2007	11:38	1357
		10/1/2007	10:15	10/2/2007	11:54	1539
		11/13/2007	9:55	11/13/2007	10:55	60
		12/17/2007	14:10	12/17/2007	15:12	62
		1/22/2008	12:56	1/22/2008	13:56	60
		3/3/2008	11:41	3/3/2008	12:41	60
		4/17/2008	9:40	4/17/2008	10:40	60
		6/3/2008	9:46	6/4/2008	11:43	1557
		7/14/2008	10:27	7/15/2008	10:26	1439
	POST1	10/7/2009	11:54	10/8/2009	8:49	1255
		11/11/2009	12:35	11/11/2009	13:33	58
		12/14/2009	12:25	12/14/2009	13:50	85
		1/19/2010	13:53	1/19/2010	13:56	3
		3/2/2010	13:10	3/2/2010	14:28	78
		4/19/2010	8:42	4/19/2010	9:42	60
		5/25/2010	11:20	5/26/2010	15:07	1667
		7/7/2010	16:34	7/8/2010	15:47	1393
		8/17/2010	12:05	8/18/2010	9:30	1285
	POST2	9/30/2010	7:50	9/30/2010	8:52	62
		11/9/2010	11:59	11/9/2010	12:58	59
		12/14/2010	11:52	12/14/2010	11:59	7
		1/24/2011	10:27	1/24/2011	10:33	6
		3/7/2011	13:51	3/7/2011	15:31	100
		4/11/2011	10:18	4/11/2011	10:42	24
		8/15/2011	9:09	8/16/2011	11:00	1551

Appendix Table 12-B-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
OLYM-0%	PRE1	11/1/2006	15:30	11/2/2006	8:50	1040
		12/11/2006	10:40	12/11/2006	11:28	48
		1/23/2007	10:34	1/23/2007	11:34	60
		3/5/2007	13:56	3/5/2007	15:00	64
		4/16/2007	12:48	4/17/2007	9:38	1250
		5/23/2007	11:25	5/24/2007	9:58	1353
		7/9/2007	14:20	7/10/2007	11:02	1242
	PRE2	8/22/2007	10:43	8/23/2007	7:55	1272
		10/3/2007	9:35	10/3/2007	10:50	75
		11/13/2007	14:40	11/13/2007	15:40	60
		12/17/2007	10:58	12/17/2007	11:58	60
		1/23/2008	11:24	1/23/2008	12:24	60
		3/3/2008	14:51	3/3/2008	15:51	60
		4/15/2008	8:50	4/15/2008	9:57	67
		5/27/2008	13:34	5/28/2008	7:44	1090
		7/7/2008	10:44	7/8/2008	10:01	1397
	POST1	10/7/2009	14:30	10/8/2009	11:21	1251
		11/11/2009	8:34	11/11/2009	9:28	54
		12/15/2009	13:31	12/15/2009	14:30	59
		1/19/2010	10:21	1/19/2010	10:31	10
		3/2/2010	9:55	3/2/2010	10:55	60
		4/20/2010	9:21	4/20/2010	10:21	60
		5/25/2010	8:45	5/26/2010	13:39	1734
		7/6/2010	11:45	7/7/2010	8:28	1243
		8/17/2010	10:12	8/18/2010	7:53	1301
	POST2	9/29/2010	10:42	9/29/2010	11:44	62
		11/9/2010	9:52	11/9/2010	10:49	57
		12/14/2010	9:46	12/14/2010	9:55	9
		1/19/2011	12:45	1/19/2011	13:00	15
		3/7/2011	10:30	3/7/2011	11:30	60
		4/11/2011	12:18	4/11/2011	12:50	32
		5/19/2011	9:05	5/19/2011	10:01	56
		7/5/2011	12:32	7/6/2011	9:08	1236
		8/15/2011	11:33	8/16/2011	9:07	1294

Appendix Table 12-B-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
WIL1-REF	PRE1	10/16/2006	12:15	10/17/2006	12:10	1435
		12/4/2006	12:57	12/5/2006	12:54	1437
		1/9/2007	12:42	1/9/2007	13:42	60
		2/21/2007	10:12	2/22/2007	9:35	1403
		4/4/2007	8:45	4/5/2007	8:45	1440
	PRE2	5/16/2007	9:00	5/17/2007	8:35	1415
		6/26/2007	11:30	6/27/2007	11:40	1450
		8/8/2007	9:26	8/9/2007	9:15	1429
		9/17/2007	12:12	9/18/2007	11:49	1417
		10/29/2007	11:55	10/30/2007	11:49	1434
		12/12/2007	12:55	12/12/2007	13:57	62
		1/7/2008	14:23	1/7/2008	15:23	60
		2/19/2008	15:22	2/19/2008	16:22	60
		4/1/2008	9:49	4/1/2008	10:51	62
	POST1	5/11/2009	9:05	5/12/2009	8:17	1392
		6/29/2009	12:36	6/30/2009	10:18	1302
		8/4/2009	11:00	8/5/2009	6:59	1199
		9/15/2009	9:55	9/16/2009	8:32	1357
		10/26/2009	8:42	10/26/2009	9:42	60
		12/7/2009	9:32	12/7/2009	10:31	59
		1/5/2010	12:49	1/5/2010	13:49	60
		2/16/2010	12:40	2/16/2010	13:40	60
		3/30/2010	7:23	3/30/2010	8:28	65
	POST2	5/11/2010	10:05	5/12/2010	6:53	1248
		6/21/2010	13:18	6/22/2010	7:02	1064
		8/3/2010	11:10	8/4/2010	9:35	1345
		9/14/2010	14:33	9/15/2010	11:06	1233
		10/26/2010	12:18	10/26/2010	13:16	58
		12/7/2010	12:41	12/7/2010	13:42	61
		1/3/2011	12:26	1/3/2011	13:26	60
		2/17/2011	9:24	2/17/2011	10:25	61
		3/30/2011	10:08	3/30/2011	10:31	23

Appendix Table 12-B-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
WIL1-100%	PRE1	10/18/2006	10:20	10/19/2006	11:04	1484
		12/6/2006	13:26	12/7/2006	12:00	1354
		1/18/2007	10:19	1/18/2007	11:19	60
		2/20/2007	14:40	2/20/2007	15:20	40
		4/2/2007	12:08	4/3/2007	11:34	1406
	PRE2	5/14/2007	10:55	5/15/2007	11:05	1450
		6/28/2007	11:00	6/29/2007	10:17	1397
		8/6/2007	9:15	8/7/2007	8:40	1405
		9/17/2007	10:36	9/18/2007	7:26	1250
		10/31/2007	9:00	11/1/2007	8:36	1416
		12/11/2007	11:02	12/11/2007	12:02	60
		1/8/2008	9:15	1/8/2008	10:16	61
		2/21/2008	9:44	2/21/2008	10:44	60
		3/31/2008	14:02	3/31/2008	15:02	60
	POST1	5/11/2009	10:05	5/11/2009	11:05	60
		6/30/2009	7:04	7/1/2009	6:42	1418
		8/4/2009	13:07	8/5/2009	9:00	1193
		9/15/2009	7:12	9/16/2009	8:21	1509
		10/27/2009	12:18	10/27/2009	13:18	60
		12/8/2009	12:48	12/8/2009	13:47	59
		1/6/2010	11:13	1/6/2010	12:12	59
		2/18/2010	11:54	2/18/2010	13:02	68
		4/1/2010	8:40	4/1/2010	9:44	64
	POST2	5/11/2010	12:45	5/12/2010	9:03	1218
		6/21/2010	14:48	6/22/2010	9:21	1113
		8/3/2010	13:38	8/4/2010	12:05	1347
		9/14/2010	16:05	9/15/2010	14:34	1349
		10/27/2010	10:30	10/27/2010	11:30	60
		12/8/2010	11:17	12/8/2010	12:17	60
		1/4/2011	11:34	1/4/2011	12:34	60
		2/17/2011	12:49	2/17/2011	13:50	61
		3/30/2011	8:26	3/30/2011	8:44	18

Appendix Table 12-B-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
WIL1-FP	PRE1	10/18/2006	9:24	10/19/2006	8:50	1406
		12/6/2006	9:55	12/7/2006	10:13	1458
		1/9/2007	13:38	1/9/2007	14:38	60
		2/21/2007	13:22	2/22/2007	11:05	1303
		4/4/2007	12:15	4/5/2007	10:25	1330
	PRE2	5/16/2007	10:55	5/17/2007	10:15	1400
		6/28/2007	9:10	6/29/2007	8:43	1413
		8/8/2007	12:30	8/9/2007	11:54	1404
		9/19/2007	11:19	9/20/2007	8:30	1271
		10/31/2007	10:15	11/1/2007	9:57	1422
		12/11/2007	13:45	12/11/2007	14:50	65
		1/8/2008	12:32	1/8/2008	13:32	60
		2/21/2008	13:20	2/21/2008	14:20	60
		4/1/2008	7:33	4/1/2008	8:33	60
	POST1	5/12/2009	9:56	5/12/2009	10:53	57
		6/29/2009	13:49	6/30/2009	8:59	1150
		8/4/2009	12:11	8/5/2009	8:00	1189
		9/14/2009	13:57	9/15/2009	8:44	1127
		10/26/2009	11:20	10/26/2009	12:20	60
		12/7/2009	11:58	12/7/2009	12:59	61
		1/6/2010	9:15	1/6/2010	10:15	60
		2/18/2010	9:17	2/18/2010	10:17	60
		3/31/2010	10:14	3/31/2010	11:14	60
	POST2	5/11/2010	11:53	5/12/2010	7:59	1206
		6/21/2010	14:14	6/22/2010	8:11	1077
		8/3/2010	12:37	8/4/2010	11:21	1364
		9/14/2010	15:36	9/15/2010	13:04	1288
		10/27/2010	8:30	10/27/2010	9:30	60
		12/8/2010	9:35	12/8/2010	10:35	60
		1/4/2011	10:06	1/4/2011	11:06	60
		2/17/2011	11:10	2/17/2011	12:10	60
		3/29/2011	13:05	3/29/2011	14:05	60

Appendix Table 12-B-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
WIL1-0%	PRE1	10/16/2006	9:20	10/17/2006	8:20	1380
		12/4/2006	10:10	12/5/2006	9:17	1387
		1/8/2007	10:36	1/8/2007	11:36	60
		2/20/2007	11:50	2/20/2007	12:45	55
		4/2/2007	8:19	4/3/2007	8:12	1433
	PRE2	5/14/2007	8:24	5/15/2007	8:00	1416
		6/26/2007	8:33	6/27/2007	8:08	1415
		8/6/2007	11:40	8/7/2007	12:53	1513
		9/19/2007	13:38	9/20/2007	11:41	1323
		10/29/2007	8:41	10/30/2007	8:20	1419
		12/27/2007	10:32	12/27/2007	11:32	60
		1/7/2008	9:41	1/7/2008	10:41	60
		2/19/2008	10:52	2/19/2008	11:52	60
		3/31/2008	9:19	3/31/2008	10:47	88
	POST1	5/13/2009	8:08	5/13/2009	9:05	57
		6/29/2009	8:12	6/30/2009	12:28	1696
		8/4/2009	8:09	8/5/2009	11:29	1640
		9/14/2009	8:34	9/15/2009	12:21	1667
		10/27/2009	8:15	10/27/2009	9:15	60
		12/8/2009	9:24	12/8/2009	10:23	59
		1/5/2010	9:13	1/5/2010	10:16	63
		2/16/2010	9:04	2/16/2010	10:05	61
		3/29/2010	8:46	3/29/2010	9:46	60
	POST2	5/11/2010	7:53	5/12/2010	11:07	1634
		6/21/2010	10:29	6/22/2010	12:10	1541
		8/3/2010	8:24	8/4/2010	7:26	1382
		9/14/2010	11:13	9/15/2010	7:56	1243
		10/26/2010	8:21	10/26/2010	9:11	50
		12/7/2010	9:21	12/7/2010	10:22	61
		1/3/2011	9:23	1/3/2011	10:23	60
		3/1/2011	9:30	3/1/2011	10:30	58
		3/29/2011	8:30	3/29/2011	9:17	47

APPENDIX 12-C. ANNUAL LITTERFALL INPUT DATA

Appendix Table 12-C-1. Annual litterfall input in grams ash-free dry mass per square meter per day (g AFDM m^{-2} day^{-1}) for the Type N Study sites by station, component, and treatment year. OLYM = Olympic block; WIL1 = Willapa 1 block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2; TOTAL = total litterfall; LEAF = total leaf litterfall (CONIF + DECID); CONIF = coniferous litterfall; DECID = deciduous litterfall; WOOD = wood litterfall; MISC = miscellaneous litterfall.

Component	**Treatment Year**	**OLYM-REF**				**OLYM-100%**			
		L1	**L2**	**L3**	**L4**	**L1**	**L2**	**L3**	**L4**
TOTAL	PRE1	1.45	0.85	0.34	1.36	1.21	1.51	0.38	0.99
	PRE2	0.76	0.69	0.31	1.22	0.73	1.29	4.15	0.83
	POST1	0.64	1.47	0.50	1.24	0.52	2.31	1.30	1.38
	POST2	0.47	1.16	0.31	1.36	0.51	1.87	1.11	1.53
LEAF	PRE1	0.37	0.38	0.29	0.95	0.86	0.91	0.32	0.81
	PRE2	0.54	0.38	0.27	0.90	0.57	0.81	0.61	0.53
	POST1	0.35	0.68	0.41	0.86	0.45	1.23	0.83	0.68
	POST2	0.34	0.46	0.26	1.02	0.42	1.34	0.55	0.81
CONIF	PRE1	0.37	0.38	0.29	0.05	0.20	0.30	0.16	0.80
	PRE2	0.54	0.38	0.27	0.04	0.33	0.01	0.53	0.51
	POST1	0.35	0.68	0.41	0.09	0.31	0.09	0.80	0.67
	POST2	0.34	0.46	0.26	0.08	0.31	0.08	0.52	0.79
DECID	PRE1	0.00	0.00	0.00	0.90	0.66	0.61	0.16	0.01
	PRE2	0.00	0.00	0.00	0.86	0.24	0.79	0.07	0.01
	POST1	0.00	0.00	0.00	0.77	0.14	1.14	0.02	0.01
	POST2	0.00	0.00	0.00	0.95	0.10	1.27	0.03	0.01
WOOD	PRE1	0.80	0.36	0.03	0.30	0.33	0.46	0.04	0.15
	PRE2	0.12	0.23	0.02	0.22	0.12	0.35	3.28	0.24
	POST1	0.19	0.57	0.06	0.28	0.04	0.56	0.41	0.61
	POST2	0.06	0.56	0.01	0.08	0.06	0.41	0.49	0.67
MISC	PRE1	0.27	0.11	0.01	0.12	0.02	0.14	0.02	0.04
	PRE2	0.10	0.09	0.01	0.09	0.04	0.13	0.27	0.06
	POST1	0.10	0.21	0.03	0.10	0.03	0.52	0.07	0.09
	POST2	0.06	0.14	0.04	0.25	0.03	0.12	0.07	0.05

Appendix Table 12-C-1. (continued)

Component	Treatment Year	OLYM-FP				OLYM-0%			
		L1	L2	L3	L4	L1	L2	L3	L4
TOTAL	PRE1	1.42	2.12	1.18	1.38	1.50	2.49	1.99	1.35
	PRE2	1.99	3.62	1.22	1.55	1.82	1.77	1.44	2.53
	POST1	1.06	2.31	0.39	3.23	0.30	0.05	0.60	0.22
	POST2	1.33	2.78	0.29	0.47	0.17	0.04	0.04	0.15
LEAF	PRE1	0.86	1.23	0.30	0.72	0.91	1.08	1.00	0.73
	PRE2	1.24	1.65	0.30	0.41	0.93	1.07	0.92	0.82
	POST1	0.74	1.11	0.27	0.88	0.15	0.01	0.01	0.07
	POST2	0.92	1.62	0.06	0.27	0.09	0.00	0.00	0.06
CONIF	PRE1	0.24	0.15	0.29	0.72	0.26	0.31	0.21	0.22
	PRE2	0.14	0.06	0.30	0.41	0.40	0.28	0.14	0.24
	POST1	0.31	0.25	0.27	0.88	0.10	0.01	0.00	0.01
	POST2	0.21	0.25	0.06	0.27	0.06	0.00	0.00	0.00
DECID	PRE1	0.62	1.08	0.00	0.00	0.65	0.78	0.79	0.51
	PRE2	1.10	1.60	0.00	0.00	0.53	0.79	0.78	0.58
	POST1	0.43	0.86	0.00	0.00	0.05	0.00	0.00	0.05
	POST2	0.71	1.38	0.00	0.00	0.03	0.00	0.00	0.06
WOOD	PRE1	0.48	0.70	0.79	0.54	0.45	1.24	0.86	0.40
	PRE2	0.62	1.47	0.80	0.99	0.77	0.60	0.47	1.59
	POST1	0.25	0.96	0.10	2.23	0.05	0.02	0.59	0.12
	POST2	0.29	0.73	0.22	0.17	0.01	0.00	0.03	0.01
MISC	PRE1	0.08	0.19	0.10	0.12	0.13	0.17	0.12	0.22
	PRE2	0.13	0.50	0.12	0.15	0.11	0.09	0.05	0.12
	POST1	0.07	0.24	0.02	0.12	0.10	0.02	0.00	0.03
	POST2	0.12	0.43	0.01	0.03	0.07	0.04	0.01	0.08

Appendix Table 12-C-1. (continued)

Component	Treatment Year	WIL1-REF				WIL1-100%			
		L1	L2	L3	L4	L1	L2	L3	L4
TOTAL	PRE1	7.62	5.17	7.92	6.28	1.83	1.50	1.53	6.13
	PRE2	1.27	0.51	1.33	1.20	3.16	4.68	4.13	2.21
	POST1	1.20	0.78	2.19	2.97	2.18	0.95	7.35	1.18
	POST2	0.64	1.02	2.22	0.40	0.85	1.19	0.84	0.75
LEAF	PRE1	1.95	1.49	1.53	2.70	0.86	0.62	0.55	1.76
	PRE2	0.59	0.40	1.00	0.96	0.76	1.87	1.01	1.11
	POST1	0.46	0.41	0.97	1.02	0.86	0.50	2.20	0.71
	POST2	0.33	0.33	0.97	0.25	0.47	0.35	0.58	0.44
CONIF	PRE1	1.95	1.46	0.63	2.26	0.85	0.61	0.55	1.75
	PRE2	0.59	0.39	0.30	0.67	0.73	1.86	1.00	1.10
	POST1	0.46	0.36	0.49	0.99	0.84	0.50	2.19	0.69
	POST2	0.33	0.28	0.52	0.23	0.43	0.35	0.57	0.43
DECID	PRE1	0.00	0.03	0.90	0.44	0.02	0.00	0.00	0.00
	PRE2	0.00	0.01	0.70	0.29	0.03	0.01	0.02	0.01
	POST1	0.00	0.05	0.49	0.03	0.02	0.00	0.01	0.01
	POST2	0.00	0.05	0.45	0.02	0.04	0.00	0.00	0.01
WOOD	PRE1	5.29	3.52	5.95	3.18	0.89	0.78	0.91	4.28
	PRE2	0.61	0.09	0.15	0.17	2.29	2.49	2.91	0.93
	POST1	0.64	0.31	0.91	1.86	1.22	0.39	4.98	0.44
	POST2	0.25	0.64	0.94	0.11	0.35	0.76	0.23	0.30
MISC	PRE1	0.38	0.17	0.44	0.40	0.08	0.10	0.07	0.09
	PRE2	0.07	0.02	0.18	0.07	0.10	0.32	0.20	0.17
	POST1	0.10	0.06	0.31	0.08	0.09	0.06	0.17	0.03
	POST2	0.05	0.05	0.32	0.04	0.04	0.07	0.03	0.02

Appendix Table 12-C-1. (continued)

Component	Treatment Year	WIL1-FP				WIL1-0%			
		L1	L2	L3	L4	L1	L2	L3	L4
TOTAL	PRE1	0.78	1.24	1.63	7.85	1.51	1.72	0.65	3.45
	PRE2	2.20	2.25	2.04	13.78	1.42	1.62	4.49	3.02
	POST1	1.99	3.13	0.21	0.09	1.57	0.01	0.22	0.02
	POST2	0.39	1.65	0.13	0.08	1.13	0.01	0.03	0.05
LEAF	PRE1	0.50	0.59	0.62	0.62	1.01	1.39	0.51	1.15
	PRE2	0.68	0.83	0.78	3.29	0.84	1.07	2.18	1.33
	POST1	0.90	1.12	0.12	0.06	0.61	0.00	0.02	0.00
	POST2	0.24	0.32	0.07	0.06	0.33	0.00	0.00	0.00
CONIF	PRE1	0.50	0.57	0.60	0.62	0.44	0.90	0.24	1.14
	PRE2	0.67	0.80	0.73	3.29	0.26	0.55	1.79	1.33
	POST1	0.90	1.12	0.12	0.06	0.25	0.00	0.01	0.00
	POST2	0.24	0.32	0.07	0.05	0.14	0.00	0.00	0.00
DECID	PRE1	0.01	0.02	0.01	0.01	0.58	0.49	0.27	0.01
	PRE2	0.01	0.03	0.05	0.00	0.58	0.52	0.38	0.00
	POST1	0.00	0.00	0.00	0.00	0.36	0.00	0.01	0.00
	POST2	0.00	0.00	0.00	0.00	0.19	0.00	0.00	0.00
WOOD	PRE1	0.22	0.58	0.93	7.10	0.30	0.22	0.08	2.20
	PRE2	1.33	1.20	1.15	10.09	0.44	0.43	2.21	1.60
	POST1	0.96	1.59	0.08	0.03	0.82	0.00	0.19	0.00
	POST2	0.13	0.32	0.06	0.02	0.67	0.00	0.00	0.00
MISC	PRE1	0.06	0.07	0.08	0.14	0.19	0.10	0.06	0.10
	PRE2	0.19	0.23	0.11	0.40	0.14	0.13	0.11	0.09
	POST1	0.13	0.43	0.01	0.00	0.14	0.00	0.01	0.01
	POST2	0.03	1.01	0.00	0.01	0.14	0.01	0.02	0.05

APPENDIX 12-D. DESCRIPTIVE STATISTICS FOR DETRITUS EXPORT

Appendix Table 12-D-1. Descriptive statistics for detritus export in grams ash-free dry mass per day (g AFDM day^{-1}) from the Type N Study sites by component and treatment year. OLYM = Olympic block; WIL1 = Willapa 1 block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2; TOTAL = total detritus; CPOM = coarse particulate organic matter detritus (>1 mm); LEAF = total leaf detritus (CONIF + DECID); CONIF = coniferous detritus; DECID = deciduous detritus; WOOD = wood detritus; MISC = miscellaneous detritus; FPOM = fine particulate organic matter detritus (<1 mm).

Component	**Treatment Year**	**OLYM-REF**		**OLYM-100%**		**OLYM-FP**		**OLYM-0%**	
		MEAN	**SD**	**MEAN**	**SD**	**MEAN**	**SD**	**MEAN**	**SD**
TOTAL	PRE1	705.73	1273.80	723.39	1795.39	1111.29	2785.36	716.51	1668.12
	PRE2	579.51	718.38	64.48	86.78	194.68	198.23	101.93	209.71
	POST1	1155.25	2725.73	124.40	208.84	593.14	1375.81	84.66	115.63
	POST2	651.93	1327.69	276.95	440.48	4263.47	7336.59	345.24	553.53
CPOM	PRE1	451.41	741.32	614.84	1566.19	902.78	2271.34	580.89	1368.91
	PRE2	432.92	547.76	38.68	52.34	158.82	174.45	74.39	167.19
	POST1	1067.92	2587.82	97.93	162.59	503.93	1183.12	60.51	90.02
	POST2	472.48	976.70	144.09	262.43	3873.16	6793.01	268.97	461.03
LEAF	PRE1	114.48	211.86	79.18	198.53	13.41	10.41	114.94	238.75
	PRE2	82.22	118.22	15.32	21.60	37.57	51.87	18.91	40.96
	POST1	188.94	473.20	48.28	129.42	24.34	40.34	28.02	48.63
	POST2	53.35	55.14	37.52	66.84	97.17	133.27	45.84	58.53
CONIF	PRE1	109.14	201.70	24.13	55.77	10.27	6.41	38.57	93.51
	PRE2	77.61	108.38	5.72	7.27	23.23	29.86	3.45	4.95
	POST1	182.58	464.99	4.21	4.46	22.47	39.24	5.08	8.49
	POST2	49.19	52.37	21.36	42.16	95.97	134.07	15.04	28.85
DECID	PRE1	5.35	10.30	55.05	142.82	3.13	4.58	76.37	148.02
	PRE2	4.62	10.41	9.60	16.49	14.33	27.37	15.45	36.68
	POST1	6.36	12.43	44.07	125.55	1.87	3.04	22.94	43.41
	POST2	4.16	6.92	16.16	28.20	1.20	3.02	30.80	56.16
WOOD	PRE1	295.87	468.31	489.12	1260.02	45.40	65.10	169.26	384.89
	PRE2	156.27	225.58	14.43	21.62	66.20	96.13	28.97	66.37
	POST1	837.36	2057.01	40.73	62.09	30.38	69.04	16.21	22.95
	POST2	352.66	815.82	86.22	176.48	301.44	439.24	60.14	146.38
MISC	PRE1	41.05	71.82	46.54	107.86	843.98	2207.59	296.69	749.27
	PRE2	194.43	384.93	8.94	12.94	55.05	50.69	26.51	60.03
	POST1	41.62	68.56	8.92	9.49	449.20	1074.66	16.28	20.91
	POST2	66.47	122.80	20.35	23.52	3474.55	6280.42	162.98	301.36
FPOM	PRE1	254.33	542.51	108.55	231.10	208.51	514.06	135.62	299.84
	PRE2	146.59	200.07	25.80	34.93	35.86	29.41	27.54	43.74
	POST1	87.34	151.28	26.48	50.08	89.21	192.81	24.15	30.71
	POST2	179.45	351.14	132.87	222.10	390.31	559.40	76.27	118.02

Appendix Table 12-D-1. (continued)

Component	Treatment Year	WIL1-REF		WIL1-100%		WIL1-FP		WIL1-0%	
		MEAN	SD	MEAN	SD	MEAN	SD	MEAN	SD
TOTAL	PRE1	905.72	1850.66	602.76	1208.85	612.73	1289.06	1774.72	2440.54
	PRE2	95.01	191.35	19.83	22.80	19.46	23.77	78.08	83.94
	POST1	38.36	63.07	103.37	196.98	56.07	89.10	266.28	571.32
	POST2	622.76	1200.31	344.19	599.62	155.01	394.50	289.90	788.71
CPOM	PRE1	638.38	1312.64	399.38	789.45	538.22	1142.95	1441.86	2065.77
	PRE2	57.89	119.07	11.36	11.91	12.97	16.51	46.40	48.68
	POST1	27.57	47.82	76.28	151.45	43.44	72.17	177.91	425.68
	POST2	393.90	764.32	211.73	376.93	96.92	242.41	185.25	518.47
LEAF	PRE1	78.83	148.90	140.60	267.92	39.35	79.36	83.74	103.99
	PRE2	14.02	27.60	5.07	5.72	3.16	2.56	16.00	16.68
	POST1	11.92	20.88	40.00	78.84	18.73	30.80	27.78	43.32
	POST2	72.06	139.76	43.20	87.21	17.38	42.65	42.47	122.85
CONIF	PRE1	78.21	147.65	139.52	265.87	38.92	78.51	78.21	98.38
	PRE2	13.98	27.61	5.06	5.71	3.03	2.53	10.55	15.03
	POST1	11.92	20.88	39.92	78.70	18.71	30.78	26.58	43.91
	POST2	72.06	139.76	43.12	87.25	17.28	42.69	26.03	74.49
DECID	PRE1	0.62	1.25	1.08	2.06	0.43	0.85	5.53	5.83
	PRE2	0.04	0.05	0.02	0.04	0.13	0.32	5.45	7.71
	POST1	0.00	0.00	0.08	0.15	0.02	0.06	1.21	1.55
	POST2	0.00	0.00	0.08	0.23	0.10	0.30	16.45	48.36
WOOD	PRE1	245.47	495.10	205.65	414.59	50.96	100.48	408.08	778.43
	PRE2	32.94	70.59	4.04	4.09	5.16	6.93	19.11	25.46
	POST1	12.40	22.04	27.33	56.55	16.22	29.99	63.75	156.20
	POST2	208.96	418.29	106.81	194.00	34.78	77.99	108.72	311.66
MISC	PRE1	314.08	668.84	53.13	106.97	447.92	963.16	950.05	1291.41
	PRE2	10.93	21.62	2.24	3.31	4.64	7.61	11.29	15.84
	POST1	3.26	5.81	8.96	16.44	8.49	18.29	86.38	227.32
	POST2	112.88	227.22	61.72	112.65	44.76	122.36	34.06	84.18
FPOM	PRE1	267.34	538.04	203.38	419.46	74.51	146.28	332.85	429.81
	PRE2	37.12	73.44	8.48	12.06	6.49	7.48	31.68	43.46
	POST1	10.79	15.54	27.08	45.85	12.63	17.43	88.38	149.93
	POST2	228.86	436.19	132.46	223.64	58.09	152.27	104.65	270.29

CHAPTER 13 - BIOFILM AND PERIPHYTON

Aimee McIntyre, Stephanie Estrella, and Eric Lund

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13-1. ABSTRACT

We compared the response of biofilm and periphyton in headwater streams throughout western Washington to clearcut timber harvest with three alternative riparian buffer treatments and reference sites that were not harvested during a five-year study (2006–2010). Riparian buffer treatments varied by percent of the stream length buffered and included the current state Forest Practices buffer (FP treatment), and a longer (100% treatment) and no buffer (0% treatment). We collected biofilm and periphyton samples from four pairs of unglazed ceramic tiles during three pre- (2006–2008) and two post-harvest (2009–2010) years. Tiles were installed in each of 17 headwater streams during two, two-month sample intervals: early summer (June–July) and late summer (August–September). We used a sample collected from one tile in each pair in an analysis of biofilm (i.e., including bacteria, algae and inanimate organic slime, measured as ash-free dry mass) and the remaining tile from each pair in an analysis of periphyton (i.e., the algal component of biofilm, measured as chlorophyll a). We hypothesized that both ash-free dry mass and chlorophyll a would increase in clearcut sites through a reduction in shade, with the greatest increase in sites lacking a riparian buffer in the Riparian Management Zone. However, the pre- to post-harvest change in ash-free dry mass did not differ by treatment following harvest in either the early or late sample intervals (P = 0.83 and P = 0.61, respectively). The change in chlorophyll a also did not differ by treatment post-harvest (P = 0.14 and P = 0.75 for the early and late sample intervals, respectively). While we did observe post-harvest reductions in canopy across all riparian buffer treatments, that reduction did not result in the increased biofilm or periphyton we expected.

13-2. INTRODUCTION

Forested headwater streams are primarily heterotrophic with most organic matter coming from forest canopy inputs (e.g., litterfall) to form the base of the aquatic food chain (Bilby and Bisson 1992). Benthic substrates in aquatic environments are rapidly colonized by biofilms (i.e., bacteria, algae and inanimate organic slime) which form the vast majority of the microbial biomass in headwater streams (Geesey *et al.* 1978). Although often limited in headwater streams, periphyton, the algal component of biofilm (i.e., the community of algae, diatoms, and cyanobacteria on benthic substrates), contributes to most of the primary production in headwater streams (Allan 1995). Due to its association with the benthos, periphyton is affected by physical and chemical processes in the stream system, which, when combined with its short life cycle and rapid reproduction rates, make periphyton a valuable indicator of short-term environmental changes (Barbour *et al.* 1999; Stevenson and Bahls 1999).

Periphyton growth in small forested streams is limited primarily by light (Gregory 1980; Feminella *et al.* 1989; Hill *et al.* 1995; Kiffney and Bull 2000; Kiffney *et al.* 2003, 2004; Mallory and Richardson 2005; Liess *et al.* 2009) so that removal of timber from headwater stream basins has the potential to increase periphyton productivity through the reduction of shade. Other studies have observed an increase in periphyton productivity in streams bordered by clearcuts or by narrow buffers (Hansmann and Phinney 1973; Murphy and Hall 1981; Murphy *et al.* 1981; Murphy *et al.* 1986; Kiffney and Bull 2000; Kiffney *et al.* 2003; Danehy *et al.* 2007; Wilkerson *et al.* 2010).

Other factors have been found to limit or influence periphyton growth, including stream flow (Hansmann and Phinney 1973; Shortreed and Stockner 1983; Peterson and Stevenson 1992), sediment (Biggs *et al.* 1999), nutrients (Perrin and Richardson 1997; Francoeur *et al.* 1999; Kiffney and Richardson 2001; Greenwood and Rosemond 2005; Liess *et al.* 2009), temperature (Noel *et al.* 1986; Bilby and Bisson 1992; Rosemond 1994; Kiffney *et al.* 2003), and consumers (Feminella *et al.* 1989; Hill *et al.* 1995; Mallory and Richardson 2005; Connelly *et al.* 2008), all of which are also susceptible to change following timber harvest. For example, suspended sediment delivery to streams may increase following harvest (reviewed in Gomi *et al.* 2005), which can inhibit periphyton growth through accumulation in the biofilm matrix (Kiffney and Bull 2000; Kiffney *et al.* 2003) or through scouring during flood events (Allan 1995). Periphyton growth may also fluctuate seasonally, with a peak in biomass typically occurring between spring and fall (Francoeur *et al.* 1999; Kiffney *et al.* 2003; Volk 2004) and a decrease in winter as a result of high flows and scouring (Shortreed and Stockner 1983). The timing of seasonal peaks in periphyton biomass may also differ between streams of varying riparian canopy composition (i.e., hardwood vs. conifer; Volk 2004).

Periphyton is an important food source for invertebrates (Hershey and Lamberti 1998) and larval amphibians, such as Coastal Tailed Frog (*Ascaphus truei*; Kiffney and Richardson 2001). While microbial conditioning is needed to improve food quality and palatability of organic matter inputs from the forest canopy (Sedell *et al.* 1975; Triska *et al.* 1975; Meehan *et al.* 1977; Suberkropp 1998), periphyton is readily digestible and more nutritious, reproduces rapidly and can turnover quickly in response to stream scour and grazing pressure (Murphy 1998). An increase in periphyton growth after timber harvest has been shown to increase productivity of invertebrates, amphibians and fish (Murphy and Hall 1981; Kiffney and Richardson 2001; Kiffney *et al.* 2003).

We examined the response of biofilm and periphyton in treated and reference sites. In particular, we examined the response of biofilm (measured as change in the ash-free dry mass [AFDM]) and periphyton (measured as change in the amount of chlorophyll *a*) to clearcut harvest with alternative riparian buffer treatments, including a treatment with no riparian buffer (i.e., clearcut to the stream edge). We hypothesized that there would be an increase in periphyton production in clearcut sites through a reduction in shade, which would also result in a subsequent increase in biofilm, especially for sites lacking a riparian buffer in the Riparian Management Zone (RMZ).

13-3. OBJECTIVES

The primary objective of this study was to describe changes in biofilm and periphyton in non-fish-bearing (i.e., Type N) headwater streams in three different riparian buffer treatments and reference conditions. Specifically, we quantified summer biofilm and periphyton production, as measured by AFDM and chlorophyll *a*, respectively, across 17 non-fish-bearing basins. We hypothesized that both AFDM and chlorophyll *a* would increase post-harvest with a reduction in shade in the treated sites, with the greatest increase in the 0% treatment. Sediment and canopy cover, which are impacted by timber harvest activities, are known to affect biofilm accumulation and periphyton growth, so a secondary objective was to assess the relationship of biofilm and periphyton with sediment and canopy.

13-4. METHODS

We collected biofilm and periphyton samples from 17 Type N study sites (see Chapter 2 – *Study Design*) in each pre-treatment (2006–2008) and post-treatment (2009–2010) year. To minimize the confounding effects of habitat differences, we used unglazed ceramic tiles (approximately 15 × 15 cm) as substrate for algal colonization (Barbour *et al.* 1999). Ceramic tiles support algal and primary consumer communities similar to those found on natural inorganic substrata (Lamberti and Resh 1985; Kiffney and Richardson 2001).

13-4.1. TILE PLACEMENT

We installed four pairs of two tiles each (n = 8) in the mainstem channel of each of our 17 study sites during two sample intervals: early summer (June and July) and late summer (August and September). We did not sample the CASC-FP or the WIL3-REF in the 2006 early summer season because inclusion of these study sites had not been confirmed at that time. In 2009, samples were not collected for either season in the OLYM-0% and WIL2-REF1 sites. In the former, active harvest prevented data collection. In the latter, which was intended to be a buffer treatment, the harvest had not been applied. When this site was not harvested we included it as a second reference in the Willapa 2 block (see Chapter 2 – *Study Design*).

Tiles were placed throughout the mainstem channel of each study site, and response of biofilm (as AFDM) and periphyton (as chlorophyll *a*) were evaluated as the average site-wide response by year and sample interval. We placed tile pairs such that two pairs were in what would have been the buffered and two in what would have been the unbuffered RMZ, had the FP treatment been applied to each site (**Figure 13-1**). We placed tile pairs *a priori* and at least 20 m from one another, from a change in RMZ type (buffered or unbuffered, assuming a FP treatment), and from road crossings. Tiles within each pair were placed 1 m apart. We attempted to place each tile in the location we identified *a priori*, with no consideration of stream channel unit type (e.g., riffle, pool). If a tile had to be moved from its selected location (e.g., bedrock channel prevented installation, dry channel), then we placed the tile in the nearest location in which it could be successfully installed, up- or downstream. Once a tile location was established, every attempt was made to install tiles in the same location for every sample interval and year thereafter. We installed tiles in the stream thalweg in an effort to ensure that tiles would remain submerged during the entire sample interval. Sometimes we were required to move a location between the early and late summer interval due to seasonal drying of the stream.

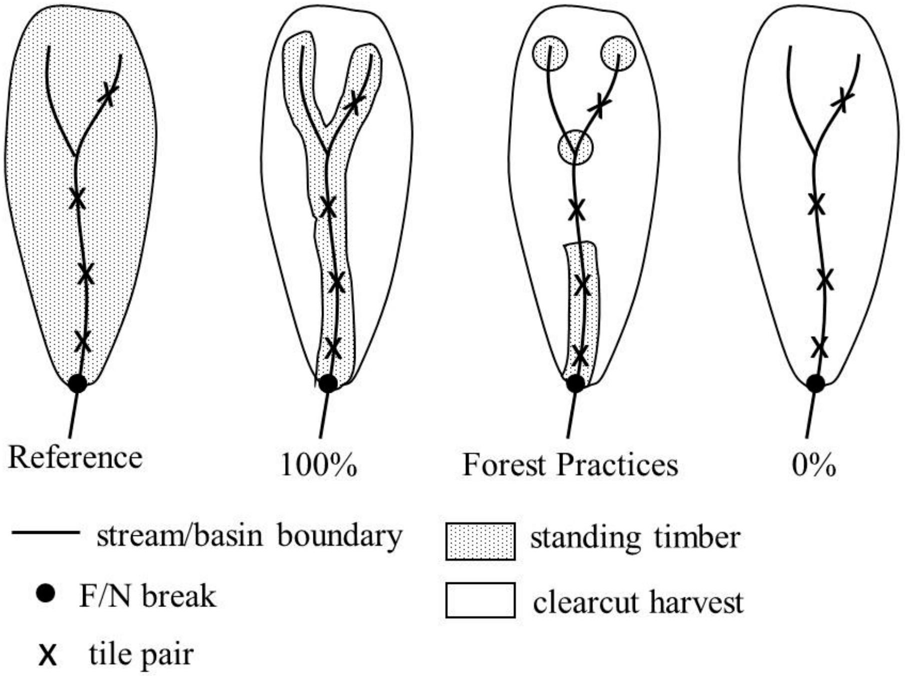

Figure 13-1. Depiction of tile pair placement by treatment, including two unbuffered and two buffered pairs in the FP treatment.

We drilled a hole (approximately 2.5 cm diameter) through the center of each tile, anchored it to the streambed with rebar, and suspended it above the channel with spacers to minimize sediment accumulation (**Figure 13-2**).

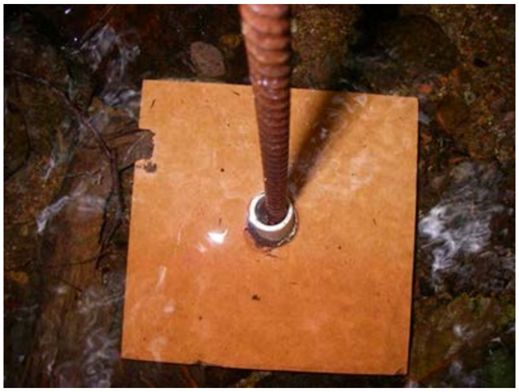

Figure 13-2. Unglazed ceramic tile (approximately 15 × 15 cm) anchored to stream bed with rebar.

After treatments had been applied we realized that the placement of the tile pairs in the WIL2-100% were located along a downstream stream reach that was not adjacent to upland harvest due to slope instability (see Chapter 3 – *Management Prescriptions*). As a result, in the second post-harvest year (2010) we added another four tile pairs upstream of the original locations and within the riparian buffer. Tiles from these additional pairs were used in the analysis. Since the tile pairs were not located in the riparian buffer during 2009 sampling, they were not included in the analysis, resulting in missing estimates of AFDM and chlorophyll *a* for both sample intervals in 2009. Other samples were not collected due to the following reasons: tile was not installed or removed, typically due to harvest activities in the vicinity; tile became exposed above the water level during the sample interval; or, tile was missing, buried or broken.

13-4.2. FIELD SAMPLING, LAB PROCESSING AND ANALYSIS

Prior to removal of tiles at the end of each sample interval we recorded the proportion of the tile that was exposed or dried due to a decrease in the water level surface. All tiles with any amount of exposure or drying were removed from the analysis. We also recorded the proportion of the tile surface that was covered with sediment so that we could evaluate the relationship between tile sediment cover and AFDM and chlorophyll *a*.

We carefully removed each tile from the rebar, and tilted it in the stream to rinse off loose sediment and debris before placing it in a sealed plastic bag. We kept tiles on ice during transport to the laboratory. Prior to processing of tiles in the laboratory at the end of each sample interval, we randomly assigned the downstream-most tile to one of the two analyses (AFDM or chlorophyll *a*). Remaining pairs alternated in the order of assignment. We measured the dimensions of each tile, including any portion of the tile that may have broken off during deployment, so that biomass could be adjusted by current tile area. We removed periphyton by brushing the top surface of each tile with a wire brush, scraping it with a razor blade, and rinsing the periphyton from the tile and tools into a Nalgene® collection bottle using distilled water. Chlorophyll *a* samples were placed in opaque bottles. All samples were returned to coolers and stored in a refrigerator until transfer to the Manchester Environmental Laboratory for analysis within 24 hours from the time of collection. The laboratory measured the concentration of periphyton following standard methods for chlorophyll *a* (10200H3M) and AFDM (SM10300C).

The AFDM samples were filtered through a glass fiber filter, dried to a constant weight at 60ºC, ignited at 500ºC in a muffle furnace, rewetted and redried at 105ºC, and weighed to estimate AFDM. The chlorophyll *a* samples were filtered through a 4.7 cm glass fiber filter and preserved with 90% aqueous acetone solution. Pigments were extracted from the plankton concentrate with aqueous acetone by sonication and their fluorescence determined using a fluorometer. The concentration of AFDM and chlorophyll *a* was measured as mg/tile and µg/tile, respectively.

We calculated the tile area (cm^2) for each tile as:

$$\begin{aligned}&Tile\ area = [tile\ length\ (cm) * tile\ width\ (cm)] - [center\\ &anchor\ hole\ (cm^2) + area\ missing\ portion\ (cm^2)]\end{aligned} \tag{13-1}$$

Next we adjusted each AFDM and chlorophyll *a* mass by tile area:

$$mass/cm^2 = mass/tile\ area\ (cm^2) \tag{13-2}$$

Finally, we calculated a site-wide average for AFDM and chlorophyll *a* for each year and sample interval. Results for AFDM are presented in mg/cm^2 and results for chlorophyll *a* are presented in µg/cm^2.

We used sample weight as a surrogate for sample volume (assumed 1 g = 1 ml). In summer 2006, the contract laboratory did not provide bottle weights and we were not able to calculate sample weight. As a result, we omitted data from the 2006 late summer sample interval for our analyses of AFDM and chlorophyll *a*. Samples were missing or excluded in other instances due to harvest delays, site inaccessibility, and drying or breaking of tiles during deployment (**Table 13-1** and **Table 13-2**).

Table 13-1. The number of chlorophyll *a* samples analyzed by site and year in the early and late summer sample intervals.

Block	**Treatment**	**Early Summer**					**Late Summer**				
		2006	**2007**	**2008**	**2009**	**2010**	**2006**	**2007**	**2008**	**2009**	**2010**
OLYM	REF	4	4	4	4	1	0	4	4	4	4
	100%	3	2	4	2	4	0	4	4	4	3
	FP	2	4	4	3	4	0	4	4	3	3
	0%	2	3	3	0	3	0	4	4	0	4
WIL1	REF	3	3	4	3	4	0	4	3	3	3
	100%	2	4	4	3	3	0	4	4	4	4
	FP	4	4	4	4	4	0	4	4	4	4
	0%	4	4	4	3	4	0	4	4	4	4
WIL2	REF1	3	3	1	0	1	0	3	4	0	4
	REF2	2	4	4	4	4	0	4	4	4	3
	100%	4	4	4	0	4	0	4	4	0	4
	0%	4	4	4	4	3	0	4	2	4	4
WIL3	REF	0	3	2	3	3	0	3	4	1	4
	100%	4	4	4	3	3	0	4	4	4	4
CASC	REF	3	1	3	2	4	0	4	3	4	4
	FP	0	3	3	3	4	0	3	2	4	4
	0%	4	2	4	3	2	0	3	3	4	4

Table 13-2. The number of ash-free dry mass (AFDM) samples analyzed by site and year in the early and late summer sample intervals.

Block	Treatment	Early Summer					Late Summer				
		2006	2007	2008	2009	2010	2006	2007	2008	2009	2010
OLYM	REF	4	4	4	3	1	0	4	4	4	4
	100%	3	3	4	4	3	0	4	4	4	4
	FP	3	3	4	2	4	0	4	4	2	3
	0%	3	2	2	0	2	0	4	4	0	4
WIL1	REF	3	4	4	3	4	0	4	3	3	3
	100%	2	4	3	4	4	0	4	4	4	4
	FP	4	4	4	4	4	0	4	4	3	3
	0%	3	4	4	4	4	0	4	4	4	4
WIL2	REF1	2	4	3	0	1	0	4	4	0	4
	REF2	3	4	4	4	4	0	3	3	4	4
	100%	4	4	4	0	4	0	4	4	0	3
	0%	4	3	4	4	1	0	4	2	4	3
WIL3	REF	0	4	4	3	3	0	4	4	2	4
	100%	0	4	4	4	3	0	4	4	4	4
CASC	REF	4	3	2	3	4	0	4	4	4	4
	FP	0	3	3	1	4	0	4	2	4	4
	0%	4	2	4	3	2	0	3	3	4	4

13-4.2.1. Statistical Analysis

Analyses following the Before-After-Control-Impact (BACI) design evaluated the generalized null hypothesis:

$$\Delta T_{\mathrm{REF}} = \Delta T_{100\%} = \Delta T_{\mathrm{FP}} = \Delta T_{0\%} \qquad (13\text{-}3)$$

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% buffer treatments, respectively.

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, and included a weight statement to account for differing numbers of subsamples (see **Table 13-1** and **Table 13-2**). In the model, block and site were random effects and the fixed effects were year, treatment, and the treatment × year interaction. 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.

The null hypotheses were evaluated with a Wald-type test using linear contrasts of the model fixed effects. The contrasts were constructed 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 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 greater than 0.1, we provided estimates of pairwise contrasts, but did not report test results for these terms. The uneven distribution of treatments among blocks required using 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 either.

This study was designed to evaluate response differences among treatments at the site scale, not to investigate within-site variability. Though some data may lend themselves to evaluation within sites, those comparisons were not intended to be formally addressed in this report.

13-4.3. RELATIONSHIP OF AFDM AND CHLOROPHYLL A WITH SEDIMENT AND CANOPY COVER

To evaluate the relationship of biofilm and periphyton with sediment and shade, we conducted post-hoc comparisons of AFDM and chlorophyll *a* values from individual tiles (not averaged as with our site-wide analyses) with sediment and canopy. We calculated the Pearson product-moment correlation coefficient (*r*), a statistical measurement of the linear association of sediment and canopy with AFDM and chlorophyll *a*. The strength of correlations will be reported based on the absolute value of *r* according to Evans (1996), whereby *r* values that are 0.00–0.19 are considered very weak, 0.20–0.39 are weak, 0.40–0.59 are moderate, 0.60–0.79 are strong, and 0.80–1.0 are very strong.

Sediment data were recorded as a visual estimate of the percentage of the tile covered by sediment at the time of tile collection from the stream. We also used two metrics of canopy cover collected as part of the stream cover evaluation of the study (see Chapter 7 – *Stream Temperature and Cover)*: canopy closure and effective shade. Canopy closure estimates were obtained from spherical densiometer measures collected at the water surface (Werner 2009) and effective shade estimates were obtained from hemispheric canopy photos taken 1 m above the water surface (Stohr and Bilheimer 2008). Canopy closure and effective shade data were collected at canopy stations systematically located throughout the stream channel network (see Chapter 7 – *Stream Temperature and Cover* for details on how canopy closure and effective shade were calculated, as well as locations of canopy stations). Canopy closure data were only collected in 2008, 2009 and 2010 and effective shade data were collected 2007 through 2010. Since we hypothesized that the amount of shading provided by the overstory canopy in the RMZ would impact AFDM and chlorophyll *a* values, and since the FP treatment had two tile pairs in each of the buffered and unbuffered reaches of the RMZ, we investigated the range of canopy cover related to both AFDM and chlorophyll *a* estimates by RMZ buffer type (i.e., reference, buffered and unbuffered stream reaches). We identified the canopy station that was nearest to, and located in the same post-treatment buffer type as, each periphyton tile pair and determined the distance from each tile pair to the corresponding canopy station. One periphyton tile pair from the post-harvest period was located equidistant from two canopy stations, so for that tile pair we averaged the canopy closure and effective shade values from the two stations.

13-5. RESULTS

13-5.1. ASH-FREE DRY MASS

We used 504 samples in the analysis of AFDM, 264 for the early and 240 for the late summer period. Tiles were installed an average of 59 days in each period (46–75 days). AFDM results for individual tiles ranged from 0.01 to 0.59 mg/cm^2, with more than 98% of results less than 0.25 mg/cm^2. Only six results were greater than 0.25 mg/cm^2, four from tiles located in a buffered stream reach and two from tiles located in an unbuffered reach.

13-5.1.1. Early Summer

Mean AFDM in the early summer season ranged from 0.02 to 0.12 mg/cm^2 in the pre- and 0.05 to 0.23 mg/cm^2 in the post-harvest period (**Appendix Table 13-A-1**). The pre- to post-harvest change within treatment ranged from 0.05 to 0.07 mg/cm^2 (**Table 13-3** and **Figure 13-3**). The P-value for the period × treatment contrast was 0.69, where period is a factor with levels pre- and post-harvest, indicating that the amount of change in early summer AFDM did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 13-4**).

Table 13-3. The within-treatment change, standard error (SE) of the estimate and 95% confidence intervals (CI) for ash-free dry mass (AFDM; mg/cm^2) in the early summer season, i.e., (post − pre).

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	0.07	0.01	0.04	0.10
100%	0.05	0.02	0.02	0.08
FP	0.06	0.02	0.03	0.10
0%	0.05	0.02	0.01	0.08

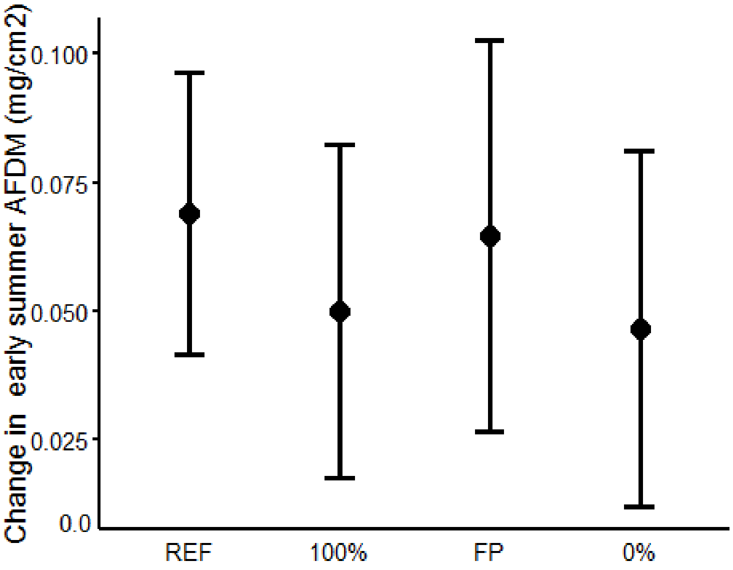

Figure 13-3. The within-treatment change and 95% confidence intervals (CI) for ash-free dry mass (AFDM; mg/cm^2) in the early summer season, i.e., (post – pre).

Table 13-4. The between-treatment difference in the change, standard error (SE) of the estimate and 95% confidence intervals (CI) for ash-free dry mass (AFDM; mg/cm^2) in the early summer season, i.e., [TrtA(post − pre) − TrtB(post − pre)]. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Estimate	SE	95% CI	
			Lower	Upper
100% vs. REF	−0.02	0.02	−0.06	0.02
FP vs. REF	0.00	0.02	−0.05	0.04
0% vs. REF	−0.02	0.02	−0.07	0.02
0% vs. FP	−0.02	0.03	−0.07	0.03
0% vs. 100%	0.00	0.02	−0.05	0.04
FP vs. 100%	0.01	0.02	−0.04	0.06

13-5.1.2. Late Summer

Mean AFDM in the late summer season ranged from 0.02 to 0.16 mg/cm^2 in the pre- and 0.02 to 0.26 mg/cm^2 in the post-harvest period (**Appendix Table 13-A-1**). The pre- to post-harvest change within treatment ranged from 0.04 to 0.06 mg/cm^2 (**Table 13-5** and **Figure 13-4**). The P-value for the period × treatment contrast was 0.61, indicating that the amount of change in late summer AFDM did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 13-6**).

Table 13-5. The within-treatment change, standard error (SE) of the estimate and 95% confidence intervals (CI) for ash-free dry mass (AFDM; mg/cm^2) in the late summer season, i.e., (post − pre).

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	0.04	0.01	0.01	0.06
100%	0.06	0.02	0.02	0.09
FP	0.05	0.02	0.01	0.09
0%	0.06	0.02	0.03	0.09

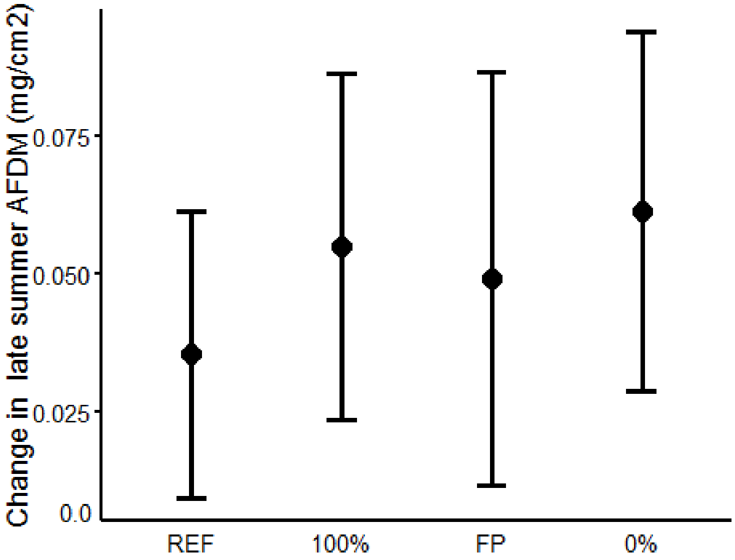

Figure 13-4. The within-treatment change and 95% confidence intervals (CI) for ash-free dry mass (AFDM; mg/cm^2) in the late summer season, i.e., (post − pre).

Table 13-6. The between-treatment difference in the change, standard error (SE) of the estimate and 95% confidence intervals (CI) for ash-free dry mass (AFDM; mg/cm^2) in the late summer season, i.e., [TrtA(post − pre) − TrtB(post − pre)]. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Estimate	SE	95% CI	
			Lower	Upper
100% vs. REF	0.02	0.02	−0.02	0.06
FP vs. REF	0.01	0.02	−0.03	0.06
0% vs. REF	0.03	0.02	−0.02	0.07
0% vs. FP	0.01	0.02	−0.04	0.06
0% vs. 100%	0.01	0.02	−0.04	0.05
FP vs. 100%	−0.01	0.02	−0.05	0.04

13-5.2. CHLOROPHYLL A

We used 503 samples in the analysis of chlorophyll *a*, 263 for the early and 240 for the late summer season. Tiles were installed an average of 59 days in each season (46–75 days). Chlorophyll *a* results for individual tiles ranged from <0.01 to 6.87 µg/cm^2, with over 97% of results less than 2.3 µg/cm^2. Only seven results were greater than 2.3 µg/cm^2, two from tiles located in reference conditions, two from tiles located in a buffered stream reach, and four from tiles located in an unbuffered reach.

13-5.2.1. Early Summer

Mean chlorophyll *a* in the early summer season ranged from 0.01 to 1.39 µg/cm^2 in the pre- and 0.07 to 1.36 µg/cm^2 in the post-harvest period (**Appendix Table 13-A-1**). The pre- to post-harvest change within treatment ranged from 0.00 to 0.38 µg/cm^2 (**Table 13-7** and **Figure 13-5**). The P-value for the period × treatment contrast was 0.12, indicating that the amount of change in early summer chlorophyll *a* did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 13-8**).

Table 13-7. The within-treatment change, standard error (SE) of the estimate and 95% confidence intervals (CI) for chlorophyll *a* (µg/cm^2) in the early summer season, i.e., (post – pre).

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	0.38	0.12	0.14	0.62
100%	0.06	0.14	−0.22	0.35
FP	0.01	0.15	−0.30	0.32
0%	0.00	0.14	−0.28	0.28

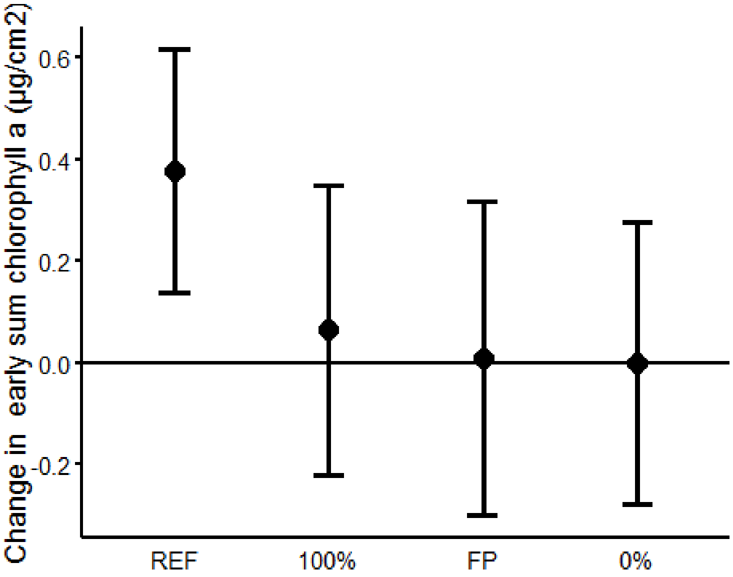

Figure 13-5. The within-treatment change and 95% confidence intervals (CI) for chlorophyll *a* (µg/cm^2) in the early summer season, i.e., (post – pre).

Table 13-8. The between-treatment difference in the change, standard error (SE) of the estimate and 95% confidence intervals (CI) for chlorophyll *a* (µg/cm^2) in the early summer season, i.e., [TrtA(post − pre) − TrtB(post − pre)]. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	**Estimate**	**SE**	**95% CI**	
			Lower	**Upper**
100% vs. REF	−0.31	0.18	−0.68	0.06
FP vs. REF	−0.37	0.19	−0.76	0.02
0% vs. REF	−0.38	0.18	−0.74	−0.01
0% vs. FP	−0.01	0.21	−0.42	0.40
0% vs. 100%	−0.07	0.20	−0.46	0.33
FP vs. 100%	−0.06	0.21	−0.48	0.36

13-5.2.2. Late Summer

Mean chlorophyll *a* in the late summer season ranged from 0.01 to 1.58 µg/cm^2 in the pre- and 0.02 to 2.37 µg/cm^2 in the post-harvest period (**Appendix Table 13-A-1**). The pre- to post-harvest change within treatment ranged from 0.27 to 0.50 µg/cm^2 (**Table 13-9** and **Figure 13-6**). The P-value for the period × treatment contrast was 0.86, indicating that the amount of change in late summer chlorophyll *a* did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and treatments (**Table 13-10**).

Table 13-9. The within-treatment change, standard error (SE) of the estimate and 95% confidence interval (CI) for chlorophyll *a* (µg/cm^2) in the late summer season, i.e., (post − pre).

Treatment	**Estimate**	**SE**	**95% CI**	
			Lower	**Upper**
REF	0.27	0.17	−0.09	0.62
100%	0.50	0.20	0.08	0.91
FP	0.40	0.24	−0.09	0.88
0%	0.39	0.21	−0.04	0.82

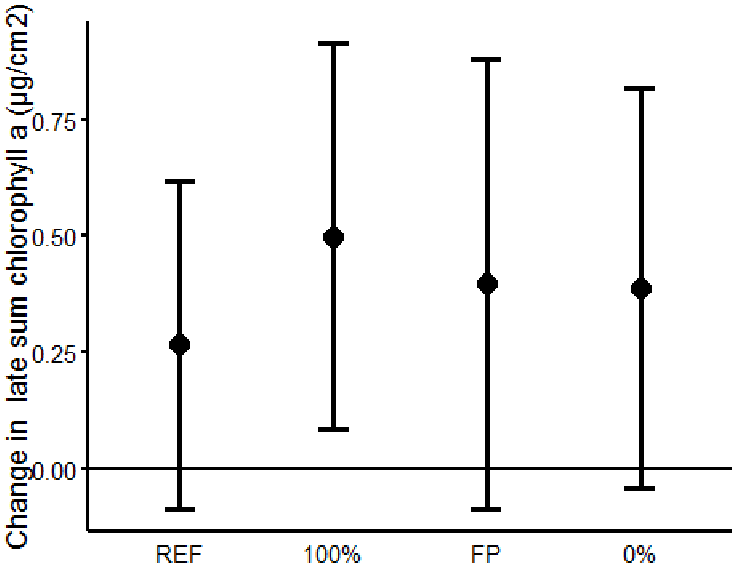

Figure 13-6. The within-treatment change and 95% confidence intervals (CI) chlorophyll *a* (µg/cm^2) in the late summer season, i.e., (post − pre).

Table 13-10. The between-treatment difference in the change, standard error (SE) and 95% confidence intervals (CI) for chlorophyll *a* (µg/cm^2) in the late summer season, i.e., [TrtA(post − pre) − TrtB(post − pre)]. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the greatest impact (fewer trees remaining in the RMZ buffer).

Contrast	Estimate	SE	95% CI	
			Lower	Upper
100% vs. REF	0.23	0.27	−0.31	0.78
FP vs. REF	0.13	0.30	−0.47	0.73
0% vs. REF	0.12	0.27	−0.43	0.68
0% vs. FP	−0.01	0.32	−0.65	0.64
0% vs. 100%	−0.11	0.29	−0.71	0.49
FP vs. 100%	−0.10	0.31	−0.74	0.54

13-5.3. SEDIMENT

The number of individual tiles for which we had both sediment and AFDM data was 468, and the number for which we had both sediment and chlorophyll *a* data was 451. We observed some fine sediment on 80% of all tiles at the time of sample collection. The percent of a tile covered by sediment ranged from 0% to 100% (mean = 34%). Average tile sediment cover across all sites was 26% and 29% in the pre- and post-harvest periods, respectively. There was a very weak negative correlation between sediment and AFDM in both the pre- and post-harvest periods (r = −0.11 and −0.14, respectively). There was a weak negative correlation between sediment and chlorophyll *a* in the pre- and post-harvest periods (r = −0.29 and −0.23, respectively).

13-5.4. CANOPY

13-5.4.1. Canopy Closure

Tiles were located anywhere between 0 and 314 m from the nearest canopy station (mean = 37 m). The number of tiles for which we had both canopy closure and AFDM and chlorophyll *a* data was 328. Canopy closure estimates associated with our tile pairs ranged from <1% to 96% (**Figure 13-7** and **Figure 13-8**). The distribution of canopy closure values was bimodal, with 88% of values greater than or equal to 67% canopy and 11% of values less than or equal to 33%. All values in the lowermost range (≤33%) were in unbuffered reaches of treated streams (FP or 0% treatments). However, canopy closure values from the unbuffered reaches were fairly evenly distributed between the lowermost and uppermost (≥66%) ranges (61% in the lowermost range and 34% in the uppermost range).

We found weak evidence of a negative correlation between canopy closure and AFDM in the pre-harvest period ($r = -0.28$) and very weak evidence of a negative correlation in the post-harvest period ($r = -0.05$). We also found very weak evidence of a negative correlation between canopy closure and chlorophyll *a* in the pre- ($r = -0.02$) and weak evidence of a positive correlation in the post-harvest period ($r = 0.21$).

13-5.4.2. Effective Shade

The number of tiles for which we had effective shade data was 453 and 449 for AFDM and chlorophyll *a*, respectively. Effective shade values 1 m above stream level ranged from 3% to 99% (**Figure 13-9** and **Figure 13-10**). The effective shade values were more equally distributed throughout the range of values than canopy closure estimates, but were grouped more distinctively by buffer type. Greater than 99% of reference values were greater than or equal to 67% canopy. Values from the buffered reaches ranged from 41% to 96%, and 92% of the values between 33% and 67% were from tile pairs located in the buffered reach. Ninety-five percent of the effective shade values from tile pairs located in unbuffered reaches were less than or equal to 33%.

We found very weak evidence of a negative correlation between effective shade and AFDM in both the pre- and post-harvest periods ($r = -0.16$ and -0.19, respectively). There was weak evidence of a negative correlation between effective shade and chlorophyll *a* in the pre-harvest period ($r = -0.24$) and a complete lack of evidence of correlation in the post-harvest period ($r = 0.00$).

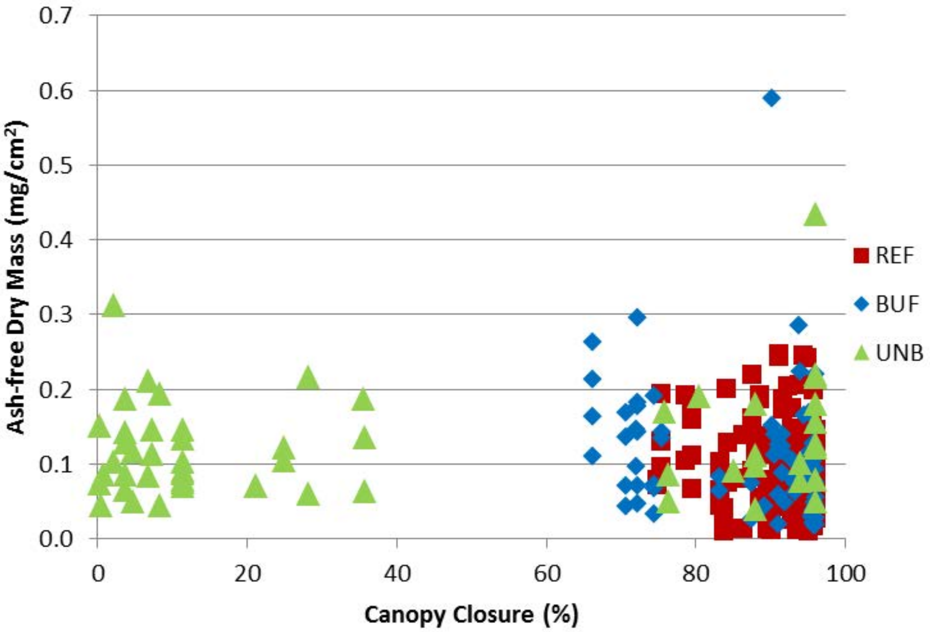

Figure 13-7. Relationship between canopy closure (%) and ash-free dry mass (mg/cm^2) for each tile by buffer type (REF = reference, BUF = buffered, UNB = unbuffered).

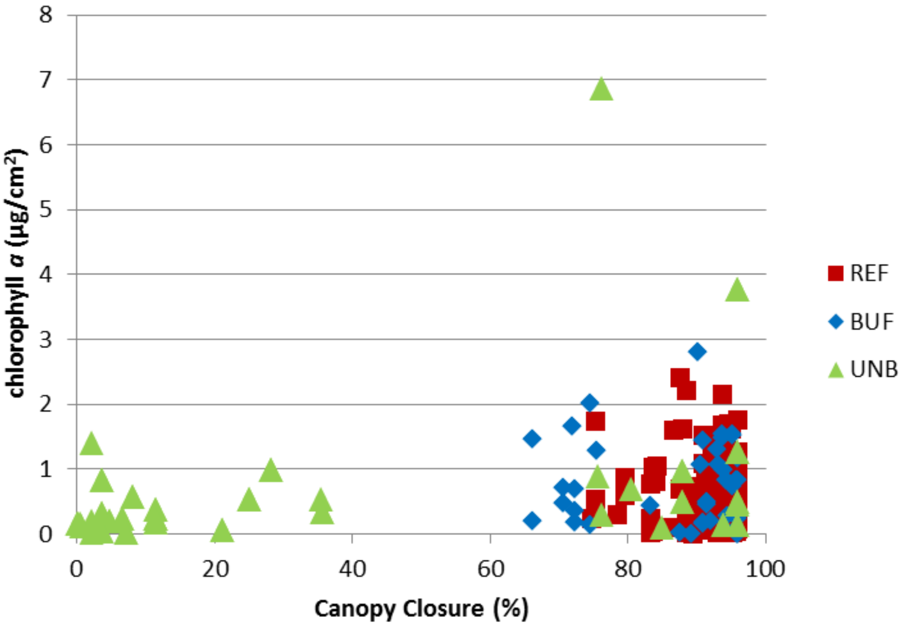

Figure 13-8. Relationship between canopy closure (%) and chlorophyll *a* (µg/cm^2) for each tile by buffer (REF = reference, BUF = buffered, UNB = unbuffered).

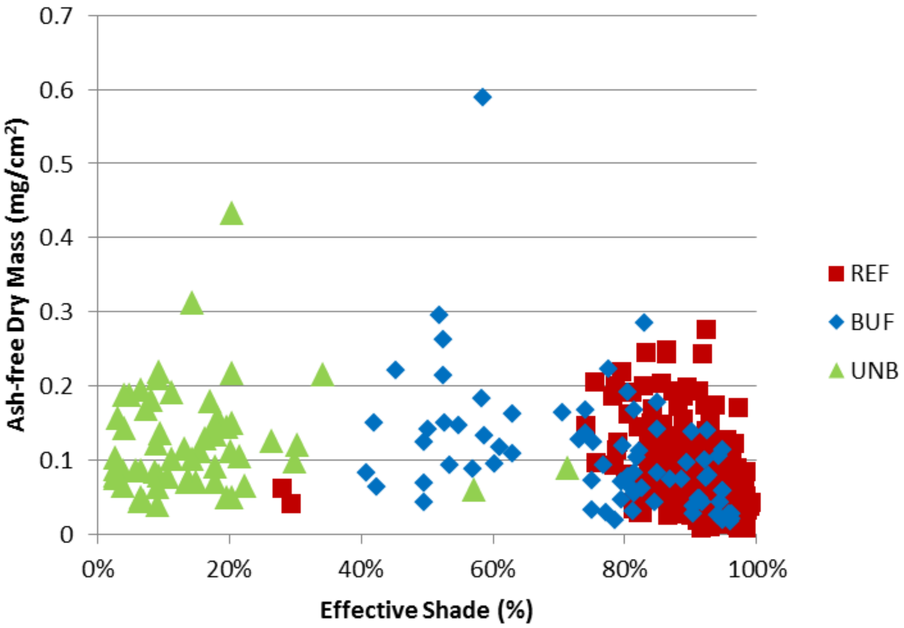

Figure 13-9. Relationship between effective shade (%) and ash-free dry mass (mg/cm^2) for each tile, by buffer (REF = reference, BUF = buffered, UNB = unbuffered).

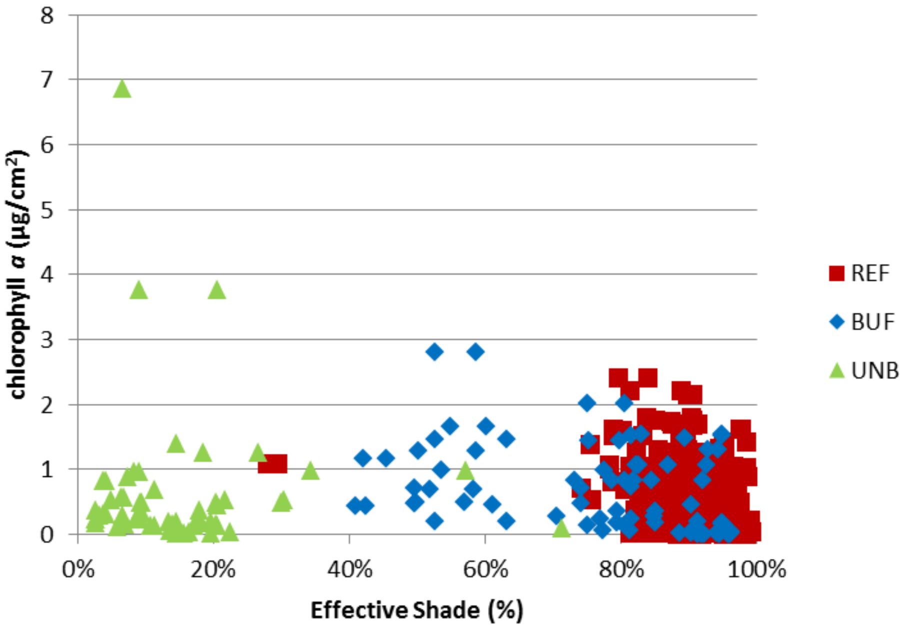

Figure 13-10. Relationship between effective shade (%) and chlorophyll *a* (µg/cm^2) for each tile, by buffer (REF = reference, BUF = buffered, UNB = unbuffered).

13-6. DISCUSSION

We found no evidence of a post-harvest increase in biofilm, as measured by AFDM, or periphyton, as measured by chlorophyll *a,* that depended on treatment. However, we did see evidence of a year effect on AFDM. Specifically, we observed an increase in AFDM in the post-harvest period across all treatments, including buffer treatments and the reference, with within-treatment changes that were similar for both the early and late summer sample intervals. The post-harvest increase in AFDM was not reflected in our estimates of chlorophyll *a*, for which we had a greater variability in response, both between treatments as well as between sample intervals. Overall, however, our results for both AFDM and chlorophyll *a* were consistent in that we lacked evidence of a treatment effect in both instances.

Our findings are unexpected. Consistent with the literature, we expected an increase in algal production associated with a reduction in shade. Indeed, periphyton production in forested headwater streams appears to be primarily light limited (Gregory 1980; Lowe *et al.* 1986; Hill *et al.* 1995; Kiffney *et al.* 2003), and the high degree of shading in headwater streams generally suppresses rates of primary production (Hill *et al.* 1995; Kiffney *et al.* 2003, 2004). In an experimental study where the amount of light reaching artificial stream channels was controlled with shade cloth, Kiffney and colleagues (2004) found a positive relationship between algal biomass and the amount of light reaching the stream. In another experiment, Mallory and Richardson (2005) showed that light was the greatest limiting factor for algae. In a canopy thinning experiment, increased light availability resulted in an approximated 10-fold increase in algal accrual rates in thinned stream reaches relative to upstream unthinned reaches (Collins *et al.* 2015). Richardson and Béraud (2014) went on to hypothesize that stream size may play a role in the impact of the removal of canopy cover in the RMZ, in particular, that smaller headwater streams are likely to have greater canopy cover than larger streams, and that removal of riparian forests may have a larger effect on algae production and other measures in smaller streams (e.g., Danehy *et al.* 2007; Reid *et al.* 2010).

We did observe a post-harvest decrease in shade in our buffer treatment sites that differed from the lack of change in the reference (see Chapter 7 – *Stream Temperature and Cover*); however, this reduction in shade did not result in the increase in algal production that we expected. Our results differ from those of several retrospective studies, where significant increases in AFDM and/or chlorophyll *a* were seen in streams with adjacent timber harvest compared with values from unharvested references (Hansmann and Phinney 1973; Murphy and Hall 1981; Murphy *et al.* 1981; Kiffney and Bull 2000; Kiffney *et al.* 2003; Danehy *et al.* 2007). In their experimental study, Wilkerson and colleagues (2010) reported an increase in chlorophyll *a* concentrations in their clearcut and 11-m wide buffer treatments. Richardson and Béraud (2014) were able to find only five replicated studies that included algae in their meta-analysis evaluating the impacts of riparian forest harvest on streams; in this analysis they found a positive, but non-significant, response of chlorophyll *a* to logging. This same analysis supported the aforementioned findings of both Danehy and colleagues (2007) and Reid and colleagues (2010) that wider streams had a greater response in chlorophyll *a* concentration following harvest than smaller streams. Still, others have reported a lack of differences in algal biomass between control and clearcut sites (Culp and Davies 1983; Göthe *et al.* 2009). Shortreed and Stockner (1983) observed no change in periphyton growth between reference and clearcut streams. In their meta-analysis of replicated studies, Richardson and Béraud (2014) found a large amount of variation in the direction and magnitude of changes across studies, which they attributed to environmental and site-specific variation.

Ultimately, the lack of a treatment effect may have resulted from the fact that approximately half of the tile pairs along unbuffered stream reaches during the post-harvest period were in areas with dense cover provided by something other than overstory canopy, likely wood in the form of logging slash. In our review of wood recruitment and loading we observed an increase in wood loading in treated sites, especially in sites that lacked a riparian buffer in the RMZ (see Chapter 6 – *Wood Recruitment and Loading*). This is also evidenced by the bimodal distribution of canopy cover values for the tiles located in unbuffered stream reaches. However, we found no evidence of anything more than a weak correlation between our AFDM or chlorophyll *a* values and canopy cover. One consideration is that we do not have canopy values for the precise tile pair locations, which could impact our ability to detect an association. It is possible that light saturation limited periphyton growth in unbuffered reaches with little shading relative to areas with heavy shading by either closed canopy or heavy wood loading. Others have noted that high light levels can inactivate chlorophyll *a* and that photosynthesis is more efficient at moderate light levels (Murphy 1998; Kiffney *et al.* 2004; Danehy *et al.* 2007). If light intensity was negatively impacting periphyton growth in our buffer treatment sites we could expect this influence to be greatest during the years immediately after harvest, when shade provided by overstory canopy is reduced. In another study, Kiffney and colleagues (2014) found a reduction in algal biomass in experimental channels related to increased density of small wood less than 10 cm diameter and 1 m in length, which they attributed to increased consumption by primary consumers.

The lack of change in chlorophyll *a* pre- to post-harvest, the overall increase in AFDM in the reference and treatments in the post-harvest period, and the lack of a strong correlation between periphyton AFDM and chlorophyll *a* and shading would suggest that something other than light is limiting periphyton growth in our study streams. Nutrient limitation is one possibility. Other studies have shown that while periphyton growth is limited primarily by light, nutrients become a limiting factor when adequate light is available (Shortreed and Stockner 1983; Murphy 1998; Kiffney and Richardson 2001). Enrichment of streams with both nitrogen and phosphorus or with phosphorus alone increases periphyton growth under varying light conditions (Shortreed and Stockner 1983; Kiffney and Richardson 2001; Silins *et al.* 2014). In our study streams, total nitrogen and nitrate export increased post-harvest. Shortreed and Stockner (1983) also observed an increase in light and nitrogen, but no change in phosphorus or periphyton growth in their treatment sites relative to their reference sites following harvest of a stream in British Columbia. This led them to conclude that periphyton growth in their stream was limited by phosphorus. However, the minimum phosphorus export concentrations recorded during the post-treatment period of our study, while variable, were higher (5–36 µg/L; see Chapter 9 – *Nutrient Export*) than in the enriched treatment sites of other studies, where maximum concentrations were approximately 5 µg/L (Kiffney and Richardson 2001; Mallory and Richardson 2005).

High stream flows and suspended sediments can scour periphyton from their substrate (Shortreed and Stockner 1983; Allan 1995), although these factors were unlikely to have influenced our lack of treatment response since flow and suspended sediment were consistently low during both summer sample intervals (see Chapter 8 – *Discharge*). Sediment can inhibit periphyton growth through accumulation in the periphyton matrix (Kiffney and Bull 2000; Kiffney *et al.* 2003). We attempted to limit sediment accumulation on tiles by suspending them in the water column; however, overall depth of these small headwater streams is limited and many tiles did end up close to, or partially supported by, the stream bed substrates. While most tiles did have some sediment accumulation at the time of sample collection, we observed no difference in the amount of sediment accumulation on tiles by treatment and there was no evidence of anything greater than a weak correlation between the proportion of a tile that was covered in fine sediment and the corresponding AFDM or chlorophyll *a* value. It should be noted, however, that we know nothing of variation in the coverage of the tiles by sediment through time, which could potentially have a complex influence on both AFDM and chlorophyll *a*.

Post-harvest changes in temperature may influence periphyton growth. Studies have measured an increase in periphyton biomass (Noel *et al.* 1986; Rosemond 1994; Kiffney *et al.* 2003) and respiration rates (Phinney and McIntire 1965; Rosa *et al.* 2013) with an increase in temperature. While we measured an increase in summer seven-day maximum stream temperatures of 1°C in the 100% and FP treatments and 3°C in the 0% treatments in the first year post-harvest, and lower but still elevated temperatures in the second year post-harvest (see Chapter 7 – *Stream Temperature and Cover*), these results did not equate to an observed difference in periphyton biomass or chlorophyll *a* in the treatment sites compared to the reference sites.

Seasonality has been shown to influence periphyton biomass accrual and community composition. Francoeur and colleagues (1999) observed the greatest biomass accrual rates during the summer, while Kiffney and colleagues (2003) found that periphyton biomass in their clearcut sites was significantly greater than in their treatment sites during the spring and summer. Streams flowing through harvested timber stands also experience a shift in community composition from filamentous green algae in the spring to diatoms during the summer, fall, and winter, compared with forested streams that consist primarily of diatoms year round (Bilby and Bisson 1992). While we sampled periphyton during the summer, we did not collect samples during the spring and so we may have missed a response in biomass or chlorophyll *a* associated with the growth of filamentous green algae.

Some researchers have found that diatoms dominate headwater streams (Danehy and Bilby 2009) and still others have shown that while diatoms dominate shaded streams, filamentous algae are most abundant seasonally in streams flowing through harvested areas (Murphy *et al.* 1981; Lowe *et al.* 1986; Noel *et al.* 1986; Bilby and Bisson 1992). While we did not investigate periphyton community composition, we did not observe the great increase in visible filamentous algae that others have noted along streams with loss of overstory canopy related to timber harvest or wildfire (Lyford and Gregory 1975; Shortreed and Stockner 1983; Silins *et al.* 2014). Species composition may better reflect the effects of harvest (Naymik *et al.* 2005). Species composition of periphyton can directly impact the taxa that feed upon it. Diatoms are more nutritious than filamentous algae, which most stream invertebrates reject (Murphy 1998). While we did see changes in macroinvertebrate scraper export, we saw variable response between treatments, which makes it difficult to determine if scrapers were responding to changes in species composition (see Chapter 14 – *Macroinvertebrate Export*).

Grazer consumers may exert top-down control of periphyton growth (Mallory and Richardson 2005; Kiffney *et al.* 2014), but their influence was probably limited in our study streams. While we did detect a post-harvest increase in macroinvertebrate scraper export in the reference and FP treatment (see Chapter 14 – *Macroinvertebrate Export*), this result does not help explain the lack of difference in biofilm and periphyton accrual between treatments. Further, we did not often observe active grazing on tiles at the time of removal. Hill and colleagues (1995) found that periphyton accrual was more strongly influenced by snail density than by light regime, with the effect of light only expressed when snail density was low. While we did at times observe evidence of grazing by snails, these observations were not common, and we do not believe it likely that snail density in our study sites approached the densities examined in this latter experiment (up to 970 snails/m^2). Coastal Tailed Frog larvae are grazers and may have an impact on periphyton production. Mallory and Richardson (2005) found that periphyton production was under simultaneous top-down (tadpole grazing) and bottom-up (light) control; however, the relative importance of these effects was stream-dependent. While we detected evidence of a significant increase in tailed frog larvae density in the 100% and FP treatments, this change was inconsistent among treatments, with no change detected in the reference and 0% treatments. Therefore, since the within-treatment change was, for the most part, similar between treatments, we find it unlikely that this increase in tailed frog larvae at only two of four treatments had an impact on our results for AFDM and chlorophyll *a* (see Chapter 15 – *Stream-associated Amphibians*).

A further important consideration is that our measures were static measures of AFDM and chlorophyll *a* accumulation at a single point in time, not measures of the rate of algal accumulation. Periphyton biomass goes through a cycle of initial accrual (colonization and growth) and loss (death, grazing, sloughing; Biggs 1996b). Since a variety of factors influence algal communities (Biggs *et al.* 1998), the cycle of accrual and loss is likely asynchronous among streams, within streams, and among substrates (Biggs 1996a), though we used artificial substrates to control for the latter. Some have recommended frequent sampling during a periphyton colonization period rather than for a standard length of time. If accrual and loss in and among our study streams was not synchronized, this could have increased variability since sampling could have occurred during variable times in the accrual and loss cycle, thus hindering our ability to detect a treatment effect even if one existed. For example, in two studies conducted on coastal streams in British Columbia (Kiffney and Bull 2000; Kiffney *et al.* 2003) researchers found that peak chlorophyll *a* biomass occurred differentially between clearcuts and controls. We did not test the time it took to reach peak biomass in each treatment group, so we cannot address this issue formally. It is possible that more frequent sampling to detect peak biomass or an increased sample size (number of tile pairs) across all of our study sites would have improved our ability to detect a difference if one exists.

Our modest sample size, large variability in instream wood cover and thus light penetrating to our sample stations, collection of samples only in summer, and our static measures of AFDM and chlorophyll *a* may have decreased our ability to detect changes. Ultimately, while we do not know for certain what physical and/or biological processes are controlling periphyton growth in our study streams, we have no evidence of a difference in the response of biofilm or periphyton production to clearcut logging in our buffer treatments.

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APPENDIX 13-A. DATA TABLES

Appendix Table 13-A-1. The average ash-free dry mass (AFDM; mg/cm^2) and chlorophyll *a* (µg/cm^2) for the early and late summer sample intervals, by block, treatment and year. "--" indicates that there was no sample for the corresponding analysis and period (spring or summer) either because no sample was collected or due to missing bottle weights from the lab contracted to do the analyses.

Block	**Treatment**	**Year**	**AFDM (mg/cm^2)**		**Chlorophyll *a* (µg/cm^2)**	
			Early	**Late**	**Early**	**Late**
OLYM	REF	2006	0.03	–	0.54	–
		2007	0.04	0.09	0.30	0.17
		2008	0.05	0.02	0.63	0.61
		2009	0.18	0.08	0.43	1.24
		2010	0.10	0.07	0.45	0.28
	100%	2006	0.02	–	0.08	–
		2007	0.03	0.02	0.02	0.01
		2008	0.06	0.02	0.41	0.02
		2009	0.05	0.02	0.22	0.02
		2010	0.06	0.04	0.12	0.05
	FP	2006	0.06	–	0.11	–
		2007	0.06	0.04	0.73	0.36
		2008	0.08	0.07	0.40	0.51
		2009	0.09	0.10	0.30	0.32
		2010	0.17	0.18	0.53	0.57
	0%	2006	0.04	–	0.03	–
		2007	0.04	0.06	0.50	0.06
		2008	0.12	0.04	0.72	0.45
		2010	0.17	0.18	0.65	2.29
WIL1	REF	2006	0.05	–	0.30	–
		2007	0.10	0.07	0.79	0.46
		2008	0.12	0.13	0.87	1.58
		2009	0.20	0.12	1.10	1.77
		2010	0.16	0.19	1.36	0.56
	100%	2006	0.04	–	1.05	–
		2007	0.12	0.04	0.91	0.34
		2008	0.09	0.16	0.56	1.11
		2009	0.22	0.15	0.38	1.13
		2010	0.10	0.26	0.58	1.32
	FP	2006	0.05	–	0.61	–
		2007	0.11	0.06	1.39	0.77
		2008	0.06	0.06	1.19	0.96
		2009	0.17	0.11	0.84	0.85
		2010	0.14	0.13	0.41	1.09

Appendix Table 13-A-1. (continued)

Block	Treatment	Year	AFDM (mg/cm^2)		Chlorophyll *a* (µg/cm^2)	
			Early	Late	Early	Late
	0%	2006	0.04	–	0.44	–
		2007	0.06	0.06	0.68	0.36
		2008	0.09	0.04	0.35	0.54
		2009	0.16	0.07	0.59	0.56
		2010	0.08	0.15	0.31	0.59
WIL2	REF1	2006	0.04	–	0.60	–
		2007	0.06	0.02	0.69	0.40
		2008	0.09	0.04	0.02	0.65
		2010	0.09	0.09	0.41	0.48
	REF2	2006	0.04	–	0.54	–
		2007	0.07	0.08	0.44	0.15
		2008	0.07	0.07	0.76	0.55
		2009	0.15	0.08	1.29	0.40
		2010	0.11	0.12	1.07	0.34
	100%	2006	0.04	–	0.15	–
		2007	0.08	0.05	0.22	0.24
		2008	0.08	0.05	0.65	0.32
		2010	0.19	0.13	1.22	1.19
	0%	2006	0.06	–	0.12	–
		2007	0.08	0.06	0.49	0.34
		2008	0.12	0.07	0.34	0.44
		2009	0.09	0.11	0.26	0.40
		2010	0.16	0.17	0.17	0.56
WIL3	REF	2007	0.09	0.02	0.13	0.22
		2008	0.04	0.03	0.15	0.15
		2009	0.07	0.06	0.85	0.08
		2010	0.12	0.09	0.97	0.70
	100%	2006	–	–	0.06	–
		2007	0.04	0.04	0.37	0.04
		2008	0.05	0.06	0.93	0.54
		2009	0.05	0.09	0.63	1.11
		2010	0.05	0.10	0.20	0.86
CASC	REF	2006	0.03	–	0.08	–
		2007	0.07	0.06	0.01	0.36
		2008	0.03	0.04	0.22	0.45
		2009	0.09	0.05	0.50	1.10
		2010	0.07	0.06	0.12	0.27
	FP	2007	0.08	0.13	0.31	0.14
		2008	0.11	0.08	0.75	0.85
		2009	0.10	0.08	1.35	2.37
		2010	0.09	0.14	0.64	0.76
	0%	2006	0.09	–	0.09	–
		2007	0.08	0.06	0.11	0.02
		2008	0.05	0.06	0.07	0.05
		2009	0.16	0.08	0.32	0.09
		2010	0.07	0.08	0.07	0.14

CHAPTER 14 - MACROINVERTEBRATE EXPORT

Stephanie Estrella and William Ehinger

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14-1. ABSTRACT

Headwater streams comprise a significant proportion of the landscape and macroinvertebrates exported from these stream networks serve as an important food source for downstream fish. Timber harvest in basins with non-fish-bearing streams may influence macroinvertebrate export through changes in organic matter inputs and primary production. We assessed the response of macroinvertebrate export from non-fish-bearing streams to timber harvest using a Before-After Control-Impact (BACI) design. The study sites were treated with the current Washington State Forest Practices buffer (FP treatment), a more extensive buffer (100% treatment), no buffer (0% treatment), or remained unharvested (reference). We collected macroinvertebrates using drift nets every six weeks and quantified macroinvertebrate export as numbers and biomass per day. Although we observed some changes after harvest, there were no major reductions in macroinvertebrate export and no major shifts in functional feeding groups associated with the three buffer treatments relative to the unharvested references within the limitations of the study design and sampling methodology. Observed changes in parasite, scraper, and Dixidae export in one or more of the buffer treatments were accompanied by changes in the references, which suggests environmental factors or natural variability in macroinvertebrate communities rather than a treatment effect. We found a treatment × period interaction for parasite and scraper export in numbers per day, and for collector-gatherer and parasite export in biomass per day ($P < 0.05$). Parasite export in numbers per day increased in the reference and decreased in the 100% and FP treatments post-harvest, whereas scraper export in numbers per day increased in the reference and in the FP treatment post-harvest ($P < 0.05$). Collector-gatherer export in biomass per day increased in the FP and 0% treatments, while parasite export in biomass per day decreased in the FP treatment ($P < 0.05$). We found a treatment × period interaction for Dixidae (Diptera) export in numbers and biomass per day, and Dixidae export increased in the reference and in the FP treatment for both metrics ($P < 0.05$). There was no treatment × period interaction for other taxonomic groups, but there were changes in export for some of the groups post-harvest and between treatments. Chironomidae and Baetis, a collector-gatherer, comprised a large proportion of individuals exported, and Baetis of biomass exported. Persistence of taxa such as Chironomidae and Baetis likely resulted from their multivoltinism, and their ability to quickly adapt to disturbances and use available food resources. Wood in the stream channels may have created depositional areas that maintained or enhanced food resources for collector-gatherers such as Baetis and some Chironomidae.

14-2. INTRODUCTION

Headwater streams comprise a significant proportion of stream networks (Benda *et al.* 2005; Richardson and Danehy 2007). In the Pacific Northwest, these streams often initiate from a bedrock channel head in mountainous terrain and consist of narrow, incised channels with steep hillslopes (Benda *et al.* 2005). Because of their high edge-to-area ratio and their close proximity to hillslopes, headwater streams are typically strongly coupled with hillslope and riparian processes, which provide the headwaters with a source of sediment, wood and other organic material, and shade from insolation (Benda *et al.* 2005; Richardson *et al.* 2005; Richardson and Danehy 2007).

Headwater stream ecosystems receive most of their energy from riparian vegetation through the contribution of allochthonous inputs, or those originating outside the stream channel (Cummins *et al.* 1983; Gregory *et al.* 1991; Bilby and Bisson 1992). Autotrophic production is generally limited in headwater streams, especially in the mesic Pacific Northwest, because of shading from riparian vegetation (Richardson and Danehy 2007). Allochthonous inputs are retained in headwater streams as detritus in depositional areas upstream of wood dams, where they are made available for processing and consumption by microbes and aquatic macroinvertebrates (Anderson *et al.* 1978; Bilby and Likens 1980; Bilby 1981; Gregory *et al.* 1991). Consequently, macroinvertebrate taxa of headwater systems typically consist of those specialized in shredding allochthonous inputs and collecting the resulting particulate organic matter.

Removal of trees from hillslopes and riparian areas can reduce or eliminate the sources of allochthonous inputs and shade to headwater streams. This can potentially reduce the quantity and change the composition of the instream detritus, and thus decrease the abundance of macroinvertebrate shredders that are dependent on allochthonous food sources (Bilby and Bisson 1992). On the other hand, as inputs of organic material decrease and insolation increases, periphyton growth may also increase, in which case stream energy production may shift from a primarily allochthonous-based system to an autochthonous-based system (Bilby and Bisson 1992; Wipfli and Gregovich 2002; Richardson and Danehy 2007). This response could result in an increase in the abundance of macroinvertebrate scrapers that graze on periphyton.

Studies have found significantly higher densities and/or biomass of benthic macroinvertebrates in streams following timber harvest (e.g., Newbold *et al.* 1980; Murphy *et al.* 1981; Hawkins *et al.* 1982; Noel *et al.* 1986; Fuchs *et al.* 2003; Haggerty *et al.* 2004; Hernandez *et al.* 2005; Danehy *et al.* 2007; Richardson and Béraud 2014). Functional feeding groups, however, have varied in their response to timber harvest. Density and/or biomass of benthic collector-gatherers (Hawkins *et al.* 1982; Haggerty *et al.* 2004; Hernandez *et al.* 2005) and benthic shredders (Hawkins *et al.* 1982; Kobayashi *et al.* 2010) were higher in streams following harvest. While some studies found higher densities and/or biomass of benthic scrapers (Hernandez *et al.* 2005) and benthic predators (Murphy and Hall 1981) in streams in harvested basins, others did not find a difference (Fuchs *et al.* 2003; Haggerty *et al.* 2004; Danehy *et al.* 2007). Gravelle and colleagues (2009) did not observe any significant changes in functional feeding group composition following timber harvest.

Few studies have measured the response of macroinvertebrate drift to timber harvest. Drift describes the phenomenon when macroinvertebrates unintentionally or intentionally enter the water column and are carried downstream with the current. Passive drift occurs when an invertebrate loses its hold and is scoured from the substrate, usually during an increase in stream flow (Wiley and Kohler 1984) or sedimentation (Culp *et al.* 1986; Suren and Jowett 2001; Larsen and Ormerod 2010). Active drift, on the other hand, is voluntary and may occur in response to abiotic conditions (Wiley and Kohler 1980; James *et al.* 2009), food resources (Richardson 1991; Richards and Minshall 1988; Hinterleitner-Anderson *et al.* 1992; Siler *et al.* 2001), overpopulation (Waters 1961), competition (Hildebrand 1974), and presence of predators (Wiley and Kohler 1984; Lancaster 1990). Drift can indicate poor stream conditions as macroinvertebrates drift to more suitable habitats, or favorable stream conditions leading to an overabundance of benthic invertebrates. In a study of the response of macroinvertebrate drift to different degrees of tree retention, Musslewhite and Wipfli (2004) did not find a relationship between drift export and tree retention treatment. Other studies, however, have shown a response in macroinvertebrate drift to other land use practices that result in changes in flows (Collier and Quinn 2003; Tonkin and Death 2013) and sediment and chemical inputs (Lauridsen and Friberg 2005; O'Callaghan *et al.* 2015; Magbanua *et al.* 2016).

Non-fish-bearing headwater streams are more extensive than fish-bearing streams in the landscape and export macroinvertebrates throughout the year. Given this, Type N streams may contribute significant quantities of drifting invertebrates to downstream habitats (Piccolo and Wipfli 2002; Wipfli and Gregovich 2002; Wipfli 2005). Timber harvest practices that alter the quantity of macroinvertebrates exported from non-fish-bearing streams may therefore have implications for downstream fish populations. Composition of the export is also important, as fish stomach content analyses have shown a greater proportion of Diptera and Ephemeroptera (Meehan 1996) and a greater proportion of invertebrates that feed on algae or algal-derived detritus (Bilby and Bisson 1992) in the diets of fish.

The Type N Experimental Buffer Treatment Study (Type N Study) was designed to evaluate the response of a variety of riparian and instream processes to different stream buffer treatments pre- and post-harvest. This component of the study examined the changes in numbers and biomass of macroinvertebrate drift, or export, from Type N Waters treated with different riparian buffers. We hypothesized that changes in habitat conditions (e.g., water temperature, sedimentation) and a decrease in allochthonous inputs may initiate drift in some taxa and functional feeding groups, such as shredders. In addition, we expected that an increase in light from timber removal would stimulate periphyton growth and thus provide an alternative food resource for other taxa and feeding groups, such as collector-gatherers and scrapers, which would increase in numbers and/or biomass.

14-3. OBJECTIVES

The overarching objective of this part of the study was to examine the magnitude and direction of change in macroinvertebrate export after timber harvest in three experimental buffer treatments. The research questions were:

1) What is the magnitude of change in macroinvertebrate numbers and biomass exported from each of three experimental buffer treatments following timber harvest relative to an unharvested reference site?

2) What are the differences in the magnitude of the change in macroinvertebrate numbers and biomass exported among the three experimental buffer treatments?

14-4. METHODS

14-4.1. SITE DESCRIPTION

We limited sampling of macroinvertebrate export to the Olympic and Willapa 1 blocks because of the time and cost required to collect and process samples. In addition, these blocks were being monitored for discharge and we needed to quantify flows to estimate macroinvertebrate exports. The Olympic and Willapa 1 blocks consisted of eight study sites with two replicates of each treatment (**Figure 14-1**). The 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. The sites were located in managed Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*)-dominated, second-growth forests on private, state, and federal land with stand ages ranging from 30 to 80 years old. Sites were located in areas dominated by competent lithology types, with average Np channel gradients ranging from 16 to 31% (see **Table 2-6** in Chapter 2 – *Study Design*). These sites can be considered as representative of Type N basins located in second-growth forests on lands managed for timber production, dominated by competent lithologies, located in western Washington, and consistent with our other site selection criteria (size, gradient, etc.; see **Table 2-1** in Chapter 2 – *Study Design*).

14-4.2. EXPERIMENTAL TREATMENTS

The four sites within each block received one of the four following treatments (see **Figure 2-2** in Chapter 2 – *Study Design*). A 30-ft (9.1-m) equipment limitation zone was maintained along all Type Np and Ns Waters, regardless of assigned treatment.

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 the entire perennial stream length buffered with a two-sided 50-ft (15.2-m) riparian buffer,

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 Riparian Management Zone (RMZ), including buffers prescribed for sensitive sites (side-slope and headwall seeps, headwater springs, Type Np intersections and alluvial fans), and

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

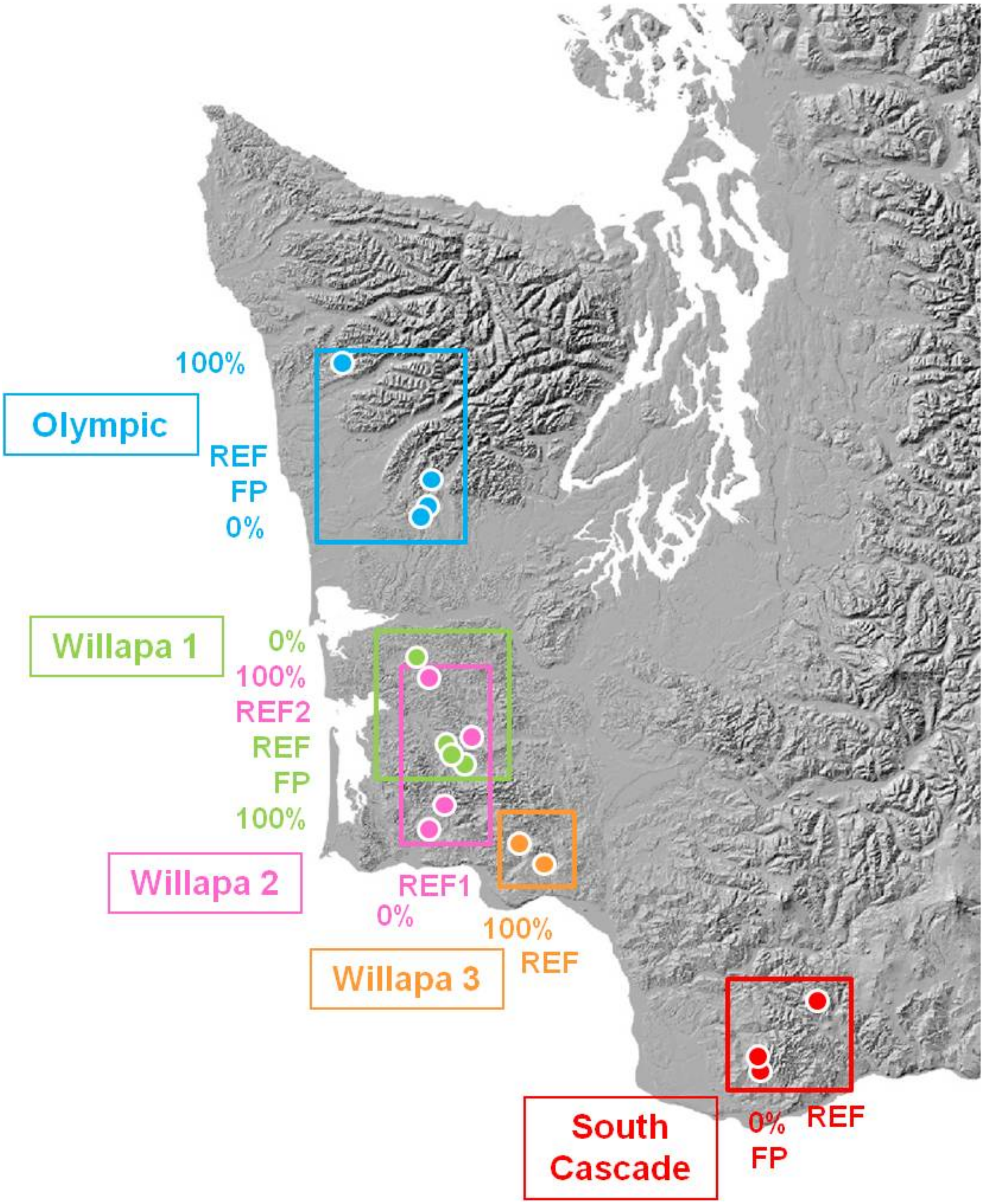

Figure 14-1. Distribution of the Type N study sites and treatments. Study sites are blocked and based on geography. Drift samples were collected from the Olympic (color-coded blue) and Willapa 1 (color-coded green) blocks. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment.

14-4.3. SAMPLE COLLECTION

We collected macroinvertebrates from the drift in conjunction with the detritus export sampling (described in Chapter 12 – *Litterfall Input and Detritus Export*). Drift sampling began in the fall of 2006 after installation of hydrological monitoring equipment. In 2008 and 2009, the sites received one of four treatments. Sampling ceased in all sites at the end of September 2011 when at least two years of post-harvest data collection were completed in all sites.

We collected drift samples at the location of the hydrological monitoring equipment to enable flow quantification. We installed the hydrological equipment as close to the F/N break as logistically feasible, although stream channel morphology often restricted installation of the equipment to stream reaches well away from the F/N break (**Table 14-1**). In the OLYM-REF and OLYM-100%, we installed the hydrological equipment in culverts, which were located well upstream of the F/N break but in locations representative of the treatment units (**Figure 14-2**). The flumes installed in the OLYM-FP and OLYM-0%, on the other hand, were located 50 m and 240 m, respectively, downstream of both the F/N break and the treatment units (**Figure 14-3**). While the flumes in the Willapa block sites were not located exactly at the F/N break, they were still in locations representative of their respective treatments (**Figure 14-4**).

Table 14-1. Distance of the hydrological measuring equipment and drift net to the F/N break and treatment unit, and the adjacent riparian management prescription. A positive distance denotes an upstream direction from the F/N break or treatment unit, while a negative distance a downstream direction. OLYM = Olympic Block; WIL1 = Willapa Block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; NAH = no adjacent harvest; BUF = buffered; UNB = unbuffered (no riparian buffer).

Block	Treatment	Equipment Type	Distance from		Riparian Management Prescription
			F/N Break (m)	Treatment Unit (m)	
OLYM	REF	Culvert	357	–	NAH
	100%	Culvert	229	0	BUF
	FP	Flume	−50	−50	NAH[1]
	0%	Flume	−240	−240	NAH
WIL1	REF	Flume	−25	–	NAH
	100%	Flume	140	0	BUF
	FP	Flume	10	0	BUF
	0%	Flume	19	0	UNB

[1] Flume downstream of the F/N break. There was no adjacent harvest until May 2011, when the surrounding stand was harvested and a Type F buffer was left adjacent to the stream. The Type F buffer is visible in **Figure 14-3a**.

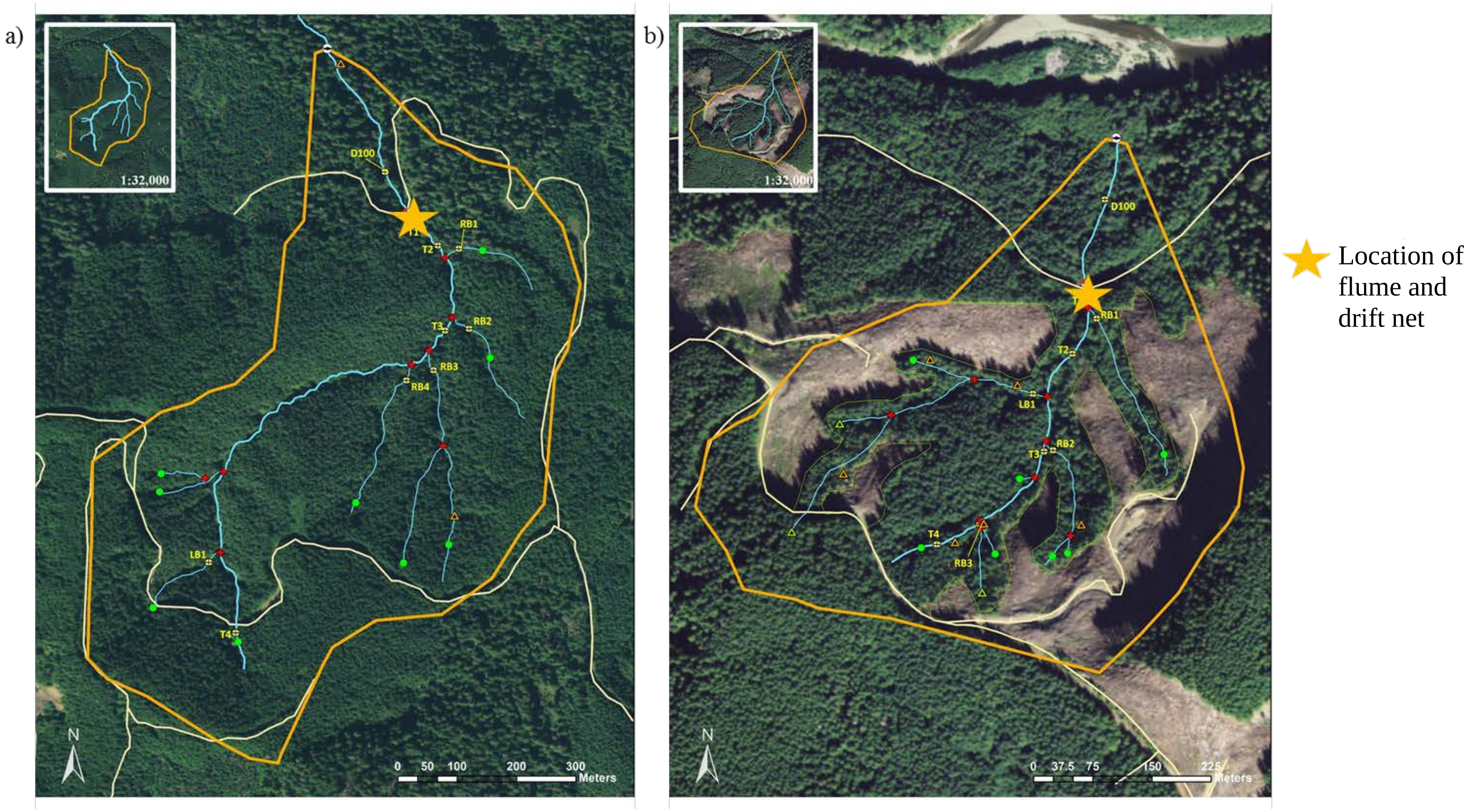

Figure 14-2. Location of the culvert and drift net installed in the Olympic Block (a) unharvested reference (OLYM-REF) and (b) 100% treatment (OLYM-100%).

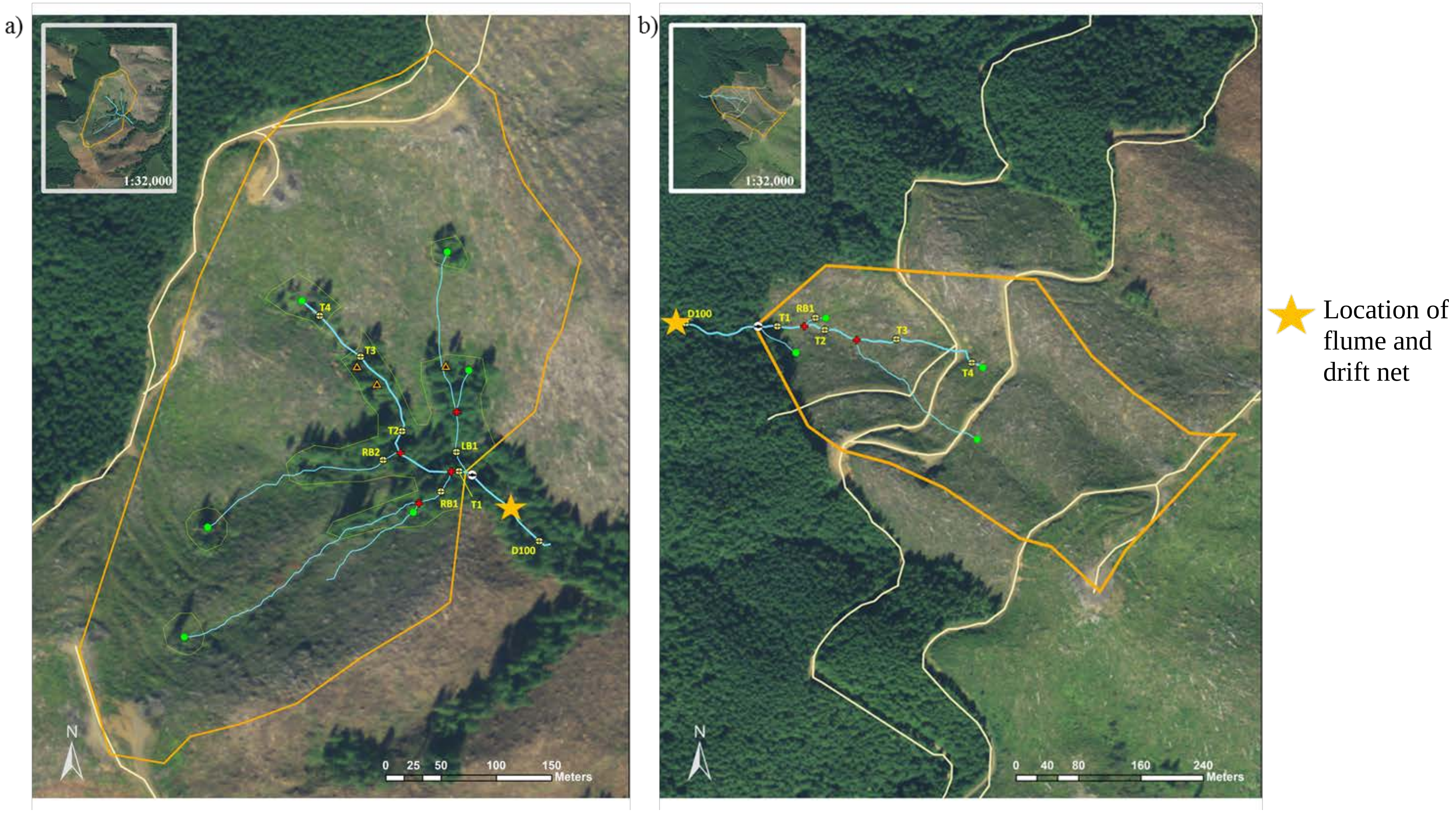

Figure 14-3. Location of the flume and drift net installed in the Olympic Block (a) Forest Practices (FP) treatment (OLYM-FP) and (b) 0% treatment (OLYM-0%). In the OLYM-FP, there was no harvest adjacent to the flume until May 2011, when the surrounding stand was harvested and a Type F buffer was left along the stream channel.

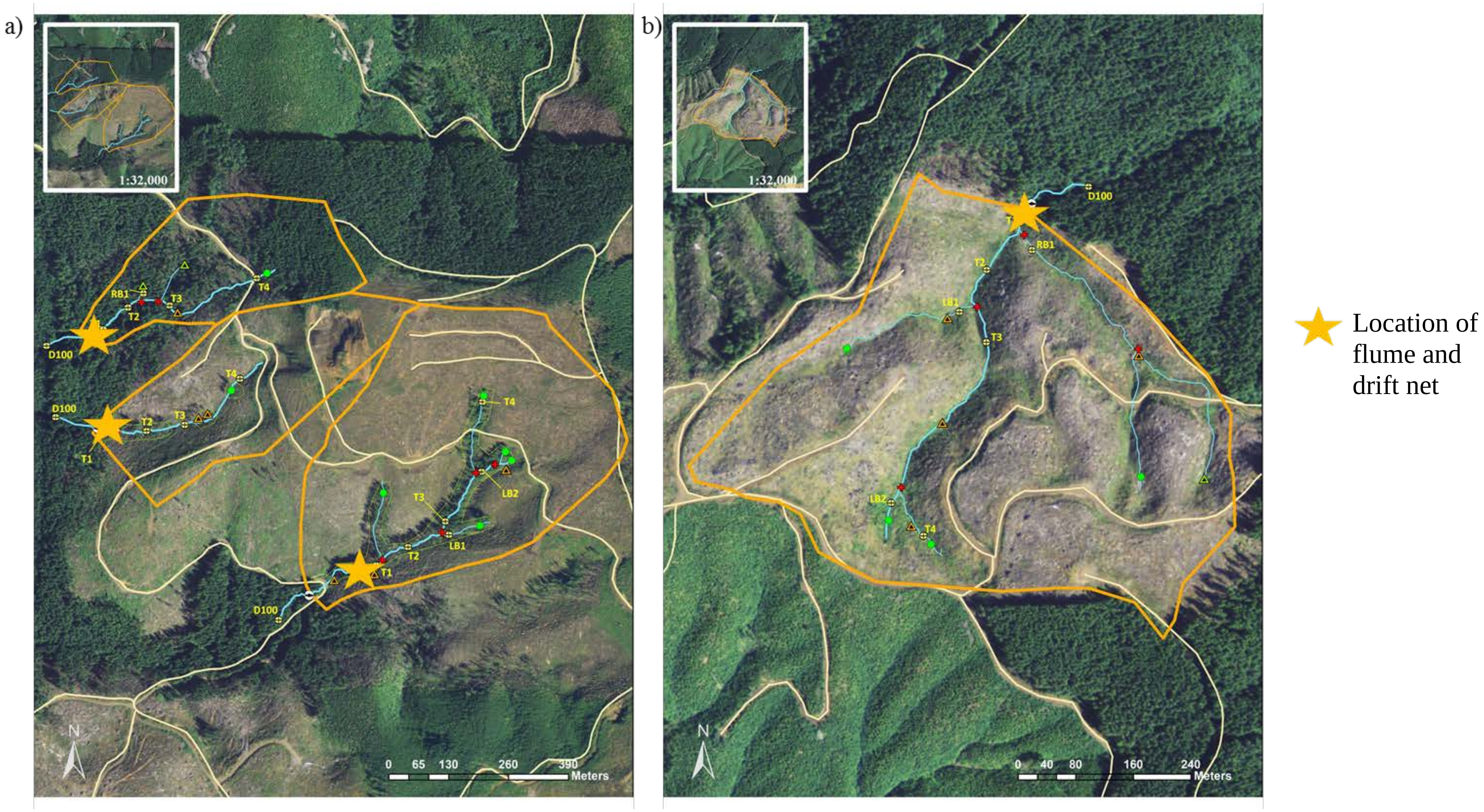

Figure 14-4. Location of the flume and drift net installed in the Willapa 1 Block (a) (from north to south) unharvested reference (WIL1-REF), Forest Practices (FP) treatment (WIL1-FP), and 100% treatment (WIL1-100%), and (b) 0% treatment (WIL1-0%).

In the six flume sites, we wrapped a 250-μm mesh drift net around an aluminum lip installed below the mouth of the flume and secured with zip ties to eyebolts attached to the flume (**Figure 14-5a**). In the two culvert sites, we attached the drift net with zip ties to a metal rod that extended along the streambed across the stream channel perpendicular to flow (**Figure 14-5b**). We installed two other metal rods oriented vertically into the streambed to support the upper portion of the drift net to allow sampling of the water column. Although this method did not sample the entire stream cross-section, the net sampled most of the cross-section and always included the thalweg, the area of most active transport. The area that was not sampled consisted mostly of the depositional margins of the stream channel.

Figure 14-5. Drift net deployment in the Type N Study sites with a flume (a) and without a flume (b).

We sampled drift for approximately one continuous 24-hour period every six weeks throughout the year for a total of about nine samples per year per site. The number of samples per year varied for some of the sites, however, depending on the harvest schedule, site accessibility, and flow conditions (**Appendix Table 14-A-1**). Field visit scheduling and high flows during some of the fall, winter, and spring sampling efforts often forced shorter deployment periods to prevent the mesh from clogging with detritus and the loss of the net (**Appendix Table 14-A-1**). While we were able to sample high flows in some cases, we were not able to target specific storm events. For each sampling effort, we recorded the drift net deployment and retrieval date and time, and estimated the proportion of the flow volume sampled by the drift net (Estrella 2006). We used *in situ* pressure transducers to measure stage height at 10-minute intervals. At the six flume sites, discharge was calculated from the flow versus stage height curve for that size flume (see Chapter 8 – *Discharge*). For the two culvert sites, discharge was estimated from a site-specific flow versus stage height curve developed over the course of the study (described in Chapter 8 – *Discharge*).

We preserved the samples in 70% ethanol. Drift samples analyzed for the trophic pathways component of the study (see Supplement 2 – *Stable Isotopes Analysis*) were stored in stream water and kept on ice until processed.

14-4.4. SAMPLE PROCESSING

We rinsed the drift samples through 1-mm and 250-μm nested sieves and sorted the macroinvertebrates from the detritus. All specimens were identified, measured to the nearest millimeter, and enumerated by length class. We identified most specimens to the lowest practical taxonomic level as designated in Plotnikoff and White (1996) using keys in Merritt and Cummins (1996) and Stewart and Stark (1993). We left some taxonomic groups, such as Acarina and Collembola, at higher taxonomic levels. We classified macroinvertebrates into functional feeding group using Merritt and Cummins (1996) and Wisseman (1998). We did not identify Chironomidae (Diptera) past the family level. At the level of taxonomic resolution used, we could not assign feeding group to chironomids so we considered this taxon its own feeding group. Adult macroinvertebrates were classified into the functional feeding group of their larval stage. Although we collected both aquatic and terrestrial macroinvertebrates, we were not able to classify some of the taxa as aquatic or terrestrial at the level of taxonomic resolution used for the taxa (e.g., Acarina and Collembola).

14-4.5. STATISTICAL ANALYSIS

We quantified macroinvertebrate export rate by numbers of individuals and by biomass exported per day. We initially included all samples in the analysis, but because invertebrate drift is highest from sunset to sunrise (e.g., Waters 1965; O'Hop and Wallace 1983; Stewart and Szczytko 1983; Allan and Russek 1985; Hetrick *et al.* 1998a) we omitted the samples that were not collected overnight (**Appendix Table 14-A-1**). After we omitted those samples, there were only two winter samples remaining (WIL1-REF PRE1 22 February 2007 and WIL1-FP PRE1 22 February 2007) so we omitted those as well. For each of the remaining samples, we calculated numbers per day by dividing the number of each taxon by the drift net deployment time and then standardizing to a 24-hour sampling period and a 100% sampled stream flow volume.

We used published taxon-specific length-weight regression coefficients, derived from the relationship between the length of an invertebrate and its dry weight, to estimate biomass for each taxon and length class. We assigned family level coefficients to larva identified to family and genus, and order level coefficients to larva identified to order or family (if a family level coefficient was not given) using Benke and colleagues (1999). Similarly, we assigned family level coefficients to aquatic and terrestrial adults identified to family, and order-level coefficients to adults identified to order or family using Sabo and colleagues (2002). We used Francis (2009) and Hodar (1996) to assign coefficients to groups not listed elsewhere. Biomass per day was then calculated by multiplying the mass of each taxon and length class by numbers per day. For each sampling effort, we summed macroinvertebrate export rate in numbers and biomass per day by total macroinvertebrates, functional feeding group, and dominant macroinvertebrate order, family, or other taxonomic group.

The stream volume per 10-minute interval was summed for the entire deployment period and then standardized to a 24-hour day to give flow in cubic meters per day (m^3 day^{-1}). If part or all of the drift net detached from the flume or rebar during deployment (**Appendix Table 14-A-1**), the proportion of flow sampled when the net was installed and when the net was removed was averaged (this assumes that macroinvertebrate export was uniform across the stream cross-section). Flow was then divided by the basin area above the hydrological equipment to give flow per hectare.

We designated consistent treatment years for all four sites in a block because the landowners determined the timing and duration of the timber harvests, which were not consistent between sites, and because macroinvertebrate drift is dependent on stream flows and season. The pre-harvest period (PRE) consisted of the two pre-harvest years (PRE1 and PRE2) from the time when the first sample was collected in the fall of 2006 to the time when harvest first began in any site of the block. The post-harvest period (POST) consisted of the two post-harvest years (POST1 and POST2) following completion of harvest in the last site of the block. The harvest schedule, inaccessibility of sites due to harvest, snow, and road conditions, and inability to sample because of high stream flows resulted in differences in the number of samples collected per site pre- and post-harvest (**Appendix Table 14-A-1**).

We analyzed macroinvertebrate export for total macroinvertebrates, functional feeding group, and dominant order, family, or other taxonomic group with each value (i.e., each drift sample) as an observation. The data were not normally distributed so we calculated the base 10 log of the calculated means +1. We analyzed the data using a generalized linear mixed effects model (GLMM) analysis of variance (ANOVA) with treatment (REF, 100%, FP, and 0%), period (PRE and POST), and the treatment × period interaction as fixed effects, and site, flow per area, and season as random 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 the dependent variables). We used the Kenward-Roger method (Kenward and Roger 1997) for estimating the denominator degrees of freedom for tests of fixed effects and the contrasts because of the unbalanced design. We used SAS software version 9.4 for the GLMM analyses (SAS 2013).

We evaluated six hypotheses grouped under the two research questions:

1) What was the magnitude of change in mean macroinvertebrate export post-harvest in each treatment relative to an unharvested reference site?

We addressed this with three *post hoc* comparisons, testing the following hypothesis for each treatment:

$$\text{H0: } \Delta M_{\text{REF}} = \Delta M_{100} \quad (14\text{-}1)$$

$$\text{H0: } \Delta M_{\text{REF}} = \Delta M_{\text{FP}} \quad (14\text{-}2)$$

$$\text{H0: } \Delta M_{\text{REF}} = \Delta M_{0} \quad (14\text{-}3)$$

where: ΔM_{REF} is change in macroinvertebrate export from the unharvested reference site,
ΔM_{100} is change in macroinvertebrate export from the 100% treatment,
ΔM_{FP} is change in macroinvertebrate export from the FP treatment, and
ΔM_{0} is change in macroinvertebrate export from the 0% treatment.

2) Did the magnitude of the change differ among treatments?

We addressed this with three *post hoc* comparisons testing the hypotheses:

$$\mathrm{H0}: \Delta M_{100} = \Delta M_{\mathrm{FP}} \tag{14-4}$$

$$\mathrm{H0}: \Delta M_{100} = \Delta M_{0} \tag{14-5}$$

$$\mathrm{H0}: \Delta M_{\mathrm{FP}} = \Delta M_{0} \tag{14-6}$$

We hypothesized that changes in habitat conditions and a decrease in allochthonous inputs may initiate drift in some taxa and functional feeding groups, such as shredders. In addition, we expected that an increase in light from timber removal would stimulate periphyton growth and thus provide an alternative food resource for other taxa and feeding groups, such as collector-gatherers and scrapers.

We present the estimates of the GLMM ANOVA (**Tables 14-2**, **14-4**, **14-6**, and **14-8**) and estimates of the effects and the associated 95% confidence intervals for each comparison (**Tables 14-3**, **14-5**, **14-7**, and **14-9**). The P-values were not adjusted for the multiple comparisons because the large number of comparisons relative to the limited replication of each treatment (two) increases the chance of a Type II error and can mask subtle treatment effects. Instead, we consider the P-value, effect size, patterns of the effect size across the treatments, and sample size when interpreting the results.

14-5. RESULTS

14-5.1. TOTAL MACROINVERTEBRATES

14-5.1.1. *Numbers per Day*

The GLMM ANOVA showed that there was no treatment × period interaction for total macroinvertebrate export in numbers per day (P = 0.4975; **Table 14-2**). Post-hoc comparisons between the PRE and POST treatment periods showed an increase in total macroinvertebrate export in the reference (P = 0.0015) and in the FP treatment (P = 0.0122; **Table 14-3**). There was no difference between the reference and the buffer treatments (P >0.05; **Table 14-3**). Descriptive statistics for total macroinvertebrate export in numbers per day from each study site by treatment year are provided in **Appendix Table 14-B-1**.

Table 14-2. Results of the GLMM ANOVA for macroinvertebrate export in numbers per day (# day^{-1}) for total invertebrates and functional feeding group by treatment (TRMT), period (TRYR), and the treatment × period interaction (TRMT × TRYR). Comparisons in **bold** print indicate P <0.05. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Total Invertebrates and Functional Feeding Group	Effect	Num DF	Den DF	F-value	P-value
Total Invertebrates	TRMT	3	3.21	0.24	0.8651
	TRYR	1	102	18.43	<0.0001
	TRMT × TRYR	3	101	0.8	0.4975
Chironomidae	TRMT	3	2.42	0.2	0.8875
	TRYR	1	104	16.34	0.0001
	TRMT × TRYR	3	103	2.07	0.1086
Collector-Filterers	TRMT	3	3.9	0.25	0.8602
	TRYR	1	117	7.93	0.0057
	TRMT × TRYR	3	117	0.37	0.7763
Collector-Gatherers	TRMT	3	3.91	0.29	0.8311
	TRYR	1	117	44.73	<0.0001
	TRMT × TRYR	3	117	0.92	0.4350
Omnivores	TRMT	3	18.5	0.44	0.7294
	TRYR	1	118	0.02	0.8927
	TRMT × TRYR	3	118	1.33	0.2671
Parasites	TRMT	3	1.54	0.08	0.9651
	TRYR	1	95.2	4.4	0.0386
	TRMT × TRYR	3	94.5	4.15	**0.0083**
Predators	TRMT	3	2.81	0.67	0.6279
	TRYR	1	101	0.11	0.7404
	TRMT × TRYR	3	101	1.85	0.1430
Scrapers	TRMT	3	3.97	0.39	0.7694
	TRYR	1	116	16.84	<0.0001
	TRMT × TRYR	3	116	4.45	**0.0054**
Shredders	TRMT	3	3.51	0.54	0.6823
	TRYR	1	112	10.4	0.0017
	TRMT × TRYR	3	111	0.1	0.9624
Unknown	TRMT	3	4.25	1.02	0.4680
	TRYR	1	118	23.07	<0.0001
	TRMT × TRYR	3	117	0.66	0.5782

Table 14-3. Results of hypothesis tests described in section 14-4.5. for macroinvertebrate export in numbers per day (# day^{-1}) for total invertebrates and functional feeding group. Comparisons a, b, c, and d compare pre- and post-harvest macroinvertebrate export for each treatment, and comparisons e, f, g, h, i, and j compare post-harvest changes in macroinvertebrate export between the treatments. Comparisons in **bold** print indicate P <0.05. P-values were not adjusted for multiple comparisons.

Total Invertebrates and Functional Feeding Group	**Comparison**	**Change (# day^{-1})**	**P-value**	**95% C.I.**	
				Lower	**Upper**
Total Invertebrates	a. REF-Pre vs. Post	0.4367	**0.0015**	0.1713	0.7021
	b. 100%-Pre vs. Post	0.1864	0.1805	−0.0878	0.4605
	c. FP-Pre vs. Post	0.3598	**0.0122**	0.0802	0.6394
	d. 0%-Pre vs. Post	0.2020	0.1492	−0.0737	0.4777
	e. REF vs. 100%	−0.2503	0.1961	−0.6318	0.1312
	f. REF vs. FP	−0.0769	0.6930	−0.4624	0.3085
	g. REF vs. 0%	−0.2347	0.2265	−0.6174	0.1479
	h. 100% vs. FP	0.1734	0.3817	−0.2181	0.5649
	i. 100% vs. 0%	0.0156	0.9367	−0.3731	0.4043
	j. FP vs. 0%	−0.1578	0.4271	−0.5504	0.2349
Chironomidae	a. REF-Pre vs. Post	0.6635	**<0.0001**	0.3447	0.9824
	b. 100%-Pre vs. Post	0.3032	0.0815	−0.0386	0.6450
	c. FP-Pre vs. Post	0.3051	0.0804	−0.0376	0.6479
	d. 0%-Pre vs. Post	0.0945	0.5789	−0.2422	0.4313
	e. REF vs. 100%	−0.3603	0.1294	−0.8277	0.1071
	f. REF vs. FP	−0.3584	0.1320	−0.8265	0.1097
	g. REF vs. 0%	−0.5690	**0.0167**	−1.0327	−0.1052
	h. 100% vs. FP	0.0020	0.9936	−0.4821	0.4860
	i. 100% vs. 0%	−0.2086	0.3906	−0.6885	0.2712
	j. FP vs. 0%	−0.2106	0.3868	−0.6911	0.2699
Collector-Filterers	a. REF-Pre vs. Post	0.3255	0.1074	−0.0719	0.7228
	b. 100%-Pre vs. Post	0.1062	0.5965	−0.2900	0.5024
	c. FP-Pre vs. Post	0.3425	0.1032	−0.0705	0.7556
	d. 0%-Pre vs. Post	0.3746	0.0722	−0.0344	0.7835
	e. REF vs. 100%	−0.2193	0.4405	−0.7804	0.3418
	f. REF vs. FP	0.0171	0.9531	−0.5561	0.5902
	g. REF vs. 0%	0.0491	0.8649	−0.5211	0.6193
	h. 100% vs. FP	0.2363	0.4152	−0.3360	0.8087
	i. 100% vs. 0%	0.2684	0.3525	−0.3010	0.8378
	j. FP vs. 0%	0.0320	0.9133	−0.5492	0.6133
Collector-Gatherers	a. REF-Pre vs. Post	0.5872	**0.0006**	0.2554	0.9191
	b. 100%-Pre vs. Post	0.4541	**0.0076**	0.1232	0.7850
	c. FP-Pre vs. Post	0.7968	**<0.0001**	0.4519	1.1417
	d. 0%-Pre vs. Post	0.4404	**0.0119**	0.0989	0.7819
	e. REF vs. 100%	−0.1331	0.5748	−0.6017	0.3355
	f. REF vs. FP	0.2096	0.3877	−0.2690	0.6881
	g. REF vs. 0%	−0.1469	0.5425	−0.6230	0.3293
	h. 100% vs. FP	0.3427	0.1583	−0.1353	0.8206
	i. 100% vs. 0%	−0.0137	0.9544	−0.4893	0.4618
	j. FP vs. 0%	−0.3564	0.1486	−0.8418	0.1289

Table 14-3. (continued)

Total Invertebrates and Functional Feeding Group	Comparison	Change (# day^{-1})	P-value	95% C.I.	
				Lower	Upper
Omnivores	a. REF-Pre vs. Post	0.0206	0.8099	−0.1486	0.1898
	b. 100%-Pre vs. Post	0.0053	0.9506	−0.1643	0.1750
	c. FP-Pre vs. Post	−0.1264	0.1598	−0.3032	0.0505
	d. 0%-Pre vs. Post	0.1240	0.1645	−0.0516	0.2996
	e. REF vs. 100%	−0.0153	0.8997	−0.2549	0.2243
	f. REF vs. FP	−0.1470	0.2368	−0.3917	0.0978
	g. REF vs. 0%	0.1034	0.4025	−0.1404	0.3473
	h. 100% vs. FP	−0.1317	0.2895	−0.3767	0.1134
	i. 100% vs. 0%	0.1187	0.3375	−0.1254	0.3629
	j. FP vs. 0%	0.2504	**0.0490**	0.0012	0.4996
Parasites	a. REF-Pre vs. Post	0.3311	**0.0494**	0.0009	0.6613
	b. 100%-Pre vs. Post	−0.3830	**0.0352**	−0.7389	−0.0272
	c. FP-Pre vs. Post	−0.3760	**0.0343**	−0.7237	−0.0284
	d. 0%-Pre vs. Post	−0.2987	0.0858	−0.6403	0.0429
	e. REF vs. 100%	−0.7141	**0.0043**	−1.1995	−0.2288
	f. REF vs. FP	−0.7071	**0.0043**	−1.1866	−0.2277
	g. REF vs. 0%	−0.6298	**0.0099**	−1.1049	−0.1548
	h. 100% vs. FP	0.0070	0.9778	−0.4903	0.5043
	i. 100% vs. 0%	0.0843	0.7351	−0.4088	0.5775
	j. FP vs. 0%	0.0773	0.7533	−0.4101	0.5647
Predators	a. REF-Pre vs. Post	0.2286	0.0739	−0.0225	0.4798
	b. 100%-Pre vs. Post	−0.1754	0.1841	−0.4354	0.0847
	c. FP-Pre vs. Post	−0.0619	0.6427	−0.3260	0.2022
	d. 0%-Pre vs. Post	−0.0781	0.5533	−0.3388	0.1826
	e. REF vs. 100%	−0.4040	**0.0288**	−0.7655	−0.0426
	f. REF vs. FP	−0.2906	0.1169	−0.6550	0.0739
	g. REF vs. 0%	−0.3067	0.0958	−0.6687	0.0552
	h. 100% vs. FP	0.1135	0.5450	−0.2571	0.4840
	i. 100% vs. 0%	0.0973	0.6012	−0.2708	0.4654
	j. FP vs. 0%	−0.0162	0.9311	−0.3873	0.3549
Scrapers	a. REF-Pre vs. Post	0.8738	**<0.0001**	0.5152	1.2324
	b. 100%-Pre vs. Post	0.0592	0.7454	−0.3011	0.4195
	c. FP-Pre vs. Post	0.5018	**0.0095**	0.1251	0.8785
	d. 0%-Pre vs. Post	0.0880	0.6420	−0.2860	0.4620
	e. REF vs. 100%	−0.8146	**0.0019**	−1.3229	−0.3063
	f. REF vs. FP	−0.3720	0.1592	−0.8921	0.1480
	g. REF vs. 0%	−0.7858	**0.0033**	−1.3039	−0.2677
	h. 100% vs. FP	0.4426	0.0953	−0.0786	0.9638
	i. 100% vs. 0%	0.0288	0.9127	−0.4905	0.5481
	j. FP vs. 0%	−0.4138	0.1253	−0.9446	0.1170

Table 14-3. (continued)

Total Invertebrates and Functional Feeding Group	Comparison	Change (# day^{-1})	P-value	95% C.I.	
				Lower	Upper
Shredders	a. REF-Pre vs. Post	0.3195	**0.0439**	0.0089	0.6301
	b. 100%-Pre vs. Post	0.1976	0.2565	−0.1457	0.5409
	c. FP-Pre vs. Post	0.2688	0.1156	−0.0671	0.6047
	d. 0%-Pre vs. Post	0.2885	0.0862	−0.0418	0.6188
	e. REF vs. 100%	−0.1219	0.6028	−0.5847	0.3409
	f. REF vs. FP	−0.0507	0.8265	−0.5082	0.4067
	g. REF vs. 0%	−0.0310	0.8926	−0.4843	0.4224
	h. 100% vs. FP	0.0712	0.7695	−0.4090	0.5514
	i. 100% vs. 0%	0.0910	0.7057	−0.3852	0.5671
	j. FP vs. 0%	0.0198	0.9339	−0.4513	0.4908
Unknown	a. REF-Pre vs. Post	0.5107	**0.0112**	0.1183	0.9030
	b. 100%-Pre vs. Post	0.5981	**0.0033**	0.2032	0.9929
	c. FP-Pre vs. Post	0.2419	0.2441	−0.1673	0.6511
	d. 0%-Pre vs. Post	0.5883	**0.0045**	0.1860	0.9906
	e. REF vs. 100%	0.0874	0.7563	−0.4692	0.6440
	f. REF vs. FP	−0.2687	0.3497	−0.8356	0.2981
	g. REF vs. 0%	0.0777	0.7847	−0.4843	0.6396
	h. 100% vs. FP	−0.3562	0.2173	−0.9248	0.2124
	i. 100% vs. 0%	−0.0098	0.9727	−0.5734	0.5539
	j. FP vs. 0%	0.3464	0.2343	−0.2274	0.9202

14-5.1.2. *Biomass per Day*

The GLMM ANOVA showed that there was no treatment × period interaction for total macroinvertebrate export in biomass per day (mg day^{-1}) (P = 0.7605; **Table 14-4**).

Post-hoc comparisons showed that there was no difference in total macroinvertebrate export between the PRE and POST treatment periods for each treatment as well as no difference in total macroinvertebrate export between the reference and the buffer treatments (P >0.05; **Table 14-5**). Descriptive statistics for total macroinvertebrate export in mg day^{-1} from each study site by treatment year are provided in **Appendix Table 14-B-2**.

Table 14-4. Results of the GLMM ANOVA for macroinvertebrate export in biomass per day (mg day^{-1}) for total invertebrates and functional feeding group by treatment (TRMT), period (TRYR), and the treatment × period interaction (TRMT × TRYR). Comparisons in **bold** print indicate $P < 0.05$. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Total Invertebrates and Functional Feeding Group	Effect	Num DF	Den DF	F-value	P-value
Total Invertebrates	TRMT	3	2.42	0.59	0.6712
	TRYR	1	105	0.03	0.8678
	TRMT × TRYR	3	105	0.39	0.7605
Chironomidae	TRMT	3	2.63	1.25	0.4423
	TRYR	1	108	12.17	0.0007
	TRMT × TRYR	3	106	1.62	0.1887
Collector-Filterers	TRMT	3	5.01	0.36	0.7831
	TRYR	1	97.6	1.13	0.2912
	TRMT × TRYR	3	97.3	0.42	0.7426
Collector-Gatherers	TRMT	3	15.3	4.23	0.0231
	TRYR	1	120	29.53	<0.0001
	TRMT × TRYR	3	120	3.18	**0.0265**
Omnivores	TRMT	3	3.79	0.55	0.6748
	TRYR	1	117	0.48	0.4901
	TRMT × TRYR	3	117	1.96	0.1239
Parasites	TRMT	3	2.96	0.1	0.9541
	TRYR	1	103	4.92	0.0288
	TRMT × TRYR	3	102	3.4	**0.0206**
Predators	TRMT	3	18.8	1.16	0.3514
	TRYR	1	116	0	0.9946
	TRMT × TRYR	3	116	1.06	0.3691
Scrapers	TRMT	3	14.4	0.34	0.7957
	TRYR	1	118	3.83	0.0526
	TRMT × TRYR	3	118	1.42	0.2408
Shredders	TRMT	3	16.3	0.88	0.4725
	TRYR	1	90.3	0.51	0.4775
	TRMT × TRYR	3	89.5	0	0.9997
Unknown	TRMT	3	4	1.23	0.4079
	TRYR	1	116	2.84	0.0944
	TRMT × TRYR	3	116	2.97	**0.0348**

Table 14-5. Results of hypothesis tests described in section 14-4.5. for macroinvertebrate export in biomass per day (mg day^{-1}) for total invertebrates and functional feeding group. Comparisons a, b, c, and d compare pre- and post-harvest macroinvertebrate export for each treatment, and comparisons e, f, g, h, i, and j compare post-harvest changes in macroinvertebrate export between the treatments. Comparisons in **bold** print indicate P <0.05. P-values were not adjusted for multiple comparisons.

Total Invertebrates and Functional Feeding Group	**Comparison**	**Change (mg day^{-1})**	**P-value**	**95% C.I.**	
				Lower	**Upper**
Total Invertebrates	a. REF-Pre vs. Post	0.1000	0.5911	−0.2679	0.4679
	b. 100%-Pre vs. Post	0.1316	0.4851	−0.2406	0.5037
	c. FP-Pre vs. Post	−0.1212	0.5397	−0.5120	0.2695
	d. 0%-Pre vs. Post	−0.0465	0.8110	−0.4316	0.3385
	e. REF vs. 100%	0.0316	0.9051	−0.4917	0.5548
	f. REF vs. FP	−0.2212	0.4156	−0.7579	0.3154
	g. REF vs. 0%	−0.1465	0.5864	−0.6790	0.3860
	h. 100% vs. FP	−0.2528	0.3551	−0.7924	0.2868
	i. 100% vs. 0%	−0.1781	0.5110	−0.7135	0.3573
	j. FP vs. 0%	0.0747	0.7876	−0.4739	0.6233
Chironomidae	a. REF-Pre vs. Post	0.3644	**0.0015**	0.1434	0.5854
	b. 100%-Pre vs. Post	0.2440	0.0526	−0.0028	0.4908
	c. FP-Pre vs. Post	0.2067	0.0864	−0.0301	0.4436
	d. 0%-Pre vs. Post	0.0104	0.9296	−0.2230	0.2438
	e. REF vs. 100%	−0.1204	0.4722	−0.4513	0.2105
	f. REF vs. FP	−0.1576	0.3368	−0.4815	0.1663
	g. REF vs. 0%	−0.3540	**0.0312**	−0.6754	−0.0325
	h. 100% vs. FP	−0.0373	0.8293	−0.3790	0.3045
	i. 100% vs. 0%	−0.2336	0.1752	−0.5729	0.1057
	j. FP vs. 0%	−0.1963	0.2445	−0.5288	0.1362
Collector-Filterers	a. REF-Pre vs. Post	−0.0310	0.8597	−0.3780	0.3160
	b. 100%-Pre vs. Post	0.0872	0.6230	−0.2632	0.4376
	c. FP-Pre vs. Post	0.0762	0.6810	−0.2907	0.4431
	d. 0%-Pre vs. Post	0.2493	0.1754	−0.1135	0.6120
	e. REF vs. 100%	0.1182	0.6357	−0.3750	0.6113
	f. REF vs. FP	0.1072	0.6744	−0.3978	0.6122
	g. REF vs. 0%	0.2803	0.2704	−0.2216	0.7822
	h. 100% vs. FP	−0.0110	0.9659	−0.5182	0.4963
	i. 100% vs. 0%	0.1621	0.5249	−0.3421	0.6663
	j. FP vs. 0%	0.1731	0.5067	−0.3429	0.6890
Collector-Gatherers	a. REF-Pre vs. Post	0.3159	0.0566	−0.0090	0.6407
	b. 100%-Pre vs. Post	0.2389	0.1496	−0.0873	0.5650
	c. FP-Pre vs. Post	0.9048	**<0.0001**	0.5641	1.2456
	d. 0%-Pre vs. Post	0.3655	**0.0343**	0.0275	0.7036
	e. REF vs. 100%	−0.0770	0.7410	−0.5373	0.3833
	f. REF vs. FP	0.5889	**0.0146**	0.1182	1.0597
	g. REF vs. 0%	0.0497	0.8342	−0.4191	0.5185
	h. 100% vs. FP	0.6659	**0.0060**	0.1943	1.1376
	i. 100% vs. 0%	0.1267	0.5944	−0.3430	0.5964
	j. FP vs. 0%	−0.5393	**0.0280**	−1.0192	−0.0593

Table 14-5. (continued)

Total Invertebrates and Functional Feeding Group	Comparison	Change (mg day^{-1})	P-value	95% C.I.	
				Lower	Upper
Omnivores	a. REF-Pre vs. Post	0.2275	0.4303	−0.3418	0.7967
	b. 100%-Pre vs. Post	−0.1877	0.5139	−0.7553	0.3799
	c. FP-Pre vs. Post	−0.6517	**0.0312**	−1.2434	−0.0600
	d. 0%-Pre vs. Post	0.2073	0.4849	−0.3786	0.7931
	e. REF vs. 100%	−0.4151	0.3085	−1.2190	0.3887
	f. REF vs. FP	−0.8791	**0.0361**	−1.7002	−0.0581
	g. REF vs. 0%	−0.0202	0.9611	−0.8370	0.7966
	h. 100% vs. FP	−0.4640	0.2647	−1.2839	0.3559
	i. 100% vs. 0%	0.3950	0.3396	−0.4208	1.2107
	j. FP vs. 0%	0.8590	**0.0433**	0.0263	1.6916
Parasites	a. REF-Pre vs. Post	0.1489	0.1178	−0.0383	0.3362
	b. 100%-Pre vs. Post	−0.1520	0.1312	−0.3501	0.0461
	c. FP-Pre vs. Post	−0.2513	**0.0132**	−0.4488	−0.0537
	d. 0%-Pre vs. Post	−0.1801	0.0688	−0.3744	0.0141
	e. REF vs. 100%	−0.3009	**0.0308**	−0.5735	−0.0284
	f. REF vs. FP	−0.4002	**0.0044**	−0.6724	−0.1280
	g. REF vs. 0%	−0.3291	**0.0173**	−0.5988	−0.0593
	h. 100% vs. FP	−0.0993	0.4831	−0.3790	0.1804
	i. 100% vs. 0%	−0.0281	0.8410	−0.3055	0.2493
	j. FP vs. 0%	0.0711	0.6115	−0.2059	0.3482
Predators	a. REF-Pre vs. Post	0.1286	0.5450	−0.2909	0.5481
	b. 100%-Pre vs. Post	0.2457	0.2501	−0.1753	0.6666
	c. FP-Pre vs. Post	−0.2167	0.3316	−0.6569	0.2235
	d. 0%-Pre vs. Post	−0.1605	0.4687	−0.5978	0.2768
	e. REF vs. 100%	0.1171	0.6971	−0.4772	0.7114
	f. REF vs. FP	−0.3453	0.2631	−0.9534	0.2628
	g. REF vs. 0%	−0.2891	0.3467	−0.8951	0.3169
	h. 100% vs. FP	−0.4624	0.1354	−1.0715	0.1467
	i. 100% vs. 0%	−0.4062	0.1876	−1.0131	0.2008
	j. FP vs. 0%	0.0562	0.8580	−0.5643	0.6767
Scrapers	a. REF-Pre vs. Post	0.4943	**0.0129**	0.1065	0.8822
	b. 100%-Pre vs. Post	−0.0305	0.8772	−0.4199	0.3590
	c. FP-Pre vs. Post	0.2683	0.1948	−0.1391	0.6756
	d. 0%-Pre vs. Post	0.0536	0.7935	−0.3508	0.4579
	e. REF vs. 100%	−0.5248	0.0611	−1.0744	0.0249
	f. REF vs. FP	−0.2261	0.4277	−0.7885	0.3364
	g. REF vs. 0%	−0.4408	0.1219	−1.0010	0.1195
	h. 100% vs. FP	0.2987	0.2961	−0.2649	0.8623
	i. 100% vs. 0%	0.0840	0.7675	−0.4774	0.6454
	j. FP vs. 0%	−0.2147	0.4603	−0.7887	0.3593
Shredders	a. REF-Pre vs. Post	−0.0525	0.7046	−0.3269	0.2218
	b. 100%-Pre vs. Post	−0.0566	0.6883	−0.3356	0.2223
	c. FP-Pre vs. Post	−0.0371	0.7963	−0.3219	0.2478
	d. 0%-Pre vs. Post	−0.0546	0.6996	−0.3354	0.2262
	e. REF vs. 100%	−0.0041	0.9835	−0.3953	0.3871
	f. REF vs. FP	0.0155	0.9381	−0.3799	0.4109
	g. REF vs. 0%	−0.0020	0.9918	−0.3945	0.3904
	h. 100% vs. FP	0.0196	0.9226	−0.3789	0.4180
	i. 100% vs. 0%	0.0020	0.9919	−0.3935	0.3975
	j. FP vs. 0%	−0.0175	0.9307	−0.4175	0.3824

Table 14-5. (continued)

Total Invertebrates and Functional Feeding Group	Comparison	Change (mg day^{-1})	P-value	95% C.I.	
				Lower	Upper
Unknown	a. REF-Pre vs. Post	0.3286	0.1274	−0.0953	0.7525
	b. 100%-Pre vs. Post	0.6525	**0.0030**	0.2269	1.0781
	c. FP-Pre vs. Post	−0.1125	0.6176	−0.5576	0.3326
	d. 0%-Pre vs. Post	−0.1292	0.5635	−0.5710	0.3126
	e. REF vs. 100%	0.3239	0.2877	−0.2768	0.9246
	f. REF vs. FP	−0.4411	0.1579	−1.0557	0.1735
	g. REF vs. 0%	−0.4578	0.1413	−1.0701	0.1544
	h. 100% vs. FP	−0.7650	**0.0154**	−1.3808	−0.1492
	i. 100% vs. 0%	−0.7817	**0.0130**	−1.3952	−0.1682
	j. FP vs. 0%	−0.0167	0.9580	−0.6438	0.6104

14-5.2. FUNCTIONAL FEEDING GROUPS

Seven functional feeding groups were represented in the drift samples included in the analysis. These included collector-filterers, collector-gatherers, omnivores, parasites, predators, scrapers, and shredders. As previously stated, we assigned individuals of the family Chironomidae their own “feeding group”. Finally, we assigned those taxa whose feeding group was unknown or that were represented by early instars that could not be identified to family or genus to the Unknown feeding group.

14-5.2.1. *Numbers per Day*

The GLMM ANOVA showed evidence of a strong treatment × period interaction for parasite ($P = 0.0083$) and scraper ($P = 0.0054$) export in numbers per day (**Table 14-2**). There was no treatment × period interaction for the Chironomidae, collector-filterer, collector-gatherer, omnivore, predator, shredder, and unknown feeding groups ($P > 0.05$).

Post-hoc comparisons between the PRE and POST treatment periods showed an increase in export in numbers per day of the Chironomidae, collector-gatherer, parasite, scraper, shredder, and unknown feeding groups in the reference, an increase in the collector-gatherer and unknown feeding groups in the 100% and 0% treatments, and an increase in the collector-gatherer and scraper feeding groups in the FP treatment ($P < 0.05$; **Table 14-3**). Parasite export decreased in the 100% and FP treatments ($P < 0.05$).

Post-hoc comparisons between the reference and the treatments showed a decrease in export in numbers per day of parasites, predators, and scrapers in the 100% treatment relative to the reference, a decrease in parasites in the FP treatment relative to the reference, and a decrease in Chironomidae, parasites, and scrapers in the 0% treatment relative to the reference ($P < 0.05$; **Table 14-3**). Omnivore export increased in the 0% treatment relative to the FP treatment ($P < 0.05$). Descriptive statistics for export by functional feeding group in numbers per day from each study site by treatment year are provided in **Appendix Table 14-B-1**.

Collector-gatherers comprised a large proportion of individuals exported per day in both the PRE and POST periods from the reference and from the treatments, and increased in proportion from the PRE to POST period in the reference and in the treatments (**Figure 14-6**). The dominant collector-gatherers were *Baetis* and early instar baetid mayflies (Ephemeroptera: Baetidae), and to some extent amphipods (Crustacea), springtails (Collembola), and *Dixa* (Diptera: Dixidae). Chironomidae, which could be either collector-gatherers or predators, were also abundant during the PRE and POST periods in the reference and in the treatments. While Chironomidae increased in proportion from the PRE to POST period in the reference and in the 100% treatment, their proportion decreased in the FP and 0% treatments post-harvest. Parasites, which consisted entirely of aquatic and terrestrial mites (Acarina), were comparable in proportion to Chironomidae during the PRE period, but decreased in proportion in the POST period. Collector-filterers, omnivores, predators, scrapers, shredders, and the unknown feeding group were present in smaller proportions and their response was more variable across treatments and periods.

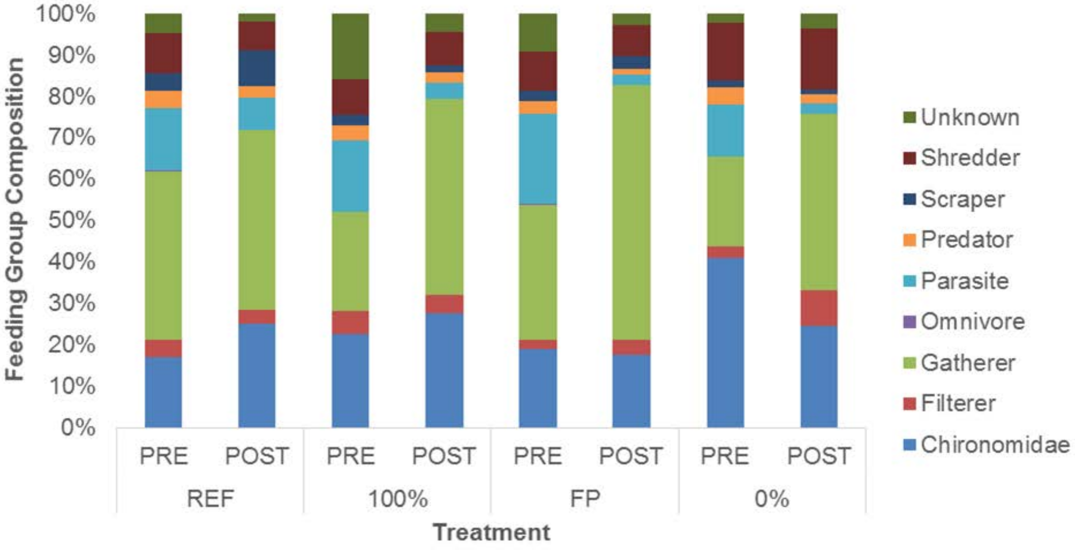

Figure 14-6. Composition of macroinvertebrate export in numbers per day (# day^{-1}) from the Type N Study treatments by functional feeding group and treatment period. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period.

14-5.2.2. *Biomass per day*

The GLMM ANOVA showed evidence of a strong treatment × period interaction for collector-gatherer (P = 0.0265), parasite (P = 0.0206), and unknown (P = 0.0348) export in mg day^{-1} (**Table 14-4**). There was no treatment × period interaction for the Chironomidae, collector-filterer, omnivore, predator, scraper, and shredder feeding groups (P >0.05).

Post-hoc comparisons between the PRE and POST treatment periods showed an increase in export in mg day^{-1} of the Chironomidae and scraper feeding groups in the reference, an increase in the unknown feeding group in the 100% treatment, and an increase in the collector-gatherer feeding group in the FP and 0% treatments (P <0.05; **Table 14-5**). Omnivore and parasite export decreased in the FP treatment (P <0.05).

Post-hoc comparisons between the reference and the buffer treatments showed a decrease in export in mg day^{-1} of parasites in the 100% treatment relative to the reference, a decrease in omnivores and parasites in the FP treatment relative to the reference, and a decrease in Chironomidae and parasites in the 0% treatment relative to the reference ($P < 0.05$; **Table 14-5**). In addition, there was a decrease in the unknown feeding group in the FP and 0% treatments relative to the 100% treatment, and a decrease in collector-gatherers in the 0% treatment relative to the FP treatment ($P < 0.05$). Collector-gatherer export increased in the FP treatment relative to the reference and in the FP treatment relative to the 100% treatment, while omnivore export increased in the 0% treatment relative to the FP treatment ($P < 0.05$). Descriptive statistics for export by functional feeding group in mg day^{-1} from each study site by treatment year are provided in **Appendix Table 14-B-2**.

Collector-gatherers comprised much of the biomass exported per day during the POST period in the reference and in the FP treatment, and were comparable in proportion to predator biomass in the 0% treatment (**Figure 14-7**). Amphipods and *Baetis* were numerous and made up most of the collector-gatherer biomass. Predator biomass was considerable during the PRE period in the reference and in the 0% treatment, and during the POST period in the 100% treatment and 0% treatment. Most of the predator biomass was made up of a few, but large, individuals, such as dragonflies (adult Odonata), *Doroneuria* (Plecoptera: Perlidae), and *Calineuria californica* (Plecoptera: Perlidae). The proportion of omnivore biomass overwhelmingly dominated export during the PRE period in the 100% and FP treatments and consisted mostly of crayfish (Crustacea) and *Pteronarcys* (Plecoptera: Pteronarcyidae). Changes in proportions of Chironomidae, collector-filterers, parasites, scrapers, shredders, and the unknown feeding group were more variable across treatments and periods.

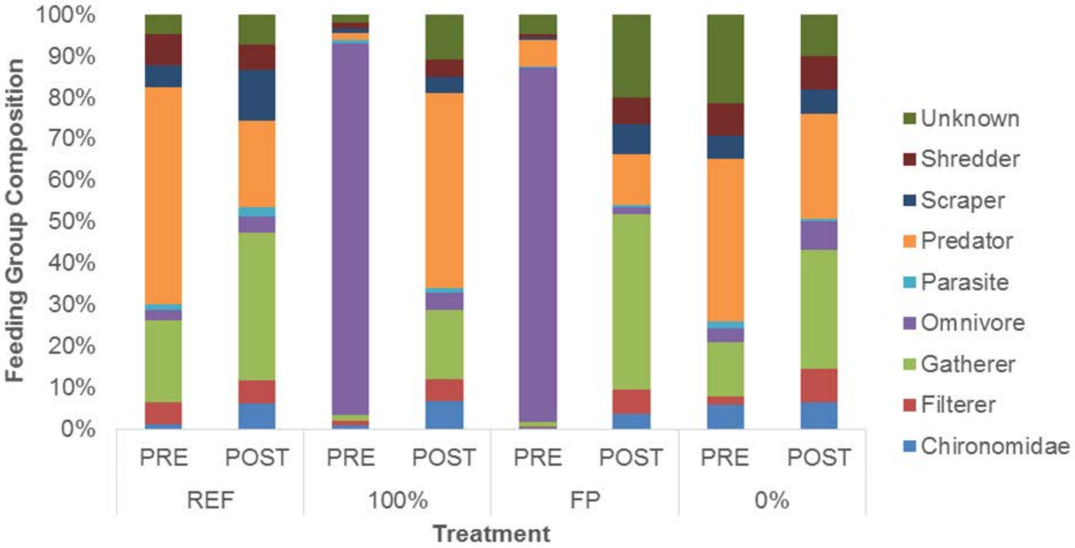

Figure 14-7. Composition of macroinvertebrate export in biomass per day (mg day^{-1}) from the Type N Study treatments by functional feeding group and treatment period. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period.

14-5.3. TAXONOMIC COMPOSITION

About 132 macroinvertebrate taxa were represented in the drift samples included in the analysis. These included members of the insect orders Collembola (springtails), Coleoptera (beetles), Diptera (true flies), Ephemeroptera (mayflies), Hemiptera (true bugs), Hymenoptera (ants, bees, wasps), Lepidoptera (butterflies, moths), Megaloptera (alderflies, dobsonflies), Odonata (dragonflies), Orthoptera (crickets, grasshoppers), Plecoptera (stoneflies), and Trichoptera (caddisflies), as well as arachnids (mites, spiders, opiliones, pseudoscorpions), crustaceans (amphipods, isopods, crayfish), molluscs (clams, snails), myriapods (centipedes, millipedes), and segmented annelids (worms, leeches). The GLMM ANOVA was run on the major aquatic insect orders — Coleoptera, Diptera, Ephemeroptera, Plecoptera, and Trichoptera — and on a select number of dominant macroinvertebrate families and other taxonomic groups. Percent composition of each macroinvertebrate taxon in the drift is provided in **Appendix Tables 14-C-1** and **14-C-2**.

14-5.3.1. *Numbers per Day*

The GLMM ANOVA showed evidence of a strong treatment × period interaction for Dixidae (Diptera; $P = 0.0352$) export in numbers per day (**Table 14-6**). There was no treatment × period interaction for Collembola, Coleoptera, Diptera, Ephemeroptera, Plecoptera, Trichoptera, Simuliidae (Diptera), Baetidae (Ephemeroptera), and Nemouridae (Plecoptera) ($P > 0.05$).

Table 14-6. Results of the GLMM ANOVA for macroinvertebrate export in numbers per day (# day^{-1}) for dominant invertebrate order, family, or other group by treatment (TRMT), period (TRYR), and the treatment × period interaction (TRMT × TRYR). Comparisons in **bold** print indicate $P < 0.05$. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Dominant Invertebrate Order, Family, or Other Group	Effect	Num DF	Den DF	F-value	P-value
Collembola	TRMT	3	1.83	0.27	0.8452
	TRYR	1	110	0	0.9953
	TRMT × TRYR	3	110	0.62	0.6017
Coleoptera	TRMT	3	2.9	0.51	0.7060
	TRYR	1	115	3.85	0.0521
	TRMT × TRYR	3	115	0.88	0.4516
Diptera	TRMT	3	1	0.18	0.9011
	TRYR	1	87.3	9.86	0.0023
	TRMT × TRYR	3	86.9	1.99	0.1221
Diptera: Dixidae	TRMT	3	19.4	0.47	0.7089
	TRYR	1	119	9.55	0.0025
	TRMT × TRYR	3	119	2.96	**0.0352**
Diptera: Simuliidae	TRMT	3	18.8	0.43	0.7363
	TRYR	1	119	3.26	0.0734
	TRMT × TRYR	3	119	0.12	0.9456
Ephemeroptera	TRMT	3	4.97	0.01	0.9973
	TRYR	1	115	61.99	<0.0001
	TRMT × TRYR	3	115	1	0.3936

Table 14-6. (continued)

Dominant Invertebrate Order, Family, or Other Group	Effect	Num DF	Den DF	F-value	P-value
Ephemeroptera: Baetidae	TRMT	3	4.02	0.12	0.9453
	TRYR	1	115	68.84	<0.0001
	TRMT × TRYR	3	115	0.65	0.5858
Plecoptera	TRMT	3	4.14	1.52	0.3355
	TRYR	1	115	9.44	0.0026
	TRMT × TRYR	3	115	0.13	0.9394
Plecoptera: Nemouridae	TRMT	3	16.1	0.79	0.5153
	TRYR	1	116	17.77	<0.0001
	TRMT × TRYR	3	116	0.32	0.8143
Trichoptera	TRMT	3	4.01	0.52	0.6897
	TRYR	1	116	22.13	<0.0001
	TRMT × TRYR	3	116	0.35	0.7900

Post-hoc comparisons between the PRE and POST treatment periods showed an increase in export in numbers per day of Diptera in the reference, Dixidae in the reference and in the FP treatment, and Nemouridae and Trichoptera in the reference and in the FP and 0% treatments ($P < 0.05$; **Table 14-7**). In addition, Ephemeroptera and Baetidae export increased in the reference and in the 100%, FP, and 0% treatments ($P < 0.05$). Post-hoc comparisons between the reference and the buffer treatments showed a decrease in export in numbers per day of Diptera and Dixidae in the 100% treatment relative to the reference and in the 0% treatment relative to the reference ($P < 0.05$; **Table 14-7**). Descriptive statistics for export by order in numbers per day from each study site by treatment year are provided in **Appendix Table 14-B-3**.

Table 14-7. Results of hypothesis tests described in section 14-4.5. for macroinvertebrate export in numbers per day (# day^{-1}) for dominant invertebrate order, family, or other group. Comparisons a, b, c, and d compare pre- and post-harvest macroinvertebrate export for each treatment, and comparisons e, f, g, h, i, and j compare post-harvest changes in macroinvertebrate export between the treatments. Comparisons in **bold** print indicate $P < 0.05$. P-values were not adjusted for multiple comparisons.

Dominant Invertebrate Order, Family, or Other Group	Comparison	Change (# day^{-1})	P-value	95% C.I.	
				Lower	Upper
Collembola	a. REF-Pre vs. Post	0.2568	0.2567	−0.1896	0.7031
	b. 100%-Pre vs. Post	−0.0611	0.7902	−0.5147	0.3926
	c. FP-Pre vs. Post	−0.0292	0.9029	−0.5016	0.4433
	d. 0%-Pre vs. Post	−0.1638	0.4879	−0.6301	0.3026
	e. REF vs. 100%	−0.3178	0.3245	−0.9542	0.3186
	f. REF vs. FP	−0.2859	0.3852	−0.9359	0.3640
	g. REF vs. 0%	−0.4205	0.1994	−1.0661	0.2250
	h. 100% vs. FP	0.0319	0.9233	−0.6231	0.6869
	i. 100% vs. 0%	−0.1027	0.7550	−0.7533	0.5479
	j. FP vs. 0%	−0.1346	0.6885	−0.7985	0.5292

Table 14-7. (continued)

Dominant Invertebrate Order, Family, or Other Group	Comparison	Change (# day^{-1})	P-value	95% C.I.	
				Lower	Upper
Coleoptera	a. REF-Pre vs. Post	0.2373	0.0705	−0.0202	0.4948
	b. 100%-Pre vs. Post	0.0084	0.9498	−0.2564	0.2733
	c. FP-Pre vs. Post	0.0379	0.7835	−0.2344	0.3101
	d. 0%-Pre vs. Post	0.2435	0.0757	−0.0256	0.5125
	e. REF vs. 100%	−0.2289	0.2222	−0.5982	0.1405
	f. REF vs. FP	−0.1994	0.2940	−0.5742	0.1753
	g. REF vs. 0%	0.0062	0.9739	−0.3662	0.3785
	h. 100% vs. FP	0.0294	0.8783	−0.3504	0.4092
	i. 100% vs. 0%	0.2350	0.2199	−0.1424	0.6125
	j. FP vs. 0%	0.2056	0.2895	−0.1772	0.5884
Diptera	a. REF-Pre vs. Post	0.5896	**0.0004**	0.2699	0.9093
	b. 100%-Pre vs. Post	0.1024	0.5508	−0.2368	0.4416
	c. FP-Pre vs. Post	0.2560	0.1350	−0.0814	0.5934
	d. 0%-Pre vs. Post	0.1007	0.5472	−0.2308	0.4321
	e. REF vs. 100%	−0.4872	**0.0406**	−0.9532	−0.0212
	f. REF vs. FP	−0.3336	0.1571	−0.7984	0.1312
	g. REF vs. 0%	−0.4889	**0.0377**	−0.9494	−0.0285
	h. 100% vs. FP	0.1536	0.5252	−0.3247	0.6319
	i. 100% vs. 0%	−0.0017	0.9942	−0.4758	0.4724
	j. FP vs. 0%	−0.1553	0.5152	−0.6283	0.3176
Diptera: Dixidae	a. REF-Pre vs. Post	0.6476	**0.0005**	0.2903	1.0049
	b. 100%-Pre vs. Post	−0.0068	0.9702	−0.3654	0.3518
	c. FP-Pre vs. Post	0.4468	**0.0199**	0.0720	0.8216
	d. 0%-Pre vs. Post	0.0543	0.7731	−0.3178	0.4264
	e. REF vs. 100%	−0.6544	**0.0117**	−1.1607	−0.1482
	f. REF vs. FP	−0.2008	0.4440	−0.7187	0.3170
	g. REF vs. 0%	−0.5933	**0.0245**	−1.1092	-0.0775
	h. 100% vs. FP	0.4536	0.0860	−0.0652	0.9723
	i. 100% vs. 0%	0.0611	0.8154	−0.4557	0.5779
	j. FP vs. 0%	−0.3925	0.1438	−0.9207	0.1357
Diptera: Simuliidae	a. REF-Pre vs. Post	0.1420	0.5155	−0.2890	0.5729
	b. 100%-Pre vs. Post	0.1343	0.5398	−0.2982	0.5669
	c. FP-Pre vs. Post	0.3013	0.1894	−0.1507	0.7534
	d. 0%-Pre vs. Post	0.2274	0.3178	−0.2214	0.6762
	e. REF vs. 100%	−0.0076	0.9803	−0.6182	0.6029
	f. REF vs. FP	0.1594	0.6143	−0.4652	0.7840
	g. REF vs. 0%	0.0854	0.7862	−0.5368	0.7076
	h. 100% vs. FP	0.1670	0.5981	−0.4587	0.7927
	i. 100% vs. 0%	0.0931	0.7680	−0.5303	0.7164
	j. FP vs. 0%	−0.0740	0.8185	−0.7110	0.5631
Ephemeroptera	a. REF-Pre vs. Post	1.0775	**<0.0001**	0.5898	1.5651
	b. 100%-Pre vs. Post	0.7198	**0.0046**	0.2270	1.2127
	c. FP-Pre vs. Post	1.3094	**<0.0001**	0.7954	1.8233
	d. 0%-Pre vs. Post	0.8769	**0.0009**	0.3675	1.3863
	e. REF vs. 100%	−0.3576	0.3091	−1.0509	0.3357
	f. REF vs. FP	0.2319	0.5180	−0.4766	0.9404
	g. REF vs. 0%	−0.2006	0.5743	−0.9058	0.5046
	h. 100% vs. FP	0.5895	0.1038	−0.1225	1.3016
	i. 100% vs. 0%	0.1571	0.6615	−0.5517	0.8658
	j. FP vs. 0%	−0.4325	0.2390	−1.1561	0.2912

Table 14-7. (continued)

Dominant Invertebrate Order, Family, or Other Group	**Comparison**	**Change (# day^{-1})**	**P-value**	**95% C.I.**	
				Lower	**Upper**
Ephemeroptera: Baetidae	a. REF-Pre vs. Post	1.0830	**<0.0001**	0.5861	1.5799
	b. 100%-Pre vs. Post	0.8430	**0.0011**	0.3443	1.3417
	c. FP-Pre vs. Post	1.3404	**<0.0001**	0.8188	1.8620
	d. 0%-Pre vs. Post	0.9969	**0.0002**	0.4788	1.5151
	e. REF vs. 100%	−0.2400	0.5007	−0.9440	0.4639
	f. REF vs. FP	0.2574	0.4805	−0.4630	0.9778
	g. REF vs. 0%	−0.0861	0.8127	−0.8039	0.6318
	h. 100% vs. FP	0.4975	0.1748	−0.2242	1.2191
	i. 100% vs. 0%	0.1540	0.6722	−0.5651	0.8731
	j. FP vs. 0%	−0.3435	0.3567	−1.0787	0.3917
Plecoptera	a. REF-Pre vs. Post	0.2855	0.0672	−0.0205	0.5916
	b. 100%-Pre vs. Post	0.1611	0.3017	−0.1465	0.4686
	c. FP-Pre vs. Post	0.2484	0.1290	−0.0733	0.5700
	d. 0%-Pre vs. Post	0.2781	0.0869	−0.0409	0.5971
	e. REF vs. 100%	−0.1245	0.5710	−0.5583	0.3094
	f. REF vs. FP	−0.0372	0.8686	−0.4812	0.4068
	g. REF vs. 0%	−0.0074	0.9736	−0.4495	0.4347
	h. 100% vs. FP	0.0873	0.6984	−0.3578	0.5323
	i. 100% vs. 0%	0.1170	0.6018	−0.3261	0.5602
	j. FP vs. 0%	0.0298	0.8967	−0.4233	0.4828
Plecoptera: Nemouridae	a. REF-Pre vs. Post	0.4429	**0.0140**	0.0912	0.7947
	b. 100%-Pre vs. Post	0.2477	0.1673	−0.1053	0.6007
	c. FP-Pre vs. Post	0.4758	**0.0120**	0.1066	0.8451
	d. 0%-Pre vs. Post	0.3670	**0.0498**	0.0003	0.7336
	e. REF vs. 100%	−0.1952	0.4393	−0.6936	0.3031
	f. REF vs. FP	0.0329	0.8985	−0.4770	0.5428
	g. REF vs. 0%	−0.0760	0.7677	−0.5841	0.4321
	h. 100% vs. FP	0.2282	0.3782	−0.2827	0.7390
	i. 100% vs. 0%	0.1193	0.6434	−0.3897	0.6283
	j. FP vs. 0%	−0.1089	0.6794	−0.6292	0.4115
Trichoptera	a. REF-Pre vs. Post	0.4810	**0.0040**	0.1564	0.8055
	b. 100%-Pre vs. Post	0.2547	0.1218	−0.0689	0.5783
	c. FP-Pre vs. Post	0.4067	**0.0186**	0.0693	0.7441
	d. 0%-Pre vs. Post	0.4250	**0.0131**	0.0910	0.7591
	e. REF vs. 100%	−0.2263	0.3302	−0.6846	0.2320
	f. REF vs. FP	−0.0742	0.7541	−0.5424	0.3939
	g. REF vs. 0%	−0.0559	0.8124	−0.5216	0.4098
	h. 100% vs. FP	0.1520	0.5208	−0.3155	0.6196
	i. 100% vs. 0%	0.1703	0.4697	−0.2947	0.6354
	j. FP vs. 0%	0.0183	0.9393	−0.4565	0.4931

Diptera and Ephemeroptera comprised most of the individuals exported per day in both the PRE and POST periods from the reference and from the buffer treatments (**Figure 14-8**). The dominant Diptera were Chironomidae, while the dominant Ephemeroptera were *Baetis* and early instar baetid mayflies.

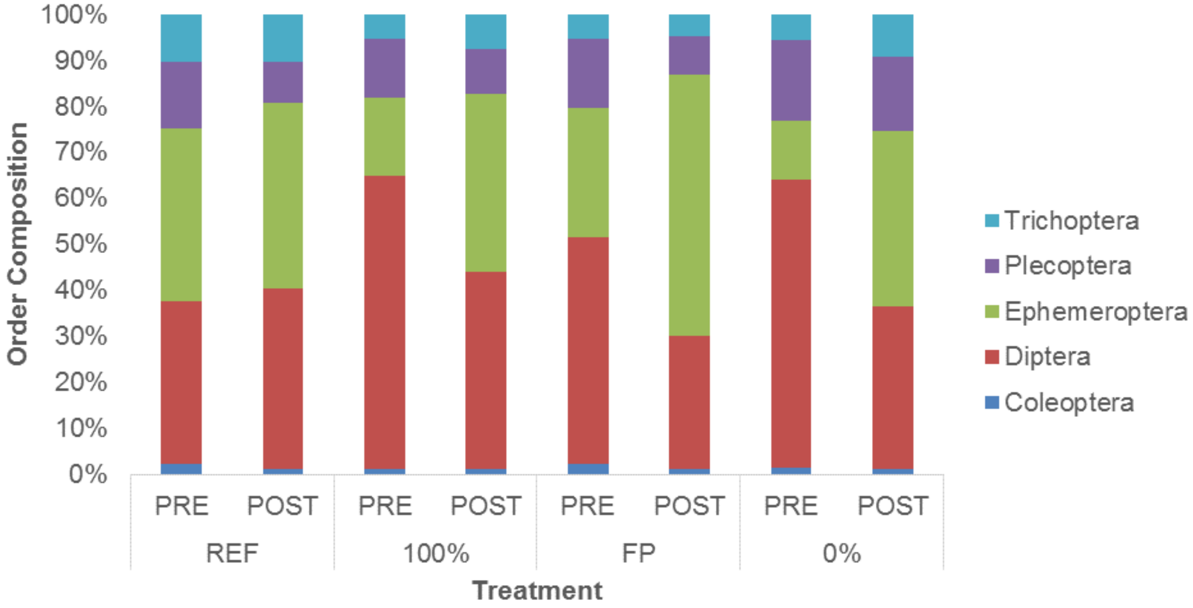

Figure 14-8. Composition of macroinvertebrate export in numbers per day (# day^{-1}) from the Type N Study treatments by order and treatment period. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period.

14-5.3.2. *Biomass per Day*

The GLMM ANOVA showed evidence of a moderate treatment × period interaction for Dixidae ($P = 0.0584$) export in mg day^{-1} (**Table 14-8**). There was no treatment × period interaction for Crustacea, Coleoptera, Diptera, Ephemeroptera, Plecoptera, Trichoptera, Baetidae, Heptageniidae (Ephemeroptera), Perlidae (Plecoptera), Perlodidae (Plecoptera), Hydropsychidae (Trichoptera), and Rhyacophilidae (Trichoptera) ($P > 0.05$).

Table 14-8. Results of the GLMM ANOVA for macroinvertebrate export in biomass per day (mg day^{-1}) for dominant invertebrate order, family, or other group by treatment (TRMT), period (TRYR), and the treatment × period interaction (TRMT × TRYR). Comparisons in **bold** print indicate P <0.05. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Dominant Invertebrate Order, Family, or Other Group	Effect	Num DF	Den DF	F-value	P-value
Crustacea	TRMT	3	4	0.86	0.5316
	TRYR	1	111	0.06	0.8149
	TRMT × TRYR	3	111	0.22	0.8826
Coleoptera	TRMT	3	3.85	0.56	0.6699
	TRYR	1	113	0.13	0.7178
	TRMT × TRYR	3	113	0.18	0.9129
Diptera	TRMT	3	2.68	0.71	0.6135
	TRYR	1	107	3.67	0.0580
	TRMT × TRYR	3	106	1.64	0.1844
Diptera: Dixidae	TRMT	3	3.84	1.47	0.3531
	TRYR	1	115	6.97	0.0095
	TRMT × TRYR	3	115	2.56	0.0584
Ephemeroptera	TRMT	3	14.2	0.36	0.7822
	TRYR	1	117	38.36	<0.0001
	TRMT × TRYR	3	117	1.84	0.1433
Ephemeroptera: Baetidae	TRMT	3	16.2	0.42	0.7406
	TRYR	1	118	55.11	<0.0001
	TRMT × TRYR	3	118	1.02	0.3861
Ephemeroptera: Heptageniidae	TRMT	3	12.9	0.58	0.6382
	TRYR	1	119	5.08	0.0260
	TRMT × TRYR	3	119	1.8	0.1516
Plecoptera	TRMT	3	2.86	0.12	0.9395
	TRYR	1	107	0.48	0.4909
	TRMT × TRYR	3	107	1.16	0.3295
Plecoptera: Perlidae	TRMT	3	2.7	0.03	0.9923
	TRYR	1	108	0.04	0.8474
	TRMT × TRYR	3	108	0.46	0.7077
Plecoptera: Perlodidae	TRMT	3	3.91	2.87	0.1698
	TRYR	1	116	0	0.9459
	TRMT × TRYR	3	116	0.39	0.7586
Trichoptera	TRMT	3	3.95	1.38	0.3715
	TRYR	1	105	0.44	0.5103
	TRMT × TRYR	3	105	0.09	0.9677
Trichoptera: Hydropsychidae	TRMT	3	5.19	0.19	0.9006
	TRYR	1	115	1.21	0.2746
	TRMT × TRYR	3	115	0.32	0.8145
Trichoptera: Rhyacophilidae	TRMT	3	5.41	0.64	0.6180
	TRYR	1	110	0.05	0.8181
	TRMT × TRYR	3	110	0.63	0.5963

Post-hoc comparisons between the PRE and POST treatment periods showed an increase in export in mg day^{-1} of Diptera in the reference, Dixidae and Heptageniidae in the reference and in the FP treatment, and Ephemeroptera in the reference and in the FP and 0% treatments ($P < 0.05$) (**Table 14-9**). In addition, Baetidae export increased in the reference and in the 100%, FP, and 0% treatments ($P < 0.05$).

Post-hoc comparisons between the reference and the treatments showed a decrease in export in mg day^{-1} of Dixidae and Heptageniidae in the 100% treatment relative to the reference, and an increase in Dixidae and Ephemeroptera in the FP treatment relative to the 100% treatment ($P < 0.05$; **Table 14-7**). Descriptive statistics for export by order in mg day^{-1} from each study site by treatment year are provided in **Appendix Table 14-B-4**.

Table 14-9. Results of hypothesis tests described in section 14-4.5. for macroinvertebrate export in biomass per day (mg day^{-1}) for dominant invertebrate order, family, or other group. Comparisons a, b, c, and d compare pre- and post-harvest macroinvertebrate export for each treatment, and comparisons e, f, g, h, i, and j compare post-harvest changes in macroinvertebrate export between the treatments. Comparisons in **bold** print indicate $P < 0.05$. P-values were not adjusted for multiple comparisons.

Dominant Invertebrate Order, Family, or Other Group	Comparison	Change (mg day^{-1})	P-value	95% C.I.	
				Lower	Upper
Crustacea	a. REF-Pre vs. Post	0.1490	0.5857	-0.3910	0.6889
	b. 100%-Pre vs. Post	−0.1468	0.5924	-0.6888	0.3951
	c. FP-Pre vs. Post	−0.0891	0.7559	-0.6553	0.4772
	d. 0%-Pre vs. Post	−0.0439	0.8771	-0.6053	0.5175
	e. REF vs. 100%	−0.2958	0.4452	-1.0609	0.4692
	f. REF vs. FP	−0.2380	0.5478	-1.0205	0.5444
	g. REF vs. 0%	−0.1929	0.6246	-0.9718	0.5860
	h. 100% vs. FP	0.0578	0.8841	-0.7260	0.8416
	i. 100% vs. 0%	0.1029	0.7943	-0.6774	0.8832
	j. FP vs. 0%	0.0451	0.9109	-0.7523	0.8425
Coleoptera	a. REF-Pre vs. Post	−0.0250	0.8972	-0.4069	0.3570
	b. 100%-Pre vs. Post	0.0501	0.7982	-0.3373	0.4375
	c. FP-Pre vs. Post	−0.0186	0.9274	-0.4215	0.3843
	d. 0%-Pre vs. Post	−0.1503	0.4572	-0.5495	0.2489
	e. REF vs. 100%	0.0751	0.7850	-0.4689	0.6191
	f. REF vs. FP	0.0064	0.9819	-0.5488	0.5615
	g. REF vs. 0%	−0.1253	0.6539	-0.6778	0.4271
	h. 100% vs. FP	−0.0687	0.8081	-0.6276	0.4902
	i. 100% vs. 0%	−0.2004	0.4768	-0.7567	0.3558
	j. FP vs. 0%	−0.1317	0.6463	-0.6989	0.4354
Diptera	a. REF-Pre vs. Post	0.4439	**0.0047**	0.1390	0.7487
	b. 100%-Pre vs. Post	0.1033	0.5109	−0.2070	0.4136
	c. FP-Pre vs. Post	0.0497	0.7590	−0.2705	0.3697
	d. 0%-Pre vs. Post	0.0076	0.9620	−0.3079	0.3231
	e. REF vs. 100%	−0.3406	0.1236	−0.7755	0.0944
	f. REF vs. FP	−0.3942	0.0799	−0.8362	0.0478
	g. REF vs. 0%	−0.4363	0.0512	−0.8750	0.0024
	h. 100% vs. FP	−0.0537	0.8119	−0.4994	0.3921
	i. 100% vs. 0%	−0.0957	0.6689	−0.5381	0.3467
	j. FP vs. 0%	−0.0421	0.8531	−0.4915	0.4074

Table 14-9. (continued)

Dominant Invertebrate Order, Family, or Other Group	Comparison	Change (mg day^{-1})	P-value	95% C.I.	
				Lower	Upper
Diptera: Dixidae	a. REF-Pre vs. Post	0.3071	**0.0048**	0.0954	0.5188
	b. 100%-Pre vs. Post	−0.0720	0.5039	−0.2846	0.1407
	c. FP-Pre vs. Post	0.2576	**0.0235**	0.0353	0.4800
	d. 0%-Pre vs. Post	0.0851	0.4463	−0.1355	0.3056
	e. REF vs. 100%	−0.3791	**0.0137**	−0.6791	-0.0790
	f. REF vs. FP	−0.0495	0.7503	−0.3564	0.2575
	g. REF vs. 0%	−0.2220	0.1530	−0.5277	0.0837
	h. 100% vs. FP	0.3296	**0.0360**	0.0220	0.6372
	i. 100% vs. 0%	0.1571	0.3120	−0.1493	0.4634
	j. FP vs. 0%	−0.1726	0.2773	−0.4857	0.1406
Ephemeroptera	a. REF-Pre vs. Post	0.7347	**0.0006**	0.3233	1.1462
	b. 100%-Pre vs. Post	0.3085	0.1418	−0.1046	0.7216
	c. FP-Pre vs. Post	1.0023	**<0.0001**	0.5702	1.4345
	d. 0%-Pre vs. Post	0.5909	**0.0074**	0.1619	1.0199
	e. REF vs. 100%	−0.4262	0.1504	−1.0092	0.1568
	f. REF vs. FP	0.2676	0.3763	−0.3291	0.8643
	g. REF vs. 0%	−0.1438	0.6327	−0.7382	0.4506
	h. 100% vs. FP	0.6938	**0.0233**	0.0960	1.2916
	i. 100% vs. 0%	0.2824	0.3497	−0.3132	0.8779
	j. FP vs. 0%	−0.4114	0.1835	−1.0204	0.1975
Ephemeroptera: Baetidae	a. REF-Pre vs. Post	0.6546	**0.0006**	0.2871	1.0221
	b. 100%-Pre vs. Post	0.4892	**0.0098**	0.1203	0.8582
	c. FP-Pre vs. Post	0.9546	**<0.0001**	0.5688	1.3405
	d. 0%-Pre vs. Post	0.7237	**0.0003**	0.3407	1.1068
	e. REF vs. 100%	−0.1654	0.5306	−0.6861	0.3554
	f. REF vs. FP	0.3000	0.2671	−0.2329	0.8329
	g. REF vs. 0%	0.0691	0.7970	−0.4617	0.6000
	h. 100% vs. FP	0.4654	0.0869	−0.0685	0.9992
	i. 100% vs. 0%	0.2345	0.3844	−0.2974	0.7663
	j. FP vs. 0%	−0.2309	0.4021	−0.7746	0.3128
Ephemeroptera: Heptageniidae	a. REF-Pre vs. Post	0.4856	**0.0174**	0.0870	0.8842
	b. 100%-Pre vs. Post	−0.0912	0.6529	−0.4916	0.3092
	c. FP-Pre vs. Post	0.4307	**0.0440**	0.0117	0.8496
	d. 0%-Pre vs. Post	0.1050	0.6180	−0.3106	0.5206
	e. REF vs. 100%	−0.5767	**0.0455**	−1.1417	-0.0117
	f. REF vs. FP	−0.0549	0.8512	−0.6331	0.5233
	g. REF vs. 0%	−0.3806	0.1932	−0.9564	0.1953
	h. 100% vs. FP	0.5218	0.0771	−0.0577	1.1013
	i. 100% vs. 0%	0.1961	0.5023	−0.3810	0.7732
	j. FP vs. 0%	−0.3257	0.2767	−0.9158	0.2644
Plecoptera	a. REF-Pre vs. Post	0.0179	0.9161	−0.3181	0.3539
	b. 100%-Pre vs. Post	0.1773	0.3387	−0.1883	0.5428
	c. FP-Pre vs. Post	−0.2014	0.2770	−0.5667	0.1640
	d. 0%-Pre vs. Post	−0.2424	0.1826	−0.6006	0.1158
	e. REF vs. 100%	0.1594	0.5259	−0.3371	0.6558
	f. REF vs. FP	−0.2193	0.3830	−0.7156	0.2771
	g. REF vs. 0%	−0.2603	0.2956	−0.7514	0.2308
	h. 100% vs. FP	−0.3786	0.1494	−0.8954	0.1382
	i. 100% vs. 0%	−0.4196	0.1070	−0.9314	0.0921
	j. FP vs. 0%	−0.0410	0.8740	−0.5526	0.4706

Table 14-9. (continued)

Dominant Invertebrate Order, Family, or Other Group	Comparison	Change (mg day^{-1})	P-value	95% C.I.	
				Lower	Upper
Plecoptera: Perlidae	a. REF-Pre vs. Post	−0.0795	0.7439	−0.5605	0.4015
	b. 100%-Pre vs. Post	0.2168	0.3810	−0.2716	0.7052
	c. FP-Pre vs. Post	−0.0549	0.8307	−0.5623	0.4526
	d. 0%-Pre vs. Post	−0.1788	0.4824	−0.6814	0.3239
	e. REF vs. 100%	0.2962	0.3936	−0.3893	0.9817
	f. REF vs. FP	0.0246	0.9445	−0.6746	0.7238
	g. REF vs. 0%	−0.0993	0.7778	−0.7950	0.5965
	h. 100% vs. FP	−0.2716	0.4463	−0.9759	0.4327
	i. 100% vs. 0%	−0.3955	0.2658	−1.0964	0.3053
	j. FP vs. 0%	−0.1239	0.7316	−0.8382	0.5904
Plecoptera: Perlodidae	a. REF-Pre vs. Post	0.1032	0.4931	−0.1941	0.4005
	b. 100%-Pre vs. Post	−0.0076	0.9602	−0.3079	0.2927
	c. FP-Pre vs. Post	0.0486	0.7598	−0.2652	0.3623
	d. 0%-Pre vs. Post	−0.1232	0.4339	−0.4339	0.1875
	e. REF vs. 100%	−0.1108	0.6046	−0.5333	0.3118
	f. REF vs. FP	−0.0546	0.8028	−0.4869	0.3776
	g. REF vs. 0%	−0.2264	0.2992	−0.6564	0.2036
	h. 100% vs. FP	0.0561	0.7984	−0.3782	0.4905
	i. 100% vs. 0%	−0.1156	0.5972	−0.5477	0.3165
	j. FP vs. 0%	−0.1717	0.4427	−0.6133	0.2698
Trichoptera	a. REF-Pre vs. Post	0.1202	0.4872	−0.2216	0.4620
	b. 100%-Pre vs. Post	−0.0027	0.9878	−0.3491	0.3438
	c. FP-Pre vs. Post	0.0475	0.7938	−0.3120	0.4070
	d. 0%-Pre vs. Post	0.0687	0.7018	−0.2863	0.4236
	e. REF vs. 100%	−0.1229	0.6178	−0.6096	0.3637
	f. REF vs. FP	−0.0727	0.7718	−0.5688	0.4233
	g. REF vs. 0%	−0.0516	0.8360	−0.5443	0.4412
	h. 100% vs. FP	0.0502	0.8424	−0.4490	0.5494
	i. 100% vs. 0%	0.0714	0.7760	−0.4245	0.5672
	j. FP vs. 0%	0.0212	0.9339	−0.4841	0.5264
Trichoptera: Hydropsychidae	a. REF-Pre vs. Post	−0.2071	0.1986	−0.5245	0.1102
	b. 100%-Pre vs. Post	0.0111	0.9456	−0.3096	0.3318
	c. FP-Pre vs. Post	−0.0982	0.5619	−0.4325	0.2362
	d. 0%-Pre vs. Post	−0.0671	0.6890	−0.3986	0.2643
	e. REF vs. 100%	0.2182	0.3401	−0.2330	0.6694
	f. REF vs. FP	0.1090	0.6405	−0.3520	0.5700
	g. REF vs. 0%	0.1400	0.5467	−0.3189	0.5989
	h. 100% vs. FP	−0.1092	0.6413	−0.5725	0.3540
	i. 100% vs. 0%	−0.0782	0.7376	−0.5394	0.3830
	j. FP vs. 0%	0.0311	0.8963	−0.4397	0.5018
Trichoptera: Rhyacophilidae	a. REF-Pre vs. Post	0.1477	0.2838	−0.1240	0.4194
	b. 100%-Pre vs. Post	−0.0649	0.6421	−0.3408	0.2110
	c. FP-Pre vs. Post	−0.0900	0.5348	−0.3765	0.1965
	d. 0%-Pre vs. Post	−0.0578	0.6876	−0.3422	0.2265
	e. REF vs. 100%	−0.2126	0.2790	−0.5998	0.1747
	f. REF vs. FP	−0.2377	0.2354	−0.6325	0.1572
	g. REF vs. 0%	−0.2055	0.3027	−0.5988	0.1878
	h. 100% vs. FP	−0.0251	0.9007	−0.4228	0.3726
	i. 100% vs. 0%	0.0071	0.9719	−0.3892	0.4033
	j. FP vs. 0%	0.0322	0.8748	−0.3715	0.4358

Plecoptera comprised most of the biomass exported per day in the PRE period from the reference and from the FP and 0% treatments (**Figure 14-9**). During the PRE period in the 100% treatment, Diptera were the most abundant. Most of the Plecoptera and Diptera biomass was made up of a few, but large, individuals, such as *Doroneuria*, *Calineuria californica*, *Pteronarcys*, and *Tipula* (Diptera: Tipulidae). Ephemeroptera made up a larger proportion of the biomass in the POST period from the reference and from the buffer treatments and consisted predominantly of *Baetis* and *Ironodes* (Ephemeroptera: Heptageniidae).

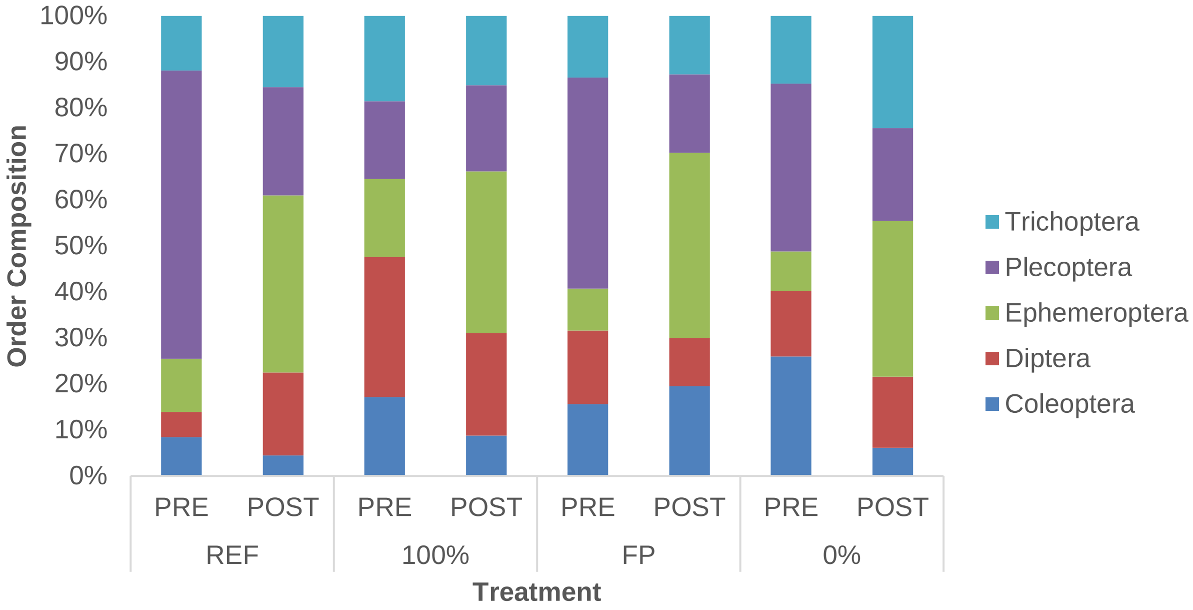

Figure 14-9. Composition of macroinvertebrate export in biomass per day (mg day^{-1}) from the Type N Study treatments by order and treatment period. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period.

14-6. DISCUSSION

We did not detect a significant treatment × period interaction for total macroinvertebrate export in numbers or biomass per day. Although we did observe a pre- to post-harvest increase in total macroinvertebrate export in numbers per day from the FP treatment, we observed the same increase in the reference, which suggests an effect across years rather than a buffer treatment effect. Environmental factors such as climate or the natural variability in macroinvertebrate communities may explain these observations. Similarly, in a BACI study of drift response to different tree retention treatments in headwater stream basins of southeast Alaska, Musslewhite and Wipfli (2004) did not detect a relationship between drift export and tree retention treatment, although they observed a decrease in mean number of individuals and in biomass export from most of their treatments, including their unharvested reference.

Although we expected that the removal of timber would cause a decrease in allochthonous inputs, which may initiate drift in some taxa and functional feeding groups, and that the increase in light, especially in the FP and 0% treatments, would stimulate periphyton growth and thus provide an alternative food resource for other taxa and feeding groups, the observed functional feeding group response was variable. There was a treatment × period interaction for parasite and scraper export in numbers per day, and for collector-gatherer, parasite, and unknown export in biomass per day, but export of the other feeding groups, including predators and shredders, did not change.

Parasite export in numbers per day increased from the pre- to post-harvest periods in the reference, but decreased in numbers and/or biomass per day in the 100% and FP treatments. There was also a decrease in parasite export in numbers and biomass per day from the 100%, FP, and 0% treatments relative to the reference, and parasites decreased in proportion in the post-harvest period. In the Type N Study, parasites consisted entirely of aquatic and terrestrial mites. Aquatic mites, depending on the stage of their life cycle, may parasitize other aquatic invertebrates or adult insects emerging from the stream, while terrestrial mites may enter the stream when searching for emerging adult hosts, or may fall into the stream from the surrounding riparian vegetation (Smith *et al.* 2001). A reduction in parasites in the riparian buffer treatments may indicate unfavorable stream conditions leading to a reduction in suitable hosts for aquatic mites, or may have resulted from the removal of timber and other riparian vegetation that serve as a substrate for terrestrial mites, at least in the FP and 0% treatments. As with total macroinvertebrate export, however, there was also a change in parasite numbers in the reference.

Scraper export in numbers per day increased from the pre- to post-harvest periods in the reference and in the FP treatment, and decreased in the 100% and 0% treatments relative to the reference. While an increase in scraper export from the FP treatment could indicate an overabundance of benthic scrapers resulting from an increase in autochthonous production, we observed the same response in the reference and did not see a pre- to post-harvest change in the 100% or 0% treatments. In addition, we did not detect a post-harvest change in periphyton ash-free dry mass or in chlorophyll *a* between the reference and buffer treatments (see Chapter 13 – *Biofilm and Periphyton*). If scrapers were drifting to more favorable habitat and food resources downstream, we would expect to see a similar response in the other buffer treatments, particularly in the 0% treatment.

Collector-gatherer export in biomass per day increased from the pre- to post-harvest periods in the FP and 0% treatment. In addition, collector-gatherer export in biomass per day increased in the FP treatment relative to the reference and relative to the 100% treatment, but decreased in the 0% treatment relative to the FP treatment. Collector-gatherers also comprised a large proportion of the export in numbers and biomass per day pre- and post-harvest. Studies of benthic macroinvertebrates have found an increase in numbers and/or biomass of collector-gatherers in unbuffered streams following harvest (Hawkins *et al.* 1982; Haggerty *et al.* 2004; Hernandez *et al.* 2005), and noted that collector-gatherers were dominant in all of their study sites before and after harvest (Gravelle *et al.* 2009). While drift is not always comparable to standing stocks of benthic invertebrates, an increase in collector-gatherer export in biomass per day in the Type N Study may indicate that benthic collector-gatherers were abundant, possibly in response to food resources, and had reached their carrying capacity, which then initiated a drift response.

Wood inputs to our study sites may have maintained or enhanced food resources available to collector-gatherers, and even to shredders, resulting in no change in shredder export following harvest. Substantial amounts of large and small wood were added to our study sites during an historic windstorm in the Coast Range (see Chapter 4 – *Unanticipated Disturbance*), and following harvest we found an increase in wood cover and in total and functional small wood and large wood pieces (see Chapter 6 – *Wood Recruitment and Loading*). Wood functions as a roughness element in headwater streams, creating step habitat and acting as a reservoir for sediment and organic matter (Bilby and Likens 1980; Bilby 1981; Benda *et al.* 2005). In addition, wood and its associated biofilm serve as a food source for some taxa and as a surface for net attachment, pupation, and oviposition for others (Anderson *et al.* 1978). Hetrick and colleagues (1998a, 1998b) found that an increase in wood maintains the retention of organic matter in harvested streams to levels comparable with reference streams. Other studies have also attributed an increase in benthic collector-gatherers to an influx of wood and detrital accretion (Cole *et al.* 2003; Haggerty *et al.* 2004), and have found a higher proportion of shredders in association with wood (Hernandez *et al.* 2005).

The presence of wood in our study sites may have also prevented a shift from an allochthonous to an autochthonous-based system, and may explain the lack of a consistent response in scraper export between our buffer treatments. Studies have shown an increase in periphyton biomass in streams bordered by clearcuts or narrow buffers (Murphy and Hall 1981; Kiffney and Bull 2000; Kiffney *et al.* 2003; Danehy *et al.* 2007), but cover provided by wood, in combination with the steep slopes characteristic of incised headwater stream channels, may have limited the amount of radiation reaching our streams (Kobayashi *et al.* 2010). In addition, the reduced velocity/ depositional environment created by wood increases the recruitment of finer sediments (Jackson *et al.* 2001), which may also inhibit periphyton growth. Wallace and colleagues (1995) observed a decrease in benthic scraper abundance and biomass after the addition of wood to streams.

Export of the unknown feeding group in biomass per day increased from the pre- to post-harvest periods in the 100% treatment, and decreased in the FP and 0% treatments relative to the 100% treatment. Most of the biomass of the unknown feeding group consisted of diplopods (millipedes), adult Coleoptera (beetles), and adult and immature lepidopterans (butterflies and moths). Although the level of taxonomic resolution did not allow designation of some taxa as aquatic or terrestrial, the taxa comprising most of the biomass in the unknown feeding group were terrestrial. The increase in export of the unknown feeding group in the 100% treatment may indicate a favorable change in riparian conditions within the buffer following timber harvest, or that more of these taxa were forced into the buffer with the removal of timber from the rest of the stream basin, although if either of these situations were the case, we should have seen a similar response in the FP treatment, but did not. The observed change in export of the unknown feeding group may have been due to chance, as these few, but large, individuals ended up in the drift.

Predator and omnivore export did not change in response to harvest, but predators made up a large proportion of the export in biomass per day in the 0% treatment before harvest and in the 100% and 0% treatments after harvest. Omnivores made up a large proportion of the biomass in the 100% and FP treatments before harvest. As with the unknown feeding group, the predator and omnivore feeding groups consisted of few, but large, individuals. These included the predators Odonata (dragonflies), *Doroneuria*, and *Calineuria californica*, and the omnivores Decapoda (crayfish) and *Pteronarcys*, which are all semivoltine taxa. It takes more than a year for semivoltine taxa to cycle through a generation, so these taxa may be more susceptible to significant environmental changes and disturbances than those with shorter life cycles. Wallace and Gurtz (1986) also did not detect a change in larger, semivoltine taxa and suggested that because of their low fecundity and voltinism the taxa were not able to respond quickly enough to exploit new food resources observed in their streams after harvest.

Of the dominant taxa included in the analysis, only the Dixidae showed a treatment × period interaction for export in numbers and biomass per day. Dixid export in numbers and biomass per day increased from the pre- to post-harvest period in the reference and in the FP treatment, decreased in numbers in the 100% and 0% treatments relative to the reference, and decreased in biomass in the 100% treatment relative to the reference and FP treatment. In addition, larval *Dixa* were one of the dominant collector-gatherers exported in numbers per day. Elliott and Tullett (1977) also found that *Dixa* are prominent in the drift, especially when searching for pupation sites. While an increase in dixid export in the FP treatment may indicate a response to stream conditions or resources, there was also an increase in export in the reference with no similar change in the 100% and 0% treatments.

Though there was no change in the major macroinvertebrate orders in response to harvest, Diptera and Ephemeroptera comprised most of the export in numbers per day from the reference and the buffer treatments in the pre- and post-harvest periods, while Ephemeroptera made up a large proportion of the biomass in the post-harvest period. Chironomidae were the dominant dipterans, and as a “feeding group” were also abundant in both harvest periods in the reference and buffer treatments. Likewise, *Baetis* were one of the dominant ephemeropterans and one of the most abundant collector-gatherers exported in numbers and biomass per day. Musslewhite and Wipfli (2004) also collected a large proportion of Diptera and Ephemeroptera in their drift samples, consisting primarily of Chironomidae and *Baetis*.

Chironomidae and *Baetis* have been observed to rapidly colonize areas after disturbances, including debris flows (Lamberti *et al.* 1991; Anderson 1992; Kobayashi *et al.* 2010), wildfires (Vieira *et al.* 2004), and volcanic eruptions (Anderson 1992). *Baetis* mayflies have been documented as prolific drifters (Stewart and Szczytko 1983; Anderson 1992), and are vigorous swimmers and crawlers (Mackay 1992). Baetis also drifts in the presence of predators, such as *Doroneuria* (Lancaster 1990). In addition, both Chironomidae and *Baetis* are considered generalist feeders that are able to exploit the earliest food materials on disturbed substrates (Mackay 1992) or short-term increases in primary productivity following timber harvest (Wallace and Gurtz 1986). Chironomidae and *Baetis* are also multivoltine, which means that the taxa have short life cycles and are able to cycle through multiple generations in one year. Wallace and Gurtz (1986) noted that changes in stream temperatures and food availability following timber harvest lead to higher growth rates of *Baetis* mayflies, which, combined with shorter life cycles and high fecundity, improved survivorship of the taxon.

As Wipfli and Musslewhite (2004) pointed out, an increase in macroinvertebrate export could indicate greater productivity in the headwater stream, but could also indicate a problem. Macroinvertebrate drift can occur in response to abiotic factors such as temperature, stream flows (Wiley and Kohler 1984), and sedimentation (Culp *et al.* 1986; Suren and Jowett 2001), as well as biotic factors such as food availability (Richards and Minshall 1988; Richardson 1991; Hinterleitner-Anderson *et al.* 1992; Siler *et al.* 2001), overpopulation (Waters 1961), competition

(Hildebrand 1974), and presence of predators (Wiley and Kohler 1984; Lancaster 1990). An increase in flows following harvest, for example, could dislodge macroinvertebrates from the substrate and cause them to inadvertently enter the drift (Hershey and Lamberti 1998). Also, increases in sediment suspended in the water column or deposited on the streambed may initiate drift behavior (Culp *et al.* 1986; Shaw and Richardson 2001; Suren and Jowett 2001). We did see an increase in discharge in the FP and 0% treatments, but no change in discharge in the 100% treatment and no change in turbidity in response to harvest (see Chapter 8 – *Discharge* and Chapter 10 – *Sediment Processes).* Although we did not see a treatment effect in within-site sediment input or storage, we also did not sample areas obscured by wood and so we may not have a complete picture of sediment impacts on macroinvertebrates at the microhabitat scale. An assessment of benthic macroinvertebrates, as well as a measure of sediment composition and detrital resources at each benthic sampling site, may have helped in the interpretation of the drift data.

Emergent macroinvertebrate sampling may have also helped to assess instream productivity and aid in the interpretation of the drift data. Emergent adult insects fly upstream to lay their eggs; stable isotope analysis could determine if adults are laying eggs in the same stream basin or if they are moving downstream (Hershey *et al.* 1993), or perhaps laterally to other basins with more suitable habitat conditions. Also, a finer level of taxonomic resolution would have allowed us to classify some taxa, such as Acarina and Collembola, as aquatic or terrestrial. Terrestrial invertebrates comprise about half of the diet of juvenile salmonids (Wipfli 1997; Kawaguchi and Nakano 2001; Allan *et al.* 2003), and an analysis of terrestrial invertebrates may have provided more information regarding changes in this resource and changes to the riparian environment in general.

Although functional feeding group analyses are widespread, classification of macroinvertebrate taxa into functional feeding groups may not be reliable for assessing changes in stream production. Taxa in the families Baetidae and Ephemerellidae (Ephemeroptera), Leuctridae and Nemouridae (Plecoptera), and Limnephilidae (Trichoptera) have generalized mouth parts that may be suitable for consuming coarse particulate organic matter (e.g., leaves and algae), periphyton, and fine particulate organic matter (Hawkins *et al.* 1982). *Baetis* and Amphipoda are classified as collector-gatherers in Wisseman (1998), which is the classification we used in our study. Other studies, however, classify *Baetis* as scrapers (Anderson 1992; Bilby and Bisson 1992) and Amphipoda as predators and shredders (Macneil *et al.* 1997). In addition, Chironomidae could be either collector-gatherers or predators, but we were not able to assign Chironomidae to their respective feeding groups because of taxonomic resolution. Inconsistent classifications make comparing results between studies problematic, and may affect interpretation of study results.

Error associated with the flow estimates for some of the study sites could have introduced error into our drift export estimates. We calculated flow from a flow versus stage-height curve for the size of flume used in each flume site, or from a site-specific flow versus stage-height curve developed for the two culvert sites (see Chapter 8 – *Discharge*). The flow versus stage relationships were good for the flume sites, but those for the two culvert sites did not hold during summer low flows so we set flow at a fixed value when flow was less than 3 L s^{-1}. This may have led to error in our drift export estimates as flow per area was a random effect in the GLMM model, but the numbers and biomass of macroinvertebrates drifting during the summer were low and the amount of error may be negligible.

Stream drying during the summer months could potentially influence drift. Clarke and colleagues (2010) found that the abundance, richness, and community composition of macroinvertebrates inhabiting debris dams in headwater streams changed in response to varying degrees of flow permanence. Likewise, DelRosario and Resh (2000) observed lower densities, richness, and diversity of benthic macroinvertebrates in intermittent relative to perennial streams, and Feminella (1996) found that richness and diversity of benthic macroinvertebrates were positively correlated with permanent flow. The Type N Study sites were largely perennial, and although some portions of our sites dried during the summer these areas were located more than 100 m upstream from the F/N break and our hydrology and drift sampling locations. Macroinvertebrates drift short distances (less than 100 m) during periods of low flow (Danehy *et al.* 2011), so channel drying in the upstream reaches of the study sites most likely had no influence on the drift that we sampled.

Short drift distances during low flows (Danehy *et al.* 2011), however, may have affected our drift observations in the OLYM-FP and OLYM-0%. While we tried to install our hydrological equipment, and thus drift nets, as close to the F/N break as possible, stream channel morphology ultimately restricted installation of the equipment in the OLYM-FP and OLYM-0% to stream reaches 50 m and 240 m downstream, respectively, of the treatment units. In the OLYM-FP, stream size and riparian stand conditions at the F/N break were similar to that at the flume location and there were no tributary inputs within the 50-m distance between the F/N break and flume location. Drift samples collected from the OLYM-FP may thus be representative of drift from the treatment unit. The flume at the OLYM-0%, however, was a greater distance downstream from the F/N break. Though there were no tributary inputs between the F/N break and the flume location, the riparian stand changed from a clearcut in the treatment unit to a forested canopy at the flume location, which may have resulted in the drift samples from the OLYM-0% not being representative of drift from the treatment unit, particularly during low flows.

Limitations of the macroinvertebrate methodology included a low number of replicates of each treatment and sample representativeness. We limited macroinvertebrate export sampling to the eight study sites instrumented with hydrological monitoring equipment because of the time and cost required to collect and process samples and the need for flow quantification for interpreting macroinvertebrate export. This left us with two replicates of each treatment. Sampling of macroinvertebrate export was limited to the downstream end of each study site mostly because we were interested in drift export to fish-bearing reaches and because of the need for flow quantification. This, however, may have resulted in drift samples that were not representative of the treatment unit in the OLYM-0%, and to some extent in the OLYM-FP, as discussed previously. In addition, drift sampling at the downstream end of the sites does not give us a whole picture of macroinvertebrates within the Type N streams and the factors influencing drift. Although we sampled drift every six weeks throughout the year, we omitted the samples that were not collected overnight and were left with a limited number of samples for each treatment year, which may have increased variability. Other authors have noted that a sizeable number of drift samples are necessary to reduce variability and detect treatment effects (Allan and Russek 1985; Musslewhite and Wipfli 2004). Despite collecting multiple samples over consecutive days, Musslewhite and Wipfli (2004) still measured high day-to-day variation in export from their streams, and suggested that localized habitat characteristics such as gradient and substrate may influence drift.

Although we observed some changes after harvest, there were no major reductions in macroinvertebrate export and no major shifts in functional feeding groups associated with the treatments relative to the unharvested references. A similar study that measured macroinvertebrate export from small headwater streams with continuous fixed-width buffers, patch buffers, and no buffers also detected changes after harvest, but did not see any dramatic reductions in diversity or changes in community structure (Bisson *et al.* 2013). The inherent variability of drift samples coupled with the limitations of the study design and sampling methodology may have decreased our ability to detect anything but very large changes after harvest.

14-7. CONCLUSIONS

The Type N Study was designed to evaluate the response of a variety of riparian and instream processes to different stream buffer treatments before and after harvest. We hypothesized that changes in habitat conditions and allochthonous inputs may initiate drift in some macroinvertebrate taxa and functional feeding groups, such as shredders, while increases in light and periphyton growth would provide an alternative food resource for other taxa and feeding groups, such as collector-gatherers and scrapers. We did not see a change in total macroinvertebrate export, but did see a change in parasite and scraper export in numbers per day, in collector-gatherer and parasite export in biomass per day, and in Dixidae export in numbers and biomass per day. Parasite, scraper, and Dixidae export changed in one or more of the buffer treatments, but also changed in the reference, which suggests environmental factors, such as climate, or natural variability in macroinvertebrate communities rather than a treatment effect.

Collector-gatherer export changed in the FP and 0% treatments, and made up a large proportion of export in numbers and biomass per day. Chironomidae and *Baetis*, a collector-gatherer, comprised much of the proportion of individuals exported, and *Baetis* of biomass exported. Persistence of taxa such as Chironomidae and *Baetis* likely resulted from their multivoltinism, and their ability to adapt quickly to disturbances and use available food resources. Wood inputs into the stream channels in the form of slash and windthrow may have created depositional areas that maintained or enhanced food resources for collector-gatherers such as *Baetis* and some Chironomidae. The presence of wood may have also prevented a shift from an allochthonous to an autochthonous-based system.

14-8. REFERENCES

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APPENDICES

APPENDIX 14-A. DRIFT COLLECTION LOGISTICS DATA

Appendix Table 14-A-1. Drift collection date, time, and deployment period. Samples in **bold** print were included in the analysis. Samples not in bold print were omitted from the analysis. Samples in **bold red** print were included in the analysis, but denote when part or all of the drift net detached from the flume or rebar during deployment. OLYM = Olympic Block; WIL1 = Willapa Block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2.

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
OLYM-REF	PRE1	**10/30/2006**	**10:52**	**10/31/2006**	**11:56**	**1504**
		12/13/2006	12:05	12/13/2006	12:35	30
		1/25/2007	9:20	1/25/2007	10:20	60
		3/5/2007	11:28	3/5/2007	12:28	60
		4/16/2007	**10:18**	**4/17/2007**	**7:32**	**1274**
		5/23/2007	**9:30**	**5/24/2007**	**7:38**	**1328**
		7/9/2007	**10:50**	**7/10/2007**	**8:00**	**1270**
	PRE2	**8/20/2007**	**9:40**	**8/21/2007**	**11:50**	**1570**
		10/4/2007	10:05	10/4/2007	11:05	60
		11/15/2007	10:35	11/15/2007	11:35	60
		12/19/2007	11:33	12/19/2007	12:33	60
		3/5/2008	13:50	3/5/2008	14:50	60
		4/16/2008	11:18	4/16/2008	12:18	60
		5/21/2008	**9:47**	**5/22/2008**	**8:12**	**1345**
		7/14/2008	**13:34**	**7/15/2008**	**7:38**	**1084**
	POST1	**10/5/2009**	**11:30**	**10/6/2009**	**10:11**	**1361**
		11/10/2009	10:29	11/10/2009	10:52	23
		12/9/2009	12:06	12/9/2009	13:10	64
		1/20/2010	11:02	1/20/2010	12:02	60
		2/23/2010	10:31	2/23/2010	11:31	60
		4/15/2010	8:27	4/15/2010	9:27	60
		5/25/2010	**14:20**	**5/26/2010**	**10:47**	**1227**
		7/13/2010	**9:45**	**7/14/2010**	**9:49**	**1444**
		8/16/2010	**9:18**	**8/17/2010**	**15:26**	**1808**
	POST2	**9/27/2010**	**10:50**	**9/28/2010**	**10:04**	**1394**
		11/8/2010	10:18	11/8/2010	11:18	60
		1/18/2011	11:50	1/18/2011	11:55	5
		4/13/2011	8:54	4/13/2011	9:41	47
		5/18/2011	8:49	5/18/2011	9:49	60
		7/5/2011	**9:30**	**7/6/2011**	**11:58**	**1588**
		8/22/2011	**9:24**	**8/23/2011**	**8:40**	**1396**

Appendix Table 14-A-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
OLYM-100%	PRE1	**10/30/2006**	**13:36**	**10/31/2006**	**8:30**	**1134**
		12/11/2006	15:19	12/11/2006	15:49	30
		1/24/2007	10:39	1/24/2007	11:39	60
		3/6/2007	9:43	3/6/2007	10:45	62
		4/18/2007	**14:00**	**4/19/2007**	**7:15**	**1035**
		5/21/2007	**12:35**	**5/22/2007**	**7:30**	**1135**
		7/11/2007	**15:50**	**7/12/2007**	**7:55**	**965**
	PRE2	**8/20/2007**	**12:45**	**8/21/2007**	**7:20**	**1115**
		10/1/2007	**16:40**	**10/2/2007**	**7:25**	**885**
		11/14/2007	8:45	11/14/2007	9:45	60
		12/18/2007	8:58	12/18/2007	9:58	60
		1/24/2008	9:08	1/24/2008	10:08	60
		3/4/2008	9:58	3/4/2008	10:58	60
		4/14/2008	10:27	4/14/2008	11:31	64
		5/21/2008	**14:22**	**5/22/2008**	**11:10**	**1248**
		7/7/2008	**14:10**	**7/8/2008**	**7:12**	**1022**
	POST1	**10/5/2009**	**15:00**	**10/6/2009**	**7:22**	**982**
		11/10/2009	15:50	11/10/2009	16:45	55
		12/10/2009	10:31	12/10/2009	11:31	60
		1/25/2010	10:15	1/25/2010	11:17	62
		3/3/2010	8:31	3/3/2010	9:41	70
		4/12/2010	9:06	4/12/2010	10:08	62
		5/26/2010	**7:20**	**5/27/2010**	**6:59**	**1419**
		7/6/2010	**9:37**	**7/7/2010**	**9:14**	**1417**
		8/16/2010	**12:51**	**8/17/2010**	**7:20**	**1109**
	POST2	**9/27/2010**	**15:35**	**9/28/2010**	**7:33**	**958**
		11/10/2010	10:14	11/10/2010	11:14	60
		12/13/2010	13:25	12/13/2010	14:29	64
		1/18/2011	16:53	1/18/2011	17:16	23
		2/28/2011	10:22	2/28/2011	11:22	60
		4/12/2011	9:02	4/12/2011	10:00	58
		5/18/2011	12:10	5/18/2011	13:10	60
		7/6/2011	**14:37**	**7/7/2011**	**8:47**	**1090**
		8/16/2011	**14:05**	**8/17/2011**	**9:32**	**1167**

Appendix Table 14-A-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
OLYM-FP	PRE1	**11/1/2006**	**11:52**	**11/2/2006**	**11:55**	**1443**
		12/18/2006	11:38	12/18/2006	12:38	60
		1/22/2007	11:11	1/22/2007	12:11	60
		3/7/2007	10:43	3/7/2007	11:45	62
		4/18/2007	**10:08**	**4/19/2007**	**11:25**	**1517**
		5/21/2007	**9:20**	**5/22/2007**	**11:33**	**1573**
		7/11/2007	**12:50**	**7/12/2007**	**12:05**	**1395**
	PRE2	**8/22/2007**	**13:01**	**8/23/2007**	**11:38**	**1357**
		10/1/2007	**10:15**	**10/2/2007**	**11:54**	**1539**
		11/13/2007	9:55	11/13/2007	10:55	60
		12/17/2007	14:10	12/17/2007	15:12	62
		1/22/2008	12:56	1/22/2008	13:56	60
		3/3/2008	11:41	3/3/2008	12:41	60
		4/17/2008	9:40	4/17/2008	10:40	60
		6/3/2008	**9:46**	**6/4/2008**	**11:43**	**1557**
		7/14/2008	**10:27**	**7/15/2008**	**10:26**	**1439**
	POST1	**10/7/2009**	**11:54**	**10/8/2009**	**8:49**	**1255**
		11/11/2009	12:35	11/11/2009	13:33	58
		12/14/2009	12:25	12/14/2009	13:50	85
		1/19/2010	13:53	1/19/2010	13:56	3
		3/2/2010	13:10	3/2/2010	14:28	78
		4/19/2010	8:42	4/19/2010	9:42	60
		5/25/2010	**11:20**	**5/26/2010**	**15:07**	**1667**
		7/7/2010	**16:34**	**7/8/2010**	**15:47**	**1393**
		8/17/2010	**12:05**	**8/18/2010**	**9:30**	**1285**
	POST2	9/30/2010	7:50	9/30/2010	8:52	62
		11/9/2010	11:59	11/9/2010	12:58	59
		12/14/2010	11:52	12/14/2010	11:59	7
		1/24/2011	10:27	1/24/2011	10:33	6
		3/7/2011	13:51	3/7/2011	15:31	100
		4/11/2011	10:18	4/11/2011	10:42	24
		8/15/2011	**9:09**	**8/16/2011**	**11:00**	**1551**

Appendix Table 14-A-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
OLYM-0%	PRE1	**11/1/2006**	**15:30**	**11/2/2006**	**8:50**	**1040**
		12/11/2006	10:40	12/11/2006	11:28	48
		1/23/2007	10:34	1/23/2007	11:34	60
		3/5/2007	13:56	3/5/2007	15:00	64
		4/16/2007	**12:48**	**4/17/2007**	**9:38**	**1250**
		5/23/2007	**11:25**	**5/24/2007**	**9:58**	**1353**
		7/9/2007	**14:20**	**7/10/2007**	**11:02**	**1242**
	PRE2	**8/22/2007**	**10:43**	**8/23/2007**	**7:55**	**1272**
		10/3/2007	9:35	10/3/2007	10:50	75
		11/13/2007	14:40	11/13/2007	15:40	60
		12/17/2007	10:58	12/17/2007	11:58	60
		1/23/2008	11:24	1/23/2008	12:24	60
		3/3/2008	14:51	3/3/2008	15:51	60
		4/15/2008	8:50	4/15/2008	9:57	67
		5/27/2008	**13:34**	**5/28/2008**	**7:44**	**1090**
		7/7/2008	**10:44**	**7/8/2008**	**10:01**	**1397**
	POST1	**10/7/2009**	**14:30**	**10/8/2009**	**11:21**	**1251**
		11/11/2009	8:34	11/11/2009	9:28	54
		12/15/2009	13:31	12/15/2009	14:30	59
		1/19/2010	10:21	1/19/2010	10:31	10
		3/2/2010	9:55	3/2/2010	10:55	60
		4/20/2010	9:21	4/20/2010	10:21	60
		5/25/2010	**8:45**	**5/26/2010**	**13:39**	**1734**
		7/6/2010	**11:45**	**7/7/2010**	**8:28**	**1243**
		8/17/2010	**10:12**	**8/18/2010**	**7:53**	**1301**
	POST2	9/29/2010	10:42	9/29/2010	11:44	62
		11/9/2010	9:52	11/9/2010	10:49	57
		12/14/2010	9:46	12/14/2010	9:55	9
		1/19/2011	12:45	1/19/2011	13:00	15
		3/7/2011	10:30	3/7/2011	11:30	60
		4/11/2011	12:18	4/11/2011	12:50	32
		5/19/2011	9:05	5/19/2011	10:01	56
		7/5/2011	**12:32**	**7/6/2011**	**9:08**	**1236**
		8/15/2011	**11:33**	**8/16/2011**	**9:07**	**1294**

Appendix Table 14-A-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
WIL1-REF	PRE1	**10/16/2006**	**12:15**	**10/17/2006**	**12:10**	**1435**
		12/4/2006	**12:57**	**12/5/2006**	**12:54**	**1437**
		1/9/2007	12:42	1/9/2007	13:42	60
		2/21/2007	10:12	2/22/2007	9:35	1403
		4/4/2007	**8:45**	**4/5/2007**	**8:45**	**1440**
	PRE2	**5/16/2007**	**9:00**	**5/17/2007**	**8:35**	**1415**
		6/26/2007	**11:30**	**6/27/2007**	**11:40**	**1450**
		8/8/2007	**9:26**	**8/9/2007**	**9:15**	**1429**
		9/17/2007	**12:12**	**9/18/2007**	**11:49**	**1417**
		10/29/2007	**11:55**	**10/30/2007**	**11:49**	**1434**
		12/12/2007	12:55	12/12/2007	13:57	62
		1/7/2008	14:23	1/7/2008	15:23	60
		2/19/2008	15:22	2/19/2008	16:22	60
		4/1/2008	9:49	4/1/2008	10:51	62
	POST1	**5/11/2009**	**9:05**	**5/12/2009**	**8:17**	**1392**
		6/29/2009	**12:36**	**6/30/2009**	**10:18**	**1302**
		8/4/2009	**11:00**	**8/5/2009**	**6:59**	**1199**
		9/15/2009	**9:55**	**9/16/2009**	**8:32**	**1357**
		10/26/2009	8:42	10/26/2009	9:42	60
		12/7/2009	9:32	12/7/2009	10:31	59
		1/5/2010	12:49	1/5/2010	13:49	60
		2/16/2010	12:40	2/16/2010	13:40	60
		3/30/2010	7:23	3/30/2010	8:28	65
	POST2	**5/11/2010**	**10:05**	**5/12/2010**	**6:53**	**1248**
		6/21/2010	**13:18**	**6/22/2010**	**7:02**	**1064**
		8/3/2010	**11:10**	**8/4/2010**	**9:35**	**1345**
		9/14/2010	**14:33**	**9/15/2010**	**11:06**	**1233**
		10/26/2010	12:18	10/26/2010	13:16	58
		12/7/2010	12:41	12/7/2010	13:42	61
		1/3/2011	12:26	1/3/2011	13:26	60
		2/17/2011	9:24	2/17/2011	10:25	61
		3/30/2011	10:08	3/30/2011	10:31	23

Appendix Table 14-A-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
WIL1-100%	PRE1	**10/18/2006**	**10:20**	**10/19/2006**	**11:04**	**1484**
		12/6/2006	**13:26**	**12/7/2006**	**12:00**	**1354**
		1/18/2007	10:19	1/18/2007	11:19	60
		2/20/2007	14:40	2/20/2007	15:20	40
		4/2/2007	**12:08**	**4/3/2007**	**11:34**	**1406**
	PRE2	**5/14/2007**	**10:55**	**5/15/2007**	**11:05**	**1450**
		6/28/2007	**11:00**	**6/29/2007**	**10:17**	**1397**
		8/6/2007	**9:15**	**8/7/2007**	**8:40**	**1405**
		9/17/2007	**10:36**	**9/18/2007**	**7:26**	**1250**
		10/31/2007	**9:00**	**11/1/2007**	**8:36**	**1416**
		12/11/2007	11:02	12/11/2007	12:02	60
		1/8/2008	9:15	1/8/2008	10:16	61
		2/21/2008	9:44	2/21/2008	10:44	60
		3/31/2008	14:02	3/31/2008	15:02	60
	POST1	5/11/2009	10:05	5/11/2009	11:05	60
		6/30/2009	**7:04**	**7/1/2009**	**6:42**	**1418**
		8/4/2009	**13:07**	**8/5/2009**	**9:00**	**1193**
		9/15/2009	**7:12**	**9/16/2009**	**8:21**	**1509**
		10/27/2009	12:18	10/27/2009	13:18	60
		12/8/2009	12:48	12/8/2009	13:47	59
		1/6/2010	11:13	1/6/2010	12:12	59
		2/18/2010	11:54	2/18/2010	13:02	68
		4/1/2010	8:40	4/1/2010	9:44	64
	POST2	**5/11/2010**	**12:45**	**5/12/2010**	**9:03**	**1218**
		6/21/2010	**14:48**	**6/22/2010**	**9:21**	**1113**
		8/3/2010	**13:38**	**8/4/2010**	**12:05**	**1347**
		9/14/2010	**16:05**	**9/15/2010**	**14:34**	**1349**
		10/27/2010	10:30	10/27/2010	11:30	60
		12/8/2010	11:17	12/8/2010	12:17	60
		1/4/2011	11:34	1/4/2011	12:34	60
		2/17/2011	12:49	2/17/2011	13:50	61
		3/30/2011	8:26	3/30/2011	8:44	18

Appendix Table 14-A-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
WIL1-FP	PRE1	**10/18/2006**	**9:24**	**10/19/2006**	**8:50**	**1406**
		12/6/2006	**9:55**	**12/7/2006**	**10:13**	**1458**
		1/9/2007	13:38	1/9/2007	14:38	60
		2/21/2007	13:22	2/22/2007	11:05	1303
		4/4/2007	**12:15**	**4/5/2007**	**10:25**	**1330**
	PRE2	**5/16/2007**	**10:55**	**5/17/2007**	**10:15**	**1400**
		6/28/2007	**9:10**	**6/29/2007**	**8:43**	**1413**
		8/8/2007	**12:30**	**8/9/2007**	**11:54**	**1404**
		9/19/2007	**11:19**	**9/20/2007**	**8:30**	**1271**
		10/31/2007	**10:15**	**11/1/2007**	**9:57**	**1422**
		12/11/2007	13:45	12/11/2007	14:50	65
		1/8/2008	12:32	1/8/2008	13:32	60
		2/21/2008	13:20	2/21/2008	14:20	60
		4/1/2008	7:33	4/1/2008	8:33	60
	POST1	5/12/2009	9:56	5/12/2009	10:53	57
		6/29/2009	**13:49**	**6/30/2009**	**8:59**	**1150**
		8/4/2009	**12:11**	**8/5/2009**	**8:00**	**1189**
		9/14/2009	**13:57**	**9/15/2009**	**8:44**	**1127**
		10/26/2009	11:20	10/26/2009	12:20	60
		12/7/2009	11:58	12/7/2009	12:59	61
		1/6/2010	9:15	1/6/2010	10:15	60
		2/18/2010	9:17	2/18/2010	10:17	60
		3/31/2010	10:14	3/31/2010	11:14	60
	POST2	**5/11/2010**	**11:53**	**5/12/2010**	**7:59**	**1206**
		6/21/2010	**14:14**	**6/22/2010**	**8:11**	**1077**
		8/3/2010	**12:37**	**8/4/2010**	**11:21**	**1364**
		9/14/2010	**15:36**	**9/15/2010**	**13:04**	**1288**
		10/27/2010	8:30	10/27/2010	9:30	60
		12/8/2010	9:35	12/8/2010	10:35	60
		1/4/2011	10:06	1/4/2011	11:06	60
		2/17/2011	11:10	2/17/2011	12:10	60
		3/29/2011	13:05	3/29/2011	14:05	60

Appendix Table 14-A-1. (continued)

Site	Treatment Year	Net Installation		Net Retrieval		Deployment Period (minutes)
		Date	Time	Date	Time	
WIL1-0%	PRE1	**10/16/2006**	**9:20**	**10/17/2006**	**8:20**	**1380**
		12/4/2006	**10:10**	**12/5/2006**	**9:17**	**1387**
		1/8/2007	10:36	1/8/2007	11:36	60
		2/20/2007	11:50	2/20/2007	12:45	55
		4/2/2007	**8:19**	**4/3/2007**	**8:12**	**1433**
	PRE2	**5/14/2007**	**8:24**	**5/15/2007**	**8:00**	**1416**
		6/26/2007	**8:33**	**6/27/2007**	**8:08**	**1415**
		8/6/2007	**11:40**	**8/7/2007**	**12:53**	**1513**
		9/19/2007	**13:38**	**9/20/2007**	**11:41**	**1323**
		10/29/2007	**8:41**	**10/30/2007**	**8:20**	**1419**
		12/27/2007	10:32	12/27/2007	11:32	60
		1/7/2008	9:41	1/7/2008	10:41	60
		2/19/2008	10:52	2/19/2008	11:52	60
		3/31/2008	9:19	3/31/2008	10:47	88
	POST1	5/13/2009	8:08	5/13/2009	9:05	57
		6/29/2009	**8:12**	**6/30/2009**	**12:28**	**1696**
		8/4/2009	**8:09**	**8/5/2009**	**11:29**	**1640**
		9/14/2009	**8:34**	**9/15/2009**	**12:21**	**1667**
		10/27/2009	8:15	10/27/2009	9:15	60
		12/8/2009	9:24	12/8/2009	10:23	59
		1/5/2010	9:13	1/5/2010	10:16	63
		2/16/2010	9:04	2/16/2010	10:05	61
		3/29/2010	8:46	3/29/2010	9:46	60
	POST2	**5/11/2010**	**7:53**	**5/12/2010**	**11:07**	**1634**
		6/21/2010	**10:29**	**6/22/2010**	**12:10**	**1541**
		8/3/2010	**8:24**	**8/4/2010**	**7:26**	**1382**
		9/14/2010	**11:13**	**9/15/2010**	**7:56**	**1243**
		10/26/2010	8:21	10/26/2010	9:11	50
		12/7/2010	9:21	12/7/2010	10:22	61
		1/3/2011	9:23	1/3/2011	10:23	60
		3/1/2011	9:30	3/1/2011	10:30	58
		3/29/2011	8:30	3/29/2011	9:17	47

APPENDIX 14-B. DESCRIPTIVE STATISTICS

Appendix Table 14-B-1. Descriptive statistics for macroinvertebrate export in numbers per day (# day^{-1}) from the Type N Study sites by TOTAL, functional feeding group, and treatment year. OLYM = Olympic block; WIL1 = Willapa 1 block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2; TOTAL = total macroinvertebrates; CH = Chironomidae; CF = collector-filterer; CG = collector-gatherer; OM = omnivore; PA = parasite; PR = predator; SC = scraper; SH = shredder; UN = unknown.

Feeding Group	**Treatment Year**	**OLYM-REF**		**OLYM-100%**		**OLYM-FP**		**OLYM-0%**	
		MEAN	**SD**	**MEAN**	**SD**	**MEAN**	**SD**	**MEAN**	**SD**
TOTAL	PRE1	456.10	238.44	445.42	482.48	270.89	36.06	747.80	243.91
	PRE2	843.11	519.10	1045.15	1223.06	858.04	808.18	558.38	452.43
	POST1	1243.65	467.83	663.52	619.25	781.79	578.15	411.99	530.54
	POST2	2298.60	946.54	728.29	661.28	964.64	-	1009.90	709.00
CH	PRE1	98.56	79.97	218.98	242.89	128.58	65.48	432.09	370.06
	PRE2	113.86	74.79	99.38	115.34	124.24	101.81	160.92	112.93
	POST1	298.58	228.80	231.40	266.72	223.64	141.86	215.46	353.37
	POST2	715.33	529.34	247.02	337.06	148.55	-	231.02	106.38
CF	PRE1	19.71	26.16	36.15	67.95	3.33	1.93	24.11	35.93
	PRE2	35.61	22.49	56.35	100.24	11.89	5.28	8.45	4.51
	POST1	68.41	73.55	18.87	18.71	17.15	16.36	3.96	4.95
	POST2	14.92	19.69	16.26	14.08	13.00	-	10.55	3.91
CG	PRE1	79.67	45.06	69.57	83.15	46.25	51.72	125.75	144.36
	PRE2	470.97	415.46	247.08	227.53	309.33	415.50	188.24	227.13
	POST1	519.97	360.96	255.81	204.12	398.35	468.61	73.56	87.01
	POST2	884.63	383.83	226.24	140.07	597.91	-	286.02	113.34
OM	PRE1	1.47	2.13	0.00	0.00	1.77	1.29	0.00	0.00
	PRE2	0.55	0.96	1.36	2.71	1.54	1.98	0.00	0.00
	POST1	0.00	0.00	0.00	0.00	0.00	0.00	0.28	0.55
	POST2	0.44	0.77	0.00	0.00	0.93	-	2.22	3.14
PA	PRE1	109.73	72.40	45.43	29.88	53.35	56.16	44.16	25.34
	PRE2	51.26	39.67	239.50	433.48	213.31	388.81	22.46	8.35
	POST1	63.50	53.52	18.60	16.38	30.53	13.07	18.81	14.25
	POST2	228.38	142.89	71.30	110.56	41.78	-	37.74	40.78
PR	PRE1	22.69	10.68	21.04	15.95	8.05	4.16	23.73	15.37
	PRE2	34.08	15.13	23.81	29.46	25.29	20.59	22.04	7.36
	POST1	38.72	30.56	26.23	27.44	22.50	13.45	17.55	23.07
	POST2	55.73	29.44	25.06	29.44	4.64	-	39.40	41.56
SC	PRE1	15.63	24.00	17.90	32.50	2.78	2.81	7.01	4.50
	PRE2	57.85	41.27	13.02	10.02	15.75	13.27	24.58	19.64
	POST1	113.50	29.57	17.83	18.89	29.14	27.72	5.14	4.46
	POST2	220.80	228.44	13.12	11.17	41.78	-	12.22	7.83
SH	PRE1	47.43	21.94	33.10	30.28	20.55	11.77	72.61	93.29
	PRE2	70.67	30.75	27.02	32.17	29.63	20.65	122.67	91.51
	POST1	118.12	95.35	58.34	53.97	38.27	13.20	52.69	44.29
	POST2	138.00	56.12	71.07	63.20	41.78	-	305.25	315.23
UN	PRE1	61.20	106.28	3.25	4.63	6.23	6.87	18.34	7.94
	PRE2	8.25	2.56	337.63	671.64	127.05	234.88	9.02	12.79
	POST1	22.87	23.05	36.43	53.56	22.20	24.52	24.55	22.11
	POST2	40.38	23.78	58.23	68.16	74.27	-	85.48	76.82

Appendix Table 14-B-1. (continued)

Feeding Group	Treatment Year	WIL1-REF		WIL1-100%		WIL1-FP		WIL1-0%	
		MEAN	SD	MEAN	SD	MEAN	SD	MEAN	SD
TOTAL	PRE1	230.22	248.17	541.11	274.67	278.23	187.31	399.83	43.93
	PRE2	175.81	54.44	233.51	120.05	208.20	130.66	369.00	151.82
	POST1	308.27	124.60	480.78	623.97	513.52	383.43	559.69	355.59
	POST2	473.58	234.25	657.92	404.21	707.96	452.79	1649.28	920.35
CH	PRE1	55.69	76.77	116.88	75.32	17.12	16.26	96.44	33.73
	PRE2	21.47	37.16	71.76	69.45	31.03	26.42	129.53	145.55
	POST1	31.92	38.72	110.75	101.81	40.48	30.63	181.43	207.01
	POST2	70.84	66.08	117.13	99.10	80.35	83.47	266.52	114.87
CF	PRE1	8.06	11.40	15.36	6.91	4.17	4.52	13.87	11.23
	PRE2	7.99	15.16	12.53	15.81	14.08	11.19	12.17	9.44
	POST1	9.61	7.36	69.90	110.02	10.26	7.93	25.80	18.77
	POST2	39.08	42.10	15.12	21.27	49.88	48.68	225.65	222.51
CG	PRE1	53.67	48.57	138.02	115.78	119.72	97.96	75.20	43.56
	PRE2	101.57	54.33	82.59	43.14	57.50	49.58	76.48	31.63
	POST1	208.05	103.71	242.62	359.87	356.63	313.27	172.82	230.86
	POST2	235.01	70.61	444.16	259.87	483.30	264.57	927.36	686.54
OM	PRE1	0.00	0.00	1.48	2.56	0.71	0.61	0.00	0.00
	PRE2	1.39	3.11	0.00	0.00	0.23	0.51	0.80	0.81
	POST1	1.13	1.56	0.42	0.73	0.00	0.00	2.30	2.00
	POST2	1.70	1.82	0.91	1.22	0.30	0.60	0.53	0.62
PA	PRE1	71.37	80.92	77.50	85.21	35.08	30.94	136.85	115.78
	PRE2	16.90	18.91	28.91	18.02	43.48	19.34	61.35	44.13
	POST1	29.81	24.05	6.55	3.46	3.68	3.16	16.56	16.11
	POST2	22.26	14.38	9.13	8.98	8.88	5.65	20.10	17.13
PR	PRE1	11.68	7.63	23.41	6.82	13.60	8.93	18.51	5.65
	PRE2	4.87	2.88	9.12	5.70	6.61	6.55	20.52	4.82
	POST1	7.06	8.39	9.53	14.88	2.88	1.81	12.67	1.41
	POST2	21.39	14.79	6.67	8.08	6.22	6.36	24.54	21.41
SC	PRE1	1.04	1.05	18.60	13.33	19.45	30.00	1.80	1.66
	PRE2	1.42	1.54	8.39	9.40	2.87	4.25	6.71	3.61
	POST1	5.65	4.12	2.86	4.15	1.26	1.28	3.43	3.04
	POST2	36.88	33.56	6.21	4.25	25.39	33.06	12.26	14.94
SH	PRE1	27.66	32.16	145.94	107.11	59.38	68.49	47.92	49.30
	PRE2	17.99	17.66	17.75	11.69	45.22	79.26	51.74	42.70
	POST1	11.75	2.87	33.72	43.92	96.29	73.68	135.53	117.63
	POST2	25.34	6.77	45.22	43.13	35.66	25.29	139.90	136.71
UN	PRE1	1.04	1.05	3.91	3.38	9.00	9.65	9.24	10.49
	PRE2	2.21	1.77	2.47	1.95	7.19	7.50	9.70	12.18
	POST1	3.31	0.87	4.43	2.09	2.05	1.86	9.15	6.06
	POST2	21.07	11.70	13.37	11.19	18.00	9.92	32.41	16.37

Appendix Table 14-B-2. Descriptive statistics for macroinvertebrate export in biomass per day (mg day^{-1}) from the Type N Study sites by TOTAL, functional feeding group, and treatment year. OLYM = Olympic block; WIL1 = Willapa 1 block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2; TOTAL = total macroinvertebrates; CH = Chironomidae; CF = collector-filterer; CG = collector-gatherer; OM = omnivore; PA = parasite; PR = predator; SC = scraper; SH = shredder; UN = unknown.

Feeding Group	Treatment Year	OLYM-REF		OLYM-100%		OLYM-FP		OLYM-0%	
		MEAN	SD	MEAN	SD	MEAN	SD	MEAN	SD
TOTAL	PRE1	196.95	159.76	96.37	86.97	4001.99	4795.72	191.34	138.61
	PRE2	185.35	48.91	131.36	175.34	712.22	566.66	188.47	231.71
	POST1	170.34	133.15	164.11	97.62	127.15	77.87	49.57	35.54
	POST2	140.41	51.88	82.94	34.92	144.63	-	233.30	144.87
CH	PRE1	4.26	3.41	12.13	12.78	6.70	3.68	24.65	22.27
	PRE2	5.41	3.48	5.69	8.43	6.25	4.08	10.11	5.68
	POST1	8.92	4.80	7.42	9.80	7.60	4.91	6.59	10.54
	POST2	31.95	16.09	11.82	18.80	7.68	-	10.86	8.49
CF	PRE1	25.05	21.84	1.84	3.49	3.62	6.14	2.69	2.28
	PRE2	23.13	13.54	1.24	1.20	7.28	7.75	3.38	1.80
	POST1	23.20	30.47	4.11	4.38	2.44	1.82	0.84	1.18
	POST2	1.48	1.87	5.49	8.49	2.23	-	2.86	2.17
CG	PRE1	16.11	13.69	7.12	10.62	1.86	1.20	36.36	42.69
	PRE2	44.67	12.76	18.69	16.40	35.08	54.12	24.92	34.11
	POST1	47.90	16.85	23.67	25.35	29.91	22.14	15.93	13.05
	POST2	28.16	19.88	14.88	19.25	42.13	-	45.06	45.14
OM	PRE1	16.49	31.96	0.00	0.00	3913.90	4794.83	0.00	0.00
	PRE2	2.51	4.35	37.52	75.05	259.62	501.25	0.00	0.00
	POST1	0.00	0.00	0.00	0.00	0.00	0.00	0.12	0.25
	POST2	0.53	0.93	0.00	0.00	8.54	-	15.29	21.62
PA	PRE1	6.08	3.89	2.50	1.72	2.90	2.90	2.34	1.34
	PRE2	2.72	2.10	12.69	22.97	11.36	20.57	1.19	0.44
	POST1	3.37	2.84	0.99	0.87	1.62	0.69	1.00	0.76
	POST2	12.10	7.57	3.78	5.86	2.21	-	2.00	2.16
PR	PRE1	65.44	67.97	14.83	19.46	22.36	29.18	36.34	23.17
	PRE2	16.79	2.60	16.16	19.91	214.03	230.96	40.77	39.68
	POST1	21.47	16.27	92.23	107.75	13.88	10.35	10.26	7.38
	POST2	28.99	11.25	13.75	17.22	50.01	-	97.57	3.87
SC	PRE1	11.26	21.27	14.15	22.45	1.34	1.95	22.81	21.58
	PRE2	36.92	28.76	5.99	5.08	14.33	11.36	15.59	13.90
	POST1	26.44	40.03	9.94	17.89	11.96	6.89	2.80	2.87
	POST2	12.07	16.79	5.07	1.31	0.43	-	6.94	3.48
SH	PRE1	35.90	23.34	13.91	19.16	4.39	2.20	16.37	17.39
	PRE2	28.74	24.00	6.69	8.99	9.35	6.63	9.91	5.27
	POST1	22.04	30.82	7.09	7.97	15.49	19.45	7.37	5.65
	POST2	13.91	8.81	5.53	6.28	4.96	-	35.14	43.35
UN	PRE1	16.37	24.92	29.87	59.44	44.93	40.57	49.77	42.03
	PRE2	24.46	18.71	26.67	44.22	154.93	135.36	82.60	140.73
	POST1	16.99	12.87	18.65	23.00	44.25	49.79	4.67	7.45
	POST2	11.21	10.13	22.62	20.84	26.46	-	17.59	22.33

Appendix Table 14-B-2. (continued)

Feeding Group	Treatment Year	WIL1-REF		WIL1-100%		WIL1-FP		WIL1-0%	
		MEAN	SD	MEAN	SD	MEAN	SD	MEAN	SD
TOTAL	PRE1	481.35	767.50	3581.57	6069.35	224.01	86.23	350.86	299.73
	PRE2	122.97	62.82	46.03	54.59	28.62	8.44	119.49	59.63
	POST1	134.93	33.35	172.41	247.59	70.75	50.18	114.90	48.01
	POST2	219.20	92.70	91.05	110.26	195.39	46.39	279.55	181.42
CH	PRE1	1.52	1.67	4.26	0.65	1.03	0.70	6.30	0.79
	PRE2	0.79	0.84	2.20	1.93	1.14	0.46	5.60	4.86
	POST1	2.17	3.03	9.00	6.61	2.12	1.65	10.81	11.98
	POST2	3.17	2.91	6.52	6.47	3.92	2.90	13.63	6.25
CF	PRE1	1.65	1.93	12.48	14.30	4.89	6.04	8.19	3.55
	PRE2	0.63	0.78	18.34	38.61	2.86	3.26	2.69	2.71
	POST1	1.40	1.01	7.62	10.46	1.26	0.89	6.19	3.94
	POST2	10.49	16.18	9.42	12.70	20.02	23.60	35.86	28.50
CG	PRE1	14.34	9.75	9.74	2.63	12.53	11.11	39.55	41.19
	PRE2	85.97	76.32	5.54	3.00	6.36	4.23	9.09	6.61
	POST1	60.87	24.45	10.17	11.17	28.42	16.52	18.57	14.80
	POST2	93.56	41.60	32.98	47.70	112.87	60.65	100.90	61.92
OM	PRE1	0.00	0.00	3503.92	6068.96	28.40	48.98	0.00	0.00
	PRE2	1.62	3.63	0.00	0.00	2.55	5.71	21.18	19.86
	POST1	13.50	20.32	1.92	3.33	0.00	0.00	26.67	23.12
	POST2	10.39	10.95	16.92	27.68	4.06	8.12	8.66	10.59
PA	PRE1	3.78	4.29	4.11	4.52	2.48	2.18	7.25	6.14
	PRE2	1.04	1.10	1.59	0.93	2.73	0.64	3.25	2.34
	POST1	1.71	1.47	0.35	0.18	0.20	0.17	0.88	0.85
	POST2	1.18	0.76	0.48	0.48	0.72	0.73	1.07	0.91
PR	PRE1	456.42	781.58	15.45	8.40	94.40	75.35	241.26	236.71
	PRE2	20.36	42.54	9.04	13.19	4.48	5.32	35.10	38.72
	POST1	30.64	35.99	136.30	216.41	17.73	28.96	14.06	4.70
	POST2	56.36	105.67	6.23	5.82	10.10	12.24	65.08	69.13
SC	PRE1	0.72	0.85	6.70	6.22	14.41	14.66	2.45	3.27
	PRE2	3.83	7.64	4.35	6.74	0.75	0.79	4.57	4.83
	POST1	8.39	8.25	0.51	0.82	0.22	0.31	4.09	4.28
	POST2	33.54	44.46	2.88	3.85	18.04	22.28	21.72	32.99
SH	PRE1	2.66	2.60	19.16	3.47	28.62	18.94	34.15	34.94
	PRE2	3.88	3.86	4.41	6.24	2.98	3.48	7.12	4.15
	POST1	1.59	1.71	2.06	2.84	5.92	4.77	5.84	2.57
	POST2	4.90	3.28	6.11	7.65	5.05	4.90	13.07	6.72
UN	PRE1	0.26	0.26	5.76	5.56	37.25	39.09	11.70	5.51
	PRE2	4.84	7.39	0.56	0.84	4.77	7.62	30.88	47.44
	POST1	14.67	17.05	4.47	3.94	14.90	24.25	27.81	17.82
	POST2	5.61	4.45	9.51	9.96	20.61	22.83	19.57	9.42

Appendix Table 14-B-3. Descriptive statistics for macroinvertebrate export in numbers per day (# day^{-1}) from the Type N Study sites by order and treatment year. OLYM = Olympic block; WIL1 = Willapa 1 block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2.

Order	Treatment Year	OLYM-REF		OLYM-100%		OLYM-FP		OLYM-0%	
		MEAN	SD	MEAN	SD	MEAN	SD	MEAN	SD
COLEOPTERA	PRE1	10.16	6.21	2.80	3.75	5.24	4.09	4.87	5.17
	PRE2	11.69	4.34	6.76	8.45	8.67	3.69	8.74	14.18
	POST1	11.40	6.28	3.63	3.65	13.92	8.61	6.12	9.98
	POST2	16.01	10.12	12.93	22.40	2.79	-	4.44	6.28
DIPTERA	PRE1	129.46	109.98	280.50	306.13	140.20	64.94	529.42	324.43
	PRE2	200.55	139.46	503.76	783.95	278.89	260.85	185.48	112.74
	POST1	410.77	280.41	309.41	322.49	297.88	190.91	252.52	415.14
	POST2	841.32	572.96	336.17	346.24	285.96	-	358.17	183.91
EPHEMEROPTERA	PRE1	45.68	52.01	46.57	82.45	6.20	6.04	23.38	23.26
	PRE2	455.41	323.59	157.44	235.35	225.24	419.85	152.77	245.39
	POST1	478.56	455.67	194.92	170.94	308.94	417.17	36.45	33.32
	POST2	627.56	737.15	101.47	124.71	510.64	-	178.84	60.92
PLECOPTERA	PRE1	50.11	26.08	38.63	42.27	21.84	12.48	72.30	100.37
	PRE2	73.15	27.57	30.86	36.75	32.07	19.42	130.64	97.95
	POST1	123.86	104.32	64.12	65.95	40.15	13.40	54.19	43.94
	POST2	140.19	54.22	69.80	65.06	40.85	-	329.67	346.61
TRICHOPTERA	PRE1	73.70	103.85	16.54	17.85	7.37	1.48	28.46	16.00
	PRE2	30.66	9.87	49.94	78.48	26.96	13.97	30.04	8.71
	POST1	109.58	46.14	44.08	58.86	37.34	19.90	29.25	24.31
	POST2	189.60	273.82	37.38	30.40	68.70	-	73.85	43.06

Order	Treatment Year	WIL1-REF		WIL1-100%		WIL1-FP		WIL1-0%	
		MEAN	SD	MEAN	SD	MEAN	SD	MEAN	SD
COLEOPTERA	PRE1	3.86	3.05	5.33	2.62	8.00	3.56	8.53	4.16
	PRE2	2.03	1.27	5.04	4.08	3.30	1.65	3.03	2.44
	POST1	3.77	2.73	3.16	0.80	2.05	1.86	6.59	3.14
	POST2	8.14	5.12	3.87	3.78	6.63	7.21	15.16	6.39
DIPTERA	PRE1	85.52	110.87	176.99	75.45	52.11	33.73	133.92	10.71
	PRE2	30.58	41.50	103.93	76.20	53.81	28.88	174.94	147.17
	POST1	59.94	62.27	125.17	118.36	47.06	34.61	209.29	215.70
	POST2	112.98	118.13	159.55	133.39	148.42	125.12	424.67	215.99
EPHEMEROPTERA	PRE1	4.20	5.59	35.43	16.00	40.23	62.41	21.50	18.24
	PRE2	12.22	14.52	42.56	33.44	30.36	47.13	35.75	28.05
	POST1	127.40	102.30	236.90	351.61	339.43	300.17	150.09	231.54
	POST2	168.09	103.58	287.63	270.40	398.19	321.16	849.61	665.53
PLECOPTERA	PRE1	33.36	29.59	150.12	110.99	62.47	72.27	50.78	54.48
	PRE2	20.62	16.28	20.07	12.77	47.08	79.96	57.09	43.30
	POST1	14.14	6.75	36.89	54.01	96.28	74.94	140.42	118.77
	POST2	34.68	15.89	44.37	42.69	37.67	26.17	143.54	134.77
TRICHOPTERA	PRE1	4.55	4.40	17.50	1.18	5.91	6.58	18.45	14.32
	PRE2	8.91	10.02	7.56	12.44	13.51	12.37	17.50	9.95
	POST1	6.48	4.53	68.94	105.66	8.60	8.67	27.46	12.77
	POST2	59.57	45.35	19.20	17.36	29.85	18.75	172.84	149.56

Appendix Table 14-B-4. Descriptive statistics for macroinvertebrate export in biomass per day (mg day^{-1}) from the Type N Study sites by order and treatment year. OLYM = Olympic block; WIL1 = Willapa 1 block; REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE1 = pre-harvest year 1; PRE2 = pre-harvest year 2; POST1 = post-harvest year 1; POST2 = post-harvest year 2.

Order	**Treatment Year**	**OLYM-REF**		**OLYM-100%**		**OLYM-FP**		**OLYM-0%**	
		MEAN	**SD**	**MEAN**	**SD**	**MEAN**	**SD**	**MEAN**	**SD**
COLEOPTERA	PRE1	31.93	35.01	34.73	54.28	33.33	38.28	65.49	104.06
	PRE2	23.09	15.43	9.45	10.74	32.94	54.25	81.87	141.38
	POST1	5.48	2.23	8.46	12.85	40.35	68.03	0.50	0.59
	POST2	9.52	8.87	6.29	10.89	4.31	-	8.64	12.21
DIPTERA	PRE1	22.56	28.34	16.44	15.22	7.94	3.28	36.40	23.41
	PRE2	14.45	9.57	55.41	99.11	20.58	12.78	22.70	22.31
	POST1	29.49	24.45	14.68	14.78	13.72	8.19	9.58	15.57
	POST2	43.04	18.25	21.87	19.72	22.00	-	43.91	44.16
EPHEMEROPTERA	PRE1	22.27	29.62	18.68	30.60	0.74	0.86	22.59	20.39
	PRE2	71.45	31.84	16.64	22.50	34.73	58.70	26.97	36.88
	POST1	67.14	58.11	28.83	37.59	34.02	18.95	14.26	11.69
	POST2	31.58	40.34	13.92	18.99	37.78	-	38.42	47.24
PLECOPTERA	PRE1	52.37	49.84	7.29	10.27	104.03	152.54	28.57	27.10
	PRE2	22.48	12.14	12.73	13.86	68.04	84.69	24.24	12.81
	POST1	29.44	45.41	8.87	11.99	10.82	5.59	4.00	3.20
	POST2	20.88	10.46	12.16	18.52	42.65	-	64.00	82.37
TRICHOPTERA	PRE1	44.61	30.43	14.72	16.88	10.53	6.46	14.88	12.20
	PRE2	36.83	13.90	2.75	1.64	47.45	51.73	25.08	23.67
	POST1	25.99	33.14	5.63	6.53	8.19	5.03	10.88	13.75
	POST2	19.26	12.41	6.64	9.19	22.30	-	23.34	22.35

Order	**Treatment Year**	**WIL1-REF**		**WIL1-100%**		**WIL1-FP**		**WIL1-0%**	
		MEAN	**SD**	**MEAN**	**SD**	**MEAN**	**SD**	**MEAN**	**SD**
COLEOPTERA	PRE1	1.37	2.01	3.09	2.96	10.03	5.41	41.36	51.08
	PRE2	6.58	7.66	2.97	4.22	5.19	7.76	7.22	11.18
	POST1	2.71	2.09	3.58	4.56	14.90	24.25	16.20	22.64
	POST2	6.24	4.11	3.80	3.39	7.22	6.37	11.65	4.37
DIPTERA	PRE1	3.64	3.82	13.13	11.19	65.83	106.13	12.86	6.03
	PRE2	1.43	0.87	5.87	5.16	3.62	1.33	22.61	26.70
	POST1	20.24	20.40	10.22	7.55	2.49	1.99	14.32	13.29
	POST2	5.84	5.29	11.34	10.63	11.15	7.95	29.71	11.64
EPHEMEROPTERA	PRE1	0.86	1.09	8.81	4.69	12.06	20.06	4.59	2.84
	PRE2	2.99	3.68	6.05	7.39	1.81	1.97	7.36	9.56
	POST1	22.99	6.40	9.81	10.63	21.71	18.04	12.64	11.11
	POST2	72.98	78.23	32.34	50.18	63.85	63.62	114.95	89.96
PLECOPTERA	PRE1	450.94	769.82	24.45	8.78	79.03	93.16	226.36	246.30
	PRE2	21.71	41.71	8.98	11.24	3.96	3.90	14.57	7.54
	POST1	18.95	36.33	23.20	22.80	22.90	33.79	11.81	2.43
	POST2	50.19	92.22	6.68	6.92	13.72	13.99	49.31	38.17
TRICHOPTERA	PRE1	8.08	8.78	18.42	23.62	5.37	6.22	28.58	29.93
	PRE2	3.09	4.50	18.37	39.43	5.47	8.38	31.48	25.64
	POST1	10.85	12.26	10.48	16.51	1.73	1.68	34.55	26.42
	POST2	24.03	12.31	15.35	19.89	23.95	30.08	65.46	56.24

APPENDIX 14-C. TAXONOMIC COMPOSITION

Appendix Table 14-C-1. Percent taxonomic composition of macroinvertebrate export in numbers per day (# day^{-1}). Each individual specimen is reported at the taxonomic level to which it was identified. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period.

	REF		100%		FP		0%	
Taxon	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**
Annelida	0.14	0.05	0.15		0.12		0.04	0.09
Hirudinea	0.02							
Arthopoda								
Chelicerata								
Arachnida								
Acarina	15.05	7.65	17.33	3.86	21.74	2.49	12.47	2.35
Araneae	0.48	0.52	0.33	0.42	0.34	0.27	0.60	0.39
Opiliones		0.02	0.03		0.10	0.01	0.08	0.03
Pseudoscorpiones	0.06	0.04	0.03			0.01	0.04	
Crustacea								
Amphipoda	6.73	2.91	0.04		0.41	3.03	0.18	0.09
Decapoda		0.01	0.01	0.03	0.05			
Isopoda							0.01	0.01
Uniramia								
Myriapoda								
Chilopoda				0.01				
Diplopoda	0.05	0.02	0.01		0.08		0.05	
Hexapoda								
Collembola	4.19	6.13	6.51	9.71	9.94	3.25	4.20	0.86
Coleoptera	0.46	0.39	0.23	0.34	0.90	0.55	0.66	0.55
Amphizoidae								
Amphizoa	0.05						0.01	
Chrysomelidae	0.04				0.03	0.01		
Curculionidae	0.03	0.09	0.01	0.02	0.03	0.02	0.03	
Elmidae	0.05	0.07	0.03	0.01	0.02		0.01	0.06
Heterlimnius	0.54	0.07	0.15	0.01	0.11		0.12	0.05
Lara avara	0.21	0.07	0.15	0.14	0.15	0.03	0.10	0.01
Narpus			0.02				0.03	
Hydrophilidae	0.21	0.24	0.24	0.13	0.13	0.46	0.15	0.28
Psephenidae		0.02						
Acneus	0.03		0.07	0.23	0.13		0.01	
Diptera		0.16		1.04		0.54		0.42
Blephariceridae				0.02				
Cecidomyiidae	0.08	0.02	15.30	0.17	7.51	0.07	0.17	
Ceratopogonidae	0.61	0.43	0.55	0.21	0.39	0.02	0.13	0.07
Chironomidae	17.05	25.01	22.65	27.68	19.05	17.56	40.83	24.57
Dixidae	0.03			0.02				
Dixa	4.50	4.07	4.60	4.33	3.86	5.00	7.13	5.26
Dixella					0.03		0.09	
Meringodixa	0.03	0.46	0.87	0.85	0.12	0.55	0.28	0.50
Dolichopodidae	0.02		0.01		0.08		0.01	
Empididae							0.01	0.02
Chelifera	0.04		0.09	0.03	0.13		0.04	0.01
Clinocera	0.02	0.02		0.07	0.02	0.02	0.07	
Oreogeton			0.02	0.01	0.04		0.02	
Phoridae			0.03		0.05	0.01		

Appendix Table 14-C-1. (continued)

	REF		100%		FP		0%	
Taxon	PRE	POST	PRE	POST	PRE	POST	PRE	POST
Platypezidae			0.03					
Psychodidae								
Pericoma	0.13	0.33	0.22		0.31		0.14	0.01
Psychoda			0.05					
Sciomyzidae		0.01						
Simuliidae	3.04	1.64	2.66	1.74	0.95	2.22	1.78	2.81
Syrphidae							0.03	
Thaumaleidae	0.02	0.05	0.26	0.23	0.06	0.01	0.02	0.02
Tipulidae	0.02	0.05		0.01	0.02		0.04	0.02
Dicranota	0.07	0.09	0.32	0.13	0.18	0.15	0.27	0.20
Hexatoma	0.04	0.02					0.11	0.05
Limnophila			0.03					
Limonia			0.03	0.01				
Ormosia							0.01	
Pedicia	0.04	0.01	0.03		0.05		0.04	
Rhabdomastix		0.01					0.01	
Tipula	0.08	0.01	0.10		0.03			
Ephemeroptera		0.06		0.07		0.16	0.02	0.04
Ameletidae								
Ameletus	0.07	0.52	0.13	0.54	0.50	0.28	0.06	0.26
Baetidae	4.33	14.15	1.80	13.67	1.03	19.53	0.75	15.34
Baetis	18.87	13.73	8.75	17.10	15.49	28.12	7.79	19.02
Ephemerellidae	0.14	0.24			0.07	0.06	0.06	0.04
Drunella	0.02	0.01	0.03				0.01	
Drunella coloradensis/flavilinea	0.40	0.16	0.02	0.01	0.28	0.23	0.23	0.29
Drunella doddsi		0.01					0.01	
Heptageniidae	0.73	0.46	0.29	0.18	0.28	0.23	0.14	0.10
Cinygma		0.02	0.01			0.06	0.02	
Cinygmula		0.18			0.06	0.07	0.02	0.03
Epeorus	1.95	2.33	0.52	0.34	0.09	1.01	0.15	0.18
Ironodes	0.53	0.84	0.90	0.21	0.91	0.42	0.51	0.36
Rhithrogena	0.02	0.02			0.01			
Leptophlebiidae	0.27	0.28	0.07	0.44	0.02	0.68	0.11	0.45
Paraleptophlebia	0.07	0.20	0.45	0.50	0.02	0.63	0.43	0.32
Hemiptera	0.21	0.12	0.02	0.24	0.05	0.14	0.28	0.36
Hymenoptera	0.03	0.02		0.03	0.05	0.04	0.06	0.03
Lepidoptera	0.02	0.03	0.18	0.07	0.19	0.12	0.05	0.05
Megaloptera/Neuroptera		0.03		0.04			0.04	0.01
Odonata				0.05	0.03			0.01
Orthoptera	0.02		0.02		0.02			
Plecoptera		0.02		0.19		0.03		0.02
Capniidae/Leuctridae	0.85	0.65	1.12	0.74	1.58	0.80	2.09	1.01
Chloroperlidae	0.05	0.02	0.04		0.03		0.01	0.02
Kathroperla perdita	0.11		0.02		0.08		0.10	0.02
Paraperla							0.03	
Sweltsa	0.70	0.50	0.74	0.60	0.73	0.20	0.61	0.41
Leuctridae								
Moselia infuscata		0.10		0.18		0.14		0.09

Appendix Table 14-C-1. (continued)

	REF		100%		FP		0%	
Taxon	PRE	POST	PRE	POST	PRE	POST	PRE	POST
Nemouridae	0.34	0.57	1.52	1.19	0.56	0.47	1.63	3.53
Amphinemura/Malenka			0.19			0.03	0.03	0.01
Nemoura					0.12		0.03	
Nemoura/Soyedina			0.04					
Soyedina	1.49	0.30	1.69	0.45	1.98	0.04	0.22	0.35
Zapada	2.43	1.85	3.06	3.44	3.10	4.93	7.45	7.80
Peltoperlidae	0.33	0.01	0.10		0.15		0.01	0.02
Sierraperla							0.01	
Yoraperla	3.63	3.07	0.57	1.17	1.25	0.67	1.39	1.75
Perlidae			0.01	0.01	0.03	0.03	0.03	0.01
Calineuria californica	0.07	0.01	0.01	0.04	0.03	0.03	0.07	
Doroneuria	0.24	0.01	0.02	0.01	0.07		0.08	0.02
Perlodidae	0.14	0.16	0.12	0.19	0.09	0.12	0.18	0.12
Isoperla	0.08	0.05	0.20	0.24	0.11	0.04	0.47	0.45
Megarcys	0.03	0.08			0.03	0.01	0.03	0.02
Perlinodes aureas						0.01		
Rickera sorpta						0.03		
Setvena							0.03	
Skwala	0.02							0.01
Pteronarcyidae					0.03			
Pteronarcys	0.02				0.03			
Trichoptera	0.11	1.14		1.99	0.37	1.17	0.56	1.89
Brachycentridae								
Micrasema	0.10	0.10	0.04	0.26	0.10	0.09	0.24	0.12
Glossosomatidae		0.04		0.02		0.03		
Agapetus			0.03		0.02			
Anagapetus	0.07	0.12	0.04	0.03	0.02	0.03	0.05	0.01
Glossosoma	0.21	0.24	0.06	0.27	0.14	0.45	0.13	0.09
Hydropsychidae	0.12	0.06	1.93	0.88	0.20	0.10	0.08	0.15
Arctopsyche/Parapsyche	0.33	0.17	0.08	0.26	0.15	0.04	0.33	0.13
Hydropsyche			0.01					
Parapsyche		0.12		0.21		0.03		0.01
Lepidostomatidae								
Lepidostoma	0.16	0.06	0.01	0.24	0.25	0.14	0.22	0.17
Limnephilidae	3.80	0.01	0.04	0.18	0.05	0.02	0.32	0.18
Chyranda centralis							0.07	0.01
Clostoeca disjuncta							0.04	
Cryptochia	0.07	0.03	0.09	0.23	0.18	0.02	0.16	0.01
Ecclisocosmoecus scylla			0.13	0.11			0.03	0.07
Hydatophylax hesperus	0.03							
Psychoglypha	0.15	0.07		0.03	0.11	0.03	0.05	0.12
Philopotamidae			0.24	0.02		0.03		
Dolophilodes	0.09	0.59			0.08	0.05	0.03	0.01
Wormaldia	0.64	0.84	0.54	1.24	0.88	1.22	0.65	5.41
Polycentropodidae	0.03							
Rhyacophilidae								
Rhyacophila	1.15	0.66	0.55	0.43	0.66	0.12	0.92	0.27

Appendix Table 14-C-1. (continued)

	REF		100%		FP		0%	
Taxon	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**
Uenoidae	0.04	4.15	0.22			0.72		
Neophylax	0.21	0.01	0.05	0.02	0.29	0.01	0.62	0.03
Neothremma	0.23	0.08						0.01
Mollusca								
Bivalvia				0.07			0.04	
Gastropoda	0.13	0.04	0.09	0.08	0.28	0.07	0.17	0.04

Appendix Table 14-C-2. Percent taxonomic composition of macroinvertebrate export in biomass per day (mg day^{-1}). Each individual specimen is reported at the taxonomic level to which it was identified. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment; PRE = pre-harvest period; POST = post-harvest period.

	REF		**100%**		**FP**		**0%**	
Taxon	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**
Annelida	0.11	0.26	0.09		0.03		0.09	0.27
Hirudinea	<0.01							
Arthopoda								
Chelicerata								
Arachnida								
Acarina	1.44	2.43	0.68	1.02	0.40	0.74	1.71	0.70
Araneae	0.38	0.95	0.13	0.76	0.14	1.39	2.75	1.81
Opiliones		0.56	0.10		0.22	0.13	1.05	1.48
Pseudoscorpiones	0.06	0.07	0.01			0.02	0.13	
Crustacea								
Amphipoda	13.30	14.63	0.01		0.17	16.30	0.62	0.98
Decapoda		0.92	88.36	3.24	82.70			
Isopoda							0.09	0.03
Uniramia								
Myriapoda								
Chilopoda				0.18				
Diplopoda	1.55	2.17	<0.01		1.54		1.69	
Hexapoda								
Collembola	0.13	0.37	0.07	0.18	0.03	0.37	0.09	0.06
Coleoptera	1.99	2.27	1.12	3.38	1.43	11.65	13.93	4.98
Amphizoidae								
Amphizoa	0.12						0.23	
Chrysomelidae	0.02				0.05	<0.01		
Curculionidae	<0.01	0.02	<0.01	0.07	<0.01	0.01	<0.01	
Elmidae	0.01	0.07	<0.01	<0.01	<0.01		<0.01	0.07
Heterlimnius	0.45	0.18	0.06	0.03	0.02		0.12	0.12
Lara avara	3.48	0.08	0.37	0.09	0.06	2.17	1.92	<0.01
Narpus			<0.01				5.66	
Hydrophilidae	0.78	0.76	0.12	0.51	0.06	0.70	0.53	0.21
Psephenidae		0.04						
Acneus	0.01		<0.01	0.32	0.01		<0.01	
Diptera		0.22		1.24		0.65		1.03
Blephariceridae				<0.01				
Cecidomyiidae	<0.01	0.01	0.18	0.01	0.06	0.01	0.01	
Ceratopogonidae	0.03	0.25	0.05	0.04	0.01	0.01	0.02	0.04
Chironomidae	1.23	6.07	0.80	6.62	0.31	3.65	5.90	6.35
Dixidae	0.01			<0.01				
Dixa	0.71	1.96	0.30	1.62	0.12	2.20	2.04	2.69
Dixella					<0.01		0.03	
Meringodixa	0.01	0.14	0.05	0.33	<0.01	0.21	0.06	0.28
Dolichopodidae	0.03		0.01		0.01		<0.01	
Empididae							<0.01	<0.01
Chelifera	<0.01		0.01	<0.01	<0.01		0.01	<0.01
Clinocera	0.01	0.01		0.02	<0.01	0.01	0.02	
Oreogeton			<0.01	0.01	<0.01		0.02	
Phoridae			<0.01		<0.01	<0.01		
Platypezidae			0.01					

Appendix Table 14-C-2. (continued)

	REF		100%		FP		0%	
Taxon	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**
Psychodidae								
Pericoma	0.01	0.10	0.01		0.01		0.04	<0.01
Psychoda			0.01					
Sciomyzidae		0.03						
Simuliidae	0.39	1.41	0.12	1.08	0.03	0.93	0.35	1.13
Syrphidae							0.09	
Thaumaleidae	<0.01	0.01	0.02	0.11	<0.01	0.02	<0.01	0.01
Tipulidae	0.03	1.13		<0.01	<0.01		0.25	0.06
Dicranota	0.05	1.83	0.06	0.09	0.03	0.15	0.22	0.21
Hexatoma	0.05	0.02					3.09	1.74
Limnophila			0.02					
Limonia			<0.01	0.03				
Ormosia							0.08	
Pedicia	0.01	0.03	<0.01		0.52		0.03	
Rhabdomastix		<0.01					<0.01	
Tipula	1.89	0.71	1.34		0.57			
Ephemeroptera		0.40		0.04		0.54	0.05	0.11
Ameletidae								
Ameletus	0.18	1.98	0.01	2.69	0.06	1.21	0.07	1.95
Baetidae	0.04	0.45	0.01	0.36	<0.01	0.53	0.01	0.46
Baetis	3.77	13.31	0.60	10.74	0.60	18.81	2.87	18.13
Ephemerellidae	0.10	0.77			<0.01	0.03	0.13	0.20
Drunella	<0.01	0.01	<0.01				0.01	
Drunella coloradensis/flavilinea	0.85	0.85	0.05	0.14	0.04	1.63	0.51	3.05
Drunella doddsi		<0.01					<0.01	
Heptageniidae	1.23	1.21	0.09	0.49	0.03	1.28	0.29	0.26
Cinygma		0.37	0.02			0.40	0.06	
Cinygmula		0.46			0.01	0.22	0.06	0.13
Epeorus	0.83	2.07	0.11	0.12	<0.01	2.16	0.18	0.27
Ironodes	2.37	6.96	0.70	2.30	0.18	2.09	2.92	4.59
Rhithrogena	0.01	0.43			0.02			
Leptophlebiidae	0.03	0.22	<0.01	0.27	<0.01	0.38	0.03	0.24
Paraleptophlebia	0.02	0.19	0.06	0.52	<0.01	0.74	0.25	0.30
Hemiptera	0.11	0.15	<0.01	0.12	0.01	0.24	0.72	0.43
Hymenoptera	0.70	0.15		2.44	0.38	0.13	2.24	0.31
Lepidoptera	0.01	0.01	0.65	3.32	0.88	6.05	1.35	1.53
Megaloptera/Neuroptera		0.17		0.27			0.48	0.11
Odonata				38.11	2.64			4.55
Orthoptera	0.32		<0.01		0.19			
Plecoptera		0.01		0.02		0.55		<0.01
Capniidae/Leuctridae	0.28	0.63	0.16	0.74	0.30	1.12	1.44	1.65
Chloroperlidae	0.03	0.02	0.03		0.01		0.01	<0.01
Kathroperla perdita	0.15		<0.01		0.12		0.09	0.14
Paraperla							0.72	
Sweltsa	0.59	1.31	0.36	1.84	0.20	0.55	0.73	0.97
Leuctridae								
Moselia infuscata		0.05		0.07		0.07		0.02

Appendix Table 14-C-2. (continued)

	REF		100%		FP		0%	
Taxon	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**
Nemouridae	0.03	0.05	0.07	0.06	0.01	0.04	0.16	0.26
Amphinemura/Malenka			0.01			0.03	0.01	0.02
Nemoura					<0.01		0.01	
Nemoura/Soyedina			<0.01					
Soyedina	0.15	0.07	0.08	0.04	0.02	0.01	0.02	0.04
Zapada	0.40	0.69	0.24	1.00	0.13	1.59	1.13	2.67
Peltoperlidae	0.01	0.01	0.06		0.01		<0.01	0.41
Sierraperla							0.05	
Yoraperla	2.33	4.09	0.23	1.11	0.09	0.56	0.93	1.52
Perlidae			<0.01	0.04	0.35	2.34	0.08	0.01
Calineuria californica	2.23	0.18	0.11	3.68	0.88	4.70	8.39	
Doroneuria	42.19	8.62	0.23	0.29	0.80		13.31	4.77
Perlodidae	0.10	0.34	0.01	0.15	0.03	0.34	1.00	0.17
Isoperla	0.04	1.66	0.08	0.38	0.06	0.08	2.96	4.56
Megarcys	2.40	0.43			0.10	0.05	0.18	0.17
Perlinodes aureas						0.40		
Rickera sorpta						0.31		
Setvena							0.26	
Skwala	0.12							0.33
Pteronarcyidae					0.01			
Pteronarcys	0.05				1.68			
Trichoptera	0.02	0.15		0.11	<0.01	0.11	0.03	0.16
Brachycentridae								
Micrasema	0.05	0.14	<0.01	0.05	0.01	0.04	0.11	0.01
Glossosomatidae		0.03		<0.01		<0.01		
Agapetus			<0.01		<0.01			
Anagapetus	0.01	0.29	<0.01	0.01	0.01	0.01	0.05	0.01
Glossosoma	0.09	0.16	0.01	0.31	0.01	1.08	0.08	0.23
Hydropsychidae	0.71	0.03	0.02	0.07	0.03	0.01	0.27	0.04
Arctopsyche/Parapsyche	3.89	0.26	0.95	0.26	0.22	0.06	0.72	0.47
Hydropsyche			<0.01					
Parapsyche		1.76		2.63		2.25		0.01
Lepidostomatidae								
Lepidostoma	0.25	0.36	0.01	0.79	0.05	0.72	0.43	0.53
Limnephilidae	0.08	0.02	<0.01	0.17	0.04	0.04	0.05	0.05
Chyranda centralis							1.40	0.68
Clostoeca disjuncta							0.09	
Cryptochia	0.02	0.01	0.02	0.05	0.06	0.02	0.06	0.01
Ecclisocosmoecus scylla			0.11	0.13			0.06	0.15
Hydatophylax hesperus	0.53							
Psychoglypha	0.46	2.23		0.87	0.40	1.50	3.55	6.85
Philopotamidae			0.01	<0.01		<0.01		
Dolophilodes	0.05	1.22			0.01	0.23	0.07	0.01
Wormaldia	0.19	1.07	0.09	1.16	0.08	2.34	0.57	6.37
Polycentropodidae	0.06							
Rhyacophilidae								
Rhyacophila	2.97	3.99	0.57	0.87	0.40	1.05	3.87	5.52

Appendix Table 14-C-2. (continued)

	REF		100%		FP		0%	
Taxon	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**	**PRE**	**POST**
Uenoidae	<0.01	0.14	<0.01			0.02		
Neophylax	0.19	<0.01	0.02	0.07	0.08	<0.01	1.27	0.29
Neothremma	0.09	0.10						0.02
Mollusca								
Bivalvia				0.03			<0.01	
Gastropoda	0.38	0.01	0.04	0.06	0.21	0.08	0.72	0.04

CHAPTER 15 - STREAM-ASSOCIATED AMPHIBIANS

Aimee McIntyre, Marc Hayes, Eric Lund, Jay Jones, Frithiof T. Waterstrat, Andrew J. Kroll, Reed Ojala-Barbour, and Timothy Quinn

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15-1. ABSTRACT

Stream-associated amphibians, frequently the dominant vertebrates in and along non-fish-bearing headwater streams, have experienced declines in local abundance and range contractions as a result of disease, competition with introduced species, and habitat loss and degradation. We compared headwater stream-associated amphibian (Coastal Tailed Frog [Ascaphus truei], and torrent [Rhyacotriton] and giant [Dicamptodon] salamanders) occupancy, density and body condition before and after clearcut timber harvest of small headwater basins in four treatments. Treatments included an unharvested reference and three riparian buffer treatments, including (1) the current Washington State Forest Practices riparian leave-tree buffer (FP treatment); (2) a more extensive buffer (100% treatment); and (3) no buffer (0% treatment). To account for the potential variation of detection probabilities in our amphibian density estimates, we used rubble-rouse sampling and contemporary statistical models. We conducted three separate analyses on amphibian density (1) in the most-downstream Type Np reach (i.e., 200 m directly upstream from the fish end-point, the F/N break); (2) stream network-wide, including stream reaches obstructed with heavy slash loading; and (3) stream network-wide excluding stream reaches obstructed with heavy slash loading.

All stream-associated amphibians continued to occupy clearcut headwater streams, regardless of treatment, throughout the two-year post-harvest study period. We were most confident in the results obtained from our stream network-wide density estimates, based on our concern that animals may have moved in and out of the downstream-most Type Np reach differentially by treatment, adding unwanted variability and potentially influencing study results. Treatment effects in the two-year post-harvest study period and based on stream network-wide linear density were variable among genera and, for tailed frogs, among life stages. We found support for a negative effect of buffer treatments on the density of giant salamanders in the FP treatment. Giant salamander density (all life stages combined) decreased between pre- and post-harvest sampling periods relative to the change in the reference and other buffer treatments (82% decrease relative to the reference when excluding obstructed reaches), and showed no change in either the 100% or 0% treatment. In contrast, larval Coastal Tailed Frog density increased in the 100% and FP treatments relative to the reference and 0% treatment; these increases were 4.1 and 8.2 times greater, respectively, than the change in the reference when excluding obstructed reaches. Post-metamorphic Coastal Tailed Frog also showed a post-harvest increase in density, but only in the 0% treatment; that increase was 5.5 times greater than in the reference when excluding obstructed reaches. Torrent salamanders (all life stages combined) failed to show clear evidence of a response between treatments, with the exception that when we included obstructed reaches in the analysis, we estimated the post-harvest change in torrent salamander density in the 0% treatment to be 3.3 times greater than the change in the reference. This was the only result obtained from analyses of stream network-wide density including obstructed reaches that differed statistically from results obtained when we did not include obstructed reaches in the analyses. We did not detect a treatment effect on body condition for any species. We conclude that, with the exception of the negative response to harvest for giant salamanders in the FP treatment, the current Forest Practice’s buffer effectively maintained stream-associated amphibian taxa in the two years post-harvest.

15-2. INTRODUCTION

Pacific Northwest headwater streams support a variety of species not found in large, higher-order river systems (Meyer *et al.* 2007). Fish densities decline in smaller streams, so these low-order headwaters offer refuge from fish predators common in higher-order streams (Richardson and Danehy 2007) and stream-associated amphibians often replace fish as the dominant vertebrate predators in and along headwater streams (Burton and Likens 1975; Bury *et al.* 1991). For example, Murphy and Hall (1981) found that the Coastal Giant Salamander (*Dicamptodon tenebrosus*) was the dominant vertebrate in headwater streams in the Oregon Cascades, making up as much as 99% of instream predator biomass in some sites. Moreover, for streams in the Pacific Northwest, aquatic amphibians are 10 times more abundant with four times the biomass than has been reported for salmonid fishes (Bury *et al.* 1991). Some contend that stream-associated amphibian species are both specifically adapted to the physical conditions of headwater streams (Kiffney *et al.* 2003) and found in abundance in the absence fish (Kroll *et al.* 2008).

The structure of headwater streams in western Washington can be quite variable due to the interaction between local disturbances (e.g., weather, tree mortality, landslides) and stream size, location and topographic setting. Biota associated with these low-order streams may be particularly predisposed to large variations in population size or local extirpation (Fagan 2002). Indeed, providing a conceptual model for amphibian response, Bury and Corn (1988) indicated either extirpation or a significant decline followed by slow recovery were the two likely outcomes of timber harvest for many stream-associated species. If amphibian populations are extirpated in a headwater basin, opportunities for recolonization from adjacent headwater streams may be restricted by larger downstream reaches that form barriers to dispersal (Lowe and Bolger 2002; Richardson and Danehy 2007) or gaps in overhead canopy (Cecala *et al.* 2014).

Amphibians are frequently considered among the vertebrate groups most susceptible to environmental modification, and thus preferred for monitoring environmental conditions (Wake 1991). Many amphibian species have experienced declines in local abundance 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). Stream-associated amphibians are considered important indicator species because of their limited dispersal abilities, dual life histories, and explicit microhabitat and physiological requirements (Welsh and Ollivier 1998; Lawler *et al.* 2010).

Many studies have suggested that stream-associated amphibians are particularly sensitive to forest practices. In the early 20th century, Noble and Putnam (1931) noted that the removal of trees via harvest and fire resulted in the disappearance of Coastal Tailed Frogs (*Ascaphus truei*), allegedly because of increased stream temperatures due to a loss of shade associated with overhead canopy. One concern with early publications, however, is that they are observational, with no supporting data presented. Additionally, forest practices used in the past are not equivalent to current practices, and retrospective studies may represent different impacts and recovery patterns. Studies that are more recent have compared amphibian occurrence or relative abundance between unharvested and harvested stands. For example, Corn and Bury (1989) reported a higher frequency of occurrence of Coastal Tailed Frogs in unlogged watersheds and fewer giant salamanders in logged streams. In another study, Steele and colleagues (2003) encountered the lowest number of Cascade Torrent Salamander (*Rhyacotriton cascadae*) captures in zero to 24-year old forests compared to forests that were 25 to 60 years old. However, one concern with these more recent studies is that they were retrospective, with historical distributions of study species unknown, so the role of forest practices in creating or maintaining these observed differences is weakly inferred. More recently, experimental studies have included direct comparisons of stream-associated amphibian relative abundance before and after harvest. For example, Jackson and colleagues (2007) found that giant salamander and Coastal Tailed Frog populations declined in the period immediately following timber harvest. However, a concern with some of these most recent experimental studies is that they do not account for the probability of detection; if detection is confounded with treatment then the observed response could be an artifact of sampling and not a true reflection of abundance (e.g., MacKenzie and Kendall 2002; Tyre *et al.* 2003).

Conversely, others studies have found a lack of correlation between forestry practices and amphibian abundance. For example, in a retrospective study, Murphy and Hall (1981) observed no overall difference in Coastal Giant Salamander density between clearcut and old-growth sites. In an experimental study, O’Connell and colleagues (2000) found no differences in the densities of Coastal Tailed Frog larvae or Coastal Giant Salamander larvae between clearcut headwater basins with variable width buffers of two size classes averaging 15.4 and 30.5 m and forested reference headwater basins. However, the former study is limited by the fact that it was retrospective in nature, lacking pre-harvest occupancy and density data for amphibians, and in the latter, the proportion of the study stream reach sampled was extremely limited (5% of total) and may not have been representative. Detection was not considered in either study.

Interpretational complexities arising from ambiguous study results and weak inference led us to design an experiment to evaluate the effectiveness of the Forest Practices riparian buffer prescription in mitigating potential forest management impacts to stream-associated amphibians in non-fish-bearing streams. We evaluated the response of stream-associated amphibians to clearcut timber harvest in headwater basins located throughout western Washington. We selected study sites on hard rock lithologies, which are more likely to support the stream-associated amphibian species selected for this study than basins underlain by more erosion prone, softer rock lithologies (Wilkins and Peterson 2000), and verified amphibian presence prior to including sites in the study. We collected data to establish baseline amphibian occupancy and density prior to timber harvest and allow direct comparison to post-harvest estimates.

A final important consideration regarding the interpretation of the response of stream-associated amphibians and similar taxa to forest management relates to detection probabilities (Bailey *et al.* 2004; Mazerolle *et al.* 2007). The amphibians selected for inclusion in this study are cryptic and frequently concealed, and detection probabilities for these species are variable and depend on ecological processes (e.g., life-history characteristics and environmental conditions; Pollock *et al.* 2002; Schmidt 2004; McIntyre *et al.* 2012). Since our study covers a five-year period, we were concerned with the influence of changing stream conditions on amphibian detection and, in particular, the potential for confounding between amphibian detection and treatment. Our study design incorporated methods to deal explicitly with these issues.

During Forests and Fish (USFWS 1999) negotiations leading to the development of Washington State's Forest Practices rules, stakeholders agreed to protect six stream-associated amphibians in Type N Waters. These species were selected based on their close association with headwater streams and the presumption that they were the most susceptible and least resilient to the impacts of forest management compared with other species of amphibians in forestlands in Washington State. In 2001, one of the selected species (tailed frog [*Ascaphus spp.*]) was partitioned into two species, raising the total number of covered species to seven, including Coastal and Rocky Mountain Tailed Frogs (*A. truei* and *A. montanus*); Olympic, Columbia and Cascade Torrent Salamanders (*R. olympicus*, *R. kezeri*, and *R. cascadae*); and Dunn's (*Plethodon dunni*) and Van Dyke's (*P. vandykei*) Salamanders (hereafter FP-designated amphibians).

At the time of Forest Practices negotiations, almost no published studies addressed the efficacy of riparian buffers for Type N Waters or provided clear or compelling guidance addressing conservation needs of stream-associated amphibians. Moreover, the few studies available were retrospective (Bisson *et al.* 2002; Raphael *et al.* 2002) or lacked the statistical power needed to confidently interpret how forest practices affected these species or their habitats (O'Connell *et al.* 2000; Jackson *et al.* 2003).

The stream-associated amphibians selected for inclusion in this study included four of the seven FP-designated amphibians: Coastal Tailed Frog and the three species of torrent salamanders. We did not address Dunn's and Van Dyke's Salamanders in this study because we could not effectively sample them with our instream sampling methods due to the lack of an aquatic larval stage. To maximize our efforts, we did not include Rocky Mountain Tailed Frog because its distribution is limited to the southeast corner of the state and it does not co-occur with any other FP-designated species (see **Figure 2-1**). Though Coastal and Cope's (*D. copei*) Giant Salamanders are not FP-designated amphibians, we included them in our study for two reasons. First, Cope's Giant Salamander is one of only two instream-breeding amphibian species distributed throughout our entire study area and for this reason was selected for inclusion in the amphibian genetics part of this study (along with Coastal Tailed Frog, see Spear and colleagues [2011] for results from the pre-harvest period). Second, since Cope's and Coastal Giant Salamanders are extremely difficult to differentiate in the field (Nussbaum 1976; Good 1989; Foster and Olson 2014) and hybridization is known to occur (Spear *et al.* 2011), we had to include Coastal Giant Salamander by default. Though life history characteristics for the two species are largely similar, one notable difference includes the propensity for Cope's Giant Salamanders to remain instream, reproducing as neotenic adults (sexually mature adults with juvenile characteristics), whereas Coastal Giant Salamanders are much more likely to transform into terrestrial adults which can disperse overland (Nussbaum 1970, 1976).

In Washington State, the Coastal Tailed Frog is found throughout the Cascades, Olympics and Coast Ranges. The species is highly specialized for life in cool, fast-flowing waters such as those found in headwater streams (Metter 1964; Nussbaum *et al.* 1983; Bury 1988; Jones *et al.* 2005). The species is commonly encountered in streams composed of coarse inorganic substrates (Hawkins *et al.* 1988; Diller and Wallace 1999; Dupuis and Steventon 1999; Adams and Bury 2002). Larvae exhibit thermo-regulatory behavior, routinely avoiding stream temperatures greater than 22°C (de Vlaming and Bury 1970) and adults show a seasonal preference for stream temperatures of approximately 11°C for oviposition sites (Karraker *et al.* 2006).

In Washington State, three species of torrent salamanders exist, all of which are covered under Forest Practices rules. The distributions of these species do not overlap. The Olympic Torrent Salamander occurs exclusively on the Olympic Peninsula, extending south to near the Chehalis River. The Columbia Torrent Salamander occurs in the Coast Range, south of the Chehalis River. The Cascade Torrent Salamander occurs on the west slope of the Cascade Range to south of the Cowlitz River (Jones *et al.* 2005). Torrent salamanders require cool water (Stebbins and Lowe 1951; Jones *et al.* 2005), are usually found in splash zones, riffles and seeps (Nussbaum and Tait 1977; Nussbaum *et al.* 1983; Cudmore and Bury 2014), and appear to be associated with the uppermost extent of the headwater system (Olson and Weaver 2007). Pollett and colleagues (2010) found that Cascade Torrent Salamanders were nearly absent when stream temperatures warmed to greater than 14°C for more than 35 consecutive hours.

Cope's Giant Salamanders are found throughout the Cascades, Olympics and Coast Ranges, whereas Coastal Giant Salamanders are found throughout the Coast and Cascade Ranges, exclusive of the Olympic Peninsula (Jones *et al.* 2005; Foster and Olson 2014). Larval and neotenic individuals of both giant salamanders are restricted to aquatic environments, occurring in streams and standing water in coniferous and mixed forests (Jones *et al.* 2005). Bury (2008) found that while Coastal Giant Salamanders have a slightly higher critical thermal maxima than torrent salamanders, they still appear to be sensitive to elevated stream temperatures (e.g., near 30°C).

Some of the habitat qualities upon which FP-designated and other stream-associated amphibians appear to rely, such as instream substrate composition and stream temperature, are affected by timber harvest and associated activities (Grizzel and Wolff 1998; Johnson and Jones 2000; Moore *et al.* 2005; Araujo *et al.* 2014). Stream-associated amphibians have been negatively associated with fine substrates (Welsh 1993; Welsh and Ollivier 1998; Dupuis and Steventon 1999; Wahbe and Bunnell 2003) which may increase in streams adjacent to clearcuts (Corn and Bury 1989; Dupuis and Steventon 1999; Jackson *et al.* 2001). Clearcut timber harvest has been shown to increase stream temperatures in some streams (Moore *et al.* 2005; Janisch *et al.* 2012).

15-3. OBJECTIVES

Our primary objective was to describe changes in stream-associated amphibian (including both FP-designated amphibians and two species of giant salamander) demographics in headwater streams during the first two years following clearcut harvest. We compared the response of amphibians in alternative riparian buffer treatments that differed in the length of a two-sided 50-ft (15.2-m) riparian buffer maintained in the Riparian Management Zone (RMZ). Treatments included a reference (REF) and three riparian buffer treatments: full buffer (100%), ≥50% buffered (buffered under current Forest Practices rules [FP]), and no buffer (0%). Specifically, we quantified changes in amphibian occupancy, density, and body condition across 17 Type N basins described in Chapter 2 − *Study Design*.

15-4. METHODS

We focused this study on six stream-associated species: Coastal Tailed Frog, three species of torrent salamanders (Cascade, Columbia and Olympic) and two species of giant salamanders (Coastal and Cope's; **Figure 15-1**), hereafter, focal amphibians. Since distinguishing between Coastal and Cope's Giant Salamander larvae is difficult in the field (Nussbaum 1970) and hybridization exists in some locations (Steele *et al.* 2009) we combined counts for analysis. We 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 our study sites. This assumes that ecology and response to disturbance among species is similar, an assumption based on the fact that the species were only relatively recently identified as distinct (Good and Wake 1992) and habitats are described similarly across all three species (Jones *et al.* 2005). Additionally, we recorded observations of all other species of amphibians.

We used two amphibian sampling methods: light-touch and rubble-rouse. In our study proposal (Hayes *et al.* 2005), we proposed using light-touch to determine focal amphibian occupancy and rubble-rouse to determine density, consistent with the traditional use of these methods. We focused our rubble-rouse effort for estimating density solely on the downstream-most Type Np reach (i.e., 200 m directly upstream from the F/N break, hereafter, lower Np reach). Coastal Tailed Frog larvae are typically encountered in greater densities in downstream reaches, and post-metamorphs move annually down- and upstream for the breeding and non-breeding seasons (Hayes *et al.* 2006). Additionally, potential effects of timber harvest on habitat may accumulate in the downstream reaches, where the cumulative upstream harvested area is at its greatest.

After we initiated amphibian sampling in 2006, an alternative means of estimating species abundance or density was developed and tested. *N*-mixture models (Royle 2004) are an alternative to traditional mark-recapture and removal sampling and have been utilized to estimate amphibian abundance and density in several studies (McKenny *et al.* 2006; Chelgren *et al.* 2011; Price *et al.* 2011; McIntyre *et al.* 2012). 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; Guillera-Arroita *et al.* 2014). We incorporated a multi-pass light-touch sampling methodology at a subset of plots in 2008, 2009 and 2010, which allowed us to adjust our amphibian counts for detection probability in order to estimate stream network-wide amphibian abundance. This new method was of special interest. It allowed us to adjust our existing count data going back to our first year of sampling in 2006. It addressed our lingering concern at the possibility that amphibian detection was confounded with treatment (e.g., amphibian detection may decline in sites harvested with a more severe treatment). It required less handling time of individuals than mark-recapture methods and allowed us to sample a greater proportion of each study site (greater sample sites frequently results in smaller variances in derived population estimates and greater statistical power to detect population trends [Dodd and Dorazio 2004]). Finally, it allowed us to use light-touch sampling methodology, which is comparatively less invasive than rubble-rouse (Quinn *et al.* 2007), a feature especially important for our long-term study to ensure that changes in abundance were due to buffer treatment and not sampling. Though we incorporated this method into our study in 2008, we continued to sample the lower Np reach utilizing rubble-rouse; both for consistency across years, as well as to be able to compare the responses estimated using both sampling methodologies.

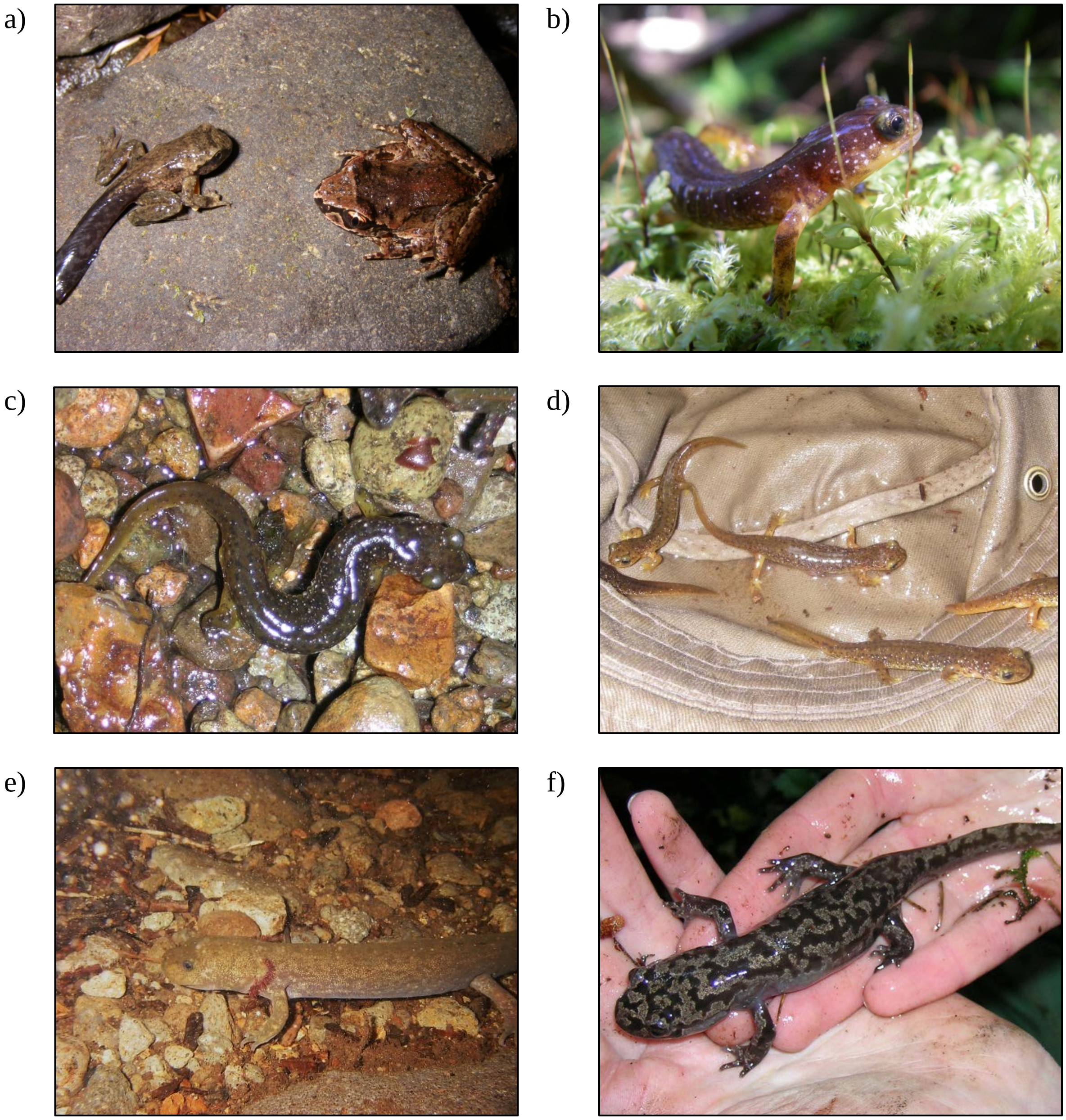

Figure 15-1. Focal amphibian species included in the Type N Study: (a) metamorphosing (left) and post-metamorphic Coastal Tailed Frog; (b) post-metamorphic Olympic Torrent Salamander; (c) post-metamorphic Columbia Torrent Salamander; (d) post-metamorphic Cascade Torrent Salamander; (e) neotenic giant salamander; and (f) post-metamorphic giant salamander. Photo credits: Eric M. Lund (a, b, e), Jennifer A. Dhundale (c, f), Frithiof T. Waterstrat (d).

15-4.1. LIGHT-TOUCH SAMPLING

The light-touch method (Lowe and Bolger 2002) has been commonly used to survey headwater amphibian populations in the Pacific Northwest (e.g., Welsh *et al.* 1997; Lowe and Bolger 2002; Steele *et al.* 2003; Russell *et al.* 2004; Quinn *et al.* 2007). The use of light-touch sampling has the advantage of allowing us to repeat sample the same sites through time while minimizing disturbance (O'Donnell *et al.* 2007; Quinn *et al.* 2007). It also allows us to cover a larger area than would have been possible with other common amphibian sampling methods (e.g., rubble-rouse; Quinn *et al.* 2007). Finally, it allows us to adjust count data by detection probability, adjusted for covariates, to estimate stream network-wide abundance. With this method, 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. We returned substrates to their original position following sampling and took special care to preserve in-channel structures such as steps.

We conducted light-touch sampling in all study years, 2006−2010. In two pre- (2006 and 2007) and two post-harvest years (2009 and 2010) we conducted systematic light-touch surveys along all Type N basin tributaries starting at the F/N break and continuing up to each channel head, with one exception[1]. Sampling included the lower Np reach (i.e., 200 m upstream from the F/N break). We conducted additional light-touch sampling systematically over all tributaries for a minimum of half of the remaining stream length for shorter streams (mainstem channel <800 m long) and a quarter of the remaining length for longer streams (mainstem channel >800 m long). The systematic sampling was typically conducted in two consecutive 10-m stream intervals (20 m total), spaced 20 m apart for shorter streams and 60 m apart for longer streams (e.g., **Figure 15-2**). Our targeted systematic sample reaches (hereafter, study reaches) were consistent in 2006, 2007, 2009 and 2010; however, additional stream length was sampled when time permitted. As a result, the proportion of the stream length sampled by site and year was often more than the minimum target length (**Table 15-1**). Due to a large storm that resulted in substantial windthrow at some sites (see Chapter 4 − *Unanticipated Disturbance Events*) we conducted a third, unplanned year of pre-harvest sampling in 2008; however, due to time and budget constraints, systematic light-touch sampling was restricted to the 200 m upstream from the F/N break.

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[1] We did not sample the WIL2-REF1 site in 2009 because this site was originally intended to be a buffer treatment site and the harvest had not been applied. When harvest was not applied, the site was included as a second reference in the Willapa 2 block (see Chapter 2 – *Study Design*).

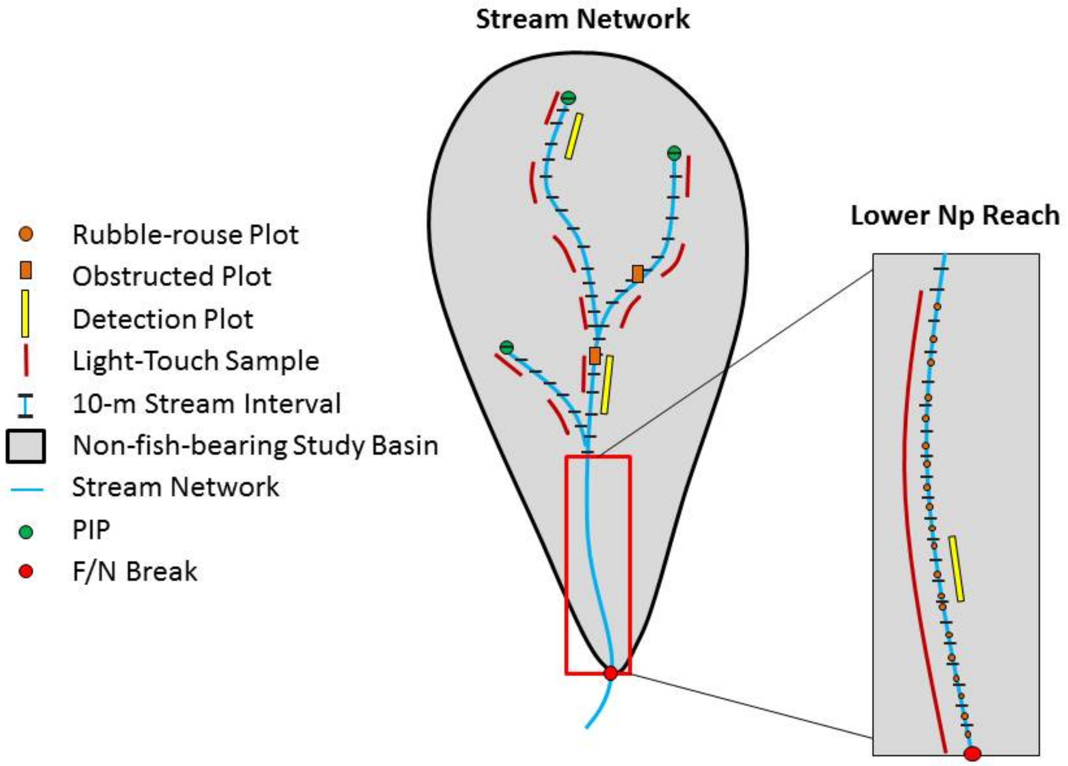


Figure 15-2. Schematic of plot layout for amphibian sampling. Includes light-touch sampling of 10-m stream intervals, rubble-rouse sampling of 1-m plots located throughout the lower Np reach (i.e., typically 200 m upstream from the F/N break), 3-m rubble-rouse plots located in stream reaches obstructed by in-channel wood, primarily in the form of logging slash and blowdown, and 30-m detection plots sampled with light-touch.

We sampled all study reaches, even if there was no evidence of surface water at the time of sampling. When we encountered amphibians in dry reaches, we recorded them as such. When we encountered obstructed stream reaches (e.g., downed tree, logging slash) that prevented access to a portion of the stream or made it impossible to turn or see under cover objects, we skipped that portion of stream and continued sampling above the obstructed reach. We recorded the location and length of each obstructed reach.

Table 15-1. Target minimum proportion of stream length to be sampled during systematic amphibian light-touch sampling and corresponding actual stream length sampled by site for two pre-harvest years (2006 and 2007) and two post-harvest years (2009 and 2010).

Block	Treatment	Target Minimum Proportion of Stream[1]	Actual Proportion of Stream Length Sampled			
			2006	2007	2009	2010
OLYM	REF	0.25	0.96	1.00	0.52	0.45
	100%	0.25	0.93	1.00	0.66	0.75
	FP	0.25	0.86	1.00	0.69	0.58
	0%	0.50	1.00	1.00	0.98	1.00
WIL1	REF	0.50	1.00	1.00	0.79	0.83
	100%	0.50	1.00	1.00	0.82	0.85
	FP	0.50	0.86	1.00	0.72	0.72
	0%	0.25	0.99	1.00	0.66	0.69
WIL2	REF1	0.50	0.49	1.00	–[2]	0.74
	REF2	0.50	0.99	1.00	0.75	0.81
	100%	0.25	0.55	1.00	0.52	0.54
	0%	0.25	0.93	0.94	0.63	0.56
WIL3	REF	0.25	0.26	0.95	0.43	0.45
	100%	0.25	0.44	0.99	0.65	0.67
CASC	REF	0.25	0.29	0.89	0.78	0.66
	FP	0.50	1.00	1.00	0.77	0.67
	0%	0.50	0.91	0.86	0.74	0.61

[1] Target minimum proportion for the remainder of stream above the 200 m reach upstream of the F/N break.
[2] We did not sample the WIL2-REF1 in 2009 (see Chapter 2 − *Study Design*).

15-4.1.1. Sampling of plots for detection estimation

We estimated detection probabilities for our light-touch sampling method along a portion of some study reaches utilizing an *N*-mixture model (Royle 2004), which accounted for spatial and annual variation in detection (McIntyre and colleagues [2012] verified the utility of *N*-mixture models for use with stream-associated amphibians). We estimated detection probabilities for 30-m long plots (hereafter, detection plots) distributed throughout each of the study sites during three sample years (2008−2010). We sampled 37, 65, and 56 detection plots in 2008, 2009, and 2010, respectively (**Table 15-2**). We chose a 30-m plot length to maximize the likelihood of detecting the focal amphibian genera. Previous research using the same sampling method demonstrated that one of our three focal amphibian genera was detected with high confidence when approximately 15 m of stream length was surveyed (Quinn *et al.* 2007).

Table 15-2. The number of 30-m detection plots sampled by buffer type and year. All plots in 2008 reflected reference conditions since buffer treatments had not yet been applied.

Buffer Type	**Number of Plots**		
	2008	**2009**	**2010**
Reference (REF)	37	20	21
Buffered (BUF)	-	27	19
Unbuffered (UNB)	-	18	16

For all years 2008−2010, we distributed detection plots across stream orders (first-, second- and third-order) in all study sites and in buffered/unbuffered reaches for FP treatment sites in the post-harvest years (e.g., **Figure 15-2**). Plot locations in 2008 were not confined to the lower Np reach sampled with systematic light-touch. In 2008, we placed five detection plots in stream reaches that were completely dry at the time of sampling. We randomly located the remaining plots along perennially flowing stream reaches with no more than 5 m of consecutively dry streambed. We randomly established new plot locations each year. We established three, 10-m transects along the stream channel for each plot, with survey flags at 10, 20, and 30 m. If we could not survey part of a detection plot due to an obstructed reach that prohibited access or visibility, we delineated a sampling break, and marked and measured the breaks to ensure that repeat samples were consistent. In 2008, we added a distance upstream that equaled the total sampling breaks encountered so that sampled length was 30 m. Of the 42 plots sampled in 2008, only five sites required breaks in sampling due to an obstructed reach, with total break length for a plot ranging from 1 to 15 m (mean = 7 m). Further, in 2008, the length of two plots was limited to the full extent of the third-order stream reaches available to sample (i.e., 18 and 24 m). In 2009 and 2010, we did not add additional length to account for obstructed portions; however, sampled plots were required to be greater than 15 m. We surveyed each detection plot in conjunction with our systematic light-touch surveys on three separate occasions and conducted surveys between one and seven days apart (**Table 15-3**); however, the majority of repeat samples were conducted on consecutive days. Waiting at least one day between surveys minimizes the chance of introducing bias due to a behavioral response to sampling. One sampler conducted each survey and to reduce bias repeat surveys were conducted by different samplers. We included the animals detected during our first visit to detection plots in our stream network-wide light-touch amphibian summaries. We recorded stream temperature within each detection plot at the time of each sample (three samples per plot) using a waterproof digital thermometer (accuracy ± 1°C) with the intent of including temperature in our detection models for amphibians (see 15-4.5.3. Calculating Density).

Table 15-3. The survey date range, average days between surveys, and range of days between surveys of 30-m detection plots (2008−2010).

Year	Date Range	Range of Days Between Surveys
2008	7 July−21 August	1−5
2009	6 July−6 October	1−7
2010	1 July−29 September	1−5

15-4.2. RUBBLE-ROUSE SAMPLING

Rubble-rouse sampling is a common technique for surveying stream-associated amphibians in the Pacific Northwest (Bury and Corn 1991; Bury *et al.* 1991; Welsh and Lind 1996; Wilkins and Peterson 2000; Quinn *et al.* 2007). We used standard rubble-rouse procedures (Bury and Corn 1991) and assumed a detection probability of 1. We blocked off a section of the stream with downstream and upstream nets to prevent animals from escaping or entering plots. 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, to a depth of 30 cm or until bedrock was reached (**Figure 15-3a**). We sifted the remaining unconsolidated fine substrates and carefully removed the nets and examined them for animals. We replaced coarse substrates in the channel (**Figure 15-3b**) and returned animals to the stream. We conducted rubble-rouse sampling only after light-touch sampling had been completed.

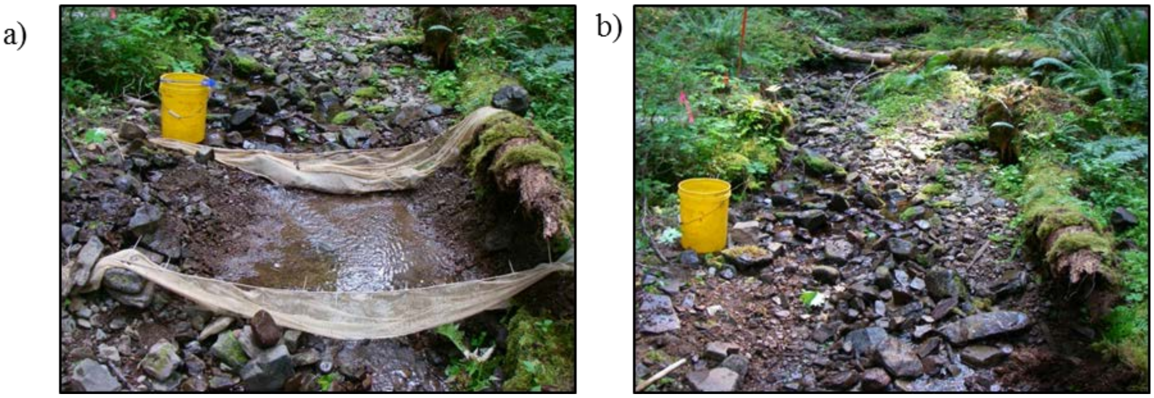


Figure 15-3. Rubble-rouse sampling for the Type N Study: (a) Placement of up- and down-stream nets and removal of course substrate to fines and small gravel; and (b) Replacement of stream substrates after sampling was completed. Photo credits: Eric M. Lund.

15-4.2.1. Lower Np Reach Rubble-rouse Sampling

We conducted rubble-rouse sampling in the lower Np reach in two pre- (2006 and 2007) and two post-harvest (2009 and 2010) years. We used a stratified random sampling approach where we established one, 1-m long rubble-rouse plot in each of 20 contiguous 10-m stream intervals located upstream of the F/N break (**Figure 15-2**). When time and conditions permitted, we sampled more plots. 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 for some sites and years, as well as placement of plots more than 200 m upstream of the F/N break. Of note, the extent of surface flow in the OLYM-0% limited the number of sampled units. The total number of 1-m rubble-rouse plots across all sites and years ranged from 9 to 27 (mean = 20) and the extent of our sampling ranged from 20 to 290 m of stream length directly above the F/N break. Though the length of stream sampled above the F/N break sometimes varied from year to year, the overall difference among years was never greater than 70 m.

We measured the wetted width of the stream at each 1-m rubble-rouse plot location to the nearest decimeter and calculated the area (A) of each plot sampled as:

$$A = l^* w \qquad (15\text{-}1)$$

where: l is the length of the 1-m rubble-rouse plot (equal to 1 m by definition), and
w is the wetted stream width at the plot location.

15-4.2.2. Obstructed Reach Rubble-rouse Sampling

During the post-harvest period, we were unable to use the light-touch method along all study reaches at all study sites due to obstructed reaches formed by in-channel wood, much of which was associated with logging activity (**Figure 15-4a**). We estimated wood cover in post-harvest years (2009 and 2010) throughout every study site as the percentage of each 10-m stream interval covered by new wood to the nearest 10% (see Chapter 6 − *Wood Recruitment and Loading* for methods). We defined new wood as fallen or cut trees, branches, twigs and leaves (if still attached) that appeared to have entered the stream during the preceding year. New wood included “green wood” (pieces with green leaves and intact bark that lacked evidence of aging) and “weathered wood” (pieces that did not meet the definition of green wood but were located on or above pieces identified as green wood, suggesting entry into the stream was recent). We determined through a pilot study that we could not effectively conduct light-touch surveys when wood cover for a 10-m stream interval was ≥70%. Therefore, we subsampled these obstructed reaches with a rubble-rouse approach.

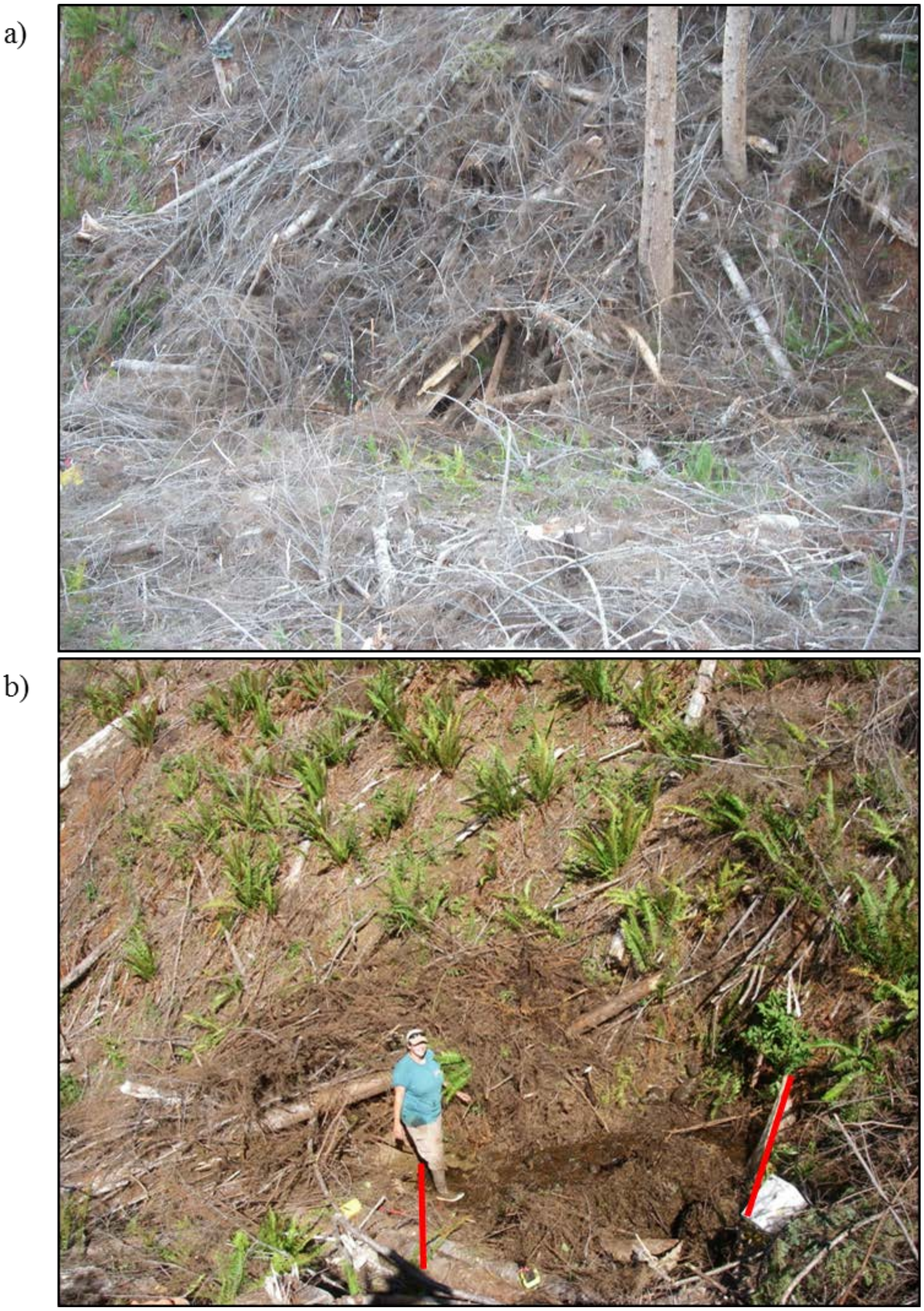


Figure 15-4. Amphibian sampling in obstructed reaches for the Type N Study: (a) an example of a stream reach obstructed to light-touch amphibian sampling due to instream wood in the form of logging slash; and (b) a 3-m long obstructed plot, sampled with rubble-rouse, located in a similarly obstructed reach. Placement of down- and upstream nets for amphibian capture is delineated in red. Wood and logging slash were removed from the stream using handheld saws, loppers and clippers. Photo credit: Frithiof T. Waterstrat.

We used rubble-rouse sampling to estimate amphibian density in a systematic subset of obstructed reaches (e.g., **Figure 15-2**). To do this, we first removed in-channel wood using handheld saws, loppers and clippers to access the stream (**Figure 15-4b**). Since there was an increased time investment in accessing the stream, we increased plot length from 1 to 3 m to increase the likelihood of detecting individuals. Furthermore, while we minimized disturbance by avoiding placement of 1-m rubble-rouse plots around steps in the lower Np reach, we could not avoid steps while sampling in obstructed reaches because steps were not apparent prior to the removal of wood. We replaced all removed substrates back in the stream prior to releasing captured amphibians. We prioritized placement of the downstream 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.

We based the number of 3-m rubble-rouse plots in obstructed reaches (hereafter, obstructed plots) on the proportion of stream obstructed in a given year. We sampled 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. In some instances, when new wood consisted of large downed trees suspended above the stream channel, we were able to conduct light-touch sampling in a reach that had ≥70% new wood. In these circumstances, the number of obstructed plots was reduced accordingly. We employed obstructed plots within each of the three buffer treatments: 100%, FP, and 0%. The total number of plots sampled was 22 in 2009 and 26 in 2010 (**Table 15-4**).

Table 15-4. The number of obstructed plots (3-m long plots sampled using rubble-rouse) by site, year and stream order.

Block	Treatment	2009		2010	
		1st	2nd/3rd	1st	2nd/3rd
OLYM	FP	0	0	2	0
	0%	2	4	3	3
WIL1	100%	1	2	1	0
	FP	4	0	4	0
	0%	2	2	2	2
WIL2	100%	0	0	1	2
	0%	2	1	2	2
CASC	FP	2	0	2	0
Total		**13**	**9**	**17**	**9**

15-4.3. ANIMAL PROCESSING

During both rubble-rouse and light-touch sampling, we captured amphibians by hand or with a dip net and identified them to species and life stage (larva, metamorph [individuals undergoing metamorphosis for Coastal Tailed Frog], neotene [for giant salamanders] or post-metamorph). 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 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 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 conducted all amphibian surveys diurnally between 0700 and 1900 hours during the summer low-flow period, generally July−October.

15-4.4. OCCUPANCY DETERMINATION

We summarized amphibian species occupancy as the taxa detected with our stream network-wide light-touch or rubble-rouse sampling in the lower Np reach by site and year. We did not include 14 animals (0.1%) that we could not identify to genus, three in the pre- and 11 in the post-harvest period. We also did not include animals from the 3-m obstructed plots since we conducted these surveys only in the post-harvest period and they were not equally distributed across all sites and treatments.

15-4.5. DENSITY ESTIMATION

We estimated amphibian density at two spatial scales: lower Np reach (hereafter, lower Np density) and stream network-wide or site (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.

15-4.5.1. Lower Np Density

We used data from our 1-m rubble-rouse plots in the lower Np reach to estimate lower Np density. We calculated amphibian density (D_{spp}) as:

$$D_{spp} = C_{spp} / A \tag{15-2}$$

where: C_{spp} is the number of captures by species for a 1-m rubble-rouse plot, and
A is the area calculated for the corresponding rubble-rouse plot.

We averaged density for each species (and stage for Coastal Tailed Frog) across all 1-m rubble-rouse plots to estimate amphibian density by site and year.

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, only data from 2007, 2009 and 2010 are included in the formal analysis.

15-4.5.2. 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 systematically collected throughout the entire study site using the light-touch method. We established plots (hereafter single-pass plots) at each study site according to the following sets of endpoints: the F/N break, tributary junctions, buffer type boundaries and uppermost points of perennial flow (PIPs; **Figure 15-5**). We assigned each single-pass plot to one combination of covariates. Covariates included two stream order types (first-order and second/third-order) and three buffer types (reference [unharvested], buffered [two-sided riparian buffer] and unbuffered [no riparian buffer on either side of the stream]), thus each single-pass plot was defined by a single buffer type and stream order. We summed the total stream length represented by each of the unique combinations of stream segment types (buffer type, stream order, obstructed). The number of single-pass plots at a site ranged from 2 to 21. The buffer type assigned to all single-pass plots during the pre-harvest period (2006−2008) was reference. We summed amphibian count data by segment type, and calculated the proportion of the stream length sampled by segment type for each site and year. We encountered very few focal amphibians (1.6%) in reaches of the stream that were dry at the time of sampling. Further, we made only two detections of focal amphibian taxa in our five dry detection plots in 2008. This finding was not surprising since the focal amphibians encountered were mostly larvae or neotenes, or torrent salamander adults most typically found in close association with water. As a result, dry stream reaches were not included in the analysis of stream network-wide density and dry plots were not included in our estimates of detection.

We field-verified the stream order (Strahler 1952) for each single-pass plot 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 StowAway TidbiT thermistors (Onset Computer Corporation, Bourne, Massachusetts; see Chapter 7 − *Stream Temperature and Cover*). Temperature sensors were spaced from the F/N break to the PIP on the mainstem tributary 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.

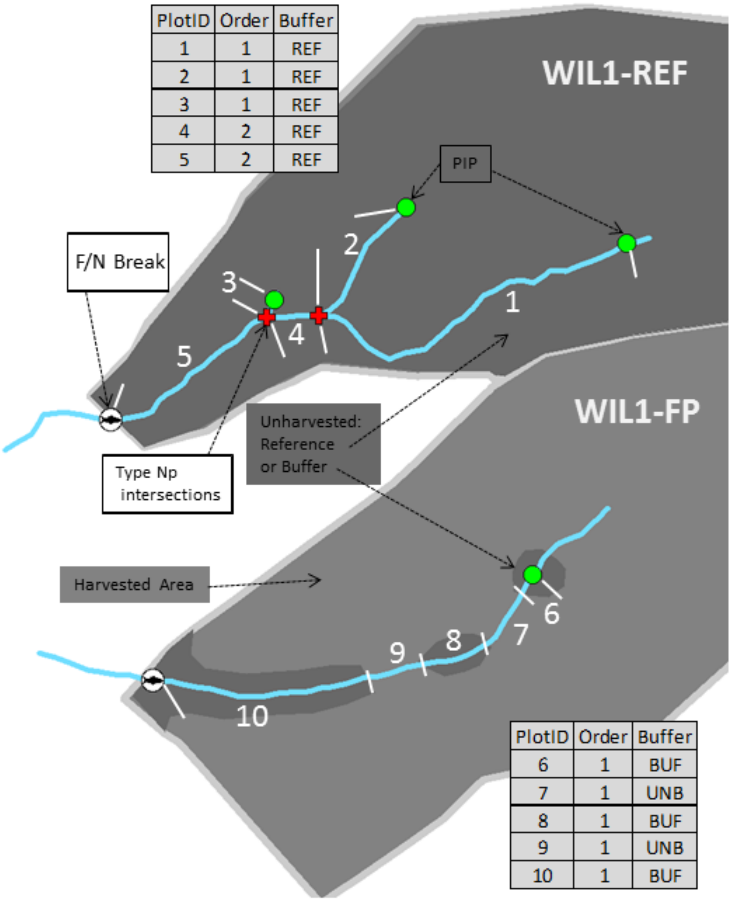


Figure 15-5. An example of single-pass plot delineation for light-touch sampling of focal amphibians at two study sites included in the Type N Study, Plots 1−5 in the WIL1-REF and plots 6−10 in the WIL1-FP. We established single-pass plots from the F/N break to each PIP by delineating breaks between tributary junctions and buffer types (BUF = buffered, UNB = unbuffered). Type N basins are delineated in gray: dark gray = reference conditions or stream buffers, light gray = clearcut harvest.

15-4.5.3. Calculating density

We calculated stream network-wide amphibian density for each study site and year as a linear density (number of animals/30 m) in four steps. First, we estimated detection probability at the 30-m detection plot level. Next, we divided observed counts in all single-pass plots by the detection probability estimated for each different combination of covariates (stream order, temperature and buffer type). We then calculated 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 normalized density to 30 m. Finally, we calculated a stream network-wide weighted average of adjusted single-pass plot-level densities based on total stream lengths for each stream order and buffer type combination.

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. 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 random basin intercept. The detection model for these two genera contained covariates for stream order, temperature, year and buffer type. Due to low counts for Coastal Tailed frog larvae and post-metamorphs, we were not able to fit the same abundance and detection models we used for torrent and giant salamanders. The abundance model for the tailed frogs included covariates for stream order, year, and buffer type. The detection model contained only buffer type.

In the abundance model for all taxa, 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 plots 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, we defined pre-harvest buffer type for all study sites as a reference condition, 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 (2006−2010), using the appropriate covariate data. We did not have the replicated count data for pre-harvest years 2006 or 2007 needed to estimate detection probability, so we based estimates for detection probabilities for those years on pre-harvest year 2008. We justified this approach based on: (1) all years 2006−2008 are in the reference state; (2) relevant covariate data were collected during 2006 and 2007; and (3) detection probability estimates for 2009 and 2010 were close for all species. While we did not statistically evaluate the impacts of the windthrow event on amphibian density or detection, we felt comfortable with our ability to apply our standard light-touch sampling technique throughout the stream network at all study sites, regardless of the severity of windthrow. Additionally, we conducted a sensitivity analysis by fitting the Before-After-Control-Impact (BACI) model without 2006 and 2007 data and comparing results to the full analysis. Across all species, the results were sufficiently similar that we felt comfortable including the 2006 and 2007 data, which provided better precision on our estimates due to larger sample sizes.

We divided observed counts for all single-pass plots and the first visit for detection plots by the estimated mean detection probability to derive abundance. We did not incorporate uncertainty in detection probability in this analysis (an assessment of the impact of this uncertainty is included in **Appendix 15-A)**. 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 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. We report two separate estimates obtained from our stream network-wide sampling: (1) linear density estimates based solely on light-touch counts in single-pass plots adjusted for imperfect detection collected at detection plots; and (2) linear density as described in (1) plus captures from 3-m obstructed plots that assumed a detection probability of 1. 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 2009 and 2010. 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:

$$\widetilde{N}_{ijh} = C \cdot \frac{\sum_k \widetilde{N}_{ijhk}}{\sum_k c_{ijhk}} \tag{15-3}$$

where: *k* indexes plot,
$\widetilde{N}_{ijhk}$ 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:

$$\widetilde{N}_{ij} = \sum\nolimits_h w_{ijh} \cdot \widetilde{N}_{ijh} \tag{15-4}$$

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

We present stream network-wide abundance both including and excluding obstructed reaches, primarily because many previous studies have not included estimates for amphibian density in areas obstructed with heavy slash loading. Estimating stream network-wide density both ways allows direct comparison of results using two different strategies. If results based on the differing analyses are not similar, then one of two things could be the cause, either density of amphibians in obstructed reaches was different from density in unobstructed reaches, or our light-touch sampling did not census the same proportion of the population as our sampling in obstructed reaches.

15-4.6. BODY CONDITION

We used body length and mass data gathered during rubble-rouse sampling conducted in our 1-m plots in the lower Np and our 3-m plots in obstructed reaches in two pre- (2006 and 2007) and two post-harvest years (2009 and 2010) 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:

$$\widehat{M}_i = M_i \left[\frac{L_0}{L_i}\right]^{b_{SMA}} \tag{15-5}$$

where: M_i and L_i 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 value 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 and tailed frog larvae because tails are important sites for fat storage in many species (Sheridan and Kao 1998). We used snout-vent length (SVL) for post-metamorphic tailed frogs. We had TL and weight data for 481 tailed frog larvae, 1,644 torrent salamanders and 1,131 giant salamanders. We had SVL and weight data for 97 tailed frog post-metamorphs. We did not include animals in the analysis that were injured (e.g., missing part of a tail or leg), gravid, or that showed evidence of prior tissue removal for genetic or stable isotope analysis. Animals excluded from the analysis included 10 tailed frog larvae, 94 torrent salamanders, and 143 giant salamanders, encountered during both the pre- and post-harvest periods. We also excluded 6 post-metamorphic tailed frogs and 5 post-metamorphic giant salamanders from the analysis.

To calculate SMI for each genera, and life stage for tailed frogs, we examined scatter plots of mass versus length 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 were considered improbable and removed from the data set. We identified 60 (12.5%) larval tailed frogs, 14 (1.1%) giant salamanders, and 52 (2.6%) torrent 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 genera, site, and sample year combinations, we averaged the SMI for each genus, life stage for tailed frogs, and site across years within the pre- and post-harvest periods rather than for each single year. We calculated SMI for 122 and 289 tailed frog larvae, 30 and 58 tailed frog post-metamorphs, 625 and 1,213 torrent salamanders, and 500 and 587 giant salamanders in the pre- and post-harvest periods, respectively.

15-4.7. STATISTICAL ANALYSIS

Analysis of density and body condition evaluated the generalized null hypothesis:

$$\Delta T_{\text{REF}} = \Delta T_{100\%} = \Delta T_{\text{FP}} = \Delta T_{0\%} \tag{15-6}$$

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. 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 among 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 −1.5 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.

15-5. RESULTS

15-5.1. OCCUPANCY

The number of amphibian detections from systematic light-touch sampling and rubble-rouse sampling in the lower Np reach combined across all study years was 13,507 individuals, of which 97% were focal amphibians. We observed 16 amphibian species across the study sites on at least one occasion during the pre-harvest period (2006 and 2007), while we observed 15 species across the study sites during the post-harvest period (2009 and 2010; **Table 15-5**). We observed spatial patterns in amphibian distribution between the lower Np reach (within 200 m upstream of the F/N break) and the remaining upstream reach. Though only 34% of our sampling effort (total meters of stream length sampled across sites and years) was focused in the lower Np reach, we observed 61% of the tailed frog larvae in this reach. Conversely, torrent salamanders were more equally distributed throughout the Np basin, with 36% of total captures observed in the lower Np reach and 64% in the remaining upstream reach, where 66% of our effort was focused. As a result, we did not include data from 2008 in our investigation of occupancy since sampling in 2008 covered less of the entire stream network and focused on sampling in the lower Np reach.

15-5.1.1. Coastal Tailed Frog

We detected Coastal Tailed Frog in all pre- and post-harvest years at 14 of 17 sites, with a total of 1,661 detections. Where detected, the number of tailed frogs per site and year ranged from 2 to 268 (mean = 57). We had one study site in which we detected tailed frog solely in the post-harvest period (OLYM-100%). In this instance detection consisted of two post-metamorphic individuals in a single year. We detected nine egg masses at five sites (OLYM-REF, OLYM-FP, WIL1-FP, WIL2-REF2, WIL2-0%) across all study years, one in the pre- and eight in the post-harvest period. The proportion of larvae, metamorphs, and post-metamorphs was 54, 19, and 27% in the pre-, versus 33, 48, and 20% in the post-harvest period, respectively. While the proportion of metamorphosing individuals increased across all treatments in the post-harvest period, we observed the greatest increases in the FP and 0% treatment sites (**Figure 15-6**). We did not detect tailed frog in the CASC-FP or CASC-0% in either period. Prior to verifying the F/N break during study reconnaissance sampling, these two sites were considered one Type N basin. We detected one Coastal Tailed Frog larva during reconnaissance surveys; however, movement of the F/N break upstream split the basin into two study sites. While we did not detect tailed frogs in either site with our systematic light-touch or rubble-rouse sampling in the lower Np reach, we did observe one post-metamorphic Coastal Tailed Frog during the first post-harvest year in the CASC-FP on the second visit to one of our detection plots. The movement of the F/N break upstream apparently limited the resulting two Type N basins to an area that tailed frogs use infrequently. In 2009, we detected four tailed frogs in two 3-m obstructed plots in the OLYM-0%, even though the species went undetected with both light-touch and rubble-rouse sampling in the lower Np reach in the same site and year.

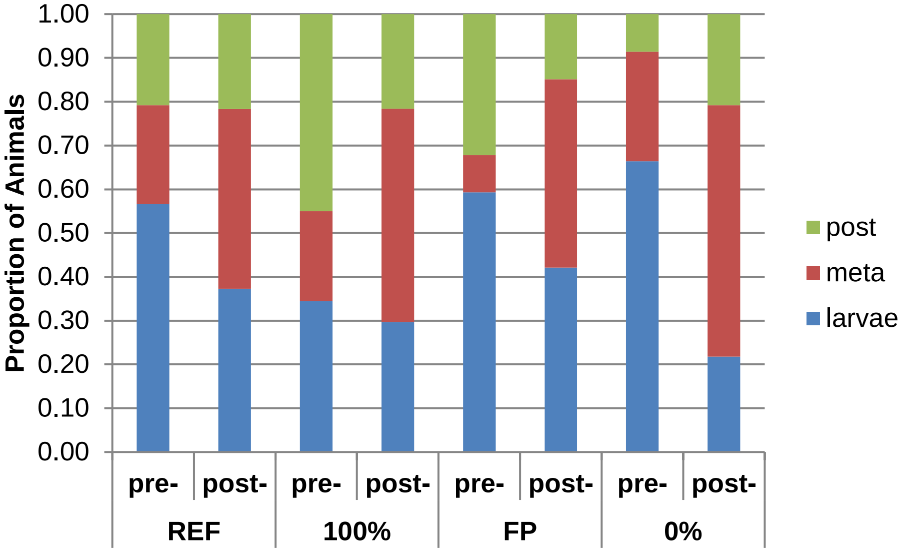


Figure 15-6. The proportions of larvae, metamorphosing (meta) and post-metamorphic (post) Coastal Tailed Frogs observed by treatment for the pre- and post-harvest periods.

15-5.1.2. Giant Salamanders

We detected one or both giant salamander species (Cope's and Coastal) in all 17 sites in all pre- and post-harvest years, with a total of 3,599 detections. Where detected, the number of giant salamanders per site and year ranged from 3 to 290 (mean = 55). The proportion of larvae, neotene and post-metamorphs was 88, 11, and 1% in the pre-, versus 90, 9, and 1% in the post-harvest period, respectively (we could not classify 1 and 2% of individuals to stage in the pre- and post-harvest periods, respectively). Our sampling methodology was not designed to focus on post-metamorphic individuals, which are not restricted to the stream corridor; however, we did detect 34 (<1%) post-metamorphic individuals. Post-metamorphic individuals were encountered nearly equally across both periods (18 in the pre- and 16 in the post-harvest period) and across all treatments in the pre- and post-harvest periods (**Appendix Table 15-B-3**). We identified 17 individuals to species with genetic analysis: Coastal (n = 5), Cope's (n = 9), hybrid (including backcross to Cope's and F2; n = 3).

The genetic analysis confirmed that Cope's Giant Salamanders were present at all study sites. We detected Coastal Giant Salamanders in 11 of 17 sites during both the pre- and post-harvest periods. We never detected the species in the four sites in the Olympic block or the WIL1-0% and detected it in the WIL2-REF in only the pre- and the WIL2-100% in only the post-harvest period (see **Appendix 15-B** for a summary of the proportions of giant salamander species across all study sites).

15-5.1.3. Torrent Salamanders

We detected torrent salamanders in all pre- and post-harvest years in 15 of 17 sites, with a total of 8,243 detections. Where detected, the number of torrent salamanders per site and year ranged from 1 to 415 (mean = 131). In the OLYM-FP, we detected the Olympic Torrent Salamander in only one of two pre-harvest years (six individuals: three larvae and three post-metamorphs). In the OLYM-0%, we detected one to three Olympic Torrent Salamanders in both of the pre- and one of two post-harvest years. Detections in the latter site totaled only five over three years (one post-metamorph in 2006, two post-metamorphs and one larva in 2007, and one larva in 2010). The proportion of larvae and post-metamorphs was 87% and 13% in the pre-, versus 79% and 21% in the post-harvest period, respectively. In 2010, we detected four torrent salamanders in one 3-m obstructed plot in the OLYM-FP, even though the species went undetected with both light-touch and rubble-rouse sampling in the lower Np reach in the same site and year.

15-5.1.4. Other Amphibian Species

The numbers of non-focal amphibian species we detected across sites was limited relative to the numbers of focal species, with a maximum of 208 individuals across 16 study sites for the most commonly encountered species, and as few as two individuals encountered in one or two sites for the least commonly encountered species (**Table 15-6**). Western Red-backed Salamander (*Plethodon vehiculum*) was the most commonly encountered non-focal species, followed by Northern Red-legged Frog (*Rana aurora*) and Van Dyke's Salamander (*P. vandykei*). We encountered the remaining non-focal species at few sites, with few detections throughout the entire study period. No obvious patterns existed in the changes in detections of non-focal species across sites and periods. Changes in detection of a species occurred in both directions (detected in the pre- and not the post-harvest period, and vice versa), and occurred in all treatments with a similar frequency.

Table 15-5. Focal amphibian taxa detected during systematic light-touch sampling conducted stream network-wide (s) and rubble-rouse sampling restricted to the lower Np reach (l) across sites and periods (pre-harvest [2006‒2008] and post-harvest [2009‒2010]). Incidences where taxa were detected at a site in one period but not the other are identified by a **bold** cell border. Instances where a focal species was detected utilizing only one of the two sampling methods are highlighted gray.

Treatment	Block	Coastal Tailed Frog		Giant Salamanders		Torrent Salamanders	
		Pre	**Post**	**Pre**	**Post**	**Pre**	**Post**
REF	OLYM	sl	sl	sl	sl	s	sl
	WIL1	sl	sl	sl	sl	sl	sl
	WIL2-1	sl	sl	sl	sl	sl	sl
	WIL2-2	sl	sl	sl	sl	sl	sl
	WIL3	sl	sl	sl	sl	sl	sl
	CASC	sl	sl	sl	sl	s	sl
100%	OLYM	-	sl	sl	sl	sl	sl
	WIL1	sl	sl	sl	sl	sl	sl
	WIL2	sl	sl	sl	sl	sl	sl
	WIL3	s	sl	sl	sl	sl	sl
FP	OLYM	sl	sl	sl	sl	s	-
	WIL1	sl	sl	sl	sl	sl	sl
	CASC	-	-	sl	sl	sl	sl
0%	OLYM	sl	l	sl	sl	s	s
	WIL1	sl	sl	sl	sl	sl	sl
	WIL2	sl	sl	sl	sl	sl	sl
	CASC	-	-	sl	sl	sl	sl

Table 15-6. Non-focal species detected in study sites, including total detections across the study period, the total number of sites in which the species was detected, and the numbers of sites in which they were detected in the pre- and post-harvest periods.

Common Name	Latin Name	Total Detections	Total Sites Detected	Number of Sites	
				Pre	Post
Western Red-backed Salamander	*Plethodon vehiculum*	208	16	16	15
Northern Red-legged Frog	*Rana aurora*	65	11	9	9
Van Dyke’s Salamander	*P. vandykei*	59	11	10	7
Rough-skinned Newt	*Taricha granulosa*	19	6	4	4
Dunn’s Salamander	*P. dunni*	15	4	3	3
Pacific Chorus Frog	*Pseudacris regilla*	8	4	2	3
Larch Mountain Salamander	*P. larselli*	6	1	1	1
Northwestern Salamander	*Ambystoma gracile*	2	2	1	1
Western Toad	*Anaxyrus boreas*	2	2	1	1
Ensatina	*Ensatina eschscholtzii*	2	1	1	0

15-5.2. DENSITY

15-5.2.1. Lower Np Density

The following estimates are derived from our 1-m rubble-rouse plots in the lower Np reach, assuming a detection probability of 1.

15-5.2.1.a. Coastal Tailed Frog—larvae

Where detected, larval tailed frog densities ranged from 0.04 to 2.55 animals per m^2. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of larval tailed frog density (**Table 15-7** and **Figure 15-7**). The P-value for the period × treatment contrast was 0.19, indicating that the amount of change in larval tailed frog density did not depend on treatment. We estimated the difference in the change for each of the six combinations among references and buffer treatments (**Table 15-8**).

Table 15-7. The within-treatment change for the natural log (ln) of larval Coastal Tailed Frog density (animals/m^2), i.e., [ln(post) − ln(pre)], and 95% confidence intervals (CI) estimated with rubble-rouse sampling in the lower Np reach for each of four treatments. SE is the standard error.

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	0.80	0.46	−0.15	1.74
100%	0.79	0.69	−0.63	2.21
FP	−0.13	0.68	−1.54	1.28
0%	−0.66	0.55	−1.78	0.47

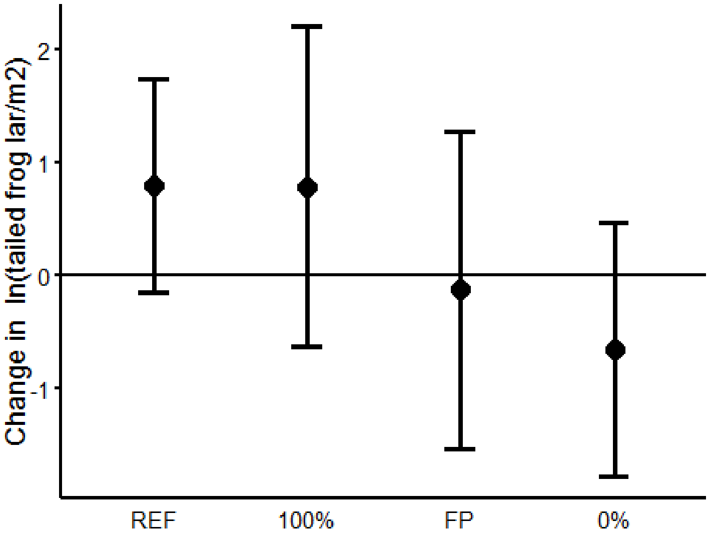


Figure 15-7. The within-treatment change for the natural log (ln) of larval Coastal Tailed Frog density (animals/m^2), i.e., [ln(post) − ln(pre)], and 95% confidence interval (CI) estimated with rubble-rouse sampling in the lower Np reach for each of four treatments.

Table 15-8. The between-treatment difference in the change in the natural log (ln) of larval Coastal Tailed Frog density (animals/m^2), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]), and 95% confidence intervals (CI) estimated with rubble-rouse sampling in the lower Np reach for all pairwise comparisons. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the Riparian Management Zone [RMZ] buffer). SE is the standard error.

Contrast	Estimate	SE	95% CI	
			Lower	Upper
100% vs. REF	−0.01	0.82	−1.71	1.69
FP vs. REF	−0.93	0.82	−2.62	0.77
0% vs. REF	−1.45	0.71	−2.92	0.01
0% vs. FP	−0.53	0.87	−2.33	1.28
0% vs. 100%	−1.44	0.88	−3.26	0.37
FP vs. 100%	−0.92	0.97	−2.92	1.08

15-5.2.1.b. Coastal Tailed Frog — post-metamorphs

Where detected, post-metamorphic tailed frog densities ranged from 0.03 to 0.56 animals per m^2. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of post-metamorphic tailed frog density (**Table 15-9** and **Figure 15-8**). The P-value for the period × treatment contrast was 0.07, indicating that the amount of change in tailed frog density depended on treatment. We estimated the change in the FP treatment to be different from the change in the reference (P = 0.07), 100% (P = 0.01) and 0% treatments (P = 0.03; **Table 15-10**).

Table 15-9. The within-treatment change for the natural log (ln) of post-metamorphic Coastal Tailed Frog density (animals/m^2), i.e., [ln(post) − ln(pre)], and 95% confidence intervals (CI) estimated with rubble-rouse sampling in the lower Np reach for each of four treatments. SE is the standard error.

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	−0.19	0.38	−0.97	0.59
100%	0.51	0.50	−0.52	1.55
FP	−1.54	0.60	−2.78	−0.31
0%	0.50	0.69	−0.92	1.92

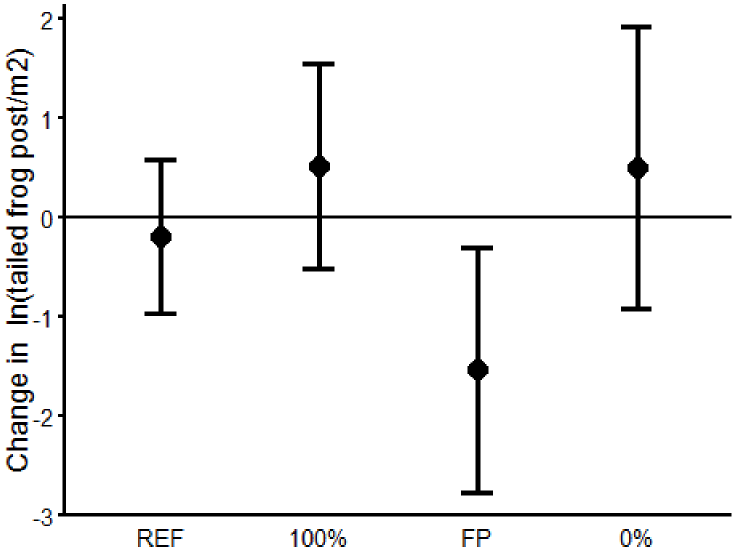


Figure 15-8. The within-treatment change for the natural log (ln) of post-metamorphic Coastal Tailed Frog density (animals/m^2), i.e., [ln(post) − ln(pre)], and 95% confidence interval (CI) estimated with rubble-rouse sampling in the lower Np reach for each of four treatments.

Table 15-10. The between-treatment difference in the change in the natural log (ln) of post-metamorphic Coastal Tailed Frog density (animals/m^2), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]), and 95% confidence interval (CI) estimated with rubble-rouse sampling in the lower Np reach for all pairwise comparisons. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Estimate	SE	P-value	95% CI	
				Lower	Upper
100% vs. REF	0.71	0.63	0.27	−0.59	2.00
FP vs. REF	−1.35	0.71	0.07	−2.81	0.11
0% vs. REF	0.69	0.78	0.39	−0.93	2.31
0% vs. FP	2.04	0.91	0.03	0.16	3.92
0% vs. 100%	−0.01	0.85	0.99	−1.77	1.74
FP vs. 100%	−2.06	0.78	0.01	−3.66	−0.45

15-5.2.1.c. Torrent salamanders

Where detected, torrent salamander densities ranged from 0.03 to 8.85 animals per m^2. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of torrent salamander density (**Table 15-11** and **Figure 15-9**). The P-value for the period × treatment contrast was 0.30, indicating that the amount of change in torrent salamander density did not depend on treatment. We estimated the difference in the change for each of the six combinations among references and buffer treatments (**Table 15-12**).

Table 15-11. The within-treatment change for the natural log (ln) of torrent salamander density (animals/m^2), i.e., [ln(post) − ln(pre)], and 95% confidence intervals (CI) estimated with rubble-rouse sampling in the lower Np reach for each of four treatments. SE is the standard error.

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	−0.09	0.16	−0.42	0.23
100%	0.17	0.26	−0.37	0.72
FP	−0.37	0.16	−0.70	−0.03
0%	−0.36	0.31	−0.99	0.27

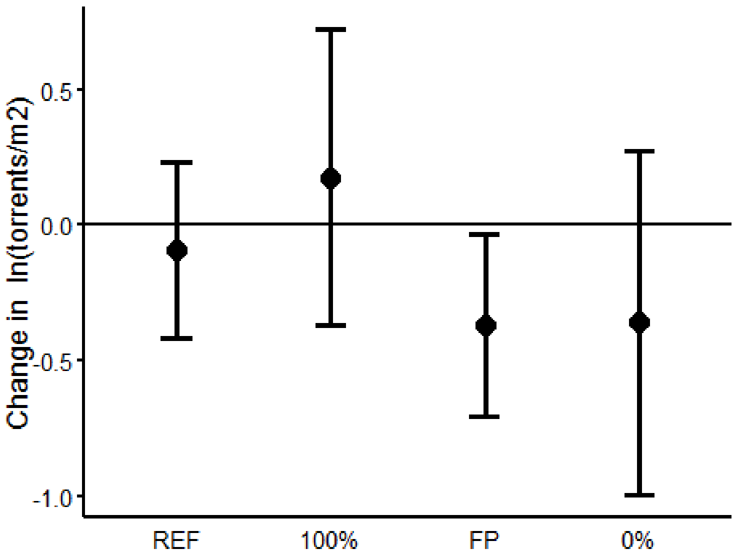


Figure 15-9. The within-treatment change for the natural log (ln) of torrent salamander density (animals/m^2), i.e., [ln(post) − ln(pre)], and 95% confidence interval (CI) estimated with rubble-rouse sampling in the lower Np reach for each of four treatments.

Table 15-12. The between-treatment difference in the change in the natural log (ln) of torrent salamander density (animals/m^2), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]), and 95% confidence intervals (CI) estimated with rubble-rouse sampling in the lower Np reach for all pairwise comparisons. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Estimate	SE	95% CI	
			Lower	Upper
100% vs. REF	0.27	0.31	−0.37	0.90
FP vs. REF	−0.28	0.23	−0.74	0.19
0% vs. REF	−0.27	0.34	−0.98	0.44
0% vs. FP	0.01	0.35	−0.71	0.72
0% vs. 100%	−0.53	0.40	−1.37	0.30
FP vs. 100%	−0.54	0.31	−1.18	0.10

15-5.2.1.d. Giant salamanders

Giant salamander densities ranged from 0.06 to 4.98 animals per m^2. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of giant salamander density (**Table 15-13** and **Figure 15-10**). The P-value for the period × treatment contrast was 0.10, indicating that the amount of change in giant salamander density depended on treatment. We estimated the change in the FP treatment to be different from the change in the reference (P = 0.09) and 100%

treatments (P = 0.02). We estimated the difference in the change for each of the six combinations among references and buffer treatments (**Table 15-14**).

Table 15-13. The within-treatment change for the natural log (ln) of giant salamander density (animals/m^2), i.e., [ln(post) − ln(pre)], and 95% confidence intervals (CI) estimated with rubble-rouse sampling in the lower Np reach for each of four treatments. SE is the standard error.

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	−0.26	0.23	−0.73	0.20
100%	0.07	0.26	−0.46	0.61
FP	−0.88	0.26	−1.41	−0.34
0%	−0.40	0.29	−1.00	0.19

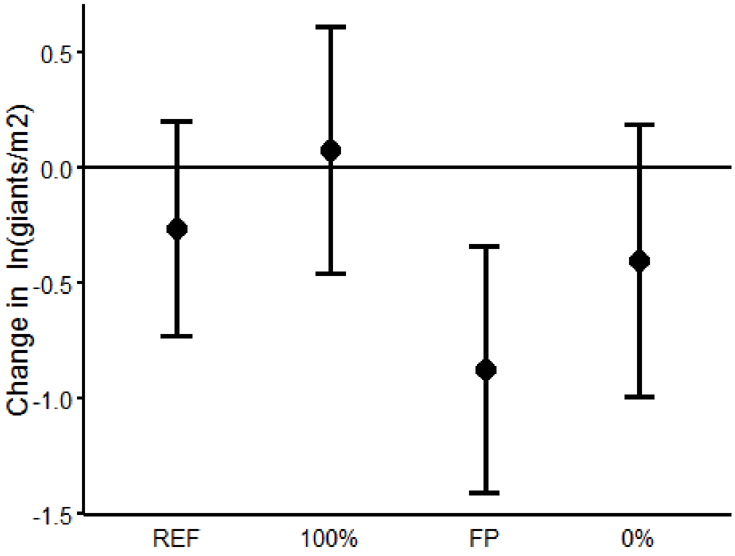


Figure 15-10. The within-treatment change for the natural log (ln) of giant salamander density (animals/m^2), i.e., [ln(post) − ln(pre)], and 95% confidence interval (CI) estimated with rubble-rouse sampling in the lower Np reach for each of four treatments.

Table 15-14. The between-treatment difference in the change in the natural log (ln) of giant salamander density (animals/m^2), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]), and 95% confidence interval (CI) estimated with rubble-rouse sampling in the lower Np reach for all pairwise comparisons. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Estimate	SE	P-value	95% CI	
				Lower	Upper
100% vs. REF	0.34	0.34	0.33	−0.37	1.05
FP vs. REF	−0.61	0.34	0.09	−1.32	0.10
0% vs. REF	−0.14	0.37	0.71	−0.89	0.61
0% vs. FP	0.47	0.39	0.23	−0.33	1.27
0% vs. 100%	−0.48	0.39	0.23	−1.27	0.32
FP vs. 100%	−0.95	0.37	0.02	−1.70	−0.20

15-5.2.2. Stream Network-wide Density

15-5.2.2.a. Detection plots

We made 2,880 observations of amphibians in detection plots across three sample years (2008–2010). Of those, 2,821 (98%) were focal amphibians. We did not include individuals from eight (<0.3%) observations that could not be identified to species because they were not captured (n = 6) or were plethodontid salamanders with non-characteristic forms (n = 2). There were 51 (<2%) encounters of six non-focal species: Dunn’s (n = 2), Van Dyke’s (n = 8), and Red-backed Salamanders (n = 18); Pacific Chorus (n = 3) and Northern Red-legged Frogs (n = 17); and Rough-skinned Newt (n = 3). The stream temperatures recorded for single-pass plots (2006, 2007, 2009 and 2010) ranged from 6.3°C to 16.1°C (mean = 10.7 °C), and the temperatures recorded for detection plots (2008−2010) ranged from 6.4°C to 16.5°C (mean = 11.1°C).

We detected Coastal Tailed Frog larvae in 52 (34%) detection plots; however, we detected tailed frog larvae during every visit in only 11 (21%) of those plots. We did not detect tailed frog larvae until the second visit in 12 (8%) plots and detected them for the first time on the third visit in five (3%) plots. Across three visits, we made 272 observations of tailed frog larvae. Overall, the numbers of tailed frog larvae we observed in a plot during a single visit ranged from 0 to 15 individuals (mean = 0.59). We estimated that mean detection probabilities across buffer types ranged from 0.03 (credible interval: 0.01−0.11) to 0.14 (credible interval: 0.04−0.27), averaged across years (2006−2010; **Table 15-15**).

We detected Coastal Tailed Frog post-metamorphs in 49 (32%) detection plots; however, we detected tailed frog post-metamorphs during every visit in only 5 (10%) of those plots. We did not detect tailed frog post-metamorphs until the second visit in 11 (7%) plots and detected them for the first time on the third visit in 18 (12%) plots. Across three visits, we made 96 observations of tailed frog post-metamorphs. Overall, the numbers of tailed frog post-metamorphs we observed in a plot during a single visit ranged from 0 to 6 individuals (mean =

0.21). We estimated that mean detection probabilities across buffer types ranged from 0.03 (credible interval: 0.00−0.33) to 0.06 (credible interval: 0.01−0.22), averaged across years (2006−2010; **Table 15-15**).

Table 15-15. Detection probability estimates and 95% credible intervals for larval and post-metamorphic Coastal Tailed Frogs by buffer type (reference, buffered, unbuffered) and giant and torrent salamanders for each buffer type and stream order (first- and second-order). Values are presented for the mean temperature and averaged across years (2006−2010).

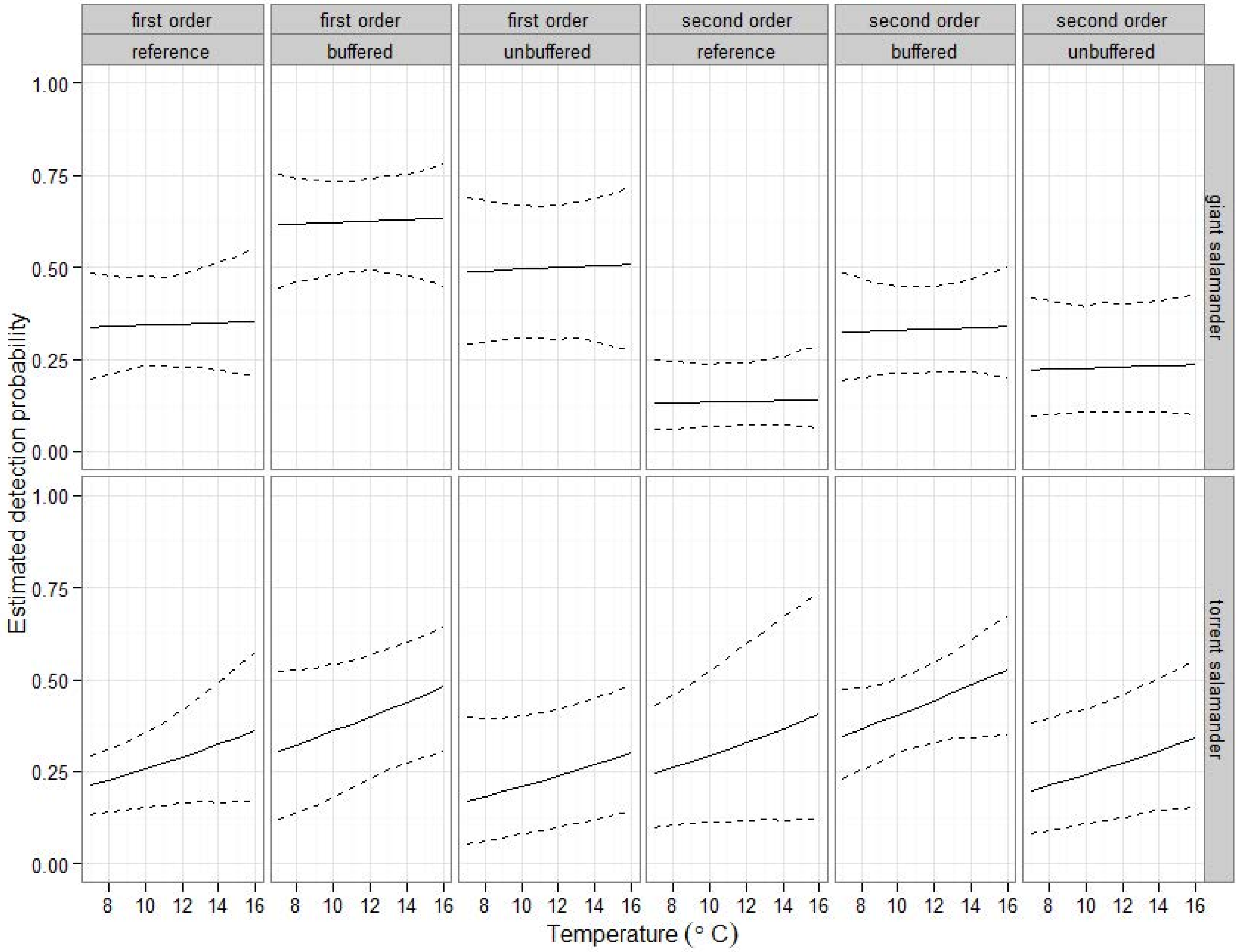
Genus	Life Stage	Order	Buffer	Mean	95% Credible Interval Lower	95% Credible Interval Upper
Coastal Tailed Frog	Larvae	All	Ref	0.14	0.04	0.27
			Buf	0.03	0.01	0.11
			Unb	0.11	0.02	0.28
	Post-metamorph	All	Ref	0.06	0.02	0.17
			Buf	0.06	0.01	0.22
			Unb	0.03	0.00	0.33
Giant Salamanders	All	1st	Ref	0.34	0.23	0.47
			Buf	0.62	0.49	0.72
			Unb	0.49	0.32	0.66
		2nd	Ref	0.13	0.07	0.24
			Buf	0.33	0.21	0.44
			Unb	0.22	0.11	0.39
Torrent Salamanders	All	1st	Ref	0.27	0.16	0.38
			Buf	0.39	0.22	0.56
			Unb	0.23	0.10	0.41
		2nd	Ref	0.31	0.11	0.56
			Buf	0.43	0.32	0.52
			Unb	0.26	0.13	0.42

We detected torrent salamanders in 117 (76%) detection plots; however, we detected torrent salamanders during every visit in only 80 (68%) of those plots. We did not detect torrent salamanders until the second visit in 7 (5%) plots and detected them for the first time on the third visit in 3 (2%) plots. Across three visits, we made 2,341 observations of torrent salamanders. Overall, the number of torrent salamanders we observed in a plot during a single visit ranged from 0 to 71 (mean = 5.07). We estimated that mean detection probabilities across buffer type and stream order ranged from 0.23 (credible interval: 0.10−0.41) to 0.43 (credible interval: 0.32−0.52), based on mean stream temperature averaged across years (2006−2010; **Table 15-15** and **Figure 15-11**).

We detected giant salamanders in 97 (63%) detection plots; however, we detected giant salamanders during every visit in only 53 (55%) of those plots. We did not detect giant salamanders until the second visit in 12 (8%) plots and detected them for the first time on the third visit in 3 (2%) plots. Across three visits, we made 1,094 observations of giant salamanders. Overall, the number of giant salamanders we observed in a plot during a single visit ranged from 0 to 52 (mean = 2.37). We estimated that mean detection probabilities across buffer type and stream order ranged from 0.13 (credible interval: 0.07−0.24) to 0.62 (credible interval: 0.49−0.72), for mean stream temperature averaged across years (2006−2010; **Table 15-15** and **Figure 15-11**).

Overall, we did observe differences in mean detection based on model terms we included in our detection probability estimates. For example, mean giant salamander detection probability is nearly two times greater in the buffered versus reference reaches along first-order streams, and nearly three times greater along second-order streams (**Table 15-15**). Parameter estimates from the N-mixture models for all species are included in **Appendix 15-C**.

Figure 15-11. Detection probability estimates for giant and torrent salamanders by temperature for each buffer type (reference, buffered, unbuffered) and stream order (first- and second-order). Values are averaged across post-harvest years (2009−2010). Dashed lines are 95% credible intervals.



15-5.2.2.b. Summary of animal density in obstructed reaches

We detected Coastal Tailed Frogs, torrent salamanders and giant salamanders in 3-m obstructed plots during both post-harvest years. Animal linear density in obstructed plots where we detected individuals ranged from 1 to 20 individuals per stream meter, dependent on taxa and year (**Table 15-16**). For comparison, we calculated average linear density based on rubble-rouse sampling in the lower Np reach and found that densities were similar to (e.g., tailed frog) or less than (e.g., torrent and giant salamanders) those estimated for obstructed plots (**Table 15-16**). We also encountered four egg masses in obstructed plots: one Coastal Tailed Frog, one Columbia Torrent Salamander and two giant salamander egg masses from three study sites (WIL1-FP, WIL1-0%, and WIL2-0%). We detected only two other amphibian species, Van Dyke's and Western Red-backed Salamanders, in obstructed plots: 1 to 2 individuals in 10 plots over both sample years.

Table 15-16. The total number of animals detected in 3-m obstructed plots in the post-harvest period (2009−2010), the number of obstructed plots in which each taxa was detected for 2009 and 2010, and the average post-harvest linear density (animals/stream m) estimated for obstructed plots and 1-m rubble-rouse plots in the lower Np reach.

Taxa	Animals	Plots (%) 2009 (n = 6)	Plots (%) 2010 (n = 8)	Density (Animals/m) Obstructed	Density (Animals/m) Lower Np Reach
Coastal Tailed Frog	18	2 (33%)	3 (38%)	1−3 ($\bar{x}$ = 2)	<1−3 ($\bar{x}$ = 1)
Torrent Salamanders	514	5 (83%)	8 (100%)	1−20 ($\bar{x}$ = 5)	<1−7 ($\bar{x}$ = 2)
Giant Salamanders	173	6 (100%)	6 (75%)	1−19 ($\bar{x}$ = 6)	<1−4 ($\bar{x}$ = 1)

15-5.2.2.c. Coastal Tailed Frog—larvae

Where detected, larval tailed frog density (animals/30 m) when including obstructed reaches and adjusted for detection ranged from 0.2 to 106.3 (**Appendix Table 15-D-1**). Estimates of stream network-wide abundance extrapolated from our 30-m density estimates ranged from 7 to 1,168 individuals. Had we not accounted for detection our stream network-wide abundance estimates would have ranged from 1 to 142 individuals. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of larval tailed frog density (animals/30 m; **Table 15-17** and **Figure 15-12**). The P-value for the period × treatment contrast was <0.0001, indicating that the amount of change in larval tailed frog density depended on treatment. We estimated the changes in the 100% and FP treatments to be different from the changes in the reference (P <0.01 and P <0.0001, respectively) and 0% treatments (P = 0.01 and P <0.0001, respectively; **Table 15-18**). We detected no difference in the within-treatment changes estimated for the 0% treatment and the reference.

Where detected, larval tailed frog density (animals/30 m) when excluding obstructed reaches and adjusted for detection ranged from 0.2 to 116.7 (**Appendix Table 15-D-1)**. Estimates of stream network-wide abundance extrapolated from our 30-m density estimates ranged from 7 to 1,168 individuals. Had we not accounted for detection, our stream network-wide abundance estimates would have ranged from 1 to 142 individuals. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of larval tailed frog density (animals/30 m; **Table 15-17** and **Figure 15-12**). The P-value for the period × treatment contrast was <0.0001, indicating that the amount of change in larval tailed frog density depended on treatment. We estimated the change in the 100% and FP treatments to be different from the change in the reference (P = 0.02 and P <0.0001, respectively) and 0% treatments (P = 0.01 and P <0.001, respectively; **Table 15-18**). We detected no difference in the within-treatment changes estimated for the 0% treatment and the reference. The pairwise comparisons that we identified as significant were the same regardless of whether obstructed reaches were included or excluded from the analysis.

Table 15-17. The within-treatment change for the natural log (ln) of stream network-wide larval Coastal Tailed Frog density (animals/30 m), i.e., [ln(post) – ln(pre)], and 95% confidence intervals (CI) estimated with light-touch sampling adjusted for detection, including and excluding obstructed reaches, for each of four treatments. SE is the standard error.

Treatment	**Including Obstructed Reaches**				**Excluding Obstructed Reaches**			
	Estimate	**SE**	**95% CI**		**Estimate**	**SE**	**95% CI**	
			Lower	**Upper**			**Lower**	**Upper**
REF	0.7	0.30	0.1	1.3	0.7	0.30	0.1	1.3
100%	2.1	0.32	1.4	2.7	2.1	0.31	1.5	2.8
FP	2.7	0.32	2.1	3.4	2.8	0.32	2.2	3.5
0%	0.5	0.51	−0.5	1.5	0.5	0.52	−0.5	1.5

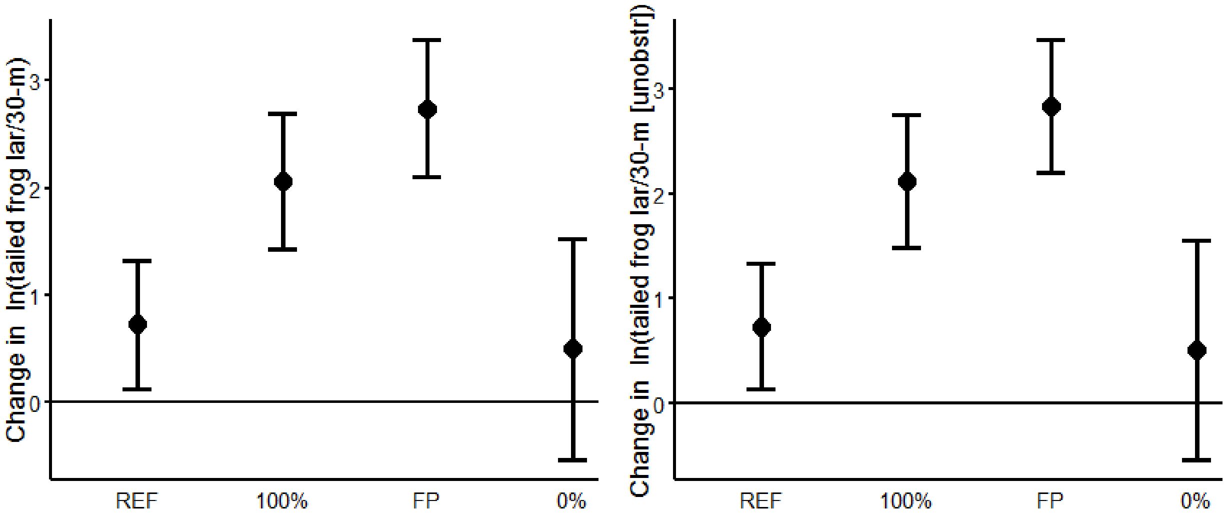


Figure 15-12. The within-treatment change for the natural log (ln) of stream network-wide larval Coastal Tailed Frog density (animals/30 m), i.e., [ln(post) − ln(pre)], and 95% confidence interval (CI) estimated with light-touch sampling adjusted for detection, including (left panel) and excluding (right panel) obstructed reaches.

Table 15-18. The between-treatment difference in the change in the natural log (ln) of stream network-wide larval Coastal Tailed Frog density (animals/m^2), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]), and 95% confidence intervals (CI), including and excluding obstructed reaches. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Including Obstructed Reaches					Excluding Obstructed Reaches				
	Est.	SE	P-value	95% CI		Est.	SE	P-value	95% CI	
				Lower	Upper				Lower	Upper
100% vs.	1.3	0.43	<0.01	0.5	2.2	1.4	0.43	0.02	0.5	2.3
FP vs. REF	2.0	0.44	<0.0001	1.1	2.9	2.1	0.43	<0.0001	1.2	3.0
0% vs. REF	−0.2	0.59	0.70	−1.4	1.0	−0.2	0.60	0.71	−1.4	1.0
0% vs. FP	−2.2	0.60	<0.001	−3.4	−1.0	−2.3	0.61	<0.001	−3.6	−1.1
0% vs. 100%	−1.6	0.60	0.01	−2.8	−0.4	−1.6	0.61	0.01	−2.8	−0.4
FP vs. 100%	0.7	0.60	0.14	−0.2	1.6	0.7	0.45	0.12	−0.2	1.6

15-5.2.2.d. Coastal Tailed Frog—post-metamorph

Where detected, post-metamorph tailed frog density (animals/30 m) when including obstructed reaches and adjusted for detection ranged from 0.3 to 35.3 (**Appendix Table 15-D-2**). Estimates of stream network-wide abundance extrapolated from our 30-m density estimates ranged from 9 to 978 individuals. Had we not accounted for detection, our stream network-wide abundance estimates would have ranged from 1 to 59 individuals. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of post-metamorph tailed frog density (animals/30 m; **Table 15-19** and **Figure 15-13**). The P-value for the period × treatment contrast was 0.11, providing no clear evidence that the amount of change in post-metamorph tailed frog density depended on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and buffer treatments (**Table 15-20**).

Where detected, post-metamorph tailed frog density (animals/30 m) when excluding obstructed reaches and adjusted for detection ranged from 0.3 to 35.3 (**Appendix Table 15-D-2**). Estimates of stream network-wide abundance extrapolated from our 30-m density estimates ranged from 16 to 978 individuals. Had we not accounted for detection, our stream network-wide abundance estimates would have ranged from 1 to 59 individuals. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of post-metamorph tailed frog density (animals/30 m; **Table 15-19** and **Figure 15-13**). The P-value for the period × treatment contrast was 0.10, indicating that the amount of change in post-metamorph tailed frog density did depend on treatment. We estimated that the change in the 0% treatment differed from the changes in the reference ($P = 0.07$), 100% ($P = 0.03$) and FP ($P = 0.02$) treatments (**Table 15-20**). The period × treatment contrast was significant when excluding obstructed reaches, but not when including obstructed reaches. However, the estimated differences in the changes for the pairwise contrasts were similar regardless of the analysis.

Table 15-19. The within-treatment change for the natural log (ln) of stream network-wide post-metamorph Coastal Tailed Frog density (animals/30 m), i.e., [ln(post) − ln(pre)], and 95% CI estimated with light-touch sampling adjusted for detection, including and excluding obstructed reaches. SE is the standard error.

Treatment	Including Obstructed Reaches				Excluding Obstructed Reaches			
	Estimate	SE	95% CI		Estimate	SE	95% CI	
			Lower	Upper			Lower	Upper
REF	0.5	0.35	−0.2	1.2	0.5	0.36	−0.2	1.2
100%	−0.1	0.36	−0.9	0.6	−0.1	0.37	−0.8	0.7
FP	0.1	0.38	−0.7	0.8	0.2	0.38	−0.6	0.9
0%	2.0	0.84	0.3	3.7	2.2	0.85	0.5	4.0

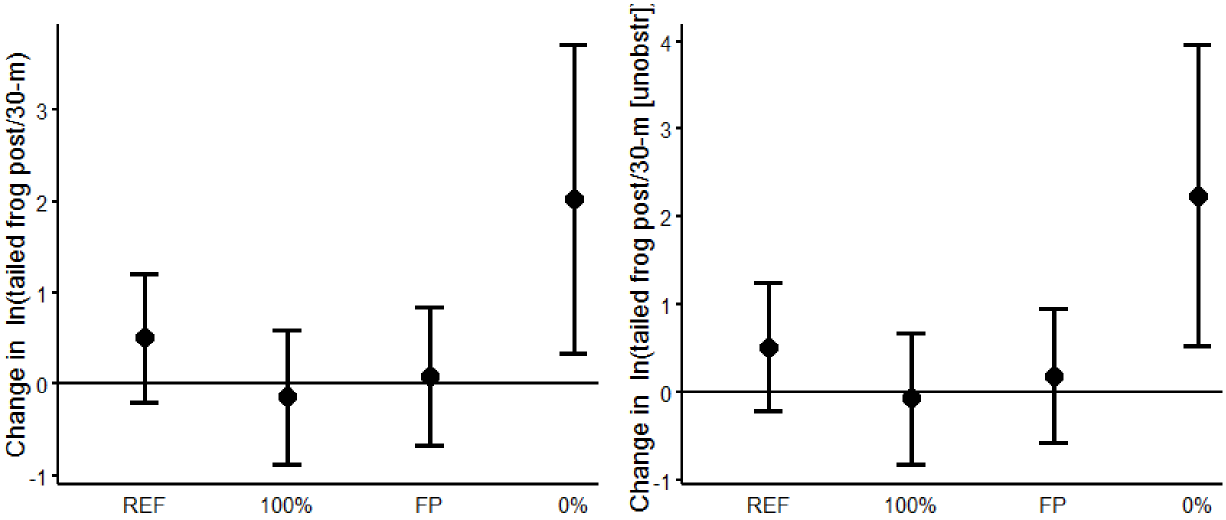


Figure 15-13. The within-treatment change for the natural log (ln) of stream network-wide post-metamorph Coastal Tailed Frog density (animals/30 m), i.e., [ln(post) – ln(pre)], and 95% confidence interval (CI) estimated with light-touch sampling adjusted for detection, including (left panel) and excluding (right panel) obstructed reaches.

Table 15-20. The between-treatment difference in the change in the natural log (ln) of stream network-wide post-metamorph Coastal Tailed Frog density (animals/m^2), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]), and 95% confidence intervals (CI), including and excluding obstructed reaches. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

	Including Obstructed Reaches				Excluding Obstructed Reaches				
Contrast	**Estimate**	**SE**	**95% CI**		**Estimate**	**SE**	**P-value**	**95% CI**	
			Lower	**Upper**				**Lower**	**Upper**
100% vs. REF	−0.7	0.50	−1.7	0.4	−0.6	0.52	0.26	−1.6	0.5
FP vs. REF	−0.4	0.51	−1.5	0.6	−0.3	0.52	0.53	−1.4	0.7
0% vs. REF	1.5	0.91	−0.3	3.3	1.7	0.93	0.07	−0.1	3.6
0% vs. FP	1.9	0.92	0.1	3.8	2.1	0.93	0.03	0.2	3.9
0% vs. 100%	2.2	0.91	0.3	4.0	2.3	0.93	0.02	0.4	4.2
FP vs. 100%	0.2	0.53	−0.8	1.3	0.3	0.53	0.64	−0.8	1.3

15-5.2.2.e. Torrent salamanders

Where detected, torrent salamander density (animals/30 m) when including obstructed reaches and adjusted for detection ranged from 0.1 to 99.9 (**Appendix Table 15-D-3**). Estimates of stream network-wide abundance extrapolated from our 30-m density estimates ranged from 2 to 2,992 individuals. Had we not accounted for detection, our stream network-wide abundance estimates would have ranged from 1 to 2,834 individuals. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of torrent salamander density (**Table 15-21** and **Figure 15-14**). The P-value for the period × treatment contrast was <0.01, indicating that the amount of change in torrent salamander density depended on treatment. We estimated the change in the 0% treatment to be different from the changes in the reference (P <0.001), 100% (P <0.01) and FP (P <0.001) treatments (**Table 15-22**).

Where detected, torrent salamander density (animals/30 m) when excluding obstructed reaches and adjusted for detection ranged from 0.1 to 40.6 (**Appendix Table 15-D-3**). Estimates of stream network-wide abundance extrapolated from our 30-m density estimates ranged from 2 to 2,323 individuals. Had we not accounted for detection, our stream network-wide abundance estimates would have ranged from 1 to 676 individuals. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of torrent salamander density (**Table 15-21** and **Figure 15-14**). The P-value for the period × treatment contrast was 0.29, indicating that the amount of change in torrent salamander linear density did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and buffer treatments (**Table 15-22**). There was a difference in the significance of one of the pairwise contrasts dependent on whether we included or excluded obstructed reaches in our analysis. When we included obstructed reaches in the analysis, the change in the 0% treatment differed from the changes estimated for the other buffer treatments and the reference. However, when we excluded obstructed reaches from the analysis, there was no difference in the changes estimated among any of the treatments, including the reference.

Table 15-21. The within-treatment change for the natural log (ln) of stream network-wide torrent salamander density (animals/30 m), i.e., [ln(post) − ln(pre)], and 95% confidence intervals (CI) estimated with light-touch sampling adjusted for detection, including and excluding obstructed reaches. SE is the standard error.

Treatment	**Including Obstructed Reaches**				**Excluding Obstructed Reaches**			
	Estimate	**SE**	**95% CI**		**Estimate**	**SE**	**95% CI**	
			Lower	**Upper**			**Lower**	**Upper**
REF	0.3	0.23	−0.2	0.8	0.3	0.22	−0.1	0.7
100%	0.4	0.25	−0.1	0.9	0.2	0.26	−0.3	0.8
FP	0.1	0.30	−0.5	0.7	−0.4	0.35	−1.1	0.3
0%	1.5	0.26	1.0	2.1	−0.2	0.37	−0.9	0.6

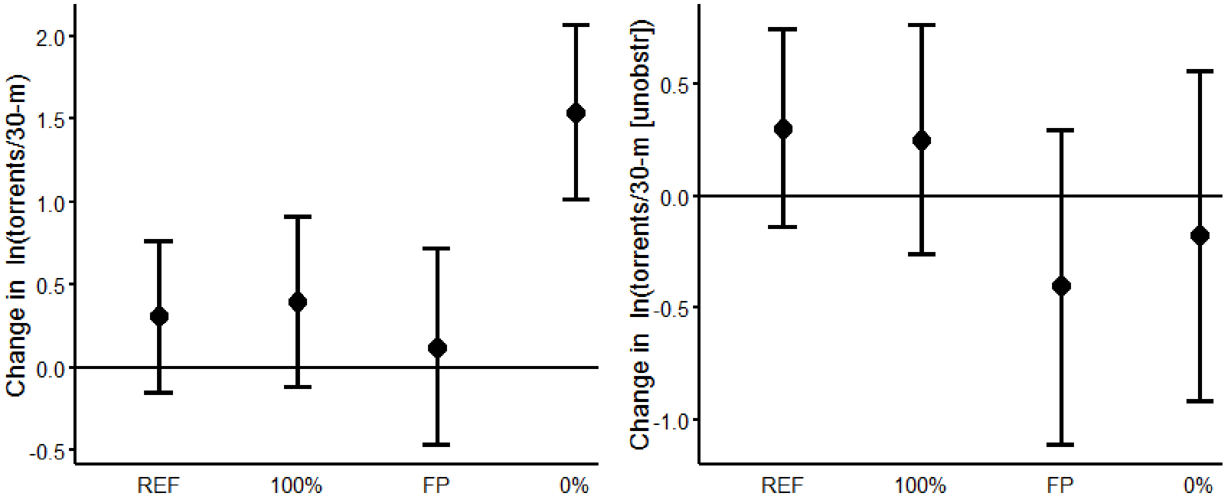


Figure 15-14. The within-treatment change for the natural log (ln) of stream network-wide torrent salamander linear density (animals/30 m), i.e., [ln(post) − ln(pre)], and 95% confidence interval (CI) estimated with light-touch sampling adjusted for detection, including (left panel) and excluding (right panel) obstructed reaches.

Table 15-22. The between-treatment difference in the change in the natural log (ln) of torrent salamander density (animals/m^2), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]), and 95% confidence intervals (CI), including and excluding obstructed reaches. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Including Obstructed Reaches					Excluding Obstructed			
	Est.	SE	P-value	95% CI Lower	95% CI Upper	Est.	SE	95% CI Lower	95% CI Upper
100% vs. REF	0.1	0.3	0.79	−0.6	0.8	−0.1	0.34	−0.7	0.6
FP vs. REF	−0.2	0.3	0.63	−0.9	0.6	−0.7	0.41	−1.5	0.1
0% vs. REF	1.2	0.3	<0.00	0.5	1.9	−0.5	0.43	−1.3	0.4
0% vs. FP	1.4	0.3	<0.00	0.6	2.2	0.2	0.50	−0.8	1.2
0% vs. 100%	1.1	0.3	<0.01	0.4	1.9	−0.4	0.45	−1.3	0.5
FP vs. 100%	−0.3	0.3	0.49	−1.1	0.5	−0.7	0.43	−1.5	0.2

15-5.2.2.f. Giant salamanders

Where detected, giant salamander density (animals/30 m) when including obstructed reaches and adjusted for detection ranged from 0.2 to 39.6 (**Appendix Table 15-D-4**). Estimates of stream network-wide abundance extrapolated from our 30-m density estimates ranged from 5 to 998 individuals. Had we not accounted for detection, our stream network-wide abundance estimates would have ranged from 1 to 586 individuals. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of giant salamander density (animals/30 m; **Table 15-23** and **Figure 15-15**). The P-value for the period × treatment contrast was 0.03, indicating that the amount of change in giant salamander density depended on treatment. We estimated the change in the FP treatment to be different from the change in the reference (P = 0.03), 100% (P = 0.09) and 0% (P <0.01) treatments (**Table 15-24**).

Where detected, giant salamander density (animals/30 m) when excluding obstructed reaches and adjusted for detection ranged from 0.2 to 39.6 (**Appendix Table 15-D-4**). Estimates of stream network-wide abundance extrapolated from our 30-m density estimates ranged from 5 to 998 individuals. Had we not accounted for detection, our stream network-wide abundance estimates would have ranged from 1 to 207 individuals. We estimated the pre- to post-harvest change within treatment for the natural log (ln) of giant salamander density (**Table 15-23** and **Figure 15-15**). The P-value for the period × treatment contrast was <0.01, indicating that the amount of change in giant salamander density depended on treatment. We estimated the change in the FP treatment to be different from the change in the reference (P <0.001), 100% (P <0.01) and 0% (P = 0.02) treatments (**Table 15-24**). The pairwise comparisons that we identified as differing significantly were the same regardless of whether obstructed reaches were included or excluded from the analysis.

Table 15-23. The within-treatment change for the natural log (ln) of stream network-wide giant salamander density (animals/30 m), i.e., [ln(post) − ln(pre)], and 95% confidence intervals (CI) estimated with light-touch sampling adjusted for detection, including and excluding obstructed reaches. SE = standard error.

Treatment	Including Obstructed Reaches				Excluding Obstructed Reaches			
	Estimate	SE	95% CI		Estimate	SE	95% CI	
			Lower	Upper			Lower	Upper
REF	0.2	0.27	−0.3	0.7	0.2	0.22	−0.2	0.6
100%	0.0	0.32	−0.7	0.6	−0.1	0.27	−0.7	0.4
FP	−0.8	0.34	−1.5	−0.1	−1.6	0.41	−2.4	−0.7
0%	0.6	0.30	0.0	1.2	−0.2	0.31	−0.9	0.4

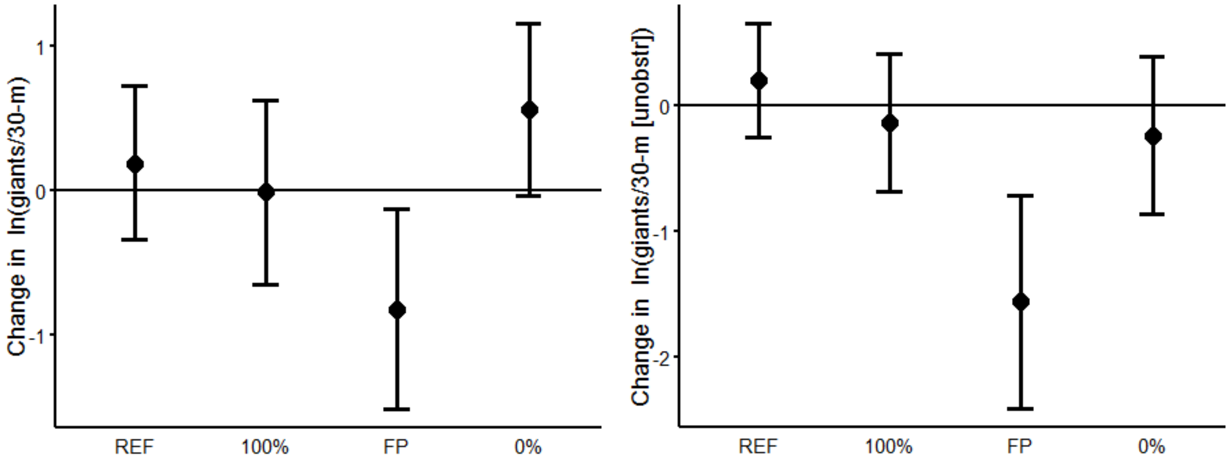


Figure 15-15. The within-treatment change for the natural log (ln) of stream network-wide giant salamander density (animals/30 m), i.e., [ln(post) − ln(pre)], and 95% confidence interval (CI) estimated with light-touch sampling adjusted for detection, including (left panel) and excluding (right panel) obstructed reaches.

Table 15-24. The between-treatment difference in the change in the natural log (ln) of stream network-wide giant salamander linear density (animals/m^2), i.e., (ln[TrtA(post)] − ln[TrtA(pre)]) − (ln[TrtB(post)] − ln[TrtB(pre)]), and 95% confidence intervals (CI), including and excluding obstructed reaches. TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Including Obstructed Reaches					Excluding Obstructed Reaches				
	Estimate	SE	P-value	95% CI		Estimate	SE	P-value	95% CI	
				Lower	Upper				Lower	Upper
100% vs. REF	−0.2	0.42	0.63	−1.0	0.6	−0.3	0.35	0.34	−1.0	0.4
FP vs. REF	−1.0	0.43	0.03	−1.9	−0.1	−1.7	0.47	<0.001	−2.7	−0.8
0% vs. REF	0.4	0.40	0.36	−0.4	1.2	−0.4	0.38	0.25	−1.2	0.3
0% vs. FP	1.4	0.45	<0.01	0.5	2.3	1.3	0.52	0.02	0.3	2.3
0% vs. 100%	0.6	0.44	0.20	−0.3	1.4	−0.1	0.41	0.8	−0.9	0.7
FP vs. 100%	−0.8	0.47	0.09	−1.7	0.1	−1.4	0.50	<0.01	−2.4	−0.4

15-5.2.3. Summary of Density Results

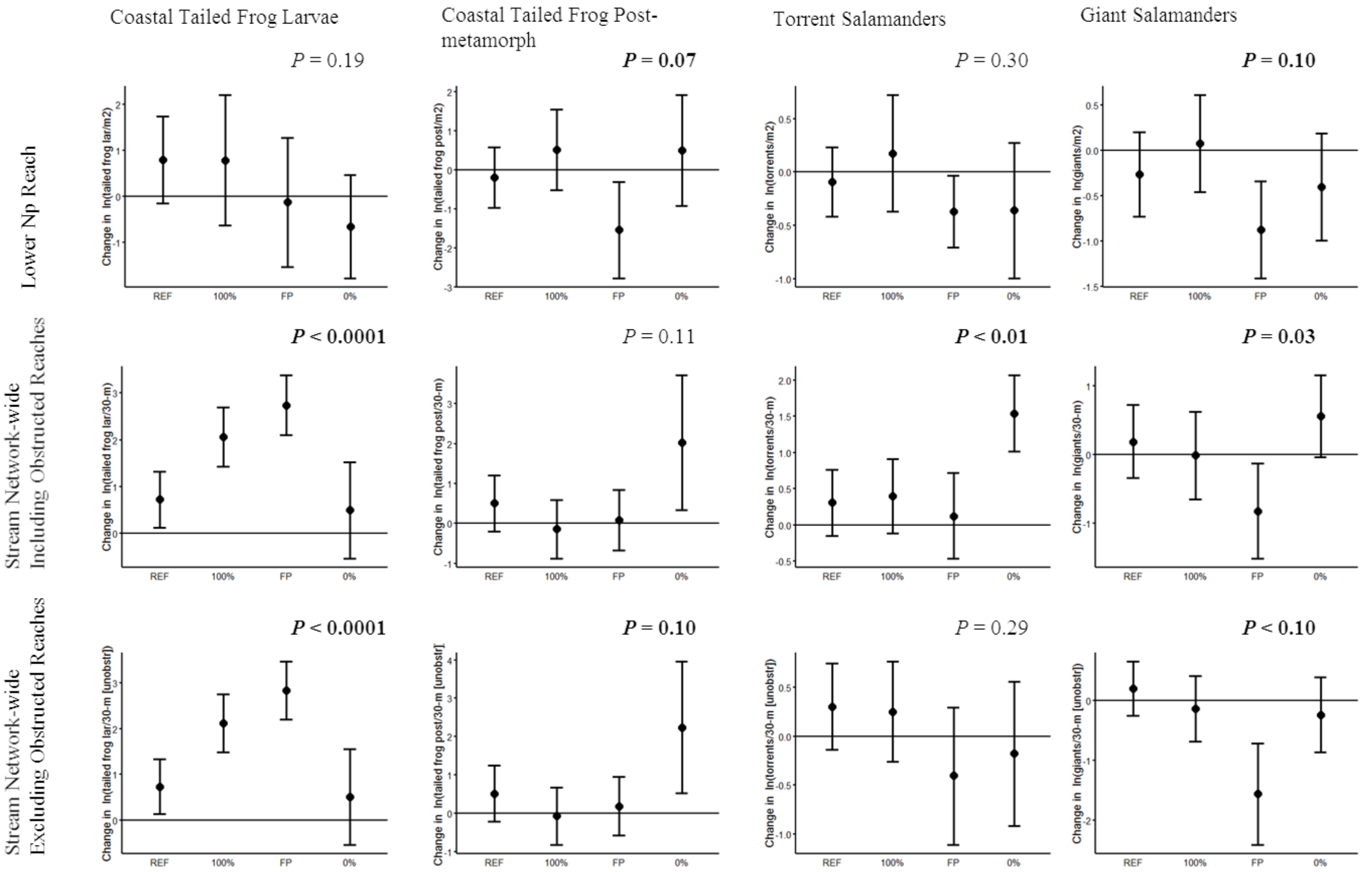


Figure 15-16. Within-treatment change for the natural log (ln) of Coastal Tailed Frog larvae and post-metamorphs, and torrent and giant salamander density, i.e., [ln(post) − ln(pre)], and 95% confidence interval (CI) for: (1) **Lower Np reach** (animals/m^2), rubble-rouse sampling approximately 200-m stream reach upstream from F/N break); (2) **Stream network-wide excluding obstructed reaches** (animals/30 m), systematic light-touch sampling, adjusted for the probability of detection; and (3) **Stream network-wide including obstructed reaches** (animals/30 m) adjusted for the probability of detection. P-values are treatment × period interaction term, values less than or equal to 0.1 are bolded.

15-5.3. BODY CONDITION

15-5.3.1. Coastal Tailed Frog—larvae

Where detected, average larval tailed frog SMI ranged from 0.64 to 2.74 in the pre- and 0.53 to 2.49 in the post-harvest period. The pre- to post-harvest change within treatment ranged from −0.91 to 0.65 (**Table 15-25** and **Figure 15-17**). The P-value for the period × treatment contrast was 0.19, indicating that the amount of change in larval tailed frog SMI did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and buffer treatments (**Table 15-26**).

Table 15-25. The within-treatment change (post − pre) and 95% confidence intervals (CI) for larval Coastal Tailed Frog scaled mass index (SMI). SE is the standard error.

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	−0.11	0.28	−0.76	0.54
100%	−0.91	0.49	−2.03	0.21
FP	−0.41	0.49	−1.53	0.71
0%	0.65	0.44	−0.37	1.68

Table 15-26. The between-treatment difference in the change ([TrtA(post − pre)] − [TrtB(post − pre)]) and 95% confidence intervals (CI) for larval Coastal Tailed Frog scaled mass index (SMI). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Estimate	SE	95% CI	
			Lower	Upper
100% vs. REF	−0.80	0.56	−2.10	0.49
FP vs. REF	−0.30	0.56	−1.60	0.99
0% vs. REF	0.76	0.52	−0.45	1.97
0% vs. FP	1.07	0.66	−0.45	2.58
0% vs. 100%	1.57	0.66	0.05	3.08
FP vs. 100%	0.50	0.69	−1.08	2.09

15-5.3.2. Coastal Tailed Frog—post-metamorphs

We did not detect Coastal Tailed Frog post-metamorphs in three study sites (CASC-100%, CASC-FP and CASC-0%) and no pre-harvest estimates of SMI existed for five sites (OLYM-100%, OLYM-FP, WIL2-100%, WIL2-0% and WIL3-100%). Numbers of tailed frogs for which SMI were calculated for a site and period ranged from 1 to 15 individuals (mean = 4). We did not conduct a formal analysis because we lacked estimates for some sites and periods and had small sample sizes at the remaining sites.

15-5.3.3. Torrent salamanders

Where detected, average torrent salamander SMI ranged from 0.61 to 1.02 in the pre- and 0.25 to 1.07 in the post-harvest period. The pre- to post-harvest change within treatment ranged from −0.23 to −0.03 (**Table 15-27** and **Figure 15-17**). The P-value for the period × treatment contrast was 0.27, indicating that the amount of change in torrent salamander SMI did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and buffer treatments (**Table 15-28**).

Table 15-27. The within-treatment change (post − pre) and 95% confidence intervals (CI) for torrent salamander scaled mass index (SMI). SE is the standard error.

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	−0.18	0.06	−0.32	−0.05
100%	−0.08	0.06	−0.22	0.06
FP	−0.23	0.09	−0.42	−0.03
0%	−0.03	0.07	−0.19	0.13

Table 15-28. The between-treatment difference in the change ([TrtA(post–pre)] − [TrtB(post − pre)]) and 95% confidence intervals (CI) for torrent salamander scaled mass index (SMI). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Estimate	SE	95% CI	
			Lower	Upper
100% vs. REF	0.11	0.09	−0.09	0.30
FP vs. REF	−0.04	0.11	−0.28	0.19
0% vs. REF	0.15	0.09	−0.06	0.36
0% vs. FP	0.20	0.11	−0.06	0.45
0% vs. 100%	0.04	0.09	−0.17	0.26
FP vs. 100%	−0.15	0.11	−0.39	0.09

15-5.3.4. Giant salamanders

Where detected, average giant salamander SMI ranged from 1.61 to 2.72 in the pre- and 1.37 to 2.23 in the post-harvest period. The pre- to post-harvest change within treatment ranged from −0.33 to 0.24 (**Table 15-29** and **Figure 15-17**). The P-value for the period × treatment contrast was 0.97, indicating that the amount of change in giant salamander SMI did not depend on treatment. We estimated the difference in the change for each of the pairwise contrasts among the six combinations among references and buffer treatments (**Table 15-30**).

Table 15-29. The within-treatment change (post − pre) and 95% confidence intervals (CI) for giant salamander scaled mass index (SMI). SE is the standard error.

Treatment	Estimate	SE	95% CI	
			Lower	Upper
REF	−0.24	0.11	−0.47	−0.01
100%	−0.33	0.13	−0.61	−0.04
FP	−0.26	0.15	−0.59	0.07
0%	−0.28	0.13	−0.56	0.01

Table 15-30. The between-treatment difference in the change ([TrtA(post − pre)] − [TrtB(post − pre)]) and 95% confidence intervals (CI) for giant salamander scaled mass index (SMI). TrtA is the first treatment listed in each paired comparison and, for consistency, is the treatment with the largest harvest effect (fewer trees remaining in the RMZ buffer). SE is the standard error.

Contrast	Estimate	SE	95% CI	
			Lower	Upper
100% vs. REF	−0.09	0.17	−0.45	0.28
FP vs. REF	−0.02	0.19	−0.42	0.38
0% vs. REF	−0.04	0.17	−0.41	0.33
0% vs. FP	−0.02	0.20	−0.45	0.42
0% vs. 100%	0.05	0.19	−0.36	0.45
FP vs. 100%	0.07	0.20	−0.37	0.50

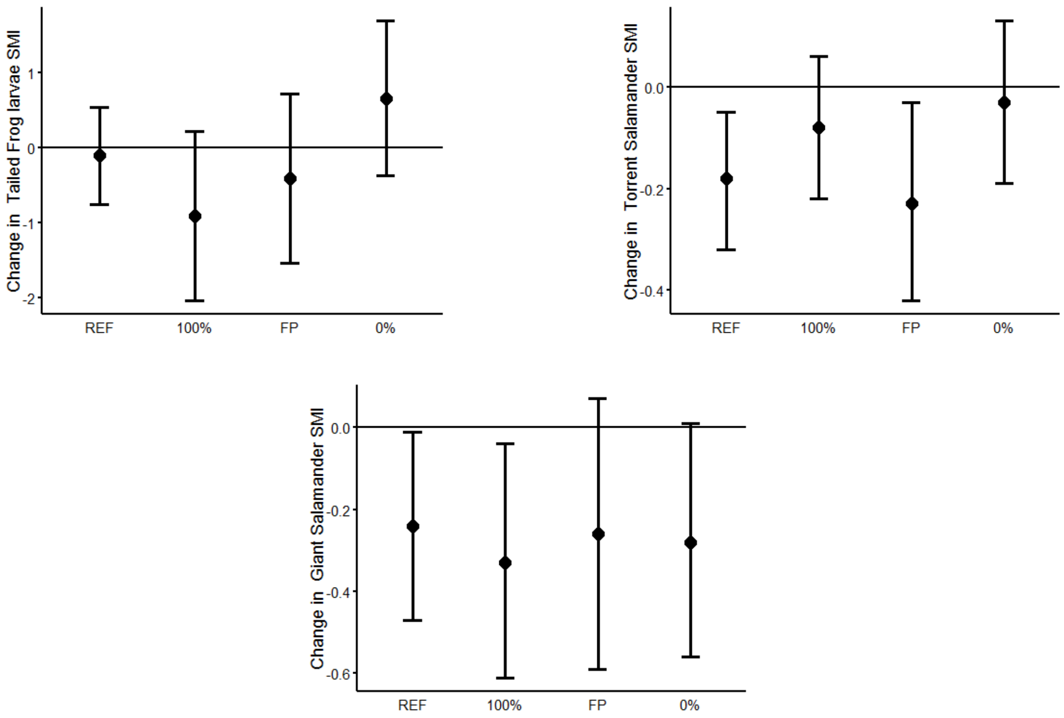


Figure 15-17. The within-treatment change (post − pre) in scaled mass index (SMI) and 95% confidence interval (CI) for larval Coastal Tailed Frog (top left panel), torrent salamander (top right panel), and giant salamander (bottom panel).

15-6. DISCUSSION

15-6.1. OCCUPANCY

Some early observations (Noble and Putnam 1931) and retrospective studies (Bisson *et al.* 2002; Raphael *et al.* 2002) suggested that stream-associated amphibians might be locally extirpated when overstory canopy is removed via timber harvest. In our study, Olympic Torrent Salamander went from detected in the pre-harvest to not detected in the post-harvest period in one site (in the OLYM-FP), but that species was represented by only six individuals that we encountered in only one of three pre-harvest sample years. We do not believe that this population was blinking in and out on an annual basis, but rather that low detection probability coupled with small population size (i.e., few individuals available for detection) resulted in detection of occupancy in some years while the population went undetected in other years, even with rigorous sampling. Conversely, Coastal Tailed frog went from not detected in the pre-harvest to detected in the post-harvest period in the Olympic 100% treatment when we found two post-metamorphic individuals in the final study year. While the light-touch method has been demonstrated to be effective for sampling stream-associated amphibians (Lowe and Bolger 2002; Hayes *et al.* 2006; Quinn *et al.* 2007; Kroll *et al.* 2008; Kroll *et al.* 2010), failure to detect a species when present can be an issue when the population size is small or density is low, or when individuals are difficult to sample and thus detect (Gu and Swihart 2004). We made more than 10,000 amphibian observations throughout the study period, 97% of which were focal species, suggesting that both the method and sampling effort were adequate for detecting focal species. However, detection probability may be strongly associated with density (Royle and Nichols 2003; MacKenzie *et al.* 2005) and failing to allow for the possibility that focal species were present, but undetected, could lead to biased estimates of site occupancy (MacKenzie *et al.* 2003). For these reasons, we interpret changes in detection of occupancy with caution. We conclude no evidence of local extirpation along headwater streams in the two-year period immediately following timber harvest, regardless of the degree of protection afforded to the stream by the riparian buffer. However, we cannot address the long-term impacts of timber harvest removal on focal stream-associated amphibians, especially as it relates to successful and continued reproduction.

15-6.2. DENSITY

Our density results differed depending upon the methodology used, in particular, density estimates for the lower Np reach versus those based on stream network-wide abundance estimates normalized to 30 m and accounting for detection probability. Below we discuss specific differences in estimates obtained between methodologies for each of the focal amphibian taxa. When we designed our study, rubble-rouse sampling was considered the most effective and efficient means for estimating density of stream-associated amphibians. However, with the confirmation that Royle's (2004) *N*-mixture models were effective for estimating detection probabilities of our focal amphibians (McIntyre *et al.* 2012), we added it while continuing with rubble-rouse sampling in the lower Np reach. We focused our rubble-rouse sampling on the lower Np reach (i.e., 200 m directly upstream from the F/N break) for two reasons: (1) Coastal Tailed Frog larvae are typically encountered in greater abundances in these downstream-most reaches and post-metamorphs move annually down- and upstream for the breeding and non-breeding seasons (Hayes *et al.* 2006); and (2) because any effects of timber harvest on habitat may accumulate in the downstream reaches where the cumulative upstream harvested area is at its greatest. However, focal amphibians may be distributed differentially throughout the non-fish-bearing stream network. Hayes and colleagues (2006) hypothesized that adult female Coastal Tailed Frogs breed in upstream areas in the fall, move downstream to oviposit and return upstream for the non-breeding season, with larvae located further downstream than post-metamorphs throughout the majority of the year. Others have found torrent salamanders to be associated with the uppermost extent of headwaters (Olson and Weaver 2007), under flow conditions favorable for creating the riffles and seeps with which they are so commonly associated (Nussbaum and Tait 1977; Nussbaum *et al.* 1983). We evaluated whether these spatial patterns were also expressed in our own data (see 15-5.1 Occupancy), and found that they were. Because our study stream networks varied substantially in total length (see Chapter 3 − *Management Prescriptions*), the application of rubble-rouse sampling to a standardized length at the bottom of Type N water reaches across all study sites may have added unwanted variability. Further, changes in the distributions of focal amphibians throughout the stream network, including those that may have been in response to treatment, may have gone undetected with our rubble-rouse sampling design. This could have affected density estimates in the lower Np reach, especially in our FP treatment sites, where riparian buffers were implemented differentially between the lower Np reach and reaches farther up in the stream network.

The potential exists that the distribution of amphibians throughout the entire Type Np network may have changed in the post-harvest period, either as a result of differences in the timing of annual migrations up and down the stream, or the locations of protective riparian buffers. Due to the uneven distribution of riparian buffers across sites, and because riparian buffers were disproportionately located in the lower Np reach, we saw a potential for confounding between amphibian distribution and buffer treatment. Consequently, we rely on our results from stream network-wide estimates of amphibian density to draw conclusions since the latter rely on estimates of abundance throughout the entire Type Np stream network that are not affected by changes in distribution. Uncertainty in estimates that control for imperfect detection have less bias than classic approaches (e.g., counts) that do not account for detection (Mazerolle *et al.* 2007). Many have reported on the benefits of including detection probabilities when estimating population parameters (Nichols and Pollock 1983; Pollock *et al.* 1990; MacKenzie *et al.* 2002; Guillera-Arroita *et al.* 2014), concluding that, even when assumptions are violated, estimation methods that account for the probability of detection generally produce estimates with less bias than those founded on methods that do not account for detection.

Few differences were apparent in results obtained from our stream network-wide density estimates when we included and excluded obstructed reaches. In fact, the significance in the between-treatment differences only varied for the torrent salamander in the 0% treatment. When we included obstructed reaches in our analysis, we detected a post-harvest increase in torrent salamander density in the 0% treatment that was 3.3 (95% CI: 1.7–6.7) times greater than for the reference. When we excluded obstructed reaches from our analysis, we detected no differences in the pre- to post-harvest change between treatments. There are two possible explanations for this difference: (1) density of torrent salamanders was greater in obstructed reaches; or (2) we only detected a proportion of the torrent salamander population with our light-touch sampling; these possibilities are not mutually exclusive. Stream-associated amphibians are known to move up and down between the surface and deeply buried in the substrate on a seasonal basis. The light-touch method, which we used for our stream network-wide estimates and adjusted based on estimates of detection, samples surface-active animals; conversely, rubble-rouse sampling, which we used to sample obstructed reaches, was developed to survey the entire population, regardless of an individual's location in the stream substrate (**Figure 15-18**). Even when accounting for detection, if deeply buried animals were never available to be sampled with light-touch (i.e., they were never surface active during the sample period), estimates of density obtained using only light-touch could have differed from those that included rubble-rouse sampling. The differing results we obtained including and excluding animals detected with rubble-rouse sampling in obstructed reaches may reflect the fact that torrent salamanders were not moving up and down in the substrate during our sample period. If salamanders located deeper in the substrate were unavailable for sampling with light-touch, our detection probability estimates would be inflated, resulting in adjusted density estimates that are biased low. Of note, we found only the torrent salamander in relatively high numbers in obstructed reaches, which may explain why results obtained when including and excluding obstructed reaches differed only for this taxa. For simplicity, we will focus our back-transformations and discussion on the results obtained when excluding obstructed reaches, though we will also include a back-transformed estimate for each taxon and treatment for which the estimated difference in the change between two treatments differed between the two methods.

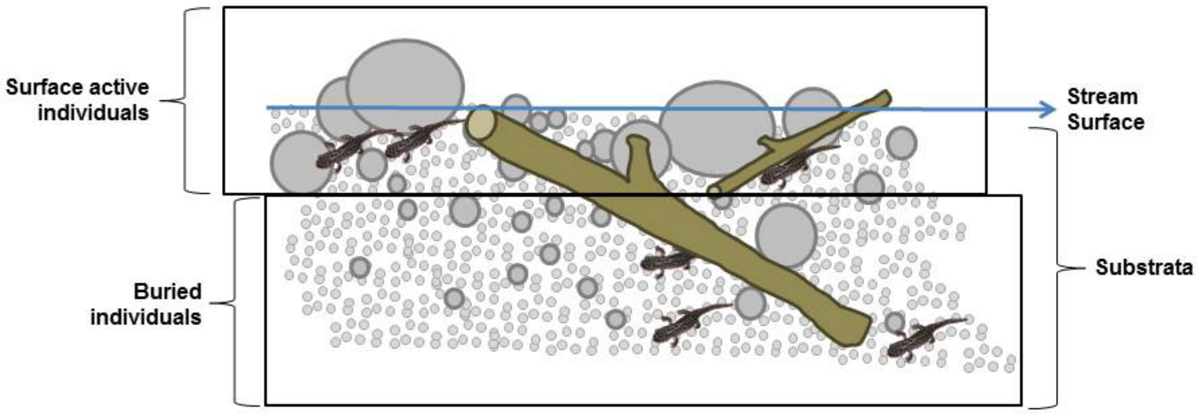


Figure 15-18. A cross-section of a headwater stream depicting the stream surface and subsurface substrates (substrata), and theoretical distribution of amphibians between those that are surface active and thus available for detection with light-touch sampling at a given moment, versus those that are buried deep in the substrata and thus not available for detection at that same moment.

Basin-wide density estimates for all species, whether adjusted or unadjusted for detection probability, showed wide ranges across basins and years (**Appendix 15-D**). These ranges reflect both process variation in density (e.g., due to treatment effects, temporal variation, and spatial variation) and variation due to sampling. The former is accounted for in the BACI mean model, while the latter is incorporated into the experimental error term. It is important to note that the basin-wide density estimates are averaged across hundreds and in some cases thousands of meters of stream length. This level of extensive survey effort reduces sampling variation and moderates the impact of individual reaches with large counts. Thus, ranges in basin-year level density values are primarily a reflection of cross-landscape spatial variation, temporal variation, treatment variation and their interactions, as well as unmodeled heterogeneity in the detection process.

15-6.2.1. Coastal Tailed Frog

Results for Coastal Tailed Frog densities were contradictory depending on whether we considered the lower Np or stream network-wide density. Specifically, while we lacked clear evidence that the amount of change in Coastal Tailed Frog density in the lower Np reach depended on treatment, we had strong evidence of an increase in the 100% and FP treatments that differed from the change in the reference and 0% treatments when considering stream network-wide density. The pre- to post-harvest change within treatment for larval tailed frog density in the lower Np reach was 60% (95% CI: 93% decrease - 2.2 times greater) and 77% (95% CI: 95% decrease - 1.0 times greater) lower in the FP and 0% treatments relative to the reference, respectively. However, there was no clear evidence of a significant difference among treatments, including the reference. We detected a post-harvest increase in stream network-wide larval tailed frog density in the 100% and FP treatments, excluding obstructed reaches, that was 4.1 (95% CI: 1.6–10.0) and 8.2 (95% CI: 3.3–20.1) times greater, respectively, than for the reference. These increases were significantly greater than the other treatments and were consistent with results including obstructed reaches.

The pre- to post-harvest change in post-metamorphic tailed frog density in the lower Np reach was 74% (95% CI: 94% decrease–1.1 times greater) lower in the FP treatment than the reference, a decrease that was significantly less than other treatments, including the reference. However, we detected a post-harvest increase in stream network-wide post-metamorph tailed frog density in the 0% treatment, excluding obstructed reaches, that was 5.5 (95% CI: 0.9−36.6) times greater than for the reference, an increase that was significantly greater than other treatments. This change was consistent with results including obstructed reaches; however, in the latter there was not a significant difference in the change between treatments.

Our stream network-wide density results differ from the findings and/or conclusions of some studies of Coastal Tailed Frogs. In a study similar to our own, with clearcut harvest treatments applied in a BACI design, Jackson and colleagues (2007) concluded that clearcut timber harvest without riparian buffers appeared to have at least a short-term negative effect on tailed frog populations, despite the fact that differences among their treatments were not statistically significant. However, this study used single-pass sampling to obtain population estimates and neither estimated detectability nor adjusted for it. Several retrospective studies comparing previously harvested with late-seral or old-growth stands have found that tailed frogs were nearly absent from, or markedly less abundant in, stands with a history of timber harvest (Corn and Bury 1989; Ashton *et al.* 2006). Welsh and Lind (2002) found the highest densities of tailed frogs in late seral forests. Dupuis and Steventon (1999) observed a similar trend, but found that the presence of riparian buffers ameliorated the impacts of timber harvest. Similarly, Stoddard and Hayes (2005) found that adult tailed frogs were positively associated with riparian buffers and larval tailed frogs with forested stands greater than 105 years old. These latter studies also did not account for detection, which may explain why results differ from our own. However, another possible explanation for differing results between these and our own is that these studies represent a potential population response over a longer timeframe than our study, which evaluated the response of focal amphibians over only the initial two years post-harvest. One additional explanation is that there is a geographic cline in response: all but one of these studies was conducted farther south than our own, in Oregon or California. Sites located farther to the south are more prone to increases in stream temperatures that may be deleterious to stream-associated amphibians than sites located farther north in Washington and British Columbia.

Conversely, other studies have observed no differences among previously clearcut and older stands. Richardson and Neill (1998) found no difference in the occurrence of tailed frog larvae between clearcut, second-growth and old-growth stands. Likewise, Matsuda and Richardson (2005) found that the numbers of tailed frogs captured were similar between clearcut and mature, second-growth sites (>81 years old). These studies were retrospective, with historical distributions unknown, detection probabilities were not calculated, and they were both conducted in British Columbia. In another retrospective study that did include estimates of detection probability, Kroll and colleagues (2008) found that tailed frog occupancy was positively associated with stand age. In their experimental study, O’Connell and colleagues (2000) found no differences in the densities of Coastal Tailed Frog larvae between clearcut headwater basins with variable width buffers of two size classes and forested reference basins; however, the proportion of the study stream length sampled was extremely limited (5% of total) and detection was not considered. In another experimental study conducted in western Oregon, Olson and colleagues (2014) concluded that upland forest thinning with variable width riparian buffers ranging from 6 to 145 m wide did not result in severe, persistent declines or local extirpations of Coastal Tailed Frog, Coastal Giant Salamander or torrent salamanders in the 10 years following harvest.

We hypothesize that the positive influence of our intermediate treatments (100% and FP) on tailed frog larvae was likely due to a combination of factors influencing reproduction and growth, including decreased canopy and effective shade and subsequent increases in stream temperature. We found a linear relationship between canopy, effective shade and stream temperature with treatment (see Chapter 7 − *Stream Temperature and Cover*). While increased stream temperature can result in an increase in instream periphyton production (Kiffney *et al.* 2003), we observed no difference in instream periphyton biomass between treatments (see Chapter 13 − *Biofilm and Periphyton*). However, intermediate treatments could have resulted in patches of increased diatom production that went undetected in our stream network-wide analysis of chlorophyll *a* and ash-free dry mass (AFDM). Larval tailed frogs are grazers, scraping periphyton from rocks, and so an increase in periphyton production could have positively influenced tailed frog larvae survival and growth. While increased tailed frog larvae could have also resulted in increased grazing, which has the potential to mask the increase in periphyton production, we lacked evidence of a grazing effect on periphyton standing stock (see Chapter 13 − *Biofilm and Periphyton*). Finally, all three of our giant salamander analyses concluded a negative effect on (i.e., decrease in) giant salamander density in the FP treatment, a consistent result that is difficult to disregard or explain. Since giant salamanders are known to prey on larval tailed frogs (among other things; Nussbaum *et al.* 1983), a decrease in giant salamanders at FP treatment sites could have contributed to the positive increase observed for tailed frog larvae. Alternatively, Feminella and Hawkins (1994) found that tailed frog larvae spent a greater amount of time hidden in crevices when predatory giant salamanders were present, so a decrease in giant salamanders in FP treatment sites could have resulted in increased tailed frog larvae activity. It is possible that increased activity resulted in what appeared to be an increase in abundance, if animals that were buried in the substrate, and thus not detected by light-touch sampling, spent more time active on the surface and available for detection.

Considering the potential mechanisms for the increased densities observed in the 100% and FP treatments, the lack of change in the 0% treatment may seem difficult to explain. Light saturation for algal production occurs at less than full sunlight (Murphy 1998), which may explain the lack of increase in larval density in our 0% treatment. Further, the increased stream temperature in the 0% treatment may have had a negative impact on tailed frog larvae, which exhibit thermo-regulatory behavior (de Vlaming and Bury 1970), and could have migrated downstream into the fish-bearing reach in an attempt to avoid higher stream temperatures. In fact, we observed an increase in the daily maximum stream temperature in the 0% treatment that averaged 2.1°C for the July–August period during the first post-harvest year. This could translate to post-harvest temperatures as high as 15.2°C for some 0% treatment sites (see Chapter 7 − *Stream Temperature and Cover*). Both of these factors could explain the lack of increase in larval density in our 0% treatment. Similarly, in another nearly complete Forests and Fish Adaptive Management study, the Shade Study, investigators observed a tendency for a greater increase in the number of tailed frogs in treatments that maintained an intermediate level of shade (i.e., 70% shade retention) compared with an untreated reference and no (0%) and low (30%) shade retention treatments. However, this increase was not statistically different from the changes observed in the other treatments (J. MacCracken, M. Hayes, personal communication).

While we combined tailed frog larvae and individuals in the process of metamorphosis in our analyses of tailed frog larvae, we did observe twice as many metamorphosing tailed frog larvae during the post-harvest period compared to the pre-harvest period. We cannot say what factors may have contributed to these changes (e.g., temperature, light); however, we observed this increased occurrence of metamorphs across all treatments, including the reference. Studies have shown that tailed frog occupancy increases with basin order and that adults move seasonally up and downstream, probably for breeding (Hayes *et al.* 2006). We also observed unequal distribution of tailed frogs within the Type Np basin, with 61% of tailed frog detections in the lower Np reach (within 200 m of the F/N break) where only 34% of our sampling effort was focused. We recognize that differences in the pre- to post-harvest density changes estimated for the lower Np reach versus stream network-wide could be a result of the different spatial scales at which sampling was conducted, especially if there was movement into or out of the lower Np reach that differed among years. Further, if the environmental conditions or timing of the conditions that influenced metamorphosis differed among treatments, or among buffered and unbuffered reaches, then there is reason to hypothesize that they may have affected tailed frog distribution during time of sampling, especially since the riparian buffer in the FP treatment are focused in the lower Np reach.

15-6.2.2. Torrent Salamanders

Results obtained for torrent salamanders differed dependent on the sampling methodology and analysis. We did not detect a significant difference in the pre- to post-harvest change between treatments for torrent salamander density in the lower Np reach or stream network-wide when excluding obstructed reaches. However, we detected a post-harvest increase in stream network-wide torrent salamander density in the 0% treatment, including obstructed reaches that was 3.3 (95% CI: 1.7–6.7) times greater than for the reference. This increase was significantly greater than the other treatments. Unfortunately, at this point we cannot differentiate between the possibility that there was a true difference in torrent salamander density dependent on whether we included and excluded obstructed reaches and the possibility that the difference was a sampling artifact. However, overall we lack evidence of a negative effect of buffer treatment on torrent salamanders, regardless of which results are used from which to draw inference.

These findings are consistent with those of Jackson and colleagues (2007) and Olson and colleagues (2014), who concluded that torrent salamanders were not greatly affected by timber harvest or upland forest thinning, respectively. Russell and colleagues (2004) detected no relationship between torrent salamander occupancy or relative abundance and stand age. Conversely, some have reported lower occurrence or densities of torrent salamanders in managed versus old-growth stands (Corn and Bury 1989; Bury *et al.* 1991; Russell *et al.* 2005). Still others have found that torrent salamander numbers and occupancy were greatest in mid-rotation stands (Steele *et al.* 2003; Kroll *et al.* 2008) and Vesely and McComb (2002), Stoddard and Hayes (2005), and Pollett and colleagues (2010) found that the impacts of forest harvest appeared to be ameliorated by the presence of a riparian buffer.

Though conclusions regarding the importance of prior management history have differed substantially, the implications of sedimentation seem consistent. Some previous research has indicated that torrent salamanders are negatively correlated with fine sediments (i.e., mud, silt and sand; Diller and Wallace 1996; Welsh and Lind 1996; Welsh and Ollivier 1998). Corn and Bury (1989) found this correlation only in previously logged stands, concluding a negative effect from increased sedimentation resulting from harvest. While we did detect more sand and finer substrates in our treated sites in the post-harvest period, we found no evidence of a significant difference between treatments (see Chapter 11 − *Stream Channel Characteristics*); further, we found no evidence of an increase in sediment inputs or exports among treatments (see Chapter 10 − *Sediment Processes*).

15-6.2.3. Giant Salamanders

We observed a pre- to post-harvest decrease in giant salamander density in the FP treatment that differed significantly from the other treatments in all three density comparisons (lower Np density, and stream network-wide density including and excluding obstructed reaches). Density of giant salamanders, excluding obstructed reaches, decreased by 82% (95% CI: 55–93%) in the FP treatment relative to the reference, a decrease that was significantly greater than all other treatments, including the reference. Giant salamanders are thought to tolerate a relatively wide variety of habitats (Nussbaum *et al.* 1983; Bury and Corn 1988; Leuthold *et al.* 2012). Correlations of giant salamander density and occupancy with stand age or timber harvest history are quite variable. For example, some have concluded a lack of correlation (Bury *et al.* 1991; Leuthold *et al.* 2012), while others have concluded increased biomass of giant salamanders in streams adjacent to clearcuts (Murphy and Hall 1981), increased relative abundance in late-seral forested streams (Ashton *et al.* 2006), and a positive association between occupancy and stand age (Kroll *et al.* 2008). Corn and Bury (1989) and Murphy and Hall (1981) found that the response of giant salamanders to timber harvest depended on stream gradient and hypothesized that the negative relationship between giant salamanders and timber harvest in low gradient channels was likely due to increased sediment that reduced habitat complexity and cover.

Giant salamanders are tolerant to a broad range of temperatures (Adams and Bury 2002), which may make them more resilient to potential changes in stream temperature resulting from timber harvest. Further, canopy removal may increase food resources, potentially enhancing populations of invertebrate prey (Murphy and Hall 1981; Bury and Corn 1988). In their experimental study, Jackson and colleagues (2007) found that giant salamanders were very sensitive to the immediate impacts of harvest, but that recovery of populations in clearcut streams began as early as two years after harvest. 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 after harvest, though upland harvest in this study was timber thinning rather than clearcut harvest. Potential negative impacts may be short-lived (e.g., two years or less), possibly due to recolonization from source populations in downstream fish-bearing reaches, as hypothesized by Jackson and colleagues (2007).

Though the observed decline in giant salamander density in the FP treatment was consistent across all analyses, we find this relationship difficult to explain, especially considering the lack of treatment effect in the 0% treatment. However, since our sampling methodology was limited to instream sampling, we were not able to account for terrestrial individuals in our estimates of giant salamander density or abundance. Changes in riparian conditions may have influenced the proportion of individuals that transformed and became terrestrial, or the proportion of terrestrial individuals that stayed in- or near-stream. For example, Matsuda and Richardson (2005) found an increased incidence of overland movement of post-metamorphic Coastal Giant Salamanders in old-growth compared with clearcut sites. Further, it has been documented that Coastal Giant Salamanders more frequently metamorphose into terrestrial adults, whereas Cope's Giant Salamanders are more likely to become neotenic and stay instream (Nussbaum 1970, 1976). Theoretically, this could result in what appears to be a differential response between sites composed predominately of one species or the other, especially if treatments resulted in more or less metamorphosis and migration away from, or concentration in, the stream channel. However, we think it is important to consider the fact that very few post-metamorphic giant salamanders were encountered (<1%), and that the total proportion of post-metamorphic giant salamanders encountered did not vary substantially between treatments or periods. Finally, for those that we had species identified via genetic tissue analysis, we actually detected a greater incidence of Cope's (n = 9) than Coastal post-metamorphs (n = 5).

We did not observe increases in sediment inputs or exports in any treatment (see Chapter 10 − *Sediment Processes*), and though an increasing trend in the amount of fines and sand with treatment existed, we found no significant difference among treatments, including the reference (see Chapter 11 − *Stream Channel Characteristics*). Coe and colleagues (2009) found that invertebrate densities were higher on wood than cobble substrates in an experimental study evaluating the effects of wood additions in large river drainages in Washington. The 0% treatment had more instream wood in the form of logging slash and blowdown than the FP treatment (see Chapter 6 − *Wood Recruitment and Loading*). Though we did not observe an overall increase in macroinvertebrate numbers or biomass export in any treatment, macroinvertebrate abundance may have been greater in areas of heavy wood loading, which were greatest in the 0% treatment. Our result suggests that giant salamander populations are limited by multiple factors, that may be acting individually or in combination, through complex interactions, a result observed in other similar studies of aquatic vertebrate populations (Kiffney and Roni 2007).

15-6.2.4. Obstructed Reaches

We detected all three focal genera in obstructed reaches, with some detected in substantial numbers. We observed torrent salamanders in the greatest densities, with as many as 60 individuals encountered in one 3-m obstructed plot. Further, we detected one Coastal Tailed Frog, one Columbia Torrent Salamander, and two giant salamander egg masses in obstructed plots (mean = 0.02/m), indicating that reproduction continued to occur in these reaches. In comparison, seven Coastal Tailed Frog and four giant salamander egg masses were detected during rubble-rouse sampling in the lower Np reach (mean = 0.008/m). It is evident that focal amphibian densities generally did not decline nor did amphibians cease breeding activity in stream reaches heavily loaded by instream wood in the two years following harvest.

Similar to the findings of Jackson and colleagues (2007), we found that stream reaches lacking a riparian buffer had the greatest increase in instream wood (see Chapter 6 − *Wood Recruitment and Loading*) comprised of a matrix of large and small branches, needles and fine sediment. While the effects of timber harvest on stream-associated genera have been mixed, studies consistently show that sedimentation in the form of increased fines is negatively associated with stream-associated amphibian occurrence and density (Hawkins *et al.* 1983; Diller and Wallace 1996, 1999; Welsh and Lind 1996; Welsh and Ollivier 1998; Dupuis and Steventon 1999). Perhaps this consistent association has resulted in the assumption that amphibians are absent in areas highly loaded with slash and fine sediment, though few have actually attempted to quantify amphibians in these areas due to the difficulty in applying traditional sampling methods. Although Jackson and colleagues (2007) excavated through instream slash to access the stream and evaluate amphibian density, they conducted light-touch surveys in those reaches whereas we conducted rubble-rouse sampling. In addition, they did not include a comparison of amphibian counts between slash- and non-slash-filled stream reaches in their study results.

15-6.3. PROBABILITY OF DETECTION

Detection probabilities frequently vary relative to a number of factors including species (Bailey *et al.* 2004; Price *et al.* 2011; McIntyre *et al.* 2012), life stages (Sagar *et al.* 2007; Price *et al.* 2011), habitat types (Kroll *et al.* 2008), environmental covariates (e.g., stream temperature; McIntyre *et al.* 2012) and sampling methods (Bailey *et al.* 2004; Quinn *et al.* 2007). Further complicating stream-associated amphibian monitoring are their relatively low detection probabilities, which makes drawing reliable inferences about trends challenging (Bailey *et al.* 2004; Mazerolle *et al.* 2007). The resulting uncertainty can confound research and monitoring efforts (Kroll 2009), potentially leading to biased inferences regarding habitat relationship and/or management effects (Bailey *et al.* 2004; MacKenzie 2006), which could potentially contribute to misguided management actions. The use of counts to index abundance are typically biased low (Otis *et al.* 1978) and assuming that patterns based on these data are unbiased can result in erroneous conclusions (Yoccoz *et al.* 2001; Mazerolle *et al.* 2007). Previous research on the impacts of timber harvest on stream-associated amphibians has established that streams can be covered with slash or clouded with sediment after harvest (deMaynadier and Hunter 1995; Jackson *et al.* 2001, 2007), which likely results in violating the assumption of constant detection before and after harvest for some sampling methods (e.g., light touch).

We note that our estimated detection probabilities for some taxa varied by life stage, stream temperature, stream riparian condition, or stream order. For example, giant salamander detection probability for unbuffered first-order streams was approximately 40% greater than for reference first-order streams (0.49 vs. 0.34). In this case, a comparison of raw counts would have underestimated abundance in reference detection plots by approximately 40% compared with unbuffered plots. Using unadjusted counts would have resulted in a bias at a scale that is likely to have been meaningful.

There are important caveats to consider when interpreting the detection probability results. The N-mixture models used to estimate detection probability rely on assumptions of population closure and availability for detection. We designed our sampling methods to minimize the chances of migration during survey, but individual movement between sub-surface strata could not be controlled and would violate the closure assumption. The form of both the abundance and detection models will also affect estimates of detection probability. We chose model forms *a priori*, with a goal to minimize treatment bias in estimates of detection probability. Including design variables in both the abundance and detection portions of the model acts to de-confound treatment impacts on detection and abundance, thereby reducing bias. This approach also implicitly adjusted for chance differences in unmodeled covariates across treatment basins, thereby adjusting for these effects without knowing their source. However, this *a priori* model selection approach cannot guarantee that the resulting models were particularly good for predicting detection. A search for different model terms may have uncovered improved prediction models. Our decision to not conduct a model search was based on our objectives to limit treatment bias, to limit overfitting associated with model selection (Chatfield 1995), and to handle constraints associated with the available data.

An implicit assumption with our approach to adjusting raw counts was that unmodeled factors strongly associated with detection probability were represented in the same proportion of our sampled length as in the full length of the stream networks. Another important assumption was that detection probabilities in 2006 and 2007 were identical to those estimated for 2008. Although detection probabilities were similar for 2009 and 2010, we cannot know if this was true for 2006 - 2008. If detection probabilities differed much between these pre-harvest years, then estimates of treatment differences may have been affected.

As noted in the methods section, our results (i.e., contrast estimates, confidence intervals, and P-values) are based on adjustments using point estimates of detection probability and do not incorporate uncertainty in these estimates. The reason for this approach was the lack of tractable alternatives within the analysis framework used in this study. This approach carries an assumption that uncertainty in the detection probability estimates has a negligible impact on any statistical inference. We assessed this assumption with a post-hoc analysis reported in **Appendix 15-A**. The results of this analysis suggest that ignoring detection probability uncertainty likely had little impact on the reported results for torrent and giant salamanders, but may have substantially under-represented inferential uncertainty for both larval and post-metamorph tailed frog results. That is, confidence intervals and P-values for stream network-wide density treatment differences are likely too small for both larval and post-metamorph tailed frogs. For all species, the post-hoc analysis suggested little impact on contrast point estimates. Consequently, the reader is cautioned to be aware of this limitation when considering potential differences in tailed frog densities reported in the results.

15-6.4. BODY CONDITION

Amphibian body condition in terms of nutrient storage has been correlated with movement (Lowe *et al.* 2006) and survival (Reading 2007; Scott *et al.* 2007), and may be considered a sign of both overall health as well as reproductive success (Moya-Larano *et al.* 2008). In theory, body condition reflects an animal's energy reserves and can be associated with environmental characteristics such as habitat quality and prey availability (Stevenson and Woods 2006; Pope and Matthews 2009). We had no evidence of a treatment effect on body condition for any species, though we were unable to include tailed frog post-metamorphs in our analysis due to a small sample size. However, if injury resulted from buffer treatments or the process of timber removal in riparian areas, we may have introduced bias by removing animals with evidence of injury from our analyses of body condition, if the injuries incurred caused reduced fitness or body condition. We believe the potential for this is slight. The proportion of animals removed from the analyses did not differ substantially between the pre- and post-harvest periods for any genus and we actually had greater numbers of individuals included in the post- than the pre-harvest period for all comparisons. However, the precision of our scales (0.1 g) limited our ability to detect small changes in body condition reflected by individual body weight changes of less than 0.1 g.

Except for the near-complete Shade Study, we are not aware of any other study that uses the SMI to compare amphibian body condition as a function of buffer treatments. However, many studies have detected associations between amphibian body condition and either riparian stand density or the amount of light reaching the stream. MacCracken (2002) found that the mean mass/individual of Columbia Torrent Salamanders was greater in streams where the surrounding tree canopy had been reduced by 30% to 50%; however, this may have been confounded with lower salamander density. Kiffney and colleagues (2004) found that, under experimental conditions where streams had been artificially shaded to create four levels of ambient photosynthetically active radiation (PAR; 2, 10, 22, and 100% of full exposure), Coastal Tailed Frog larvae lost mass at the two lowest light levels, while relative growth rate was seven times greater in the full exposure (100% PAR) than the 22% PAR treatment. In another experiment, Mallory and Richardson (2005) found that relative growth rates of tailed frog larvae were 14% higher in unshaded treatments.

Increased solar flux resulting from forest harvest may result in increased primary production in streams and rivers (Mallory and Richardson 2005), which leads to increased invertebrate and vertebrate production (Murphy 1998). Curiously, while we did observe a post-harvest increase in periphyton AFDM across all treatments, including the reference (see Chapter 13 − *Biofilm and Periphyton*), this did not translate to either increased macroinvertebrate abundance or improvements in amphibian body condition in any treatment.

15-6.5. FORESTRY EFFECTS ON STREAM-ASSOCIATED AMPHIBIANS

Investigations into the effects of timber management on stream-associated amphibians have had contradictory results (Richardson and Béraud 2014). Many studies have concluded a negative effect of forest harvest on Coastal Tailed Frogs, torrent and/or giant salamanders (Corn and Bury 1989; Dupuis and Steventon 1999; Jackson *et al.* 2007) or a positive effect of stand age on the same genera (Corn and Bury 1989; Welsh and Lind 2002; Steele *et al.* 2003; Stoddard and Hayes 2005; Ashton *et al.* 2006; Kroll *et al.* 2008; Pollett *et al.* 2010). Other studies have indicated no impact of forest harvest (Kelsey 1995; Matsuda and Richardson 2005; Jackson *et al.* 2007) or a lack of correlation with stand age (Welsh 1990; Welsh and Lind 1996; Richardson and Neill 1998; Welsh and Lind 2002). These apparently conflicting results may be due in part to the retrospective nature of some studies, unequal distributions of reference versus managed sites across the landscape of interest, latitudinal variance whereby the same or similar species respond differentially dependent on their location within their geographic range, or a lack of accounting for detection in estimates of amphibian abundance (Kroll 2009; Hayes and Quinn 2015). Further, the differing responses between some of these studies and our own could also be the result of the short-term nature of our study. While retrospective studies have limitations, any difference in amphibian occupancy and/or abundance detected between recent clearcuts and older mature forests in these studies may reflect the longer time period since harvest, and amphibian populations investigated in our study may have not yet responded completely to the buffer treatments.

Timber harvest has the potential to alter several aspects of headwater streams, including stream and bank morphology, sediment dynamics, stream temperature, primary productivity, nutrients and flow dynamics. Forest management can affect sources and mobilization of sediment in headwater streams (Hassan *et al.* 2005). Sediment that would otherwise be transferred through the stream network may become trapped in woody slash that resulted from timber harvest (Jackson *et al.* 2001). A number of studies have suggested a negative association between focal amphibian species occupancy and density and the amount of instream fines and sand (Hawkins *et al.* 1983; Diller and Wallace 1996, 1999; Welsh and Ollivier 1998; Dupuis and Steventon 1999; Wilkins and Peterson 2000; Stoddard and Hayes 2005). Several possible causes for negative impacts of fine sediment on stream-associated amphibians have been proposed, including elimination of critical microhabitats through the filling of interstitial spaces (Corn and Bury 1989; Welsh and Ollivier 1998), impaired respiration from the clogging of gills of aquatic amphibians (Kelsey 1995), and, for Coastal Tailed Frog larvae, limited ability to adhere to rocks and feed (Bury and Corn 1988). It needs to be mentioned, however, that the only study that directly addressed sedimentation effects on stream-associated amphibians (i.e., Welsh and Ollivier 1998), was not done in the context of forest practices, and exhibits a bias in stream size between reference and treatment streams (Hayes and Quinn 2015). In our analysis of sediment, we found no correlation between treatment and sediment input, storage or yield (see Chapter 10 − *Sediment Processes*); however, we did note an increasing trend in the proportion of sand and finer substrates relative to larger sediment size classes as buffer length decreased (see Chapter 11 − *Stream Channel Characteristics*). However, we had no evidence that this change differed among treatments.

Timber harvest can cause increased stream temperatures through the removal of overhead canopy (Beschta *et al.* 1987), which may detrimentally affect stream-associated amphibians (Bury and Corn 1988). All focal amphibians have been found to preferentially select cool waters or avoid areas with higher stream temperatures (de Vlaming and Bury 1970; Karraker *et al.* 2006; Bury 2008; Pollett *et al.* 2010). However, Jackson and colleagues (2001) found that stream cover provided by logging slash effectively insulated some clearcut streams lacking riparian buffers in the RMZ against temperature increases. We observed a greater increase in July - August daily maximum stream temperatures in our harvested sites relative to the reference, with the greatest average increase of 2.1°C in the 0% treatment during the first post-harvest year. With pre-harvest daily maximum temperatures ranging from 8.9°C to 13.1°C across all study sites during this period, this could translate to absolute post-harvest temperatures as high as 15.2°C for some 0% treatment sites (see Chapter 7 − *Stream Temperature and Cover*). The critical aspect of this issue is whether this degree of temperature change over pre-harvest conditions translates to a biologically risky condition. Currently, no stress temperature information exists for stream-associated amphibians; however, in stream basins impacted by the 1980 eruption of Mt. St. Helens, Coastal Tailed Frogs were found in the smallest densities in areas highly affected by the eruption, with little intact forest and maximum temperatures near 20°C (Hawkins *et al.* 1988). In fact, conditions appeared to be near optimal in basins partially deforested by the eruption, providing both suitable habitat for adults (i.e., cool, moist microhabitats for reproduction) as well as a food source for tadpoles (i.e., periphyton), at least when stream temperatures remained below 18°C (Hawkins and Sedell 1991).

Timber harvest can result in greater amounts of sunlight reaching a stream, potentially increasing primary production and changing the composition of periphyton (Hawkins *et al.* 1983). Beschta and colleagues (1987) suggested that increased sunlight and/or stream temperature caused a shift in the species composition of periphyton away from diatoms, the primary food source for larval tailed frogs (Altig and Brodie 1972; Nussbaum *et al.* 1983). The food base for macroinvertebrate scrapers also includes diatoms, among other things, and this shift in species composition may affect their abundance (Hawkins *et al.* 1983), thereby impacting food sources for predatory stream-associated salamanders and post-metamorphic frogs. We detected no differences in the changes in periphyton AFDM or chlorophyll *a* among treatments (see Chapter 13 − *Biofilm and Periphyton*); however, we did not evaluate periphyton species composition and have no way of knowing if the proportion of nutritious diatoms in the periphyton matrix changed as a function of treatment. In fact, different diatoms possess varying levels of protein, and vary in quality to the grazers that consume them. For example, members of the freshwater diatom *Rhopalodiaceae* contain N-fixing endosymbiotic cyanobacteria, making them higher in protein than other diatoms (Furey *et al.* 2014) and the relative abundance of N-fixing algae has been shown to be negatively correlated with nitrogen concentrations (Porter *et al.* 2008). Given that we observed a post-harvest increase in total-N and nitrate-N across all buffer treatments, is it quite likely that shifts in food quality also existed.

Many studies on the impacts of timber harvest have reported an increase in stream concentrations of nitrate-N (Likens *et al.* 1970; Brown *et al.* 1973; Harr and Fredriksen 1988). While phosphorus concentration in the stream can also increase after harvest, because it has a tendency to be absorbed onto organic material and clay particles, phosphorus is generally much less mobile in the soil than nitrate. Consequently, increases in concentrations of phosphorus after timber harvest are more likely to be the result of sediment inputs to the channel via erosion. Amphibians are likely influenced by nutrient availability on some level. For example, Kiffney and Richardson (2001) found that Coastal Tailed Frog larvae can be resource (e.g., food) limited when light is not a limiting factor and that growth rates and mass of larvae were greater in channels supplemented with phosphorous, suggesting that larvae were food limited at phosphorus concentrations found in natural streams (Hawkins *et al.* 1988). We observed post-harvest increases in total-N and nitrate-N export across all treatments with the magnitude of change consistent with the severity of the treatment (i.e., greatest increase in the 0% treatment). Conversely, total-P concentrations were not correlated with treatments in a consistent way (see Chapter 9 − *Nutrient Export*). Even with an increase in total-N and nitrate-N, we did not see a corresponding increase in periphyton AFDM, or macroinvertebrate numbers or biomass export, as was seen by Kiffney and Richardson (2001) in their evaluation of the effects of nutrient enrichment in headwater streams.

Decreases in the quantity or quality of habitat from timber harvest could affect stream-associated amphibian populations by inducing emigration or immigration (Peterman *et al.* 2011), or through increased competition and predation (Crawford and Semlitsch 2007). Animals associated with streams may move along stream channels or terrestrially between streams (Fagan 2002; Grant *et al.* 2007), although larval amphibian movement is restricted to the stream. Wahbe and Bunnell (2001) found that larval Coastal Tailed Frog movement declined with an increasing density of log jams. Cecala and colleagues (2014) found that Black-bellied Salamanders (*Desmognathus quadramaculatus*), a highly aquatic headwater salamander native to the eastern U.S., resisted movement across even relatively small (i.e., 13-m) gaps in the riparian canopy. Movement of post-metamorphic individuals can occur along the stream network or overland. Terrestrial movement may be limited if no habitats exist to provide adequate environmental conditions (Grant *et al.* 2010). For example, Matsuda and Richardson (2005) detected post-metamorphic Coastal Giant Salamanders moving overland in old-growth sites but observed none moving in clearcut sites; however, they did not detect a relationship between timber harvest and movement of post-metamorphic Coastal Tailed Frogs. Conversely, Wahbe and colleagues (2004) found that adult Coastal Tailed Frogs traveled farther from streams in old growth than in clearcut sites, and that juveniles showed a stronger affinity for streams located within clearcut versus old growth stands, and Hawkes and Gregory (2012) noted an absence of tailed frogs from uplands 10 years following clearcut logging. Stream-adjacent timber harvest may have affected the proportion of post-metamorphic giant salamanders and tailed frogs that remained near streams. We are not aware of any study that investigates direct mortality of stream-associated amphibians in recently harvested streams.

The large windthrow event that affected sites during the pre-harvest period in December 2007 reflects the underlying natural variability of managed forestlands that occurs throughout western Washington. We did not do a formal statistical analysis comparing amphibian density 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 were impacted, and that the impact was asymmetric across those sites. Fortunately, 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 event greatly affected the response of stream-associated amphibians to treatment; however, the timing of the windthrow event may have affected our ability to distinguish differences among buffer treatments.

15-7. CONCLUSIONS

In the two-year post-harvest period, we found few differences in the densities of focal stream-associated amphibian species across a gradient of disturbance related to harvest. We observed either a lack, or a positive effect, of timber harvest on larval Coastal Tailed Frog density for all buffer treatments and analyses, and a positive response of post-metamorph tailed frog density to the 0% treatment in our stream network-wide analyses. We did observe a negative effect on post-metamorph tailed frog density in the FP treatment when considering only the lower Np reach. However, we have substantial concerns with the validity of this result considering the possibility of local migrations into and out of the lower Np reach and the possibility that these movements may be confounded with treatment since the buffered portion of the FP treatment is mostly restricted to the lower Np reach. We observed either a lack of, or a positive effect of, timber harvest on torrent salamander density for all buffer treatments and analyses. A consistent significant negative response of giant salamander density existed in the FP treatment across all analyses. This result is somewhat puzzling considering the lack of an effect noted for both treatments with greater (100% treatment) and lesser (0% treatment) protection; however, it highlights that giant salamander populations are likely limited by multiple factors, that may be acting individually or in combination through complex interactions (including, but not limited to, habitat, resources, predation and competition). Furthermore, we did not detect a treatment effect on body condition for any species and we continued to find all focal amphibian species in areas lacking a riparian buffer, where logging slash accumulations were greatest.

We conclude that, overall, the current Forest Practice's regulatory buffer established under the Forests and Fish agreement was effective in maintaining FP-designated stream-associated amphibian populations in the two years immediately after timber harvest. However, since our sampling methodology focused on instream sampling, we do not adequately address impacts to terrestrial post-metamorphic Coastal Tailed Frog and Coastal Giant Salamanders in this study. Understanding the basis of the unique negative response of giant salamanders in only the FP treatment will require further investigation. Further, at the scale of Type N basins and on competent (i.e., hard rock) lithologies, clearcut harvest without riparian buffering in the RMZ (0% treatment) maintained focal amphibian populations in the short-term. However, understanding the scale of potential timber harvest impacts on stream-associated amphibian populations at our study sites will require study over a longer temporal scale of at least eight years to allow for one generational turnover of focal amphibian populations. For example, population census over the short-term does not describe potential impacts to reproduction over the long-term. We recommend future investigations at the same study sites after at least one generational turnover to understand the long-term consequences of timber harvest with our riparian buffer configurations. Only longer-term study of the impacts of clearcut timber harvest with alternative riparian buffer prescriptions can provide guidance on the effectiveness of the current Forest Practices rules and their ability to maintain viable populations of the stream-associated amphibians of interest.

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APPENDICES

APPENDIX 15-A. ASSESSMENT OF THE IMPACT OF THE UNCERTAINTY IN DETECTION PROBABILITY ON THE ANALYSIS AND RESULTS

The BACI analysis of stream network-wide amphibian density used the weighted average adjusted density for all basins and years as the response. We adjusted observed counts for imperfect detection by dividing them by estimated individual detection probabilities. However, the adjustments did not take into account uncertainty in the detection probability estimates.

The purpose of this appendix is to outline a post-hoc assessment of the impact of detection probability uncertainty on the experimental results for stream network-wide amphibian density, and to discuss the results of this assessment.

The basic idea behind our approach is that instead producing a single set of stream network-wide density values based on detection probability mean parameters, we will generate a large number (e.g., 500) of density values using a posterior sample of detection probability parameters. A BACI model can be fit to each dataset and the variation across these results can be used to illustrate uncertainty due to the unknown detection probability parameters.

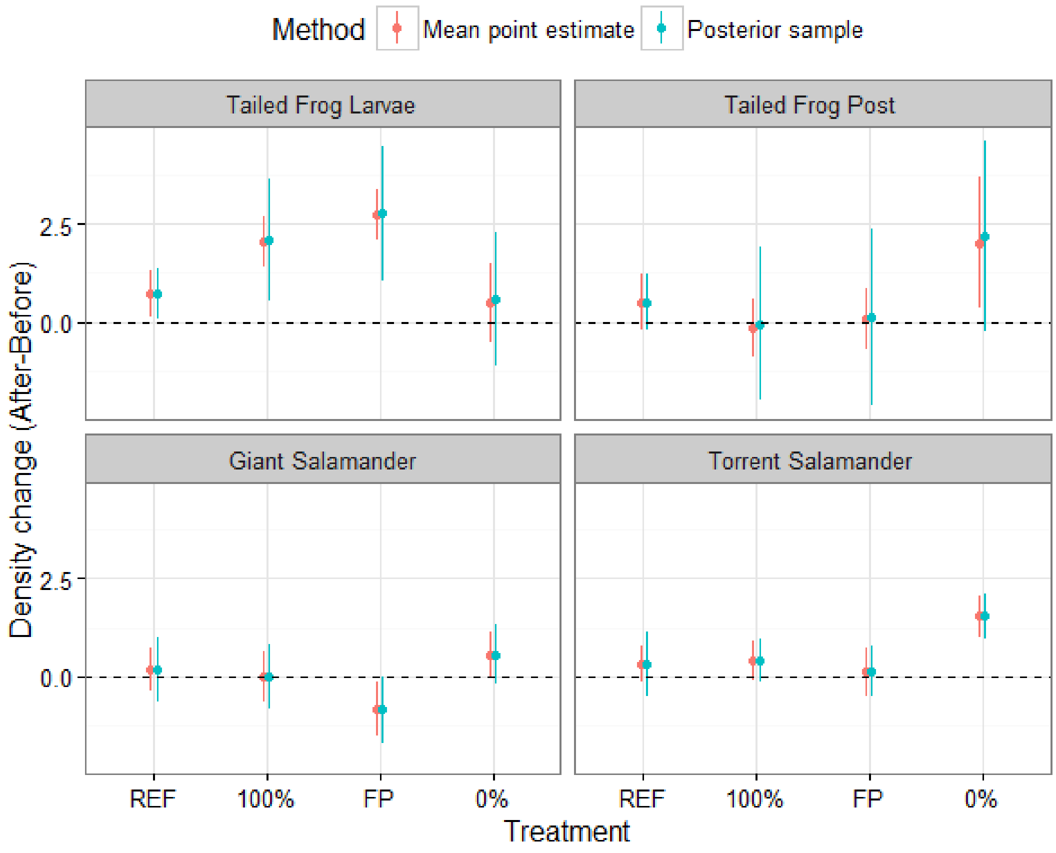
Specifically, we used the steps below to incorporate detection probability uncertainty into estimates and confidence intervals from the BACI analysis. This approach follows closely the method for multiple imputation (Rubin 1987).

1) Draw a sample s from the posterior distribution.
2) Calculate detection probabilities using sample s and covariate data for each single-pass light touch sample.
3) Adjust observed counts by dividing by the calculated detection probabilities in step 2; aggregate the adjusted counts to obtain stream network-wide density estimates, by year.
4) Fit the BACI design model to the stream network-wide density estimates in step 3 and record contrast estimates and standard errors.
5) Repeat steps 1 - 4 S times.
6) Calculate the mean of the squared standard error over the S samples for each contrast; calculate the variance of contrast mean estimates over the S samples. Sum these two quantities.
7) Calculate the sample average over all S contrast mean estimates.

The square root of the sum in step 6 is an estimate of contrast standard error that incorporates both experimental error and uncertainty in the estimated detection probability. One limitation of this approach is that it cannot be easily used to adjust experiment-wise F-test P-values for uncertainty in detection probability. Before-after mean estimates and confidence intervals using

S = 500 are shown for each treatment in **Appendix Figure 15-A-1** below. Between-treatment contrasts of before-after differences are reported in **Appendix Table 15-A-1**.

The results below suggest that ignoring uncertainty in detection probability estimates, as was done in the results reported in Chapter 15 − *Stream-associated Amphibians*, likely had negligible impact on BACI point estimates for all species and all comparisons. In addition, it appears to have had only a small impact on the size of confidence intervals for RHSP and DISP, indicating that inference related to differences among treatments was relatively insensitive to the use of mean point estimates for detection probability. However, BACI point estimate uncertainty for both larval and post-metamorph tailed frogs was considerably larger when accounting for detection probability uncertainty than when using mean detection estimates. This suggests that inference from the BACI analysis for both life stages of the tailed frog may be anti-conservative, that is, confidence intervals and P-values reported in Chapter 15 − *Stream-associated Amphibians* may over-represent our confidence in the magnitude of treatment differences.



Appendix Figure 15-A-1. Estimated stream network-wide density change (After-Before) for each treatment. Separate estimates and 95% confidence intervals are given for two different methods of deriving change estimates: using mean point estimates of detection probability to adjust raw counts, and using a sample of detection probabilities from the posterior distribution to incorporate estimate uncertainty.

Appendix Table 15-A-1. Pairwise treatment comparisons of After-Before density differences. The middle columns show results reported in Chapter 15 − *Stream-associated Amphibians* using point estimates of mean detection probability to adjust raw counts. The right-hand columns show results that incorporate uncertainty in the detection probability estimates. The critical t-value for calculating 95% confidence intervals (CI) is approximately 2.01.

Species	**Contrast**	**Using Mean Detection**		**With Detection Uncertainty**	
		Estimate	**SE**	**Estimate**	**SE**
Tailed Frog Larvae	100% vs. REF	1.33	0.43	1.34	0.84
	FP vs. REF	2.01	0.44	1.98	0.92
	0% vs. REF	−0.23	0.59	−0.19	0.85
	0% vs. FP	−2.24	0.60	−2.17	1.04
	0% vs. 100%	−1.56	0.60	−1.53	1.00
	FP vs. 100%	0.68	0.45	0.64	0.50
Tailed Frog Post-metamorph	100% vs. REF	−0.65	0.50	−0.58	1.01
	FP vs. REF	−0.43	0.51	−0.42	1.17
	0% vs. REF	1.51	0.91	1.65	1.28
	0% vs. FP	1.94	0.92	2.07	1.58
	0% vs. 100%	2.16	0.91	2.23	1.48
	FP vs. 100%	0.22	0.52	0.16	0.63
Giant Salamander	100% vs. REF	−0.20	0.42	−0.20	0.46
	FP vs. REF	−1.01	0.43	−1.03	0.51
	0% vs. REF	0.37	0.40	0.36	0.46
	0% vs. FP	1.38	0.45	1.39	0.47
	0% vs. 100%	0.57	0.44	0.56	0.45
	FP vs. 100%	−0.81	0.47	−0.83	0.48
Torrent Salamander	100% vs. REF	0.09	0.34	0.11	0.47
	FP vs. REF	−0.18	0.37	−0.17	0.53
	0% vs. REF	1.23	0.35	1.23	0.49
	0% vs. FP	1.41	0.39	1.40	0.40
	0% vs. 100%	1.14	0.37	1.12	0.38
	FP vs. 100%	−0.27	0.39	−0.28	0.40

APPENDIX 15-B. GIANT SALAMANDER SPECIES COMPOSITION

We collected giant salamander tail tissue samples from individuals captured in our 17 study sites during rubble-rouse sampling in the lower Np reach and systematic light-touch conducted stream network-wide (2006 through 2010), rubble-rouse sampling in obstructed reaches (2009 and 2010), and during incidental surveys conducted in six basins in 2011 to augment post-harvest sample sizes. Samples were identified as Cope's or Coastal Giant Salamander or as hybrids of the two species using the methods outlined by Spear and colleagues (2011). We summarized the proportions of all sampled individuals that were each of the two species, or were hybrids, by site and period (pre- and post-harvest).

We collected 1,608 giant salamander tissue samples. The proportion of captured individuals sampled at a site ranged from 0.05 to 0.42. We collected only a limited number of samples from sites located in the Olympic Block (8−36; mean = 18) during the first two years of sampling (2006 and 2007) to confirm the absence of the Coastal Giant Salamander in that region. The vast majority of samples (95%) came from the Willapa Hills and the southern Cascades regions, where the two species co-occur. The number of samples collected from these sites ranged from 23 to 228 (mean = 109; **Appendix Table 15-B-1**).

As expected, no Coastal Giant Salamanders were detected in sites located in the Olympic Block (n = 72). Of 167 samples collected in the two northernmost sites in the Willapa Hills (WIL1-0% and WIL2-100%), only one (0.6%) was identified as Coastal Giant Salamander. We detected this individual in the WIL2-100% during the second post-harvest sample year in 2010. The Chehalis River was previously assumed to be the northernmost extent of the range of Coastal Giant Salamanders; however, our data suggests that the northern boundary of that species may occur south of the Chehalis River, or that the occurrence of this species is limited towards the northern extent of its range.

Our sampling methodology did not focus on post-metamorphic individuals, which are not restricted to the stream corridor. We detected 34 (<1%) post-metamorphic individuals: 18 in the pre- and 16 in the post-harvest period (**Appendix Table 15-B-2**). We encountered post-metamorphs across all treatments. We identified 17 individuals to species with genetic analysis: Coastal (n = 5), Cope's (n = 9), hybrid (including backcross to Cope's and F2; n = 3).

Hybridized individuals (n = 26) were detected in six study sites, all of which were located in the Willapa Hills (**Appendix Figure 15-B-1**). In eight of the 11 southernmost basins, the majority of sampled individuals were identified as Coastal Giant Salamander (52% to 82%; mean = 67%). In the remaining three sites, all located in the Willapa Hills, 64% to 83% (mean = 76%) of sampled individuals were identified as Cope's Giant Salamander. Some small changes in species composition were observed in some study sites, though there does not appear to be a pattern by treatment or period (**Appendix Figure 15-B-2**).

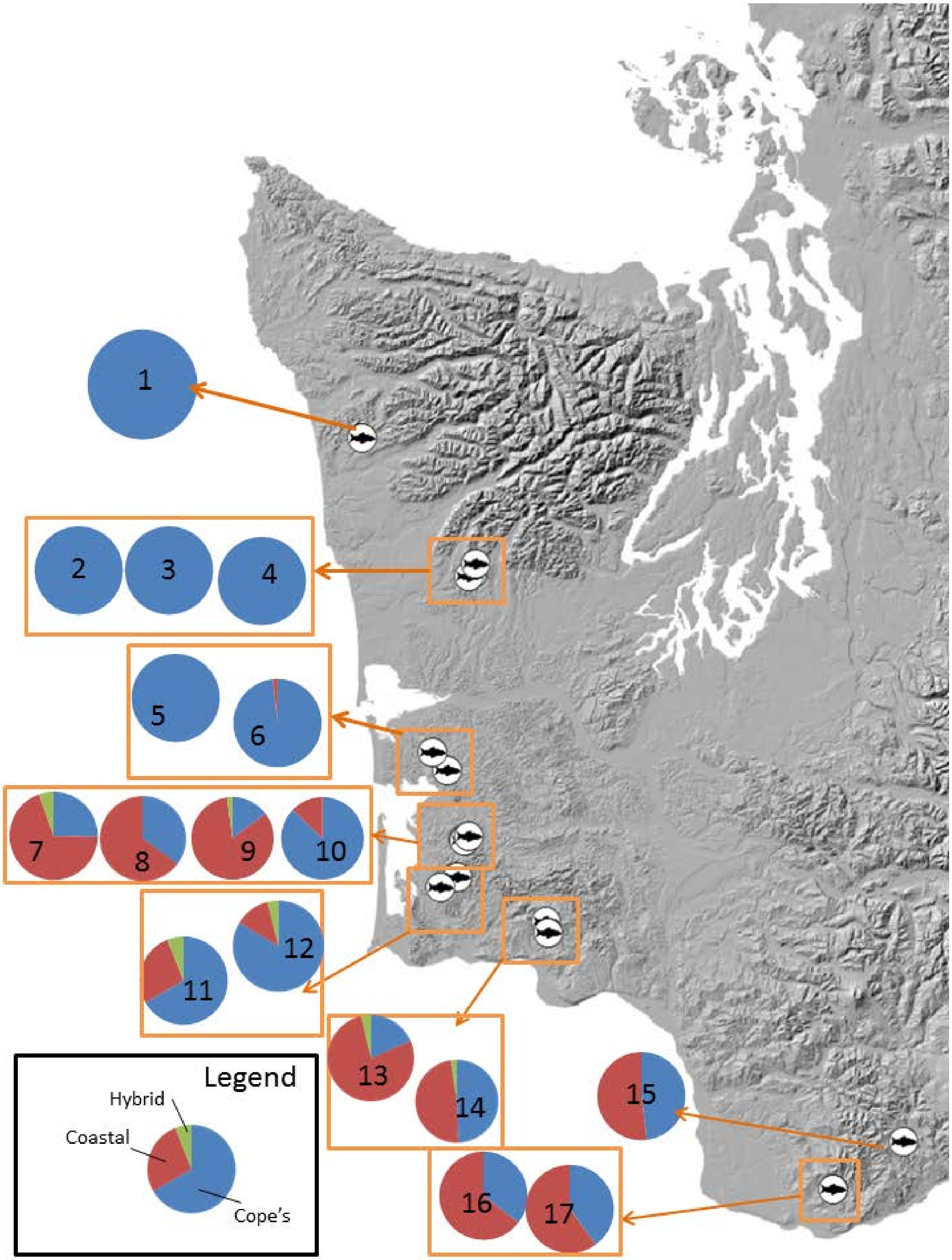
Appendix Table 15-B-1. The number of giant salamanders included in the genetic analysis, the proportion of all observed giant salamanders that were sampled (combined numbers from light-touch, rubble-rouse in the lower Np reach, rubble-rouse in obstructed reaches, and incidental sampling), and the proportion of each sample identified as Cope's and Coastal Giant Salamanders or hybrids (2006−2011).

Site	Block	Treatment	Samples	Proportion Sampled	Proportion of Sample		
					Cope's	Coastal	Hybrid
1	OLYM	100%	14	0.08	1.00	0.00	0.00
2	OLYM	REF	14	0.05	1.00	0.00	0.00
3	OLYM	FP	8	0.16	1.00	0.00	0.00
4	OLYM	0%	36	0.32	1.00	0.00	0.00
5	WIL1	0%	105	0.41	1.00	0.00	0.00
6	WIL2	100%	62	0.26	0.98	0.02	0.00
7	WIL1	100%	193[1]	0.39	0.15	0.82	0.02
8	WIL1	REF	118	0.58	0.35	0.65	0.00
9	WIL1	FP	96	0.42	0.27	0.69	0.04
10	WIL2	REF2	31	0.48	0.87	0.13	0.00
11	WIL2	0%	83[2]	0.37	0.64	0.29	0.06
12	WIL2	REF1	128	0.38	0.84	0.13	0.04
13	WIL3	100%	162	0.32	0.19	0.78	0.04
14	WIL3	REF	84[2]	0.37	0.49	0.48	0.02
15	CASC	REF	146	0.26	0.48	0.52	0.00
16	CASC	0%	127	0.30	0.35	0.65	0.00
17	CASC	FP	228	0.20	0.42	0.58	0.00
Across all sites			**1,608**	**0.30**	**0.49**	**0.49**	**0.02**

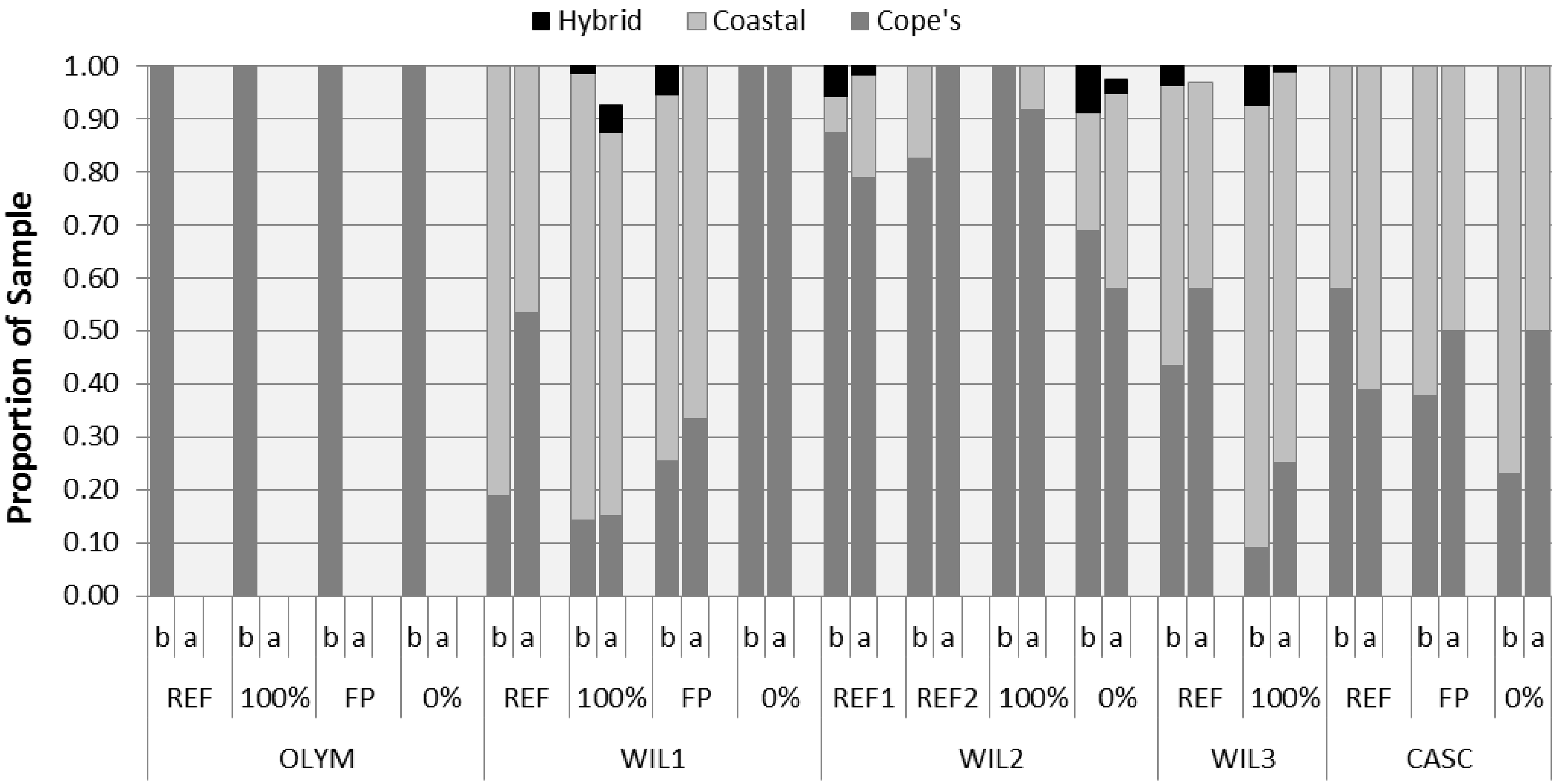
[1] Two individuals could not be confidently identified to species.
[2] One individual could not be confidently identified to species.

Appendix Table 15-B-2. The numbers of post-metamorphic giant salamanders detected by period (pre- and post-harvest) and treatment.

Treatment	Pre-harvest	Post-harvest
REF	3	6
100%	6	3
FP	3	3
0%	6	4
Total	18	16



Appendix Figure 15-B-1. Distribution of giant salamanders (Cope's, Coastal and hybrids) across 17 study sites (2006–2011). Sample size and proportion of each species for numbered sites are presented in **Appendix Table 15-B-1**.



Appendix Figure 15-B-2. The proportion of the giant salamanders sampled at each site that were identified as Cope’s, Coastal or hybrid Giant Salamanders during the pre- (b; 2006–2007) and post-harvest (a; 2009–2011) periods. No samples were collected from the Olympic block in the post-harvest period (since only Cope’s Giant Salamanders are known to occur on the Olympic Peninsula). When bars do not equal 1.0 there was a portion of the sampled population that could not be confidently identified to species (note that this situation only occurred in sites where hybridization was also identified).

APPENDIX 15-C. PARAMETER ESTIMATES FROM N-MIXTURE MODELS

Appendix Table 15-C-1. Parameter estimates from the N-mixture models for all species. For definitions of parameter names see **Appendix Table 15-C-2**.

Taxa	Parameter Name	Mean	SD	2.5%	25%	50%	75%	97.5%
Torrent Salamander	mu.b0	2.01	0.37	1.24	1.77	2.00	2.26	2.74
	sig.b0	1.26	0.24	0.88	1.09	1.24	1.41	1.79
	betaBuf	0.94	0.22	0.52	0.79	0.94	1.09	1.38
	betaUnb	−0.07	0.25	−0.56	−0.23	−0.07	0.09	0.41
	betaOrd	−0.38	0.23	−0.81	−0.53	−0.38	−0.21	0.05
	betaYr09	0.96	0.26	0.50	0.79	0.95	1.13	1.52
	betaYr10	0.20	0.36	−0.38	−0.07	0.15	0.42	1.00
	betaBuf09	−1.37	0.36	−2.07	−1.62	−1.36	−1.12	−0.69
	betaBuf10	−0.97	0.39	−1.79	−1.22	−0.95	−0.69	−0.28
	betaUnb09	−0.73	0.47	−1.61	−1.06	−0.76	−0.42	0.25
	betaUnb10	−1.02	0.53	−2.03	−1.36	−1.02	−0.67	0.03
	alpha0	−0.32	0.23	−0.79	−0.47	−0.30	−0.15	0.08
	alphaT	0.08	0.05	−0.02	0.05	0.08	0.12	0.18
	alphaOrd	0.18	0.38	−0.52	−0.11	0.16	0.45	0.92
	alphaYr09	−0.61	0.37	−1.35	−0.86	−0.58	−0.34	0.06
	alphaYr10	−0.71	0.51	−1.74	−1.07	−0.67	−0.34	0.16
	alphaBuf09or10	0.49	0.56	−0.55	0.07	0.49	0.88	1.56
	alphaUnb09or10	−0.28	0.66	−1.62	−0.72	−0.28	0.19	0.97
Giant Salamander	mu.b0	1.20	0.38	0.48	0.94	1.20	1.45	1.97
	sig.b0	1.19	0.24	0.82	1.03	1.16	1.34	1.76
	betaBuf	0.25	0.23	−0.20	0.10	0.25	0.41	0.71
	betaUnb	−0.38	0.29	−0.94	−0.57	−0.38	−0.19	0.18
	betaOrd	1.62	0.22	1.23	1.46	1.61	1.75	2.09
	betaYr09	0.02	0.34	−0.67	−0.20	0.03	0.25	0.70
	betaYr10	−0.24	0.33	−0.90	−0.46	−0.23	−0.02	0.40
	betaBuf09	−0.82	0.31	−1.46	−1.02	−0.81	−0.61	−0.25
	betaBuf10	−1.64	0.32	−2.28	−1.86	−1.63	−1.43	−1.06
	betaUnb09	−0.49	0.40	−1.28	−0.75	−0.49	−0.22	0.29
	betaUnb10	−0.80	0.41	−1.62	−1.07	−0.80	−0.53	0.00
	alpha0	−0.31	0.28	−0.91	−0.50	−0.29	−0.12	0.18
	alphaT	0.01	0.05	−0.08	−0.03	0.01	0.05	0.12
	alphaOrd	−1.22	0.29	−1.81	−1.42	−1.21	−1.02	−0.69
	alphaYr09	−0.27	0.40	−1.05	−0.54	−0.28	−0.01	0.53
	alphaYr10	−0.41	0.39	−1.15	−0.66	−0.42	−0.15	0.37
	alphaBuf09or10	1.16	0.33	0.55	0.93	1.16	1.38	1.84
	alphaUnb09or10	0.64	0.40	−0.16	0.38	0.64	0.91	1.42

Appendix Table 15-C-1. (continued)

Taxa	Parameter Name	Mean	SD	2.5%	25%	50%	75%	97.5%
Tailed Frog larvae	mu.b0	−1.35	0.67	−2.65	−1.80	−1.35	−0.92	−0.05
	sig.b0	1.91	0.38	1.31	1.66	1.86	2.12	2.80
	betaOrd	2.19	0.24	1.75	2.02	2.18	2.34	2.68
	betaYr09	−0.18	0.23	−0.65	−0.34	−0.18	−0.03	0.28
	betaYr10	−0.58	0.23	−1.04	−0.74	−0.59	−0.43	−0.14
	betaBuf09or10	3.10	0.87	1.48	2.51	3.07	3.66	4.95
	betaUnb09or10	1.40	0.80	−0.04	0.87	1.32	1.86	3.19
	alpha0	−1.84	0.52	−3.03	−2.15	−1.79	−1.47	−1.00
	alphaBuf09or10	−1.45	0.87	−3.28	−2.02	−1.43	−0.85	0.17
	alphaUnb09or10	−0.22	0.86	−2.10	−0.71	−0.14	0.35	1.35
Tailed Frog post-metamorphs	mu.b0	0.03	0.61	−1.12	−0.39	0.01	0.42	1.25
	sig.b0	1.12	0.28	0.66	0.91	1.09	1.29	1.77
	betaOrd	0.83	0.25	0.37	0.66	0.82	0.99	1.34
	betaYr09	1.09	0.38	0.38	0.82	1.08	1.33	1.88
	betaYr10	−0.37	0.45	−1.24	−0.67	−0.38	−0.07	0.50
	betaBuf09or10	−0.29	1.05	−2.19	−1.02	−0.38	0.35	2.04
	betaUnb09or10	0.11	1.18	−2.03	−0.73	0.07	0.90	2.55
	alpha0	−2.73	0.61	−4.01	−3.13	−2.70	−2.31	−1.63
	alphaBuf09or10	−0.07	1.08	−2.47	−0.70	0.04	0.69	1.79
	alphaUnb09or10	−0.51	1.27	−3.03	−1.35	−0.50	0.37	1.88

Appendix Table 15-C-2. Definitions of parameter names presented in **Appendix Table 15-C-1**.

Taxa	Parameter Name	Description
Torrent and Giant Salamanders	mu.b0	Abundance intercept (log-scale)
	sig.b0	Among-basin standard deviation (estimate) in mean abundance
	betaBuf	Contrast for buffered reach vs. control (pre-harvest)
	betaUnb	Contrast for unbuffered reach vs. control (pre-harvest)
	betaOrd	Contrast for 2nd/3rd-order stream vs. 1st-order stream
	betaYr09	Contrast for 2009 abundance vs. 2008 abundance
	betaYr10	Contrast for 2010 abundance vs. 2008 abundance
	betaBuf09	Buffered reach × 2009 year interaction
	betaBuf10	Buffered reach × 2010 year interaction
	betaUnb09	Unbuffered reach × 2009 year interaction
	betaUnb10	Unbuffered reach × 2010 year interaction
	alpha0	Detection probability intercept (logit-scale)
	alphaT	Detection probability temperature coefficient
	alphaOrd	Detection contrast for 2nd/3rd-order streams vs. 1st-order stream (Log-odds-ratio)
	alphaYr09	Detection contrast for year 2009
	alphaYr10	Detection contrast for year 2010
	alphaBuf09or10	Post-harvest buffered reach detection contrast with control
	alphaUnb09or10	Post-harvest unbuffered reach detection contrast with control
Coastal Tailed Frog − larvae and post	mu.b0	Abundance intercept (log-scale)
	sig.b0	Among-basin standard deviation (estimate) in mean abundance
	betaOrd	Contrast for 2nd/3rd-order stream vs. 1st-order stream
	betaYr09	Contrast for 2009 abundance vs. 2008 abundance
	betaYr10	Contrast for 2010 abundance vs. 2008 abundance
	betaBuf09or10	Post-harvest buffered reach abundance contrast with control
	betaUnb09or10	Post-harvest unbuffered reach abundance contrast with control
	alpha0	Detection probability intercept (logit-scale)
	alphaBuf09or10	Post-harvest buffered reach detection contrast with control
	alphaUnb09or10	Post-harvest unbuffered reach detection contrast with control

APPENDIX 15-D. DENSITY ESTIMATES FOR STREAM-ASSOCIATED AMPHIBIANS, WITH AND WITHOUT ADJUSTMENTS FOR DETECTION

Appendix Table 15-D-1. Estimated density (animals/30 m) of Coastal Tailed Frog larvae adjusted for detection (A) and not adjusted for detection (U), both excluding obstructed reaches (i.e., unobstructed stream) and including obstructed reaches.

Block	**Treat-ment**	**Unobstructed Stream**										**Including Obstructed Reaches**									
		2006		**2007**		**2008**		**2009**		**2010**		**2006**		**2007**		**2008**		**2009**		**2010**	
		A	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**
OLYM	REF	4.2	0.6	5.1	0.7	4.3	0.6	12.1	1.6	5.9	0.8	4.2	0.6	5.1	0.7	4.3	0.6	12.1	1.6	5.9	0.8
	100%	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	FP	0.5	<0.1	0.4	<0.1	0.5	<0.1	12.7	0.5	1.3	<0.1	0.5	<0.1	0.4	<0.1	0.5	<0.1	12.7	0.5	1.1	<0.1
	0%	0.0	0.0	0.4	<0.1	2.1	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.4	<0.1	2.1	0.3	1.3	1.3	2.0	2.0
WIL1	REF	2.7	0.4	0.0	0.0	0.7	<0.1	0.5	<0.1	0.0	0.0	2.7	0.4	0.0	0.0	0.8	<0.1	0.5	<0.1	0.0	0.0
	100%	4.5	0.6	4.3	0.6	8.9	1.2	40.4	1.5	29.7	1.0	4.6	0.6	4.3	0.6	8.9	1.2	35.8	1.3	28.3	1.0
	FP	22.6	3.1	6.2	0.9	2.0	0.3	116.0	4.1	116.7	4.4	22.6	3.1	6.2	0.9	2.0	0.3	106.3	3.8	104.5	4.5
	0%	1.2	0.2	5.6	0.8	3.7	0.5	2.3	0.2	2.8	0.3	1.2	0.2	5.6	0.8	3.7	0.5	1.1	0.1	1.4	0.2
WIL2	REF1	0.7	<0.1	1.8	0.2	9.8	1.4	-	-	10.0	1.3	0.7	<0.1	1.8	0.2	9.8	1.4	-	-	10.0	1.3
	REF2	0.5		0.0	0.0	0.3	<0.1	0.3	<0.1	0.6	<0.1	0.5	<0.1	0.0	0.0	0.3	<0.1	0.3	<0.1	0.6	<0.1
	100%	0.3	<0.1	0.2	<0.1	0.4	<0.1	25.6	1.6	5.0	0.7	0.4	<0.1	0.2	<0.1	0.4	<0.1	25.6	1.6	4.7	0.6
	0%	0.5	<0.1	1.2	0.1	2.0	0.3	15.8	1.8	0.4	<0.1	0.5	<0.1	1.2	0.1	2.0	0.3	12.8	1.5	0.3	<0.1
WIL3	REF	1.3	0.2	0.2	0.0	2.1	0.3	3.1	0.4	3.6	0.5	1.3	0.2	0.2	0.0	2.1	0.3	3.1	0.5	3.6	0.5
	100%	0.0	0.0	0.0	0.0	0.6	0.1	4.4	0.1	0.9	<0.1	0.0	0.0	0.0	0.0	0.6	0.1	4.4	0.2	0.9	<0.1
CASC	REF	0.6	<0.1	1.0	0.1	0.0	0.0	0.0	0.0	0.5	<0.1	0.6	<0.1	1.0	0.1	0.0	0.0	0.0	0.0	0.5	<0.1
	FP	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0%	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Appendix Table 15-D-2. Estimated density (animals/30 m) of Coastal Tailed Frog post-metamorphs adjusted for detection (A) and not adjusted for detection (U), both excluding obstructed reaches (i.e., unobstructed stream) and including obstructed reaches.

Block	Treat-ment	Unobstructed Stream										Including Obstructed Reaches									
		2006		2007		2008		2009		2010		2006		2007		2008		2009		2010	
		A	U	A	U	A	U	A	U	A	U	A	U	A	U	A	U	A	U	A	U
OLYM	REF	0.6	0.0	1.1	<0.1	1.1	<0.1	11.1	0.7	1.8	0.2	0.6	0.0	1.1	<0.1	1.1	<0.1	11.1	0.7	1.8	0.2
	100%	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	<0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	<0.1
	FP	0.0	0.0	0.0	0.0	1.2	<0.1	0.7	<0.1	0.0	0.0	0.0	0.0	0.0	0.0	1.2	<0.1	0.7	<0.1	0.0	0.0
	0%	0.0	0.0	0.8	<0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.8	<0.1	0.0	0.0	0.4	0.4	2.5	2.5
WIL1	REF	0.9	<0.1	6.1	0.4	0.0	0.0	0.0	0.0	1.1	<0.1	0.9	<0.1	6.1	0.4	0.0	0.0	0.0	0.0	1.1	<0.1
	100%	2.1	0.2	12.8	0.8	16.1	1.0	13.7	0.8	4.1	0.2	2.1	0.2	12.8	0.8	16.1	1.0	12.2	0.7	3.9	0.2
	FP	10.7	0.6	35.3	2.2	2.3	0.1	30.9	1.7	5.4	0.3	10.7	0.6	35.3	2.2	2.3	0.2	28.6	1.8	4.9	0.3
	0%	0.7	0.0	0.4	0.0	0.0	0.0	6.7	0.3	3.9	0.1	0.7	<0.1	0.4	0.0	0.0	0.0	3.3	0.1	2.9	1.0
WIL2	REF1	1.6	0.1	0.0	0.0	4.0	0.2	-	-	0.0	0.0	1.6	<0.1	0.0	0.0	4.0	0.2	-	-	0.0	0.0
	REF2	0.0	0.0	3.0	0.2	1.3	<0.1	3.3	0.2	3.1	0.2	0.0	0.0	3.0	0.2	1.3	<0.1	3.3	0.2	3.1	0.2
	100%	0.9	0.1	2.7	0.2	0.0	0.0	2.4	0.1	1.2	<0.1	0.9	<0.1	2.7	0.2	0.0	0.0	2.4	0.1	1.1	<0.1
	0%	0.0	0.0	0.5	<0.1	0.5	<0.1	1.7	<0.1	3.6	0.2	0.0	0.0	0.5	<0.1	0.5	<0.1	1.4	<0.1	2.6	0.1
WIL3	REF	0.0	0.0	1.2	<0.1	1.2	<0.1	1.4	0.1	2.0	0.1	0.0	0.0	1.2	<0.1	1.2	<0.1	1.5	0.1	2.0	0.1
	100%	0.7	<0.1	6.2	0.4	4.1	0.3	2.8	0.1	0.6	<0.1	0.7	<0.1	6.2	0.4	4.1	0.3	2.8	0.2	0.6	<0.1
CASC	REF	0.0	0.0	4.3	0.3	10.1	0.6	9.5	0.5	0.6	<0.1	0.0	0.0	4.3	0.3	10.1	0.6	9.5	0.6	0.6	<0.1
	FP	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	0%	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Appendix Table 15-D-3. Estimated density (animals/30 m) of torrent salamanders adjusted for detection (A) and not adjusted for detection (U), both excluding obstructed reaches (i.e., unobstructed stream) and including obstructed reaches.

Block	**Treat-ment**	**Unobstructed Stream**										**Including Obstructed Reaches**									
		2006		**2007**		**2008**		**2009**		**2010**		**2006**		**2007**		**2008**		**2009**		**2010**	
		A	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**
OLYM	REF	0.5	0.2	0.4	0.1	5.0	1.9	0.9	0.3	1.1	0.3	0.5	0.3	0.4	0.2	5.0	1.9	0.9	0.3	1.1	0.3
	100%	4.7	2.1	3.3	1.5	24.1	10.8	10.5	4.8	8.6	3.2	0.0	0.0	0.4	0.2	1.7	0.7	0.0	0.0	1.8	1.8
	FP	0.0	0.0	0.4	0.2	1.7	0.7	0.0	0.0	0.0	0.0	4.7	2.1	3.3	1.5	24.2	10.8	10.5	4.8	8.6	3.2
	0%	0.1	<0.1	0.4	0.2	0.0	0.0	0.0	0.0	1.0	0.2	0.1	<0.1	0.4	0.2	0.0	0.0	0.0	0.0	4.0	3.8
WIL1	REF	11.5	4.9	3.6	1.6	4.7	1.9	12.2	3.7	4.3	1.2	11.5	4.9	3.6	1.6	4.7	1.9	12.2	3.7	4.3	1.2
	100%	21.3	9.4	23.4	11.1	2.4	1.2	23.6	10.4	17.4	6.5	27.9	11.4	25.2	10.5	15.4	6.6	44.3	29.2	28.2	20.2
	FP	27.9	11.4	25.2	10.5	15.4	6.6	27.0	10.5	14.2	5.2	21.3	9.5	23.4	11.1	2.4	1.2	37.3	25.5	21.4	11.0
	0%	3.7	1.6	5.8	2.7	20.1	8.8	4.4	1.3	5.5	1.2	3.7	1.6	5.8	2.7	20.1	8.8	23.9	22.4	59.9	57.7
WIL2	REF1	5.9	2.3	36.5	13.9	14.5	5.6	-	-	28.8	6.8	5.9	2.3	36.5	13.9	14.5	5.6	-	-	28.8	6.8
	REF2	5.6	2.2	6.6	2.8	0.9	0.5	10.0	2.7	7.1	1.9	5.6	2.2	6.6	2.8	0.9	0.5	10.0	2.7	7.1	1.9
	100%	6.7	2.7	7.7	3.6	7.1	3.4	24.2	9.2	11.8	4.1	6.7	2.8	7.7	3.6	7.1	3.4	24.2	9.2	13.8	6.5
	0%	16.0	6.8	23.5	11.1	11.0	5.2	15.0	4.6	14.8	3.9	16.0	6.8	23.5	11.2	11.0	5.2	71.7	63.3	99.9	92.1
WIL3	REF	10.2	3.7	6.9	2.8	19.8	7.6	29.0	7.8	21.7	5.2	10.3	3.7	6.9	2.8	19.8	7.6	29.0	7.8	21.7	5.2
	100%	5.3	1.9	9.6	4.5	19.6	7.4	13.2	5.9	8.0	3.1	5.3	1.9	9.6	4.5	19.6	7.4	13.2	5.9	8.0	3.1
CASC	REF	18.7	6.7	6.6	2.8	40.6	14.7	20.0	4.9	11.8	2.8	18.7	6.7	6.6	2.8	40.6	14.7	20.0	4.9	11.8	2.8
	FP	5.0	2.1	17.0	7.4	24.1	10.4	14.7	5.5	1.7	0.5	5.0	2.1	17.0	7.4	24.1	10.4	15.6	6.5	1.9	0.7
	0%	1.3	0.6	0.8	0.4	1.3	0.6	3.7	1.2	1.7	0.4	1.3	0.6	0.8	0.4	1.4	0.6	3.7	1.3	1.7	0.4

Appendix Table 15-D-4. Estimated density (animals/30 m) of giant salamanders adjusted for detection (A) and not adjusted for detection (U), both excluding obstructed reaches (i.e., unobstructed stream) and including obstructed reaches.

Block	**Treat-ment**	**Unobstructed Stream**										**Including Obstructed Reaches**									
		2006		**2007**		**2008**		**2009**		**2010**		**2006**		**2007**		**2008**		**2009**		**2010**	
		A	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**	**A**	**U**
OLYM	REF	3.6	0.7	2.8	0.6	3.0	0.5	6.1	0.9	3.1	0.4	3.6	0.7	2.8	0.6	3.0	0.5	6.1	0.9	3.1	0.4
	100%	0.5	<0.1	0.6	0.1	7.5	2.8	4.2	1.3	2.6	0.6	0.5	<0.1	0.6	0.1	7.5	2.8	4.2	1.3	2.6	0.6
	FP	1.2	0.2	0.7	0.2	1.7	0.3	0.6	0.3	1.2	0.5	1.2	0.2	0.7	0.2	1.7	0.3	0.6	0.3	1.0	0.4
	0%	2.0	0.3	0.4	0.1	0.3	<0.1	0.0	0.0	1.5	0.3	2.0	0.3	0.4	0.1	0.3	<0.1	0.9	0.9	9.9	9.6
WIL1	REF	7.9	1.4	0.7	0.2	2.4	0.8	5.5	0.8	6.0	0.9	7.9	1.4	0.7	0.2	2.4	0.8	5.5	0.8	6.0	0.9
	100%	14.0	2.6	11.6	2.3	3.1	0.6	7.9	2.8	4.4	1.4	14.0	2.6	11.6	2.3	3.1	0.6	16.1	11.6	4.1	1.3
	FP	9.8	4.1	8.6	3.7	1.3	0.6	1.7	1.0	1.6	1.0	9.8	4.1	8.6	3.7	1.3	0.6	3.6	3.0	12.6	12.1
	0%	6.8	1.3	7.1	1.4	3.6	0.9	5.4	1.3	2.6	0.7	6.8	1.3	7.1	1.4	3.6	0.9	6.3	4.4	6.8	5.7
WIL2	REF1	7.8	1.7	17.2	4.7	6.3	1.8	-	-	16.3	3.5	7.9	1.8	17.3	4.7	6.3	1.8	-	-	16.3	3.5
	REF2	1.4	0.3	0.9	0.3	0.5	<0.1	1.2	0.3	0.2	<0.1	1.4	0.3	0.9	0.3	0.5	<0.1	1.2	0.3	0.2	<0.1
	100%	2.7	0.5	5.7	1.1	5.9	1.4	8.6	1.6	5.3	1.0	2.7	0.5	5.7	1.1	5.9	1.4	8.6	1.6	5.9	1.8
	0%	2.8	0.6	5.1	1.1	8.7	2.0	4.6	1.0	5.5	1.3	2.8	0.6	5.1	1.1	8.7	2.0	12.7	9.8	22.6	19.6
WIL3	REF	0.8	0.1	1.7	0.3	3.8	0.7	4.7	0.7	3.3	0.5	0.8	0.1	1.7	0.3	3.9	0.7	4.8	0.7	3.3	0.5
	100%	1.6	0.3	8.5	1.7	17.8	3.4	10.6	3.8	3.8	1.4	1.6	0.3	8.5	1.7	17.8	3.4	10.6	3.8	3.8	1.4
CASC	REF	8.8	1.5	12.5	2.3	22.3	4.6	15.2	2.4	9.3	1.6	8.9	1.5	12.5	2.3	22.3	4.6	15.2	2.4	9.3	1.6
	FP	13.9	3.2	36.7	7.6	39.6	9.1	11.7	4.6	1.6	0.6	13.9	3.2	36.7	7.6	39.6	9.1	11.9	5.0	2.6	1.6
	0%	3.6	1.5	7.7	3.3	17.2	7.3	12.4	5.7	2.4	1.1	3.6	1.5	7.8	3.3	17.2	7.3	12.4	5.7	2.4	1.1

CHAPTER 16 - DOWNSTREAM FISH

Jason Walter, Brian Fransen, Jack Giovanini, Steve Duke, Robert Bilby, Graham Mackenzie, and Renata Tarosky

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16-1. ABSTRACT

Headwater streams comprise a significant proportion of the cumulative stream length in mountainous catchments, and are an important component to the ecology of lotic systems. These small streams provide critical habitat for fish such as coastal cutthroat trout, the species most often found at the upstream extent of fish distribution in western Washington. Few published studies exist characterizing the fish populations in these unique habitats, or their sensitivity to modern timber harvest practices and those that do report various conflicting responses. The original intent of the Type N Study was to include an evaluation of fish response in the stream segments immediately downstream from timber harvests treated with current Washington Forest Practices buffers (FP treatment), more extensive buffers (100% treatment), or no buffers (0% treatment), relative to unharvested reference sites. For a variety of reasons, however, 11 of the 17 Type N sites included in the final study design were dropped from the fish component of the study. Due to a resulting lack of replication, an evaluation of fish response to upstream timber harvest and different riparian buffer prescriptions, as originally intended, was not possible. Instead, we modified our objectives to treat the remaining six sites as a case study, with the intent of providing insight into the characteristics of cutthroat trout and their habitats at the upstream extent of fish distribution. Fish and stream habitat data were collected twice annually (July and October) between 2006 and 2010. Cutthroat trout density and population structure were highly variable, both spatially and temporally, across sites, months, and years. This variability in total fish abundance did not appear to be correlated with physical stream habitat metrics such as gradient and percent pool area that were also variable across sites. There was, however, suggestive evidence of a positive relationship between age-0 fish density and percent pool area in both the July and October samples ($P = 0.05$ and 0.06, respectively). Analysis of variance revealed that fish condition was consistently higher in July than October ($P < 0.001$). We found no relationship between fish condition and density. Consistently low recapture rates for PIT-tagged fish over the course of the study provides strong evidence of a high level of fish emigration from and/or mortality within study reaches. The percent of PIT-tagged fish that were recaptured during multiple surveys dropped exponentially through time with only 28.6, 5.7, 2.6, and 0.6% of fish being recaptured one, two, three, and four times, respectively. A general linear model fit to size data from recaptured PIT-tagged fish revealed that a log-linear relationship exists between specific growth and initial fish size ($P = 0.002$). For each additional one gram of initial weight, growth rate was reduced by 7.1%. We found no relationship between fish density and growth in either the July–October or the October–July interval. This work documents the low abundance and growth of coastal cutthroat trout in stream reaches at the upstream extent of fish distribution in western Washington. We found that these habitats tended to support lower densities of cutthroat trout than typically reported in the published literature for headwater basins as a whole, and that the fish in these habitats grew more slowly and were smaller on average with a lower condition factor than fish reported in these studies.

16-2. INTRODUCTION

Headwater streams account for more than 70% of the cumulative stream channel length in mountainous catchments (Lowe and Likens 2005), and are critical to the ecological function of aquatic systems. They are important to nutrient cycling, in the recruitment and transport of sediment and woody debris, and provide habitat for a variety of aquatic organisms, including fish (Meyer and Wallace 2001; Gomi *et al* 2002; Rosenfeld *et al.* 2002; Richardson and Danehy 2007). The Forestry Module of the Forests and Fish Report (FFR) established numerous goals, including providing compliance with the Endangered Species Act (ESA) for aquatic and riparian-dependent species on non-federal forest lands, and restoring and maintaining riparian habitat on non-federal forest lands to support a harvestable supply of fish (USFWS 1999). Headwater fish-bearing streams in western Washington often do not support ESA-listed fish species, or fish that are subject to significant commercial or sport harvest. Headwater fishes and the streams they inhabit are nonetheless of great interest to FFR stakeholders due, at least in part, to their importance in determining the regulatory division between fish-bearing (Type F) and non-fish-bearing (Type N) waters, and the associated change in riparian buffer requirements for each stream type.

Coastal cutthroat trout (*Oncorhynchus clarkii clarkii*) typically rear in small headwater streams (Rosenfeld *et al.* 2000; Rosenfeld *et al.* 2002; Reeves *et al.* 2011), and are the fish species that most often occupies stream habitat at the extreme upstream extent of fish distribution in western Washington (Connolly 1997; Trotter 2000; Fransen *et al.* 2006). Few published studies exist characterizing the fish populations in these unique habitats, or their sensitivity to modern timber harvest practices. Studies that have investigated the effects of timber harvest on trout in headwater streams have typically focused on habitats well downstream from the upper extent of fish use or in headwater catchments in their entirety, assessed the fish population long after timber harvest had occurred, and/or reported a variety of conflicting responses. The Alsea Watershed Study (Brown 1972), for example, was implemented in the 1950s to investigate the effects of various timber harvest practices on water quality and fish populations in a set of three watersheds. Researchers reported a one-third reduction in cutthroat trout numbers in a treatment watershed (Needle Branch) that was entirely clearcut and burned, but found no significant changes in fish populations or habitats in nearby patch-cut (Deer Creek) or control (Flynn Creek) watersheds over the same time period (Hall and Lantz 1969; Brown 1972). Similarly, in another more recent study, Young and colleagues (1999) found significantly lower cutthroat trout abundance associated with timber harvest. In contrast, Murphy and colleagues (1981) found higher trout abundance and biomass in stream reaches within clearcuts (5–17 years after harvest) than in nearby, old-growth (>450 years old) forest stands. Similar increases in trout abundance and biomass in stream segments associated with clearcuts, relative to unlogged stands, were also reported by Aho and Hall (1976) and Bisson and Sedell (1984). In even greater contrast, De Groot and colleagues (2007) did not detect any logging treatment effects on abundance or condition of cutthroat trout. Connolly and Hall (1999) also reported no difference in trout biomass, density, or mean weight among three management categories (unlogged, logged 20–30 years ago, and logged 40–60 years ago). Most recently, Bateman and colleagues (2015) found that timber harvest in fishless headwaters did not have a significantly negative effect on downstream coastal cutthroat trout populations, and that the only statistically significant response of downstream fish to upstream logging was an increase in late-summer biomass of age-1+ coastal cutthroat trout. Regarding fish distribution, Latterell and colleagues (2003) concluded that forest management has likely not impacted the upstream extent of trout distribution in headwater streams.

One of the first projects initiated in the reconfigured Cooperative Monitoring, Evaluation and Research Committee (CMER) and the Instream Scientific Advisory Group (ISAG) following FFR negotiations was the development of a monograph summarizing available information and literature related to headwater fish populations (Trotter 2000). In the published CMER report, several hypotheses about the ecology of headwater fish populations were proposed. The Type N Study, as proposed, provided an opportunity to assess the effectiveness of current Forest Practice prescriptions, generate new information characterizing headwater fish populations, and evaluate existing hypotheses on cutthroat trout at the upstream extent of fish distribution, including some related to food availability, habitat limitations, recruitment, and mortality. A fish component was therefore included in the final Type N study design.

16-3. OBJECTIVES

The evaluation of fish response to upstream timber harvest was included for all sites in the original Type N Study Plan. The plan specified the inclusion of 20 sites with a basin area less than 49 ha) distributed over five blocks across western Washington. A description of the buffer treatments for all sites can be found in Chapter 2 – *Study Design*.

Initial site selection for the Type N study relied, in part, on the location of the modeled fish/no-fish (F/N) break to establish the downstream extent of each potential sample basin. Due to error in the location of modeled F/N breaks relative to the actual upstream extent of fish use, and the relatively low likelihood of fish use in streams meeting the maximum 49 ha threshold, knowledge of actual fish use was required to identify sites that were suitable for inclusion in the study. In addition, because we were originally interested in evaluating fish response in the stream reach immediately downstream from and most directly impacted by upstream timber harvest, a minimum continuous distance of 100 m downstream from the F/N break within each subject stream without a junction with a significant and/or fish bearing tributary was a specific requirement for inclusion in the fish component of the study. Other site selection criteria for inclusion in the fish component of the study were that resident cutthroat trout were the only salmonid species present in the potential fish sampling reach, and the potential reach was not within or adjacent to an area of recent timber harvest activity.

During the site selection process in 2005 and early 2006, the location of the actual F/N break was determined within 48 candidate sites using specific protocols for conducting presence/absence electrofishing surveys on forestlands in Washington State (Washington Forest Practices Board 2002). Based on the results of these surveys many potential candidate sites were eliminated from consideration for inclusion in the overall Type N study due to the large size (>49 ha) of the basin upstream of the actual Type F/N break. The majority of the remaining candidate sites that were selected for inclusion in the overall study were not appropriate for inclusion in the fish component due to the close proximity (<100 m) of the actual Type F/N break to a significant or fish-bearing tributary confluence downstream, the presence of a salmonid species other than cutthroat trout in the potential fish sampling reach, and/or recent timber harvest activity adjacent to the potential fish sampling reach. Only six of the final 17 Type N sites across two of the blocks (OLYM-0%, OLYM-FP, WIL1-0%, WIL1-FP, WIL1-100%, WIL1-REF) met all of the criteria for inclusion in the fish component of the study **(Figure 16-1)**.

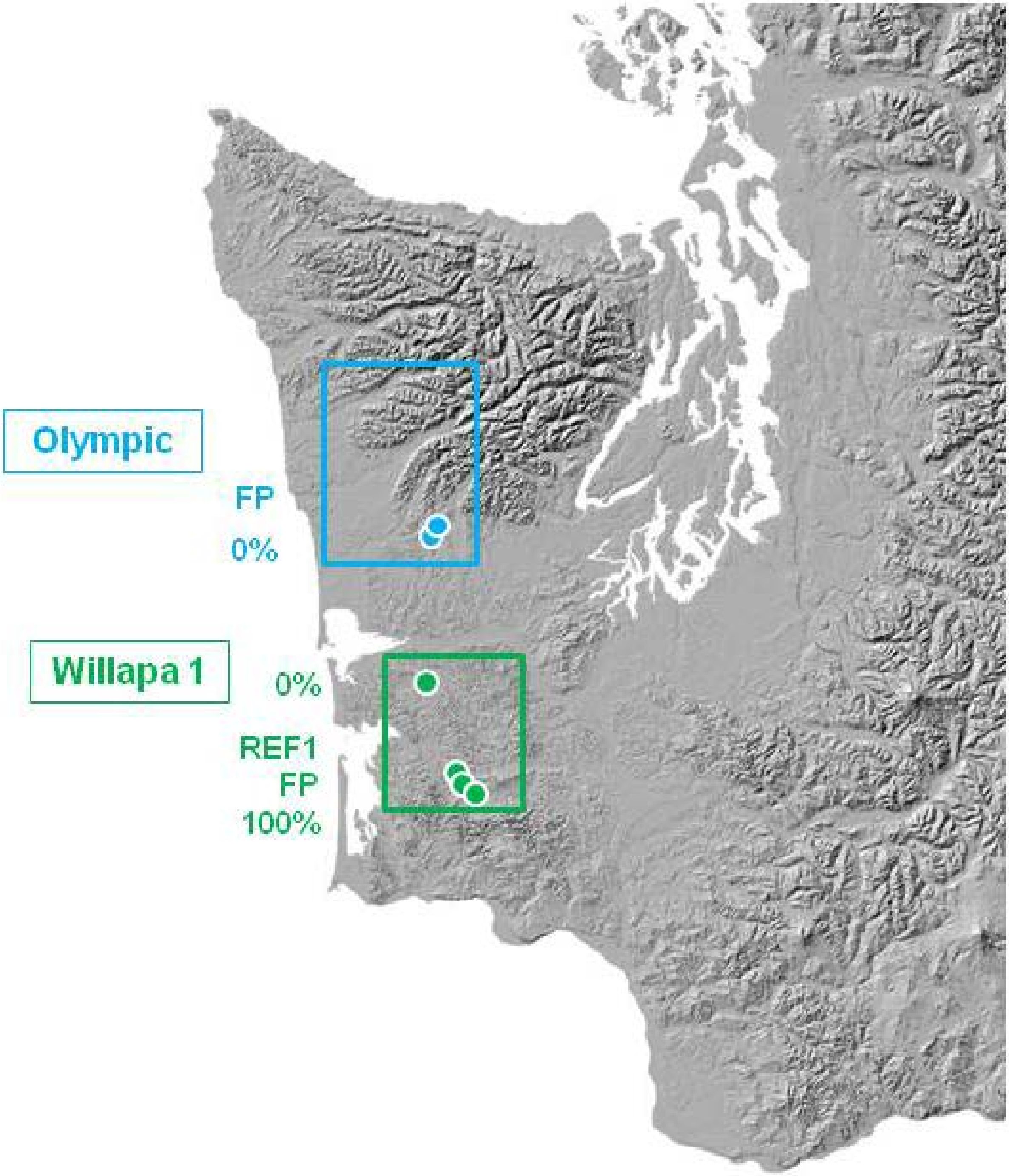

Figure 16-1. Distribution of the six Type N study sites and treatments included in the fish component of the study. Study sites are blocked and based on geography. REF = reference; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment.

With only one complete block (all four treatments) and one incomplete block (only 0% and FP treatments represented) of sites we had a lack of replication and therefore a statistically rigorous evaluation of fish response to upstream timber harvest and the effectiveness of different riparian buffer prescriptions as part of the Type N study was not appropriate. The study design and the selected suite of 17 final sites were, however, appropriate for evaluating the other physical and biological response variables of interest identified in the Type N Study Plan. In addition, the evaluation of fish response was secondary to many of these other variables. Therefore, a decision was made to move forward with the overall Type N Study despite the obvious limitations to the fish component. Given these limitations, our research objectives for the fish component of the study were modified to treat the remaining suite of sites as a case study, with the intent of simply providing insight into the characteristics of cutthroat trout populations and their habitats at the upstream extent of fish distribution.

16-4. METHODS

16-4.1. DATA COLLECTION

Fish population and physical stream habitat data were collected between 2006 and 2010. Sampling was conducted two times per year in July and October. This twice-yearly sampling approach provided us the opportunity to not only assess fish abundance, size, and condition at specific points in time, but also to assess growth, mortality and emigration over two specific time intervals each year.

We conducted surveys along a 100-m long reach immediately downstream of the previously identified Type F/N break. Prior to electrofishing, we isolated the survey reach by installing channel-spanning block nets at both the upstream and downstream end of the reach. We then conducted electrofishing surveys in a downstream direction using a three-pass removal methodology (Zippin 1956; Carle and Strub 1978). We used a Smith-Root Model LR-24 backpack electrofishers for fish sampling. The electrofisher setting at each site was set using the ‘Quick Setup’ process available on the LR-24, which automatically sets the voltage of the electrofisher to produce a 30 Hz, 12% duty cycle at 25 watts average output, relative to the water being sampled. We measured fork length (mm) and weight (to the nearest 0.01 g) for all sampled cutthroat trout, and implanted all cutthroat trout with a fork length >60 mm with Passive Integrated Transponder (PIT) tags unless the condition of a specific individual brought into question its ability to survive the tagging process. After allowing adequate time for recovery, fish were re-distributed throughout the stream reach from which they were sampled (**Figure 16-2**).

Figure 16-2. Conducting fish surveys for the Type N Study: (a) electrofishing a typical Type N study reach; (b) typical catch from a single electrofishing pass through a Type N study reach (note 5-gallon bucket for scale); (c) representative coastal cutthroat trout from three different cohorts on length board for scale; and (d) PIT tagging of a coastal cutthroat trout.

We collected physical stream habitat data at all sites on each sample date using modified stream channel unit criteria developed by Bisson and colleagues (1982). We identified individual channel units as a pool, riffle, cascade, or rapid, and measured for length (m), wetted width (m), maximum depth (cm) and gradient (%). We recorded a single length and a single wetted-width measurement to the nearest 0.1 m at the lateral and longitudinal mid-point of each channel unit using a hand-held Tru-Pulse laser rangefinder (Laser Technology, Inc.), and we recorded a single gradient measurement to the nearest 1% for the entire length of each channel unit using a hand-held clinometer. We measured maximum depth to the nearest centimeter for each channel unit using a handheld stadia rod, and measured bankfull width to the nearest 0.1 m every 25 m over the length of the study reach.

16-4.2. DATA ANALYSIS

We used α = 0.05 for all statistical analyses, although we report all P-values, including those that are greater than α, to provide more information for the reader. We provide estimates and 95% confidence intervals (CI) for regression model parameters of scientific interest where appropriate. Lastly, we examined residuals for normality and homogenous variance when necessary, with no issues found.

16-4.2.1. Fish Density and Population Structure

We estimated age classes visually from length-frequency histograms by assigning individual fish to one of two age classes, age-0 (young of the year) or age-1+ (adult fish), based on a threshold that split the histogram into two distinct cohorts. Length-frequency was not useful for segregating age classes of fish older than one year. We estimated total population with 95% CI for each age class for both sample events each year at all sites using a small sample estimator (Zippin 1956; Carle and Strub 1978). We then calculated linear density by dividing the estimated total number of fish present by the length of stream sampled.

16-4.2.2. Fish Density Relative to Physical Stream Habitat

To assess the association of linear fish density with gradient and percent pool area (the habitat variables found to vary significantly between sites) we fit regression models with fish density as the response and gradient or percent pool area as the explanatory variables. We fit these models separately for the age-0 and age-1+ cohorts for each July and October sample events. We fit the model in R (R Core Team 2014) using the lm function and we tested for a non-zero slope.

16-4.2.3. Fish Condition

We evaluated fish condition after fitting a regression of the natural log of length and the natural log of weight for all fish captured over the entire study. This allowed for the calibration of condition values to the study population. We calculated condition of each fish as the ratio of the actual weight divided by the expected weight for a fish of given length based on the regression. A condition value of greater than one indicated that a fish was heavier than the estimated population mean for its given length, or ‘more fit’. A ratio of less than one indicated that a fish was lighter than the mean for its length, or ‘less fit’. We performed a test for differences for the pooled populations by age and sample month using an analysis of variance model fit in SAS 9.3 using the GLM procedure (SAS Institute 2011).

To assess the association of fish condition and linear fish density, we fit a regression model with mean fish condition by site as the response and linear fish density by site as the explanatory variable. We then tested for a non-zero slope. This model was fit separately for the July and October samples in R, using the *lm* function (R Core Team 2014).

16-4.2.4. PIT Tagging and Recapture

We summarized PIT-tag implants and recaptures by site, year, and sample month. We evaluated the potential for size-dependent influence on fish emigration and mortality within our sites by fitting a regression model to the relationship between tag recovery and fish size and testing the slope for significance. The model was fit in SAS 9.3 using the GLM procedure (SAS Institute 2011).

16-4.2.5. Fish Growth

Specific growth rate (g/g/day) provides for the measurement and comparison of growth across different sizes of fish. We calculated specific growth between sample dates in two ways: (1) we estimated the specific growth of age-0 fish, which were too small to tag across a representative range of fish size, using the pooled average weight of all age-0 fish sampled; and (2) we calculated specific growth rate for recaptured tagged fish. Specific growth was calculated using the following equation:

$$\frac{ln(wt_2) - ln(wt_1)}{t_2 - t_1} \tag{16-1}$$

where: wt_1 is the weight of an individual tagged fish or cohort at t_1
wt_2 is the weight of that fish or cohort at t_2
t_1 is time 1
t_2 is time 2

To examine the relationship between specific growth and initial size of PIT-tagged fish we used a general linear model to fit an analysis of covariance (Milliken and Johnson 2002). We used the natural log transformation of the specific growth response to account for increasing variability in growth rates as initial fish size decreases. The final model was:

$$ln(\text{specific\_growth}_i) = \text{sample\_month} + \text{weight}_i \tag{16-2}$$

where: specific_growth_i is the growth of fish i
sample_month is July or October
weight_i is the weight in grams of fish i

The models were fit in SAS 9.3 using the GLM procedure (SAS Institute 2011).

We fit a linear model to compare specific growth rate of age-0 fish for the intervals between sample dates (July–October and October–July). The data has the form of a randomized block design with repeated measures (Littell *et al.* 2006), where sites play the role of blocks and sample year is the repeated measure. To account for correlation between years, we used AIC (Burnham and Anderson 2002) to select the compound symmetry correlation structure from several candidates. The compound symmetric correlation structure has equal variances at all years and equal covariance between observations within a year (Littell *et al.* 2006). The fixed effects portion of the model is then:

$$\text{specific\_growth} = \text{sample\_month} + \text{year} + \text{sample\_month} * \text{year} \qquad (16\text{-}3)$$

A random site effect was included to account for variation of samples collected over multiple years at single sites. The model was fit in SAS 9.3 using the Mixed procedure (SAS Institute 2011). The Mixed procedure is used to fit linear statistical models that contain both fixed and random effects (Littell *et al.* 2006).

We evaluated for a density-dependent influence on fish growth by testing for a non-zero slope on a regression of growth and density for the age-0 cohort and for recovered tagged fish. These models were fit separately for the two distinct intervals between sample dates (July–October and October–July) in R using the *lm* function (R Core Team 2014). The *lm* function fits least square regression models.

16-5. RESULTS

Resident cutthroat trout were the only salmonid species encountered within the six Type N survey reaches. A small number of sculpin (*Cottus spp.*) were found at three of the sites (WIL1-FP, WIL1-100%, WIL1-REF); however, those data were not included in our analysis and are not further reported here. Over the five years of fish sampling (2006–2010), a total of 1,183 cutthroat trout were sampled via three-pass electrofishing surveys. Estimated removal efficiency was 97% over three passes, across all sites and years. Average sampling efficiency across all passes was 68.7% (64, 63, and 79% on passes one, two, and three, respectively), as calculated by the ratio of actual fish captured over the estimated total number of fish available on each pass (i.e., fish caught on prior passes were removed from the estimated total number of fish remaining). Direct sampling mortality impacted a total of 40 of the 1,183 (3%) fish collected over the course of the study.

Physical stream habitat metrics collected at each site over the course of the study period were relatively consistent, showing no significant change year-to-year. Due to this temporal consistency in the habitat data we chose to not report habitat values for each sample date, but instead to average both the July and October stream habitat data for each site across all years and report those values along with associated standard deviation, where appropriate (**Table 16-1**). Measured habitat metrics that reflect the small size of our study sites such as stream bankfull width and pool depth, were relatively consistent across all sites. Average bankfull width ranged from 1.7 m in the WIL1-100% to 2.2 m in the OLYM-0%. Average maximum pool depth ranged from 12.4 cm in the WIL1-FP to 19.8 cm in the OLYM-FP. Despite the consistently small size of the study reaches, however, they varied widely in some habitat characteristics as exemplified by other metrics such as reach gradient and percent pool area. Stream gradient ranged from 5.7% in the WIL1-REF to 17.3% in the OLYM-0%, representing two distinct channel types, step-pool and cascade channels, respectively (Montgomery and Buffington 1998). Percent pool area was also highly variable across sites, ranging from 11% in the OLYM-0% to 42% in the WIL1-100%.

Table 16-1. Average of physical channel characteristics measured each year from 2006–2010 (± 1 standard deviation) for the six study sites included in the fish component of the Type N study for both July and October sample dates.

Site	Basin Area (ac)	Sample Period							
		July				October			
		Grad. (%)	BFW (m)	% Pool Area	Average Max. Pool Depth (cm)	Grad. (%)	BFW (m)	% Pool Area	Average Max. Pool Depth (cm)
OLYM-0%	41	17.3 (±0.8)	2.2 (±0.4)	11 (±4.3)	14.8 (±3.0)	15.8 (±4.1)	2.2 (±0.3)	14 (±7.1)	16.0 (±2.8)
WIL1-REF	32	5.9 (±0.6)	1.8 (±0.3)	33 (±5.6)	14.5 (±2.8)	5.7 (±0.8)	1.9 (±0.4)	39 (±12.3)	14.1 (±2.9)
WIL1-0%	29	12.8 (±2.3)	2.0 (±0.2)	26 (±2.6)	15.5 (±1.8)	12.3 (±1.7)	2.2 (±0.3)	25 (±5.6)	17.2 (±2.5)
OLYM-FP	76	7.0 (±1.6)	2.1 (±0.5)	34 (±8.9)	16.3 (±3.1)	7.0 (±0.9)	2.0 (±0.3)	31 (±3.8)	19.8 (±4.0)
WIL1-FP	37	10.2 (±1.2)	1.7 (±0.7)	17 (±4.6)	12.6 (±2.2)	9.9 (±0.9)	2.0 (±0.3)	16 (±6.7)	12.4 (±2.6)
WIL1-100%	69	5.8 (±0.5)	1.7 (±0.2)	42 (±3.1)	17.2 (±2.2)	6.1 (±0.7)	1.7 (±0.2)	41 (±7.3)	17.7 (±3.8)

16-5.1. FISH DENSITY AND POPULATION STRUCTURE

Fish density and population structure were highly variable, both spatially and temporally, across sites, sample months, and years. Sampled cutthroat trout ranged in length from 25–154 mm. We assigned individuals to one of two distinct age classes: age-0 (young-of-the-year) and age-1+ fish based on their length (**Figure 16-3**). The OLYM-FP and WIL1-0% consistently supported cutthroat trout populations with representation of both age-0 and age-1+ fish at higher abundance relative to the other sites. The highest value we recorded for total cutthroat density over the course of the study was 0.71 fish/m in the OLYM-FP during the October 2008 sample. The OLYM-0% consistently lacked representation of the age-0 age class, with older age cutthroat occurring at low abundance. In contrast, the WIL1-100% supported relatively abundant age-0 fish but older age cutthroat were at low density or not detected at all. The WIL1-REF and WIL1-FP supported few fish in most years, with single year spikes in abundance of age-0 cutthroat. Abundant age-0 cutthroat in those years did not translate to a corresponding increase in older age cutthroat in subsequent years (**Figure 16-4**).

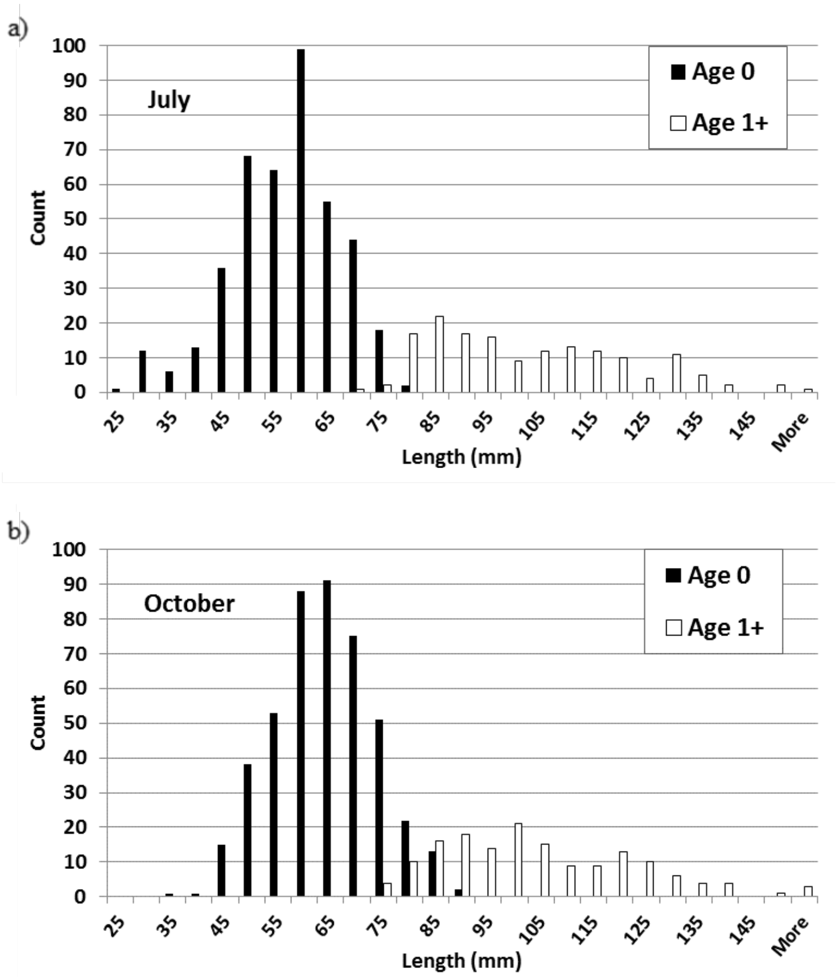

Figure 16-3. Aggregated length-frequency histograms for all a) July and b) October samples with age class assignment.

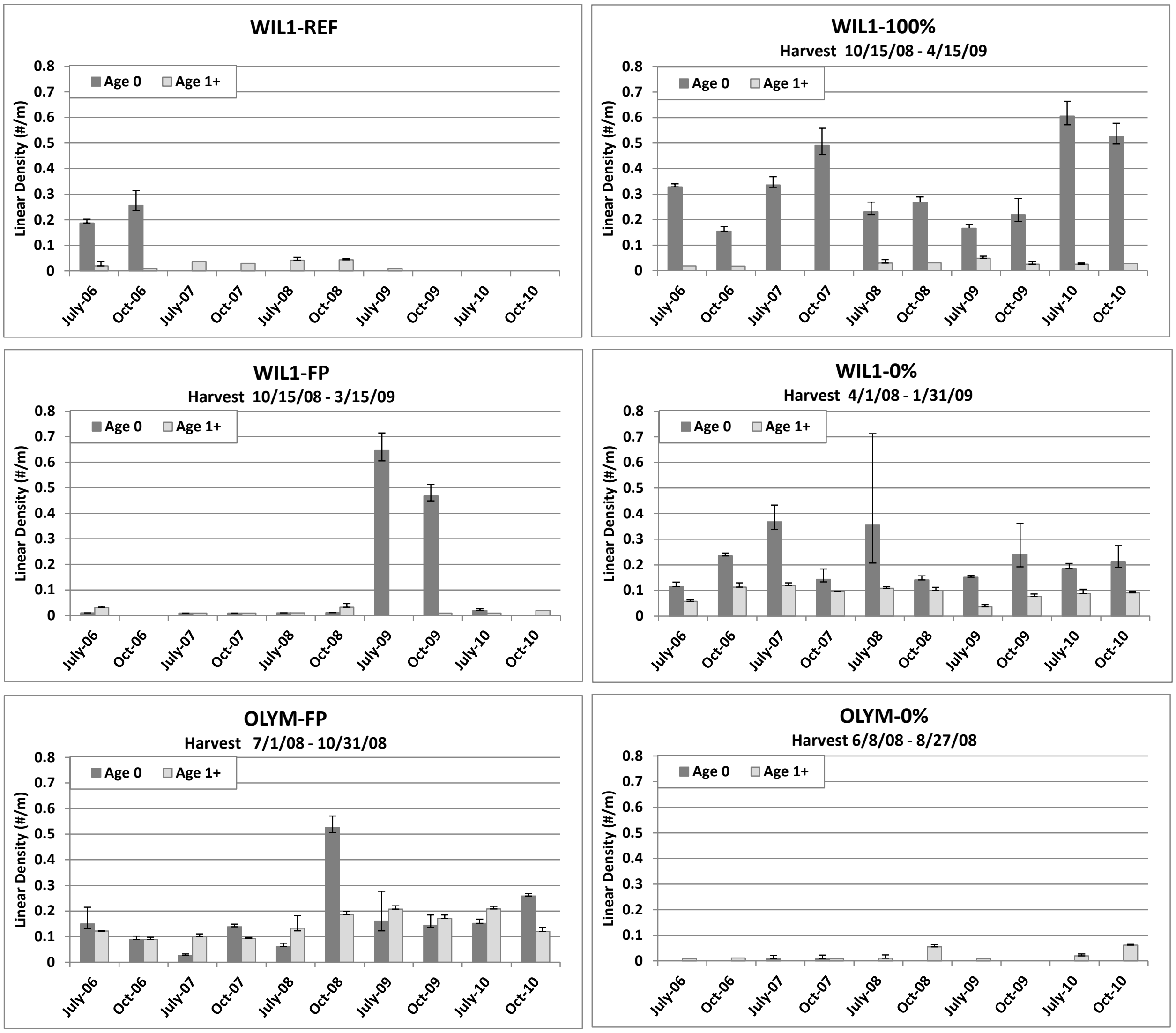

Figure 16-4. Linear density (fish/m) of age-0 and age-1+ cutthroat trout across all sites and sample dates.

16-5.2. FISH DENSITY RELATIVE TO PHYSICAL STREAM HABITAT

Variability in total fish density was not correlated with physical stream habitat metrics such as stream gradient and percent pool area that varied across sites. In addition, we found no relationship between age-0 or age-1+ fish density and stream gradient in the July or October samples (all P-values >0.15). We also found no relationship between age-1+ fish density and percent pool area in the July or October samples (P = 0.14 and 0.92, respectively). There was, however, suggestive evidence of a positive relationship between age-0 fish density and percent pool area in both the July and October samples (P = 0.05 and 0.06, respectively; **Figure 16-5).**

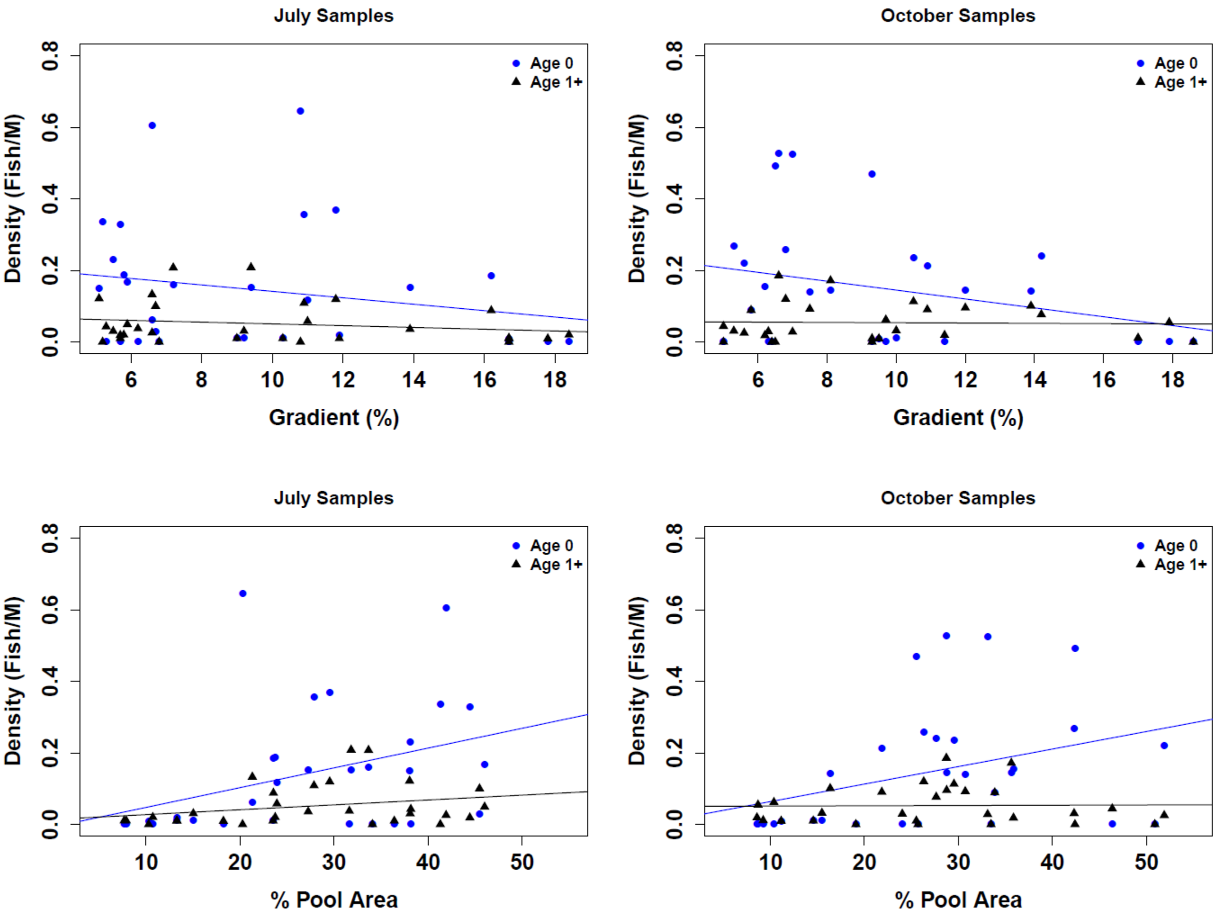

Figure 16-5. Linear fish density vs. 'reach gradient' (top row) and '% pool area' (bottom row) for both age-0 and age-1+ fish on all sample dates. The left panels represent the July samples and the right panels represent the October samples. The lines are regression fit and are color coded by cohort (blue is age-0 and black is age-1+).

16-5.3. FISH CONDITION

We found a strong relationship between fish length and weight, and used that relationship to develop an equation to calculate an expected fish weight (**Figure 16-6**). Average fish condition (actual weight/expected weight) across all sites and years for age-0 fish in July was 0.98, relative to 0.94 in October. Similarly, fish condition for age-1+ fish was 1.00 in July and 0.95 in October. Analysis of variance revealed that there was an effect of sample month on fish condition for both age groups ($P < 0.001$). There was no evidence ($P > 0.05$) of an age effect or of a sample month × age interaction. In addition, we found no relationship between fish density and fish condition in the July or October samples ($P = 0.96$ and 0.54, respectively; **Figure 16-7**).

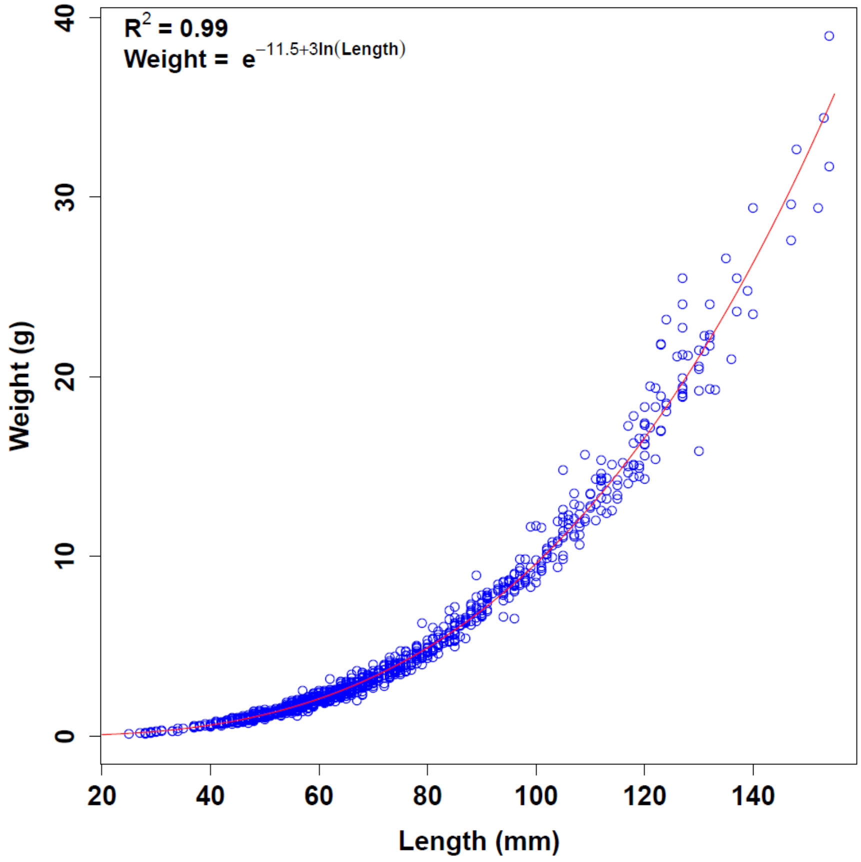

Figure 16-6. Relationship between length and weight of cutthroat trout across all sample sites and years.

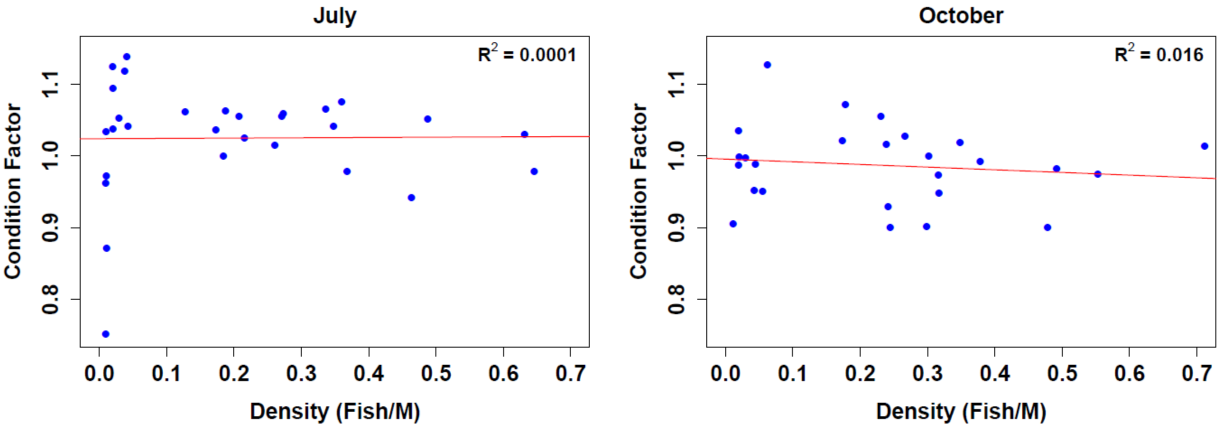

Figure 16-7. Average fish condition factor vs. linear fish density by site for all July and October samples. The red line is regression fit.

16-5.4. PIT TAGGING AND RECAPTURE

In total, we implanted 419 coastal cutthroat trout with PIT tags across our six study catchments over the course of the study. We removed 66 fish that were PIT tagged in the final sampling period (October 2010) from the analysis of tagged fish because there was no subsequent sampling period where those fish could have been potentially recaptured. Therefore, information on fish emigration and mortality presented here is based on the 353 fish implanted with PIT tags between July 2006 and July 2010.

Consistently low PIT tag recapture rates over the course of the study period provide strong evidence of a high level of fish emigration from and/or mortality within our study reaches. Of the 353 fish that could have been recaptured at least once, only 101 (28.6%) of those individuals were recaptured at least one time (**Table 16-2**). The percent of PIT-tagged fish that were recaptured during multiple surveys dropped exponentially, with only 5.7% and 2.6% of eligible PIT-tagged fish being recaptured during two and three future surveys, respectively. There were 179 individual fish that were PIT tagged during or before the October 2008 sample date, and therefore candidates for recapture in four future surveys. Of these 179 fish, only 1 (0.6%) was recaptured on four separate occasions. No PIT-tagged fish were recaptured more than four times.

Table 16-2. Recapture rates of PIT-tagged fish. Note that recapture rates were calculated using only fish that could have potentially been recaptured during a given number of surveys.

Potential Recapture Events	PIT Tags Implanted	Number Recaptured	% Recaptured
1	353	101	28.6
2	299	17	5.7
3	230	6	2.6
4	179	1	0.6
5	154	0	0.0

We also investigated the rate of PIT tag recovery seasonally, focusing separately on the interval between the July and October samples in a given calendar year, and the interval between the October sample and the July sample of the next year. A larger proportion of the population of fish either died or emigrated out of the site in the October–July interval relative to the July–October interval (94.7% versus 58.8%, respectively). This may be due, in part, to the fact that the average October–July interval (269 days) was much longer than that from July–October (96 days). The rate of decline, or percent decline per day, however, was higher for the July–October interval relative to that of October–July (0.61%/day versus 0.35%/day, respectively; **Table 16-3**).

There was evidence of a relationship between the probability of PIT-tag recovery and average fish length ($P = 0.09$). A 10 mm increase in fish length was associated with a reduction in recovery of 2.7% (95% CI is −5.7% to 0.3%; **Figure 16-8**).

Table 16-3. Summary of recapture rates of PIT-tagged fish that were recaptured in the sampling period immediately following that in which they were tagged.

Interval	Average # Days	PIT Tags Implanted	Actual Recaptures	% Decline	% Decline/Day
July–October	96	182	75	58.8	0.61
October–July	269	171	9	94.7	0.35

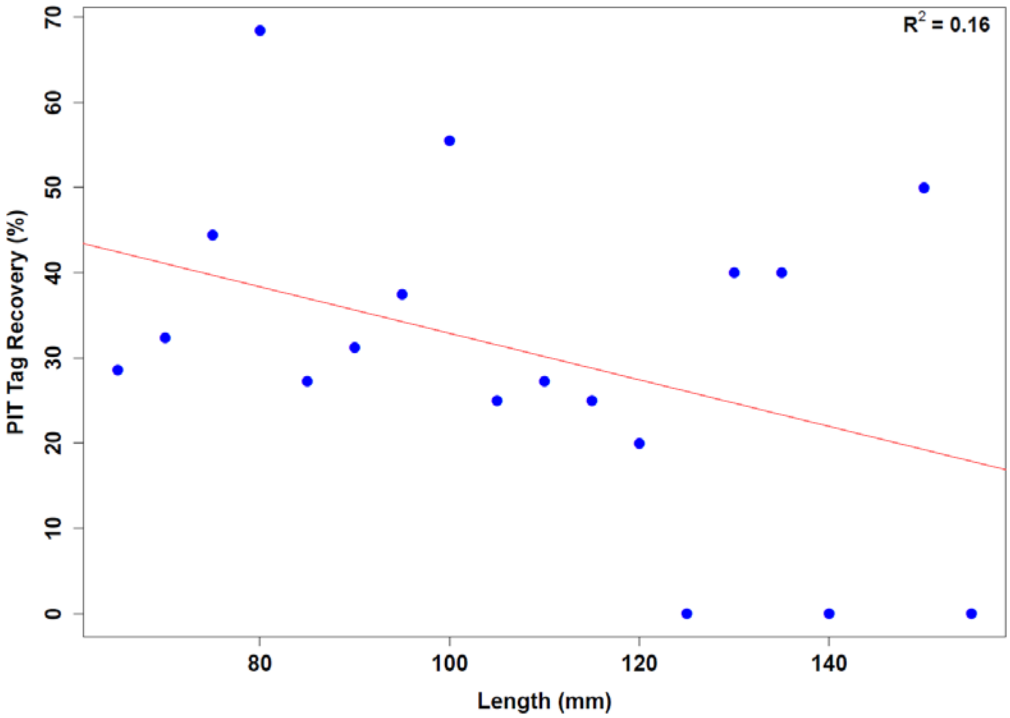

Figure 16-8. Percent PIT-tag recovery versus fish length. The red line is regression fit.

16-5.5. FISH GROWTH

In addition to assessing fish condition, we also investigated growth of individual (PIT-tagged) fish, and fish within the age-0 cohort as a whole. A general linear model fit to size data from recaptured PIT-tagged fish revealed that a log-linear relationship exists between specific growth and initial fish size (P = 0.002; **Figure 16-9**). The trend lines for the two distinct growth intervals between sample dates (July–October versus October–July) do not differ in slope on the log scale (P = 0.89). For each additional 1 gram of initial weight, the growth rate is reduced by 7.1% (95% CI is 2.9–11.2% decrease). Note, however, that the data for the July–October interval only covers about half the temporal range of that for the October–July interval for the independent variable (weight), so these relationships should be considered with some caution.

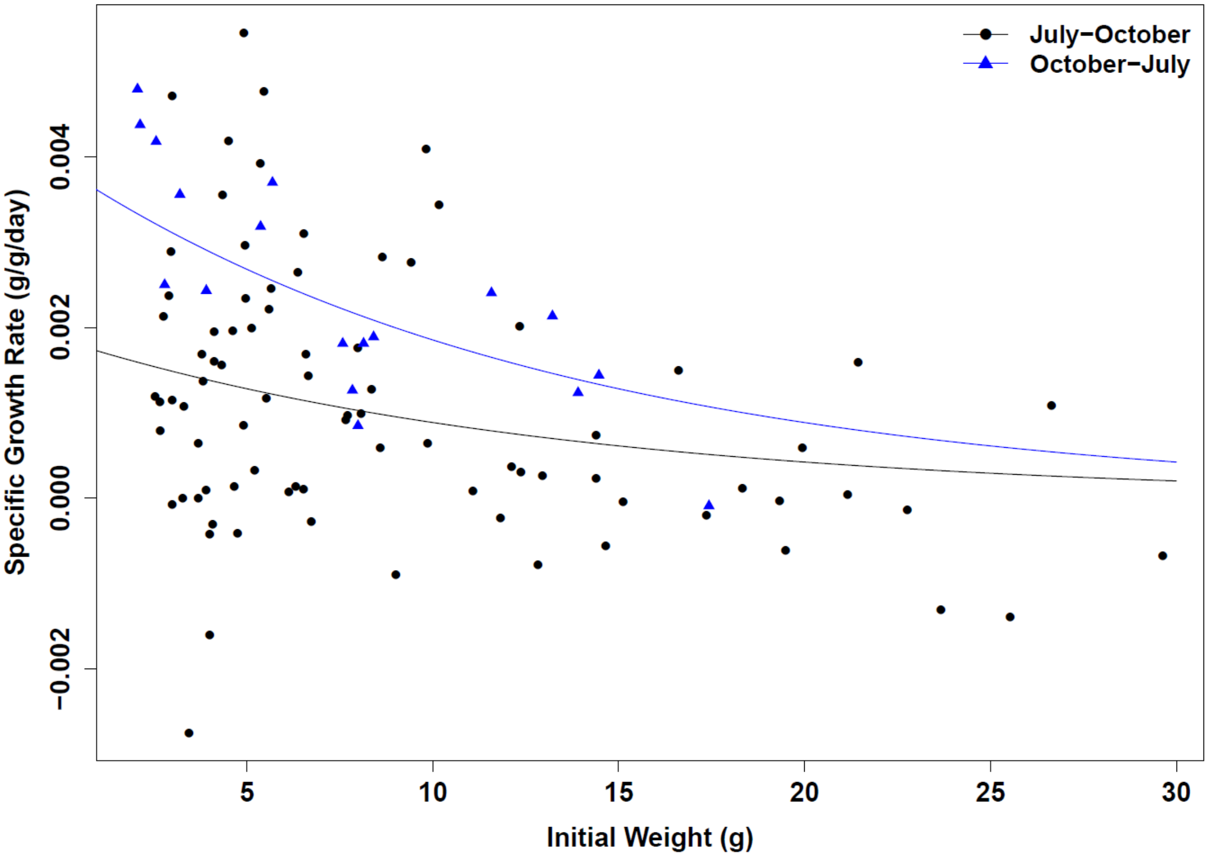

Figure 16-9. Specific growth versus initial weight for tagged fish. The lines are from an analysis of covariance fit on the log-transformed specific growth.

Comparing specific growth rate for age-0 fish, we found that neither sample month (P = 0.31) nor year (P = 0.88) had a significant effect, and no interaction existed between them (P = 0.99). Averaged over all samples, age-0 cutthroat in the interval between the July and October sample dates had a specific growth rate of 0.0058 g/g/day (95% CI is 0.0034–0.0081). Similarly, age-0 cutthroat had an average specific growth rate of 0.0042 g/g/day (95% CI is 0.0018–0.0065) in the interval between the October and July sample dates.

For age-0 fish, we found no relationship between fish density and growth in either the interval from July–October or that from October–July (P = 0.22 and 0.84, respectively). Similarly, for tagged fish, no relationship existed between density and fish growth in either the July–October or October–July interval (P = 0.63 and 0.81, respectively; **Figure 16-10**).

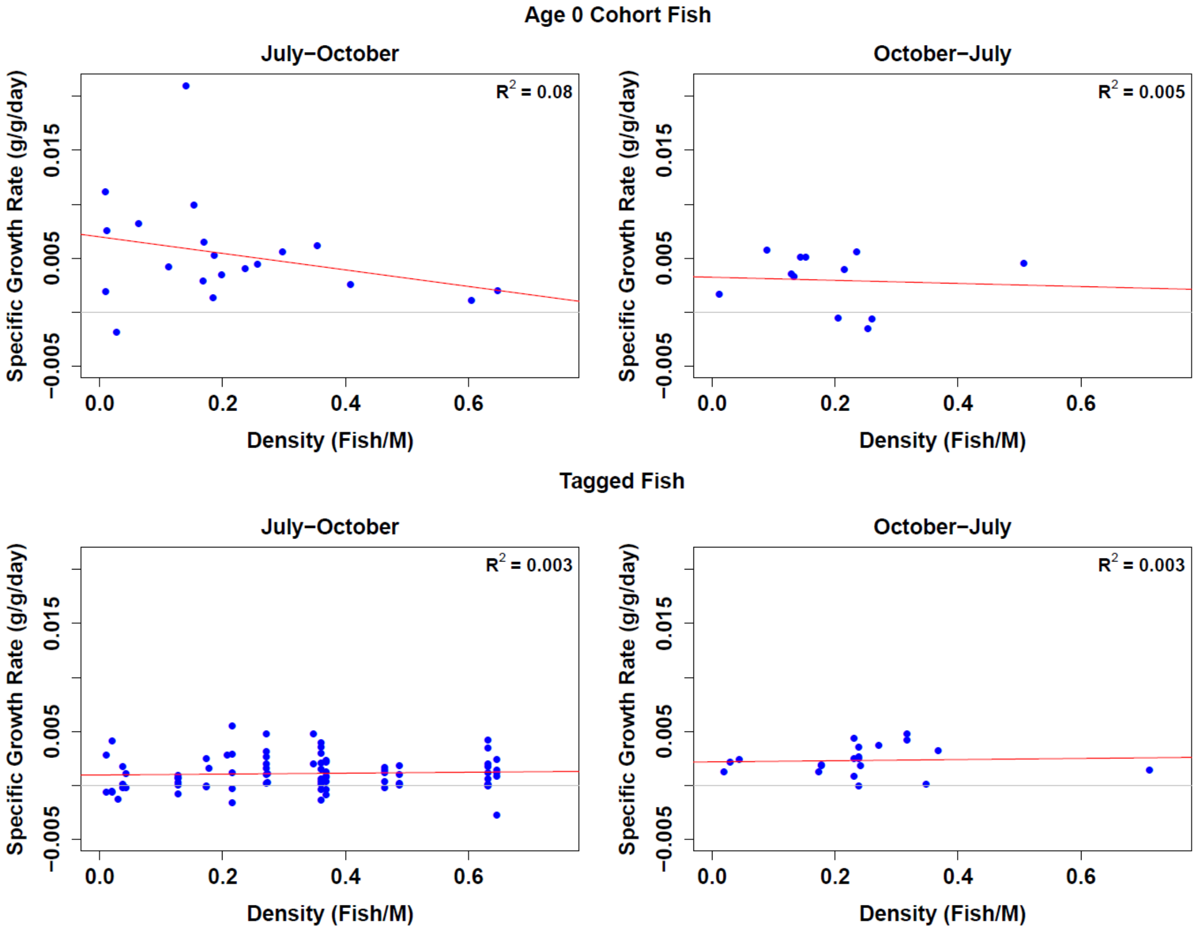

Figure 16-10. Average specific growth rate vs. linear fish density by site for age-0 cohort fish (top row) and tagged fish (bottom row). The left panels represent the July–October growth interval and the right panels represent the October–July growth interval. The red line is regression fit.

16-6. DISCUSSION

Due to a variety of logistical and biological constraints, only six of 17 candidate Type N sites were suitable for inclusion in the fish component of the study. Site selection requirements for the primary elements of the Type N study required selection of small catchments to allow for complete application of harvests and buffer treatments. The small catchment criteria resulted in the selection of several study sites well upstream from the actual extent of fish use, preventing the evaluation of fish response immediately downstream from the treatments. In other sites, stream network junctions dissected potential sample reaches and rendered them unsuitable for inclusion. This limitation prevented us from directly evaluating fish response in headwater streams to upstream timber harvest across a variety of riparian buffer prescriptions, as was the original intended focus of the Type N Study. Instead, we focused on assessing the fish populations, specifically coastal cutthroat trout, immediately downstream from the extreme upstream limit of fish distribution to help develop a better understanding of these unique habitats.

Previous work in headwater basins suggests that cutthroat trout occur at their highest densities in small streams (Murphy *et al.* 1986; Rosenfeld *et al.* 2000, 2002; Walter *et al.* 2014). For instance, Rosenfeld and colleagues (2002) estimate cutthroat trout abundance of 0.9–1.0 fish/m in small headwater streams with a channel width between one and three meters. These studies, however, have focused primarily on headwater basins in their entirety, and not specifically on the segments of stream habitat at the upstream extent of fish distribution. We consistently found cutthroat trout density in our study reaches to be relatively low in comparison with these other studies. Across all sites, samples, and years, for instance, the density of older age-class fish (age-1+) never exceeded 0.21 fish/m, and was consistently below 0.1 fish/m at four of our six study sites. Density of age-0 fish was more variable across sites and years than that of age-1+ fish, with peaks in density for some samples around 0.6 fish/m. These peaks in age-0 density, however, did not result in a corresponding increase in age-1+ fish in future samples at the same site. The high level of both spatial and temporal variability seen in age-0 fish density suggests that their abundance, or furthermore, presence, at the extreme upstream extent of fish distribution may be directly related to whether or not spawning occurred and/or the success of redds within these habitats the preceding spring. While we did not directly assess spawning in our surveys, the fact that age-0 fish have a limited ability to swim upstream in these smaller streams, where blockages to the upstream movement of fish can be common, and that there is no opportunity for fish to drift into these habitats from upstream, support this hypothesis.

We found no relationships between total linear fish density and stream habitat characteristics, such as stream gradient and percent pool area, which are often thought to be associated with increased fish abundance. Despite a lack of statistical significance, there were patterns expressed in the data suggesting a positive relationship between age-0 fish density and both increased percent pool area and lower stream gradient. The combination of lower stream gradient and increased pool area may result in lower overall stream power within some stream reaches, potentially resulting in an increased ability for smaller (age-0) fish to persist in these habitats. Other studies have documented a strong association between adult cutthroat trout abundance and these habitat variables (Hartman and Gill 1967; Hartman *et al* 1996; Rosenfeld *et al* 2000), but we saw no evidence of a density/habitat relationship for age-1+ fish similar to that seen for age-0 fish. All of our stream reaches were similar in their small size, and we hypothesize that the low productivity of these stream systems may be the single most important limiting factor for fish, particularly larger individuals with greater caloric needs. Trotter (2000) proposed a similar hypothesis, suggesting that fish density steadily declines to zero as one moves upstream toward the upper extent of fish distribution. Low fish density in the smallest of fish-bearing streams like those included in this study may be associated, at least in part, with low food availability.

We did not directly investigate fish diet or food availability in the fish component of the Type N Study, but did assess both fish specific growth rate and fish condition. Specific growth rate and condition of fish in our study reaches was lower than that observed in other studies that focused on fish in downstream habitats or throughout headwater basins in their entirety (Harvey 1998; Berger and Gresswell 2009). Harvey (1998) reports that growth rate for cutthroat trout is negatively correlated with distance upstream within the Little Jones Creek basin in California. Specific growth rate in our study reaches was higher for smaller fish, relative to larger fish, regardless of season. These results may also support Trotter's (2000) hypothesis regarding low food availability in headwater streams and its potential effect on both fish growth and density in these habitats. It is possible that larger fish may simply not be able to find sufficient food resources in these small habitats to sustain continued growth. Note that specific growth rate for age-0 fish, which were too small to tag across a representative range of fish size, and age-1+ (PIT tagged) fish were calculated differently. The specific growth of age-0 fish was estimated by calculating the pooled average weight of all age-0 fish sampled. This method may overestimate actual growth for a cohort because it does not account for size-specific mortality within that cohort, so these findings must be considered with some caution. In addition, we found that fish condition was significantly lower in October than in July for both age classes of fish, again potentially suggesting a food limitation, particularly during the summer months when stream flows are at their lowest. Boss and Richardson (2002) found that during summer months, food availability in streams can limit cutthroat trout growth and survival, supporting this hypothesis. We did not find any evidence that fish density was significantly correlated with either fish condition or growth.

Trotter (2000) also reports a variety of previous studies that suggest fish in small streams near the upstream extent of fish distribution have relatively short lifespans. While we have no way of directly assessing fish mortality in our sites, this assertion may be supported by our PIT tagging and recapture data. We found >90% of tagged individuals emigrated from the study reaches or died within one year of tagging, and no individuals persisted more than two years, providing strong evidence of a high rate of fish mortality within and/or emigration from these small stream reaches. This, coupled with the fact that age-1+ fish density was relatively consistent at individual sites over time, suggests that fish in the habitats at the upstream extent of fish distribution may be highly mobile. It should be noted that we assumed growth, movement, and mortality of PIT-tagged fish were similar to that for untagged fish, which is common in studies involving PIT tagging (Roni *et al.* 2012), and also that PIT tag loss after implanting did not impact our findings (Meyer *et al.* 2011).

This work, while limited, documents the relatively low abundance and growth rate of cutthroat trout populations in headwater stream reaches at the extreme upstream extent of fish distribution in western Washington. We found that these habitats tended to support lower densities of fish than typically reported in the reviewed literature for headwater basins as a whole, and that the cutthroat trout in these habitats grew more slowly and were smaller on average with a lower condition factor than those reported in these studies. Our findings support the hypotheses of researchers such as Trotter (2000) who contend that fish in these habitats are living, “at or near the limit of available stream resources”, be those resources food, space (habitat), or both. This work was a case study and therefore may have limited inference beyond sites similar to those that we sampled. To better understand the characteristics and dynamics of these unique habitats at the upstream extent of fish distribution, additional work directly comparing these stream reaches to the broader population of headwater streams is needed.

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CHAPTER 17 - SUMMARY AND DISCUSSION

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

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17-1. INTRODUCTION

Headwater streams comprise a majority (over 65%) of the total stream length on forestlands in western Washington (Rogers and Cooke 2007) and are largely understudied relative to their frequency in Pacific Northwest forests (Richardson and Danehy 2007). Headwater streams differ substantially from larger, typically fish-bearing, streams in their physical, chemical and biotic attributes, providing habitats for a range of unique species (Meyer *et al.* 2007; Richardson and Danehy 2007). As noted in this study, forest practices have diverse effects on headwater systems; however, the mechanisms by which forest practices act on the biotic assemblages in headwater streams are challenging to identify, largely due to interactive effects that can be difficult to disentangle (Richardson and Danehy 2007).

We used a Before-After Control-Impact (BACI) design (*sensu* Smith 2002) to develop the Type N Experimental Buffer Treatment Study (hereafter Type N Study). Study sites were headwater basins consisting of Type N streams located in managed second-growth forests. We evaluated four experimental treatments (**Figure 17-1**):

1) **Reference:** unharvested references previously in the harvest rotation but withheld from harvest for this experiment (n = 6),

2) **100% treatment:** two-sided 50-ft (15.2-m) riparian leave-tree buffer along the entire riparian management zone (RMZ; n = 4),

3) **FP treatment:** two-sided 50-ft (15.2-m) riparian leave-tree buffer along at least 50% of the RMZ, consistent with the current Forest Practices buffer prescription for Type N streams (n = 3), and

4) **0% treatment:** clearcut harvest throughout the entire RMZ (n = 4).

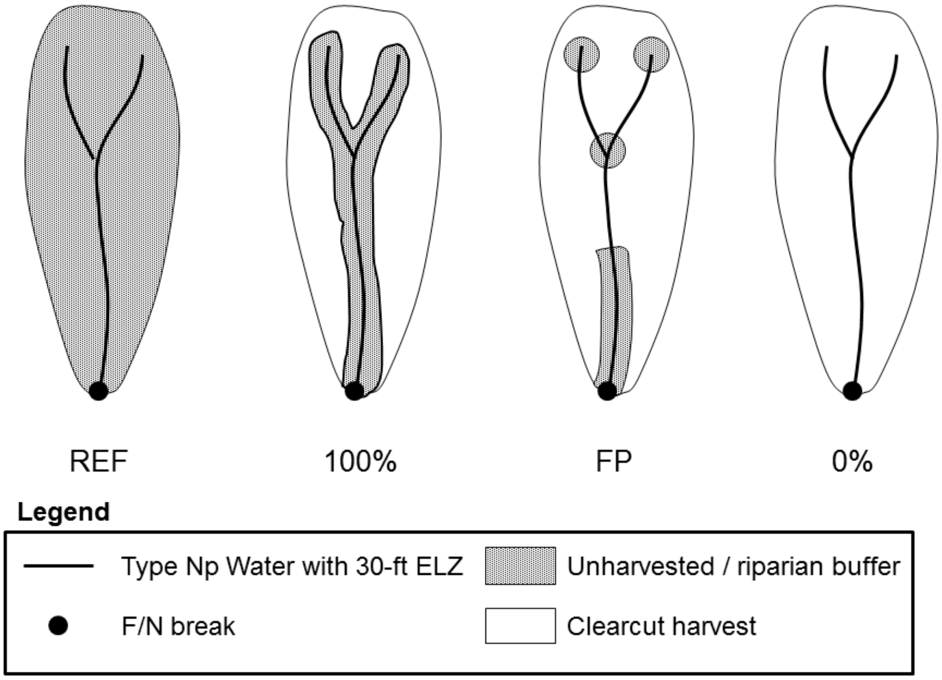

Figure 17-1. Four experimental treatments included in the Type N Study, including unharvested reference sites (REF) and sites receiving clearcut harvest with one of three riparian buffer treatments along the Type Np Water RMZ: two-sided 50-ft (15.2-m) riparian buffers of 100%, Forest Practices (FP), and 0%.

We replicated the four treatments across 17 study sites distributed across five blocks (**Table 17-1**) and collected data pre- (2006–2008) and post-harvest (2009–2011). The FP and 100% treatments included 56-ft (17.1-m) radius buffers around Type Np intersections (i.e., the intersection of two or more Type Np Waters[1]) and perennial initiation points at the uppermost points of perennial flow (PIPs)[2]. At all sites, the entire stream length was protected by a two-sided 30-ft (9.1-m) equipment limitation zone (ELZ)[3]. The proportion of the stream length buffered for our FP treatment sites was 55, 62 and 73%, more than the minimum required under Forest Practice rules. The additional protection was related to the numbers and locations of sensitive sites that required buffering as well as, though less frequently, logistic constraints that made harvest of some stream reaches unfeasible. In four study sites, some areas outside of the RMZs (ranging from 2% to 15% of the total Type N basin area) could not be harvested, again due to regulatory and logistic constraints (see Chapter 3 – *Management Prescriptions*).

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[1] A Type Np Water (i.e., perennial non-fish-bearing stream) includes 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 (WAC 222-16-030).

[2] The uppermost point of perennial flow is the point where perennial flow in the stream channel network begins.

[3] An equipment limitation zone (ELZ) is a 30-ft (9.1-m) wide zone measured horizontally from the outer edge of the bankfull width of a Type Np Water or Ns Water (i.e., seasonal). It applies to all perennial and seasonal non-fish-bearing streams (WAC 222-16-010).

Table 17-1. Number of study sites included in the Type N Study, by block. Block acronym in parentheses.

Block	Number of Sites	Treatment			
		REF	100%	FP	0%
Olympic (OLYM)	4	1	1	1	1
Willapa 1 (WIL1)	4	1	1	1	1
Willapa 2 (WIL2)	4	2	1	0	1
Willapa 3 (WIL3)	2	1	1	0	0
South Cascade (CASC)	3	1	0	1	1
Total	17	6	4	3	4

We evaluated the effectiveness of the Type N riparian buffer prescription in meeting the goals of the Forest Practices rules. Though the primary focus was stream-associated amphibians, we also evaluated changes in riparian tree mortality and tree fall, instream wood recruitment and loading, stream temperature and shade, discharge, nutrient export, suspended sediment export, channel characteristics, litterfall input and detritus export, biofilm and periphyton, macroinvertebrate export and fish.

17-2. RIPARIAN AND INSTREAM LINKAGES IN HEADWATER STREAMS

The size of a stream and its position in the landscape influences its response to modifications of the adjacent uplands. Higher-order stream channels with broad floodplains are frequently disconnected (i.e., decoupled) from hillslopes (Church 1992). In these systems, the valley floor buffers the stream from most direct and immediate inputs of runoff, sediment, organic debris and nutrients from adjacent hillslopes. Inputs are commonly stored in the floodplain, delaying delivery to the stream channel and providing an opportunity for mixing of inputs from various sources and their modification by organisms (Ward *et al.* 1999; Amoros and Bornette 2002; Naiman *et al.* 2005). Conversely, in headwater streams, the channel often occurs in narrow V-shaped valleys with direct connections (i.e., coupling) between the hillslope and channel (Gomi *et al.* 2002; Richardson and Danehy 2007).

In this study, we predicted that the unique relationship between the stream and adjacent hillslopes in headwater stream basins would influence the response of study sites to forest management. We identified linkages between variables directly influenced by forest management activities, in this case clearcut timber harvest with variable length riparian buffer treatments, and response variables (**Figure 17-2**). Based on relevant literature, we identified linkages between upland and riparian areas with instream features (e.g., wood, channel, temperature, shade), key input processes (e.g., sediment, litter, wood) and instream biotic responses (e.g., biofilm, macroinvertebrates, amphibians). These linkages and how they are supported or not supported by our study results, provide the basis for the structure and presentation of our results.

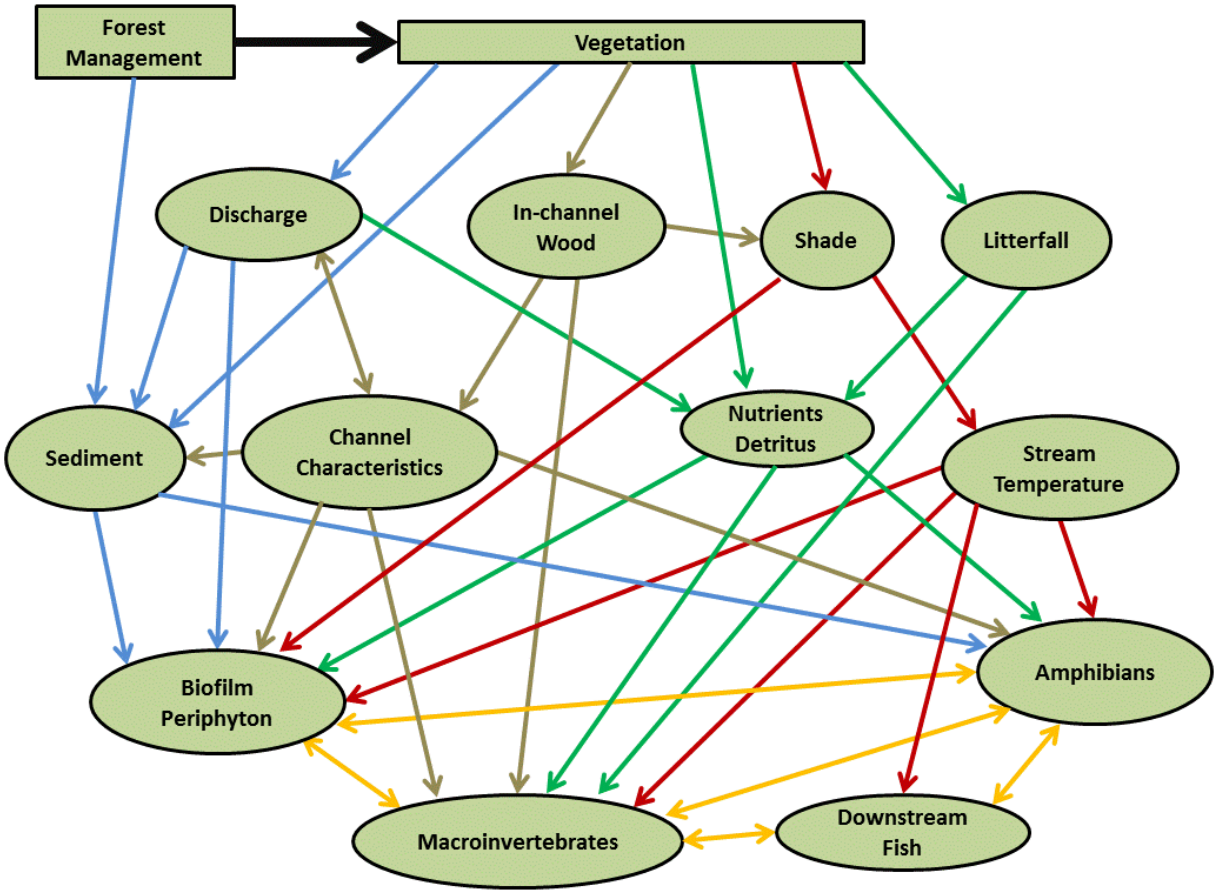

Figure 17-2. Linkages identified between upland and riparian areas, headwater stream conditions, and biotic communities. Rectangles are variables that were directly manipulated as part of the experimental study design. Ovals are dependent variables with the potential to be directly or indirectly affected by timber management activities, specifically riparian buffer treatments. Different colored arrows describe the linkages between forest management and response metrics. Arrows of a like color link variables that are suspected to either directly (directional arrows) or reciprocally (bidirectional arrows) influence each other.

17-3. SUMMARY OF RESULTS AND INTERACTIONS AMONG RESPONSE VARIABLES

We summarize what we regard as the most important study results, including response variables that differed significantly by treatment (**Table 17-2** and **Table 17-3**) and variables for which we expected a change but did not detect one. A comprehensive list of all of the response variables included in the study is presented in **Supplement 1**. We present results as the estimated pre- to post-harvest change by treatment, relative to the change in the reference, unless otherwise stated. We also discuss the interactions among response variables and the implications for ecological functions in headwater streams. We describe response details, including the hypotheses on which we based evaluations and the magnitudes of change in response, in the preceding chapters.

Table 17-2. Summary of statistically significant results for responses included in post-harvest comparisons between treatments (i.e., statistical comparisons not intended to be done in the framework of the overall BACI design). Only variables with a significant treatment effect are displayed (i.e., P-value for a difference between treatments ≤0.1; see Supplement 1 for a complete list of responses). Arrows indicate the direction of the difference in the change for each pairwise comparison. Significance is indicated as ** for P ≤0.05 and * for 0.05> P ≤0.10. Pairwise treatment comparisons are always in the order of the first treatment listed, e.g., REF v. 100% indicates the direction of change in the 100% treatment relative to the change in the reference. For simplicity, metrics listed under the same variable group and having the same response are listed in the same row, separated by an ‘&’. N/A indicates comparisons that were not made because treatment implementation required the complete removal of trees in the 0% treatment.

			P-values					
			REF v.			**100% v.**		**FP v.**
Chapter	**Variable**	**Units**	**100%**	**FP**	**0%**	**FP**	**0%**	**0%**
5 - Stand Structure and Tree Mortality Rates in Riparian Buffers	Tree mortality	% stems/yr & % basal area/yr RMZ		↑ **	N/A	↑ *	N/A	N/A
		% stems/yr PIP	↑ *	↑ **	N/A	↑ **	N/A	N/A
		% basal area/yr PIP	↑ *	↑ **	N/A		N/A	N/A
	Change in stand structure	% Δ stems & % Δ basal area RMZ		↑ **	N/A	↑ **	N/A	N/A
		% Δ stems PIP		↑ **	N/A	↑ **	N/A	N/A
		% Δ basal area PIP	↑ **	↑ **	N/A	↑ **	N/A	N/A
6 - Wood Recruitment and Loading	Tree fall rates	Tree fall RMZ (stems/yr)		↑ **	N/A	↑ *	N/A	N/A
		Tree fall PIP (stems/yr)	↑ **	↑ **	N/A		N/A	N/A
	Large wood recruitment	Pieces/ha/yr RMZ					↓ *	↓ **
		Volume/ha/yr RMZ			↓ **		↓ **	↓ **
		Pieces/ha/yr PIP	↑ **				↓ **	↓ *
		Volume/ha/yr PIP	↑ **	↑ *			↓ **	↓*
	New wood cover (i.e., slash and windthrow)	% cover post yr 1		↑ **	↑ **			
		% cover post yr 2			↓ **		↓**	↓ **

Table 17-3. Summary of statistically significant results for responses included in the BACI analysis. Only variables with a significant treatment effect are displayed (i.e., P-value for a difference between treatments ≤0.1; see Supplement 1 for a complete list of responses). Arrows indicate the direction of the difference in the change for each pairwise comparison. Significance is indicated as ** for P ≤0.05 and * for 0.05> P ≤0.10. Pairwise treatment comparisons are always in the order of the first treatment listed, e.g., REF v. 100% indicates the direction of change in the 100% treatment relative to the change in the reference. For simplicity, metrics listed under the same variable group and having the same response are listed in the same row, separated by an ‘&’. -- indicates comparisons that were not made.

			P-values					
			REF v.			**100% v.**		**FP v.**
Chapter	**Variable**	**Metric**	**100%**	**FP**	**0%**	**FP**	**0%**	**0%**
6 - Wood Recruitment and Loading	Wood Loading	Total small wood	↑ **	↑ *	↑ **		↑ **	↑ *
		Functional small wood	↑ **	↑ **	↑ **			
		Total & Functional large wood	↑ **	↑ **	↑ **			
7 - Stream Temperature and Cover	Riparian Cover	Canopy and topographic density post yr 1		↓ **	↓ **	↓ **	↓ **	↓ **
		Canopy and topographic density post yr 2	↓ **	↓ **	↓ **	↓ **	↓ **	↓ **
		Effective shade post yr 1 & 2	↓ **	↓ **	↓ **	↓ **	↓ **	↓ **
		Canopy closure 1-m post yr 1 & 2		↓ **	↓ **	↓ **	↓ **	↓ **
		Canopy closure 0-m post yr 1			↓ **		↓ **	↓ **
		Canopy closure 0-m post yr 2		↓ **	↓ **		↓ **	↓ **
	Water Temperature	Daily max post yr 1 & 2	↑ **	↑ **	↑ **		↑ **	↑ **
		Daily min post yr 1	↑ **	↑ **	↑ *		↑ *	
		Daily min post yr 2	↑ **		↑ **			↑ **
		Diel range post yr 1	↑ **	↑ **	↑ **		↑ **	↑ **
		Diel range post yr 2			↑ **		↑ **	↑ **
8 - Discharge	Discharge	Mean discharge		↑ **	↑ **	--	--	--
		Baseflow	↓ *	↑ *	↑ *	--	--	--
9 - Nutrient Export	Nutrients	Total nitrogen load	↑ *	↑ **	↑ **		↑ **	
		Nitrate-nitrogen load	↑ **	↑ **	↑ **		↑ **	
		Total phosphorus load	↑ **	↑ **	↑ **			

Table 17-3. (continued)

			P-values					
			REF v.			100% v.		FP v.
Chapter	Variable	Metric	100%	FP	0%	FP	0%	0%
11 - Stream Channel Characteristics	Stream Hydrology	Wetted stream & Bankfull channel widths			↓**		↓ **	↓ **
	Stream Channel Units	Pool length	↑ **	↑ **	↑ **			
		Channel rise by steps			↓**		↓ **	↓ **
12 - Litterfall Input and Detritus Export	Litterfall Input	Total litterfall			↓*		↓ **	
		Total leaf (conifer+deciduous) & Deciduous			↓**		↓ **	↓ **
	Detritus Export	Total detritus			↓**	↓**	↓ **	
		Total coarse particulate	↑ *		↓**	↓**	↓ **	
		Total leaf (conifer+deciduous)					↓ **	
		Total conifer					↓ **	↓ *
		Total wood & Total fine particulate			↓**	↓**	↓ **	
		Total miscellaneous				↓**	↓ **	
14 - Macroinvertebrate Export	Functional Feeding Group	Gatherer export (mg day^{-1})		↑ **		↑**		↓ **
		Parasite export (# day^{-1} & mg day^{-1})	↓ **	↓ **	↓ **			
		Scraper export (# day^{-1})	↓ **		↓ **	↑ *		
		"Unknown" export (mg day^{-1})				↓ **	↓ **	
	Taxonomic Composition	Dixidae export (# day^{-1})	↓ **		↓ **	↑ *		
		Dixidae export (mg day^{-1})	↓ **			↑ **		
15 - Stream-associated Amphibians	Amphibian Density	Coastal Tailed Frog larval density	↑ **	↑ **			↓ **	↓ **
		Coastal Tailed Frog post-metamorph density			↑ *		↑ **	↑ **
		Giant salamander density		↓ **		↓ **		↑ **

17-3.1. TREE MORTALITY, WOOD RECRUITMENT AND LOADING, AND STREAM CHANNEL CHARACTERISTICS

We identified linkages between riparian forest management practices, wood recruitment and in-channel wood loading (**Figure 17-2**). There are several mechanisms by which forest management affects wood recruitment (i.e., new wood entering into or over the bankfull channel) and loading (i.e., total wood [e.g., pieces, volume] within the bankfull channel) through effects on streamside forests (Benda *et al.* 2003; Boyer *et al.* 2003; Hassan *et al.* 2005). Clear-cut harvest of streamside forests can result in large inputs of mostly small wood from tops and branches that enter the stream during harvest (Gomi *et al.* 2001; Jackson *et al.* 2001; Maxa 2009). For example, in a study of the impacts of clear-cut harvest on headwater streams in the Coast Range of Washington, over 98% of the stream length was covered with organic debris (i.e., logs, branches, twigs and needles) 1 to 2 m (3.3 to 6.6 ft) deep (Jackson *et al.* 2001). Consistent with these findings, we observed extensive input of logging slash (mostly small wood <10 cm [4 in] diameter) where streamside forests were harvested to the edge of the stream channel (i.e., 0% treatment and unbuffered portions of the FP treatment). No additional wood recruitment occurred in these stream reaches during the post-harvest period due to the absence of standing trees following timber harvest.

Where buffers are retained adjacent to the channel, empirical studies have noted elevated tree mortality rates in the newly established riparian buffers (Grizzel *et al.* 2000; Liquori 2006; Bahuguna *et al.* 2010; Schuett-Hames *et al.* 2012). Exposure of buffer trees to winds after the adjacent timber is removed often results in tree uprooting and stem breakage, with short-term increases in wood recruitment and loading (Grizzel and Wolff 1998; Jackson *et al.* 2001; Schuett-Hames *et al.* 2012). We observed an increase in mean tree mortality and tree fall rates in newly established FP treatment RMZ and PIP buffers. Mortality and tree fall in the FP treatment RMZs and in FP and 100% treatment PIP buffers were significantly higher than comparable areas in unharvested reference sites. While mortality in the 100% treatment RMZ buffers was also elevated, the increase did not differ statistically from the reference sites, but was significantly lower than in the FP RMZs. We observed higher large wood recruitment rates in both RMZ and PIP buffers; however, the comparisons largely lacked significance due to variation among sites.

The response of wood recruitment and loading to forest management and/or natural disturbance varies dependent on the type and magnitude of disturbance to streamside forests (Spies *et al.* 1988; Bragg 2000). The severity and frequency of disturbances affects both the magnitude and timing of wood inputs (Benda and Sias 2003). Wood input processes can be characterized as either chronic or episodic (i.e., catastrophic). Chronic wood input is the recruitment of individual or small groups of trees from streamside forests that occurs gradually over relatively long periods and is typically the result of competition, wind, insects or disease. Chronic wood input provides a relatively stable supply of wood to the channel, producing a wood loading regime that gradually increases with stand age (Hedman *et al.* 1996; Warren *et al.* 2009). Episodic wood input refers to the mortality and input of a larger numbers of trees over shorter periods, typically as a result of severe disturbance events such as high winds, fire, insect outbreaks or landslides (Bragg 2000; Benda *et al.* 2003). Wood input related to severe episodic disturbances typically results in a high initial peak in wood loading followed by decreased wood loading during the stand initiation stage, when there is little recruitment potential from the young forest (Liquori 2000). Disturbance severity modulates the pattern of wood input related to severe episodic events. In contrast to chronic wood inputs, episodic disturbances have longer recurrence intervals (RIs; Benda *et al.* 2003) that generally result in greater oscillations in wood load, with the potential for increased wood load over time (Bragg 2000). Clearcut harvest of streamside trees results in the greatest reduction in wood input and loading over time. Although there is input of logging debris during harvest, removal of streamside trees eliminates potential future wood recruitment for an extended period, resulting in low wood loading levels (Beechie *et al.* 2000; Meleason *et al.* 2003).

We observed site- and plot-scale differences in wood recruitment and loading associated with differences in disturbance frequency and severity in both the pre- and post-harvest periods, regardless of treatment. For example, during the pre-harvest period, a storm with hurricane-force winds affected the southwest Washington Coast. Sites in the Willapa 1 and Willapa 2 blocks sustained extensive tree mortality, tree fall and large wood recruitment due to windthrow, while sites outside of the path of the storm did not. Post-harvest mortality, tree fall and large wood recruitment rates were elevated in riparian buffers at sites exposed to storm-force winds compared to sites where storm-force winds did not occur.

We identified linkages between in-channel wood and channel characteristics, shade, discharge, and sediment. Wood is a primary determinant of channel structure (Harmon *et al.* 1986; Montgomery and Buffington 1998; Gomi *et al.* 2002; Brookshire and Dwire 2003; Hoover *et al.* 2006), especially in small headwater streams (Bilby and Bisson 1998) that typically lack discharge rates capable of transporting wood out of the system (Keller and Swanson 1979; Gurnell *et al.* 2002). Wood functions as a roughness element, creating areas of low energy and slowing the transport of, and acting as a reservoir for, sediment and organic materials (Bilby and Ward 1989; Bilby and Bisson 1998; Benda *et al.* 2005).

While maintaining a riparian buffer along headwater streams served to prevent recruitment of logging slash to the stream, we found variable, but often substantial, quantities of logging slash in stream channels with no riparian buffer. When we considered windthrow and slash in combination, post-harvest in-channel wood cover was greater in the FP and 0% treatments than in the reference in the first year post-harvest. Numbers of pieces of both large and small wood increased in all buffer treatments in the post-harvest period, with concurrent increases in functional wood (i.e., contributing to step formation, bank stability, or hydraulic roughness). The only significant difference in the number of wood pieces between treatments was for total small wood, which had a post-harvest increase that was greater in the 0% than in the 100% and FP treatments. While we did observe an increase in the number of large wood pieces contributing to stream function, most recruited trees were suspended over the stream channel due to narrow channels and valley confinement. In fact, 90% of recruited trees were completely suspended above the stream. While these pieces do not currently contribute to pool, formation, step formation or sediment storage, they do provide shade and cover, a source of small wood as the bark and branches slough into the channel, and we expect them to provide in-channel functions in the future as they decay, break up and enter the stream channel. Finally, we found that small wood played a functional instream role in all sites, with approximately 50% of small wood pieces contributing to instream function, regardless of treatment.

Inputs of wood and logging slash have been shown to restrict wetted stream width (Carlson *et al.* 1990). In our study sites, the pre- to post-harvest change in both wetted and bankfull widths was less in the 0% treatment than all other treatments, including the reference. We also found that the proportion of the channel rise attributed to steps was less in the 0% treatment than the other treatments. We attribute this to the fact that fluvial power in study streams was insufficient to sort harvest-derived wood and form distinctive steps, at least over the two years immediately following timber harvest. Rather, some steps were covered by a matrix of mixed organic (e.g., branches, leaves, needles) and inorganic material (e.g., fines), resulting in a reduction in percent channel rise attributed to steps. Additionally, though we did not directly assess sediment storage within the basin, our visual surveys indicated that, while not statistically significant, instream fines were highest in the 0% treatment, moderate in the FP treatment, and least abundant in the 100% treatment. We attribute this pattern to the fine sediment and litter accumulations associated with increased amounts of in-channel wood and slash at buffer treatment sites. Finally, we observed an increase in pool length in all buffer treatments, which we attribute to wood and debris dams, which dominate pool formation (Lisle 1986; Montgomery *et al.* 1995; Montgomery *et al.* 2003). Instream wood at the downstream ends of pools reduces the rate of water discharge in small streams, increasing pool length by extending the upstream limits of pools, at least during the low flow period (Lisle 1986). We conclude that changes in stream channel characteristics can be attributed to wood recruitment, an increase in channel roughness, and subsequent hydraulic alteration (Jackson *et al.* 2001).

17-3.2. SHADE AND STREAM TEMPERATURE

In forested environments, shade provided by riparian vegetation attenuates incoming solar radiation and is often the single most important variable influencing summer stream temperature (Johnson and Jones 2000; Danehy *et al.* 2005; Groom *et al.* 2011a). Removal of overstory trees within or adjacent to the RMZ decreases riparian canopy cover, allowing more light to reach the stream. Early studies of the direct effects of forest harvest on stream temperature demonstrated dramatic decreases in shade and increased summer stream temperature after harvest (Brown and Krygier 1970; Harris 1977; Feller 1981; Holtby and Newcombe 1982; Beschta and Taylor 1988). More recent studies of contemporary practices (Johnson and Jones 2000; Gomi *et al.* 2006; Gravelle and Link 2007; Groom *et al.* 2011b; Janisch *et al.* 2012; Rex *et al.* 2012; Kibler *et al.* 2013) demonstrated much smaller temperature increases, consistent with the greater shade retention provided by current forest practices and with other site-specific factors that affect the response of streams to upland timber removal.

The reductions in effective shade and canopy closure (1 m above water surface) were consistent with the intensity of buffer treatments. We observed the greatest reduction by the end of the second post-harvest year in the 0% treatment (72% and 78%, respectively), a moderate reduction in the FP treatment (36% and 27%, respectively) and the smallest reduction in the 100% treatment (10% and 5%, respectively). Reduction in canopy closure measured at the water surface by the end of the second year were 2, 15, and 45% in the 100%, FP, and 0% treatments, respectively, due to the cover provided by overlying slash in the unbuffered portions of the stream channel and understory regrowth.

Consistent with reductions in shade, we observed increases in the maximum and minimum daily stream temperatures and diel ranges across all buffer treatments. Mean pre-harvest, July-August maximum daily water temperature averaged 11.1, 12.3, 10.6, and 11.7°C at the REF, 100%, FP, and 0% treatments, respectively. The mean July-August response in maximum daily temperature was 0.7°C in the 100% and 1.1°C in the FP treatment. This is similar to Janisch and colleagues (2012) who detected 0.6° and 1.1°C increases with buffers similar to the 100% and FP treatments. However, the response in our 0% (2.3°C) treatment was greater than the 1.7°C increase seen by Gomi and colleagues (2006) and the 1.5°C increase by Janisch and colleagues (2012). Stream temperature tended to increase with distance downstream from the PIP. While increases in summer maximum daily stream temperatures were consistent with our predictions, the increase seen in the minimum daily stream temperatures was unexpected. Few other studies have evaluated minimum temperature and the results are variable (Johnson and Jones 2000; Mellina *et al.* 2002; Guenther *et al.* 2014). Another unexpected result was that maximum daily stream temperature generally increased post-harvest over a broad seasonal range at most sites, including from early spring through the fall. There are few studies that evaluate year-round temperature change and none using a comparable analytical method. Nonetheless, within the first two years after harvest we observed partial recovery of summer maximum daily stream temperature approximately 100-m downstream of the study reach, but mean monthly maximum temperature was still 0.2°C to 1.6°C higher than pre-harvest in the five sites where this could be evaluated.

17-3.3. DISCHARGE AND SEDIMENT

Forest harvest can alter the magnitude and timing of flow in headwater streams as a result of reductions in interception and evapotranspiration (Lewis *et al.* 2001; Johnson *et al.* 2007; Kuraś *et al.* 2012). Further, changes in the magnitude and/or timing of flow in headwater streams can impact the frequency and magnitude of sediment transporting events (Gomi *et al.* 2005; Alila *et al.* 2009; Kaufmann *et al.* 2009), which are a function of sediment supply and transport capacity (Schumm 1971).

We measured discharge and estimated suspended sediment export (SSE) in the Olympic and Willapa 1 blocks. The change in total water yield was positive for all treatment sites, with discharge increasing in proportion to the area of each basin harvested. However, we observed differences in water yield per unit area of harvest (i.e., specific discharge), with the 100% treatment exhibiting the smallest increase per unit area and the FP and 0% treatments exhibiting larger increases. These observations are consistent with literature showing greater evapotranspiration rates in riparian areas (Tsang *et al.* 2014).

Baseflow response (i.e., RI <~2 day) differed among buffer treatments, with baseflows decreasing in both the OLYM-100% and WIL1-100% and significantly increasing in both the WIL1-FP and WIL1-0%. Baseflow response in the FP treatment only increased significantly for the wetter portion of the baseflow period. For events with an RI between 2 and 7 days, which are likely to be associated with low to moderate intensity rainfall, specific discharge (i.e., volume of water normalized to basin area) increased significantly by 1.5 to 7 mm/day in all treated sites except the OLYM-100% which still exhibited a reduction in flow for that RI. Discharge increased for larger events (e.g., RI >7 day), but the effect size varied by block as well as treatment. For example, the 1.5-year RI (e.g., peak flow) response, which is responsible for most of the geomorphic work, increased significantly in the OLYM-FP and OLYM-0% but not in the OLYM-100% or in any of the sites in the Willapa 1 block.

Although discharge with RIs greater than 7 days increased in all treatment sites, we did not detect a significant change in SSE following harvest. Study sites appeared to be supply limited with respect to suspended sediment, both before and after harvest, with observed turbidity below 3.1 NTU over 95% of the time. The total annual SSE (2 to 108 tons/km^2/yr) was within the range of reported suspended yields for unmanaged small catchments, though greater than expected for sites dominated by competent lithologies (Gomi *et al.* 2005). Eight storm events dominated the suspended sediment budgets of the six buffer treatment sites. These storm events typically occurred in late fall or early winter and generally resulted in SSE spikes in about half of the sites. Given the limited number of sites and sediment producing events, it was not possible to identify strong relationships between changes in discharge and sediment export.

17-3.4. LITTERFALL INPUT, AND DETRITUS AND NUTRIENT EXPORT

Tree removal from stream-adjacent riparian areas reduces short-term litterfall inputs to the stream (Kiffney and Richardson 2010), at least in the short term as vegetation communities reestablish (Schuett-Hames *et al.* 2011). In some cases, the addition of logging slash to streams during harvest helps to retain available litterfall within the stream channel (Brookshire and Dwire 2003; Hoover *et al.* 2006). Research suggests that there is little change in instream detritus storage with a reduction in litterfall inputs (Hetrick *et al.* 1998; Wallace *et al.* 1999; Richardson and Béraud 2014), and little difference in total detritus export from a wide range of treatments following harvest (Piccolo and Wipfli 2002; Wipfli and Gregovich 2002; Musslewhite and Wipfli 2004). In our study, total litterfall was slightly higher after harvest in the 100% treatment, lower in the FP treatment and lowest in the 0% treatment. We noted significant differences only in the 0% treatment for deciduous litterfall and total leaf (deciduous plus conifer) litterfall. Total detritus export from the 0% treatment also decreased following harvest, a response driven by a decrease in both total and coarse particulate organic matter and wood. A decrease in litterfall input and detritus export in the 0% treatment reflects the removal of all trees from that treatment, but may also indicate an increase in detritus retention from the addition of slash and windthrow to the streams.

Nitrogen (N) export is the product of instream concentrations and stream discharge. Vegetation removal via logging can decrease dissolved inorganic nitrogen uptake, resulting in higher nitrate-N concentrations in soil water (Feller *et al.* 2000) and in the stream (Dahlgren 1998). Increased runoff results in an increased capacity for leaching soluble nutrients from the soil and higher inputs to the stream. Mean total N and nitrate-N concentrations increased in all buffer treatment sites. This, along with greater annual discharge, increased N export after harvest. The magnitude of the increase ranged from 8.2 to 32.9 kg ha^{-1} yr^{-1} (7 to 358% increase) for total N, and from 6.3 to 30.0 kg ha^{-1} yr^{-1} (13% to 327% increase) for nitrate-N. These increases were greatest in the 0% treatment, intermediate in the FP treatment, and lowest in the 100% treatment. The relative magnitude of the changes among sites was consistent with our expectations of increased N export with an increase in the proportion of the watershed harvested. Higher post-harvest export of total-N and nitrate-N in all buffer treatments was correlated to the increase in annual runoff, which was correlated with the proportion of the watershed harvested. The increases seen in nitrogen export were similar to the results of earlier studies. The implication of the increase in N export on downstream receiving waters depends upon the N load being delivered and the timing of delivery to the receiving waters.

Like nitrogen, phosphorus (P) exports reflect a combination of instream concentrations and stream discharge. However, unlike nitrate-N, phosphorus is readily adsorbed onto organic material and clay particles and is generally much less mobile in the soil than nitrate (Feller 2005). Increases in phosphorus export after forest harvest are more likely the result of soil disturbance and erosion leading to sediment input to the channel. We did not detect significant post-harvest differences in the concentration of soluble reactive phosphorus (SRP) or total-P in any treatment. However, total-P export in buffer treatments increased after harvest by 21% to 50%. This increase was a function of the low pre-harvest exports, stable concentrations, and increased post-harvest runoff. We observed very little bank disturbance and little evidence of sediment delivery to the channel across all study sites. That we did not observe differences between treatments was likely the result of the riparian buffer and ELZ requirements along the harvested stream reaches, which minimized ground disturbance near the stream and largely prevented sediment delivery.

17-3.5. BIOFILM AND PERIPHYTON

Biofilm and periphyton production are controlled by a variety of factors including discharge, shade and light, nutrients, stream temperature, sediment, channel characteristics, and grazer density (i.e., macroinvertebrates and larval Coastal Tailed Frog; see Stevenson *et al.* 1996). Periphyton production in forested headwater streams is thought to be largely light-limited (Kiffney *et al.* 2003; Mallory and Richardson 2005; Collins *et al.* 2015) with the high degree of shading generally suppressing rates of primary production (Hill *et al.* 1995; Kiffney *et al.* 2003, 2004). Many studies have reported an increase in the level of trophic support provided by autotrophic production following thinning or removal of riparian vegetation that increases light reaching the stream and elevates water temperature (Bilby and Bisson 1992; Kiffney *et al.* 2003; Tiegs *et al.* 2008). Further, increased instream nutrients can increase autotrophic production (Kiffney and Richardson 2001; Greenwood and Rosemond 2005; Liess *et al.* 2009). Although there was a reduction in stream shade across all buffer treatments, along with an increase in stream temperature and nitrogen, there was no detectable increase in biofilm (as measured by ash free dry mass) or periphyton (as measured by chlorophyll *a*).

Biofilm is composed of organic material with embedded algal, bacterial and fungal cells, and its isotopic signature reflects the origin of the dissolved organic matter from which it is composed. Biofilm consisting primarily of terrestrially-derived organic matter, which typically displays a δ^{13}C value lower than that for biofilm containing a significant amount of algal material (Rosenfeld and Roff 1992; Mulholland *et al.* 2000), would be expected prior to application of the treatments, as the low light levels experienced by these streams inhibit algal growth. An increase in the δ^{13}C values and/or δ^{15}N values for biofilm after application of the buffer treatments would indicate an increase in the proportion of algal material in the biofilm matrix. However, we found no evidence of an increased algal abundance in our results for ash free dry mass, chlorophyll *a*, or as reflected by stable isotope ratios for biofilm. The fact that substantial post-harvest increases in light reaching the streams in the FP and 0% treatment sites did not appear to have generated a change in C isotope ratios suggests that autotrophic energy sources may played a limited role in the support of food webs in these small streams, regardless of light level. Further, a lack of difference in δ^{13}C values between biofilm and leaf litter suggests that the biofilm encrusting the streambed is likely derived largely from terrestrial organic matter rather than organic matter fixed within the stream. Heterotrophic sources of energy appeared to dominate these small headwater streams both before and after harvest, regardless of treatment.

Other studies have reported results similar to our own, with a lack of differences in biofilm or periphyton growth between control and clearcut sites (Culp and Davies 1983; Shortreed and Stockner 1983; Göthe *et al.* 2009). In their meta-analysis of replicated studies, Richardson and Béraud (2014) concluded that the magnitude of periphyton response to logging was reduced in steeper, narrower streams (similar to our study sites) relative to lower-gradient, wider streams. Cover provided by wood, in combination with the steep slopes characteristic of incised headwater stream channels, may have limited the amount of radiation reaching streams (Kobayashi *et al.* 2010). In addition, the reduced velocity and depositional environment created by instream wood can increase fine sediment storage (Jackson *et al.* 2001), which can inhibit periphyton growth through accumulation in the periphyton matrix (Kiffney and Bull 2000; Kiffney *et al.* 2003). Finally, stream flows influence algal production (Hansmann and Phinney 1973; Shortreed and Stockner 1983; Peterson and Stevenson 1992), and the increased flows we observed in both the FP and 0% treatments may have had a negative impact on overall biofilm standing stock.

17-3.6. MACROINVERTEBRATE EXPORT

Macroinvertebrates respond to a variety of instream dynamics including biofilm and periphyton standing stock, in-channel wood load and channel characteristics, nutrients and detritus, litterfall, stream temperature, and competition and predation (i.e., amphibians and downstream fish; e.g., Wiley and Kohler 1984; Lancaster 1990; Richardson 1991; Siler *et al.* 2001). Since shading from riparian vegetation generally limits autotrophic production in forested headwater streams (Richardson and Danehy 2007), litterfall inputs are the main source of energy (Cummins *et al.* 1983; Gregory *et al.* 1991; Bilby and Bisson 1992). As a result, macroinvertebrate taxa of headwater systems typically consist of those specialized in shredding litterfall and collecting the resulting particulate organic matter. Tree removal has the potential to shift the macroinvertebrate community away from shredder and collector taxa and toward macroinvertebrate scrapers (Newbold *et al.* 1980; Hawkins *et al.* 1982; Hawkins 1988) as a result of decreased litterfall (Bilby and Bisson 1992) and increased periphyton biomass (Kiffney and Bull 2000; Kiffney *et al.* 2003; Danehy *et al.* 2007). We did not, however, observe this shift.

Total macroinvertebrate export in numbers and biomass per day did not change in response to buffer treatments in this study, but the response of functional feeding groups varied. Export of collector-gatherer biomass increased in the FP treatment relative to both the reference and 100% treatment. Previous studies of benthic macroinvertebrates have found an increase in numbers and/or biomass of collector-gatherers in unbuffered streams following harvest (Hawkins *et al.* 1982; Haggerty *et al.* 2004; Hernandez *et al.* 2005). While drift is not always comparable to standing stocks of benthic invertebrates, an increase in collector-gatherer export may indicate that benthic collector-gatherers were abundant, possibly in response to food resources, and reached their carrying capacity, which then initiated a drift response. Other mechanisms that initiate drift include responses to temperature and stream flows (Wiley and Kohler 1984), sedimentation (Culp *et al.* 1986; Suren and Jowett 2001), food availability (Richardson 1991;

Hinterleitner-Anderson *et al.* 1992; Siler *et al.* 2001), competition (Hildebrand 1974) and presence of predators (Wiley and Kohler 1984; Lancaster 1990). We also observed a decrease in $\delta^{15}N$ for macroinvertebrate collector-gatherers, which is indicative of a decrease in the proportion of biofilm in their diets.

Export of Chironomidae and *Baetis*, a collector-gatherer, did not change in response to harvest, even though both taxa comprised much of the proportion of individuals exported pre- and post-harvest, and *Baetis* comprised most of the biomass exported. An abundance of these taxa likely resulted from their multivoltinism, or short life cycles, which allow them to cycle through multiple generations in one year, and their ability to quickly adapt to disturbances and use available food resources. Both Chironomidae and *Baetis* are considered generalist feeders that are able to exploit the earliest food materials on disturbed substrates (Mackay 1992) or short term increases in primary productivity following timber harvest (Wallace and Gurtz 1986).

We observed a decline in scraper export in numbers per day in the 100% and 0% treatments, and in parasite export in numbers and biomass per day in the 100%, FP, and 0% treatments relative to the reference. In addition, we measured a decrease in Dixidae (Diptera) export in numbers per day in the 100% and 0% treatments, and in biomass per day in the 100% treatment relative to the reference. While export of these feeding groups and taxon changed in some of the treatments relative to the reference, export from the reference sites increased significantly during the post-harvest period, which may suggest environmental factors or natural variability rather than a treatment effect.

17-3.7. STREAM-ASSOCIATED AMPHIBIANS

Stream-associated amphibian (i.e., Coastal Tailed Frog [*Ascaphus truei*], and torrent [*Rhyacotriton*] and giant [*Dicamptodon*] salamanders) presence and abundance are associated with a suite of factors. Factors known to influence stream-associated amphibians include stream temperature (Bury and Corn 1988; Bury 2008; Pollett *et al.* 2010), nutrients (Kiffney and Richardson 2001), sediment (Dupuis and Steventon 1999; Wilkins and Peterson 2000; Stoddard and Hayes 2005), food availability (e.g., periphyton standing stock and macroinvertebrates; Kiffney and Richardson 2001), competitors and predators (e.g., macroinvertebrates and downstream fish; Crawford and Semlitsch 2007; Richardson and Danehy 2007; Kroll *et al.* 2008).

Larval Coastal Tailed Frog density increased in intermediate treatments (i.e., 100% and FP) relative to both the reference and 0% treatment. While increased stream temperature can result in increased instream periphyton production (Kiffney *et al.* 2003), we observed no difference in instream periphyton biomass between treatments; however, increased diatom production that went undetected in our analysis of periphyton growth could have been present in discrete patches. These patches have been associated with accumulations of organic substrates such as wood, which we found in increased densities across all buffer treatment sites, and can support higher levels of periphyton biomass than inorganic substrates (Coe *et al.* 2009). Larval tailed frogs are grazers, so an increase in periphyton production, even in discrete patches, could have positively influenced larval survival. Further, periphyton production has been shown to increase with an increase in nutrient availability (Kiffney and Richardson 2001), so the total-N and nitrate-N increase we observed across all buffer treatments had the capacity to positively influence tailed frog larvae indirectly.

Considering the potential mechanisms for the increased densities observed in the 100% and FP treatments, the lack of change of tailed frog larval density in the 0% treatment may seem difficult to explain. However, light saturation for algal production occurs at less than full sunlight (Murphy 1998). Further, timber harvest and subsequent increases in sunlight reaching the stream and stream temperature may have altered the composition of periphyton (Hawkins *et al.* 1983; Beschta *et al.* 1987), resulting in a shift in periphyton species composition away from diatoms, which are the preferred food source for larval tailed frogs (Altig and Brodie 1972; Nussbaum *et al.* 1983). Different diatoms possess varying levels of protein, and vary in quality to the grazers that consume them. For example, members of the freshwater diatom *Rhopalodiaceae* contain N-fixing endosymbiotic cyanobacteria, making them higher in protein than other diatoms (Furey *et al.* 2014) and the relative abundance of N-fixing algae has been shown to be negatively correlated with nitrogen concentrations (Porter *et al.* 2008). Given that we observed a post-harvest increase in total-N and nitrate-N across all buffer treatments, is it likely that shifts in food quality also existed, even though we did not detect a difference in biofilm or periphyton standing crop between treatments. Further, the increased stream temperature in the 0% treatment may have had a negative impact on tailed frog larvae, which exhibit thermo-regulatory behavior (de Vlaming and Bury 1970). Tailed frog larvae have been found to use cool waters (Karraker *et al.* 2006; Bury 2008; Pollett *et al.* 2010), and the increased stream temperature we observed in the 0% treatment in particular may have induced downstream migration into the fish-bearing reaches.

Based on more limited data than what was available for larvae (i.e., fewer individuals detected), post-metamorphic Coastal Tailed Frog showed an increased post-harvest density in the 0% treatment, which may seem contradictory to our findings for larval tailed frogs. However, terrestrial movement of amphibians may be limited if upland habitats do not provide adequate environmental conditions. For example, changes in temperature and moisture (Grant *et al.* 2010), and canopy gaps related to timber harvest activity, can limit terrestrial movements of post-metamorphic individuals (Popescu and Hunter 2011). Thus, one possible explanation for our observation is that overland movement of post-metamorphic frogs was restricted in the 0% treatment due to unfavorable conditions, which in turn resulted in a concentration of individuals along the stream channel. Another possibility is that a greater abundance of resources led post-metamorphic individuals to remain in place rather than emigrating away from the stream.

We saw no difference in torrent salamander density between treatments, a finding that is consistent with those of some other studies of the impacts of timber harvest on torrent salamanders in headwater streams (Jackson *et al.* 2007; Olson *et al.* 2014). While we found no difference in density between our three riparian buffer treatments, some researchers have concluded that the impacts of forest harvest on torrent salamanders are ameliorated by the presence of riparian buffers (Vesely and McComb 2002; Stoddard and Hayes 2005). However, these studies were conducted in drier locales (e.g., in Oregon) and differences between those and our own results may be related to latitudinal and coastal-interior gradients. Finally, previous research on the impacts of forestry management on torrent salamanders has implicated a negative response to increased fine sediments, i.e., mud, silt and sand (Diller and Wallace 1996; Welsh and Lind 1996; Welsh and Ollivier 1998). We had no evidence of a significant increase in sediment inputs or exports between treatments, which may explain the lack of treatment effect on torrent salamanders, at least in part.

We detected a post-harvest decline in giant salamander density in the FP treatment, a change that we did not observe in the other buffer treatments. We find this relationship difficult to explain, especially considering the lack of response in the 0% treatment. Based on a preponderance of literature, we expected giant salamanders to be the species least likely to demonstrate a negative response to harvest. Giant salamanders have been found to tolerate a relatively wide variety of habitats (Nussbaum *et al.* 1983; Bury and Corn 1988; Leuthold *et al.* 2012) and stream temperatures (Adams and Bury 2002). Some researchers have concluded a lack of correlation between timber harvest and giant salamander density (Leuthold *et al.* 2012) or even a positive relationship between giant salamander biomass and clearcut harvest, at least for steeper streams greater than 9% (Murphy and Hall 1981). Jackson and colleagues (2007) concluded that giant salamanders were very sensitive to the immediate impacts of clearcut harvest. However, in their study, the vast majority (98%) of the stream length in the harvest treatment equivalent to our 0% treatment was buried with dense matrices of post-harvest logging debris, which impacted their ability to thoroughly sample and may have had implications for their findings. Interestingly, our giant salamander results are supported somewhat by the findings from our analysis of stable isotopes, insofar as results for the 100% and 0% treatments differed from those for the FP treatment. Specifically, post-harvest C isotope ratios for giant salamanders were higher in both the 100% and 0% treatment than in the FP treatment, indicating that food resource availability or utilization may have differed between treatments. Further, $\delta^{15}N$ for gathering macroinvertebrates in the FP treatment appeared to be lower than in the reference and 100% treatment.

The decreased giant salamander density in the FP treatment could have contributed to the positive increase observed for tailed frog larvae in that treatment, since giant salamanders are known to prey on larval tailed frogs, among other things (Nussbaum *et al.* 1983). However, vertebrate prey items are consumed by giant salamanders opportunistically, and only larger individuals are capable of consuming rare, large prey (Parker 1994). Alternatively, Feminella and Hawkins (1994) found that tailed frog larvae spent a greater amount of time hidden in crevices when predatory giant salamanders were present, so a decrease in giant salamanders in FP treatment sites could have resulted in increased tailed frog larval activity and probability of detection. Importantly, if the two species of giant salamanders observed in this study, Cope’s Giant Salamander (*D. copei*) and Coastal Giant Salamanders (*D. tenebrosus*), responded differently to timber harvest, we might not have been able to detect a treatment effect.

17-3.8. DOWNSTREAM FISH

Small headwater streams provide critical habitat for fish such as Coastal Cutthroat Trout (*Oncorhynchus clarkii clarkii*), the species most often found at the upstream extent of fish distribution in western Washington (Connolly 1997; Trotter 2000; Fransen *et al.* 2006). Few published studies characterize the fish populations in these distinctive habitats, or their sensitivity to modern timber harvest practices. Studies that have investigated the effects of timber harvest on trout in headwater streams have typically focused on habitats well downstream from the upper extent of fish use or in headwater catchments in their entirety, assessed the fish population long after timber harvest had occurred, and/or reported a variety of conflicting responses (Hall and Lantz 1969; Brown 1972; Aho and Hall 1976; Murphy *et al.* 1981; Bisson and Sedell 1984; Connolly and Hall 1999; Young *et al*. 1999; De Groot *et al*. 2007; Bateman *et al*. 2015), The original intent of the Type N Study was to include an evaluation of fish response to upstream timber harvest in the stream segments immediately downstream from the timber harvested Type N study sites. However, due to a variety of logistic and biological constraints (see Chapter 16 – *Downstream Fish*) only six of the 17 study sites were suitable for inclusion in the fish component of the study. This limitation, and a resulting lack of replication between treatments, prevented us from directly evaluating fish response to upstream timber harvest and alternative riparian buffer treatments. Instead, we focused on assessing the fish populations, specifically Coastal Cutthroat Trout, immediately downstream from the extreme upstream limit of fish distribution.

Previous work in headwater basins suggests that cutthroat trout occur at their highest densities in small streams (Murphy *et al.* 1986; Rosenfeld *et al.* 2000; Rosenfeld *et al.* 2002; Walter *et al.* 2014). For instance, Rosenfeld and colleagues (2002) estimate cutthroat trout abundance of 0.9–1.0 fish/m in small headwater streams with a channel width between one and three meters. These studies, however, focused primarily on headwater basins in their entirety, and not specifically on the segments of stream habitat at the upstream extent of fish distribution. We consistently found cutthroat trout density to be relatively low in comparison with other studies. We also found no relationships between total linear fish density and physical stream habitat characteristics, such as stream gradient and percent pool area, which have been associated with increased fish abundance. Despite a lack of statistical significance, there were patterns expressed in the data suggesting a positive relationship between age-0 fish density and both increased percent pool area and lower stream gradient. We hypothesize that the combination of lower stream gradient and increased pool area may result in lower overall stream power within some stream reaches, potentially resulting in an increased ability for smaller (age-0) fish to persist in these habitats. Despite other studies that have documented a strong association between adult cutthroat trout abundance and these habitat variables (Hartman and Gill 1968; Hartman *et al.* 1996; Rosenfeld *et al.* 2000), we saw no evidence of a density/habitat relationship for age-1+ fish that was similar to that seen for age-0 fish. All of our stream reaches were similarly small, and we hypothesize that the low productivity of these stream systems may be the single most important limiting factor for fish, particularly larger individuals with greater caloric needs. Trotter (2000) proposed a similar hypothesis, suggesting that fish density steadily declines to zero as one moves upstream toward the upper extent of fish distribution, and that low fish density in the smallest of fish-bearing streams, like those included in this study, may be associated, at least in part, with low food availability.

We did not directly investigate fish diet or food availability; however, we did assess both fish-specific growth rate and fish condition. Specific growth rate and condition of fish in our study reaches was lower than that observed in other studies that focused on fish in downstream habitats or throughout headwater basins in their entirety (Harvey 1998; Berger and Gresswell 2009). These results may also support Trotter’s (2000) hypothesis regarding low food availability in headwater streams and its potential effect on both fish growth and fish density in these habitats. We hypothesize that larger fish may simply not be able to find sufficient food resources in these small habitats to sustain continued growth. We did not find any evidence that fish density was significantly correlated with either fish condition or growth.

This work, while limited, documents the relatively low abundance and growth rates of cutthroat trout populations at the upstream extent of their distribution in the headwater stream reaches evaluated in this study. We found that these unique habitats tended to support lower densities of fish than typically reported in the reviewed literature for headwater basins as a whole, and that the cutthroat trout in these habitats grew more slowly and were smaller on average, with a lower condition factor than those reported in other studies. Our findings support the hypotheses of researchers such as Trotter (2000) who contend that fish in these habitats are living "at or near the limit of available stream resources", be those resources food, space (habitat), or both. This work was a case study, so its scope of inference is limited. To understand better the characteristics and dynamics of these unique habitats at the upstream extent of fish distribution, additional work directly comparing these stream reaches to the broader population of headwater streams is needed.

17-4. RIPARIAN PRESCRIPTION EFFECTIVENESS

We conducted an experimental study comparing different timber harvest treatments that varied in the length of the riparian buffer retained within RMZs of Type N Waters relative to untreated reference conditions. Our objective was to evaluate the effectiveness of Forest Practices rules for westside Type N Waters in maintaining riparian structures, functions and processes important to the riparian forest and associated aquatic system (e.g., water quality and aquatic populations). We compared the magnitude, direction (positive or negative), and duration of change in riparian forest stand conditions related to: riparian inputs (i.e., light [measured as shade], litterfall, sediment, and wood) instream responses (amphibians, water temperature, and channel characteristics), and downstream components (exports of nutrients, detritus, macroinvertebrates, and sediment; temperature; discharge; and fish in the downstream fish-bearing [Type F] reach).

The effects of timber harvest varied in relation to the proportion of stream length with riparian buffers. We found more statistically significant changes, with greater magnitudes of change, in the 0% treatment than in the other buffer treatments (**Table 17-2** and **Table 17-3**), especially for physical characteristics that included wood loading, riparian cover, discharge, water temperature, litterfall, and exports of detritus and nutrients. Overall, the 100% treatment was the most effective in maintaining pre-harvest conditions, followed by the FP and then the 0% treatment. While the 100% and FP treatments had similar responses for several metrics (i.e., large wood recruitment, wood cover/loading, water temperature, discharge, channel unit metrics, and Coastal Tailed Frog density), we also found responses that differed between the two treatments, including tree mortality, stand structure, riparian cover, detritus export, macroinvertebrate export, and giant salamander density. Differences in the response of treatments did not appear to have a dramatic negative impact on the biota found at higher trophic levels (i.e., macroinvertebrates and amphibians). Reductions in shade and increasing stream temperatures in treated streams did not result in post-harvest changes in biofilm or periphyton, and total macroinvertebrate export did not change measurably in response to treatment. Only giant salamanders responded negatively in the FP treatment. Importantly, since the proportion of the stream length buffered in all FP treatment sites was more than the minimum required under Forest Practice's rules, some consistent results between the 100% and FP treatments may be related to the similarity in buffer lengths between these treatments.

We evaluated the effectiveness of the westside Type Np riparian rules in meeting the Forests and Fish Report (USFWS 1999) functional Resource Objectives for heat/water temperature, large wood/organic inputs, sediment, hydrology, and stream-associated amphibians. Resource Objectives and Critical Questions are outlined in the Cooperative Monitoring, Evaluation and Research Committee (CMER) Work Plan. Where we had the opportunity, we also evaluated some additional responses. Consistent with our study design (see Chapter 2 – *Study Design*), inference from this study is best limited to similar western Washington Type N basins located on competent lithologies and in second-growth managed forestlands. Approximately 29% of western Washington Forests and Fish-regulated lands are composed of competent lithologies (P. Pringle, Washington State Department of Natural Resources, personal communication, September 2005).

17-4.1. STAND STRUCTURE AND TREE MORTALITY RATES IN RIPARIAN BUFFERS

We found differences in mortality rates and magnitude of change in riparian stand structure between the treatments. The greatest change in stand structure occurred in treatments where riparian trees were harvested (i.e., the 0% treatment and the unbuffered portions of the FP treatment sites). Removal of streamside trees returned these areas to the stand-initiation stage of development. In the treatments with buffers (100% treatment and buffered portions of FP treatment sites), the greatest change in stand structure occurred in FP and 100% treatment PIP buffers, where we observed significant decreases in density (FP treatment) and basal area (FP and 100% treatments) relative to reference PIPs due to mortality from windthrow. Windthrow-induced mortality in PIP buffers has been observed in another CMER study (Schuett-Hames *et al.* 2012), and is likely due to the small size of the buffers, topographic exposure to wind due to their location on the upper slopes at the tops of drainage systems, and the removal of surrounding trees by clearcut harvest (Mitchell *et al.* 2001; Ruel *et al.* 2001; Rollerson *et al.* 2009). Tree mortality in the buffered portions of the FP treatment RMZs was significantly higher than in either the 100% treatment or unharvested reference RMZs. However, mortality rates in RMZs with riparian buffers were highly variable, with little mortality in most buffer stands and higher mortality in a smaller subset. Unless the rates of mortality change significantly over time, FP and 100% RMZ buffers, which experienced low mortality, will continue developing as single cohort, conifer dominated stands. The future trajectory for the sub-set of buffer stands with higher mortality is uncertain. Success of natural conifer regeneration will likely determine if these stands develop as multi-cohort conifer stands, or become dominated by broadleaf trees or shrubs. Windthrow was the primary cause of mortality and tree fall in the FP and 100% RMZ and PIP buffers. We observed higher rates of windthrow in the coastal blocks (Willapa 1 and Willapa 2) than in sites located further inland.

17-4.2. WOOD RECRUITMENT AND LOADING

All treatments experienced inputs in woody material in the two-year post-harvest period, but the quantity and characteristics of wood inputs varied between and within treatments. We found a marked difference between the buffered and clearcut RMZs in the timing and characteristics of wood recruitment. Streams adjacent to clearcut areas (0% treatment and unbuffered portions of the FP treatment sites) received substantial inputs of logging slash (mostly branches, tops and non-merchantable stems) during harvest, even with an ELZ and the additional rules intended to minimize wood input during harvest. Conversely, RMZ and PIP buffers (100% treatment and buffered portions of the FP treatment sites) prevented input of logging slash into the stream from upland clearcut harvest. In the two years following harvest, little additional wood recruitment to the stream occurred in the clearcut RMZs because few if any trees remained. In the FP and 100% treatment buffers, wood input was mostly due to windthrow and consisted primarily of stems of uprooted and broken trees. Recruitment rates were highly variable due to differences in wind disturbance among study sites.

We attributed differences in wood loading to both treatment and the variability in disturbance frequency and severity (e.g., windthrow events) between sites. However, regardless of disturbance variability, we detected an overall post-harvest increase in wood loading in all buffer treatments that, for the most part, did not differ statistically. The only exception was for the number of small wood pieces (≤10 cm diameter), which had a post-harvest increase that was greater in the 0% than in the 100% and FP treatments.

The stand structure two years post-harvest has implications for future wood input and wood loading. In the areas with clearcut harvest to the edge of the stream, recruitment of large wood from riparian areas will require the establishment and development of a new forest stand. Wood loading is likely to decrease as logging slash decays over time; however, these channels are likely to receive another pulse of logging slash during the next harvest. Future wood recruitment from riparian buffers in the 100% and FP treatments will depend on several factors, including existing recruitment potential (the density and size of the current stand), ingrowth of new trees, and the magnitude and frequency of disturbances, such as wind, that cause tree mortality and wood input.

17-4.3. STREAM TEMPERATURE AND COVER

None of the three buffer treatments was effective at preventing a significant loss in shade or a significant increase in stream temperature. Shade retention was greatest in the 100% treatment, moderate in the FP treatment, and least in the 0% treatment. July–August seven-day average maximum daily temperature increased in all buffer treatments. At the harvest unit boundary the 100% and FP treatments were not significantly different from each other (approximately 1.2℃ increase) but each was significantly more effective at maintaining temperatures than the 0% treatment (approximately 3.2℃ increase). The results were similar when measured at the F/N break; although the magnitude of the temperature increase in this location was slightly lower (**Table 17-4**).

Downstream of the harvest unit the seven -day average July–August water temperature was lower than at the harvest unit boundary, but still elevated compared to pre-harvest temperatures at five of the six sites where it could be assessed. This suggests that recovery is occurring but not complete within approximately 100 m below the harvest unit.

Higher temperatures were not isolated to the summer months. We also noted significant increases in maximum daily temperature at nearly all monitoring locations and in nearly all buffer treatment sites from the spring through the fall months in both post-harvest years. We noted smaller, but significant, increases in minimum daily temperature, most frequently for the July-August period; however, these did not differ among buffer treatments.

Table 17-4. Post-harvest increase in the seven -day average temperature response (°C) for each treatment and post-harvest year (Post 1, Post 2) at the buffer treatment (harvest unit boundary) and the F/N break (location nearest the F/N junction). Numbers in bold indicate a significant (P <0.05) increase from pre-harvest conditions.

Treatment	Post 1	Post 2
Buffer Treatment		
100%	1.2	1.2
FP	1.4	1.0
0%	3.4	3.0
F/N Break		
100%	0.9	0.6
FP	1.4	1.0
0%	3.1	2.7

17-4.4. DISCHARGE

Discharge changes associated with harvest are likely to be complex and extend over much longer timescales than those analyzed as part of this study (Stednick 1996; Moore and Wondzell 2005; Perry and Jones 2016). As a result, it is not possible to provide a comprehensive assessment of treatment effectiveness. We found evidence that hydrologic regimes were altered in all six buffer treatment sites evaluated, including the two FP treatment sites. The magnitude of baseflows (RI <2) decreased in both of the 100% treatment sites while storm flow (e.g., RI ~2–30) increased in five of the six buffer treatment sites, including both the FP and 0% treatment sites.

The resource objectives for discharge specifically mention the prevention of increased peak flows (CMER Work Plan). We found no evidence of increased annual peak flows in the Willapa 1 block sites or the OLYM-100%, but did observe peak flow increases in the OLYM-FP and OLYM-0%. The difference in peak flow response between the two blocks may be due to differences in runoff generation processes, especially rain-on-snow. The findings and literature suggest that, given the importance of rain-on-snow in peak flow generation, prescription effectiveness related to flow is unlikely to be uniform across the landscape of interest.

17-4.5. SEDIMENT PROCESSES

Study sites appeared to be supply limited with respect to suspended sediment, both before and after harvest, a finding consistent with the fact that the study was restricted to sites underlain by relatively competent lithologies. Eight storm events dominated suspended sediment budgets, which generally resulted in SSE spikes in about half of the study sites included in this evaluation. While the magnitude of the spikes varied by storm, it did not significantly vary by treatment. Further, we did not find strong relationships between changes in discharge and sediment export. We did not identify windthrow as a major driver in SSE, probably because most rootpits resulting from windthrow did not appear to deliver fine sediments to the channel. Stream-delivering surface erosion was largely restricted to small areas of the streambank and was unlikely to measureably contribute to SSE.

Although SSE increased during or post-harvest at several sites, when evaluated relative to the unharvested reference sites, we could identify no significant change in sediment supply or export in any of the treatments, including the 0% treatment. Our ability to detect a buffer treatment effect may have been hampered by the low replication (n = 2) of each treatment, the limited number of sediment generating events, and inconsitency in sediment export across sites for a given storm event owing to the limited sediment supply and stochastic nature of sediment inputs. Strict observence of ELZs, which restricted equipment disturbances within 30 ft (9.1 m) of the stream and preserved riparian vegetation (see resource objectives for sediment in the CMER Work Plan) may have played a role, though this was not explicitily examined. Any harvest-related changes in SSE appears to have been less than the natural variability. Given the acknowledged limitations of the sediment portion of the study, we cannot draw major conclusions about the effectiveness of forestry activities on erosion processes at this time.

17-4.6. STREAM CHANNEL CHARACTERISTICS

We measured channel characteristics to interpret harvest effects on macroinvertebrate and amphibian habitat. With the single exception of pool length, which increased in all three buffer treatments, channel characteristics in the 100% and FP treatments did not differ from the reference. The increase in pool length was likely the result of increased wood inputs to the treatment stream relative to the reference (Lisle 1986). Stream wetted width, bankfull width, and the proportion of the channel rise attributed to steps increased in the post-harvest period for the reference, 100% treatment and FP treatment, a consistent change resulting from year effects or annual patterns; however, these changes were not observed in the 0% treatment. Large wood, which increased in all treated sites, can be concentrated towards the channel margins (Ralph *et al.* 1994), restricting stream wetted width during the summer (Carlson *et al.* 1990) and affecting bankfull width. Fluvial power in our small debris-driven study streams was likely insufficient to sort wood and form distinctive steps during the two years immediately post-harvest. Rather, wood frequently accumulated in dense matrices of branches, organic debris and fine sediment, covering or filling the stream channel in some reaches. As a result, organic and inorganic materials likely covered some steps in the 0% treatment, affecting the percent channel rise attributed to steps.

17-4.7. LITTERFALL INPUT AND DETRITUS EXPORT

Riparian tree removal in the RMZs of the 0% treatment and unbuffered portions of the FP treatment reduced litterfall inputs in both treatments post-harvest. However, we noted significant differences in litterfall inputs only in the 0% treatment, and annual litterfall inputs continued to remain low in the 0% treatment at the end of the second post-harvest year. The decrease in litterfall inputs also resulted in a significant decrease in instream detritus exports from the 0% treatment. Riparian buffers in the 100% and FP treatments maintained litterfall inputs and detritus exports that did not differ statistically from reference conditions.

17-4.8. NUTRIENT EXPORT

Though resource objectives and critical questions that address nutrients were lacking, we measured this response to evaluate the effect of treatments on the quantity of instream nitrogen and phosphorus exported from headwater streams. Post-harvest N export increased at all buffer treatment sites. 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 and the relative magnitude of the changes was consistent with our expectations of increased N export with an increase in the proportion of the watershed harvested. However, the difference between the 100% and FP treatments and between the FP and 0% treatments were not significant. Nitrogen export was also correlated with an increase in annual runoff, which was correlated with the proportion of the watershed harvested. In contrast to N, total P concentration did not change post-harvest. That we did not observe differences in phosphorus between treatments was likely the result of the riparian buffer and ELZ requirements, which minimized ground disturbance near the stream and largely prevented sediment delivery.

17-4.9. BIOFILM AND PERIPHYTON

Though resource objectives and critical questions that address biofilm and periphyton were lacking, we measured these responses to evaluate the effects of treatments on food resources for instream biota, including macroinvertebrates and stream-associated amphibians. We observed post-harvest reductions in canopy, but no concomitant increases in biofilm and chlorophyll *a*, across all riparian buffer treatments. The lack of a treatment effect may have resulted from a variety of factors, including the fact that approximately half of the biofilm and chlorophyll *a* collection locations in unbuffered stream reaches were located in areas shaded by something other than overstory canopy (e.g., wood). Alternatively, photosynthetic efficiency may be greater at moderate light levels (Hill *et al.* 1995), providing for the possibility that light saturation limited periphyton growth in unbuffered reaches. Other possible factors influencing biofilm accumulation and periphyton production in our study streams include nutrient limitation (Kiffney and Richardson 2001), stream flow and scour (Allan 1995), and sediment accumulations in the periphyton matrix (Kiffney *et al.* 2003), among others. Ultimately, while we do not know exactly what physical and/or biological processes are controlling periphyton growth in our study streams, we have no evidence of a difference in response between buffer treatments.

17-4.10. MACROINVERTEBRATE EXPORT

No resource objectives or critical questions address macroinvertebrates specifically. However, we assessed the response of macroinvertebrate export from Type N basins to evaluate the effect of treatments on food resources for stream-associated amphibians and downstream fish. We did not observe a change in total macroinvertebrate export in numbers and biomass per day in response to harvest, nor did we see a change in most functional feeding groups. Export of collector-gatherer biomass increased in the FP treatment relative to both the reference and 100% treatment. While we observed changes in export for the scraper and parasite functional feeding groups and in Dixidae (Diptera) from some of the treatments, a change also occurred in the reference sites. Wood inputs to our study sites may have maintained or enhanced food resources available to collector-gatherers. Other studies have attributed an increase in benthic collector-gatherers to an influx of wood and detrital accretion (Cole *et al.* 2003; Haggerty *et al.* 2004). The presence of wood in our study sites may have also prevented a shift from a primarily heterotrophic-based system to an autotrophic-based system, and may explain the lack of a response in other functional feeding groups.

17-4.11. STREAM-ASSOCIATED AMPHIBIANS

We used stream-associated amphibian occupancy, density and genetics as measures of population viability for evaluating impacts of alternative riparian buffers on Coastal Tailed Frog, and torrent and giant salamanders. We did not observe changes in stream-associated amphibian occupancy that were associated with treatment in the two years post-harvest. This finding is consistent with our own predictions and based on current literature examining the effect of timber management on stream-associated amphibians (e.g., Jackson *et al.* 2007; Pollett *et al.* 2010; Olson and Burton 2014). However, we did find support for a decrease in density for giant salamanders in the FP treatment, an unexpected response that would require study over a longer period to understand fully. In contrast, Coastal Tailed Frog larval density increased in the 100% and FP treatments relative to the reference and 0% treatment. Post-metamorphic Coastal Tailed Frog density increased in the 0% treatment relative to other treatments. We did not detect clear evidence of a response for torrent salamanders. Evaluation of a genetic response requires generational turnover of amphibian populations that will require a minimum of seven to eight years after treatment implementation. Consequently, we report on the results from this component of the study separately.

The density of stream-associated amphibians covered under Forest Practices Rules (which does not include giant salamanders) did not decline in any buffer treatment, including in the 0% treatment where the July–August seven-day average maximum daily temperature increased by 3.2℃ in the two years post-harvest. We found all taxa (including giant salamanders) in stream reaches of clearcut RMZs that were covered by dense matrices of wood, organic debris (e.g., leaves and needles), and fine sediment (i.e., obstructed reaches). However, we note that short-term changes to amphibian movement, food resources, competition, predation, quantity and quality of habitat, altered oviposition sites, ova developmental conditions, or egg or larval success may only become apparent with longer-term study. Even so, the fact that stream-associated amphibians inhabited our study sites in second-growth forests, on forestlands managed for timber production, confirms that Forest Practices Rules have been sufficient for amphibian persistence to date.

17-4.12. TEMPORAL PERSPECTIVE

The scope of potential long-term response for all variables can only be understood with continued monitoring. For example, there will be opportunities for continued windthrow from riparian buffers and associated in-channel wood recruitment, sediment delivery, or delayed response to the reproductive success of stream-associated amphibians, among others. Conversely, for metrics for which we saw an immediate response, we could not adequately address the time needed for recovery to pre-harvest levels in our two years of post-harvest sampling. For example, summer stream temperatures were generally lower in the second post-harvest year relative to the first year, but were still not fully recovered to pre-harvest levels.

This study was originally proposed to be conducted throughout an entire harvest rotation (i.e., 30 to 40 years in western Washington). To address a recognized need, we have continued to monitor study sites continuously for some metrics (e.g., stream temperature) and periodically for others (e.g., stream-associated amphibians). These data have been collected through eight years post-harvest. The results from this extended study period will provide additional information for understanding the effectiveness of the current Forest Practices rules and buffer alternatives.

Pending the results from the evaluation of data collected through eight years post-harvest, future sampling may be warranted, at least for some variables. Even though one reference site was recently harvested in January 2016 and three more are anticipated to be harvested between prior to 2020, continued monitoring may prove valuable, especially considering the scarcity of long-term data of this nature. It may be necessary to substitute reference sites from a similar long-term study (Ehinger *et al.* 2011) in place of some of the reference sites that we anticipate will be harvested in the coming years.

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GLOSSARY

Alluvial fan (WAC 222-16-010): an erosional land form consisting of a cone-shaped deposit of water-borne, often coarse-sized sediments.

(a) The upstream end of the fan (cone apex) is typically characterized by a distinct increase in channel width where a stream emerges from a narrow valley;

(b) The downstream edge of the fan is defined as the sediment confluence with a higher order channel; and,

(c) The lateral margins of a fan are characterized by distinct local changes in sediment elevation and often show disturbed vegetation.

Alluvial fan does not include features formed under climatic or geologic conditions that are not currently present or that are no longer dynamic. Alluvial fan is one of five sensitive site categories.

Aquatic resources (WAC 222-16-010): water quality, fish, Columbia Torrent Salamander (*Rhyacotriton kezeri*), Cascade Torrent Salamander (*R. cascadae*), Olympic Torrent Salamander (*R. olympicus*), Dunn's Salamander (*Plethodon dunni*), Van Dyke's Salamander (*P. vandykei*), Coastal Tailed Frog (*Ascaphus truei*), Rocky Mountain Tailed Frog (*Ascaphus montanus*) and their respective habitats.

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.

Block: the arrangement of experimental units (in this, case study basins) into groups (blocks) that are similar. The result is a reduction in sources of variability, resulting in greater precision.

Breccia: a rock composed of broken fragments of minerals or rock cemented together by a fine-grained matrix that can be either similar to or different from the composition of the fragments.

Channel head: the most upslope part of a channel network, where overland and/or subsurface flow accumulates to form a defined channel (i.e., exposed mineral substrates) downslope.

Channel reach: stream channel reach that exhibits similar bedforms over stretches of stream that are many channel widths in length (e.g., colluvial, bedrock, cascade, step-pool, plane-bed, and pool-riffle reaches; Montgomery and Buffington 1998).

Channel unit: morphologically distinct area that extends up to several channel widths in length and is spatially embedded within a channel reach (e.g., pools, riffles, cascades, steps; Montgomery and Buffington 1998).

Clearcut (WAC 222-16-010): a harvest method in which the entire stand of trees is removed in one timber harvesting operation.

Diameter at breast height (dbh; WAC 222-16-010): the diameter of a tree at 4.5 feet above the ground measured from the uphill side.

Equipment limitation zone (WAC 222-16-010): a 30-foot (9.1-m) wide zone measured horizontally from the outer edge of the bankfull width of a Type Np or Ns Water. It applies to all perennial and seasonal non-fish-bearing streams.

Experimental treatment: Experimental treatments include three clearcut harvest treatments that vary in the length of perennial non-fish-bearing stream length buffered and reference sites:

(a) *0% Buffer:* Clearcut harvest with no riparian buffer;

(b) *Forest Practices Buffer:* 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 tributary junctions);

(c) *100% Buffer:* 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 (i.e., fish end point, point of last known fish use, fish-bearing water boundary): Point at which a Type F (fish-bearing) Water becomes a Type Np (perennial non-fish-bearing) Water.

Fish (WAC 222-16-010): for the purposes of the Forest Practice rules means species of the vertebrate taxonomic groups of *Cephalospidomorphi* and *Osteichthyes*.

Fish habitat (WAC 222-16-010): habitat that is used by any fish at any life stage at any time of the year, including potential habitat likely to be used by fish which could be recovered by restoration or management and includes off-channel habitat.

Green recruitment trees (WAC 222-16-010): trees left after harvest for the purpose of becoming future wildlife reserve trees under WAC 222-30-020(11).

Headwall seep (WAC 222-16-010): a seep located at the toe of a cliff or other steep topographical 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. Headwall seep is one of five 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 extent of Type Np Water (i.e. perennial initiation point). Headwater spring is one of five sensitive site categories.

Herbicide (WAC 222-16-010): any substance or mixture of substances intended to prevent, destroy, repel, or mitigate any tree, bush, weed, algae and aquatic weeds.

Mainstem channel: The dominant stream thread as defined by the greatest surface water volume.

Merchantable timber (WAC 222-16-010): a stand of trees that will yield logs and/or fiber suitable in size and quality for the production of lumber, plywood, pulp or other forest products, or of sufficient value at least to cover all costs of harvest and transportation to available markets.

Riparian buffer: see *Riparian management prescriptions*

Riparian function (WAC 222-16-010): includes bank stability, the recruitment of wood, leaf litter fall, nutrients, sediment filtering, shade, and other riparian features that are important to both riparian forest and aquatic system conditions.

Riparian management prescriptions: The prescription applied within the RMZ along Type Np Waters:

- *(a) Riparian buffer (BUF):* includes both stream and sensitive sites buffers:
    - *a. Stream buffer:* two-sided 50-ft(15.2-m) no-harvest buffer measured horizontally from the bankfull width of the Type Np Water;
    - *b. Sensitive site buffers:* no-harvest buffers measured horizontally from the specified sensitive site, as follows:
        1. *Headwall seep buffer:* no timber harvest is permitted in an area within 50 feet of the outer perimeter of a soil zone perennially saturated from a headwall seep;
        2. *Side-slope seep buffer:* no timber harvest is permitted in an area within 50 feet of the outer perimeter of a soil zone perennially saturated from a side-slope seep;
        3. *Type Np intersection buffer:* no timber harvest is permitted within a 56-ft (17.1-m) radius buffer patch centered on the point of intersection of two or more Type Np Waters;
        4. *Headwater spring buffer:* no timber harvest is permitted within a 56-ft (17.1-m) radius buffer patch centered on a headwater spring (i.e., uppermost point of perennial flow); and,
        5. *Alluvial fan buffer:* no timber harvest is permitted within an alluvial fan.
- *(b) Unbuffered (UNB): no riparian buffer adjacent to the stream edge.*

(c) *No adjacent harvest (NAH): no harvest adjacent to the stream. These are areas of the basin that could not be harvested due to regulatory or landowner constraints (e.g., unstable slopes, stand age).*

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 *Riparian Management Prescriptions*).

Road construction (WAC 222-16-010): the establishment of any new sub-grade including widening, realignment, or modification of an existing road prism, with the exception of replacing or installing drainage structures, for the purposes of managing forest land under Title 222 WAC.

Road maintenance (WAC 222-16-010): any road work specifically related to maintaining water control or road safety and visibility (such as: grading, spot rocking, resurfacing, roadside vegetation control, water barring, ditch clean out, replacing or installing relief culverts, cleaning culvert inlets and outlets) on existing forest roads.

Salvage (WAC 222-16-010): the removal of snags, down logs, windthrow, or dead and dying material.

Sensitive sites (WAC 222-16-010): areas near or adjacent to Type Np Waters that include one or more of the following: headwall seep, side-slope seep, Type Np intersection, headwater spring, alluvial fan.

Side-slope seep (WAC 222-16-010): a seep within 100 feet of a Type Np Water located on side-slopes greater than 20 percent, connected to the stream channel network via overland flow, and characterized by loose substrate 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. Side-slope seep is one of five sensitive site categories.

Slash (WAC 222-16-010): pieces of woody material more than 3 cubic feet in size resulting from forest practice activities.

Spatially intermittent streams: segments of streams that normally go dry.

Stream-associated amphibian: any species of amphibian (including frogs, toads, salamanders and newts) that depend on headwater streams and their riparian environments for breeding habitat, rearing habitat, cover, and/or food.

Stream order (e.g., first-order, second-order, and third-order)*:* definition of stream size based on a hierarchy of tributaries where perennial streams with no upstream tributaries are first-order streams, the confluence of two first-order streams are second-order streams, the confluence of two second-order streams are third-order streams, and so on. Streams of lower order joining a higher order stream do not change the order of the higher stream. Thus, if a first-order stream joins a second-order stream, it remains a second-order stream. It is not until a second-order stream combines with another second-order stream that it becomes a third-order stream. See Strahler (1952).

Temporary road (WAC 222-16-010): a forest road that is constructed and intended for use during the life of an approved forest practices application/notification. All temporary roads must be abandoned in accordance to WAC 222-24-052(3).

Tributary: a secondary stream thread 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, which are within the bankfull width of defined channels and contain fish habitat.

Type Np basin: the extent or area of land where surface water from rain and melting snow or ice converges to a single point, in this case where the Type Np Water joins the Type F Water at the N/F break. The basin includes both the streams that convey the water as well as the land surfaces from which water drains into those channels.

Type Np intersection (WAC 222-16-10): the intersection of two or more Type Np Waters. Type Np intersection is one of five 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.

Type Ns Water (i.e., non-perennial, seasonally intermittent stream; WAC 222-16-030): all segments of natural waters within the bankfull width of 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 by physically connected by an above-ground channel system to Type S, F, or Np Waters.

Type S Water (WAC 222-16-030): all waters, within their bankfull width, inventoried as “shorelines of the state” under chapter 90.58 RCW and the rules promulgated pursuant to chapter 90.58 RCW including periodically inundated areas of their associated wetlands.

Uppermost point of perennial flow (PIP; i.e., perennial initiation point, upper extent of a Type Np Water, initiation point of perennial flow): point which is the start of perennial flow in the stream channel network. PIPs include both headwater spring and headwall seep sensitive sites categories.

Wildlife reserve trees (WAC 222-16-010): defective, dead, damaged, or dying trees which provide or have the potential to provide habitat for wildlife species.

Windthrow (i.e., *blowdown*; WAC 222-16-010): a natural process by which trees are uprooted or sustain severe trunk damage by the wind.

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Strahler, A.N. 1952. Hypsometric (area-altitude) analysis of erosional topography. *Bulletin of the Geological Society of America* 63:1117–1142.

SUPPLEMENT 1 – RESPONSE VARIABLES INCLUDED IN THE TYPE N STUDY

Table S1-1. Complete list of all response variables included in the Type N Study, including the chapter in which they are discussed.

Chapter	Variable	Metric
5 - Stand Structure and Tree Mortality Rates in Riparian Buffers	Stand Condition	Post-harvest tree mortality
		Post-harvest change in stand structure
6 - Wood Recruitment and Loading	Wood Loading	Post-harvest tree fall rates
		Total large wood
		Functional large wood
		Total small wood
		Functional small wood
		Post-harvest new wood cover year 1
		Post-harvest new wood cover year 2
		Post-harvest large wood recruitment piece count
		Post-harvest large wood recruitment volume
7 - Stream Temperature and Cover	Riparian Cover	Canopy and topographic density post year 1
		Canopy and topographic density post year 2
		Effective shade post year 1
		Effective shade post year 2
		Canopy closure 1-m post year 1
		Canopy closure 1-m post year 2
		Canopy closure 0-m post year 1
		Canopy closure 0-m post year 2
	Water Temperature	Daily maximum post year 1
		Daily maximum post year 2
		Daily minimum post year 1
		Daily minimum post year 2
		Diel range post year 1
		Diel range post year 2
	Air Temperature	Air temperature
8 - Discharge	Discharge	Baseflow
		Total discharge
		Mean discharge
9 - Nutrient Export	Nutrients	Nitrate-nitrogen concentration
		Phosphorus concentration
		Total nitrogen load
		Nitrate-nitrogen load
		Phosphorus load

Table S1-1. (continued)

Chapter	Variable	Metric
10 - Sediment Processes	Sediment	Stream-delivering surface erosion
		Road surface erosion
		Uprooted trees
		Turbidity and suspended sediment export
11 - Stream Channel Characteristics	Stream Hydrology	Wetted stream width
		Stream depth
		Dry length
	Stream Channel	Bankfull width
		Substrate dominated by fines and sand
	Stream Channel Units	Pool length
		Pool maximum depth
		Pool density
		Riffle length
		Riffle density
		Step height
		Step density
		Proportion of steps keyed by wood
		Diameter of wood key piece in steps
		Channel rise by steps
12 - Litterfall Input and Detritus Export	Litterfall Input	Total
		Leaf (coniferous + deciduous)
		Coniferous
		Deciduous
		Wood
		Miscellaneous
	Detritus Export	Total
		Course particulate
		Leaf (coniferous + deciduous)
		Coniferous
		Deciduous
		Wood
		Miscellaneous
		Fine particulate
13 - Biofilm and Periphyton	Biofilm and Periphyton	Ash-free dry mass (AFDM)
		Chlorophyll *a*

Table S1-1. (continued)

Chapter	Variable	Metric
14 – Macroinvertebrate Export	Total Macroinvertebrates	Numbers Biomass
	Functional Feeding Group - Numbers	Chironomidae Collector-filterer Collector-gatherer Omnivore Parasite Predator Scraper Shredder Unknown
	Functional Feeding Group - Biomass	Chironomidae Collector-filterer Collector-gatherer Omnivore Parasite Predator Scraper Shredder Unknown
	Taxonomic Composition - Numbers	Collembola Coleoptera Diptera Diptera: Dixidae Diptera: Simuliidae Ephemeroptera Ephemeroptera: Baetidae Plecoptera Plecoptera: Nemouridae Trichoptera
	Taxonomic Composition - Biomass	Crustacea Coleoptera Diptera Diptera: Dixidae Ephemeroptera Ephemeroptera: Baetidae Ephemeroptera: Heptageniidae Plecoptera Plecoptera: Perlidae Plecoptera: Perlodidae

Table S1-1. (continued)

Chapter	Variable	Metric
14 – Macroinvertebrate Export (continued)	Taxonomic Composition - Biomass (continued)	Trichoptera Trichoptera: Hydropsychidae Trichoptera: Rhyacophilidae
15 - Stream-associated Amphibians	Amphibian Occupancy	Coastal Tailed Frog Giant salamander Torrent salamander
	Amphibian Density	Coastal Tailed Frog larvae Coastal Tailed Frog post-metamorph Giant salamander Torrent salamander
	Amphibian Body Condition	Coastal Tailed Frog larvae Giant salamander Torrent salamander
16 - Downstream Fish	Fish Density	Fish density
	Fish Quality	Fish quality
Supplement 2 – Stable Isotopes	Stable isotopes - $\delta^{13}C$	Giant salamanders >50 mm Giant salamanders ≤50 mm Torrent salamander post-metamorph Torrent salamander larvae Gatherers Shredders Detritus Litter Periphyton
	Stable isotopes - $\delta^{15}N$	Giant salamanders >50 mm Giant salamanders ≤50 mm Torrent salamander post-metamorph Torrent salamander larvae Gatherers Shredders Detritus Litter Periphyton

SUPPLEMENT 2 – STABLE ISOTOPES ANALYSIS

Robert Bilby, Stephanie Estrella, Aimee McIntyre, Graham Mackenzie, Jamie Thornton,
and Marc Hayes

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DISCLAIMER

This appendix was originally intended to be a chapter in the main body of the Type N report. The information that would have been included in the chapter has been included as an appendix because it was not approved through CMER's Independent Scientific Peer Review (ISPR) process as a stand-alone chapter. The purpose of the research described here was to help determine if the application of the riparian buffer treatments affected trophic linkages (measured with stable isotopes) in the study streams. The primary hypothesis was related to the idea that canopy removal altered the proportion of algal material in instream biofilm, and this alteration could lead to a change in the consumption of algae by primary consumers or of food items supported by algae for secondary consumers at higher trophic levels (e.g., invertebrates and amphibians). This type of shift in trophic system organization has been documented for larger, fish-bearing streams when riparian canopy cover is reduced. Unfortunately, consistently collecting all sample types proved to be logistically impossible. While we were able to collect biofilm samples across all study sites and in both sample periods (pre- and post-harvest), we failed to collect enough samples that represented all invertebrate functional feeding groups and amphibian genera, especially Coastal Tailed Frogs. As a result, the findings for this component of the study were not robust enough to receive ISPR approval. Nonetheless, we included study context, methods and results for the stable isotopes analysis as an appendix to preserve the work, describe the lessons we learned, and to provide supporting evidence for periphyton, macroinvertebrate, and stream-associated amphibian findings reported in the body of the Type N report.

We refer to some information generated by the stable isotopes effort in the body of the report, specifically as it relates to conclusions and limitations including:

1) **A lack of evidence for an increased production of the algal component of biofilm:** We were able to obtain sufficient biofilm samples from all study sites and from both sample periods. We found no isotopic evidence for an increase in algal content in the biofilm. This conclusion is consistent with the chlorophyll *a* and biofilm biomass values reported in Chapter 13 – *Periphyton and Biofilm*. ISPR reviewers questioned our ability to support linkages between our findings for the stable isotopes and periphyton components of the study based on limitations clearly documented by the authors. The final ISPR reviewer indicated that she was "not inclined to accept the authors' claim that those results [for periphyton] provide a strong line of evidence complementary to the results [of stable isotopes] they present in this manuscript." We acknowledge that limitations to the sampling methodologies for biofilm and periphyton could have precluded our ability to detect differences had they existed, nonetheless, the lack of evidence for increased production of the algal component of biofilm was consistent between our periphyton and stable isotopes analyses. We clearly discuss methodology limitations here and in the Periphyton and Biofilm Chapter. Importantly, ISPR and CMER approved the Biofilm and Periphyton Chapter. In contrast to reviewers for the stable isotopes component, reviewers for the Biofilm and Periphyton Chapter did not identify concerns with study design limitations.

2) **The role of autotrophic energy sources in headwater streams:** A corollary to our conclusion that there was no increase in algal content of the biofilm is the conclusion/hypothesis that autotrophic energy sources appear to play a limited role in the support of food webs in these small streams, regardless of light level. This point is supported by the facts that the biofilm and detritus isotope values are very similar to values for terrestrial litter, and that those values did not

change post-harvest. The very low chlorophyll *a* values for the biofilm also support the conclusion that there is very little in-channel production of organic matter. Again, though we acknowledge limitations to both the biofilm and stable isotopes studies, we note that each study supports the other and were unexpected.

3) **Insufficient sampling of food items for amphibian genera:** Although our samples for invertebrates and amphibians were relatively small, our preliminary results suggested that amphibians were eating items that we did not sample either adequately or at all. In other words, we found differences in N and C isotope values between the insects we sampled and the isotopic values of predatory amphibians.

   Limitations in the study design and sampling methodology for the stable isotope component of the study undoubtedly contributed to small samples for some invertebrate functional feeding groups. Specifically, collection of invertebrates by drift sampling (as a part of the sampling design for the macroinvertebrate component of the study, Chapter 14 – *Macroinvertebrate Export*) proved to be inadequate for obtaining individuals from all functional groups. We are addressing this issue by collecting a comprehensive set of isotope samples as a component of the Type N re-sample effort. The primary goal of the re-sample effort for isotopes will be to generate a better understanding of the dietary habits of Coastal Tailed Frogs, and torrent and giant salamanders in these small streams. To help ensure that we obtain samples consistently and capture the full range of possible diet items, the type of samples collected has been expanded to include terrestrial invertebrates, and the sampling methods for invertebrates and amphibians has been modified to improve our ability to collect adequate samples of all invertebrate functional groups and amphibian genera at most study sites.

   As noted above, the intent of including this information in the final report was always to supplement information in other chapters, regardless of whether it was included as a standalone chapter or an appendix. The results provided a clear indication that our understanding of the dietary habits of stream-associated amphibians in small headwater streams is still incomplete. This was the primary conclusion of the stable isotope analysis and a conclusion with which ISPR reviewers agreed.

S2-1. ABSTRACT

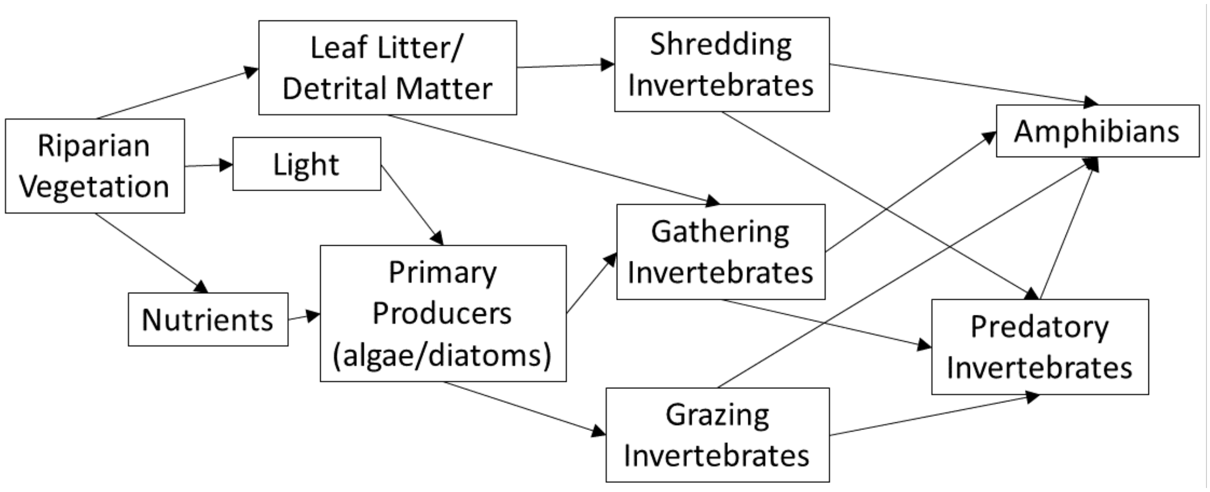
Canopy modification along forested streams has been associated with an increase in the contribution of algae to the trophic support of the system in numerous studies. However, the effect of canopy modification on food webs of very small, fishless streams has not been thoroughly evaluated. The Type N Study provided an opportunity to assess this question. Stable isotopes have been used for more than 30 years to study food web dynamics. This technique is especially applicable to the question of shifts in trophic system organization due to canopy modification because the carbon (C) and nitrogen (N) isotopic signature of algae differs from that of terrestrially-derived organic matter. A post-harvest change in C and N stable isotope values for stream organic matter and biota would be expected if algae became a more significant source of energy. We collected samples of organic matter sources (leaf litter, biofilm and instream detritus), macroinvertebrate and stream-associated amphibians to assess stream trophic response before and after timber harvest. Four buffer treatments were evaluated: clearcut harvest with the current Washington State Forest Practices riparian leave-tree buffer (FP treatment), a more extensive riparian buffer (100% treatment), and no buffer (0% treatment), compared with unharvested reference sites. We found relatively few differences among treatments before and after harvest. The responses we did observe were for N isotope ratios for gathering macroinvertebrate and C isotope ratio for giant salamanders ≤50 mm snout-vent length. The cause of these changes, however, did not appear to be related to a change in the contribution of algae to the trophic support of the streams as no change in either C or N isotopic ratios was observed for biofilm or detritus, the two major food sources for primary consumers. The fact that substantial post-harvest increases in light reaching the streams in some treated sites did not generate a significant change in isotope ratios suggests that these very small streams are tightly coupled to the bordering terrestrial environment, even under conditions apparently conducive to instream plant growth.

S2-2. INTRODUCTION

Modification of forest canopy density along fish-bearing streams has often been associated with shifts in trophic system organization. This type of response has been consistently reported for streams in the coastal rainforests of the Pacific Northwest (Murphy *et al.* 1981; Bilby and Bisson 1992; Kiffney *et al.* 2014), although response in streams in other regions of western North America with less-dense forest cover have varied (Fuchs *et al.* 2003; Melody and Richardson 2007).

Reduction in riparian vegetation, as often occurs with timber harvest, has the potential to influence food web organization in several ways (**Figure S2-1**; Richardson *et al.* 2005). Increased light resulting from removal or thinning of streamside vegetation often results in an increase in instream algal production (Murphy *et al.* 1981). Increased in-channel primary production has been associated with increased abundance of invertebrates that ingest algae, referred to as scrapers (Newbold *et al.* 1980; Hawkins *et al.* 1982). Reduction in riparian vegetation also often results in decreased input of terrestrial organic matter to streams (Bilby and Bisson 1992). Despite the reduction in input of leaves, needles and other types of terrestrial organic matter, there is little evidence that the type of macroinvertebrate that utilize this material (shredders) decline in response to canopy modification. Increased growth rate, production and/or density of vertebrate animals have often been reported following reductions in riparian canopy cover. This type of response has been reported for cutthroat trout (Murphy and Hall 1981), juvenile coho salmon (Bilby and Bisson 1992; Kiffney *et al.* 2014) and steelhead (Wilzbach *et al.* 2005).

Figure S2-1. Conceptual model of food web organization in the headwater streams examined in this study. Removal or thinning of riparian vegetation can increase light input, which may affect primary production. Reduced riparian cover also can reduce delivery of organic matter and nutrients to the channel. These types of changes have the potential to cause alterations in energy flow through the food web.



Quantitative examination of energy flow in trophic systems has been facilitated through the application of stable isotope analysis (Peterson and Fry 1987). This technique has been applied to food web analyses for the last three decades and has greatly enhanced the understanding of the organization of the food webs of various ecosystems. Stable isotopes, especially of carbon (C) and nitrogen (N), are useful for this purpose due to the difference in isotopic signature among system energy sources. For example, terrestrial organic matter typically contains lower proportions of the heavier C isotope (^{13}C) than aquatic algae. This difference in 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. N stable isotopes can be used in conjunction with 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 N (^{15}N) increases in concentration with each trophic exchange also makes N isotopes useful for determining the trophic level of an organism (DeNiro and Epstein 1978).

Nearly all research on trophic response to canopy modifications that has been conducted 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 would 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).

This study evaluated the response of stream food webs to riparian canopy modification through a basin-scale, manipulative experiment. In this study, small streams received one of four buffer treatments: unharvested reference sites (REF), clearcut harvest with riparian buffering along the entire perennial Type N (i.e., Np) channel (100% treatment), clearcut harvest with riparian buffering per the current Washington Forest Practice Rules, which requires a riparian buffer along a minimum of 50% of the Type Np stream length (FP treatment), and clearcut harvest with no riparian buffer (0% treatment). Specific details on the treatments may be found in Chapter 2 – *Study Design*. We evaluated the hypothesis that an increase in light reaching the streams at sites where canopy modification was most severe (FP and 0% treatments) would result in an increase in primary production in the channel, which would be reflected in a change in the stable isotope ratios of biofilm encrusting the streambed surfaces, which would then be transferred to invertebrates and amphibians. We assume that there is sufficient distinction in the isotopic signatures of terrestrial and algal organic matter sources to be able to determine if buffer treatments cause a shift in the degree to which these sources support higher trophic levels.

We conducted a comprehensive evaluation of the physical, chemical and biological responses to the application of these buffer treatments and we describe those results in the chapters in the main body of this report. This appendix examines changes in trophic organization related to application of the riparian buffer treatments as indicated by C and N stable isotope values.

S2-3. METHODS

We made stable isotope determinations for nine sample categories. Amphibians sampled included giant salamanders (*Dicamptodon spp*.) ≤50 mm snout-vent length (SVL), giant salamanders >50mm SVL, torrent salamander (*Rhyacotriton spp*.) post-metamorphs and torrent salamander larvae. In addition, we sampled macroinvertebrate gatherers and macroinvertebrate shredders. We also sampled materials that formed the base energy sources for the stream food web, including terrestrial organic matter (leaf litter) and two types of instream organic matter (detritus and biofilm). We intended this range of samples to enable us to assess if the buffer treatments altered the stable isotope ratios of the instream organic matter sources and if these changes were then transferred through the food web.

We attempted to collect samples of amphibians across all 17 study sites including six reference sites, four 100% buffer treatment sites, three FP treatment sites and four 0% treatment sites (a detailed description of the study sites may be found in Chapter 3 – *Management Prescriptions*). However, we obtained few samples from sites located in the South Cascade block; therefore, we did not include the three sites from this block (CASC-REF, CASC-FP and CASC-0%) in the analysis. Samples for macroinvertebrates, biofilm, detritus and leaf litter were restricted to the eight study sites in the Olympic and Willapa 1 blocks (see **Figure 2-3** in Chapter 2 – *Study Design*). We collected biofilm twice a year while macroinvertebrates, detritus and leaf litter were collected four times per year. We were not able to obtain samples of amphibians and macroinvertebrates at all sites for all sample dates. However, we analyzed 640 total samples during the study. The number of samples analyzed for each sample type, treatment and period (pre- and post-harvest) is provided in **Table S2-1**.

S2-3.1. BIOFILM

We used a wire brush to scrub biofilm off cobbles (64–256 mm) collected from the streambed within the reach where we sampled drift. 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. Sample processing consisted of thawing the sample and collecting the biofilm by filtering through a glass fiber filter with a nominal pore size of 1.2 µm.

Table S2-1. Number of samples analyzed by sample type, treatment, and period (pre- and post-harvest).

Sample Type	Treatment	Number of Samples	
		Pre-harvest	Post-harvest
Giant Salamanders >50mm	REF	5	9
	100%	3	8
	FP	1	4
	0%	3	6
Giant Salamanders ≤50mm	REF	5	11
	100%	3	8
	FP	1	6
	0%	3	8
Torrent Salamander Post-metamorph	REF	5	11
	100%	4	8
	FP	2	5
	0%	3	5
Torrent Salamander Larvae	REF	5	11
	100%	4	8
	FP	2	5
	0%	3	7
Macroinvertebrate Gatherers	REF	10	15
	100%	15	14
	FP	12	11
	0%	14	12
Macroinvertebrate Shredders	REF	9	5
	100%	2	2
	FP	5	5
	0%	7	5
Detritus	REF	19	15
	100%	21	14
	FP	20	11
	0%	23	12
Litter	REF	18	15
	100%	20	13
	FP	19	14
	0%	21	11
Periphyton	REF	7	11
	100%	7	10
	FP	12	8
	0%	13	7

S2-3.2. LITTERFALL

We collected litterfall using paired traps installed at four locations at each study site (see Chapter 12 – *Litterfall Input and Detritus Export*). We lined each trap with screen and retrieved screens from each site at six-week intervals. We combined nets from the paired traps at each sample location into a single plastic bag.

Litter samples were dried, thoroughly mixed and a subsample of 10 to 20 milligrams dry weight was selected for stable isotope analysis (see *Chapter 12 – Litterfall Input and Detritus Export* for details). Given the predominance of coniferous trees at our study sites, most sample material consisted of needles, although deciduous litter increased in abundance in autumn. Litter subsamples for isotopic analysis were wrapped in aluminum foil, placed in a labeled plastic bag, and frozen until further processing.

S2-3.3. DRIFT: DETRITUS AND MACROINVERTEBRATES

We obtained both detritus and macroinvertebrate samples from drift nets placed near the downstream end of our study sites (see Chapter 12 – *Litterfall Input and Detritus Export* and Chapter 14 – *Macroinvertebrate Export*). We placed drift nets with 250-µm mesh in each stream for roughly a 24-h period every six weeks throughout the year, as flows allowed. We stored drift samples to be analyzed for stable isotopes in stream water and kept them on ice until initial sorting.

In the lab, we stored the samples in a refrigerator for at least 24 h to allow the macroinvertebrates to evacuate their gut. We then rinsed the drift samples through 1-mm and 250-µm sieves and sorted the macroinvertebrates from the detritus. We designated detritus retained on the 1-mm sieve as coarse particulate organic matter (CPOM) and the detritus retained on the 250-µm sieve as fine particulate organic matter (FPOM). Quarterly, we dried each CPOM sample (see Chapter 12 – *Litterfall Input and Detritus Export*), thoroughly mixed each sample, and set aside 10 to 20 milligrams dry weight for the stable isotope analysis. We refer to this material as detritus for the remainder of this appendix. As with the leaf litter samples, we wrapped dried detritus samples in aluminum foil and froze them until further processing.

Macroinvertebrates were identified using keys in Merritt and Cummins (1996) and Stewart and Stark (1993). We consistently collected sufficient macroinvertebrate tissue to enable stable isotope determination for taxa in two functional feeding groups; gatherers, which feed by gathering fine particles of benthic organic material, and shredders, which ingest larger pieces of terrestrially- derived organic matter. We selected the two most abundant taxa in each feeding group for isotope analysis. Mayflies of the genus *Baetis* (Ephemeroptera: Baetidae) and dipteran larvae of the genus *Dixa* (Diptera: Dixidae) were, by far, the most common gatherer taxa analyzed. The most common shredder taxa were stoneflies of the genera *Yoroperla* (Plecoptera: Peltoperlidae) and *Zapada* (Plecoptera: Nemouridae). We attempted to include about 500 mg wet weight of each taxon in the sample used for isotopic determination. All captured individuals were included when less than 500 mg wet weight was collected. We stored each sorted sample in a labeled vial filled with water and froze the samples until further processing.

S2-3.4. STREAM-ASSOCIATED AMPHIBIANS

We collected tissue samples from two stream-associated amphibian genera: torrent salamanders and giant salamanders. Two species of torrent salamanders were present at the sampled sites, namely Olympic Torrent Salamander (*R. olympicus*) in the Olympics and Columbia Torrent Salamander (*R. kezeri*) in the Willapa Hills. Two species of giant salamanders, namely Cope's (*D. copei*) and Coastal (*D. tenebrosus*) in the Willapa Hills study sites, however, Cope's Giant Salamander is the only species present at the Olympic sites. We collected tissue from two developmental stages, or size classes, for each of the genera. Torrent salamanders were defined as larvae or post-metamorphs based on the presence or absence of gills. Giant salamanders were segregated into individuals with SVL $\leq$50 mm and >50 mm. We provide the rationale for the size break in the amphibian chapter of this report (Chapter 15 – *Stream-associated Amphibians*).

Our goal was to collect tissue samples from ten individuals for each taxa, life stage, study site, and year. However, insufficient numbers of animals were collected at many of the sites and sample dates to meet this goal. We collected tail tissue from all salamanders. We attempted to collect samples that were approximately 0.5 cm^2 in size; however, this was not possible for smaller animals for which we collected a smaller sample.

We used sterilized dissecting scissors or a razor blade to remove tissue and placed samples in a 1.5-ml sample vial on ice. We labeled each sample with the study site, taxa and life stage (or size). Once we returned to the lab, we placed all samples from a study site, taxa and life stage (or size) together in a plastic bag and stored them in the freezer until further processing.

S2-3.5. STABLE ISOTOPES 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^{o} 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.

S2-3.6. STATISTICAL ANALYSIS

Analyses evaluated the generalized null hypothesis:

$$\Delta T_{\mathrm{REF}} = \Delta T_{100\%} = \Delta T_{\mathrm{FP}} = \Delta T_{0\%} \qquad \text{(S2-1)}$$

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 ≤ 0.05, 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.

S2-4. RESULTS

We observed a treatment × period interaction effect for only two of the sample categories; $\delta^{13}C$ for giant salamanders ≤ 50 mm in length ($P = 0.02$) and $\delta^{15}N$ for macroinvertebrate gatherers ($P = 0.03$; **Table S2-2**). Pairwise comparisons for the giant salamanders indicated differences in $\delta^{13}C$ values between the 0% and the FP treatments ($P = 0.02$) and between the FP and 100% treatments ($P < 0.01$; **Table S2-3**). C isotope ratios were higher in both the 0% treatment and 100% treatment than in the FP treatment. There were also pre-post treatment differences detected for the 100% treatment ($P = 0.04$) and the FP treatment ($P = 0.02$; **Table S2-3**). The $\delta^{13}C$ values decreased following harvest in the FP treatment but increased after harvest in the 100% treatment. Differences in $\delta^{15}N$ for gatherers occurred between the FP treatment and reference sites ($P = 0.02$) and between the FP and 100% treatments ($P < 0.01$; **Table S2-4).** In both cases, the FP treatment exhibited lower $\delta^{15}N$ values. A pre-post treatment difference was detected for the FP treatment ($P < 0.0001$) and 0% treatment ($P = 0.03$) with a decrease in N isotope ratio following the application of the buffer treatments. We found no treatment effect in any other sample category ($P > 0.05$).

Table S2-2. P-values of the generalized linear mixed effects model ANOVA analysis. The analysis compared among treatment types and pre-post treatment application (treatment × period). P-values in **bold** print indicate P <0.05.

Sample Category	P-value	
	δ^{15}N	δ^{13}C
Giant Salamanders >50mm	0.13	0.09
Giant Salamanders ≤50mm	0.36	**0.02**
Torrent Salamander Post-metamorphs	0.63	0.77
Torrent Salamander Larvae	0.85	0.69
Macroinvertebrate Gatherers	**0.03**	0.12
Macroinvertebrate Shredders	0.68	0.68
Detritus	0.48	0.41
Litter	0.18	0.36
Periphyton	0.28	0.14

Table S2-3. Results of pairwise comparisons for δ13C for giant salamanders ≤50mm in length among treatments and pre-post treatment for each treatment type. Comparisons in **bold** print indicate P <0.05.

Treatment Comparison				
Contrasts	Estimate	95% CI		P-value
		Lower	Upper	
100% vs. REF	1.18	−0.09	2.45	0.07
FP vs. REF	−1.45	−2.91	0.01	0.05
0% vs. REF	0.47	−0.71	1.64	0.42
0% vs. FP	1.91	0.37	3.46	**0.02**
0% vs. 100%	−0.71	−2.08	0.65	0.29
FP vs. 100%	−2.63	−4.24	−1.01	**<0.01**
Pre-Post Harvest Comparison				
Contrasts	**Estimate**	**95% CI**		**P-value**
		Lower	**Upper**	
REF	−0.11	−0.86	0.64	0.76
100%	1.06	0.04	2.09	**0.04**
FP	−1.56	−2.82	−0.31	**0.02**
0%	0.35	−0.55	1.25	0.43

Table S2-4. Results of pairwise comparisons for δ 15N for macroinvertebrate gatherers among treatments and pre post treatment for each treatment type. Comparisons in **bold** print indicate P <0.05.

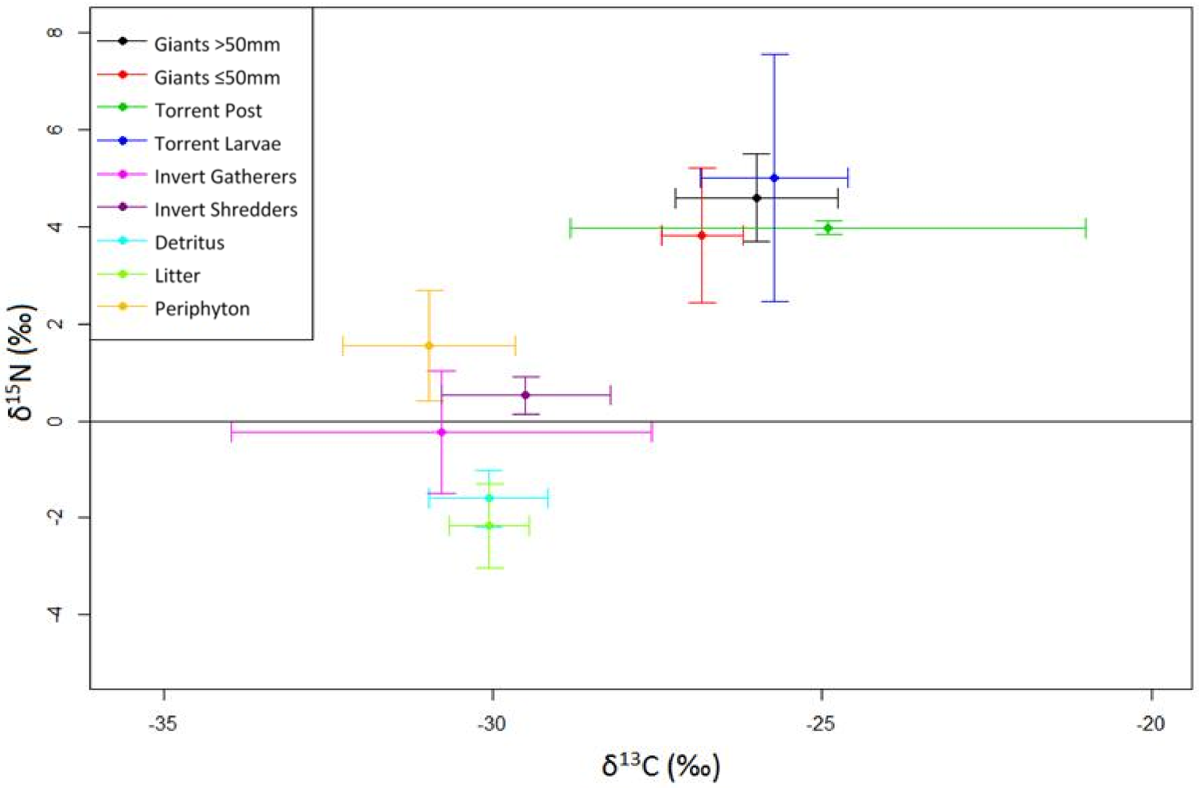
Treatment Comparison				
Contrasts	**Estimate**	**95% CI**		**P-value**
		Lower	**Upper**	
100% vs. REF	0.26	−1.19	1.71	0.71
FP vs. REF	−1.82	−3.27	−0.37	**0.02**
0% vs. REF	−0.57	−2.06	0.92	0.43
0% vs. FP	1.24	−0.21	2.69	0.09
0% vs. 100%	−0.84	−2.28	0.61	0.24
FP vs. 100%	−2.08	−3.48	−0.68	**<0.01**
Pre-Post Harvest Comparison				
Contrasts	**Estimate**	**95% CI**		**P-value**
		Lower	**Upper**	
REF	−0.65	−1.71	0.41	0.21
100%	−0.39	−1.38	0.60	0.42
FP	−2.47	−3.46	−1.48	**<0.0001**
0%	−1.22	−2.28	−0.17	**0.03**

Table S2-5. Average and (standard deviation) for N and C stable isotope values for the nine sample categories averaged across all treatments and years.

Sample Category	**$\delta\ ^{15}N$**	**$\delta^{13}C$**
Giant Salamander >50mm	4.04 (0.78)	−26.23 (0.92)
Giant Salamander ≤50mm	3.51 (0.73)	−27.24 (1.08)
Torrent Salamander Post-metamorphs	4.32 (0.81)	−25.09 (1.13)
Torrent Salamander Larvae	5.09 (0.72)	−25.96 (0.93)
Macroinvertebrate Gatherers	−0.29 (1.47)	−30.31 (3.55)
Macroinvertebrate Shredders	−0.50 (0.96)	−30.29 (2.06)
Detritus	−2.68 (1.21)	−29.31 (5.27)
Litter	−2.95 (1.00)	−29.84 (1.29)
Periphyton	0.49 (1.44)	−30.50 (2.36)

Stable isotope ratios for leaf litter, biofilm, detritus, shredders and gatherers were clustered and distinct from the values exhibited by the amphibians. $\delta^{13}C$ values for the non-amphibian samples ranged from −30.50‰ to −29.31‰ and $\delta^{15}N$ values from −2.95‰ to 0.49‰ (**Table S2-5**). In contrast, stable isotope values for amphibians were consistently higher than those for the other sample categories, with $\delta^{13}C$ values ranging from −27.24‰ to −25.09‰ and $\delta^{15}N$ values from 3.51‰ to 5.09‰. **Figure S2-2** provides an illustration of the degree of divergence between the amphibian and other sample types for the 0% treatment, but higher values for both N and C isotope ratios were observed at all sites sampled (**Table S2-5**).

Figure S2-2. Plot of δ13C versus δ15N for all sample categories analyzed for this study for the 0% treatment sites prior to the application of treatments. This figure illustrates the divergence between the organic matter and macroinvertebrate sample categories, clustered on the lower left of the plot, and the amphibian values on the upper right. The isotope values for each sample type are shown with 95% confidence intervals. The higher isotope values for the amphibians were consistent across all study sites.



S2-5. DISCUSSION

Although we were not able to consistently obtain isotope samples for all sample types, our results, in conjunction with the information developed by other components of this project, enabled us to arrive at three conclusions:

1) *There was no evidence of an increase in algal content in the biofilm in response to the buffer treatments.* Both by the isotope values and by the biofilm chlorophyll *a* and biomass values reported in Chapter 13 – *Biofilm and Periphyton* support this point.

2) *The system is primarily supported by energy subsidies from terrestrial sources, even after harvest.* The fact that the biofilm and detritus isotope values are very close to the litter values, and that they did not change post-harvest, support this point. The very low chlorophyll *a* values for the biofilm also support this conclusion; there is very little in-channel production of autotrophic organic matter.

3) *We did not sample an important component of the diet of the salamanders.* The difference in both N and C isotope values between the insects we sampled and the amphibians strongly suggest this conclusion. For this reason, we did not employ a mixing model to examine trophic organization.

Many studies of stream food webs have reported an increase in the level of trophic support provided by autotrophic production following thinning or removal of riparian vegetation (Busch 1978; Gregory 1980; Murphy and Hall 1981; Bilby and Bisson 1992). This shift is caused by increased light reaching the stream (Gregory *et al.* 1987) and, in some cases, by elevated levels of nutrients (Fredriksen 1971; Triska *et al*. 1983). However, we found little evidence that the application of the different riparian treatments altered food webs of our study streams, despite a substantial increase in light reaching the stream channel in two of the treatments (0% and FP treatments; Chapter 7 – *Stream Temperature and Cover*).

We were not able to obtain a sample of pure algae from our study sites. Therefore, we cannot verify that algae embedded in the biofilm at the study sites had an isotopic signature distinct from terrestrial organic matter at our study sites. It is possible that part of the reason for a lack of response to the treatments is due to the lack of differentiation in isotopic values between algae and terrestrial organic matter. However, our interpretation of the isotope data is consistent with the observation of no treatment effect on biofilm biomass or chlorophyll *a* (Chapter 13 – *Biofilm and Periphyton*).

Biofilm consists of a film of organic material with algal, bacterial and fungal cells embedded in it (Suberkropp 1998). The organic matrix of this film is largely produced by the flocculation of dissolved organic matter (Lock 1993). Therefore, the isotopic signature of the biofilm will reflect the origin of the dissolved organic matter that forms this matrix and the relative abundance of embedded algal, fungal and bacterial cells. An increase in the $\delta^{13}C$ values and/or $\delta^{15}N$ values for biofilm after application of the treatments would indicate an increase in the proportion of algal material. However, we found no changes in stable isotope ratios for biofilm after application of treatments ($P > 0.05$), indicating no evidence of an increase in algal abundance. The result of no increase in the proportion of algae in the biofilm after treatment suggested by the isotope data is consistent with the results reported in Chapter 13 – *Biofilm and Periphyton* of no increase in biofilm chlorophyll *a* or ash-free dry mass following treatment. Post-harvest shade provided by high levels of wood in the form of logging slash that covered portions of the stream channels in some buffer treatment sites where harvest occurred to the channel edge (see Chapter 6 – *Wood Recruitment and Loading*) may have been partially responsible for the lack of autotrophic response.

We did observe two statistically significant changes in isotope values for amphibians after application of the buffer treatments. However, these changes did not support our hypothesis that some of the treatments resulted in increased trophic support of the salamanders from autotrophic sources. $\delta^{13}C$ for giant salamanders $\leq$50 mm in length for the FP treatment were lower than those for the 0% treatment and the 100% treatment, and the FP treatment exhibited a decrease after harvest while the 100% treatment increased. A post-harvest increase in giant salamander $\delta^{13}C$ in the FP treatment would occur if algal material became a more significant source of energy for the stream food web after harvest. Streams having received the FP treatment would have received more light after application of the harvest. However, we observed no change in $\delta^{13}C$ values for biofilm and there was no evidence of an increased presence of algae. Similarly, the macroinvertebrate samples indicated no changes in $\delta^{13}C$ that would be consistent with an increase in algae in their diets. Therefore, the change in $\delta^{13}C$ for giant salamanders $\leq$50 mm cannot be attributed to an increased contribution of autotrophic production to the trophic support of the stream food web. In addition, the fact that we saw no difference in $\delta^{13}C$ values for the 0% treatment, a treatment that increased light levels more than the FP treatment, further indicates that the $\delta^{13}C$ response for the salamanders was likely not caused by increased instream primary production.

A change was also seen in $\delta^{15}N$ for macroinvertebrate gatherers ($P < 0.05$). Values for the FP treatment were lower than those for the reference sites and the 100% treatment. Pre-post treatment changes in $\delta^{15}N$ for gatherers occurred in the FP and 0% treatments, which both exhibited a post-harvest decline in $\delta^{15}N$. Here again, it is unlikely that the changes in $\delta^{15}N$ levels were due to an increase in autotrophic production in the streams. $\delta^{15}N$ values for algae are typically higher than those for terrestrial organic matter (Mulholland *et al*. 2000). The FP and 0% treatments would have received greater light following harvest, yet there was no change in biofilm $\delta^{15}N$ values and the gatherers exhibited a decrease in $\delta^{15}N$, opposite of what would be expected with an increase in algal material in their diet. We have no evidence of what factors may have contributed to the post-harvest decline in $\delta^{15}N$ values in gatherers at the FP and 0% treatments, but we also did not see any evidence that canopy modification increased the contribution of autotrophic production to the trophic support of these macroinvertebrate.

N stable isotopes also have frequently been used to indicate trophic position because $\delta^{15}N$ values increase from 2‰ to 4‰ with each trophic exchange (Minagawa and Wada 1984; Keough *et al*. 1996). Gatherers are primary consumers, however, and a change in trophic position in response to treatment application is not possible. $\delta^{15}N$ values for all biofilm samples collected during the study were higher than those for detritus and leaf litter, the other organic matter sources forming the base of the trophic system in these streams, and higher than those for gatherers (**Table S2-5**). Higher $\delta^{15}N$ values in biofilm than in terrestrially-derived organic matter has been observed in other studies (Mulholland *et al*. 2000). Therefore, the post-harvest change in $\delta^{15}N$ values seen for gatherers in the FP and 0% treatments could have been caused by a decrease in the proportion of biofilm in their diets, although it is unclear why canopy modification in the FP and 0% treatments would cause an increase in the proportion of terrestrially-derived organic matter in the diets of these macroinvertebrate. In the case of both the $\delta^{13}C$ response by the giant salamanders and the $\delta^{15}N$ response by gatherers, the cause of the changes cannot be determined from available data.

The lack of difference in $\delta^{13}C$ values between biofilm and leaf litter in this study indicates that the biofilm encrusting the streambed is largely derived from terrestrial organic matter rather than organic matter fixed within the stream. As noted earlier, biofilm containing a significant amount of algal material typically will display a $\delta^{13}C$ value higher than that for biofilm consisting primarily of terrestrial organic matter (Rosenfeld and Roff 1992; Mulholland *et al*. 2000). Biofilm consisting primarily of terrestrially-derived organic matter would be expected prior to application of the treatments as the low light levels experienced by these streams would inhibit algal growth. The fact that substantial post-harvest increases in light reaching the streams in the FP and 0% treatments did not generate a change in C isotope ratios suggests that autotrophic energy sources play a limited role in the support of food webs in these small streams, regardless of light level. A study of trophic dynamics in small alpine streams in Austrian Alps found that primary production was an important element of trophic support, although these systems were still highly dependent on terrestrial energy sources (Füreder *et al*. 2003). In contrast, autochthonous production appears to play a minimal role in the trophic dynamics of our study streams.

The only sample types that could be consistently obtained at all sites and sample dates were leaf litter, detritus and biofilm. These three sample categories were obtained across all seasons, while macroinvertebrate and amphibians were only consistently collected during the spring and summer months. The lack of complete temporal correspondence between our samples of litter, detritus and biofilm and the samples of consumers could have confounded our interpretation of trophic relationships if there were strong seasonal variations in the isotope values. However, we found significant seasonal variation only for $\delta^{15}N$ for litter, with autumn values higher than the other three seasons (ANOVA; $P \leq 0.05$), likely reflecting the influx of litter from deciduous vegetation during the fall. There was no significant seasonal variation in $\delta^{15}N$ for detritus or biofilm, and no seasonal variation in $\delta^{13}C$.

The values for both $\delta^{13}C$ and $\delta^{15}N$ exhibited by the amphibians are higher than would be expected if their primary food source was shredding and gathering macroinvertebrates (**Figure S2-1** and **Table S2-5**). Fractionation associated with trophic exchanges is usually between 2‰ and 4‰ for N and about 1‰ for C (DeNiro and Epstein 1978; Minagawa and Wada 1984). Therefore, if the gathering and shredding macroinvertebrates we sampled were the primary food source for giant and torrent salamanders, the amphibians would exhibit $\delta^{13}C$ values of about −29.5‰ and $\delta^{15}N$ of 1.5‰ to 3.5‰. However, the $\delta^{13}C$ of the amphibians averaged from −27.2‰ to −25.1‰ and the $\delta^{15}N$ values from 3.5‰ to 5.1‰. The elevated values suggest that the amphibians are utilizing a food resource that was not included in our samples. Elevated $\delta^{13}C$ values could result from consumption of terrestrial macroinvertebrate, which we did not sample. In fact, Hicks and colleagues (2008) observed consumption of terrestrial macroinvertebrates by larval torrent salamanders at our study sites in the pre-harvest period and other studies have documented the consumption of terrestrial macroinvertebrate by adult torrent salamanders (O’Donnell and Richart 2012). The higher than expected $\delta^{15}N$ values also could be caused by the consumption of predatory macroinvertebrate or predation on other amphibians, especially in the case of the giant salamanders (Sepulveda *et al*. 2012). The high values of the amphibian isotope ratios relative to the values for the macroinvertebrate we did sample indicate that the amphibian’s diet includes a broader array of items than just the most commonly available aquatic macroinvertebrate in these small streams. The high probability that we did not sample an important food resource for the amphibians precluded the use of a mixing model to determine relative proportion of various food items in amphibian diets.

The fact that we were not able to characterize fully the diet of the amphibians living in these small streams is, in part, due to the difficulty encountered in sampling. On many dates at many sites, we could not collect enough organisms in some sample categories to conduct stable isotope analysis. Obtaining some types of samples was so difficult that these organisms could not be included in the analysis. Scraping and predatory macroinvertebrates were two sample types for which we were not able to collect sufficient sample material consistently enough for analysis. Had we been able to assemble a more comprehensive set of samples, we may have been able to better characterize the diets of the amphibians utilizing these small streams and, possibly, been better able to ascertain whether the application of the buffer treatments had any impact on trophic organization.

S2-6. REFERENCES

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