Effectiveness of Forest Practices Buffer Prescriptions on Perennial Non-fish-bearing Streams on Marine Sedimentary Lithologies in Western Washington

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Washington State
Cooperative Monitoring, Evaluation, and Research Committee (CMER)
Report

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Washington State Forest Practices Board
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Washington State Department of Natural Resources
Olympia, Washington

August 2021 **CMER 2021.08.24**

Washington State Forest Practices Adaptive Management Program

The Washington Forest Practices Board (FPB) has adopted an adaptive management program in concurrence with the Forests and Fish Report (FFR) and subsequent legislation. The purpose of this program is to:

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

To provide the science needed to support adaptive management, the FPB made the Cooperative Monitoring, Evaluation, and Research Committee (CMER) a participant in the program. The FPB empowered CMER to conduct research, effectiveness monitoring, and validation monitoring in accordance with guidelines recommended in the FFR.

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 Forests and Fish Adaptive Management program. The project is part of the Type N 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.

Proprietary Statement

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TABLE OF CONTENTS

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EXECUTIVE SUMMARY

Stephanie Estrella, William Ehinger, Dave Schuett-Hames, and Greg Stewart

We assessed the effectiveness of riparian management zone prescriptions in maintaining riparian functions and processes in non-fish-bearing, perennial (Type Np) headwater streams in incompetent (easily eroded) marine sedimentary lithologies in western Washington. This study was proposed as a companion study to the Type N Experimental Buffer Treatment Study in competent (erosion-resistant) lithologies (Hard Rock Study). We evaluated the effects of the rules on riparian vegetation and wood recruitment, canopy closure and stream temperature, stream discharge and downstream transport of suspended sediment and nitrogen, and benthic macroinvertebrates (see Chapter 1 – *Introduction*). Results will inform the efficacy of current Forest Practices rules in meeting the objectives outlined in the Washington Forest Practices Habitat Conservation Plan (Schedule L-1, Appendix N).

We used a Multiple Before-After Control-Impact (MBACI) study design to compare treatment sites to reference sites (see Chapter 2 – *Study Design*). We evaluated two experimental treatments, including:

1) Reference (REF): unharvested reference sites with no timber harvest activities within the entire study site during the study period, and

2) Forest Practices treatment (TRT): clearcut harvest with a current Forest Practices (FP) riparian leave-tree buffer (i.e., clearcut harvest with a two-sided 50-ft [15.2-m] wide riparian buffer along at least 50% of the riparian management zone, including buffers prescribed for sensitive sites and unstable slopes).

The ten study sites included first-, second-, and third-order non-fish-bearing stream basins (with one treatment site divided into two sub-basins for some of the variables) located in managed second-growth conifer forests with marine sedimentary lithologies in the southwest Willapa Hills region. The study design incorporated one or two years of pre-harvest sampling (2012-2014), a harvest period (2013-2015), and at least two years of post-harvest sampling. A two-sided 30-ft equipment limitation zone applied to the entire stream length in all sites. Because of unstable slopes, total buffer area was 18 to 163% greater than a simple 50-ft buffer along 50% of the stream length.

Riparian buffers mitigate the impacts from upland timber harvest on streams by providing shade and bank stability, filtering runoff, and maintaining productivity through inputs of large wood, leaf litter, and other organic matter. We sampled riparian stands, tree mortality, and wood input and loading in riparian management zones (RMZs) and perennial initiation points (PIPs) at the uppermost point of perennial flow (see Chapter 3 – *Riparian Stand Structure and Wood Recruitment*). Implementation of complex riparian and unstable-slope prescriptions resulted in different post-harvest stand conditions, referred to as buffer types. Four RMZ buffer types included: 1) RMZ FP Buffers encompassing the full RMZ width, 2) RMZ <50ft Buffers narrower than the full RMZ width, 3) Unbuffered RMZs harvested to the edge of the channel, and 4) Reference RMZs embedded in unharvested forests. PIP buffer type included: 1) PIP FP Buffers surrounding the PIPs at treatment sites, and 2) Reference PIPs embedded in unharvested forests. In the RMZ FP Buffers and <50ft Buffers, density decreased by 33 and 51% and basal area decreased by 26 and 49%, respectively. In the PIP FP Buffers, density and basal area decreased by 52 and 46%, respectively. In the Unbuffered RMZs, density and basal area were reduced to near zero during harvest. Post-harvest mortality was 31% of density and 29% of basal area in the RMZ FP Buffers, and approximately 50% of both density and basal area in the PIP FP Buffers. Variability among treatment sites was high. Wind and physical damage from falling trees accounted for approximately 75% of mortality in RMZ FP Buffers and 81% of mortality in PIP FP Buffers, compared to <10% in the Reference RMZs and PIPs.

Buffer trees recruited into the stream provide a source of in-channel wood, which influences channel morphology and aquatic habitat and serves as a retention mechanism for sediment and organic matter. The RMZ FP Buffers and <50ft Buffers received inputs of 23 and 10 pieces of large wood per 100 m, respectively, during the post-harvest interval. Over 90% of recruited large wood volume came to rest above the bankfull channel. Channel large wood counts remained stable in the Reference RMZs through the third year post-harvest; increased in RMZ FP Buffers (8%), Unbuffered RMZs (13%), and PIP FP Buffers (25%); and decreased in RMZ <50ft Buffers (15%) and Reference PIPs (28%). Small wood frequency was highest in the Unbuffered RMZs in the first year post-harvest (13.0 pieces/bankfull width) but decreased by nearly 50% by the third year. The mean percentage of channel surface area covered by wood of all sizes in the first year post-harvest ranged from 27.6% in the Reference RMZs to 34, 43, and 42% in the RMZ FP Buffers, RMZ <50ft Buffers, and Unbuffered RMZs, respectively. Wood cover remained stable through the third year post-harvest in the Reference and Unbuffered RMZs, but increased in the RMZ FP Buffers and RMZ <50ft Buffers. The changes in stand structure in this study are similar to changes reported in the Type N Hard Rock Study (McIntyre *et al.* 2018) and Westside Type N Buffer Characteristics, Integrity, and Function (BCIF; Schuett-Hames and Stewart 2019) studies following harvest under the western Washington riparian prescriptions for Type Np streams. Aside from the greater frequency of unstable slope buffers in streams with incompetent lithologies, we did not observe obvious differences between lithologies in the effects of buffer treatments on stand structure, tree mortality, wood recruitment, or wood loading. Consistency in the results of the three studies increases confidence in these conclusions, and the current study expands the geographic and geomorphic context in which these results apply.

Shade provided by riparian vegetation influences stream temperature, which is an important determinant in biological processes and growth and survival of aquatic biota. We measured a decrease in riparian shade and an increase in water temperature in the buffer treatments after harvest (see Chapter 4 – *Stream Temperature and Cover*). Mean canopy closure decreased in the treatment sites from 97% in the pre-harvest period to 75%, 68%, and 69% in the first, second, and third post-harvest years, respectively, and was related to the proportion of stream buffered and to post-harvest windthrow within the buffer. The seven-day average temperature response increased by 0.6°C, 0.6°C, and 0.3°C in the first, second, and third post-harvest years, respectively. Changes in temperature were most closely related to canopy closure, but hyporheic exchange and stream discharge combined with canopy closure were possible factors at some sites. While the greatest change in temperature occurred during the July–August period, spring and fall temperatures were also elevated at most locations in all treatment sites.

Timber harvest can influence stream discharge through changes in canopy interception and evapotranspiration and the extension of the channel network by forest roads. Changes in runoff combined with bank disturbance from windthrow and a decrease in nutrient uptake through vegetation removal may also result in an increase in suspended sediment and nitrogen export. We measured discharge, suspended sediment export, and nitrogen export in four study sites. Despite significant effort, we were unable to develop discharge prediction equations that would allow us to estimate harvest treatment effects (see Chapter 5 – *Discharge and Suspended Sediment Export*). This was possibly due to the relatively low precipitation in the pre-harvest period, shorter than expected pre-treatment calibration periods, and differences in precipitation between sites. The suspended sediment data showed that the marine sedimentary lithologies sampled in this study were more erodible than the competent lithologies sampled in the Hard Rock Study. In this study, both the treatment and reference sites exported more sediment in the post-harvest period probably due to greater precipitation in the post-harvest period. The site with the greatest post-harvest period suspended sediment export was an unharvested reference site that happened to have streamside mass wasting upstream of the monitoring station. Mean total nitrogen (N) and nitrate-N concentration increased in all treatment sites after harvest, likely a result of reduced nitrogen uptake, while export increased at all sites due to a combination of higher concentration at the treatment sites and higher discharge at all sites (see Chapter 6 – *Nitrogen Export*). The estimated change in export was inversely related to the proportion of the stream buffered and may have been affected by the unusually dry weather and low stream discharge in the pre-harvest period.

Headwater streams contribute a substantial proportion of macroinvertebrates to downstream fish-bearing waters. Timber harvest may influence the macroinvertebrate community through changes in organic matter inputs and primary production, as well as changes in shade, temperature, discharge, sediment, and wood inputs. We found no major changes in benthic macroinvertebrate assemblages in our study sites after harvest (see Chapter 7 – *Benthic Macroinvertebrates*). While Ephemeroptera-Plecoptera-Trichoptera (EPT) richness and the Shannon H' diversity index decreased in the treatment sites after harvest, the metrics also decreased in the reference sites indicating broader environmental factors rather than a treatment effect. The response of the other metrics (total richness, EPT percent, and the fine sediment biotic index), functional feeding groups, and major taxonomic orders did not change, which possibly reflects the extensive buffers, increase in wood cover, and vegetation regrowth that provided enough shade to inhibit primary production and instream structure to retain particulate organic matter.

REFERENCES

McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn (technical coordinators). 2018. *Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington*. Cooperative Monitoring, Evaluation, and Research Report, CMER 18-100. Washington State Forest Practices Adaptive Management Program. Washington Department of Natural Resources, Olympia, WA. 890 p.

Schuett-Hames, D., and G. Stewart. 2019. *Changes in Stand Structure, Buffer Tree Mortality, and Riparian-associated Functions 10 Years after Timber Harvest Adjacent to Non-fish-bearing Perennial Streams in Western Washington*. Cooperative Monitoring, Evaluation, and Research Report, CMER 2019.10.22.B. Washington State Forest Practices Adaptive Management Program. Washington Department of Natural Resources, Olympia, WA. 39 p.

CHAPTER 1 – INTRODUCTION

Welles Bretherton and Stephanie Estrella

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1-1. INTRODUCTION

In 2001, the Washington State Forest Practices Board (WFPB) approved a comprehensive set of Forest Practices rules based on the Forests & Fish Report (USFWS 1999; WFPB 2001). The rules established guidelines for forest roads and culverts, reforestation, and timber harvest practices including protections for riparian management zones and unstable slopes, among others. The rules were designed to maintain riparian functions, such as shade, wood recruitment, and bank stability, that would (a) provide compliance with the Endangered Species Act for aquatic and riparian-dependent species, (b) restore and maintain riparian habitat to support a harvestable supply of fish, (c) meet the requirements of the Clean Water Act for water quality, and (d) keep the timber industry economically viable in the state of Washington. In addition, the rules led to the development of an adaptive management program administered by the Washington Department of Natural Resources' Adaptive Management Program and the Cooperative Monitoring, Evaluation, and Research Committee (CMER) to oversee scientific studies pertaining to the Forest Practices rules.

To study the effectiveness of riparian management zone prescriptions on Type N (non-fish-bearing) stream basins, CMER funded the Type N Experimental Buffer Treatment Study in Competent Lithologies (Hayes *et al.* 2005; Ehinger and Estrella 2007). That study (hereafter, Hard Rock Study) focused on streams with competent (erosion-resistant), volcanic lithologies with coarse substrates where Coastal Tailed Frog (*Ascaphus truei*) and other stream-associated amphibians were likely to be present. The Hard Rock Study examined the influence of buffer treatments on riparian inputs (wood, shade, litterfall, and sediment), water quality, stream-associated amphibians, and exports (streamflow, sediment, nutrients, instream detritus, and macroinvertebrates) to downstream fish-bearing waters.

This Type N Experimental Buffer Treatment Study in Incompetent Lithologies (hereafter, Soft Rock Study) was proposed as a companion study that focused on the effectiveness of the Type N riparian management zone prescriptions when applied in incompetent (more easily eroded), sedimentary lithologies. Recent studies in the Pacific Northwest have documented different responses to anthropogenic disturbance between volcanic and sedimentary lithologies in stream temperature (Dent *et al.* 2008; Hunter and Quinn 2009), relative bed stability (Kaufmann *et al.* 2009), and ecological condition (Kaufmann and Hughes 2006). These differences in response are likely the result of differences in erodibility and, therefore, sediment supply to headwater basins (Kaufmann and Hughes 2006). The Soft Rock Study included many of the same elements of the Hard Rock Study, but was restricted to incompetent, marine sedimentary lithologies. In addition, this study was limited to two experimental treatments—unharvested reference sites and sites treated with the current Forest Practices prescription. It did not include the 100% and 0% buffer treatments.

The purpose of the study was to evaluate the effects of the Forest Practices rules for westside Type N streams on canopy closure and stream temperature, stream discharge and downstream transport of suspended sediment and nitrogen, and benthic macroinvertebrates in incompetent lithologies. Results will inform the efficacy of current Forest Practices rules to meet the functional objectives listed below. This study was part of the formal adaptive management program for the Washington Forest Practices Habitat Conservation Plan (HCP; WDNR 2005) and the state Forest Practices rules and was one component in the Type N Riparian Effectiveness Program in the 2011 CMER work plan (CMER 2010).

1-2. FUNCTIONAL OBJECTIVES

The WFPB developed a series of key questions and functional objectives for adaptive management, outlined in Schedule L-1 of the Forest Practices HCP, Appendix N (WDNR 2005). These targets were developed through negotiations among private, state, federal, and tribal stakeholders and, in many cases, were not well defined.

The HCP (WDNR 2005) lists four goals of the current Forest Practices rules:

1) To provide compliance with the Endangered Species Act for aquatic and riparian-dependent species on non-Federal forestlands;

2) To restore and maintain riparian habitat on non-Federal forestlands to support a harvestable supply of fish;

3) To meet the requirements of the Clean Water Act for water quality on non-Federal forestlands; and

4) To keep the timber industry economically viable in the state of Washington.

We sought to inform the four Functional Objectives outlined in Schedule L-1 listed below. Note that several of the targets are subject to regulatory procedures outside the authority of this report, e.g., water quality criteria. This study intended to inform policymakers rather than address each target directly or completely.

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

2) **Large Woody Debris (LWD):** Develop riparian conditions that provide complex habitats for recruiting LWD.

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

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 that cause scouring, and maintaining the hydrologic continuity of wetlands.

1-3. RIPARIAN MANAGEMENT PRESCRIPTIONS FOR WESTSIDE TYPE N WATERS

1-3.1. RIPARIAN MANAGEMENT ZONES

Among other forest management practices, a Riparian Management Zone (RMZ) under Forest Practices rules protects all shorelines of the state (Type S), fish-bearing (Type F), and non-fish-bearing (Type N) waters in Washington. The transitions (type breaks) from Type F to Type N streams are determined through a field-verified, geographic information system (GIS), logistic regression model (WAC 222-16-030). These F/N type breaks are used to pinpoint the outlet locations of all Type N watersheds in western Washington. The rule definitions from WAC 222-16-030 for waters of the state are:

- **"Type S Water":** all waters, within their bankfull width, as 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 associated wetlands.

- **"Type F Water":** segments of natural waters other than Type S Waters, which are within the bankfull widths of defined channels and periodically inundated areas of associated wetlands; or within lakes, ponds, or impoundments having a surface area of 0.5 acre or greater at seasonal low water, and which in any case contain fish habitat or are described by one of four categories in WAC 222-16-030(2).

- **"Type Np Water":** all segments of natural waters within the bankfull width of defined channels that are perennial non-fish-habitat streams. Perennial streams are flowing waters that do not dry out at any time of the year under normal rainfall and include the intermittent dry potions of the perennial channel below the uppermost point of perennial flow.

- **"Type Ns Water":** all segments of natural waters within the bankfull width of the defined channels that are not Type S, F, or Np Waters. These are seasonal, non-fish-habitat streams in which surface flow is not present for some portion of a year of normal rainfall and are not downstream from any stream reach that is a Type Np Water. Ns Waters must be physically connected by an above-ground channel system to Type S, F, or Np Waters.

Riparian management prescriptions for Type N Waters vary by water type and location (east versus west of the Cascade Mountain crest). The RMZ for Type Np and Ns Waters in western Washington includes the following zones (WAC 222-30-021(2)):

1) **Equipment limitation zone:** A zone measuring 30 ft (9 m) horizontally from each outer edge of the bankfull width of Type Np or Ns Waters wherein 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 over 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 management zone:** A two-sided 50 ft (15 m) wide no-harvest riparian buffer, measured horizontally from each outer edge of the bankfull width, along at least 50% of the Type Np stream length, with the following additional restrictions:

   - Type Np Waters >1000 ft (305 m) long: RMZ of a minimum of 500 ft (152 m);
   - Type Np Waters 300 ft (91 m) to 1000 ft (305 m) long: RMZ at least equal to the greater of 300 ft or 50% of the entire length;
   - Type Np Waters <300 ft (91 m) long: RMZ buffered in its entirety.

3) **Sensitive site buffers:** No-harvest buffers specific to each sensitive site category (see WAC 222-16-010; **Table 1-1**).

Table 1-1. Sensitive site definitions and riparian management zone (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.	Two-sided 50-ft (15-m) wide no-harvest buffer around the outer perimeter of the perennially saturated area
Side-slope seep	Seeps within 100 ft (30 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.	Two-sided 50-ft (15-m) wide 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-m) radius no-harvest buffer centered on the intersection
Headwater spring	Permanent spring at the head of a perennial channel, coinciding with the uppermost extent of Type Np Waters.	56-ft (17-m) radius no-harvest buffer centered on the spring
Alluvial fan	An erosional landform consisting of a cone-shaped deposit of water-borne, often coarse-sized sediments.	No harvest within

1-3.2. UNSTABLE SLOPES

Unstable slopes cannot be harvested under Forest Practices rules without special review for compliance with State Environmental Policy Act guidelines (WAC 222-10-030). As a result, areas adjacent to Np streams that are designated as unstable landforms and processes (**Table 1-2**) are usually avoided in harvest unit layouts (Dieu *et al.* 2008). This practice can result in an extension of the RMZ buffer. These no-harvest zones vary in width due to the varying shapes of these unstable features. This often results in streamside buffers that are wider (or narrower, but longer) than the 50-foot minimum buffers otherwise established for riparian zones. In practice, and as occurred at our study sites, the additional protection for potentially unstable slopes has the potential of creating a variable-width buffer, along 50% to 100% of the length of the Np stream (**Figure 1-1**). Conversations with landowners, managers, and foresters during the site selection process revealed that this is a common occurrence throughout the region. A search of the Washington Department of Natural Resources Forest Practices Application/Notification GIS data revealed that, of the harvests that occurred within 50 ft of an Np stream, 81% contained unstable slopes (WDNR, unpublished).

Table 1-2. Unstable slope definitions under Forest Practices rules.

Unstable Slope Type	Definition
Bedrock hollow	Commonly spoon-shaped areas of convergent topography within unchanneled valleys on hillslopes steeper than 70% (~35°).
Convergent headwalls	Teardrop-shaped concave landforms, broad at the ridge top and terminating where headwaters converge into a single channel.
Deep-seated landslides	Those in which the slide plane or zone of movement is mostly below the maximum rooting depth of trees, to depths of tens to hundreds of feet.
Groundwater recharge area	The land up-gradient from an unstable slope that can contribute subsurface water to a landslide.
Inner gorges	Canyon walls created by a combination of stream down-cutting or undercutting and mass wasting on slope walls steeper than 70% (~35°).

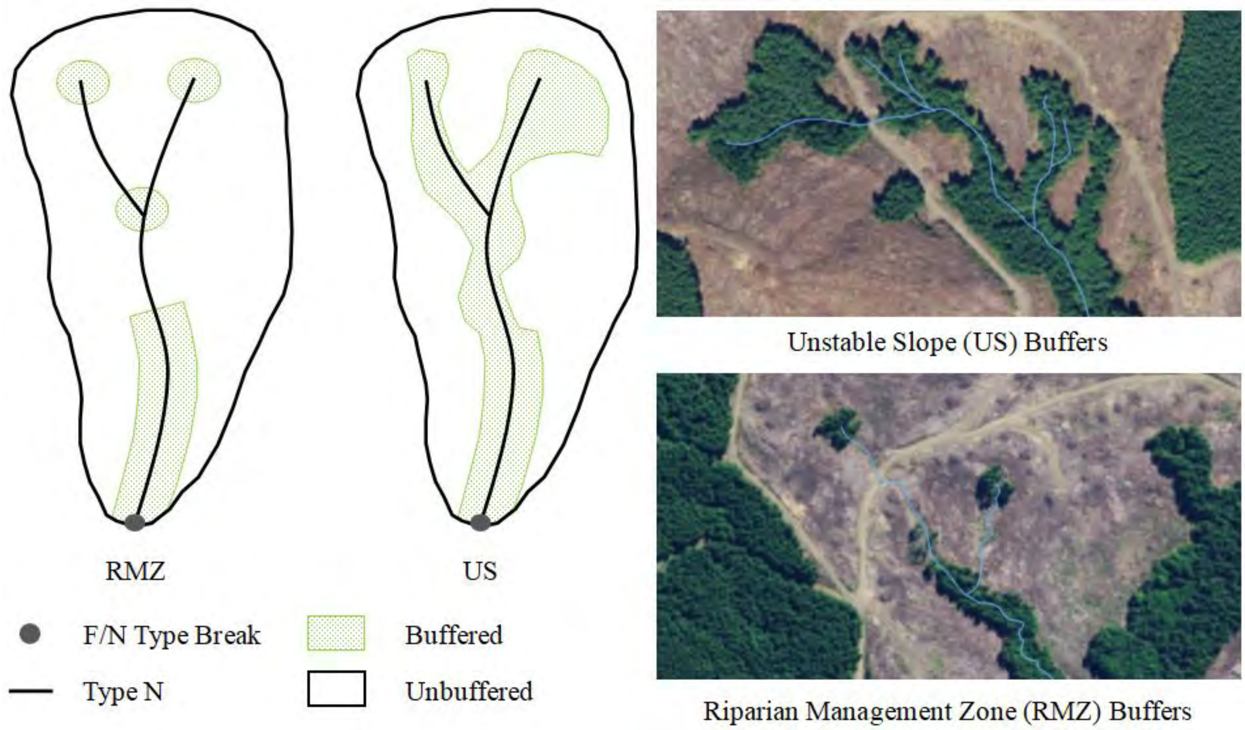


Figure 1-1. Conceptual examples of Type Np buffers with only riparian management zone (RMZ) buffers and with additional unstable slope (US) buffers (left) and examples from two Soft Rock Study treatment sites (right).

1-4. REFERENCES

CMER. 2010. *Fiscal Year 2011 CMER Work Plan*. The Cooperative, Monitoring, Evaluation, and Research Committee. Washington Department of Natural Resources, Olympia, WA.

Dent, L., D. Vick, K. Abraham, S. Schoenholtz, and S. Johnson. 2008. Summer temperature patterns in headwater streams of the Oregon coast range. *Journal of the American Water Resources Association* 44:803-813.

Dieu, J., A. Hook, G. Stewart, L. Vaugeois, C. Veldhuisen, and J. Black. 2008. *Mass Wasting Prescription-Scale Effectiveness Monitoring Project (Post-Mortem) Study Design*. Washington Department of Natural Resources, Olympia, WA. 86 pp.

Ehinger, W.J. and S.M. Estrella. 2007. *Quality Assurance Project Plan: Type N Experimental Buffer Treatment Study: Addressing Buffer Effectiveness on Riparian Inputs, Water Quality, and Exports to Fish-Bearing Waters in Basaltic Lithologies*. Washington State Department of Ecology, Lacey, WA. 101 p.

Hayes, M.P., W.J. Ehinger, R.E. Bilby, J.G. MacCracken, R. Palmquist, T. Quinn, D. Schuett-Hames, and A. Storfer. 2005. *Study Plan for the Type N Experimental Buffer Treatment Study: Addressing Buffer Effectiveness on Stream-Associated Amphibians, Riparian Inputs and Water Quality, and Exports to and Fish in Downstream (Type F) Waters in Basaltic Lithologies of the Coastal Areas and the South Cascades of Washington State*. A report submitted to the Landscape and Wildlife Advisory Group, Riparian Processes Scientific Advisory Group, and the Cooperative Monitoring, Evaluation, and Research Committee. Washington Department of Natural Resources, Olympia, WA. 68 p.

Hunter, M.A. and T. Quinn. 2009. Summer water temperatures in alluvial and bedrock channels of the Olympic Peninsula. *Western Journal of Applied Forestry* 24:103-108.

Kaufmann, P.R. and R.M. Hughes. 2006. Geomorphic and anthropogenic influences on fish and amphibians in Pacific Northwest coastal streams. *American Fisheries Society Symposium* 48:429-455.

Kaufmann, P.R., D.P. Larsen, and J.M. Faustini. 2009. Bed stability and sedimentation associated with human disturbances in Pacific Northwest streams. *Journal of the American Water Resources Association* 45:434-459.

USFWS. 1999. *Forests and Fish Report*. U.S. Fish and Wildlife Service and 11 other organizations. Washington Forest Protection Association, Olympia, WA. 177 p.

WDNR. 2005. *Final Forest Practices Habitat Conservation Plan*. Washington Department of Natural Resources, Olympia, WA.

WDNR. Unpublished. FP_GIS_FPA_. Forest Practices division. Washington Department of Natural Resources, Olympia, WA.

WFPB. 2001. *Washington Forest Practices: Rules, Board Manual, and Act*. Washington Department of Natural Resources, Olympia, WA. Variable pagination.

CHAPTER 2 – STUDY DESIGN

Welles Bretherton and William Ehinger

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

2-1. CRITICAL QUESTION

The critical question was how does clearcut harvest of Type N stream basins following the current Forest Practices riparian buffer requirements for western Washington Type N streams affect: (1) water temperature within and at the outlet of the Type N basin; (2) stream discharge and export of suspended sediment and nitrogen to downstream Type F waters; and (3) benthic macroinvertebrate communities.

2-2. EXPERIMENTAL DESIGN

2-2.1. MULTIPLE BEFORE-AFTER CONTROL-IMPACT DESIGN

We used a Multiple Before-After Control-Impact (MBACI) design to compare treatment sites to reference sites. This differs from the BACI design in that multiple reference and treatment sites were monitored. The MBACI design, with its replication of reference sites in space and time, provides an accurate estimate of the spatial and temporal variability throughout the pre- and post-treatment periods (Underwood 1994a, 1994b; Downes *et al.* 2002) and tests the assumption that any changes detected post-harvest are due to the treatment while accounting for climatic variability. In addition, having multiple control and treatment sites decreased the likelihood that the study would be compromised by the loss of one or more sites due to landowner withdrawal from the study, harvest delays, or other unforeseen circumstances. The design included data collection for a minimum two years before and two years after harvest in both the treatment and reference sites.

2-2.2. EXPERIMENTAL TREATMENTS

The entire Type N basin was harvested except those areas (rule-defined stream buffers and sensitive sites) described in **Section 1-3.** The experimental treatments included:

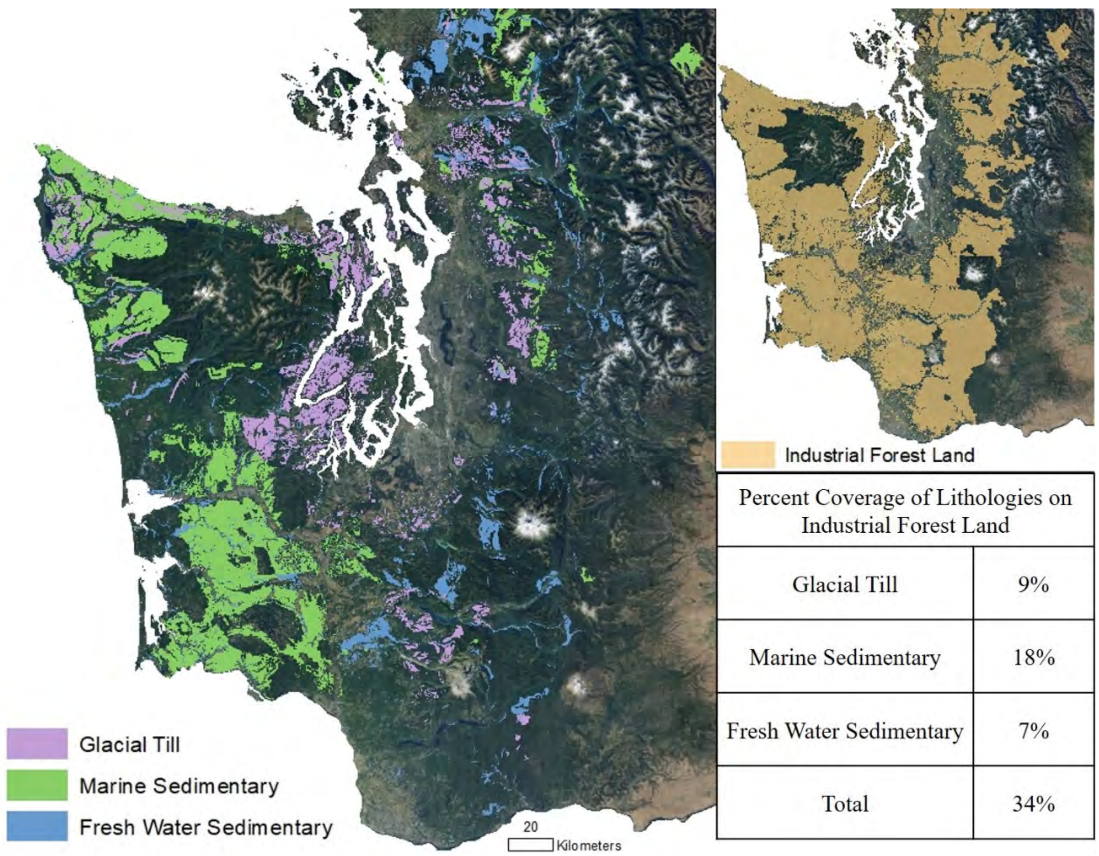
1) **Reference (REF):** unharvested reference sites with no timber harvest activities during the study period, and

2) **Forest Practices Treatment (TRT):** treatment sites with clearcut harvest following current Forest Practices rules (i.e., clearcut harvest with a two-sided 50 ft [15.2 m] wide riparian buffer along at least 50% of the perennial stream channel, including buffers prescribed for sensitive sites and unstable slopes).

The choice to designate a site as a TRT or REF was determined by the landowner's ability to harvest a treatment site during the study period or to hold off harvest of a reference site until the conclusion of the study. The success of this study hinged on the voluntary cooperation of landowners to adhere to our harvest schedule for the study's duration. Although not ideal from a study design perspective, it was impossible to randomly assign no-harvest reference treatments to harvest age stands on actively managed industrial forestlands due to the opportunity costs of not harvesting stands and the logistical costs of harvesting isolated stands. However, we did select sites that were well matched physically and geographically, which should minimize the effects of non-random assignment of treatments. The site selection process is summarized below and detailed in **Appendix A**.

2-3. SITE SELECTION

The Soft Rock Study was a companion study to the Hard Rock Study. In the latter, sites were selected based on presence of coarse substrates—where Coastal Tailed Frogs (*Ascaphus truei*) and other stream-associated amphibians were likely to be present in sufficient numbers to test responses to harvest. We used most of the same physical and stand-level criteria for the Soft Rock Study for comparability with the Hard Rock Study, except for the focus on lithologies likely to produce a fine-grained stream substrate (**Figure 2-1**). Our initial study plan called for the inclusion of glacial till and freshwater sedimentary lithologies. However, we were unable to find suitable sites representing these lithologies, so we selected from the available marine sedimentary sites following the other site selection criteria. For a detailed description of the site selection process, including modifications to the criteria, see **Appendix A**.



Percent Coverage of Lithologies on Industrial Forest Land	
Glacial Till	9%
Marine Sedimentary	18%
Fresh Water Sedimentary	7%
Total	34%

Figure 2-1. Distribution of glacial till and sedimentary lithologies across the industrial forest lands of western Washington, calculated in ArcMap (ESRI 2010).

The site selection criteria are described below. Reference sites were chosen based on proximity to the treatment sites with the same landscape criteria and a commitment from the landowner to hold off harvest until the end of the study. Site aspect was not a selection criterion and elevation was only included for the Cascade region, which was removed from consideration due to lack of suitable sites. The regional and lithological modifications mentioned above are documented in **Appendix A**. Our strategy, based on several other CMER studies, was to cast a wide net for potential sites, then, if numbers were sufficient, to select from these based on more stringent criteria. We were unable to find a sufficient number of sites, so included nearly every potential TRT site located near a suitable REF site.

Geographic location. Study sites were restricted to the area west of the crest of the Cascade Mountains in Washington State, where the westside Type N prescriptions apply.

Lithology. We initially searched for sites within marine sedimentary, glacial till, and freshwater sedimentary lithologies. Due to a paucity of suitable sites within the latter two lithologies, we chose basins largely comprised of marine sediment, as identified by the Washington Department of Natural Resources on the Southwest and Northwest Geologic Maps.

Stream gradient. Average stream gradient was restricted to 5 to 50% (3 to 27 degrees).

Basin size. Basin area was limited to less than 49 ha (120 ac). Forest Practices regulations limit the upper size of harvest units to 49 ha (120 ac) without review by an interdisciplinary team (WAC 222-30-025(1)). The entire basin was harvested, except areas described in **Section 1-3.**

Stand age. Stand age ranged from 30 to 80 years at time of harvest. Landowners indicated that 30 years was the minimum age for harvest. The maximum was set at 80 years; older stands are infrequently harvested on private and state lands thus they would not be representative of current practice in Washington.

Ownership. A single landowner controlled at least 80% of the basin area to increase the likelihood that harvests would occur on schedule.

Harvest timing. Harvest could occur between October 2013 and May 2015. No forest management activity (i.e., timber harvests, thinning, chemical application, and road construction) would occur in the reference basins until after October 2020.

Landowner commitment. To participate, landowners had to commit to the timing of harvest and access to their land for the study period.

2-4. SITE DESCRIPTIONS

Ten sites (seven treatments and three references) remained in the Soft Rock Study after identifying suitable sites and obtaining landowner commitments regarding timing of harvest and site access. These included first-, second-, and third-order non-fish-bearing streams in the Willapa Hills region of southwest Washington, draining to Crooked Creek and the Bear, Elochoman, Grays, and Naselle Rivers (**Table 2-1**; **Figure 2-2**). Treatment sites were numbered sequentially (TRT1 to TRT7) based on the percentage of stream length buffered (lowest to highest). This numbering system provided a quick reference to where sites fell along the gradient of buffer length (see **Section 2-5.** below). Stream discharge and downstream transport of suspended sediment and nitrogen were monitored in only four of the sites (REF1, REF2, TRT3, and TRT4) due to limited funding.

We originally intended TRT1 to include two sites or sub-basins of similar size, TRT1a and TRT1b, thus increasing the number of sites. However, after harvest the proportion of stream buffered in TRT1a was less than the 50% required by the Forest Practices rules. As a compromise, for some variables, e.g., temperature and canopy cover, we treated TRT1 as a single site. However, for the stand structure, tree mortality, wood input, wetted extent, and benthic macroinvertebrate metrics, we split the basin into TRT1a and TRT1b, resulting in a total of 11 sites (**Table 2-2**). The primary advantages of this split were that it normalized basin area and provided sub-basins of varying buffer length (**Table 2-1**; **Table 2-4**).

Table 2-1. Soft Rock Study site attributes, including basin aspect, elevation, and area, total length of stream network (including tributaries), mean bankfull width (BFW), mean wetted width (WW), and mean percent slope of stream channel and valley walls. Valley azimuth readings, taken at the riparian vegetation transects, were used to determine the general aspect of the basins. Average stream slope was derived from a 1m Digital Terrain Model (DTM; WDNR 2019). Valley wall slope was averaged across all slopes measured at the riparian vegetation transects. TRT1a and TRT1b are sub-basins of TRT1 (see text for details).

Study Site	Aspect	Elevation (m [ft])	Basin Area (ha [ac])	Stream Length (m [ft])	BFW (cm)	WW (cm)	Stream Slope (%)	Valley Wall Slope (%)
REF1*	SW	114 (373)	16 (40)	1456 (4777)	120	61	21	55
REF2*	SW	58 (190)	15 (38)	856 (2808)	125	50	18	53
REF3	W	46 (151)	12 (29)	697 (2287)	94	35	19	58
TRT1	NW	73 (241)	30 (76)	1827 (5994)	152	76	20	44
TRT1a	NW	73 (241)	13 (33)	797 (2615)	159	76	22	36
TRT1b	NW	74 (241)	17 (43)	930 (3051)	146	76	18	51
TRT2	NW	31 (101)	10 (25)	591 (1939)	114	52	13	58
TRT3*	NW	36 (118)	13 (31)	958 (3143)	102	54	15	58
TRT4*	NW	34 (111)	15 (38)	864 (2835)	126	49	18	60
TRT5	SW	63 (206)	14 (35)	1049 (3442)	79	32	19	64
TRT6	NE	46 (151)	12 (29)	992 (3255)	119	47	22	82
TRT7	SE	289 (947)	24 (59)	940 (3084)	192	68	30	71

*Sites with hydrology stations.

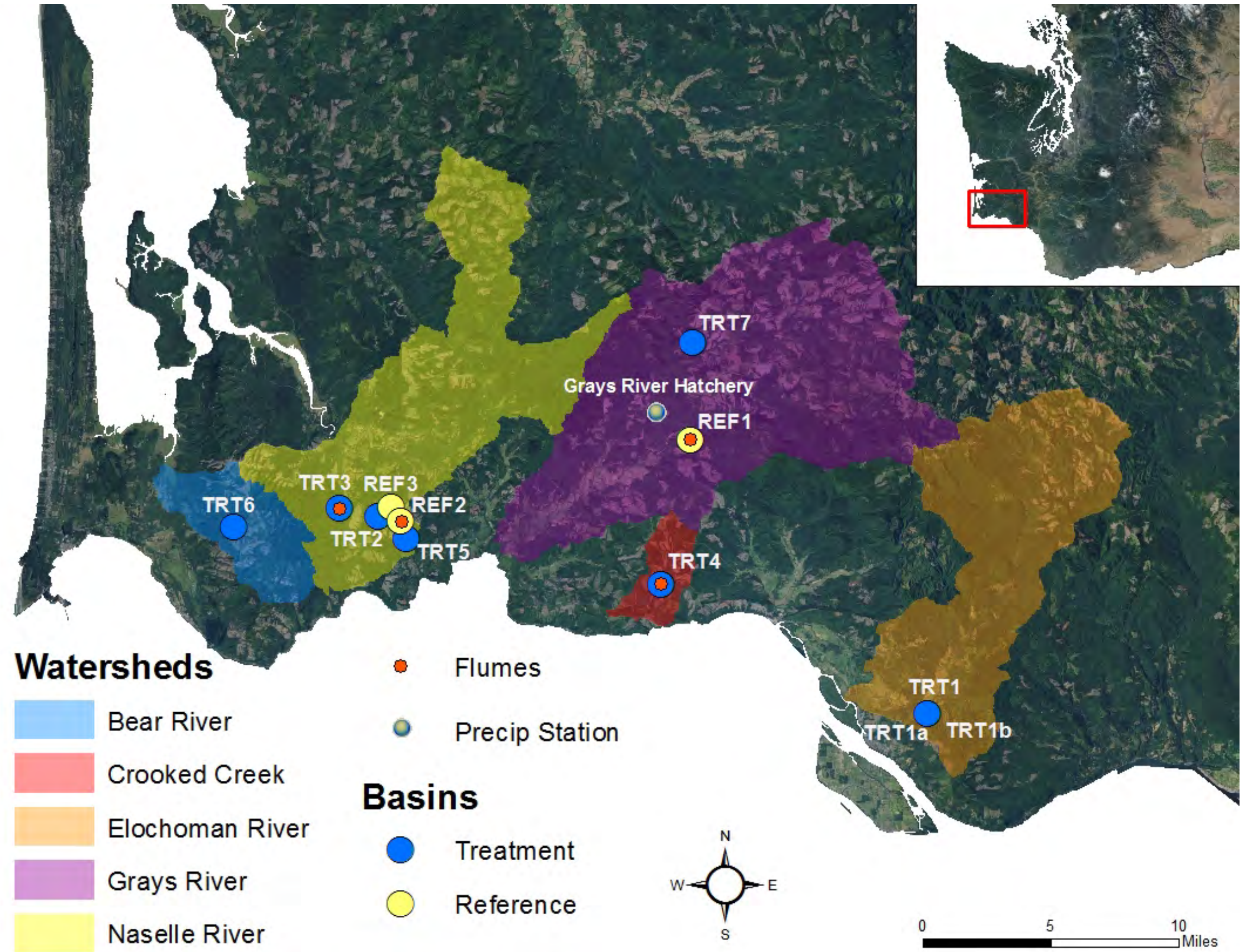


Figure 2-2. Map of the Soft Rock Study sites showing treatment type and location of flow and sediment monitoring stations (flumes). The Grays River Hatchery precipitation station (elevation 30.5 m) is also indicated.

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. Average annual precipitation over the study period (2012 to 2017) was 259 cm (102 in), as recorded by the Grays River Hatchery weather station (**Figure 2-2**; NOAA station ID: USC00453333). Thirty-year average precipitation from several weather stations in the vicinity ranged from 78 to 129 inches per year (see **Appendix A**) (PRISM Climate Group 2004).

Table 2-2. Parameters measured in the Soft Rock Study. Number of sites refers to individual basins and indicates that measurements were taken at the four flume sites (4; REF1, REF2, TRT3, TRT4), at all sites (10), or at all sites with TRT1 subdivided into two basins (11; TRT1a, TRT1b). See individual chapters for details.

<table>
<tr><th>Parameter Group</th><th>Parameter</th><th>Frequency</th><th>Number of Sites</th></tr>
<tr><td rowspan="7">Stream Measurements</td><td>Stage (ft)</td><td rowspan="2">Continuous</td><td rowspan="5">4</td></tr>
<tr><td>Turbidity (NTU)</td></tr>
<tr><td>Nitrate-nitrite (µg/L)</td><td rowspan="3">8/year, plus storm events</td></tr>
<tr><td>Total nitrogen (persulfate) (µg/L)</td></tr>
<tr><td>Suspended sediment concentration (mg/L)</td></tr>
<tr><td>Water temperature (ºC)</td><td rowspan="2">Continuous</td><td rowspan="3">10</td></tr>
<tr><td>Air temperature (ºC)</td></tr>
<tr><td rowspan="3">Riparian Measurements</td><td>Instream canopy cover (%)</td><td>Annual</td></tr>
<tr><td>Stand structure (trees/ha, basal area/ha, QMD, RD)</td><td rowspan="2">1-pre, 3-post</td><td rowspan="8">11</td></tr>
<tr><td>Tree mortality (trees/ha, basal area/ha)</td></tr>
<tr><td rowspan="4">Channel Measurements</td><td>Large wood recruitment (pieces/ha, pieces/100 m)</td><td rowspan="4">1-pre, 3-post</td></tr>
<tr><td>Large wood loading (pieces/100 m, volume/100 m)</td></tr>
<tr><td>Small wood loading (pieces/channel width)</td></tr>
<tr><td>Wood cover (% channel surface area)</td></tr>
<tr><td>Stream Network Surveys</td><td>Extent of the wetted channel (m)</td><td>Annual</td></tr>
<tr><td>Biota</td><td>Macroinvertebrates</td><td>3 times/year;
1-pre, 1-post</td></tr>
</table>

2-5. TREATMENT IMPLEMENTATION

Pre-harvest monitoring started in summer 2012, harvests began December 2013 in TRT3 and ended August 2015 in TRT5 (**Table 2-3**). At TRT3, delays in finding sites and an early harvest resulted in fewer than two pre-harvest sampling years for some parameters (discharge, turbidity, suspended sediment, nitrogen, and benthic macroinvertebrates). TRT5 was harvested relatively late (July 2015), but this did not affect pre-treatment sampling. The consequences of the asynchronous timing of harvest are discussed in the relevant chapters.

Table 2-3. Treatment implementation timeline, including the pre-harvest monitoring period (blue), post-harvest monitoring period (green), and active harvest period (grey; harvest months shown to the left).

Site	Harvest Months	2012	2013	2014	2015	2016	2017
TRT1	June - Oct						
TRT2	Oct - Nov						
TRT3	Dec - Feb						
TRT4	Nov - Mar	Pre-Harvest				Post-Harvest	
TRT5	July - Aug						
TRT6	Nov - Jan						
TRT7	Nov - May						

All timber harvests followed the Forest Practices rules (WAC 222-30). Mechanized and hand falling of timber occurred at all treatment sites as is common throughout the region. Lower gradient slopes were harvested by machine; trees on steeper gradients were cut by chainsaws and yarded by cable upslope to a tower. Both methods of transporting logs minimized ground disturbance in and near the stream. No bucking or limbing was performed on trees lying within the bankfull width of Type Np Waters and reasonable care was taken to avoid felling trees into the riparian management zone (RMZ; WAC 222-30-050). There was a two-sided 30-ft wide equipment limitation zone along the entire Type Np stream (WAC 222-30-021(2)(a)).

The addition of unstable slope buffers in all treatment sites sometimes resulted in riparian buffers that were longer (>50% of total stream length) than otherwise required (see **Section 1-3.2.** for explanation). Across all sites, the buffer area was 18 to 163% greater than a simple 50-ft-wide buffer along 50% of the stream length (**Table 2-4**). The percentage increase roughly corresponded to the percentage of stream length buffered. The addition of unstable slope buffers also resulted in riparian buffers of varying width.

Three of the seven treatment sites had average buffer widths that were slightly narrower (47 to 48 ft) than the 50 ft minimum. This reflected the addition of unstable slope buffers that averaged less than 50 ft wide. Wider unstable slope buffers at three other sites increased both the proportion of stream buffered and the mean buffer width. The distinctly wider buffer at TRT7 reflected the convergence of inner gorges high up in the stream basin and very wide and steep valley-wall buffers near the F/N break (see **Appendix B** for visual representations).

Table 2-4. The total area of the watershed harvested, total length and percentage of total length of stream network (including tributaries) buffered, mean buffer width, minimum buffer area had a 50% buffer been applied, actual buffer area (calculated in ArcMap), and percentage increase in buffer area from the minimum required length and width. TRT1a and TRT1b are sub-basins of TRT1 (see text for details).

Study Site	Harvest Area (%)	Buffer Length (%)	Buffer Length (m [ft])	Buffer Width (m [ft])	Buffer Area (ha [ac]) Minimum	Buffer Area (ha [ac]) Actual	Increase in Area (%)
TRT1	88	53	1068 (3504)	17 (56)	3.1 (7.6)	3.7 (9.1)	19
TRT1a	93	40	353 (1158)	15 (50)	1.2 (2.8)	1.0 (2.4)	N/A
TRT1b	87	63	615 (2018)	20 (65)	1.5 (3.6)	2.3 (5.6)	57
TRT2	88	54	358 (1175)	15 (50)	1.0 (2.5)	1.2 (2.9)	18
TRT3	85	58	661 (2169)	14 (47)	1.5 (3.8)	1.9 (4.7)	23
TRT4	85	92	775 (2543)	17 (56)	1.3 (3.3)	2.4 (5.9)	80
TRT5	78	95	1010 (3314)	15 (48)	1.9 (4.6)	3.1 (7.6)	65
TRT6	73	96	947 (3107)	14 (47)	1.6 (4.0)	3.1 (7.7)	94
TRT7	83	100	940 (3084)	23 (74)	1.6 (3.9)	4.2 (10.3)	163

2-6. MANAGEMENT ACTIVITIES

No new through roads were constructed in any of the study sites during the study. However, some minor spur roads were created near ridgelines to provide access to landings for logging equipment. Existing roads were rehabilitated before harvest. This included clearing the right of way, laying new rock on the road surface, and digging sediment traps with straw bales near stream crossings to capture any runoff during the active harvest period. The sediment traps remained unfilled throughout the study period. No new culverts were installed at any stream crossings (**Table 2-5**). Only TRT7 had active road use outside of the harvest period and that was largely associated with haul to and from a nearby rock quarry. The other treatment sites had native road surfaces pre-harvest and were inactive during the post-harvest period. The reference sites had native road surfaces throughout the study period and were only used by the monitoring crews.

At all treated sites, slash was gathered mechanically and piled. Slash piles were burned during the post-harvest monitoring period only in TRT7. In accordance with WAC 222-30-050, no machine piling of slash occurred within 30 ft of an unbuffered section of stream. A small amount of slash was removed from an unbuffered section of stream in TRT2 where the road crossed the stream.

Table 2-5. Road density, total length of roads, length of roads that occur within 30 m of a stream, and number of stream crossings within each study site.

Study Site	Road Density (m/m^2)	Total Road Length (m [ft])	Length Within 30 m of Stream (m [ft])	Stream Crossings
REF1	0.0045	730 (2396)	164 (541)	0
REF2	0.0060	922 (3025)	174 (573)	1
REF3	0.0045	523 (1719)	78 (258)	0
TRT1	0.0031	991 (3252)	150 (494)	2
TRT2	0.0059	588 (1932)	66 (217)	1
TRT3	0.0088	1096 (3598)	0 (0)	0
TRT4	0.0100	1547 (5077)	203 (666)	1
TRT5	0.0077	1084 (3557)	64 (212)	0
TRT6	0.0052	603 (1979)	135 (444)	2
TRT7	0.0092	2212 (7259)	129 (424)	1

Herbicide was applied at the treatment sites under the guidelines established by the Forest Practices rules (WAC 222-38). All applications were outside the required 50-ft no-spray buffer on Np streams, and there were no signs of herbicide-affected vegetation in the riparian zone.

Forest Practices rules (WAC 222-34) require reforestation of clearcuts that are not converted to other uses, or where such conversion is unlikely. All sites were replanted by hand with a mix of Douglas-fir (*Pseudotsuga menziesii*) and western hemlock (*Tsuga heterophylla*) within two years after harvest. Seedlings were planted in unbuffered portions of the RMZ. These were noted as planted trees and tracked post-harvest as part of the riparian regeneration surveys.

2-7. SCOPE OF INFERENCE, ASSUMPTIONS, AND LIMITATIONS

Our sites were limited in number and geographic area to those meeting the selection criteria and offered by cooperating landowners. Although the clustering of all sites in southwest Washington rather than in multiple incompetent lithologies across the west side of the state was unintentional, it had advantages as well as disadvantages. The former included a more uniform climate across sites and greater replication within a single lithology. The latter included limited representation of incompetent lithologies and climate.

The total number of sites was based on the budget available and the number of sites meeting the selection criteria. As noted earlier, we were able to find acceptable reference and treatment sites in one lithology and all ten sites were included in the study. There were three REF sites and seven TRT sites. Experience from the Hard Rock Study temperature analyses suggested that two reference sites were necessary (three is better) to provide multiple sites to pair with each treatment site, to estimate variability in shade over time, and to be able to evaluate the stationarity of the reference sites (i.e., were they on the same trajectory over time) with respect to stream temperature. We believe the unequal number of sites in each treatment had little effect because the mixed effects models used to estimate treatment effects were able to account for the unbalanced number of treatment replicates.

We were not able to randomly assign treatments. Harvest timing may be based on timber volume, species, age, market conditions, the logistics of moving equipment in and logs out of a site, as well as economic factors unique to each landowner. We believe the nonrandom assignment of treatments had little impact on the study results because the sites were well matched. However, the fact that the sites covered a relatively narrow range of forest conditions in western Washington means that direct inference is limited to similar conditions. This does not imply that results are not informative to other situations, but that the application of the results of this study should consider the variable in question, physical site characteristics, type and extent of forest harvest, and the physical processes involved.

2-8. REFERENCES

Downes, B.J., L.A. Barmuta, P.G. Fairweather, D.P. Faith, M.J. Keough, P.S. Lake, B.D. Mapstone, and G.P. Quinn. 2002. *Assessing Ecological Impacts: Application to Flowing Waters*. Cambridge University Press, Cambridge, UK.

ESRI. 2010. ArcMap 10.0. Environmental Systems Resource Institute, Redlands, CA.

PRISM Climate Group. 2004. *PRISM 1981-2010 Annual Precipitation*. Oregon State University, http://prism.oregonstate.edu, created 4 Feb 2004.

Underwood, A.J. 1994a. On beyond BACI: Sampling designs that might reliably detect environmental differences. *Ecological Applications* 4:3-15.

Underwood, A.J. 1994b. Spatial and temporal problems with monitoring. Pages 1010-1123 *in* P. Calow and G.E. Petts (eds.). *The Rivers Handbook: Hydrological and Ecological Principles, Volume Two*. Blackwell Scientific Publications, Oxford, UK.

WDNR. 2019. Lidar Portal. https://lidarportal.dnr.wa.gov/#46.32435:-123.77704:14.

CHAPTER 3 – RIPARIAN STAND STRUCTURE AND WOOD RECRUITMENT

Dave Schuett-Hames and Greg Stewart

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDIX TABLES

3-1. ABSTRACT

This chapter examines changes in stand structure and wood input associated with the western Washington riparian prescriptions for perennial non-fish-bearing (Type Np) streams in incompetent (sedimentary) lithology. We collected data on stand structure, tree mortality, large-wood input, and channel wood loading before and for three years after harvest at eight harvested treatment sites and three unharvested reference sites. At each site we sampled locations on the stream network where two types of prescriptions were applied: riparian management zones (RMZs) on both sides of the stream and perennial initiation points (PIPs) at the uppermost point of perennial flow. Implementation of complex riparian and unstable-slope prescriptions resulted in different post-harvest stand conditions, referred to as buffer types. There were four buffer types in the RMZs: 1) RMZ Forest Practices (FP) Buffers encompassed the full RMZ width, 2) RMZ <50ft Buffers were narrower than the full RMZ width, 3) Unbuffered RMZs had trees removed to the edge of the channel, and 4) Reference RMZs were embedded in unharvested forests. There were two PIP buffer types: 1) PIP FP Buffers surrounding the PIP at harvested sites, and 2) PIP References embedded in unharvested forests. The original study design called for statistical comparisons among buffer types, but this was not possible due to the low replication of reference sites and unanticipated differences in timing of harvest among treated sites. Consequently, we present only descriptive statistics (i.e., the averages of site means with standard errors) for each buffer type, without formal statistical comparisons of treatments or buffer types.

Pre-harvest riparian forests were 30- to 50-year-old second-growth stands dominated by western hemlock (*Tsuga heterophylla*) and Douglas-fir (*Pseudotsuga menziesii*). Stand structure three years post-harvest reflected differences in the number of trees removed during harvest and post-harvest mortality. In the RMZ FP Buffers and <50ft Buffers, density decreased by 33 and 51% and basal area decreased by 26 and 49%, respectively. In the PIP FP Buffers, density and basal area decreased by 52 and 46%, respectively. In the Unbuffered RMZs, density and basal area were reduced to near zero during harvest. Post-harvest mortality was 31% and 25% of density and 29% and 23% of basal area in the RMZ FP Buffers and RMZ <50ft Buffers, respectively, and ~50% of both density and basal area in the PIP FP Buffers. Variability among treatment sites was high. Wind and physical damage from falling trees accounted for ~75% of mortality in RMZ FP Buffers and <50ft Buffers, and 81% of mortality in PIP FP Buffers, compared to <10% in the Reference RMZs and PIPs.

The RMZ FP Buffers and <50ft Buffers received inputs of 23 and 10 pieces of large wood per 100 m, respectively, during the post-harvest interval. Over 90% of recruited large wood volume came to rest above the bankfull channel. The majority of recruited large wood pieces had attached rootwads. Channel large wood counts remained stable in the Reference RMZs through year 3 post-harvest; increased in RMZ FP Buffers (8%), Unbuffered RMZs (13%), and PIP FP Buffers (25%); and decreased in RMZ <50ft Buffers (15%) and Reference PIPs (28%). The proportion of functional pieces (those contributing to sediment retention or formation of pools, steps, or debris jams) ranged from 36 to 53% pre-harvest; remained stable through year 3 in Reference RMZs and PIPs; and decreased by ~10% in other buffer types.

Small wood frequency was highest in the Unbuffered RMZs one year post-harvest (13.0 pieces/bankfull width) but decreased by nearly 50% by year 3. RMZ FP Buffers and <50ft Buffers had 8.9 and 9.2 pieces/bankfull width in year 1, respectively, and also decreased by year 3. The mean percentage of channel surface area covered by wood of all sizes in year 1 ranged from 27.6% in the Reference RMZs to 34, 43 and 42% in the RMZ FP Buffer, RMZ <50ft Buffer, and Unbuffered RMZs. Wood cover remained stable through year 3 in the Reference and Unbuffered RMZs, but increased in the RMZ FP Buffers and <50ft Buffers.

The changes in stand structure in this study are similar to changes reported in the Type N Experimental Buffer Treatment Study in Competent Lithologies (Hard Rock Study; McIntyre *et al.* 2018) and Westside Type N Buffer Characteristics, Integrity, and Function (BCIF) Study (Schuett-Hames and Stewart 2019) following harvest under the western Washington riparian prescriptions for Type Np streams. In all three studies, RMZ and PIP reference stand structure was relatively stable in the first few years after harvest, whereas density and basal area decreased from 17 to 30% and 19 to 26%, respectively, in RMZ FP Buffers, and from 33 to 52% and 28 to 46%, respectively, in PIP FP Buffers. The changes in this study were at the upper end of these ranges, but only slightly higher than those of the Hard Rock Study. Wind and associated damage from falling trees was the dominant cause of tree mortality in all three studies, accounting for 74 to 90% of mortality in RMZ FP Buffers and 67 to 95% in PIP FP Buffers. Post-harvest input of large wood into streams was greater in RMZ and PIP FP Buffers than in the reference sites in all three studies.

Buffering of stream-adjacent unstable slopes in this study resulted in many buffers narrower than the full RMZ width in portions of the stream network where riparian buffers were not required, which was not observed in the Hard Rock Study or BCIF Study. The prevalence of unstable slope buffers reduced the stream length available for harvest to the edge of the stream.

Aside from the greater inclusion of unstable slope buffers in streams with incompetent lithologies, we did not observe obvious differences between lithologies in the effects of buffer treatments on stand structure, tree mortality, wood recruitment, or wood loading. The primary causes of change in stand structure and wood input in all three studies was the removal of trees from Unbuffered RMZs during harvest, and subsequent wind-related mortality of buffer trees. Consistency in the results of all three studies increases confidence in these conclusions, and the current study expands the geographic and geomorphic context in which these results apply.

3-2. INTRODUCTION

Buffers in riparian management zones (RMZ) adjacent to streams are an important component of Washington State's Forest Practices Habitat Conservation Plan (FPHCP) Riparian Strategy (WDNR 2005). Riparian buffers mitigate impacts from adjacent upland timber harvest and maintain productivity of aquatic systems (Gregory *et al.* 1991; MacDonald and Coe 2007; Richardson and Danehy 2007). Shade from buffer trees and other streamside vegetation moderates stream temperature (Wilkerson *et al.* 2006; DeWalle 2010; Janisch *et al.* 2012; Kibler *et al.* 2013). Tree fall from buffers contributes wood to stream channels, creating and maintaining productive aquatic habitat (Boyer *et al.* 2003; May and Gresswell 2003; Meleason *et al.* 2003; Montgomery *et al.* 2003; Martin and Shelly 2017), retaining sediment in headwater streams, and reducing sediment transport to downstream fish-bearing reaches (Montgomery and Buffington 1993; O'Connor and Harr 1994). Leaf litter and organic material from stream-adjacent vegetation support the aquatic food web (Grady 2001; Richardson *et al.* 2005). Streamside vegetation reduces sediment input by filtering runoff from adjacent uplands, while tree root systems stabilize streambanks and reduce erosion (Jackson *et al.* 2001; Litschert and MacDonald 2009).

The Forest Practices (FP) rules for riparian areas adjacent to non-fish-bearing perennial (Type Np) streams in western Washington require 15.2 m (50 ft) wide no-harvest RMZ buffers on both sides of Np streams over a minimum of 50% of the stream length in each Type Np basin, including the portion immediately upstream of fish-bearing waters (WFPB 2012). Buffers are also required on sensitive sites including perennial initiation points (PIP) located at the uppermost point of perennial flow. The remaining Np stream length can be harvested to the edge of the stream with limitations on the operation of equipment to minimize erosion. However, where there are unstable slopes, no-harvest buffers are required to prevent mass wasting.

Two previous studies by the Cooperative Monitoring, Evaluation, and Research Committee (CMER) have examined the effects of Type Np buffer prescriptions on stand structure and wood recruitment. The Type N Buffer Characteristics, Integrity, and Function (BCIF) Study documented elevated, but variable, tree mortality and large wood recruitment to the stream from RMZ and PIP buffers following harvest of adjacent uplands (Schuett-Hames *et al.* 2012; Schuett-Hames and Stewart 2019). Wind was the primary mortality agent. Shade declined after harvest due to mortality, but returned to similar levels as unharvested reference sites within five years. Streams adjacent to Unbuffered RMZs received variable, but often large, inputs of wood during harvest, but little additional wood during the decade after harvest. Shade was nearly absent in unbuffered stream reaches immediately after harvest but increased over time as shrubs and saplings became established. The Type N Experimental Buffer Treatment Study in Competent Lithologies (Hard Rock Study) compared changes in structure and wood input in basins that differed in the proportion of the Type Np stream network that was buffered (McIntyre *et al.* 2018). Patterns of mortality in Type Np RMZ and PIP buffers were similar to those of the BCIF study. Increases in spring and summer stream temperatures were also documented.

The Soft Rock Study focused on sites in incompetent sedimentary lithologies to augment earlier research on competent basalt lithologies. Although the primary goal of this study was to assess effects on stream temperature, we also examined factors likely to explain these effects, i.e., changes in shading associated with changes in riparian stand structure and wood input. This study of structural change and wood input in streams with incompetent lithologies expands the geologic and geographic context of existing studies of riparian ecosystem responses to FPHCP buffer prescriptions for Type Np streams in western Washington.

3-3. SAMPLING STRATEGY

The study sites included entire Type Np basins. There were eight treatment (TRT) sites where the Type Np riparian prescriptions were implemented during harvest and three reference (REF) sites where no harvest occurred. Although the initial intent was for harvest to occur synchronously among TRT sites, it occurred over a three-year period due to logistical constraints. Consequently, the sampling schedule differed for each of three groups of sites (**Table 3-1**). Treatment sites were sampled in the summer prior to harvest (Pre 1) and every summer for three years after harvest (Post 1-3). Reference sites were sampled each summer throughout the study period.

Table 3-1. Sampling schedule for three harvest-timing groups of treatment sites.

Harvest-Timing Group	Site	2013	2014	2015	2016	2017	2018
1	TRT3	Pre1	Post 1	Post 2	Post 3		
2	TRT1a		Pre 1	Post 1	Post 2	Post 3	
	TRT1b		Pre 1	Post 1	Post 2	Post 3	
	TRT2		Pre 1	Post 1	Post 2	Post 3	
	TRT4		Pre 1	Post 1	Post 2	Post 3	
	TRT6		Pre 1	Post 1	Post 2	Post 3	
3	TRT5		Pre 1	*	Post 1	Post 2	Post 3
	TRT7		Pre 1	*	Post 1	Post 2	Post 3

* Harvest activity not completed in time for post-harvest sampling in 2015.

Within each site (TRT or REF), we sampled two management zones with different harvest prescriptions under the Washington State Forest Practices Rules (WFPB 2012). RMZs are 15.2 m (50 ft) wide bands on both sides of westside Type Np streams along their entire length. PIPs are rule-designated sensitive sites located at the uppermost point of perennial flow.

3-3.1. RIPARIAN MANAGEMENT ZONES

In the reference sites, the RMZs were embedded in unharvested forests. In the treatment sites, implementation of the Forest Practices rules resulted in a mixture of post-harvest conditions. All treatment sites had no-harvest buffers covering the full width of the RMZ (15.2 m) on a minimum of 50% of the length of the Type Np stream network, including a 92 to 152 m (300 to 500 ft) long reach upstream of the transition point between fish-bearing and non-fish-bearing stream. Seven of eight treatment sites had unbuffered sections of the RMZ where trees were harvested to the edge of the stream.

At six of the eight treatment sites buffers were retained where there were unstable slopes in portions of the RMZ that otherwise would have been available for harvest to the stream edge. In these cases, an unharvested unstable slope buffer extended to the edge of the unstable area.

These unstable slope buffers varied in length and width according to the shape of the unstable area, but were narrower than the full width of the RMZ.

3-3.2. PERENNIAL INITIATION POINTS

The Forest Practice rules identify sensitive sites with specific management prescriptions. The most common of these is the PIP located at the uppermost point of perennial flow. PIPs occurred in all sites. PIPs in the treatment sites were surrounded by a 17.1 m (56 ft) radius no-harvest buffer with adjacent clearcut harvest, while PIPs in reference sites were embedded in unharvested forests.

3-4. METHODS

3-4.1. DATA COLLECTION

3-4.1.1. Standing and Fallen Trees

To sample stand structure, tree mortality, and tree fall in RMZs, we established strip plots at 40 m (131 ft) intervals along the Np stream channels (Marquardt *et al.* 2010). The centerline of each tree plot extended perpendicularly from the edge of the bankfull channel for a horizontal distance of 15.24 m (50 ft) on each side of the stream. Plots were 6.1 m (20 ft) wide (extending 3.05 m [10 ft] upstream and downstream from the centerline) yielding a total area of 185.9 m^2 (2000 ft^2). PIPs were sampled with circular 17.1 m (56 ft) radius plots centered on the PIP, corresponding with the buffer configuration specified in the Forest Practices rules. RMZ or PIP tree plots that intersected roads or overlapped other tree plots were excluded.

In each RMZ or PIP plot, all live and dead standing trees ≥10.2 cm (4 in) in diameter at breast height (DBH), measured at 1.4 m (4.5 ft) above the ground, were tallied on each visit. Live trees were marked with tree crayons and dead trees were painted with red tree paint, so that newly dead trees could be identified on subsequent visits. Condition (live or dead), species, and DBH were recorded for all trees. Canopy class for live trees was based on Stewart (1986); dominant and codominant trees were classified as overstory, intermediate or suppressed trees as understory, and isolated trees as open. For newly dead trees, we recorded the cause of mortality (wind, erosion, suppression, fire, insects, disease, physical damage, and unknown).

Data on large wood input (recruitment) from fallen trees was collected in the channel adjacent to each RMZ tree plot. For each newly fallen tree that originated in the plot we recorded the species and DBH. If at least 0.5 m (1.6 ft) intruded into or over the bankfull channel (termed ‘recruitment’), we recorded separate lengths and mid-point diameters for the portions that were within the bankfull channel (in-channel) and above the bankfull channel (over-channel).

3-4.1.2. Channel Large Wood Loading and Wood Cover

Channel wood plots were used to quantify large wood (LW) in and over the bankfull channel (channel LW loading). Plots extended 3.05 m upstream and downstream of the centerline of each tree plot. All new and existing pieces of wood >10 cm (4 in) in diameter and intruding ≥0.5 m (1.6 ft) into or over the bankfull channel were tallied (Gomi *et al.* 2001) and in- and over-channel length and mid-point diameters were recorded. Each piece was assigned a decay class (**Table 3-2**), and channel functions (sediment retention or formation of pools, steps, or debris jams) were recorded. Within each plot, we visually estimated the proportion of bankfull channel surface area covered by wood of any size (logs, branches, and slash) in years 1-3 post-harvest. In some plots, it was not possible to identify or count large wood pieces due to accumulations of wood and logging debris input during harvest.

3-4.1.3. Channel Small Wood

Counts of channel small wood pieces 1 to 10 cm (0.4 to 4 in) in diameter were made at shade-measurement transects located at 40-m intervals along the RMZ (see Chapter 4—*Stream Temperature and Cover*) in each post-harvest year. A tally was made of pieces that intersected the centerline within the bankfull channel. Bankfull width was measured according to Pleus and Schuett-Hames (1998).

Table 3-2. Large wood decay classes and defining characteristics. Adapted from Robison and Beschta (1990) and WDNR (1996).

Decay Class	Bark	Twigs (<3 cm)	Texture	Cross-sectional Shape	Color
1	Intact	Present	Intact	Round	Original
2	Partial	Absent	Intact	Round	Original
3	Trace	Absent	Smooth, some surface abrasion	Round	Original to darkening
4	Absent	Absent	Abrasion, some holes and openings	Round to oval	Dark
5	Absent	Absent	Vesicular, many holes and openings	Irregular	Dark

3-4.2. DATA ANALYSIS

3-4.2.1. Metrics

3-4.2.1.a. Stand structure

Live tree density, basal area, quadratic mean diameter (Curtis and Marshall 2000), and relative density (Curtis 1982) were computed for each plot in the summer before harvest (Pre 1) and the third year after harvest (Post 3). Live tree density (Den, trees/ha) was calculated by dividing the live tree count by the plot area (ha). Plot basal area (BA, m^2/ha) was calculated as the summed basal area (m^2) of live trees ($0.00007854 \times DBH^2$) divided by plot area (ha). Quadratic mean diameter (QMD, cm) was calculated as $\sqrt{(BA/Den/0.00007854)}$. Relative density ($RD_{qmd}$) was expressed as $BA/\sqrt{QMD}$. Net changes in live density and basal area over the study period were computed as Post 3 minus Pre 1 values. Percent change was calculated as 100 × Net change divided by the Pre 1 value.

3-4.2.1.b. Tree mortality

We computed the cumulative post-harvest mortality of trees ≥10 cm DBH (excluding harvested trees) in units of density (trees/ha) and basal area (m^2/ha). We also computed these metrics as a percentage of live trees present post-harvest, i.e., as cumulative post-harvest mortality in Post 3 divided by live density or basal area in the Pre 1 sample (excluding any harvested trees).

3-4.2.1.c. Large wood input

Large wood input (recruitment) to the stream channel from fallen trees originating in tree plots was calculated per 100 m of channel length by dividing the LW piece count by 6.1 m (the length of channel in RMZ tree plots) and multiplying by 100.

The volume of each recruited piece in or over the bankfull channel was estimated using the formula:

$$V = \pi \cdot r^2 \cdot L \qquad (3\text{-}1)$$

where: V is LW recruitment piece volume in m^3,
r is the midpoint radius in m, and
L is the piece length in m.

The LW recruitment volume in m^3/100 m of stream length was calculated for each tree plot by dividing the LW recruitment volume by 6.1 m and multiplying by 100. Because fallen trees are more likely to persist in streams if their roots are attached (Fox and Bolton 2007), we also computed the number and volume of pieces that had rootwads (SWRW).

3-4.2.1.d. Channel large wood loading

For each channel wood plot we computed the total number and combined in- and over-channel volume of LW pieces per 100 m of stream length for the pre-harvest (Pre 1) and post-harvest (Post 3) samples. We summed the LW piece counts and in- and over-channel volumes, then divided by the stream length (6.1 m) and multiplied by 100. We calculated the percentage of new wood pieces by dividing the number of pieces in decay classes 1 and 2 (**Table 3-2**) by the total count, and the percentage of functional pieces by dividing the number of pieces with one or more channel functions by the total count.

3-4.2.1.e. Channel small wood loading

For each small wood transect we computed the number of small wood pieces per bankfull width (count divided by the bankfull width in meters).

3-4.2.2. Analysis

For each treatment site we assigned RMZ plots to one of three RMZ buffer types based on the width of the buffer: buffers that met or exceeded the width requirement of 15.24 m (RMZ FP Buffer), buffers narrower than 15.24 m (RMZ <50ft Buffer), and areas without buffers (Unbuffered RMZ). Treatment PIP plots had a 17.1 m radius no-harvest buffer (PIP FP Buffer). Reference RMZs and Reference PIPs were embedded in unharvested forest. The distribution of plots by site and buffer type is shown in **Table 3-3**.

Table 3-3. Numbers of riparian management zone (RMZ) and perennial initiation point (PIP) plots by site and buffer type.

Site	RMZ				PIP	
	FB Buffer	**<50ft Buffer**	**Unbuffered**	**Reference**	**FP Buffer**	**Reference**
TRT1a	6		10		3	
TRT1b	13		7		2	
TRT2	5	3	3		2	
TRT3	6	5	5		4	
TRT4	10	5	2		2	
TRT5	11	6	1		5	
TRT6	10	1	1		4	
TRT7	11	2			3	
REF1				24		6
REF2				15		2
REF3				12		3

Because harvest timing varied among TRT sites, we selected appropriate survey data for each TRT site depending on harvest timing (**Table 3-1**) to calculate metrics for the year immediately prior to the start of harvest (Pre 1), year 3 post-harvest (Post 3), and the Pre 1 to Post 3 interval. For example, to calculate TRT site means for Post 3 we used 2016 data for sites in TRT harvest-timing group 1, 2017 data for harvest-timing group 2, and 2018 data for harvest-timing group 3. Then plots from each TRT site were sorted by buffer type and averaged to obtain site means, and the site means were averaged to obtain overall means for each buffer type.

Variation in harvest timing complicated selection of appropriate reference data to use in the analysis because individual TRT and REF sites were not paired. All three REF sites were sampled each year, and since there was variation among years, no single year of REF site data accounted for the interannual variability in the TRT data due to differences in harvest timing. To address this issue, we computed a weighted mean value for the REF data. This was done by selecting REF data from each of the same years used in the TRT site calculations for the different harvest-timing groups. For example, to calculate Post 3 REF values, we calculated separate REF site means (by REF buffer type) for the 2016, 2017, and 2018 surveys. Although we could have calculated a simple average giving equal weight to each of the three years, we chose to weight the values according to the number of sites in each of the TRT groups to mimic the temporal distribution of the TRT data. To compute the weighted REF values we multiplied the REF value for each year by the proportion of treatment sites in the harvest timing group (group 1 = 0.125, group 2 = 0.625, and group 3 = 0.25) to obtain the weighted mean REF value. For example, to calculate the weighted mean for Post 3 at each REF site, we multiplied the 2016 value by 0.125, the 2017 value by 0.625, and the 2018 value by 0.25, and summed the products. The site means were averaged to produce overall means for the REF RMZs and PIPs. **Appendix Table 3-1** shows the original reference site data by survey year, and the simple and weighted Pre 1 and Post 3 means.

Our original intent was to statistically compare the responses of buffer types. However, the small number of reference sites (3), unbalanced treatment design, non-random allocation of treatments, and complex implementation of buffer types (producing widely varying numbers of plots per type) prevented formal statistical analysis. Instead, we present simple descriptive statistics, focusing on buffer type means and standard errors. For each response variable we first computed the mean value of each buffer type based on plot values at each site. Site values were then averaged to generate a buffer-type mean and to calculate standard errors (SE) using JMP 13 software (SAS Institute Inc. 2017). Box plots showing the distributions of site values by buffer type were created in R (R Development Core Team 2012). These include the median (horizontal line), central quartile (or central 50% of the data, as a box), non-extreme values (whiskers), and outliers (points).

3-5. RESULTS

3-5.1. CHANGE IN STAND STRUCTURE

Prior to harvest, stream-adjacent forests were dominated by 30- to 50-year-old second-growth stands that had regenerated after previous harvests. Stands were dominated by western hemlock (*Tsuga heterophylla*) and Douglas-fir (*Pseudotsuga menziesii*). Stand structure varied among sites (**Appendix Table 3-2**). Before harvest, conifers comprised 70 to 99% of the basal area, live density ranged from 403 to 823 trees/ha (163 to 333 trees/ac), mean live basal area ranged from 39.8 to 50.2 m^2/ha (173 to 219 ft^2/ac), and relative density ranged from 47.9 to 58.4. Mean pre-harvest densities were highest, and QMDs were lowest, in the Reference RMZs and PIPs, resulting in overlap of pre-harvest basal area and relative density among treatment and reference sites (**Tables 3-4** and **3-5**).

Differences in stand structure among buffer types in Post 3 reflected differences in the number of trees removed during harvest and in post-harvest mortality. There was little change in stand structure in Reference RMZs or PIPs: mean density declined by ~5% and basal area increased by ~6% (**Table 3-4**). In contrast, there were substantial changes in structure in the RMZ FP Buffers, RMZ <50ft Buffers, and PIP FP Buffers. Density decreased by 33 and 51% and basal area by 26 and 49% in the RMZ FP and <50ft Buffers, respectively, with high variability among sites. The PIP FP Buffers decreased by 52% in density and 46% in basal area (**Figure 3-1**; **Table 3-4**). Nearly all trees were removed from Unbuffered RMZs during harvest (>99% of basal area) (**Table 3-4**).

Table 3-4. Mean live density (trees/ha) and live basal area (m^2/ha) before harvest (Pre 1) and three years after harvest (Post 3) and the net change from Pre 1 to Post 3. Buffer-type means and standard errors (SE) are based on site means (numbers of sites vary by type).

Plot Type	Buffer Type	*n*	Pre 1		Post 3		Change: Pre 1–Post 3		
			Mean	**SE**	**Mean**	**SE**	**Mean**	**SE**	**%**
Live Density (trees/ha)									
RMZ	Reference	3	730.7	173.3	681.3	147.3	-49.3	26.0	-5.6
	FP Buffer	8	484.3	62.2	332.1	72.7	-152.1	38.7	-33.1
	<50ft Buffer	6	524.1	32.5	259.8	49.2	-264.3	30.0	-51.1
	Unbuffered	7	461.3	32.3	8.5	5.5	-452.8	29.1	-98.3
PIP	Reference	3	822.8	244.3	779.5	217.9	-43.3	26.5	-4.5
	FP Buffer	8	403.4	23.6	207.3	47.6	-196.1	41.2	-51.8
Live Basal Area (m^2/ha)									
RMZ	Reference	3	41.7	1.3	44.2	1.0	2.5	0.5	6.3
	FP Buffer	8	39.8	2.2	28.1	4.4	-11.7	4.8	-25.6
	<50ft Buffer	6	47.7	3.3	25.0	5.5	-22.8	3.1	-48.7
	Unbuffered	7	50.2	3.9	0.3	0.3	-49.9	3.9	-99.1
PIP	Reference	3	43.5	2.1	46.5	1.9	3.0	0.5	6.9
	FP Buffer	8	42.3	2.1	23.1	5.1	-19.2	5.1	-45.5

Table 3-5. Quadratic mean diameter and relative density before harvest (Pre 1) and three years after harvest (Post 3). Buffer-type means and standard errors (SE) are based on site means (numbers of sites vary by type).

Plot Type	Buffer Type	*n*	Quadratic Mean Diameter (cm)				Relative Density			
			Pre 1		Post 3		Pre 1		Post 3	
			Mean	**SE**	**Mean**	**SE**	**Mean**	**SE**	**Mean**	**SE**
RMZ	Reference	3	28.0	2.5	29.7	2.5	54.8	4.4	56.4	3.7
	FP Buffer	8	33.6	1.4	35.7	1.5	47.9	2.8	34.7	4.9
	<50ft Buffer	6	34.8	1.3	35.1	2.7	56.2	3.6	31.2	5.6
	Unbuffered	7	37.4	1.6	20.5	7.8	56.7	3.5	7.3	6.0
PIP	Reference	3	27.5	3.3	29.1	3.4	58.4	6.1	60.6	5.5
	FP Buffer	8	37.1	1.4	38.8	1.8	48.3	1.9	25.9	5.6

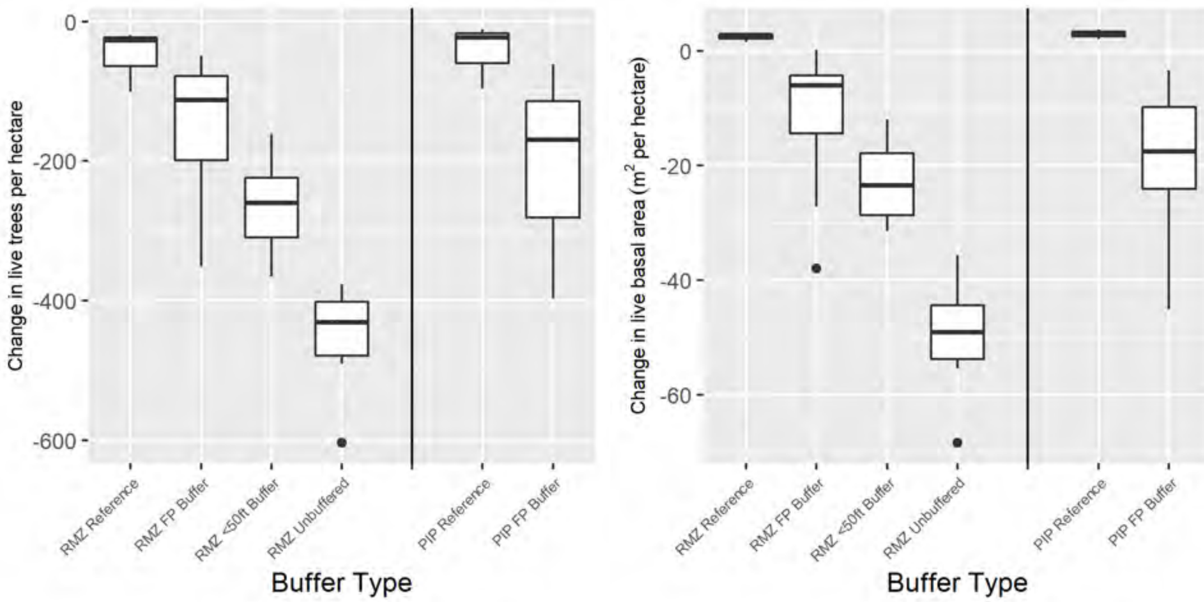


Figure 3-1. Change in live density (left panel) and live basal area (right panel) from before harvest to three years post-harvest. The horizontal line is the median, the box is the central quartile (or central 50% of the data), the whiskers are non-extreme values, and the points are outliers.

In the reference plots, mean cumulative post-harvest mortality during the 3-year post-harvest interval was only 6.5% (RMZ) and 5.0% (PIP) of live density and 3 and 2% of live basal area (**Table 3-6**). In contrast, post-harvest mortality in the RMZ FP and <50Ft Buffers was 31 and 25% of density, respectively, and 29 and 23% of basal area. Moreover, there was considerable variation in mortality among sites (**Table 3-6**; **Figure 3-2**). For example, mortality exceeded 65% in the RMZ FP Buffers at the TRT1 and TRT2 sites (outlying points in **Figure 3-2**), elevating the mean mortality of the RMZ FP Buffer type. There were no RMZ <50ft Buffers in these two treatment sites, which likely lowered the mean mortality for the <50ft Buffer type. On average, the greatest cumulative mortality occurred in the PIP FP Buffer type, reaching ~50% of residual density and basal area by Post 3. As with the RMZs, there was considerable variability among sites (**Figure 3-2**).

Windthrow and physical damage from falling trees accounted for ~75% of mortality in the RMZ FP and <50ft Buffers, and for 81% of mortality in the PIP FP Buffers. Overstory trees accounted for 62 to 76% of mortality in these buffer types, with mean diameters of 27 to 32 cm. In contrast to the treated sites, <10% of trees died due to wind or physical damage in the Reference RMZs and PIPs. Here, overstory trees accounted for ~20% of mortality and mean diameters were smaller (16 to 19 cm; **Table 3-6**).

Table 3-6. Cumulative tree mortality over the three-year post-harvest interval. Buffer-type means and standard errors (SE) are based on site means (numbers of sites vary by type).

Plot Type	Buffer Type	*n*	Density (trees/ha)		Percent of Count		Basal Area (m^2/ha)		Percent of Basal Area		Diameter (cm)	
			Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
RMZ	Reference	3	54.0	24.2	6.5	1.6	1.5	0.3	3.3	0.8	19.3	2.0
	FP Buffer	8	136.8	39.3	31.3	9.2	12.0	4.4	28.5	9.1	30.1	1.8
	<50ft Buffer	6	65.2	19.1	24.6	7.6	4.1	1.3	22.7	7.8	27.1	2.1
PIP	Reference	3	48.5	31.8	5.0	2.2	0.9	0.5	2.1	1.1	15.7	1.3
	FP Buffer	8	191.8	40.8	51.5	10.8	19.9	4.9	47.9	11.3	32.4	1.3

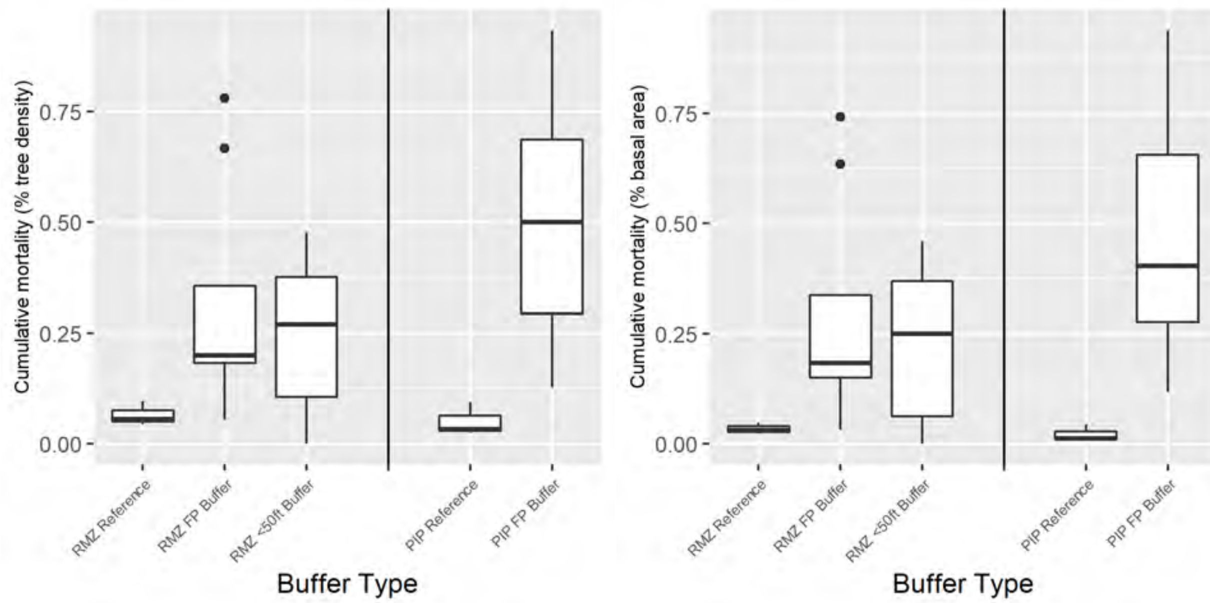


Figure 3-2. Post-harvest tree mortality as a percentage of live tree density (left panel) and basal area (right panel) over the three-year post-harvest interval. The horizontal line is the median, the box is the central quartile (or central 50% of the data), the whiskers are non-extreme values, and the points are outliers.

3-5.2. LARGE WOOD INPUT

There was little post-harvest large wood input in Reference RMZs: an average of 4.3 pieces and 0.34 m^3 of combined in- and over-channel volume per 100 m of channel. In contrast, the RMZ FP and <50ft Buffers received an average of 23 and 10 pieces/100 m and 2.3 and 0.7 m^3/100 m of large wood, respectively. Over 90% of recruited large wood volume came to rest over the bankfull channel (**Table 3-7**; **Figure 3-3**). The patterns were similar for the subset of pieces consisting of stems with attached rootwads (SWRW), which are larger, more stable and likely to persist in the channel (**Table 3-7**). The majority of recruited large wood pieces had SWRW; 60, 70, and 100% in the RMZ Reference, RMZ FP Buffer, and RMZ <50ft Buffer types, respectively.

Table 3-7. Large wood input (total pieces and stems with rootwads) in pieces and volume per 100 m of channel over the three-year post-harvest interval. Buffer-type means and standard errors (SE) are based on site means (numbers of sites vary by type).

Plot Type	Buffer Type	*n*	Count (pieces/100 m)		Volume (m^3/100 m)					
			Total		Total		In-Channel		Over-Channel	
			Mean	SE	Mean	SE	Mean	SE	Mean	SE
Total Large Wood										
RMZ	Reference	3	4.3	2.1	0.34	0.11	0.02	0.02	0.32	0.13
	FP Buffer	8	23.3	6.9	2.33	0.95	0.12	0.05	2.21	0.94
	<50ft Buffer	6	10.1	3.6	0.69	0.29	0.02	0.01	0.67	0.28
Stems with Rootwads (SWRW)										
RMZ	Reference	3	2.6	1.0	0.31	0.13	0.00	0.00	0.30	0.13
	FP Buffer	8	16.4	5.1	1.77	0.83	0.00	0.00	1.77	0.82
	<50ft Buffer	6	10.1	3.6	0.69	0.29	0.02	0.01	0.67	0.28

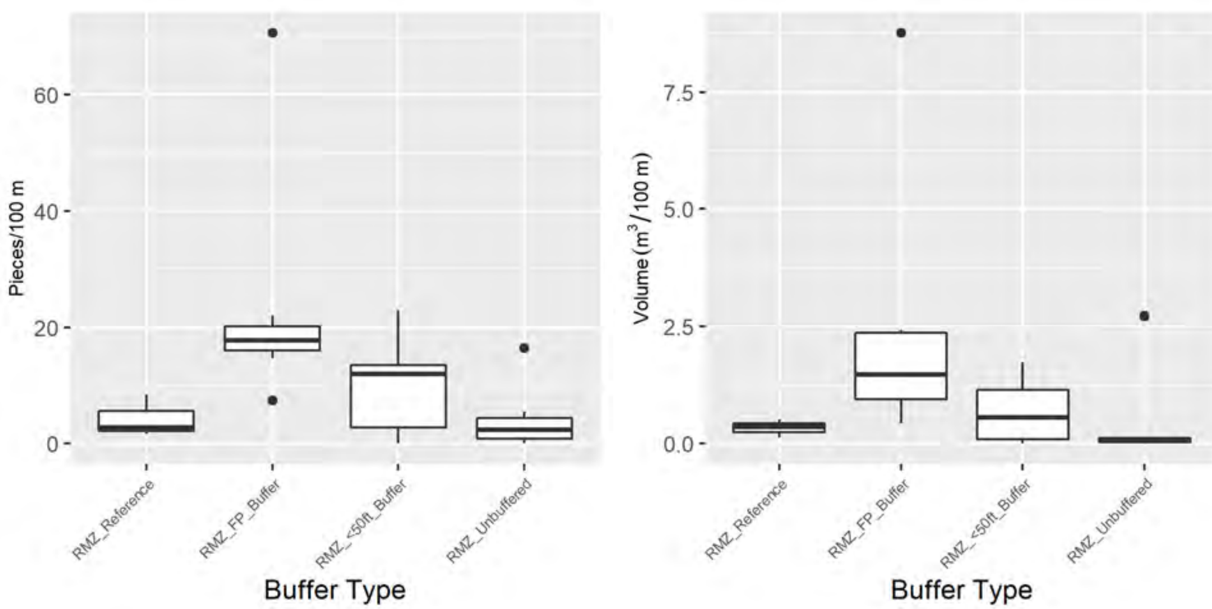


Figure 3-3. Total large wood input (pieces/100 m; left panel) and combined in- and over-channel volume (m^3/ 100 m; right panel) input over the three-year post-harvest interval. The horizontal line is the median, the box is the central quartile (or central 50% of the data), the whiskers are non-extreme values, and the points are outliers.

3-5.3. CHANNEL LARGE WOOD LOADING

Pre-harvest (Pre 1) channel large wood loading ranged from 55.8 to 111 pieces/100 m and from 9.8 to 25.2 m^3/100 m among buffer types (**Table 3-8**). Piece counts remained stable in the Reference RMZs through Post 3, increased in the RMZ FP Buffer and Unbuffered RMZs (8 and 13%, respectively), and decreased in the RMZ <50ft Buffers (15%). Piece counts increased 25% in the PIP FP Buffers and decreased 28% in the Reference PIPs. The patterns differed for volume, which decreased in the Reference RMZs, RMZ FP Buffers, and RMZ <50ft Buffers, increased in the Unbuffered RMZs and PIP FP Buffers, and remained stable in the Reference PIPs.

The proportion of functional pieces (i.e., contributing to sediment retention or formation of pools, steps, or debris jams) ranged from 36 to 53% pre-harvest (**Table 3-8**). The proportion of functional pieces were very similar between Pre 1 and Post 3 samples in the Reference RMZs and PIPs, but decreased by ~10% in the other buffer types.

Table 3-8. Channel large wood loading (piece counts and volume) before harvest (Pre 1) and three years post-harvest (Post 3). Buffer-type means and standard errors (SE) are based on site means (numbers of sites vary by type).

Plot Type	Buffer Type	*n*	Pieces/100 m Total		Volume (m^3/100 m) Total		In-Channel		Over-Channel		% Functional Total	
			Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Pre 1												
RMZ	Reference	3	84.6	8.8	22.3	4.8	12.8	2.8	9.5	2.8	48.5	0.8
	FP Buffer	8	103.3	13.1	25.2	5.6	14.0	4.0	11.2	3.2	51.5	4.3
	<50ft Buffer	6	111.0	15.0	24.7	4.6	13.0	4.2	11.7	3.6	43.0	6.7
	Unbuffered	7	65.1	13.1	16.6	6.6	9.8	5.2	6.8	3.6	40.0	9.5
PIP	Reference	3	65.6	20.4	9.8	4.0	6.6	1.9	3.2	2.1	53.3	10.3
	FP Buffer	8	55.8	13.6	12.7	4.3	9.2	3.9	3.5	0.9	35.5	9.4
Post 3												
RMZ	Reference	3	85.3	11.8	16.0	3.2	9.8	1.9	6.2	3.2	48.1	3.8
	FP Buffer	8	112.1	12.8	19.4	3.0	11.4	2.4	7.9	1.5	39.1	4.6
	<50ft Buffer	6	96.5	12.1	18.3	3.8	12.6	3.5	5.7	1.5	34.1	2.8
	Unbuffered	7	75.1	11.4	17.9	8.6	5.9	3.4	12.0	8.5	18.2	5.9
PIP	Reference	3	51.4	9.0	9.9	1.9	5.6	2.7	4.3	3.0	48.8	11.2
	FP Buffer	8	74.2	13.2	16.0	6.5	11.3	6.7	4.7	1.6	25.3	6.2

The proportion of large wood pieces in decay classes 1 and 2 (indicating recent recruitment) ranged from 16 to 27% in Pre 1 (**Table 3-9**). The greatest increase occurred in RMZ and PIP FP Buffers by Post 3, likely due to input from post-harvest tree fall.

Table 3-9. Percentage of channel large wood loading pieces in decay classes 1 and 2 before harvest (Pre 1) and three years post-harvest (Post 3). Buffer-type means and standard errors (SE) are based on site means (numbers of sites vary by type).

Plot Type	Buffer Type	*n*	Pre 1		Post 3	
			Mean	SE	Mean	SE
RMZ	Reference	3	21.0	3.6	18.7	2.7
	FP Buffer	8	21.2	2.7	30.7	5.1
	<50ft Buffer	6	25.3	5.6	24.0	5.0
	Unbuffered	7	18.0	6.4	20.5	7.1
PIP	Reference	3	15.7	8.8	9.8	4.9
	FP Buffer	8	27.2	12.3	34.0	6.6

3-5.4. CHANNEL SMALL WOOD

Post-harvest changes in the frequency of small wood pieces (1 to 10 cm in diameter) intersecting channel cross-sections in the RMZ varied by buffer type. Small wood counts in Reference RMZs were 6.8 pieces/bankfull width in Post 1 and increased slightly by Post 3. Post 1 counts were higher in the RMZ FP Buffer and <50ft Buffer (8.9 and 9.2 pieces/bankfull width) but decreased by Post 3. Small wood frequency was highest in Post 1 in the Unbuffered RMZs (13.0 pieces/bankfull width) but decreased nearly 50% by Post 3 (**Table 3-10**).

Table 3-10. Small wood pieces per bankfull width intersecting channel cross-sections before harvest (Post 1) and three years post-harvest (Post 3). Buffer-type means and standard errors (SE) are based on site means (numbers of sites vary by type).

Plot Type	Buffer Type	*n*	Post 1		Post 3	
			Mean	SE	Mean	SE
RMZ	Reference	3	6.8	1.3	7.1	1.8
	FP Buffer	8	8.9	1.1	8.1	0.8
	<50ft Buffer	6	9.2	1.8	7.7	2.1
	Unbuffered	7	13.0	2.7	6.7	1.7

3-5.5. WOOD COVER

One year after harvest the mean percentage of channel surface area covered by wood of all sizes ranged from 28% in the Reference RMZs to 34, 43, and 42% in the RMZ FP Buffer, RMZ <50ft Buffer, and Unbuffered RMZs. The percentage in Post 3 appeared unchanged in the Reference and Unbuffered RMZs, while increasing in the RMZ FP and <50ft Buffers (**Table 3-11**). The proportion of the plot that could not be surveyed due to burial by wood varied by buffer type, ranging from ≤1.5% in Reference RMZs and RMZ FP Buffers to <3% in RMZ <50ft Buffers and 20% in Unbuffered RMZs.

Table 3-11. Percentage of channel surface area covered by wood of all sizes before harvest (Post 1) and three years post-harvest (Post 3). Buffer-type means and standard errors (SE) are based on site means (numbers of sites vary by type).

Plot Type	Buffer Type	*n*	Post 1		Post 3	
			Mean	SE	Mean	SE
RMZ	Reference	3	27.6	7.5	28.0	5.7
	FP Buffer	8	33.7	2.2	39.0	3.7
	<50ft Buffer	6	43.2	4.1	45.4	6.0
	Unbuffered	7	42.0	9.7	41.8	9.0

3-6. DISCUSSION

The current study is the third conducted by CMER examining the efficacy of riparian prescriptions for Type Np streams in western Washington. Two features differentiate it from the BCIF (Schuett-Hames *et al*. 2012; Schuett-Hames and Stewart 2019) and Hard Rock studies (McIntyre *et al*. 2018). First, it focused on sites with incompetent (sedimentary) lithology, providing a basis for comparison with the Hard Rock Study, which focused on competent (basalt) lithology. Second, unstable slopes bordered significant portions of the stream channel available for unbuffered harvest within the RMZ, resulting in a substantial number of buffers narrower than the standard 15 m wide FP buffer. This was not the case in previous studies, nor was it anticipated during planning for the current study.

Inability to conduct a statistical analysis to detect potential treatment effects reduces our confidence in interpreting the results of the current study. Obstacles to a robust statistical analysis include: 1) low power to detect treatment effects due to the small number (three) of reference sites; 2) potential bias due to unbalanced distribution of buffer types among treatment sites and non-randomized assignment of treatments; and 3) unanticipated variation in harvest timing over a three-year period which complicated treatment–reference comparisons. However our confidence in interpreting the results of the current study increased because the post-harvest differences between the treatments and references were large and the responses to the Type Np riparian prescriptions were similar in direction and magnitude to those observed in the previous two studies.

The patterns of change in stand structure were similar to early patterns of change in the BCIF and Hard Rock studies. The primary cause of structural change in all three studies was removal of trees from the Unbuffered RMZs, and wind-related mortality of RMZ and PIP buffer trees following harvest. Consequently, changes at treatment sites were considerably greater than at reference sites. The greatest change in stand structure occurred in the Unbuffered RMZs where nearly all trees were removed. Density and basal area decreased in the RMZ and PIP FP Buffers in all three studies. Density declined on average by 17 to 30% and basal area by 19 to 26% in RMZ FP Buffers. Both declined to an even greater degree in PIP FP Buffers (by an average of 33 to 52% and 28 to 46%, respectively). Post-harvest declines in this study were at the upper end of these ranges, but only slightly higher than those in the Hard Rock Study. In contrast, RMZ and PIP reference stands were relatively stable in the years following harvest with densities declining by 2 to 12%, while basal area increased in some instances (range -7 to +7%).

Since no trees were removed from the RMZ FP and PIP FP Buffers during harvest of adjacent uplands, changes in stand structure were due to post-harvest mortality. Compared to the reference sites, cumulative rates of mortality ranged from two to five times higher in RMZ FP Buffers and seven to ten times higher in PIP FP Buffers in the three studies (**Table 3-12)**. Mean values in the current study were most similar to those in the Hard Rock Study. Post-harvest mortality was highly variable among sites in all three studies.

Table 3-12. Post-harvest tree mortality as a percentage of stems among CMER studies of FP Buffers on Type Np streams in western Washington.

Study	Buffer Type	*n*	Mortality (% density)	
			Mean	Range Among Sites
RMZs				
Soft Rock Study	FP Buffer	8	31.3	5-78
BCIF Study	FP Buffer	13	20	1–69
Hard Rock Study	FP Buffer	3	30	7–52
PIPs				
Soft Rock Study	FP Buffer	8	51.5	13-94
BCIF Study	FP Buffer	3	35	11–63
Hard Rock Study	FP Buffer	3	48	14–74

Wind was the dominant cause of mortality in the RMZ and PIP FP Buffers in this study, accounting for >75% of trees that died. Storm-force winds (55 to 74 mph) occurred on a total of 31 days during the study period (based on data from a nearby weather station in Astoria, Oregon), including at least two days every winter (**Appendix Table 3-3**). Wind was the dominant mortality agent for buffer trees in all three studies, accounting for 74 to 90% of stems that died in the RMZ FP Buffers and 67 to 95% in the PIP FP Buffers. In contrast, wind accounted for <16% of mortality in the reference sites in this and the BCIF Study. The pattern of elevated, but variable, buffer-tree mortality due to wind has been documented in numerous studies, including buffers of various configurations across a wide range of streams in many parts of the Pacific Northwest and beyond (Grizzel and Wolff 1988; Grizzel *et al*. 2000; Ruel *et al*. 2001; Liquori 2006; Jackson *et al*. 2007; Martin and Grotefendt 2007; Bahuguna *et al*. 2010; Urgenson *et al.* 2013; Beese *et al.* 2019). Variation in wind damage in buffers has been related to differences in topographic position, buffer size and orientation to prevailing winds, and exposure to wind due to the size of adjacent cleared areas (Ruel *et al*. 2001; Beese *et al.* 2019). The small number of sites and their limited geographic distribution precluded evaluation of factors contributing to variation in wind damage, however analysis of combined data from this and other studies of Forest Practices buffers in western Washington could provide useful insights into relationships between buffer tree mortality and factors such as stand composition and structure, site conditions such as topography and aspect, and regional factors such as proximity to the coast.

In all three studies, post-harvest input of large wood into streams was greater in RMZ and PIP FP Buffers than in the reference sites. Uprooting of trees due to windthrow produced many stems with attached rootwads. In the absence of debris flows these should persist for long periods of time. However, most pieces came to rest suspended above the bankfull channel, where they will provide shade and cover, but not in-channel functions (e.g., sediment retention or formation of pools, steps, or debris jams). Unbuffered RMZs received variable input of wood during logging, but there was no post-harvest LW input because all trees were removed.

Prior to harvest, channel large wood loading varied widely among sites in the current study. For example, large wood frequency averaged 61 to 153 pieces/100 m among sites (**Appendix Table 3-2**). This was comparable to that in the Hard Rock Study but is somewhat higher than that observed by Bilby and Ward (1991) for small streams in old- or second-growth stands in western Washington. However, channels in the latter study were larger and large wood frequency varied inversely with channel width. Large wood frequency increased by Post 3 in the RMZ and PIP FP Buffers, similar to the Hard Rock Study. Many pieces in Post 3 had little decay, indicating post-harvest input of fallen buffer trees. In contrast, large wood frequency declined or remained stable in the reference sites.

In the Unbuffered RMZs, LW piece counts increased over pre-harvest levels, but volume decreased, indicating an input of smaller-sized pieces during harvest. Although small wood pieces were initially abundant in the Unbuffered RMZs after harvest, small wood began decreasing at year 3 post-harvest, indicating that wood input as logging debris was not persistent in the channel during the post-harvest period. A similar pattern was observed in Unbuffered RMZs in the Hard Rock Study, although the decline occurred after year 5 post-harvest. We believe that our estimates of large and small wood in Unbuffered RMZs are conservative, because portions of the channel that were buried under debris could not be surveyed. Also, there was inter-annual variability in channels with disturbance from adjacent harvest due to changes in channel morphology and invasion of vegetation following harvest.

The presence of unstable slopes resulted in buffering of substantial portions of the RMZ adjacent to Type Np streams that would otherwise have been available for harvest to the stream edge under the riparian prescriptions. This was infrequent in the previous CMER study in competent lithology. Unstable slopes were often confined to steep areas adjacent to the stream, resulting in narrow buffers that did not extend to the full width of the RMZ (RMZ <50ft Buffers) because trees were removed from the portions of the RMZ not designated as unstable slopes. Direct comparison of mortality rates and LW input in RMZ FP and <50ft Buffers was complicated by the fact that RMZ <50ft Buffers were not present at two sites where post-harvest mortality was greatest in the RMZ FP Buffers. Comparing only sites where both RMZ FP and <50ft Buffers were present, mortality was greater in the RMZ <50ft Buffers than in the RMZ FP Buffers (24.6 vs. 17.7% of stems, respectively), however the volume of LW input was only about half because harvest reduced the number of standing trees available for post-harvest recruitment in the RMZ <50ft Buffers. Since the unstable slope buffers reduced the length of RMZ available for clearcut harvest adjacent to the stream, more trees were retained providing greater opportunity for future LW recruitment than had those areas been harvested.

The variability in stand conditions created by these prescriptions has important implications for future stand development and wood input from riparian areas. As stands develop over time, composition, structure, and associated wood input regimes will be determined by the initial differences in post-harvest structure, silvicultural management (e.g., planting and thinning) in Unbuffered RMZs, and mortality and regeneration in unharvested stands (Gomi *et al.* 2006).

In Unbuffered RMZs managed for timber production, the reforestation requirements of the Forest Practices rules (WFPB 2012) produce dense stands of conifers that quickly achieve canopy closure and are typically thinned to accelerate growth. Initial input of smaller wood (e.g., tops, branches, and broken stems) during harvest (Jackson *et al.* 2001) is followed by a long period with little additional input until the next harvest (Beechie *et al.* 2000). This management regime results in decreased LW abundance over time, since the young trees are harvested before natural mortality processes have an opportunity to produce significant inputs of large wood (Beechie *et al.* 2000; Bragg 2000; Gomi *et al.* 2006).

The frequency and magnitude of post-harvest mortality exerts a strong influence on stand development (Lutz and Halpern 2006). In the absence of severe disturbance, mortality from suppression and mechanical damage in the unharvested buffers will primarily affect smaller trees as the stands pass through the stem exclusion phase of development (Oliver 1980). Such stands are expected to continue to develop into mature, single cohort stands, providing a stable source of LW input over time from fine-scale mortality processes affecting individual or small groups of trees. In other locations, severe disturbance events such as high winds, fire, or disease (Harcombe *et al.* 2004; Edmonds *et al.* 2005) may kill many trees, including larger co-dominant trees. This produces an immediate pulse of LW input, with additional input as dead trees fall over time (Bragg 2000; Reilly and Spies 2016). These full or partial stand-replacement events create openings where regenerating conifers compete with hardwoods and shrubs for dominance. Successful conifer regeneration will produce a two-cohort conifer stand with an overstory of remnant large conifers from the original stand and an understory of young conifers, typically dominated by western hemlock (Franklin *et al.* 2002). Where conifer regeneration is unsuccessful, stand development will take an alternative pathway, with scattered remnant conifers surrounded by an understory of shrubs or hardwoods (Donato *et al.* 2012).

The combined effect of the complex FPHCP prescriptions for western Washington Type Np streams and the spatial variability in post-harvest mortality from wind is creating considerable diversity in riparian forest structure and wood input regimes across the landscape. A similar mosaic of post-harvest stand structure was observed in the BCIF and Hard Rock studies. This is a marked change from the more homogenous stand structures produced by widespread clearcut harvest of the riparian stands adjacent to headwater streams in the past. As stands in unharvested buffers mature, they should provide refugia of mature forest habitat and complex aquatic habitat across the managed forest landscape.

In conclusion, the results of this study are strikingly similar to those of the Hard Rock and BCIF studies. Aside from the greater prevalence of unstable slope buffers, we did not observe obvious differences attributable to incompetent lithology. Consistency across all three studies in the direction and magnitude of change in the stand structure, tree mortality, and wood recruitment in response to the prescriptions increases confidence in our assessment of the effects of the Type Np prescriptions on stand structure and wood input. In addition to reinforcing the conclusions of the previous studies, the current study substantially broadens the geographic and geomorphic scope of where the results apply.

3-7. REFERENCES

Bahuguna, D., S.J. Mitchell, and Y. Miquelajauregui. 2010. Windthrow and recruitment of large woody debris in riparian stands. *Forest Ecology and Management* 259(10):2048-2055.

Beechie, T.J., G. Pess, P. Kennard, R.E. Bilby, and S. Bolton. 2000. Modeling recovery rates and pathways for woody debris recruitment in northwestern Washington streams. *North American Journal of Fisheries Management* 20(2):436-452.

Beese, W., T. Rollerson, and C. Peters. 2019. Quantifying wind damage associated with variable retention harvesting in coastal British Columbia. *Forest Ecology and Management* 443:117-131.

Bilby, W.E., and J.W. Ward. 1991. Characteristics and function of large woody debris in streams draining old-growth, clear-cut, and second-growth forests in southwestern Washington. *Canadian Journal of Fisheries and Aquatic Sciences* 48: 2499-2508.

Boyer, K.L., D.R. Berg, and S.V. Gregory. 2003. Riparian management for wood in rivers. Pages 407-420 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell (eds.). *The Ecology and Management of Wood in World Rivers*. American Fisheries Society, Symposium 37, Bethesda, MD.

Bragg, D.C. 2000. Simulating catastrophic and individualistic large woody debris recruitment for a small riparian system. *Ecology* 81(5):1383-1394.

Curtis, R.O. 1982. A simple index of stand density for Douglas-fir. *Forest Science* 28(1):92-94.

Curtis, R.O., and D.D. Marshall. 2000. Why quadratic mean diameter? *Western Journal of Applied Forestry* 15(3):137-139.

DeWalle, D.R. 2010. Modeling stream shade: Riparian buffer height and density as important as buffer width. *Journal of the American Water Resources Association* 46:323-333.

Donato, D.C., J.L. Campbell, and J.F. Franklin. 2012. Multiple successional pathways and precocity in forest development: Can some forests be born complex? *Journal of Vegetation Science* 23:576-584.

Edmonds, R.L., J.K. Agee, and R.I. Gara. 2005. *Forest Health and Protection*. Waveland Press, Inc., Long Grove, IL.

Fox, M., and S. Bolton. 2007. A regional and geomorphic reference for quantities and volumes of instream wood in unmanaged forested basins of Washington State. *North American Journal of Fisheries Management* 27:342-359.

Franklin, J.F., T.A. Spies, R. Van Pelt, A.B. Carey, D.A. Thornburgh, D.R. Berg, D.B. Lindenmayer, M.E. Harmon, W.S. Keeton, D.C. Shaw, K. Bible, and J. Chen. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. *Forest Ecology and Management* 155:399-423.

Gomi, T., A.C. Johnson, R.L. Deal, P.E. Hennon, E.H. Orlikowska, and M.S. Wipfli. 2006. Factors affecting distribution of wood, detritus, and sediment in headwater streams draining managed young-growth red alder-conifer forests in southeast Alaska. *Canadian Journal of Forest Research* 36:725-737.

Gomi, T., R.C. Sidle, M.D. Bryant, and R.D. Woodsmith. 2001. The characteristics of woody debris and sediment distribution in headwater streams, southeastern Alaska. *Canadian Journal of Forest Research* 31:1386-1399.

Grady, J. 2001. *Effects of Buffer Width on Organic Matter Input to Headwater Streams in the Western Cascade Mountains of Washington State*. M.S. thesis. University of Washington. Seattle, WA. 46 p.

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones: Focus on links between land and water. *Bioscience* 41:540-551.

Grizzel, J.D., M. McGowan, D. Smith, and T. Beechie. 2000. *Streamside Buffers and Large Woody Debris Recruitment: Evaluating the Effectiveness of Watershed Analysis Prescriptions in the North Cascades Region.* Timber Fish and Wildlife publication TFW-MAG1-00-003. Washington Department of Natural Resources, Forest Practices Division, Olympia, WA.

Grizzel, J.D., and N. Wolff. 1998. Occurrence of windthrow in forest buffer strips and its effect on small streams in northwest Washington. *Northwest Science* 72(3):214-223.

Harcombe, P.A., S.E. Greene, M.G. Kramer, S.A. Acker, T.A. Spies, and T. Valentine. 2004. The influence of fire and windthrow dynamics on a coastal spruce-hemlock forest in Oregon, USA, based on aerial photographs spanning 40 years. *Forest Ecology and Management* 194:71-82.

Jackson, C.R., D.P. Batzer, S.S. Cross, S.M. Haggerty, and C.A. Sturm. 2007. Headwater streams and timber harvest: Channel, macroinvertebrate, and amphibian response and recovery. *Forest Science* 53:356-370.

Jackson, C.R., C.A. Sturm, and J.M. Ward. 2001. Timber harvest impacts on small headwater stream channels in the coast ranges of Washington. *Journal of the American Water Resources Association* 37:1533-1549.

Janisch, J.E., S.M. Wondzell, and W.J. Ehinger. 2012. Headwater stream temperature: Interpreting response after logging, with and without riparian buffers, Washington, USA. *Forest Ecology and Management* 270:302-313.

Kibler, K.M., A. Skaugset, L.M. Ganio, and M.M. Huso. 2013. Effect of contemporary forest harvesting practices on headwater stream temperatures: Initial response of the Hinkle Creek catchment, Pacific Northwest, USA. *Forest Ecology and Management* 310:680-691.

Liquori, M.K. 2006. Post-harvest riparian buffer response: Implications for wood recruitment modeling and buffer design. *Journal of the American Water Resources Association.* 42(1):177-189.

Litschert, S.E., and L.H. MacDonald. 2009. Frequency and characteristics of sediment delivery pathways from forest harvest units to streams. *Forest Ecology and Management* 259:143-150.

Lutz, J.A., and C.B. Halpern. 2006. Tree mortality during early forest development: A long-term study of rates, causes, and consequences. *Ecological Monographs* 76(2):257-275.

MacDonald, L.H., and D. Coe. 2007. Influence of headwater streams on downstream reaches in forested areas. *Forest Science* 53:148-168.

Marquardt, T., H. Temesgen, and P. Anderson. 2010. Accuracy and suitability of selected sampling methods within conifer dominated riparian zones. *Forest Ecology and Management* 260:313-320.

Martin, D.J., and R.A. Grotefendt. 2007. Stand mortality in buffer strips and the supply of woody debris to streams in southeast Alaska. *Canadian Journal of Forest Research* 37(1):36-49.

Martin, D.J., and A. Shelly. 2017. Temporal trends in stream habitat on managed forestlands in coastal Southeast Alaska. *North American Journal of Fisheries Management* 37:882-902.

May, C.L., and R.E. Gresswell. 2003. Large wood recruitment and redistribution in headwater streams in the southern Oregon Coast Range, U.S.A. *Canadian Journal of Forest Research* 33:1352-1362.

McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn (technical coordinators). 2018. *Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington*. Cooperative Monitoring, Evaluation, and Research Report, CMER 18-100. Washington Department of Natural Resources, Olympia, WA. 890 p.

Meleason, M.A., S.V. Gregory, and J.P. Bolte. 2003. Implications of riparian management strategies on wood in streams of the Pacific Northwest. *Ecological Applications* 13:1212-1221.

Montgomery, D.R., and J.M. Buffington. 1993. *Channel Classification, Prediction of Channel Response, and Assessment of Channel Condition*. Timber Fish and Wildlife publication TFW-SH10-93-002. Washington Department of Natural Resources, Olympia, WA.

Montgomery, D.R., B.D. Collins, J.M. Buffington, and T.B. Abbe. 2003. Geomorphic effects of wood in rivers. Pages 21-48 *in* S.V. Gregory, K.L. Boyer, and A.M. Gurnell (eds.). *The Ecology and Management of Wood in World Rivers*. American Fisheries Society, Symposium 37, Bethesda, MD.

O'Connor, M.D., and R.D. Harr. 1994. *Bedload Transport and Large Organic Debris in Steep Mountain Streams in Forested Watersheds on the Olympic Peninsula, Washington*. Timber Fish and Wildlife publication TFW-SH7-94-001. Washington Department of Natural Resources, Olympia, WA.

Oliver, C.D. 1980. Forest development in North America following major disturbances. *Forest Ecology and Management* 3:153-168.

Pleus, A.E., and D.E. Schuett-Hames. 1998. *TFW Monitoring Program Methods Manual for the Reference Point Survey*. Timber Fish and Wildlife publication TFW-AM9-98-002. Washington Department of Natural Resources. Olympia, WA.

R Development Core Team. 2012. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Reilly, M.J., and T.A. Spies. 2016. Disturbance, tree mortality, and implications for contemporary regional forest change in the Pacific Northwest. *Forest Ecology and Management* 374:102-110.

Richardson, J.S., R.E. Bilby, and C.A. Bondar. 2005. Organic matter dynamics in small streams of the Pacific Northwest. *Journal of the American Water Resources Association* 41:921-934.

Richardson, J.S., and R.J. Danehy. 2007. A synthesis of the ecology of headwater streams and their riparian zones in temperate forests. *Forest Science* 53:131-147.

Robison, G.E., and R.L. Beschta. 1990. Characteristics of course woody debris for several coastal streams of Southeast Alaska, USA. *Canadian Journal of Fisheries and Aquatic Sciences* 47:1684-1693.

Ruel, J.-C., D. Pin, and K. Cooper. 2001. Windthrow in riparian buffer strips: Effect of wind exposure, thinning, and strip width. *Forest Ecology and Management* 143(1-3):105-113.

SAS Institute Inc. 2017. Using JMP® 13. SAS Institute Inc. Cary, NC.

Schuett-Hames, D.E., A. Roorbach, and R. Conrad. 2012. *Results of the Westside Type N Buffer Characteristics, Integrity, and Function Study, Final Report*. Cooperative Monitoring, Evaluation, and Research Report, CMER 12-1201. Washington Department of Natural Resources, Olympia, WA.

Schuett-Hames, D.E., and G. Stewart. 2019. *Changes in Stand Structure, Buffer Tree Mortality and Riparian-associated Function 10 Years after Timber Harvest adjacent to Non-fish-bearing Perennial Streams in Western Washington*. Cooperative Monitoring, Evaluation, and Research Report. Washington Department of Natural Resources, Olympia, WA

Stewart, G.H. 1986. Forest development in canopy openings in old-growth *Pseudotsuga* forests of the western Cascade Range, Oregon. *Canadian Journal of Forest Research* 16:558-568.

Urgenson, L.S., C.B. Halpern, and P.D. Anderson. 2013. Level and pattern of overstory retention influence rates and forms of tree mortality in mature, coniferous forests of the Pacific Northwest, USA. *Forest Ecology and Management* 308:116-127

WDNR. 1996. *Field Procedures, Forest Resource Inventory System*. Washington Department of Natural Resources, Olympia, WA.

WDNR. 2005. *Final Forest Practices Habitat Conservation Plan*. Washington Department of Natural Resources, Olympia, WA.

WFPB. 2012. *Washington Forest Practices: Rules, Board Manual, and Act*. Washington Department of Natural Resources, Olympia, WA.

Wilkerson, E., J.M. Hagan, D. Siegel, and A.A. Whitman. 2006. The effectiveness of different buffer widths for protecting headwater stream temperature in Maine. *Forest Science* 52:221-231.

3-8. APPENDIX TABLES

Appendix Table 3-1. Reference site means (*n* = 3) for the three harvest-timing groups (see **Table 3-1**) with simple mean of the three harvest-timing groups and mean weighted by number of treatment sites in each group. Den = live tree density; BA = basal area; QMD = quadratic mean diameter; RD = relative density; LW = large wood; SW = small wood; CHAN = combined in- and over- channel; IC = in-channel; OC = over-channel; SWRW = stems with rootwad; BFW = bankfull width.

Metric	Harvest-Timing Group (number of treatment sites)			Group Means	
	Group1 (1)	Group 2 (5)	Group 3 (2)	Simple	Weighted
RMZs					
Δ live Den (trees/ha)	-23.9	-48.9	-63.1	-45.3	-49.3
Δ live BA (m^3/ha)	3.3	2.5	2.2	2.6	2.5
Δ QMD	1.5	1.6	1.9	1.7	1.7
Δ RD	2.7	1.5	1.0	1.7	1.5
Mortality (trees/ha)	32.6	56.7	58.0	49.1	54.0
Mortality %Count	4.0%	6.6%	7.5%	6.0%	6.5%
Mortality BA (m^3/ha)	0.8	1.6	1.5	1.3	1.5
Mortality %BA	1.7%	3.6%	3.6%	3.0%	3.3%
LW recruit pieces/100m	3.2	4.8	3.4	3.8	4.3
LW recruit CHAN volume/100m	0.3	0.4	0.2	0.3	0.3
LW recruit IC volume/100m	0.0	0.0	0.0	0.0	0.0
LW recruit OC volume/100m	0.3	0.4	0.2	0.3	0.3
LW recruit SWRW pieces/100m	1.5	2.4	3.4	2.4	2.6
LW recruit SWRW CHAN volume/100m	0.2	0.4	0.2	0.3	0.3
LW recruit SWRW IC volume/100m	0.0	0.0	0.0	0.0	0.0
LW recruit SWRW OC volume/100m	0.2	0.4	0.2	0.3	0.3
LW loading pieces/100m- Pre1	101.0	82.2	82.2	88.5	84.6
LW loading total volume/100m- Pre1	24.3	22.1	22.1	22.8	22.3
LW loading IC volume/100m- Pre1	14.0	12.7	12.7	13.1	12.8
LW recruit OC volume/100m- Pre1	10.4	9.4	9.4	9.7	9.5
LW loading pieces/100m- Post3	88.4	82.2	91.3	87.3	85.3
LW loading total volume/100m- Post3	18.0	16.4	14.2	16.2	16.0
LW loading IC volume/100m- Post3	11.3	10.6	7.2	9.7	9.8
LW loading OC volume/100m- Post3	6.7	5.8	7.0	6.5	6.2
SW pieces/BFW- Post1	8.5	7.0	5.6	7.0	6.9
SW pieces/BFW- Post3	5.6	8.4	6.0	6.7	7.5
PIPs					
Δ live Den (trees/ha)	-37.6	-40.7	-52.8	-43.7	-43.3
Δ live BA (m^3/ha)	3.0	3.1	2.8	3.0	3.0
Δ QMD	1.4	1.6	1.7	1.6	1.6
Δ RD	2.4	2.3	1.8	2.2	2.2
Mortality (trees/ha)	50.4	49.8	44.3	48.1	48.5
Mortality %Count	4.6%	5.0%	5.1%	4.9%	5.0%
Mortality BA (m^3/ha)	1.0	0.9	0.9	0.9	0.9
Mortality %BA	2.1%	2.0%	2.2%	2.1%	2.1%

Appendix Table 3-2. Pre-harvest stand structure and wood loading by site.

Site	Stand Structure				Large Wood Loading	
	Density		Basal Area		Frequency	Volume
	Trees/ha	% Conifer	m^3/ha	% Conifer	Pieces/100 m	m^3/100 m
REF1	1073.9	81.9%	43.9	81.0%	102.0	25.1
REF2	517.1	67.6%	41.7	69.7%	77.4	28.8
REF3	601.0	85.2%	39.6	87.2%	74.3	13.1
TRT1a	444.6	81.3%	48.0	88.6%	60.5	5.2
TRT1b	409.9	83.3%	46.0	87.5%	82.0	7.1
TRT2	648.4	78.5%	45.7	85.3%	92.5	28.2
TRT3	510.1	92.2%	38.2	98.7%	117.9	33.2
TRT4	423.5	74.7%	49.6	91.1%	79.3	26.0
TRT5	422.1	81.2%	42.3	85.3%	80.3	16.2
TRT6	446.2	98.8%	37.9	98.8%	95.0	20.5
TRT7	514.3	92.2%	43.3	92.2%	152.8	25.1

Appendix Table 3-3. Number of days with windspeed ≥ storm-force (55 mph) by hydrological year.

Year	Days (*n*)
2013	3
2014	9
2015	5
2016	4
2017	8
2018	2
Total	31

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CHAPTER 4 – STREAM TEMPERATURE AND COVER

William Ehinger and Welles Bretherton

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

4-1. ABSTRACT

We used a Before-After Control-Impact study design to estimate the changes in canopy closure and stream temperature after timber harvest in non-fish-bearing headwater streams on marine sedimentary lithologies in western Washington. Each site was an entire non-fish-bearing stream basin. The study included three no-harvest reference sites and seven sites harvested under the current forest practices rules, i.e., a clearcut harvest leaving a 50-ft (15.2-m) wide buffer along each side of the perennial stream for at least 50% of the stream length. Harvested sites were also given unstable slope buffers resulting in buffers along 52% to 100% of the stream. We monitored from summer 2012 to summer 2017. Harvest occurred from October 2013 to August 2015.

Riparian shade decreased post-harvest relative to unharvested reference sites. Mean canopy closure decreased in the harvested sites from 97% to 75%, 68%, and 69% in Post 1, Post 2, and Post 3 years, respectively, while mean canopy closure in the unharvested reference sites was never less than 96%. Declines in canopy closure were related to the proportion of stream buffered and to post-harvest windthrow within the buffer, which tended to be more severe in sites with a lower proportion of the stream buffered.

The current Forest Practices rules were not effective at preventing increases in summer water temperature. The mean increases in the seven-day temperature response at the junction with the fish-bearing stream were 0.6°C, 0.6°C, and 0.3°C in Post 1, Post 2, and Post 3, respectively. The temperature response varied among the streams and was most closely related to the loss of canopy closure. There was evidence that hyporheic exchange and stream discharge, in combination with high canopy closure, were factors at some individual sites.

In a comparison of the temperature responses in this study with results from a similar study in basalt lithology (Hard Rock Study), we found no evidence of a fundamental difference in how stream temperature responded to the loss of canopy closure. However, the data indicated that temperature response at the unbuffered Hard Rock Study sites was greater than at the buffered sites even when accounting for the proportion of stream with surface flow and length of stream with surface flow. An examination of temperature response by site aspect also suggested that canopy closure was the primary driver of temperature change.

Spring and fall temperatures were elevated at nearly all locations in all harvested sites, with the mean monthly temperature response sometimes exceeding 1.0°C even in sites more than 90% buffered.

4-2. INTRODUCTION

Non-fish-bearing “headwater” (Type N) streams constitute more than 65% of the total stream length on industrial forestlands in western Washington (Rogers and Cooke 2007). These streams serve as habitat for non-fish species and provide important subsidies of organic matter and macroinvertebrates (Wipfli *et al.* 2007), nutrients (Alexander *et al.* 2007), and cool water to downstream fish-bearing reaches. Stream temperature is an important determinant in many biological processes that may affect these subsidies and the growth and survival of aquatic biota

(Wehrly *et al.* 2007; Friberg *et al.* 2013), many of which have narrow thermal tolerances for specific life stages (Richter and Kolmes 2005).

Stream temperature is a function of the water temperature entering the reach and energy exchanges between the stream and its surroundings (see Moore *et al.* 2005b). Radiative exchanges include direct and diffuse 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 was often found to be 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, by conduction of heat from the water to the stream bed, or via flow into bed sediments. Estimates of latent and sensible heat exchange in forested environments are typically less than 10% of net radiation (Brown 1969; Johnson 2004; Moore *et al.* 2005a), while estimates of bed heat exchange range from 10% of net radiation for a step-pool stream (Moore *et al.* 2005a) to 25% in a bedrock channel (Brown 1969). Ground water inflow in summer is usually cooler than stream water and can moderate diurnal and seasonal 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.* 2005a).

Early studies of the direct effects of forest harvest on stream temperature documented large decreases in shade and increases in 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 (2005b) and, more recently, Groom and colleagues (2011), Janisch and colleagues (2012), Kibler and colleagues (2013), Bladon and colleagues (2018), and McIntyre and colleagues (2018) reported more modest temperature increases from studies of riparian buffers following contemporary forest practices in the Pacific Northwest. These studies were consistent in finding a measurable increase in summer stream temperature following timber harvest leaving no buffers or with partially buffered streams. They suggest that much of the variability among studies is likely due to differences in buffer width and length, forest management within the buffer, length of stream within the harvest unit, proportion of stream buffered, and underlying lithology. Moore and colleagues (2005a) noted other site-specific factors that play a role. For example, studies have shown that stream width and depth, flow velocity and volume, length of surface flow (Janisch *et al.* 2012), subsurface hydrology (Story *et al.* 2003), upstream hydrology (Gomi *et al.* 2006), site aspect and elevation (Beschta *et al.* 1987; Isaak and Hubert 2001; Poole and Berman 2001; Moore *et al.* 2005a), geologic setting (Janisch *et al.* 2012; Bladon *et al.* 2018), stream substrate size (Johnson and Jones 2000; Johnson 2004; Janisch *et al.* 2012), and distance downstream from a disturbance (Cole and Newton 2013) influence stream temperature response. However, few studies have monitored stream temperature for the 7 to 15 years needed for stream temperature to return to pre-harvest levels (Moore *et al.* 2005a).

Here, we report the effects of clearcut forest harvest following the current Washington State Forest Practices rules for non-fish-bearing perennial streams on stream temperature and cover. The results of this study may be used to determine the effectiveness of current forest practices rules with respect to Washington state water quality criteria. This study was conducted on sites in marine sedimentary lithologies in southwestern Washington State and complements a similar study conducted on more resistant basalt lithologies in western Washington (McIntyre *et al.* 2018; McIntyre *et al*. 2021).

4-3. METHODS

4-3.1. CANOPY CLOSURE

We used a spherical densiometer (Lemmon 1956) to measure canopy closure at 10 to 25 equally-spaced (40-m minimum) locations along the entire Type Np stream network (see **Appendix B**). Four measurements were made from mid-channel (facing upstream, downstream, right bank, and left bank) at each location and averaged (Werner 2009).

Analyses evaluated the generalized null hypothesis:

$$\Delta S_{REF} = \Delta S_{TRT} \tag{4-1}$$

where: ΔS_{REF} is the change (post-harvest minus pre-harvest) in shade (as characterized by canopy closure) in the reference sites, and

ΔS_{TRT} is the post-harvest change in shade in the treatment sites.

We used generalized linear mixed-effects models (GLMM) that incorporated both fixed and random effects for hypothesis testing. GLMM can be used to fit data that derive from non-normal distributions with monotonic link transformations. An added benefit is that mixed models can accommodate missing data as long as those data are missing at random. In matrix form, this model can be represented as:

$$Y = X\beta + Z\gamma + \epsilon \tag{4-2}$$

where: X is a vector of fixed effects,

β is a vector of unknown fixed-effects parameters,

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

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

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

Values from each site were averaged by year and analyzed using a GLMM with the GLIMMIX procedure in SAS 9.4 (SAS Institute 2013). The fixed effects were treatment (reference or harvested), period (pre-harvest, post-harvest 1, post-harvest 2, or post-harvest 3), and treatment x period interaction. Site was included as a random effect allowing the intercept to differ among sites. We estimated model parameters using residual maximum pseudo likelihood (Method = RMPL) using the beta distribution and a logit link function, as is often necessary for proportion data. 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. We checked for overdispersion by ensuring that the Chi-Square/degrees of freedom was approximately equal to one. The SAS code is included in **Table 4-1**.

Pairwise comparisons were used to estimate the effect size for the buffer treatment relative to the reference treatment in each post-harvest year where:

$$\text{Effect size} = (\mathrm{TRT}_{\mathrm{Post}\ j} - \mathrm{TRT}_{\mathrm{Pre}}) - (\mathrm{REF}_{\mathrm{Post}\ j} - \mathrm{REF}_{\mathrm{Pre}}) \qquad (4\text{-}3)$$

where: REF = reference treatment
TRT = buffer treatment
Pre = pre-harvest
Post = post-harvest
j = year post-harvest

Although the analyses were done using the Distribution = Beta and Link = logit, the effect sizes are presented in tables as percentages to better relate to the measured shade values. These were calculated using **Equation 4-3** and the least squares means transformed from Beta space to percentages using the ilink option within GLIMMIX (**Table 4-1**). We did not adjust the P-values for multiple comparisons but focused on the overall pattern of riparian cover reduction and recovery in the post-harvest years.

Table 4-1. SAS code used in mixed effects models. TRMT = treatment (reference or harvested), TRYR = treatment year (Pre, Post 1, Post 2, Post 3). CC = canopy closure, PropWet = proportion of channel with surface water, 7DTR = seven-day average temperature response relative to the reference site.

Variable	SAS code

Canopy closure	`PROC GLIMMIX METHOD=RMPL;`
	`CLASS TRMT TRYR SITE;`
	`MODEL CC=TRMT TRYR TRMT*TRYR / INTERCEPT`
	`distribution=beta link=logit DDFM=KENWARDROGER;`
	`RANDOM INT / SUBJECT=SITE TYPE=VC;`
	`LSMEANS TRMT*TRYR / CL ALPHA=0.05 ilink;`

Wetted extent	`PROC GLIMMIX`
	`CLASS TRMT TRYR SITE;`
	`MODEL PropWet= TRMT TRYR TRMT*TRYR / DISTRIBUTION=BETA`
	`LINK=LOGIT;`
	`RANDOM int / SUBJECT=SITE TYPE=VC;`
	`LSMEANS TRMT*TRYR / CL ALPHA=0.05 ilink;`

7DTR	`PROC MIXED`
	`CLASS TRYRSAS SITE;`
	`MODEL 7DTR=TRYR / DISTRIBUTION= GAUSSIAN;`
	`RANDOM INT /  SUBJECT=SITE TYPE=VC;`
	`LSMEANS TRYR / CL ALPHA=0.05;`

4-3.2. EXTENT OF WETTED CHANNEL

Each year during the low-flow period (second or third week of August), we walked the entire stream network and recorded every dry section of channel greater than 2 m in length. A section was considered dry if there was no flowing water within the channel for a continuous 2 m. These data were compiled into a GIS database.

Annual values of wetted extent, expressed as a proportion of the total Np channel length, was analyzed using a GLMM with the GLIMMIX procedure in SAS 9.4 (SAS Institute 2013). We estimated model parameters using residual pseudo likelihood, Distribution = Beta and a Link = logit function, as is often necessary for proportion data. We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals and found no evidence of either. We checked for overdispersion by ensuring that the Chi-Square/degrees of freedom was approximately equal to one. The SAS code is included in **Table 4-1**. Pairwise comparisons were used to estimate the effect size for the buffer treatment relative to the reference treatment in each post-harvest year as described above in **Equation 4-3**.

4-3.3. TEMPERATURE

We measured water temperature at 30-minute intervals using StowAway TidbiT thermistors (Onset Computer Corporation, Bourne, Massachusetts) at multiple locations within each site (**Figure 4-1**). At each location, we installed a TidbiT where there was sufficient water depth and flow to keep it submerged, and stable substrate to prevent loss of the sensor during high flows (Schuett-Hames *et al.* 1999). TidbiTs were attached to iron rebar driven into the streambed. We used zip ties to suspend the TidbiTs in the water column and leaned rocks or woody debris against the rebar to protect the sensor from direct sunlight and detection (vandalism). Portions of these streams were very shallow (<3 cm), especially near the perennial initiation point (PIP), and some sensors were installed very near the streambed surface. The likely effect of being positioned near the streambed, if any, was that in areas of upwelling, extremes in water temperature may be dampened by the influx of cooler subsurface flow.

We monitored at least four locations along the perennial stream length in each site. Locations were based on the conceptual layout of the riparian buffers (**Figure 4-1**). The intent was to measure water temperature at multiple locations along the main perennial channel from the F/N break (i.e., the transition from a non-fish-bearing to fish-bearing channel; location T1 in Figure 4-1), to the uppermost point of perennial flow (location T4), and all tributaries just above the confluence with the main channel. We monitored comparable locations in the reference (REF) sites. We installed TidbiTs in all perennial tributaries near the confluence with, but above the influence of, the main channel. Our convention for labeling these tributary 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). We also monitored a location downstream from the harvest unit in two treatment sites, which had at least 100 m of stream flowing through a fish-bearing stream buffer with no perennial tributaries, to monitor temperature response after leaving the harvest unit. Only TRT6 and TRT7 presented this opportunity.

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 the 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 noted whether sensors were submerged during field visits to measure other variables and used these records to identify specific times when a TidbiT was not submerged so that these data could receive special scrutiny.

Prior to use, all TidbiTs passed a calibration check in which they were compared 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. We rechecked the calibration on 57 TidbiTs after nearly five years of use and all 57 were within 0.2°C of the NIST thermometer. This strongly suggests that the sensors precisely and accurately recorded steam temperature within 0.2°C with no apparent drift over time.

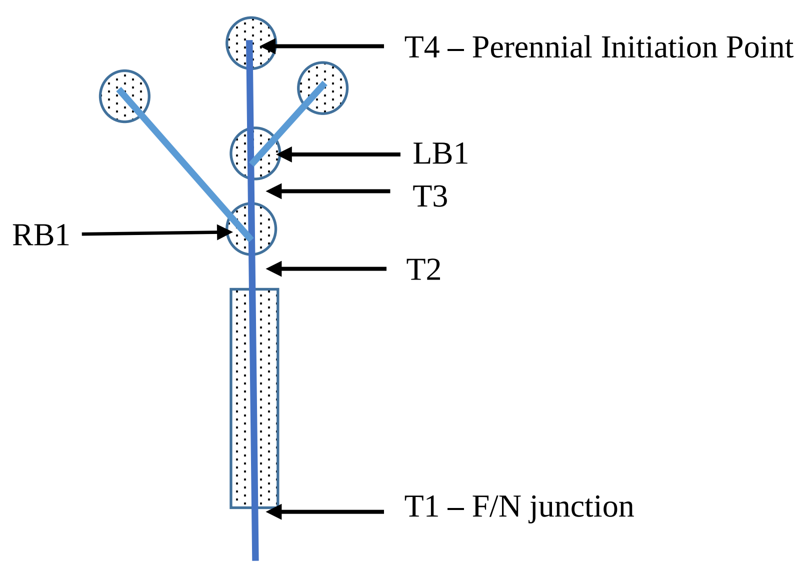

Figure 4-1. Conceptual layout of Forest Practices riparian buffers and temperature monitoring locations. Stippled areas are riparian buffers. Alphanumeric codes indicate the temperature monitoring locations ("T1", "T2", etc.) described in the text.

4-3.4. TIME SERIES ANALYSIS FOR STREAM TEMPERATURE

We calculated a daily temperature response (TR) for each monitoring location in the buffer treatment streams based upon temperature changes relative to an unharvested reference site. Our approach allowed us to select the most appropriate reference site location for each monitoring location within each treatment site and use of all daily temperature data. The daily TR values could then be summarized over the appropriate time period for further analysis. Here we used the mean monthly temperature response (MMTR) to describe the magnitude and pattern of temperature change along the stream channel and over time. We estimated the buffer treatment effects using the seven-day average TR (7DTR) in a linear mixed-effects model (LMM) analysis of variance (ANOVA). The 7DTR provides a temperature change metric analogous to the 7-day average daily maximum temperature, used in the water quality standards, but because the change in 7DTR is relative to the REF site, it has the advantage of accounting for interannual variability not related to harvest.

4-3.4.1. Calculation of Site-Specific Daily Temperature Response

Daily temperature response at each location in each treatment stream was calculated using an approach built upon and 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. The GLS regression accounts for autocorrelation between daily temperature values.

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

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 (i.e., lagged) observations plus a homoscedastic white noise term:

$$E_t = \phi_1 \varepsilon_{t-1} + \cdots + \phi_p \varepsilon_{t-p} + a_t \qquad (4\text{-}5)$$

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:

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

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:

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

The parameters of the ARMA model were determined during the GLS regression, which was conducted using the gls function from the Linear and Nonlinear Mixed-effects Models (nlme) package by Pinheiro *et al.* (2018) in 64-bit R 2.15 (R Core Team 2018). 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 were no significant ($P < 0.05$) lag one through lag twelve autocorrelation terms and the residuals were homoscedastic, relative to the predicted value and to time, and were approximately normally distributed.

If these conditions were not met with a lag six AR term, then we repeated the sequence with an MA term equal to one. If no suitable model was found using all combinations of AR terms (one through six) and MA terms (one or two), then the process was repeated using data from a different location within the same reference site.

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):

$$R^2{}_{LR} = 1 - exp(-2/_n * (logLik(x) - logLik(0))) \quad (4\text{-}8)$$

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

Pseudo R^2 is interpreted in the same manner as r^2, with $R^2 = 0$ indicating that the model explains no additional variation and $R^2 = 1$ indicating the model 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.

Each temperature monitoring location in each TRT site was paired with a location in one of the REF sites. In general, locations highest in the TRT sites tended to pair best with similar locations in the REF sites. In all cases REF2 or REF3 were better references than REF1 in terms of model fit, distribution of residuals, and pseudo-R^2. TRT5 was paired with REF2, and at all other TRT sites REF3 provided a better fit.

Step 2. Calculate the daily TR as the observed temperature minus the predicted temperature in the treatment stream:

$$TR = (y_t - \hat{y}_t) \tag{4-9}$$

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

4-3.4.2. Calculation of Location-Specific Mean Monthly Temperature Response

For each treatment year we calculated a MMTR to examine seasonal changes in maximum stream temperature. 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), the MMTR allowed us to compare across seasons within years and across years at all sites. We used the gls function within the nlme package in R to estimate the MMTR and 95% confidence intervals, using the daily TR values calculated above, for each month in the post-harvest years. We included an AR term in the model to account for the autocorrelation present and the weights = varIdent option was used to allow the variance to vary by month:

$$y_i = \beta_j + \varepsilon_{ij} \tag{4-10}$$

where: y_i is the daily temperature response,
β_j is the monthly mean response for months j=1…12, and
ε_{ij} is an error term.

The errors are modeled using an AR1 correlation structure:

$$\varepsilon_{ij_t} = \phi_1 \varepsilon_{t-1} + \cdots + \phi_p \varepsilon_{t-p} + a_{i,j} \tag{4-4}$$

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:

$$Var(\varepsilon_{ij}) = \sigma^2 \delta^2{}_j \tag{4-5}$$

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 j^{th} month and the first month (Pinheiro and Bates 2000).

We calculated MMTR on an annual basis initially because it was part of our regular (i.e., annual) data review. As we later found, analyzing longer records, often with one or more periods of missing data, sometimes produced spurious results. Spurious results could be related to the missing data or to varying AR structure over time. Regardless, we chose to avoid the issue by working with annual time series.

The large number of comparisons (months) and the large number of locations increases the likelihood of a 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.

4-3.4.3. Analysis of Buffer Treatment Effects on Temperature

Two criteria included in the water quality standards are the seven-day average maximum daily water temperature and the magnitude of human-caused changes in this metric. We used the 7DTR during July–August to estimate human-caused change in summer stream temperature. 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:

$$\Delta 7DTR_{\mathrm{TRT}} = 0 \qquad (4\text{-}6)$$

where: $\Delta 7DTR_{\mathrm{TRT}}$ is the post-harvest change, post- minus pre-harvest, in the treatments.

We used a LMM that incorporated both fixed and random effects for hypothesis testing. In matrix form, this model can be represented as:

$$Y = X\beta + Z\gamma + \epsilon \qquad (4\text{-}7)$$

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

The observations were the 7DTR for each combination of site/year. The fixed effect was period (pre-harvest, post-harvest year 1, post-harvest year 2, or post-harvest year 3). Site was included as a random effect to allow the intercept to vary by site. We used the GLIMMIX procedure in SAS (SAS Institute 2013) with Distribution = Gaussian and Link = Identity. We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals and found no evidence of either.

We used pair-wise comparisons to estimate post-harvest changes in 7DTR for each post-harvest year. We used the F/N break location in the analysis for all sites except TRT6, where we used location T2, 150 m above the F/N break, because of missing data at the F/N break sensor. The stream at this site was 96% buffered, so the T2 location still met the criterion of buffering at least 50% of the stream length.

4-3.5. STATIONARITY OF REFERENCE SITES AND SENSITIVITY OF THE METHOD

The use of a reference site assumes that in the absence of harvest the treatment and reference conditions are correlated and that this relationship does not change over the course of the study (i.e., is stationary). If this relationship changes (e.g., due to the reference basin changing over time), then spurious changes may be detected in the treatment sites. One cannot test this REF-TRT relationship after harvest (because the TRT sites were harvested), but we can evaluate the relationship between REF sites over time to ensure they are stationary relative to each other. If true, this ensures that post-harvest changes in TRT sites are not due to changes in the reference.

We used the same method described above to fit a regression model of daily maximum water temperature between the F/N break locations in the REF2 and REF3 sites. We arbitrarily set REF3 as a reference site, calibrated the model using data from 2012 to 2014, and estimated the daily TR and the MMTR in REF2 for the 2015 to 2017 period. P-values were not adjusted for multiple comparisons, so we expected more than 5% Type 1 errors. However, if the relationship between sites was stationary, we expected the errors to be evenly distributed in direction (positive or negative) and magnitude. Daily TR values were calculated and plotted for the pre- and post-harvest period. MMTR was tabulated as an index of the sensitivity of the analysis of location-specific changes in monthly average maximum daily stream temperature. Only the REF2 and REF3 sites were used in this analysis because the fit of the regression models using REF1 was poor (e.g., the relationship was nonlinear and there was a non-normal distribution of model residuals).

4-4. RESULTS

4-4.1. CANOPY CLOSURE

Initially, all sites were well shaded with average site-wide pre-harvest canopy closure ranging from 92 to 99% (**Table 4-2**). Canopy closure remained stable in the REF sites throughout the course of the study, ranging from 95 to 99%. Canopy closure decreased 5 to 59 percentage points from the pre-harvest mean in Post 1 across all TRT sites. Losses continued in Post 2 at all TRT sites except TRT3, which varied by only 2 to 3% from Post 1 through Post 3, and TRT7, where canopy closure was near (within 3%) of pre-harvest levels by Post 2. Greater declines in canopy closure were observed in TRT1 (68%), TRT2 (38%), and TRT3 (45%), the treatment sites with less than 60% of the stream length buffered, than at TRT4–TRT7, with more than 90% buffered (**Table 2-4**). Canopy closure was stable to slightly increasing from post-harvest lows at most TRT sites by 2017, but was near pre-harvest levels only in TRT7, which was buffered for 100% of its length with buffers averaging 79 ft in width.

Table 4-2. Mean canopy closure values (%) by site and year. Shaded values were measured post-harvest. TRT1 was inaccessible during harvest in summer 2014.

Year	REF1	REF2	REF3	TRT1	TRT2	TRT3	TRT4	TRT5	TRT6	TRT7
2012	98	96	97	95	99	98	97	94	95	95
2013	95	95	97	94	98	97	94	94	95	95
2014	95	95	96	—	95	55	94	94	94	92
2015	97	97	99	36	72	56	79	89	84	87
2016	97	96	97	27	59	58	71	81	76	91
2017	97	96	98	29	59	53	75	85	79	92

After harvest canopy closure at individual locations varied with buffer presence and width. We classified each measurement location in the TRT sites as Unbuffered, <50 ft, 50–75 ft, or >75 ft in width after harvest to compare with REF site locations. These are plotted by treatment year in **Figure 4-2**. Median pre-harvest values were greater than 95% for all buffer types. Post-harvest median canopy closure remained greater than 96% in the REF sites and decreased only slightly to 91% in Post 2 in buffered reaches >75 ft. The median value for 50–75 ft buffers decreased to as low of 82% in Post 2 and was 86% in Post 3. The median value for <50 ft buffers decreased to a minimum of 66% in Post 2 and was 78% in Post 3. The median value at unbuffered locations was less than 2% through Post 3. Minimum median values for all buffer types occurred in Post 2. In addition, the variability in canopy closure measurements in the <50 ft, 50–75 ft, and >75 ft buffer categories increased after harvest as shown by the larger interquartile ranges in Post 1 to Post 3.

The p-value associated with the treatment × period interaction in the GLMM ANOVA was <0.0001 (**Table 4-3**) indicating that post-harvest changes in canopy closure differed between the two treatments. Pair-wise comparisons of post-harvest changes in the TRT relative to the unharvested REF treatment indicated that decreases in canopy closure persisted through all post-harvest years (**Table 4-4**). Least squares means in the TRT sites were 22, 29, and 28 percentage points lower in Post 1, Post 2, and Post 3, respectively, while the REF varied by only 1 percentage point over the same period (**Table 4-5**).

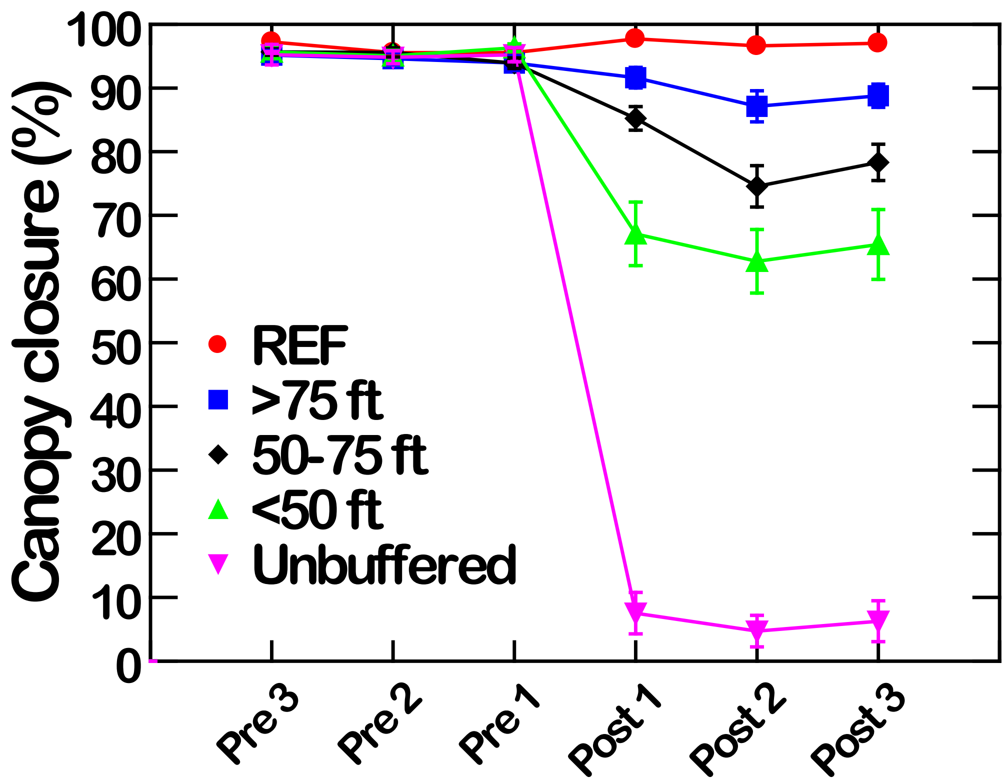

Figure 4-2. Canopy closure by buffer category and treatment year. Points are mean (±1 standard error) canopy closure of all measurement locations within REF sites (red); reaches with average buffer width greater than 75 ft (blue); average width from 50–75 ft (black); less than 50 ft (green); and no buffer (magenta).

Table 4-3. Type 3 Fixed Effects of the GLMM ANOVA for canopy cover, wetted extent, and maximum seven-day average temperature response (7DTR). Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Metric	Effect	Num DF	Den DF	F-value	P-value
Canopy Closure	Treatment	1	33	14.32	0.0003
	Period	3	33	11.19	<0.0001
	Treatment × Period	3	33	17.04	<0.0001
Wetted Extent	Treatment	1	34	0.51	0.0013
	Period	3	34	16.51	<0.0001
	Treatment × Period	3	34	8.59	0.0002
7DTR	Period	3	29.96	7.49	0.0007

Table 4-4. Post-harvest change in canopy closure in the treatment sites relative to the reference sites by year. Estimates are presented in Beta-space. P-values were not adjusted for multiple comparisons. SE = standard error; DF = degrees of freedom; C.I. = confidence intervals.

Year	Estimate	SE	DF	t-value	P-value	95% C.I.	
Post 1	-2.96	0.46	33	-6.49	<0.0001	-3.87	-2.05
Post 2	-2.92	0.41	33	-7.10	<0.0001	-3.74	-2.10
Post 3	-3.01	0.43	33	-7.02	<0.0001	-3.87	-2.15

Table 4-5. Least squares means of canopy closure presented as percent. LCL = Lower 95% confidence limit; UCL = Upper 95% confidence limit.

	Reference			Treatment		
Year	**Mean**	**LCL**	**UCL**	**Mean**	**LCL**	**UCL**
Pre	96	90	98	97	95	99
Post 1	97	92	99	75	63	85
Post 2	96	90	99	68	54	79
Post 3	97	91	99	69	56	80

4-4.1.1. Within Site Variability and Relationship with Tree Mortality

Post-harvest mortality of trees (basal area/hectare) within the riparian buffer varied widely among sites and was largely driven by windthrow (**Table 4-6**). Mortality was highest in TRT1–TRT3 where less than 60% of the stream channel was buffered, and lowest in TRT6 and TRT7 with 96% and 100% buffered, respectively. Lower mortality rates were observed in sites with longer total buffer length, but this may be an artifact of our presenting the two subbasins of TRT1 separately. When compiled as a single site, TRT1, mortality was still high but the total buffer length was comparable to TRT7. There was no apparent relationship of mortality with the total area harvested or with site aspect. However, four of the seven sites have NW aspects and all but two sites are northerly, so any quantitative relationship would be difficult to detect.

The effect of post-harvest tree mortality can be observed in the canopy closure measurements taken within the buffered reaches of the TRT sites (**Figure 4-3**). In the REF sites, with very low mortality, canopy closure measurements at nearly all locations were greater than 85% with no obvious difference in the range or variability of measurements among years. In contrast, in both sub-basins of TRT1 canopy closure in the buffer was not only lower at nearly all locations after harvest, but it more variable over time due to continued windthrow. Sites TRT2–TRT7 demonstrated varying degrees of loss in canopy closure after the intial harvest, but not as widespread or as severe as noted in TRT1, with continuing decrease in canopy closure at some locations in Post 2 and 3.

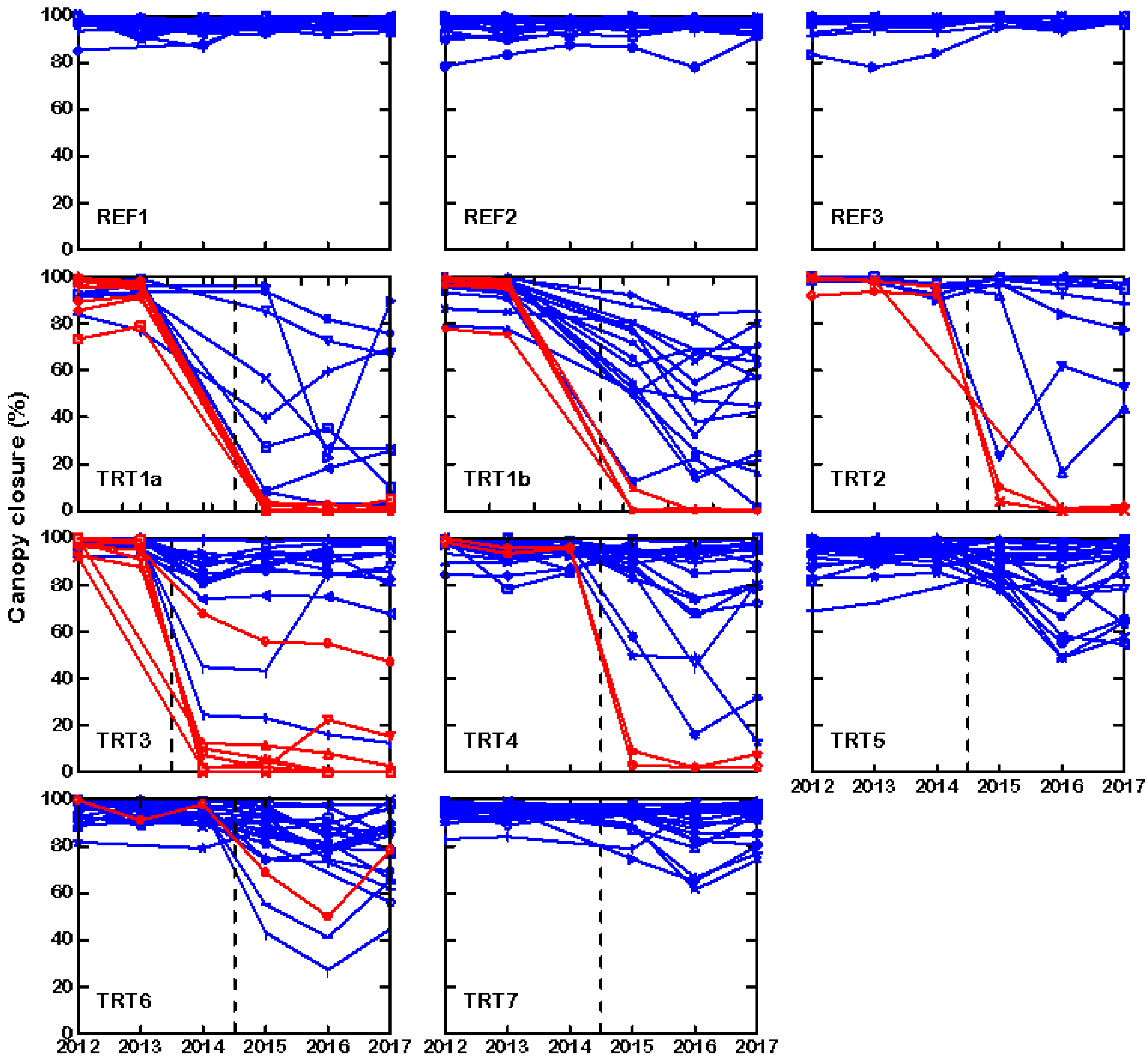

Figure 4-3. Canopy closure plotted for each location over time. Blue indicates the location was within a buffered reach; red indicates no buffer. Vertical dashed line separates pre-harvest from post-harvest measurements.

Table 4-6. Post-harvest tree mortality (based on basal area/hectare) within the perennial initiation point (PIP) and riparian management zone (RMZ) buffers, area harvested, percent stream length buffered, total buffer length, mean buffer width, and stream aspect by site. TRT1 is shown as two sub-basins to expand the range of percent stream buffered.

Site	PIP (%)	RMZ (%)	Harvested Area (ha)	Percent Buffered	Buffer Length (m)	Buffer Width (m)	Aspect
REF1	1.7	1.5	-	-	-	-	SW
REF2	0.4	0.8	-	-	-	-	SW
REF3	0.3	1.1	-	-	-	-	W
TRT1a	63.2	48.4	30.2	40	316	15	NW
TRT1b	61.0	54.8	37.4	63	585	20	NW
TRT2	22.7	18.6	21.7	54	320	15	NW
TRT3	29.3	4.6	26.3	58	560	14	NW
TRT4	12.1	14.0	32.3	92	794	17	NW
TRT5	14.6	7.0	27.2	95	999	15	SW
TRT6	5.8	6.4	21.0	96	957	14	NE
TRT7	4.3	4.9	49.1	100	940	23	SE

4-4.2. TEMPERATURE

Mean monthly maximum daily temperatures are shown in **Table 4-7**. Pre-harvest values were less than 15°C at all sites. The highest pre-harvest mean monthly temperatures occurred in July or August and ranged from 11.3 to 14.9°C in the REF sites and 10.1 to 14.8°C in the TRT sites. During and after harvest, mean monthly water temperatures were higher, but equaled or exceeded 15.0°C only in TRT1 (Post 1 to 3; by up to 1.8°C) and TRT5 (during harvest; by 0.1°C). None of the three REF sites exceeded 15°C during the study.

There was a general pattern of higher summer maximum daily water temperature from 2015 through 2017 at all sites, including the three unharvested REF sites (**Figure 4-4**). In the three REF sites, the July 2015 mean maximum daily temperature was 1.3°C to 1.8°C higher than the 2012 to 2014 July average. The mean July maximum daily temperature averaged across the three REF sites was warmer by 1.5°C, 0.5°C, and 0.1°C in 2015, 2016, and 2017, respectively, than the 2012 to 2014 average. Post-harvest mean July temperature was higher than pre-harvest at all TRT sites with the exception of TRT5 in Post 2. The pre- to post-harvest differences at each site were highest in 2015 ranging from 1.1°C in TRT5 to 2.8°C in TRT1. These post-harvest differences decreased at all sites in 2016 and 2017 from the 2015 highs.

Table 4-7. Mean monthly maximum daily temperatures at the F/N junction (transition of fish-bearing to non-fish-bearing stream). Light blue shading indicates the harvest period, and gray shading the post-harvest period.

Site	Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
	2012								12.3	11.7	10.2	9.1	8.1
	2013	6.9	7.5	7.8	8.3	9.5	10.2	11.3	11.9	12.6	9.9	8.9	6.8
REF1	2014	7.7	7.0	8.4	8.9	10.0	10.8	12.3	13.0	12.7	11.9	9.1	8.9
	2015	8.8	9.2	9.2	9.2	10.2	11.7	13.1	13.5	12.1	11.9	9.2	8.6
	2016	8.2	9.1	9.1	10.1	10.6	11.5	12.3	13.4	12.3	10.9	10.3	7.6
	2017	6.6	7.4	8.0	8.5	9.7	10.8	11.8	12.9	12.6			
	2012						10.6	11.6	13.7	12.4	10.4	9.6	8.5
	2013	7.0	7.6	8.0	8.6	10.1	11.4	13.2	14.0	13.5	10.4	9.4	7.2
REF2	2014	7.8	7.1	8.6	9.2	10.4	12.1	14.3	14.9	13.7	12.2	9.7	9.4
	2015	9.0	9.2	9.1	9.2	10.5		14.6	14.7	12.9	12.3	9.9	9.1
	2016	8.4	9.2	9.2	10.2	11.2	12.7	13.8	14.3	13.0	11.5	10.9	8.4
	2017	7.0	7.6	8.2	8.7	10.0	11.2	13.2	14.4	13.5			
	2012						10.9	11.7	13.6	12.9	10.4	9.7	8.6
	2013	6.9	7.7	8.1	8.6	10.1	11.3	12.9	13.7	13.5	10.3	9.3	7.0
REF3	2014	7.8	7.2	8.7	9.3	10.5	11.8	13.8	14.4	13.5	12.3	9.7	9.4
	2015	9.1	9.3	9.2	9.5	10.7	12.8	14.6	14.7	12.8	12.2	9.8	9.1
	2016	8.5	9.3	9.3	10.0	10.8	11.8	13.0	13.9	12.9	11.5	10.9	8.5
	2017	6.9	7.6	8.4	8.7	10.0	11.2	13.0	14.5	13.7			
	2012					10.6	10.9		14.4	13.0	10.6	9.2	7.7
	2013	5.6	6.8	7.5	8.5	10.7	12.3	14.0	14.8	14.0	10.2	8.4	5.6
TRT1	2014	6.7	5.9	8.1	9.0	11.2	12.8	15.7	16.8	15.4	13.5	8.9	8.6
	2015	8.2	9.0	9.5	10.4	12.9	15.2	16.7	16.4	14.0	12.8	9.2	8.5
	2016	7.7	9.2	9.7	11.4	12.8	14.1	15.2	16.0	14.0	12.1	11.0	7.6
	2017	5.6	7.2	8.7	9.6	11.5	13.1	14.8	15.9	14.6			
	2012						9.9	10.7	12.9	12.1	10.0	9.4	8.7
	2013	7.4	8.0	8.3	8.7	9.7	10.7	12.1	13.4	13.1	9.9	9.2	7.6
TRT2	2014	8.1	7.7	8.7	9.1	10.0	11.4	13.2	14.2	13.3			
	2015					10.7	12.3	14.0	14.4	12.8	11.9	10.1	9.8
	2016	9.1	9.6	9.5	10.1	10.8	11.8	12.7	13.6	12.7	11.6	11.2	9.5
	2017	8.2	8.6	9.0	9.4	10.2	11.0	12.3	13.7	13.2			
	2012								12.9	12.0	10.0	9.5	8.6
	2013	7.1	7.8	8.0	8.4	9.7	10.8	12.3	13.4	13.2	10.0	9.2	7.2
TRT3	2014	7.7	7.5	8.7	9.2	10.5	12.0	13.4	14.4	13.5	12.0	9.8	9.6
	2015	9.2	9.3	9.4	9.6	10.9	12.8	14.4	14.5	12.9	12.2	10.0	9.6
	2016	8.7	9.5	9.6	10.2	11.0	12.2	13.3	13.9	12.8	11.9		
	2017					10.9	11.3	12.8	13.9	13.1			
	2012						10.8	11.8	13.5	12.2	10.1	9.5	8.4
	2013	6.6	7.7	8.2	8.7	10.1	11.3	12.9	13.9	13.5	10.0	9.0	6.8
TRT4	2014	7.5	7.0	8.5	9.1	10.3	11.7	13.9	14.8	13.7	12.2	9.3	9.1
	2015	8.8	9.2	9.4	9.8	11.0	13.1	14.7	15.0	13.1	12.3	9.8	9.3
	2016	8.5	9.5	9.6	10.6	11.2	12.5	13.7	14.3	13.1	11.8	11.3	8.8
	2017	7.1	8.0	8.9	9.4	10.5	11.8	13.0	14.3	13.9			
	2012						10.9	11.9	13.3	12.4	10.6	9.7	8.5
	2013	6.9	7.7	8.1	8.8	10.1	11.2	12.9	13.6	13.4	10.4	9.4	7.1
TRT5	2014	7.7	7.2	8.6	9.1	9.7			14.8	13.8	12.2	9.6	9.3
	2015	9.0	9.3	9.2	9.4	10.8	13.2	14.9	15.1	13.7	12.7	10.3	9.5
	2016	8.6	9.4	9.4	10.4	11.5	12.5	13.5	14.6	13.5	12.1	11.5	9.0
	2017	7.1		8.7	8.7	11.1	11.6	13.1	14.4	13.8	11.8		
	2012						10.9	11.6	13.0	12.2	10.6	9.5	8.3
	2013	6.8	7.6	7.9	8.6	10.0	11.2	12.6	13.2	13.2	10.4	9.2	6.8
TRT6	2014	7.9	6.7	8.6	9.1	9.8			14.1	13.7	12.4	9.2	9.0
	2015	8.9	9.4	9.4	9.4	10.6	12.5	14.1	14.4	13.1	12.5	9.7	8.9
	2016	8.2	9.4	9.4	10.2	11.0	12.1	13.3	13.9	13.1	11.7	11.0	8.1
	2017	6.6	7.7	8.8	9.3	10.4	11.6	12.8	13.9	13.4	11.7		
	2012						9.1	10.1	11.5	11.1	9.6	8.2	7.1
	2013	6.5	6.8	7.2	7.6	8.9	9.6	10.9	11.6	11.8	9.4	8.3	6.3
TRT7	2014	7.5	6.6	7.7	8.1	9.2	10.1	11.8	12.7	12.3	11.0	8.3	8.1
	2015	8.4	8.9					12.7	13.1	11.7	11.2	8.8	8.3
	2016	8.0	8.7	8.7	9.6	10.2	11.0	11.5	12.6	11.8	10.4	9.8	7.6
	2017	6.7	7.4	8.1	8.4	9.5	10.3	11.4	12.7	12.5	10.7		

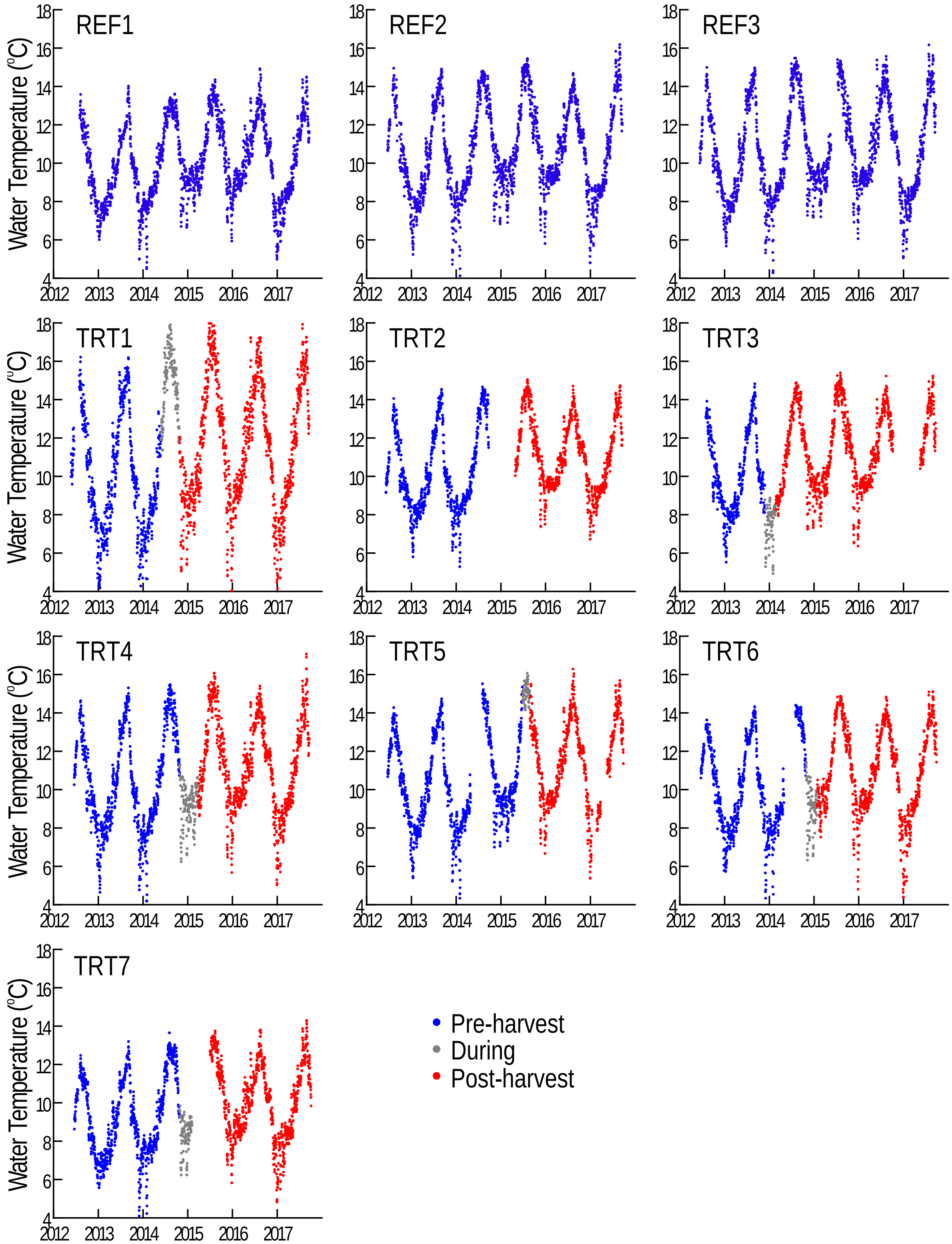

Figure 4-4. Daily maximum water temperature at F/N break (transition from fish-bearing to non-fish-bearing stream) at all study sites. Blue indicates the pre-harvest period, gray during harvest, and red the post-harvest period.

4-4.2.1. Regression Results

We used only REF2 and REF3 as reference sites in the predictive models because the models using REF1 were not acceptable due to poor model fit, generally at the upper and lower end of the temperature range; heteroscedasticity or non-normality of the residuals; or some combination of the three. REF3 provided the better temperature prediction model for all treatment sites except for TRT5, where REF2 was used. The GLS regressions at all 34 TRT locations exhibited significant lag one or greater autocorrelation in the residuals. Autoregressive lag terms in the final models ranged from one to six and 75% of sites had lag one or two AR terms. The MA term was needed in only one location. Pseudo R^2 values from all locations ranged from 0.478 to 0.873 with a median of 0.726. Pseudo R^2 values from the monitoring locations used in the analysis of buffer treatment effects on the 7DTR ranged from 0.604 to 0.873 with a median of 0.799.

4-4.2.2. Stationarity of Reference Sites

We calculated TR and MMTR using REF3 as the reference and REF2 as our 'treatment' site to evaluate stationarity (**Table 4-8**). Daily TR values in the post-calibration years ranged from -0.95 to 2.56°C with 98.5% of all TR values between -1.0 and 1.0°C (**Figure 4-5**). The MMTR ranged from -0.1 to 0.2°C in the two-year calibration period and -0.4 to 0.7°C in the three-year post-calibration period. MMTR exceeded 0.5°C in only two of the 35 (5.7%) post-calibration months and there was no apparent trend in MMTR across years. Both occurrences were during a spike in the daily TR values in June and July 2016 (**Figure 4-5**). This event was short-term and no similar event was seen in the summers of 2015 or 2017, both of which were warmer than 2016. We saw no indication of similar events in the TRT sites, suggesting this was not a modeling error or caused by the use of a particular REF site. Overall, these results suggest the two reference sites used were stationary over time and that our method should reliably detect a MMTR of approximately 0.5°C.

Table 4-8. Mean monthly temperature response (MMTR) for reference-to-reference regressions. MMTR values greater than 0.5°C and P <0.05 in magnitude are shaded red (positive).

Treatment Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
Pre 2	0.1	0.0	0.0	-0.1	0.0	-0.1	0.1	0.0	-0.1	-0.1	0.0	0.0
Pre 1	0.1	0.0	0.0	-0.1	-0.1	0.2	0.2	0.2	0.0	0.0	0.1	0.1
Post 1	0.0	0.1	0.1	-0.1	-0.2		-0.1	-0.1	-0.1	-0.1	0.0	0.1
Post 2	0.1	0.1	0.0	0.4	0.2	0.7	0.6	0.3	0.0	0.0	0.1	0.0
Post 3	-0.1	0.1	-0.1	0.0	0.0	-0.1	0.0	-0.4	-0.3	-0.1	0.0	0.0

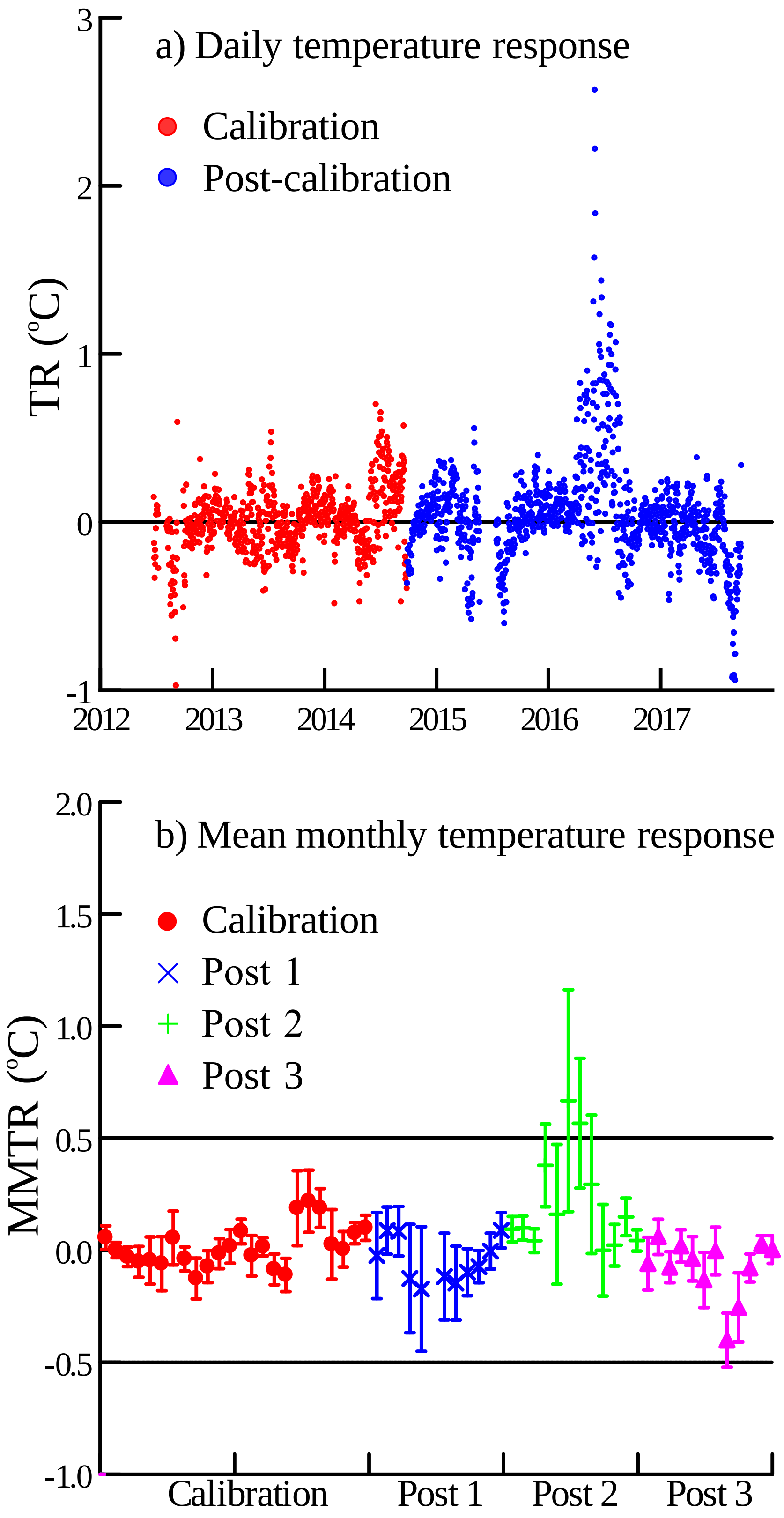

Figure 4-5. Daily temperature response (TR) and mean monthly temperature response (MMTR) for the evaluation of stationarity of the reference sites.

4-4.2.3. Seasonal Temperature Response

Our results indicated that temperatures at the TRT sites were elevated post-harvest over much of the year, often beginning in early spring and persisting well into the autumn (**Table 4-9**). Post-harvest MMTR values were greater than 0.5°C ($P < 0.05$) at most locations (monitored locations are nested within sites) in all harvested sites for four or more months per year. The increase in stream temperature after harvest is apparent when comparing the proportion of MMTRs exceeding 0.5°C in **Table 4-9** to the proportion of MMTRs calculated in the REF sites in **Table 4-8**. Of the 887 MMTR values calculated for the TRT sites in the post-harvest period, 63% (516) were greater than 0.5°C ($P < 0.05$), indicating higher temperatures, while fewer than 1% (7) were less than -0.5°C ($P < 0.05$), indicating lower temperatures. In comparison, only 5.7% (2) of the 35 post-calibration MMTRs calculated for the REF site were greater than 0.5°C and none were less than -0.5°C. We also compared the post-harvest results to the pre-harvest MMTRs calculated for the TRT locations (**Table 4-10**). Daily TR values in the pre-harvest period are the regression residuals and, in the absence of a substantial temporal (seasonal) effect, we expected very few MMTRs in **Table 4-10** would exceed 0.5°C and that they would be evenly split between positive and negative values. The pre-harvest MMTRs were equitably split with only 1.7% (9) of the 537 MMTR values calculated for the TRT sites greater than 0.5°C and 1.3% (7) less than -0.5°C ($P < 0.05$). Collectively, this suggests the pattern of post-harvest temperature increases shown in **Table 4-9** is real and likely not the result of seasonal variability not accounted for in the regression model.

The magnitude of temperature change varied across sites generally as a function of the proportion of stream buffered, tree mortality, and related measures. MMTRs tended to be highest in TRT1, the site with the lowest proportion of the stream buffered (53%), the greatest length of unbuffered stream, and the lowest post-harvest canopy closure (**Table 4-2**). Here, MMTR exceeded 2.0°C at ten of the 14 locations monitored, with MMTR values between 2.7 and 4.7°C at seven locations. Maximum MMTR values of 1.8 and 1.6°C were seen in TRT2 and TRT3 with 54% and 58% of the stream buffered, respectively. In contrast, sites TRT4, TRT5, TRT6, and TRT7, where shade levels were higher and the buffered stream length was 92% or greater, maximum MMTR values were lower, ranging from 1.0 to 1.3°C.

The timing of the maximum MMTR varied among TRT sites. MMTR tended to be higher in the summer months at sites TRT1, TRT2, and TRT3, where the buffered stream length was less than 60%. In contrast, the highest MMTRs tended to occur in the spring or fall months in TRT4–TRT7, where buffered stream length exceeded 90%. In absolute terms spring MMTR tended to be highest in TRT1, where seven locations exceed 2.0°C, compared to TRT2–TRT7 where it never exceeded 1.3°C.

Temperature response high in the stream network near the PIP varied among sites. Location T4 in TRT3 showed negligible response after harvest and the T4 location in TRT7, three meters below the PIP, cooled up to 0.7°C. However, both PIPs in TRT1, T4a and T4b, warmed more than 2.0°C and the T4 location in TRT6, only three meters below the PIP, warmed up to 1.3°C. In the summer the PIPs ranged from seeps with barely perceptible flow (TRT1, TRT2, TRT5, and TRT6) to something more akin to a spring, with easily discernible flow (TRT3, TRT4, and TRT7).

MMTR at the F/N break exceeded 1.0°C only at TRT1 and TRT7. However, at TRT7 this occurred in April and did not extend into the warmer summer months as in TRT1. At the F/N break, all sites, except TRT2, showed warming of 0.5 to 1.0°C in the July–August period. The highest July–August MMTRs at the F/N break across all sites ranged from 0.4 to 1.2°C. Nearly all sites demonstrated less summer warming at the F/N junction than at locations higher in the stream network.

Direct comparisons of temperature response at buffered vs. unbuffered reaches across study sites were confounded because TRT4 through TRT7 had little or no unbuffered stream and, although TRT1, TRT2, and TRT3 had similar proportions of unbuffered stream channel, there were insufficient temperature data to calculate MMTR at unbuffered locations in TRT2 and TRT3. TRT1 had multiple buffered and unbuffered locations and the results suggest the loss of canopy closure was a factor. The highest MMTRs, 4.7°C and 4.3°C, were at locations T4b, a PIP with a 56-ft buffer, and T300b, located more than 200 m inside a 50-ft buffer, respectively. However, the buffers at both locations experienced severe windthrow after harvest. More than 50% and 80% of standing trees in the vegetation plots in these buffers were blown down in the first and second year after harvest, respectively. Locations LB1a, T3a, and T2a were unbuffered and warmed more than 2.0°C. The MMTR at T300b and T500b, both within relatively stable buffers, exceeded 3.4°C, but this was likely impacted by the temperature response noted upstream at T4b.

Table 4-9. Mean monthly temperature response (MMTR) in the post-harvest period at each location in each treatment site. Locations are sorted by distance upstream to the perennial initiation point (PIP) within each site. Shaded cells indicate the absolute value of MMTR ≥0.5°C with an uncorrected P-value <0.05. Blue shading indicates a decrease in temperature. The three intensities of red shading indicate warming with MMTR values of 0.5–1.0°C, 1.0–2.0°C, and >2.0°C, respectively. Location superscripts 1 = PIP, 2 = unbuffered, 3 = F/N break, 4 = downstream of harvest unit.

Site	Distance from PIP	Location	Treatment Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
TRT1	0	T4a[1]	Post 1	0.2	0.6	0.8	1.4	1.8	2.0	1.9	2.0	1.8	1.5	0.1	0.1
			Post 2	0.6	0.6	0.7	1.2	0.0	1.7	1.8	1.5	1.3	1.0	0.6	0.3
	9	T4b[1]	Post 1	0.1	0.6	1.6	2.3	3.3	4.7	3.7	3.1	2.1	0.9	-0.5	-0.1
			Post 2	0.5	0.5	0.9	1.5						0.2	-0.1	0.0
	71	LB1a[2]	Post 1	0.9	1.5	1.6	1.8	2.0	2.4	2.4	1.8	1.3	0.8	0.0	0.3
			Post 2	1.1	0.6	1.1	1.1	1.1	1.4	1.2	1.1	0.9	0.9	0.4	0.4
	150	LB1b	Post 1	-0.2	0.4	0.9	1.6	2.2	3.2	2.2	1.9	1.4	0.8	-0.3	-0.2
			Post 2	0.1	0.3	0.7	1.7	2.0	2.7	2.3	2.2	1.0	0.6	-0.2	-0.1
	259	T3a[2]	Post 1	0.0	0.5	1.0	1.7	2.6	3.4	2.3	1.5	1.0	0.4	-0.3	0.0
			Post 2	0.4	0.5	1.0	1.7	1.8	2.1	1.6	1.2	0.4	0.5	0.1	0.2
	337	T2a[2]	Post 1	0.1	0.5	0.8	1.5	2.3	3.5	3.0	2.5	1.8	0.4	-0.3	0.0
			Post 2	0.4	0.5	0.8	1.3	1.4	1.7	1.4	1.3	0.4	0.4	0.1	0.2
	357	T3b	Post 1	0.0	0.4	0.5	0.9	1.1	1.6	1.1	0.8	0.5	0.2	-0.1	-0.1
			Post 2	0.3	0.3	0.5	1.0	0.9	1.5	1.1	1.1	0.4	0.4	0.0	0.1
	460	T2b	Post 1	0.1	0.4	0.5	0.8	1.0	1.9	1.3	0.9	0.8	0.4	0.0	0.1
			Post 2	0.4	0.4	0.7	0.9	1.1	1.4	1.2	0.9	0.4	0.5	0.4	0.3
	484	T300a	Post 1	0.0	0.5	1.0	1.5	1.8	2.7	1.9	1.6	1.1	0.6	-0.2	0.0
			Post 2	0.4	0.4	0.8	1.2	1.1	1.3	1.1	1.0	0.4	0.5	0.1	
	564	T1a	Post 1	0.4	0.9	1.2	1.4	1.5	1.7	0.8	0.7	0.7	0.6	0.0	0.3
			Post 2	0.5	0.7	1.1	1.3	1.2	1.5	1.2	1.1	0.6	0.7	0.4	0.4
	610	T300b	Post 1	0.1	0.7	1.2	1.9	2.7	4.3	3.3	2.7	1.6	1.1	0.0	0.1
			Post 2	0.4	0.5	0.9	1.9	1.7	2.2	1.8	1.6	0.7	0.6	0.2	0.2
	679	T500b	Post 1	0.3			2.4	2.4	3.4	2.2	1.9	1.2	0.8	0.0	0.1
			Post 2	0.5	0.4	0.7	1.1	0.8	1.8	1.3	1.1	0.6	0.5	0.3	0.3
	755	T1b	Post 1	0.3	0.8	1.3	1.7	1.8	2.2	1.2	1.0	0.9		0.1	0.2
			Post 2					1.7	1.9	1.7	1.5	1.1	0.6		
	870	D100[3]	Post 1	0.1	0.5	0.7	0.9	1.3	1.4	0.9	0.7	0.7	0.5	-0.1	0.1
			Post 2	0.4	0.5	0.8	1.3	1.2	1.4	1.1	1.2	0.6	0.5	0.1	0.2
			Post 3	0.0	0.7	0.8	0.8	1.2	0.9	0.8	0.3	0.3	0.4	0.4	0.3
TRT2	257	T300	Post 1	0.4	0.5	0.7	1.0	1.0	1.1	1.3	1.8	1.0	0.5	0.7	0.3
	325	T500	Post 1	0.3	0.3			0.8	0.8	0.8	0.9	0.8	0.4	0.5	0.2
			Post 2	0.8	0.8	1.1	1.1	0.9	0.9	0.8	1.4	1.0	0.9	1.0	0.7
			Post 3	0.7	0.7	0.9	1.1	1.0	1.2	0.8	0.9	0.7	0.4		0.7
	436	T1[3]	Post 1					0.3	0.1	0.2	0.4	0.4	0.2	0.4	
			Post 2	0.7	0.5	0.4	0.5	0.4	0.4	0.3	0.4	0.4	0.4	0.6	0.7
			Post 3	0.9	0.9	0.7	0.7	0.5	0.1	-0.1	-0.1	0.0		0.9	0.9

Table 4-9 (continued).

Site	Distance from PIP	Location	Treatment Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
TRT3	1	T4[1]	Post 1	0.1	-0.2	0.0	-0.1				0.1	-0.1	0.2	-0.1	-0.1
			Post 2	0.3	0.3	-0.3	-0.3	0.2	-0.2	-0.3	-0.2	-0.4	-0.2	-0.2	-0.1
			Post 3	-0.5	-0.5	0.0	-0.2	-0.6	-0.7	-0.8	-0.4	-0.4	-0.2	-0.2	-0.5
			Post 4	0.1	0.2	0.2	0.6	0.3	-0.1	0.0	0.2				
	17	LB1	Post 1	0.5	0.5	0.2	0.2	0.4	0.4	0.1	0.5	0.1	0.2	0.3	0.4
			Post 2	0.8	0.8	0.7	0.7	0.8	0.9	1.1	1.2	0.9	0.8		0.7
			Post 3	0.9	1.0	0.8	0.7	0.6	0.7	0.7	1.2	0.9	0.5	0.5	0.6
			Post 4		1.3	1.3	1.2	1.2	1.0	1.0	1.0	0.9			
	264	T2	Post 1	0.5	0.6	0.3	0.3	0.7	0.8	0.8	1.3	1.0	0.6	0.8	0.5
			Post 2	0.7	0.8	0.7	0.7	0.8	1.1	1.4	1.6	1.1	0.6	0.4	0.8
			Post 3	0.6	0.7	0.8	0.7	0.7	1.0	1.0	1.1	0.8	0.7	0.9	0.8
	425	T1[3]	Post 1	0.4	0.4	0.2	0.2	0.4	0.5	0.4	0.7	0.7	0.3	0.4	0.4
			Post 2	0.6	0.6	0.5	0.4	0.4	0.5	0.7	0.7	0.6	0.4	0.4	0.7
			Post 3	0.5	0.6	0.5	0.4	0.7	0.8	0.8	0.4				
			Post 4				0.5	0.3	0.2	0.2	0.2	0.5			
TRT4	242	T3	Post 1	0.6	0.7	0.9	0.9	1.0	0.8	0.5	0.9	0.3	0.4	0.6	0.4
			Post 2	0.9	0.9	0.6	0.6	0.8	0.9	1.0	1.1	0.7	0.6	0.7	1.0
	460	T2	Post 1	0.5	0.6	0.9	1.1	1.1	1.0	0.4	0.4	0.4	0.4	0.4	0.3
			Post 2	0.8	0.8	0.5	-0.1						0.7	0.7	1.1
	696	T1[3]	Post 1	0.4	0.4	0.6	0.4	0.2	0.2	0.0	0.3	0.3	0.2	0.4	0.4
			Post 2	0.3	0.7	0.6	0.6	0.4	0.6	0.6	0.4	0.3	0.4	0.6	0.7
TRT5	72	LB3	Post 1	0.9	0.7	0.8	0.6	0.6	0.6	0.7	0.1	0.4	0.4	0.5	0.8
			Post 2	0.9	1.0	1.0	0.7	0.8	0.9	0.9	0.9	0.7	0.6	0.6	0.8
	529	T1	Post 1	0.4	0.4	0.5	0.4	0.4	0.1	-0.1	0.8	0.7	0.5	0.4	0.4
			Post 2	0.5	0.2	0.5	0.4	0.2	0.1	0.4	0.5	0.5	0.6	0.6	
TRT6	3	T4[1]	Post 1	0.6	1.1	1.0	1.0	0.7				1.2	0.8	0.1	0.2
			Post 2	-0.2	0.7	1.1	1.3	0.9	1.1	0.8	0.8	0.7	0.5	0.3	0.1
	192	T3	Post 1	0.2	0.8	0.7	0.5	0.2	0.2	0.3	0.4	0.3	0.3	0.0	0.2
			Post 2	-0.1	0.6	0.6	0.6	0.5	0.6	0.4	0.5	0.4	0.2	0.2	0.2
	323	T2[3]	Post 1	0.2	0.8	0.7	0.5	0.2	0.2	0.3	0.4	0.3	0.3	0.0	0.2
			Post 2	-0.1	0.6	0.6	0.6	0.5	0.6	0.4	0.5	0.4	0.2	0.2	0.2
	573	D100[4]	Post 1		0.6	0.5	0.2	0.0	-0.1	0.1	0.2	0.3			
			Post 2	-0.2			0.5	0.2	0.4	0.4	0.5	0.3	0.1	0.2	0.0
TRT7	3	T4[1]	Post 1	0.7	0.6	0.3	0.3	0.5		0.3	0.5	-0.5	-0.1	0.6	0.7
			Post 2					0.8	0.1	-0.6	-0.7	-0.4	-0.7		
	140	LB1	Post 1	0.9	0.7	0.6	0.7	0.6	0.5	0.5	0.5	0.3	0.5	0.7	
			Post 2	1.0	1.0	0.8	0.5	0.5	0.5	0.3	0.3	0.4	0.4	0.7	1.1
	573	T1[3]	Post 1	0.6	0.7	0.8	1.1	0.8		0.6	0.8	0.7	0.5	0.3	0.4
			Post 2	0.2	0.6	0.7	0.5	0.7	0.9	0.4	0.7	0.5	0.2	0.3	0.3
	693	D100[4]	Post 1	0.3	0.4	0.8	1.1	1.0	0.5	0.5	0.6	0.4	0.1	0.0	
			Post 2	-0.1	0.5	0.7	0.7	1.0	1.1	0.7	0.7	0.5	0.2	0.2	0.0
			Post 3	0.9	0.7	0.5	0.5	0.5							

Table 4-10. Mean monthly temperature response (MMTR) in the pre-harvest period at each location in each treatment site. Locations are sorted by distance to the upstream perennial initiation point (PIP) within each site. Shaded cells indicate the absolute value of MMTR ≥0.5°C with an uncorrected P-value <0.05. Blue shading indicates MMTR is negative. Tan shading indicates MMTR is positive.

Site	Distance from PIP	Location	Treatment Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
TRT1	0	T4b	Pre 2	0.0	-0.4	0.0	0.1	-0.2	0.0	-0.2	-0.1	0.2	0.2	0.1	-0.3
	9	T4a	Pre 1	0.2	-0.3	0.0	0.1	-0.2	0.1	-0.4	0.0	0.1	0.1	-0.1	-0.3
	71	LB1a	Pre 1	0.1	-0.3	0.1	0.3	0.0	-0.4	-0.3	-0.2	0.0	-0.1	0.0	-0.1
	150	LB1b	Pre 1	0.2	-0.3	0.0	0.0	0.0	0.2	0.0	-0.1	-0.1	0.1	-0.1	-0.2
	259	T3a	Pre 1	0.2	-0.2	0.0	0.0	0.0	0.1	-0.1	0.0	0.0	0.1	-0.3	-0.1
	337	T2a	Pre 1	0.2	-0.2	-0.1	-0.4		0.0	-0.2	0.0	0.0	0.0	-0.3	-0.2
	357	T3b	Pre 1	0.1	-0.1	0.0	-0.1	0.0	0.0	-0.1	-0.1	0.1	0.0	-0.2	-0.2
	460	T2b	Pre 2	0.2	0.1	-0.1	0.0	0.0			-0.1	0.0	0.2	0.0	0.0
	460	T2b	Pre 1	0.1	-0.1	0.0	-0.1	0.0	0.0	-0.1	0.0	0.1	0.0	-0.2	-0.2
	484	T300a	Pre 1	0.1	-0.2	0.0	-0.1	0.0	0.0	-0.1	0.0	0.0	0.0	-0.2	-0.2
	564	T1a	Pre 1	0.0	0.2	0.1	0.2	0.1	-0.2	-0.1	-0.2	0.0	0.1	0.2	0.1
	610	T300b	Pre 1	0.1	-0.2	0.0	-0.1	0.1	0.1	-0.1	0.0	0.0	0.1	-0.2	-0.2
	679	T500b	Pre 1	0.1	-0.2	-0.1	-0.1	0.0	0.1	-0.2	-0.1	-0.1	0.1	-0.2	-0.2
	755	T1b	Pre 1	0.2	-0.2	0.0	-0.1	0.1	0.1	-0.1	-0.1	-0.1	0.2	0.0	-0.1
	870	D100	Pre 2	0.2	0.1	-0.1	0.0	0.1			-0.2	-0.1	0.2	0.1	0.1
	870	D100	Pre 1	0.1	-0.3	0.0	-0.1	0.0	0.1	-0.1	0.0	0.0	0.0	-0.2	-0.2
TRT2	257	T300	Pre 2	0.1	0.1	0.0	0.0	0.0	0.1	0.0	0.2	-0.1	-0.4	-0.2	0.0
	257	T300	Pre 1	0.0	0.1	0.2	0.1	0.1	0.0	0.0	0.3	0.1	-0.2	-0.1	-0.1
	325	T500b	Pre 2	0.0	0.1	0.1	0.1	-0.1	-0.2	-0.2	-0.8	0.1	-0.1	-0.2	-0.1
	325	T500b	Pre 1	0.0	0.0	0.1	0.0	0.1	0.3	0.1	0.4	0.3	-0.3	-0.2	-0.1
	436	T1	Pre 2	-0.1	0.1	0.1	0.2	0.0	-0.1	-0.1	0.0	-0.1	0.1	-0.1	-0.1
	436	T1	Pre 1	0.1	0.2	0.1	0.0	0.0	0.1	0.1	0.4	0.1	-0.3	-0.2	0.1
TRT3	1	T4	Pre 1	0.2	0.2	-0.1	-0.1	0.0	0.1	0.0	0.2	-0.1	0.2	-0.4	0.0
	17	LB1	Pre 1	0.1	0.1	0.1	0.0	-0.1	-0.1	-0.1	0.1	0.2	0.0	0.0	-0.1
	264	T2	Pre 1	0.2	0.4	0.4	0.7	0.2	0.1	0.0	-0.1	-0.4	-0.3	-0.3	0.2
	425	T1	Pre 1	0.1	0.2	0.2	0.1	-0.3	-0.2	0.0	0.0	0.0	-0.1	0.0	0.2
TRT4	242	T3	Pre 2	0.1	0.1	0.0	0.1	0.1	0.0	-0.1	0.0	0.0	0.1	-0.1	-0.1
	460	T2	Pre 2	0.0	0.1	0.1	0.1	0.2	0.1	0.0	-0.1	0.0	0.0	0.1	-0.1
	696	T1	Pre 2	0.0	0.2	0.2	0.1	0.0	0.0	-0.1	0.0	0.0	-0.1	0.0	0.0
	696	T1	Pre 1	0.0	-0.1	0.0	-0.1	-0.2	-0.1	0.1	0.3	0.3	0.1	-0.1	-0.1
TRT5	72	LB3	Pre 1	0.0	0.0	0.2	0.2	-0.1	0.0	0.1	0.2	0.1	0.2	0.5	0.0
	529	T1	Pre 2	-0.1	0.0	0.1	0.0	-0.2		-0.2	-0.2	0.0	-0.2	-0.2	-0.3
	529	T1	Pre 1	0.2	0.3	0.3	0.2	0.1			0.3	0.2	0.1	0.2	0.0
TRT6	3	T4	Pre 2	0.1	0.3	0.1	0.1	-0.1	-0.2	-0.2	-0.2	-0.1	-0.1	-0.4	-0.3
	3	T4	Pre 1	0.1	-0.2	0.1	0.1	-0.1	0.1	0.1	0.4	0.3	0.6	0.0	-0.2
	192	T3	Pre 2	0.3	0.5	0.3	0.1	-0.1	-0.2	-0.2	-0.3	-0.1	-0.2	-0.3	-0.1
	323	T2	Pre 2	-0.2	0.0	0.2	0.5	0.5	0.7	0.4		-0.2	-0.4	-0.8	-0.8
	323	T2	Pre 1	0.0	-0.3	0.1	0.2	0.0			0.3	-0.1	0.0	0.7	0.0
	573	D100	Pre 1	0.1	-0.2	0.0	0.1	0.0	0.0	0.0	0.0	0.1	0.4	0.0	-0.2
TRT7	3	T4	Pre 1	0.4	-0.1	-0.2	0.0	-0.2	-0.2	-0.1	0.1	-0.4	-0.7	0.0	0.5
	140	LB1	Pre 2	-0.1	0.0	-0.1	-0.1	-0.1	0.3	0.2	0.3	-0.2	0.0	-0.3	-0.1
	140	LB1	Pre 1	0.1	0.1	0.0	-0.1	-0.1	-0.2	0.0	0.1	0.0	0.0	0.1	0.1
	573	T1	Pre 2	-0.2	0.0	-0.1	-0.2	-0.2	0.0	-0.1	-0.2	-0.1	0.1	-0.4	-0.3
	573	T1	Pre 1	0.5	0.1	0.3	0.3	0.2	-0.1	-0.2	-0.6	-0.4	0.0	0.7	0.5
	693	D100	Pre 1	-0.1	-0.3	-0.2	-0.2	-0.6	-0.4		0.1	0.6	0.0	-0.2	

4-4.2.4. Relationship between Temperature Response, Canopy Closure, Buffets, Site Aspect, and Discharge

There was a weak (r >0.5 <0.7) to moderately strong (r >0.7) negative correlation between July MMTR and canopy closure in Post 1 and Post 2 (**Table 4-11**; **Figure 4-6**). There were moderately strong correlations in one or both years with the percent wetted channel and total length of wetted channel (**Figure 4-7**). Correlations with total length of buffered channel and percent of channel buffered were generally weak (r <0.5). We plotted the two subbasins of TRT1 to illustrate the effect of site area on the relationships. TRT1a and TRT1b bracket TRT1 in the plots of canopy closure, percent of stream buffered, and percent wetted channel and so do not fundamentally change the scatterplots. This is not true for the plots of total buffer length or total wetted length.

Table 4-11. Pearson correlation coefficients and P-values for Pearson correlations with July mean monthly temperature response.

Year	Canopy Closure	% Buffered	Buffer Length	% Wetted Channel	Wetted Channel Length
Post 1	-0.599/0.156	-0.423/0.344	0.079/0.866	0.726/0.056	0.742/0.065
Post 2	-0.809/0.027	-0.536/0.215	0.032/0.945	0.510/0.243	0.763/0.046

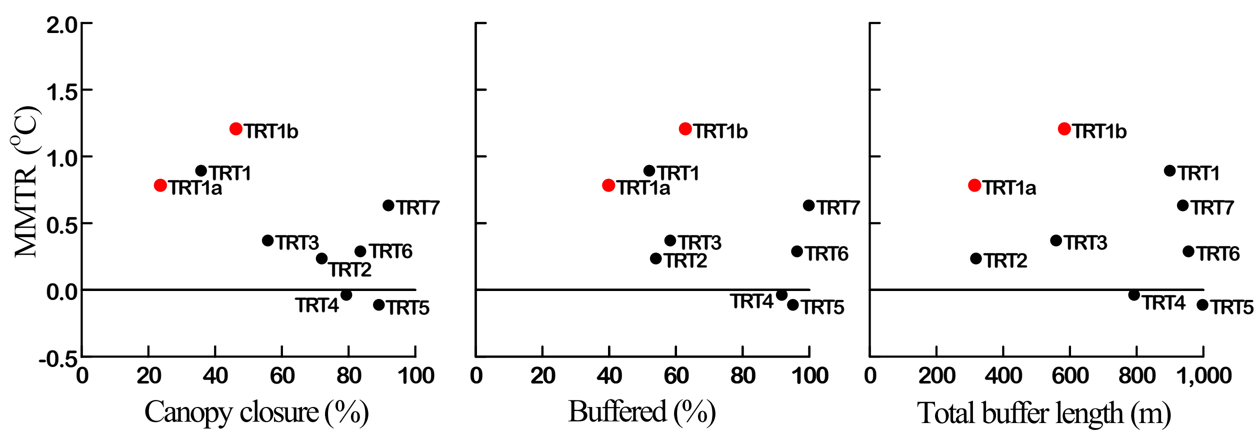

Figure 4-6. July mean monthly temperature response (MMTR) vs. canopy closure, percent of stream buffered, and total buffer length. Correlations were done using sites TRT1–TRT7. TRT1a and TRT1b, sub-basins of TRT1, are presented (in red) to illustrate the effect of basin size on the relationship. Values are from the first year post-harvest.

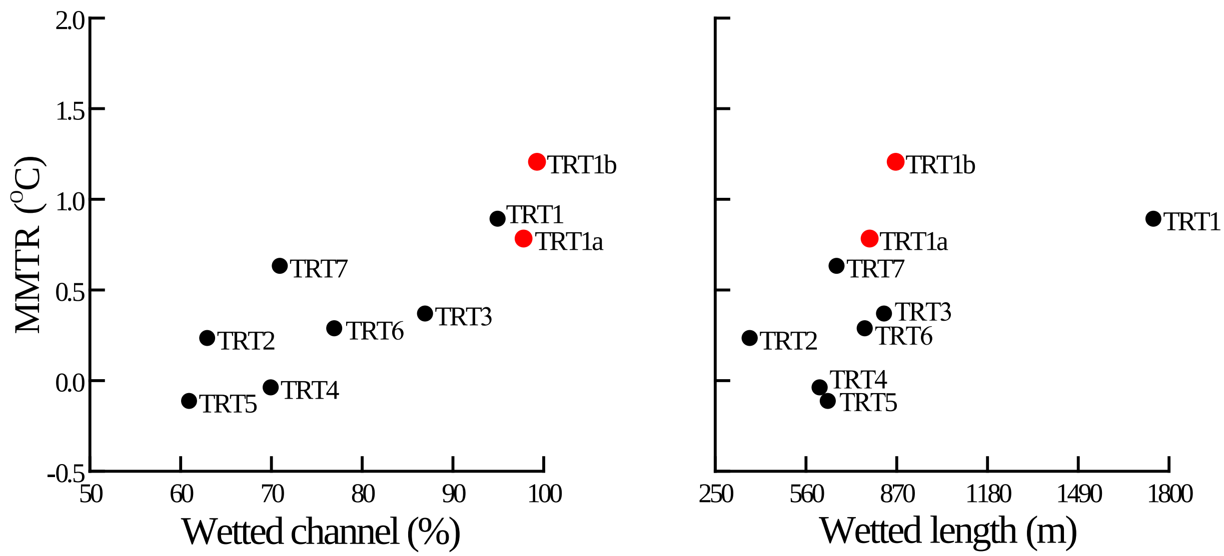

Figure 4-7. July mean monthly temperature response (MMTR) vs. percent of stream channel with surface water (wetted channel) and total wetted channel length. Correlations were done using sites TRT1–TRT7. TRT1a and TRT1b, sub-basins of TRT1, are presented (in red) to illustrate the effect of basin size on the relationship. Values are from the first year post-harvest.

We incorporated the 12 Hard Rock Study sites into the same plots below to illustrate differences between the two studies in the relationship of July MMTR in the first year after harvest and the first-year same site descriptors. Although most July MMTRs in this study tended to be lower relative to the Hard Rock Study's similarly buffered 100% and Forest Practices (FP) sites, they do fall within the cloud of points formed by the Hard Rock Study sites in the scatterplot against canopy closure, percent of stream buffered, and total buffer length (**Figure 4-8**). In the plots of July MMTR against percent wetted channel and total wetted channel length there were two distinct groups (**Figure 4-9**). One group included this study's sites plus the Hard Rock Study's 100% and FP treatment sites. The second group was comprised solely of the Hard Rock Study's 0% (unbuffered) treatment sites which had higher MMTR values across the range of x-values. We saw no apparent relationship with aspect, but within each aspect class there was a tendency of higher MMTR with lower canopy closure, with only two exceptions, one each for NE and SE aspect.

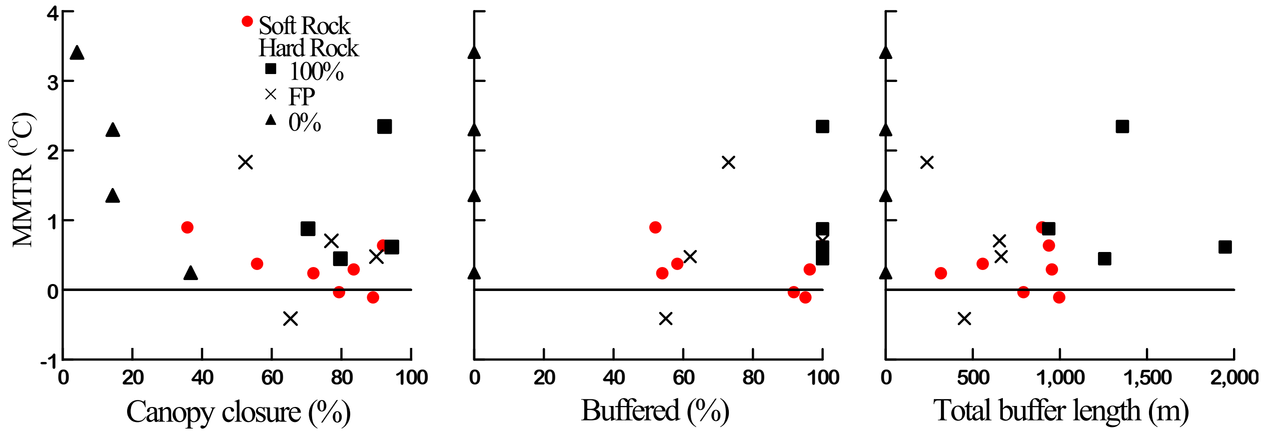

Figure 4-8. July mean monthly temperature response (MMTR) vs. canopy closure, percent of stream buffered, and total buffer length for this study (red) and the Hard Rock Study (black). Values are from the first year post-harvest.

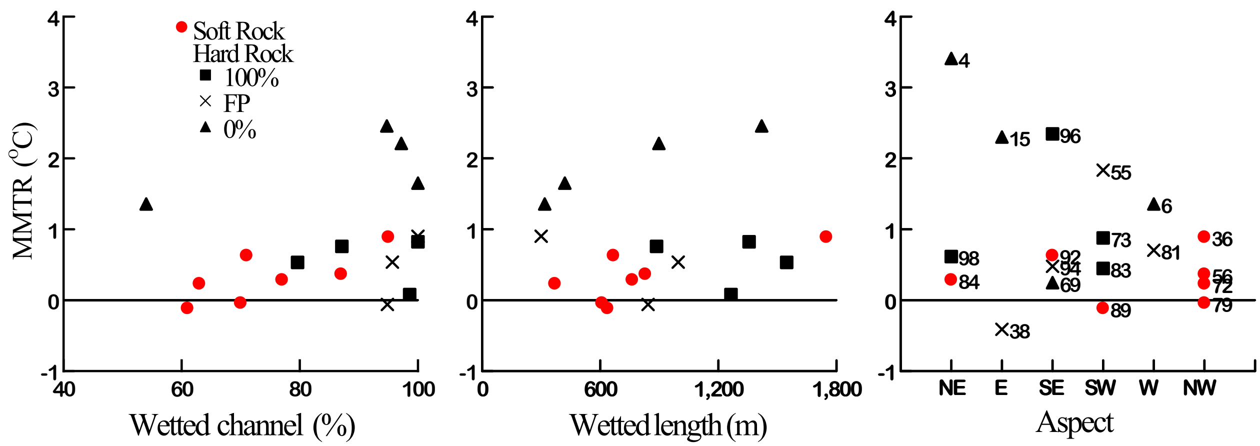

Figure 4-9. July mean monthly temperature response (MMTR) vs. percent of stream channel with surface water (wetted channel), total wetted channel length, and aspect for this study (red) and the Hard Rock Study (black). Values are from the first year post-harvest except for the Hard Rock Study values for wetted channel and wetted length, which were measured in 2010, the second year post-harvest at most sites. Numbers in the aspect plot are mean canopy closure.

4-4.2.5. Longitudinal Patterns in Surface Flow, Temperature Response, Buffer Width, and Canopy Closure

There were substantial portions of the stream at all sites with no surface water during the summer surveys (**Table 4-12**), and this varied across years and with harvest. The percentage of wetted channel was lowest in 2015, an unusually warm and dry spring and summer, at all but two sites. The p-value associated with the treatment × period interaction was 0.0002 (**Table 4-3**) indicating the post-harvest change differed between the two treatments. Pair-wise comparisons of post-harvest changes in the TRT relative to the unharvested REF treatment indicate the differences occurred in Post 1 (P = 0.0006) and Post 2 (P = 0.0002) (**Table 4-13**). The least squares mean of each treatment by year (**Table 4-14**) shows the wetted extent in the TRT sites tended to be higher after harvest relative to the REF sites. Mean wetted extent in the REF sites was 88% pre-harvest then 59 and 71% in Post 1 and Post 2, respectively. The mean in the TRT sites was 82% pre-harvest, and 75% and 86% in Post 1 and Post 2, respectively. In effect the differences between the treatments were due to a lower percentage of wetted channel in the REF sites in the first two years after harvest (2015 to 2016) while the TRT sites changed little.

Table 4-12. Percentage of the channel length with surface water by site and year. There were no surveys in 2014 at TRT1 and TRT7. Shaded cells indicate post-harvest.

Site	2013	2014	2015	2016	2017
REF1	92	90	76	81	87
REF2	87	84	54	69	88
REF3	93	81	47	64	88
TRT1	99		95	97	99
TRT2	65	62	63	67	67
TRT3	79	87	70	83	74
TRT4	75	65	70	89	83
TRT5	72	66	61	77	71
TRT6	92	90	77	78	85
TRT7	91		71	98	82

Table 4-13. Post-harvest change in wetted extent in the treatment sites relative to the reference sites by year. Estimates and confidence intervals are in Beta-space. P-values were not adjusted for multiple comparisons. SE = standard error; DF = degrees of freedom; C.I. = confidence intervals.

Year	Estimate	SE	DF	t-value	P-value	95%	C.I.
Post 1	1.20	0.32	34	3.81	0.0006	0.56	1.84
Post 2	1.40	0.34	34	4.16	0.0002	0.72	2.08
Post 3	0.18	0.37	34	0.50	0.6183	-0.56	0.93

Table 4-14. Least squares means by treatment and period expressed in percentage of stream channel with surface flow. TRMT = treatment; Period = Pre- or Post-harvest year; SE = standard error; LCL = lower 95% confidence limit; UCL = upper 95% confidence limit.

TRMT	Period	Mean	SE	LCL	UCL
REF	Pre	88	4.44	76	94
	Post 1	59	9.90	39	76
	Post 2	71	8.56	52	85
	Post 3	86	5.58	71	94
TRT	Pre	82	3.79	73	88
	Post 1	75	4.81	65	84
	Post 2	86	3.37	78	91
	Post 3	82	3.99	73	89

Longitudinal patterns in the point estimates of MMTR generally reflected the patterns in canopy closure and buffer width. In TRT1–TRT3, in which the proportion of the stream channel buffered was less than 60%, buffer width and canopy closure varied greatly along the channel length (**Figures 4-10** to **4-13**). Where canopy closure was low, July MMTR often exceeded 1.0°C in one or more years post-harvest. In nearly all cases where July MMTR exceeded 1.0°C, MMTR was lower downstream after flowing through a buffered reach where canopy closure was high. In contrast, sites TRT5–TRT7 experienced little loss in canopy closure and, although July MMTR was elevated (i.e., >0.5°C and 95% confidence intervals did not intersect zero), it never exceeded 1.0°C and varied little along the stream length (**Figures 4-15** to **4-17**). TRT4 fell between the two extremes in that canopy closure was less than 10% along a portion of the stream very high in the catchment and July MMTR at the location immediately downstream reached 1.0°C in the second year (**Figure 4-14**).

We were able to monitor locations downstream of the F/N break but above any tributaries (D100) only in TRT6 and TRT7. These locations provided estimates of temperature change after flowing through 100 m (328 ft) to 140 m (459 ft) of stream with an average buffer width of 30.5 m (100 ft) in TRT6 and 42.7 m (140 ft) in TRT7. In general, there was a negligible change in MMTR below the F/N break in either site. In TRT6, July–August MMTR ranged from 0.3 to 0.5°C at the F/N break and 0.1 to 0.5°C at D100 (**Table 4-9**). July–August MMTR in TRT7 was sometimes slightly elevated at the F/N break, ranging from 0.4 to 0.8°C, and downstream ranged from 0.5 to 0.7°C at D100.

We did not find a consistent relationship between temperature response and the presence of reaches with no surface water immediately upstream. The was one notable instance in TRT2 where relatively warm water, with August MMTR up to 1.8°C, flowed from an unbuffered reach into a buffered reach with a persistent dry reach. Summer MMTR below that point, at T1, was less than 0.4°C.

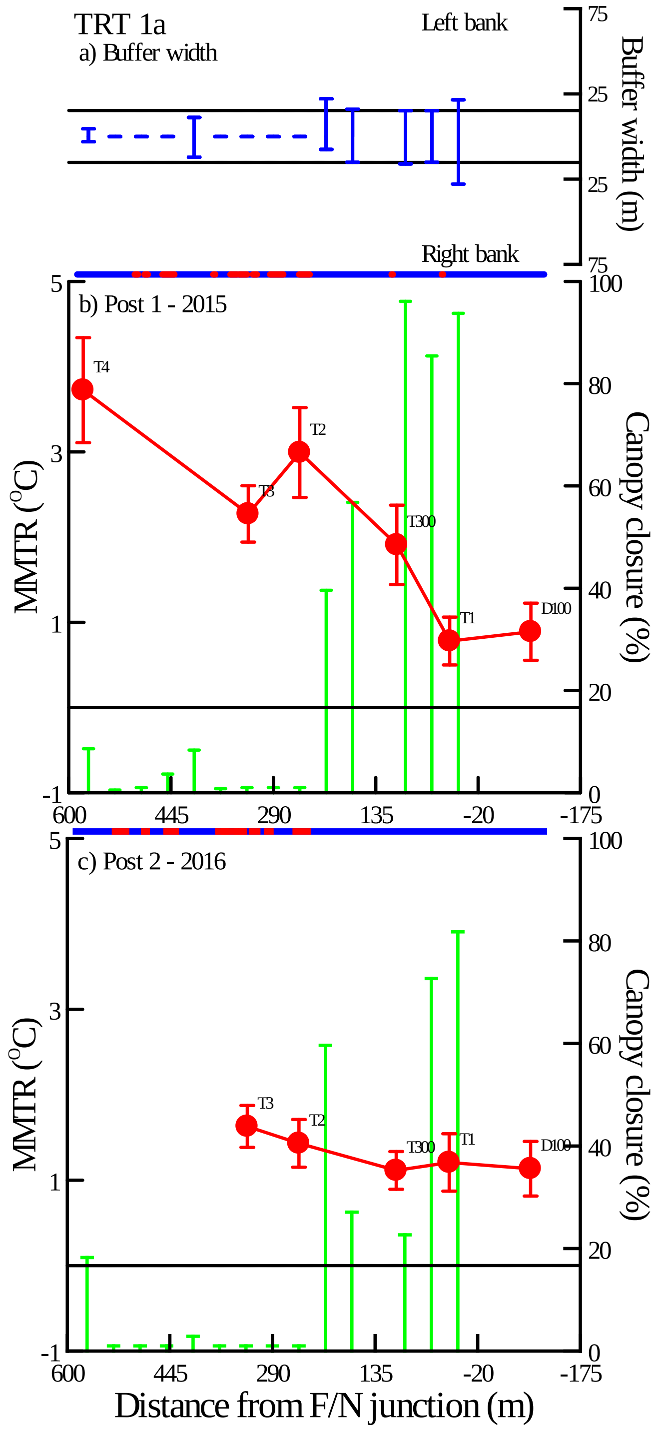

Figure 4-10. Site TRT1a buffer width, canopy closure, July mean monthly temperature response (MMTR), and wetted extent. Panel a): Buffer widths plotted by distance from F/N junction. Dashed horizontal lines indicate 50-ft (15-m) width. Panel b): First year post-harvest. Blue and red line at top of graph shows surface water and dry reaches, respectively. July MMTR with 95% confidence intervals are shown in red with locations labeled. Dashed horizontal line indicates MMTR = 0. Canopy closure is shown in green. Panel c): Second year post-harvest (same variables as Panel b).

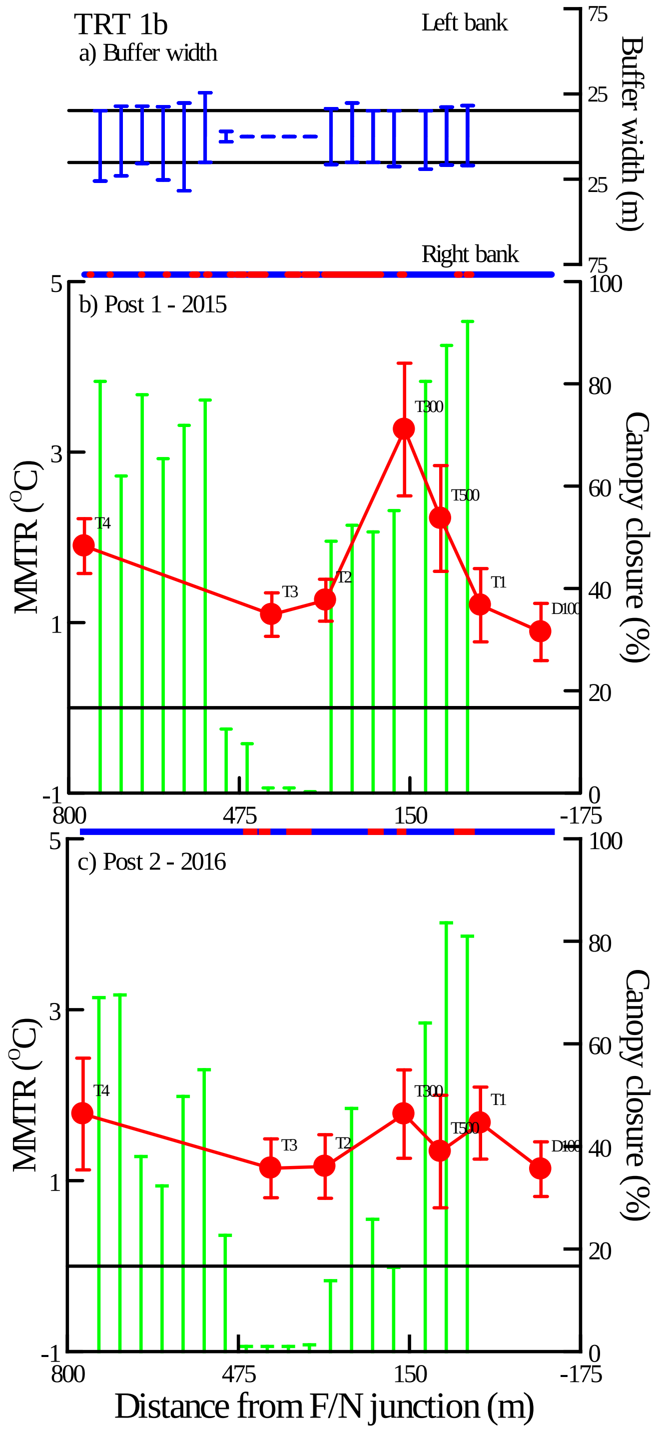

Figure 4-11. Site TRT1b buffer width, canopy closure, July mean monthly temperature response (MMTR), and wetted extent. Panel a): Buffer widths plotted by distance from F/N junction. Dashed horizontal lines indicate 50-ft (15-m) width. Panel b): First year post-harvest. Blue and red line at top of graph shows surface water and dry reaches, respectively. July MMTR with 95% confidence intervals are shown in red with locations labeled. Dashed horizontal line indicates MMTR = 0. Canopy closure is shown in green. Panel c): Second year post-harvest (same variables as Panel b).

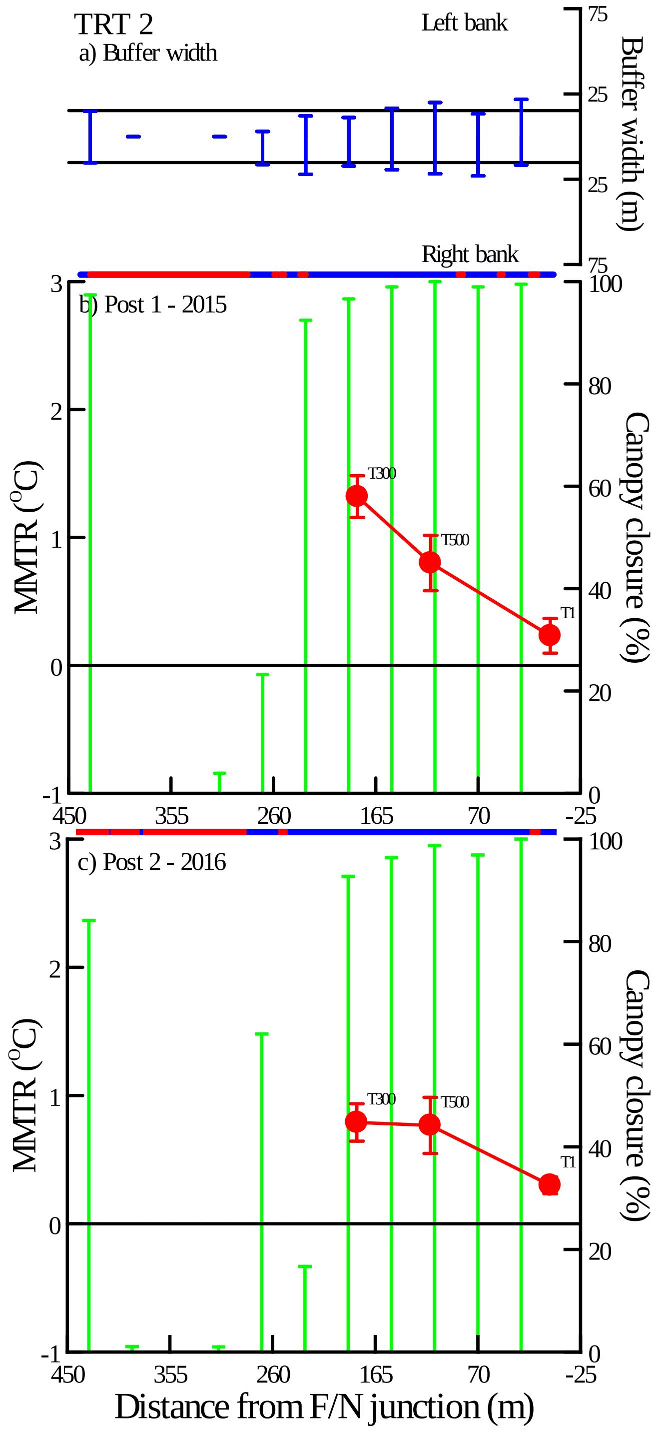

Figure 4-12. Site TRT2 buffer width, canopy closure, July mean monthly temperature response (MMTR), and wetted extent. Panel a): Buffer widths plotted by distance from F/N junction. Dashed horizontal lines indicate 50-ft (15-m) width. Panel b): First year post-harvest. Blue and red line at top of graph shows surface water and dry reaches, respectively. July MMTR with 95% confidence intervals are shown in red with locations labeled. Dashed horizontal line indicates MMTR = 0. Canopy closure is shown in green. Panel c): Second year post-harvest (same variables as Panel b).

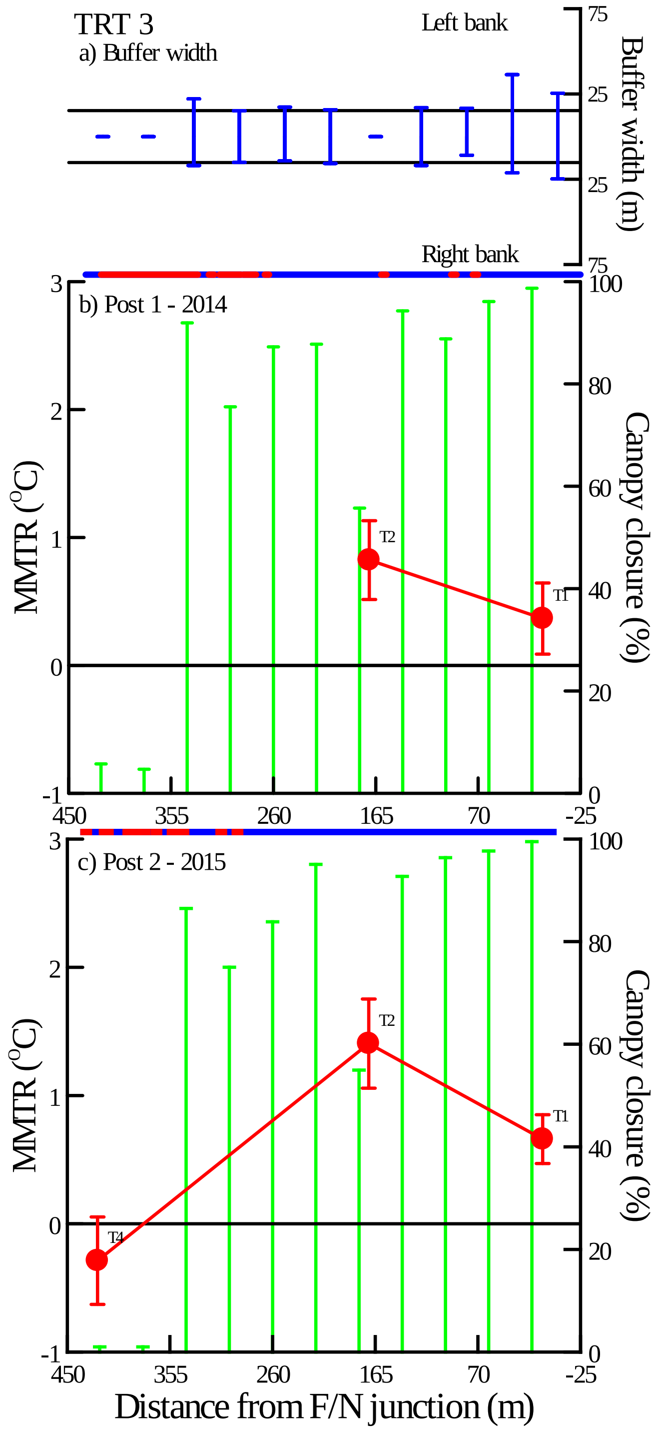

Figure 4-13. Site TRT3 buffer width, canopy closure, July mean monthly temperature response (MMTR), and wetted extent. Panel a): Buffer widths plotted by distance from F/N junction. Dashed horizontal lines indicate 50-ft (15-m) width. Panel b): First year post-harvest. Blue and red line at top of graph shows surface water and dry reaches, respectively. July MMTR with 95% confidence intervals are shown in red with locations labeled. Dashed horizontal line indicates MMTR = 0. Canopy closure is shown in green. Panel c): Second year post-harvest (same variables as Panel b).

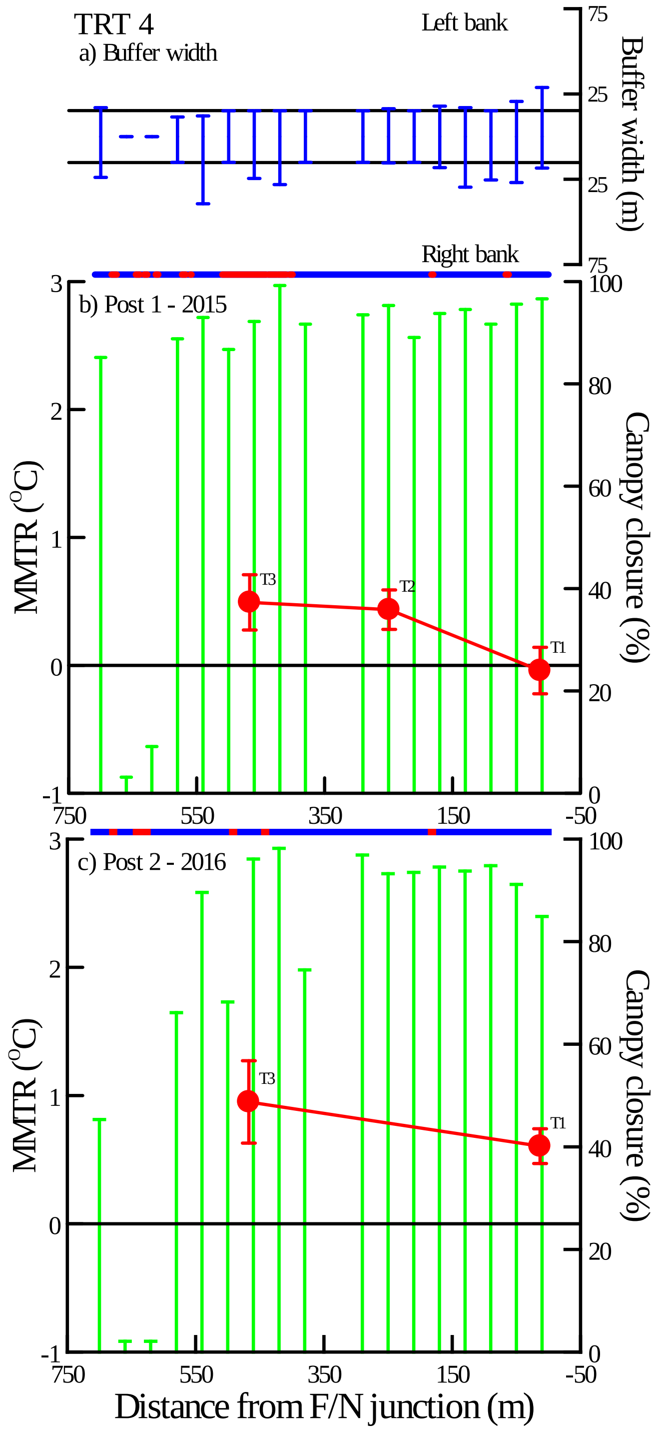

Figure 4-14. Site TRT4 buffer width, canopy closure, July mean monthly temperature response (MMTR), and wetted extent. Panel a): Buffer widths plotted by distance from F/N junction. Dashed horizontal lines indicate 50-ft (15-m) width. Panel b): First year post-harvest. Blue and red line at top of graph shows surface water and dry reaches, respectively. July MMTR with 95% confidence intervals are shown in red with locations labeled. Dashed horizontal line indicates MMTR = 0. Canopy closure is shown in green. Panel c): Second year post-harvest (same variables as Panel b).

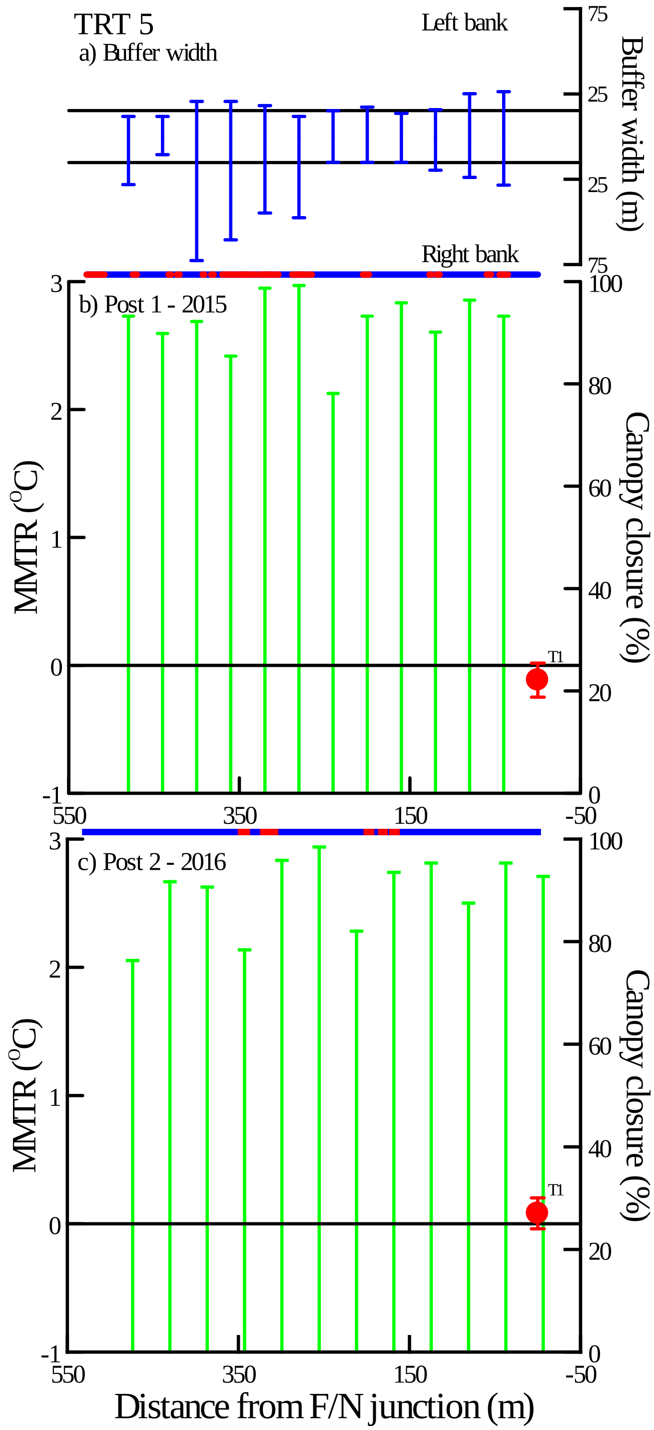

Figure 4-15. Site TRT5 buffer width, canopy closure, July mean monthly temperature response (MMTR), and wetted extent. Panel a): Buffer widths plotted by distance from F/N junction. Dashed horizontal lines indicate 50-ft (15-m) width. Panel b): First year post-harvest. Blue and red line at top of graph shows surface water and dry reaches, respectively. July MMTR with 95% confidence intervals are shown in red with locations labeled. Dashed horizontal line indicates MMTR = 0. Canopy closure is shown in green. Panel c): Second year post-harvest (same variables as Panel b).

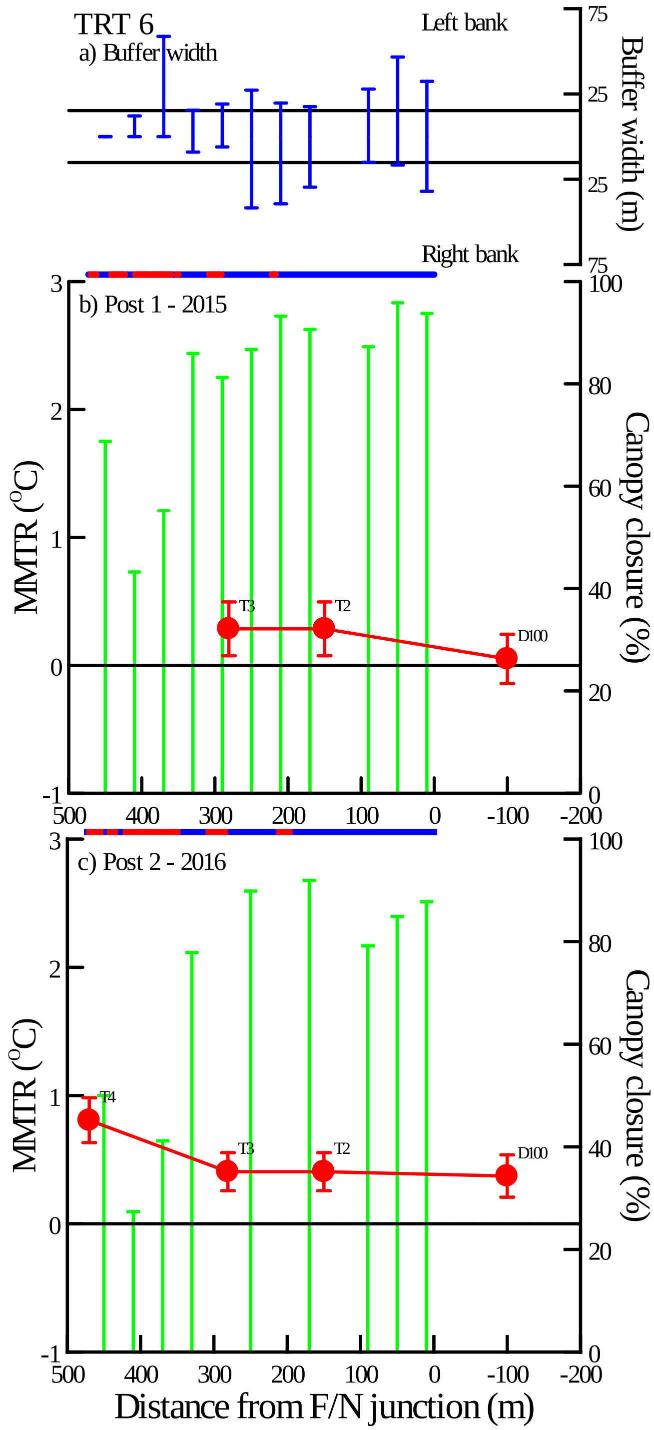

Figure 4-16. Site TRT6 buffer width, canopy closure, July mean monthly temperature response (MMTR), and wetted extent. Panel a): Buffer widths plotted by distance from F/N junction. Dashed horizontal lines indicate 50-ft (15-m) width. Panel b): First year post-harvest. Blue and red line at top of graph shows surface water and dry reaches, respectively. July MMTR with 95% confidence intervals are shown in red with locations labeled. Dashed horizontal line indicates MMTR = 0. Canopy closure is shown in green. Panel c): Second year post-harvest (same variables as Panel b).

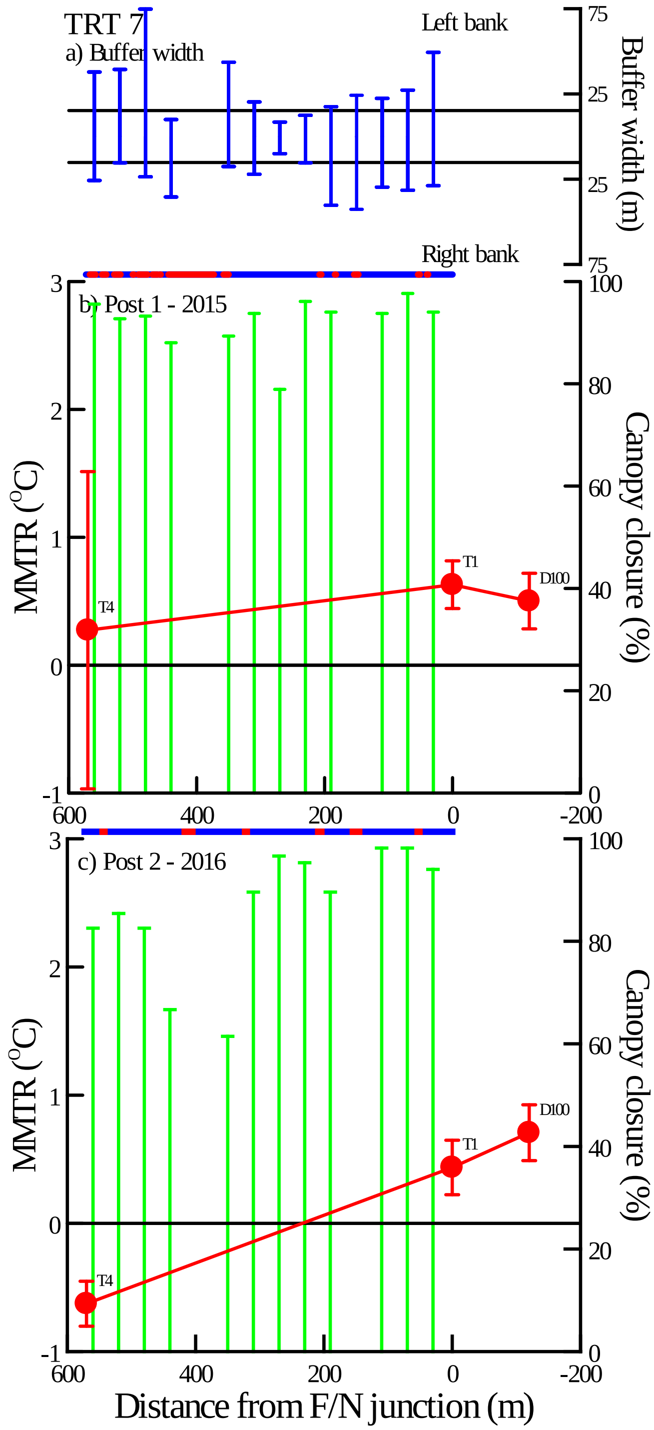

Figure 4-17. Site TRT7 buffer width, canopy closure, July mean monthly temperature response (MMTR), and wetted extent. Panel a): Buffer widths plotted by distance from F/N junction. Dashed horizontal lines indicate 50-ft (15-m) width. Panel b): First year post-harvest. Blue and red line at top of graph shows surface water and dry reaches, respectively. July MMTR with 95% confidence intervals are shown in red with locations labeled. Dashed horizontal line indicates MMTR = 0. Canopy closure is shown in green. Panel c): Second year post-harvest (same variables as Panel b).

4-4.2.6. Buffer Treatment Effects

The GLMM ANOVA for the period effect indicated a high probability the 7DTR changed from pre-harvest to post-harvest (P <0.001; **Table 4-3**). Pair-wise comparisons estimated the 7DTR increased by 0.6°C (P = 0.001), 0.6°C (P = 0.001), and 0.3°C (P = 0.049) compared to pre-harvest in the Post 1, Post 2, and Post 3 years, respectively (**Table 4-15**).

Table 4-15. Pairwise comparisons of the seven-day average temperature response (7DTR) in each post-harvest year relative to the pre-harvest period. P-values were not adjusted for multiple comparisons. SE = standard error; DF = degrees of freedom; C.I. = confidence intervals.

Year	Estimate	SE	DF	t-value	P-value	95% C.I.	
Post 1	0.6	0.16	30.0	-3.89	0.001	0.29	0.95
Post 2	0.6	0.16	30.0	-3.62	0.001	0.25	0.90
Post 3	0.3	0.16	30.0	-2.05	0.049	0.00	0.65

For comparison, we tabulated the maximum July–August seven-day average daily maximum temperatures (7DADM) for the locations used to estimate the buffer treatment effects in **Table 4-15** for each site and year (**Table 4-16**), and calculated the difference between each year and the pre-harvest average 7DADM for each TRT site. At the REF sites, we calculated the difference from the 2012 to 2014 average. A simple calculation of the average pre- to post-harvest change across the TRT sites in Post 1, Post 2, and Post 3 minus the average change in the REF sites in 2015, 2016, and 2017, respectively, yields an estimated change in 7DADM of 0.5°C, 0.3°C, and 0.0°C in Post 1, Post 2, and Post 3, respectively. Although these relatively crude estimates do not take into account that the harvest dates were not consistent for all TRT sites or that REF1 was a poor reference site for stream temperature, the values are within 0.3°C of the buffer treatment effects in **Table 4-15** and demonstrate that our estimates of treatment effects based on the 7DTR are consistent with the observed 7DADM temperatures.

Table 4-16. Seven-day average daily maximum (7DADM) temperature for July–August. For REF sites, Diff is the difference between that year and the mean 2012 to 2014 values for that site. For the treatment sites, Diff is the difference between that year and the average of the pre-harvest values. Blue shading indicates the harvest period and gray shading the post-harvest period. Mean REF is the mean 7DADM. Mean TRT is the mean across all TRT sites except in 2014 when it included only unharvested sites. TRT minus REF is the difference in the mean values for that year.

Year	2012	2013	2014	2015	2016	2017
REF1	13.0	12.2	13.2	13.8	13.9	13.6
Diff				1.0	1.1	0.8
REF2	14.4	14.2	15.0	15.1	14.8	15.3
Diff				0.6	0.2	0.7
REF3	14.3	14.0	14.5	15.2	14.3	15.1
Diff				0.9	0.0	0.8
TRT1	15.4	15.4	17.1	17.5	16.7	17.0
Diff			1.7	2.2	1.3	1.6
TRT2	13.6	13.8	14.4	14.8	14.1	14.1
Diff				0.9	0.2	0.2
TRT3	13.4	13.8	14.6	15.0	14.3	14.3
Diff			1.0	1.4	0.7	0.7
TRT4	14.3	14.4	15.1	15.6	14.8	15.0
Diff				1.0	0.2	0.4
TRT5	13.8	13.9	14.9	15.7	15.2	14.8
Diff			1.1	1.8	1.4	1.0
TRT6	13.4	13.7	14.1	14.7	14.2	14.4
Diff				1.0	0.5	0.7
TRT7	12.0	12.1	12.9	13.3	13.2	13.3
Diff				1.0	0.9	0.9
Mean REF	13.9	13.5	14.3	14.7	14.3	14.7
Mean TRT	13.7	13.9	14.7	15.2	14.6	14.7
TRT minus REF				0.5	0.3	0.0

4-5. DISCUSSION

4-5.1. CANOPY CLOSURE

Canopy closure decreased after harvest, as a function of the proportion of the stream buffered and windthrow within the remaining buffer. The mean pre-harvest canopy closure in this study was 97%, slightly higher than the mean of 89% measured by Schuett-Hames and colleagues (2012) in unharvested Type Np streams in western Washington. Immediately post-harvest, mean canopy closure decreased to 72% in this study and 76% in Schuett-Hames and colleagues (2012), a decrease of 22 and 13 percentage points, respectively. The Hard Rock Study (McIntyre *et al.*

2018), using very similar sites with a mean pre-harvest canopy closure of 96% and the same Forest Practices harvest prescription, reported mean canopy closure of 72% and 67% in Post 1 and Post 2, respectively. Although average shade loss across sites was comparable between the two studies, the range of shade loss across sites was greater in the Soft Rock Study, especially in the first post-harvest year. This was likely related to the wide range in the proportion of the stream buffered.

Mean canopy closure across TRT sites (**Table 4-5**) and within each type of buffer (**Figure 4-2**) continued to decrease into Post 2. The lower Post 2 canopy closure values reflected buffer tree mortality (**Table 4-6**), which was largely driven by windthrow in the first and second winters after harvest. Post 1 measurements were taken in the first summer after harvest. At six of the seven TRT sites, harvest ended March 1 or later so that the first full winter season was between Post 1 and Post 2.

4-5.2. TEMPERATURE

The study streams were cool pre-harvest relative to a random sample of western Washington streams on commercial forest land (Ecology 2019). This was not unexpected because stand age at our sites ranged from 30 to 80 years compared to the random sample of stand ages in Ecology (2019). Our sites were also warmer in summer (13.0 to 14.9°C), on average, than the Willapa Hills sites used in the Hard Rock Study (10.4 to 13.0°C).

4-5.2.1. Reliability of Temperature Data Analysis Methods

Our sites were limited in number and geographic area to those meeting the selection criteria and offered by cooperating landowners. In addition, only two of the three reference sites were suitable for use in the temperature analysis, leaving the temperature results dependent on only two reference sites. In spite of this we have confidence in the results as presented. Every BACI study assumes all sites are on a similar trajectory over time. We were able to demonstrate that the REF sites used in the temperature analysis were stationary, relative to each other, over time. Every BACI study assumes that changes in the TRT sites are due to the treatment. Ideally, the study would have had an equal number of reference and treatment sites, treatments would have been randomly assigned, and pairing of reference with treatment sites would have been ensured before the study began. However, funding approval, site selection, and building relationships between scientists and land managers must fit within the harvest schedule. In the end, the clustering of all sites in southwestern Washington limited the scope of direct inference, but our results suggest the factors driving temperature change were similar to those cited elsewhere in the literature.

We employed a GLS regression to calculate a daily TR for each monitoring location within each site then used the TR to describe the monthly temperature response throughout each treatment site and to estimate the buffer treatment effects on the 7DTR. By pairing each treatment site with a well-matched REF site, the daily TR accounted for interannual variability in weather. In addition, we were able to evaluate the stationarity of our reference sites over the course of the study then use the post-harvest seasonal and temporal patterns at the REF sites to provide context for the observed pattern of higher MMTRs seen at the treatment sites. Only 5.7% of reference site MMTRs exceeded 0.5°C during the three post-calibration years. This is nearly identical to the 5.6% exceedance noted in McIntyre and colleagues (2021). There was one period in summer 2016 when MMTR exceeded 0.5°C for two months. It isn't known why this occurred. The data met the quality assurance criteria and summer 2016 was not a particularly warm or dry period (relative to the study period). However, we did not see comparable events in summer 2015 or 2017 nor was there evidence of similar events in any of the TRT sites during this period, and so we are confident this did not represent a systemic problem in the analyses.

Detecting buffer treatment effects using the linear mixed effects model depends upon the number of sites, magnitude of the response, and variability in the response within treatments. **Table 4-15** suggests a minimum detectable change in 7DTR near 0.3°C. This is lower than 0.8°C suggested by McIntyre and colleagues (2021) using identical methods. The difference is likely due to the greater number of treatment sites in this study (seven sites in one buffer treatment) compared to four sites in each of three buffer treatments in McIntyre and colleagues (2021).

4-5.2.2. Buffer Treatment Effects on Seven-day Average Temperature Response

The buffers required under the current Forest Practices rules did not prevent an average increase in the 7DTR of up to 0.6°C. The magnitude of temperature response immediately after harvest was lower than studies of similar-size streams and buffers. In a study that included sites in both marine sedimentary and basalt lithologies, Janisch and colleagues (2012) reported that mean July–August temperature increased by 1.06°C in a similar (patch cut) treatment. McIntyre and colleagues (2021) reported increases in 7DTR of 1.1, 0.9, and 0.8°C in the first three years post-harvest following the same buffer rules in basalt lithologies. Guenther and colleagues (2014) observed mean July–August temperature increases of 1.64 to 3.00°C at different locations within a partial retention harvest that resulted in a 14% decrease in canopy closure. However, their stream had no harvest along the uppermost stream reach and a greater loss in riparian cover near the bottom of the harvest unit, the inverse of our 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. Bladon and colleagues (2018) reported increases in the median 7DADM (seven-day average daily maximum) temperature of 0.6°C and 1.0°C in sites with buffers of 11 m and 12 m, respectively, underlain by friable lithologies, 0.8°C in a site with a 17 m buffer in mixed lithologies, and 2.4°C and 3.3°C in sites with buffers 8 m wide along 25% and 60% of the stream length, respectively, underlain by mixed lithologies. Although the analyses differed and their lithologies ranged from 100% resistant to 100% friable, the results from their sites with 11 to 17 m wide buffers are similar to our buffer treatment.

The dominant factor affecting the magnitude of temperature increases was the loss of riparian cover. We observed a negative Pearson correlation between July MMTR and canopy closure in both Post 1 ($r = -0.597$, $P = 0.158$) and Post 2 ($r = -0.811$, $P = 0.027$) (**Table 4-11**; **Figure 4-6**). This is similar to that reported by McIntyre and colleagues (2021) in the Hard Rock Study and suggests that shade loss had a large influence. At the site scale, buffer length and percent of stream buffered, although related to canopy closure, were not useful in explaining the temperature response.

We also found moderate positive correlations between July MMTR and the percentage of the stream with surface flow and with the length of wetted channel (surface flow) (**Figure 4-7**). Janisch and colleagues (2012) also reported a correlation on similar-sized streams with fine-grained sediment on marine sedimentary lithologies and suggested the area of exposed surface water may influence temperature response. Another possibility is that a lower percentage of the channel with surface water indicates a greater degree of hyporheic or groundwater influence downstream, which could moderate higher water temperatures.

The Soft Rock Study was originally proposed because of concerns that stream temperature in the Hard Rock Study sites may be less responsive to harvest because the larger stream substrate in the Hard Rock Study sites, relative to the fine-grained Soft Rock Study sites, was associated with greater hyporheic flow and thus would buffer stream temperature changes. Contrary to expectations, the mean effect size from harvest in the Soft Rock Study was less than in the Hard Rock Study's FP treatment. Several factors may have led to a smaller temperature treatment effect in this study. On average, the Soft Rock Study sites had a greater percentage of the stream buffered, post-harvest windthrow was not as severe or widespread, and these streams tended to be deeply incised with steep valley walls, 60% vs. 45% slope in the Hard Rock Study (**Table 2-1**), which provided topographic shading in addition to that provided by the unstable slope buffers. This resulted in higher post-harvest canopy closure in the Soft Rock Study compared with the Hard Rock Study. However, **Figure 4-8** suggests that the basic relationship between temperature change and canopy closure is similar between the studies. Another factor may the degree of hyporheic/groundwater influence on stream temperature. **Figure 4-9** suggests that temperature change in both studies was influenced by the proportion of wetted channel or the length of wetted stream. Buffered sites in both studies tended to fall along the same curve, but the Soft Rock Study streams tended to have lower values of both percent wetted channel and wetted length than the comparable Hard Rock Study 100% and FP treatments, which may account for the smaller temperature increase. Temperature increases were greater at the Hard Rock Study's unbuffered sites across the range of the x-axis, suggesting that the lower canopy closure at these sites was an important factor.

We could not directly evaluate the impacts of site aspect on the temperature response. However, McIntyre and colleagues (2021) noted that within a given treatment, temperature response was greater in southerly-facing sites during the first several post-harvest years of the study. When the two studies are examined together in **Figure 4-9**, the data suggest, with few exceptions, that for any given aspect the magnitude of temperature change was related to the loss of canopy closure. Aspect may be a factor in the difference between the two studies, but our uneven distribution of sites across aspect (only one south-facing Soft Rock Study site) and the large range in the proportion of stream buffered, prevent a direct comparison.

Bladon and colleagues (2018) observed that temperature change after harvest tended to be lower at sites underlain by more permeable lithologies compared to sites on less permeable lithologies. However, this observation is confounded by the fact that their sites with a greater temperature response and on less permeable lithologies were either unbuffered or had narrower buffers (8 m) than the sites on more permeable lithologies (11 to 17 m buffers). We should also note that our assumptions about hyporheic flow were based on stream substrate size while Bladon and colleagues (2018) assumptions were based on lithology. Neither study made direct measurements.

The spring and summer of 2015 was exceptionally warm and dry and this coincided with the first year post-harvest at six of the seven TRT sites. Although this likely affected the stream temperatures at all sites, even the REF sites, we observed no evidence that it affected the calculations of the daily TR, 7DTR, or MMTR. The 2015 7DADM values in all sites tended to be higher than previous years but, as expected, the TRT sites were somewhat higher than the REF sites.

4-5.2.3. Seasonal Effects

Maximum daily temperature increased over much of the year at most locations in all sites after harvest and this persisted for one or more years (**Table 4-9**). McIntyre and colleagues (2018) found the same pattern of temperature increases after harvest in two of their three FP treatment streams and in all eight of their unbuffered and fully buffered streams in the first two years after harvest. Gomi and colleagues (2006), who used a similar method of calculating a daily temperature response, observed a similar pattern of increased temperatures beginning in the spring that peaked in the late summer and extended into the fall in three of their four unbuffered streams and in their only stream with a 10-m buffer. This pattern persisted through all four post-harvest years of their study. MacDonald and colleagues (2003) and Rex and colleagues (2012) observed higher stream temperatures after harvest throughout the ice-free season in sub-boreal, headwater streams with a variable retention buffer similar to ours. Higher temperatures persisted in both studies through the three to five years of monitoring.

Temperature increases at sites with 92 to 100% of the stream length buffered, e.g., TRT4, TRT5, TRT6, and TRT7, tended to be greater in the spring than in the summer months, while less buffered sites, TRT1, TRT2, and TRT3, tended to warm more in the summer. McIntyre and colleagues (2018) observed increased temperatures after harvest in spring and fall at all sites and noted this specific pattern of higher MMTRs in spring than in the summer in two of the twelve Hard Rock Study treatment sites. One was a 100% buffered site with variable width buffers, but generally wider than 50 ft, similar to TRT4–TRT7. At the second site, a portion of the stream was consistently subsurface in the summer months and resurfaced within a dense riparian buffer just above the monitoring location, similar to TRT2. They suggested the higher MMTRs in spring and fall were a result of greater insolation during the leaf-off period (November–February), when canopy closure in standard FP buffers was 20 percentage points lower than in the summer.

TRT2 was the only site where MMTR was moderately elevated (1.4°C) upstream but never exceeded 0.5°C at the F/N break. The stream consistently had no summer surface flow just upstream of the F/N break, which would have mixed warmer water from upstream with cooler hyporheic or groundwater flow, and very little loss of canopy closure below the dry reach. McIntyre and colleagues (2018) reported a similar situation with no post-harvest temperature change in their site with only subsurface flow above the F/N break and high shade retention in the buffer.

4-5.2.4. Downstream Effects on Stream Temperature

We had only two sites where we could measure stream temperature downstream of the F/N break. In both cases the streams were buffered along 96% or more of their length, experienced less than 1.0°C July MMTR at the F/N junction, and a -0.2°C to +0.3°C change in temperature below that. McIntyre and colleagues (2021) observed that the greater the temperature increase within the harvest unit, the greater the cooling observed immediately downstream and it was difficult to detect downstream cooling when the upstream temperature change was less than 1.0°C. The change in MMTR below the harvest unit was small, likely because temperatures were already near equilibrium. In other studies of similar-sized streams, Bladon and colleagues (2018) reported no change in the 7DADM at most locations downstream of the harvest unit in the Trask, Hinkle Creek, and Alsea studies, suggesting cooling below the harvest unit. Their monitoring locations, however, were usually much farther downstream than ours (up to 1100 m), and their harvest treatments varied from ours both in the proportion of the watershed harvested (10 to 65% vs. 73 to 88% in ours) and in the presence or width of riparian buffers. For their four sites with monitoring locations less than 100 m downstream of the harvest unit (FEN1, RUS1, GS2, and BEB1), they reported post-harvest changes of 0.0, 0.1, 0.1, and 1.0°C, respectively. While the results from the two studies are not directly comparable, both suggest rapid cooling or stable temperatures downstream in well-shaded reaches.

Overall, our observations of moderate decreases in riparian shade and an increase in the seven-day average daily temperature response of up to 0.6°C after harvest following current Washington Forest Practices rules are consistent with recent scientific literature of contemporary forest practices. The dominant factor in higher temperatures was the loss of canopy closure. Higher summer low flows after harvest may have increased the sensitivity of streams to warming because of increased exposure to solar radiation or because of lower degree of hyporheic/groundwater influence and we cannot rule out aspect as a factor, but both are clearly of less importance than canopy closure. The temperature response differed from that observed following the same Forest Practices rules in the Hard Rock Study. However, this is likely due to differences in the buffer layout (proportion of stream buffered and buffer width), site topography (steep valley walls), and the extent and severity of post-harvest windthrow, rather than fundamental differences in the response to the loss of canopy closure. Although not a focus of this study, but of importance, was the consistent pattern of higher temperatures in the spring through fall months at nearly all locations in all sites.

This study was restricted to only one lithology within the southwest corner of Washington State. An additional two years of data were collected since this report was initiated. This will extend the duration of the study to nearly one half that of the Hard Rock Study and possibly to the point when stream temperatures return to background levels. The restricted geographic extent of the study suggests informed caution when applying the results to dissimilar watersheds. The spring and summer of 2015 was unusually warm and dry. This may have impacted the results either directly, through warmer air temperatures and lower discharge, or indirectly, through a decrease in the proportion of the channel with surface flow in the REF sites. However, there was no evidence that the atypical weather in 2015 affected the estimated changes in stream temperature after harvest.

4-6. REFERENCES

Alexander, R.B., E.W. Boyer, R.A. Smith, G.E. Schwarz, and R.B. Moore. 2007. The role of headwater streams in downstream water quality. *Journal of the American Water Resources Association* 43:41-59.

Barton, K. 2012. *MuMIn: Multi-model Inference*. R package version 1.7.11. http://CRAN.R-project.org/package=MuMIn.

Beschta, R.L., R.E. Bilby, G.W. Brown, L.B. Holtby, and T.D. Hofstra. 1987. Stream temperature and aquatic habitat: Fisheries and forestry interactions. Pages 191-232 *in* E.O. Salo and T.W. Cundy (eds.). *Streamside Management: Forestry and Fishery Interactions*. Contribution No. 75, Institute of Forest Resources, University of Washington, Seattle, WA.

Beschta, R.L., and R.L. Taylor. 1988. Stream temperature increases and land use in a forested Oregon watershed. *Journal of the American Water Resources Association* 24:19-25.

Bladon, K.D., C. Segura, N.A. Cook, S. Bywater-Reyes, and M. Reiter. 2018. A multicatchment analysis of headwater and downstream temperature effects from contemporary forest harvesting. *Hydrological Processes* 32:293-304.

Brown, G.W. 1969. Predicting temperatures of small streams. *Water Resources Research* 5:68-75.

Brown, G.W., and J.T. Krygier. 1970. Effects of clear-cutting on stream temperature. *Water Resources Research* 6:1133-1139.

Cole, E., and M. Newton. 2013. Influence of streamside buffers on stream temperature response following clear-cut harvesting in western Oregon. *Canadian Journal of Forest Research* 43:993-1005.

Danehy, R.J., C.G. Colson, K.B. Parrett, and S.D. Duke. 2005. Patterns and sources of thermal heterogeneity in small mountain streams within a forested setting. *Forest Ecology and Management* 208:287-302.

Ecology. 2019. Extensive Riparian Status and Trends Monitoring Program—Stream Temperature. Phase I: Westside Type F/S and Type Np Monitoring Project.

Feller, M. 1981. Effects of clearcutting and slashburning on stream temperature in southwestern British Columbia. *Journal of the American Water Resources Association* 17:863-867.

Friberg, N., J. Bergfur, J. Rasmussen, and L. Sandin. 2013. Changing northern catchments: Is altered hydrology, temperature, or both going to shape future stream communities and ecosystem processes? *Hydrological Processes* 27:734-740.

Gomi, T., R.D. Moore, and A.S. Dhakal. 2006. Headwater stream temperature response to clear-cut harvesting with different riparian treatments, coastal British Columbia, Canada. *Water Resources Research* 42:1-11.

Groom, J.D., L. Dent, L.J. Madsen, and J. Fleuret. 2011. Response of western Oregon (USA) stream temperatures to contemporary forest management. *Forest Ecology and Management* 262:1618-1629.

Guenther, S.M., T. Gomi, and R.D. Moore. 2014. Stream and bed temperature variability in a coastal headwater catchment: Influences of surface-subsurface interactions and partial-retention forest harvesting. *Hydrological Processes* 28:1238-1249.

Harris, D. 1977. *Hydrologic Changes After Logging Two Small Oregon Coastal Watersheds*. Water Supply Paper 2037, US Geological Survey, Washington, DC. 31 p.

Holtby, B., and C.P. Newcombe. 1982. A preliminary analysis of logging-related temperature changes in Carnation Creek, British Columbia. Pages 81-99 *in* G.F. Hartman (ed.). *Proceedings of the Carnation Creek Workshop: A 10-year Review*. Canada Department of Fisheries and Oceans, Pacific Biological Station, Nanaimo, BC.

Isaak, D.J., and W.A. Hubert. 2001. A hypothesis about factors that affect maximum summer stream temperatures across montane landscapes. *Journal of the American Water Resources Association* 37:351-366.

Janisch, J.E., S.M. Wondzell, and W.J. Ehinger. 2012. Headwater stream temperature: Interpreting response after logging, with and without riparian buffers, Washington, USA. *Forest Ecology and Management* 270:302-313.

Johnson, S.L. 2004. Factors influencing stream temperature in small streams: Substrate effects and a shading experiment. *Canadian Journal of Fisheries and Aquatic Sciences* 61:913-923.

Johnson, S.L., and J.A. Jones. 2000. Stream temperature responses to forest harvest and debris flows in western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences* 57:30-39.

Kenward, M.G., and J.H. Roger. 1997. Small sample inference for fixed effects from restricted maximum likelihood. *Biometrics* 53:983-997.

Kibler, K.M., A. Skaugset, L.M. Ganio, and M.M. Huso. 2013. Effect of contemporary forest harvesting practices on headwater stream temperatures: Initial response of the Hinkle Creek catchment, Pacific Northwest, USA. *Forest Ecology and Management* 310:680-691.

Leach J.A., R.D. Moore, S. Hinch, and T. Gomi. 2012. Estimation of forest harvesting-induced stream temperature changes and bioenergetics consequences for cutthroat trout in a coastal stream in British Columbia. *Aquatic Sciences* 74:427-441.

Lemmon, P.E. 1956. A spherical densiometer for estimating forest overstory density. *Forest Science* 2:314-320.

MacDonald, J.S., E.A. MacIsaac, and H.E. Herunter. 2003. The effect of variable-retention riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of British Columbia. *Canadian Journal of Forest Research* 33:1371-1382.

Magee, L. 1990. R^2 measures based on Wald and likelihood ratio joint significance tests. *American Statistician* 49:450-253.

McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn (technical coordinators). 2018. *Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington*. Cooperative Monitoring, Evaluation, and Research Report, CMER 18-100. Washington Department of Natural Resources, Olympia, WA. 890 p.

McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D.E. Schuett-Hames, R. Ojala-Barbour, G. Stewart, and T. Quinn (technical coordinators). 2021. *Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington—Phase 2 (9 Years after Harvest)*. Cooperative Monitoring, Evaluation, and Research Report CMER 2021.07.27, Washington State Forest Practices Adaptive Management Program, Washington Department of Natural Resources, Olympia, WA.

Moore, R.D., D.L. Spittlehouse, and A. Story. 2005a. Riparian microclimate and stream temperature response to forest harvesting: A review. *Journal of the American Water Resources Association* 41:813-834.

Moore, R.D., P. Sutherland, T. Gomi, and A. Dhakal. 2005b. Thermal regime of a headwater stream within a clear-cut, coastal British Columbia, Canada. *Hydrological Processes: An International Journal* 19:2591-2608.

Pinheiro, J., and D. Bates. 2000. *Mixed-Effects Models in S and S-PLUS (Statistics and Computing)*. Springer, New York, NY.

Pinheiro, J., D. Bates, S. DebRoy, and D. Sarkar. 2018. *nlme: Linear and Nonlinear Mixed Effects Models*. R package version 3.1-137. <URL:https://CRAN.R-project.org/package=nlme>.

Poole, G.C., and C.H. Berman. 2001. Pathways of human influence on water temperature dynamics in stream channels. *Environmental Management* 27:787-802.

R Core Team. 2018. *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Rex, J.F., D.A. Maloney, P.N. Krauskopf, P.G. Beaudry, and L.J. Beaudry. 2012. Variable-retention riparian harvesting effects on riparian air and water temperature of sub-boreal headwater streams in British Columbia. *Forest Ecology and Management* 269:259-270.

Richardson, J.S., R.J. Naiman, and P.A. Bisson. 2012. How did fixed-width buffers become standard practice for protecting freshwaters and their riparian areas from forest harvest practices? *Freshwater Science* 31:232-238.

Richter, A., and S.A. Kolmes. 2005. Maximum temperature limits for Chinook, coho, and chum salmon, and steelhead trout in the Pacific Northwest. *Reviews in Fisheries Science* 13:23-49.

Rogers, L.W., and A.G. Cooke. 2007. *The 2007 Washington State Forestland Database*. Prepared for the USDA Forest Service. University of Washington, College of Forest Resources, Seattle, WA. 81 p.

SAS Institute, Inc. 2013. *SAS/STAT User's Guide*. SAS Statistical Institute, Cary, NC.

Schuett-Hames, D., A.E. Pleus, E. Rashin, and J. Matthews. 1999. *TFW Monitoring Program Method Manual for the Stream Temperature Survey*. Washington State Department of Natural Resources and Northwest Indian Fisheries Commission publication TFW-AM9-99-005.

Schuett-Hames, D.E., A. Roorbach, and R. Conrad. 2012. *Results of the Westside Type N Buffer Characteristics, Integrity, and Function Study, Final Report*. Cooperative Monitoring, Evaluation, and Research Report, CMER 12-1201. Washington Department of Natural Resources, Olympia, WA. 93 p.

Som, N.A., N.P. Zégre, L.M. Ganio, and A.E. Skaugset. 2012. Corrected prediction intervals for change detection in paired watershed studies. *Hydrological Sciences Journal* 57:134-143.

Story, A., R.D. Moore, and J.S. Macdonald. 2003. Stream temperatures in two shaded reaches below cutblocks and logging roads: Downstream cooling linked to subsurface hydrology. *Canadian Journal of Forest Research* 33:1383-1396.

Watson, F., R. Vertessy, T. McMahon, B. Rhodes, and I. Watson. 2001. Improved methods to assess water yield changes from paired-catchment studies: Application to the Maroondah catchments. *Forest Ecology and Management* 143:189-204.

Webb, B.W., and Y. Zhang. 1999. Water temperatures and heat budgets in Dorset chalk water courses. *Hydrological Processes* 12:309-321.

Wehrly, K.E., L. Wang, and M. Mitro. 2007. Field-based estimates of thermal tolerance limits for trout: Incorporating exposure time and temperature fluctuation. *Transactions of the American Fisheries Society* 136:365-374.

Werner, L. 2009. *Standard Operating Procedure for Determining Canopy Closure Using a Concave Spherical Densiometer—Model C for the Extensive Riparian Status and Trends Monitoring Program*. ECY EAP SOP 064, Washington Department of Ecology, Lacey, WA. 10 p.

Wipfli, M.S., J.S. Richardson, and R.J. Naiman. 2007. Ecological linkages between headwaters and downstream ecosystems: Transport of organic matter, invertebrates, and wood down headwater channels. *Journal of the American Water Resources Association* 43:72-85.

CHAPTER 5 – DISCHARGE AND SUSPENDED SEDIMENT EXPORT

Greg Stewart, William Ehinger, Stephanie Estrella, and Welles Bretherton

TABLE OF CONTENTS

LIST OF FIGURES

5-1. ABSTRACT

We measured stage, turbidity, and suspended sediment concentration to estimate discharge and suspended sediment export from four non-fish-bearing streams before and after timber harvest following current Forest Practices rules. The sites were located in southwestern Washington State and have been managed for timber production. Two of the sites were harvested following current Forest Practices rules that require a two-sided, 50-ft wide unharvested riparian buffer along at least 50% of the length of a non-fish-bearing stream, while two sites remained unharvested. We were unable to develop discharge prediction equations to predict the response of the treatment sites after harvest, possibly due to unusually low precipitation in the pre-harvest period, shorter than expected pre-treatment calibration periods, and differences in precipitation between sites. The suspended sediment data indicated that the marine sedimentary lithologies sampled in this study were more erodible than the competent lithologies sampled in the companion Hard Rock Study. Both the treatment and reference sites exported more sediment in the post-harvest period, likely due to higher precipitation in the post-harvest period, with the greatest suspended sediment export from a reference site with streamside mass wasting.

5-2. INTRODUCTION

Forest harvest can affect headwater stream hydrology and fluvial sediment transport (Gomi *et al.* 2005; Moore and Wondzell 2005). The removal of forest canopy reduces evaporation from canopy interception and transpiration, which changes the magnitude and timing of water delivery to the soil and affects soil moisture (Lewis *et al.* 2001; Keim and Skaugset 2003; Johnson *et al.* 2007) and snowmelt dynamics (Marks *et al.* 1998; Jones and Post 2004). Forest roads can extend the surface channel network and intercept subsurface flow thereby increasing the surface water volume and the speed at which it enters the channel (Wemple *et al.* 1996; Wemple and Jones 2003). At the same time forest harvest practices are affecting discharge, they may also increase headwater sediment supply by altering a range of processes including road surface erosion, windthrow, and bank erosion (Roberts and Church 1986; Grizzel and Wolff 1998; Araujo *et al.* 2014). The combination of changes in flow and/or sediment supply can affect the frequency and magnitude of sediment transporting events (Gomi *et al.* 2005; Alila *et al.* 2009; Kaufmann *et al.* 2009).

In general, watershed studies from the Pacific Northwest and elsewhere have found that annual water yields increase in the short term then gradually decline following recovery from timber harvest, though the magnitude and timing of change is affected by numerous factors (Bosch and Hewlett 1982; Stednick 1996; Jones and Post 2004; Brown *et al.* 2005; Moore and Wondzell 2005). 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; Harr 1983), 290 to 410 mm in the Oregon Cascades (Harr *et al.* 1982; Harr 1983, 1986), and 360 mm on Vancouver Island (Hetherington 1982). In rain-dominated areas, measurable annual runoff can increase by as much as 6 mm/year for each percent of the basin harvested above some threshold, or -2 to 8 mm/day following 100% forest removal, with strong seasonal variations in the response (Hicks *et al.* 1991; Jones and Post 2004; Moore and Wondzell 2005; Winkler *et al.* 2017).

Although changes in discharge can be expected to affect fluvial transport, sediment routing through a basin is complex. Changes in bedforms, large wood, and other channel features alter hydraulic resistance, shear stress, and in-channel sediment storage (Buffington and Montgomery 1999; Jackson *et al.* 2001; Kaufmann *et al.* 2009). Historically, forest harvest 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).

The objective of this portion of the study was to evaluate changes in discharge magnitude and frequency and suspended sediment export associated with timber harvest following current Forest Practices rules. We were unable, however, to draw strong conclusions about the response of discharge and suspended sediment export from the data collected, and so present the results and limitations of the data as a case study.

5-3. DATA COLLECTION METHODS

Discharge and suspended sediment export monitoring was conducted in four Soft Rock Study sites, two treatments (TRT3 and TRT4) and two references (REF1 and REF2; **Figure 2-2**). The TRT sites were harvested following current Forest Practices rules that require a two-sided, 50-ft wide unharvested riparian buffer along at least 50% of the length of a non-fish-bearing stream while the REF sites remained unharvested (see Chapter 2 – *Study Design*). The sites were located in the Willapa Hills in southwestern Washington State and have been managed for timber production.

At each site, stage height was measured with a pressure transducer in a stilling well in an 18- or 24-inch Montana-style Parshall flume. The data were recorded at 15-minute intervals using a Forest Technology Systems (www.ftsinc.com) Axiom H2 datalogger. We also measured stage height with a staff gauge on each site visit (approximately 6-week intervals) and used this measurement to correct for drift in the pressure transducer's stage height measurements prior to calculating discharge. Precipitation was not measured.

Turbidity was measured and recorded near the flume at 15-minute intervals using a DTS-12 turbidity sensor and Axiom H2 datalogger. The system was programmed to conduct Turbidity Threshold Sampling (TTS; Lewis and Eads 2009) so that a water sample was collected by an ISCO pump sampler when turbidity exceeded specified thresholds for two consecutive measurements and stage height exceeded baseflow levels. This ensured that water samples were collected across the range of turbidity values during flow events. Turbidity thresholds ranged from 10 to 1,600 Nephelometric Turbidity Units (NTU) on both the rising and falling limbs of the turbidity graph.

The ISCO sample bottles were collected within several days of each storm event and analyzed for suspended sediment concentration (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 sampler 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 the samples were easily identified and excluded.

5-4. ANALYSES AND RESULTS

5-4.1. DISCHARGE

The study design called for two-years of pre-treatment discharge data collection prior to harvest. Discharge measurements started in January 2013 and TRT3 was harvested in December 2013 providing only 303 days of pre-treatment discharge measurements in TRT3 and only a partial winter season when storms are most common (**Figure 5-1**). The other treatment site (TRT4) was harvested in April 2015 providing 1.75 years of pre-treatment data. As it turned out, 2013 and early 2014 was a very dry period. The mean discharge in the pre-treatment period in TRT3 and TRT4 was less than half of the mean in the post-treatment years and annual peak discharges were lower. This had serious ramifications on the intended analyses, as described below.

We analyzed the discharge data using a technique similar to Gomi and colleagues (2006) and Alila and colleagues (2009). This approach which is described in detail in McIntyre and colleagues (2018) involves using generalized least squares (GLS) regression to model the pre-treatment discharge relationship between reference and treatment sites. The GLS models allowed us to account for serial autocorrelation and non-linearity in the discharge relationship. We had difficulty getting good pre-treatments fits so we also tried fitting Generalized Additive Mixed Models (GAMM). GAMM allows the analyst to incorporate non-linear smoothing functions in the models.

Unfortunately, none of the pre-treatment regression models performed well. In TRT3, the best GLS model was an autoregressive moving average model (ARMA) (1,1) and included a 2^{nd} order polynomial relationship with our REF1 discharge and a linear fit to our REF2 discharge and had a pseudo r^2 of 0.70. The fitted model appeared to correlate well at higher discharges, but not at moderate to lower discharges which provided the bulk of the leverage (**Figure 5-2**, top left). When we plotted expected vs. observed discharge in the pre-treatment period we could see that the references failed to capture a small storm in July 2013 and the model seriously overpredicted base-flow discharge in October 2013 (**Figure 5-2**, top right). In TRT4, the best model was an ARMA(3,2) and including a 2^{nd} order polynomial with our REF2 discharge and had a pseudo r^2 of 0.63. This model did a poor job at estimating base flows prior to October 2013 and completely failed to capture the three largest pre-treatment discharge events (**Figure 5-2**, bottom right).

Before-After Control-Impact studies are predicated on the assumption that reference and treatment sites will track each other in the absence of a treatment. Unfortunately, despite significant effort (much of which is not completely documented here) and for reasons that remain unknown, but were likely related to the unusually dry and unexpectedly short pre-harvest period, we were unable to identify any relationships that would allow us to predict what we would have observed in the treatment sites in the absence of a treatment effect.

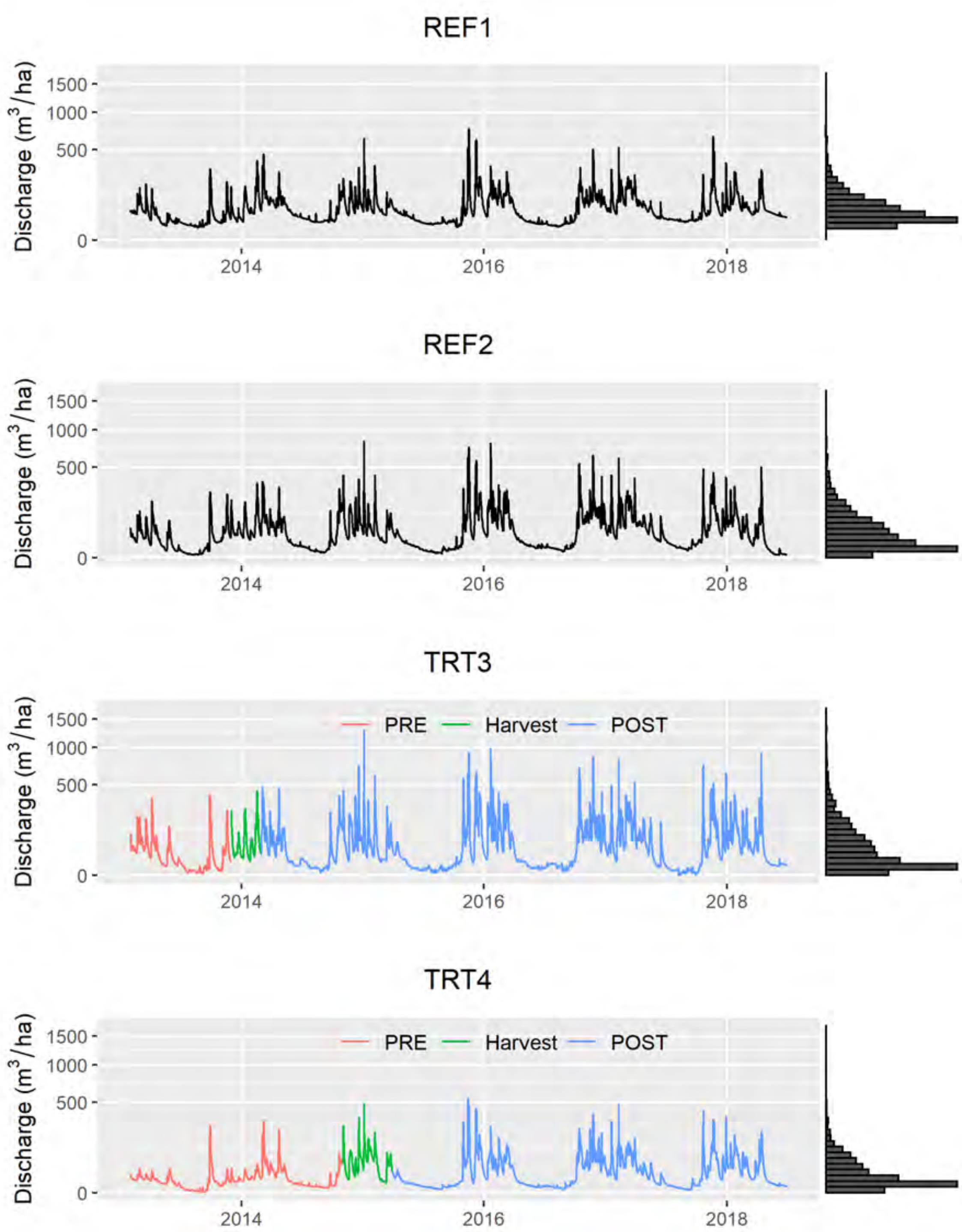

Figure 5-1. Observed daily specific discharge in the two reference (REF) and two treatment (TRT) sites where discharge was measured. Histograms give discharge frequency and color indicates treatment period. Note the short pre-treatment period in TRT3 and the relatively dry conditions in 2013. PRE = pre-harvest period; Harvest = during harvest; POST = post-harvest period.

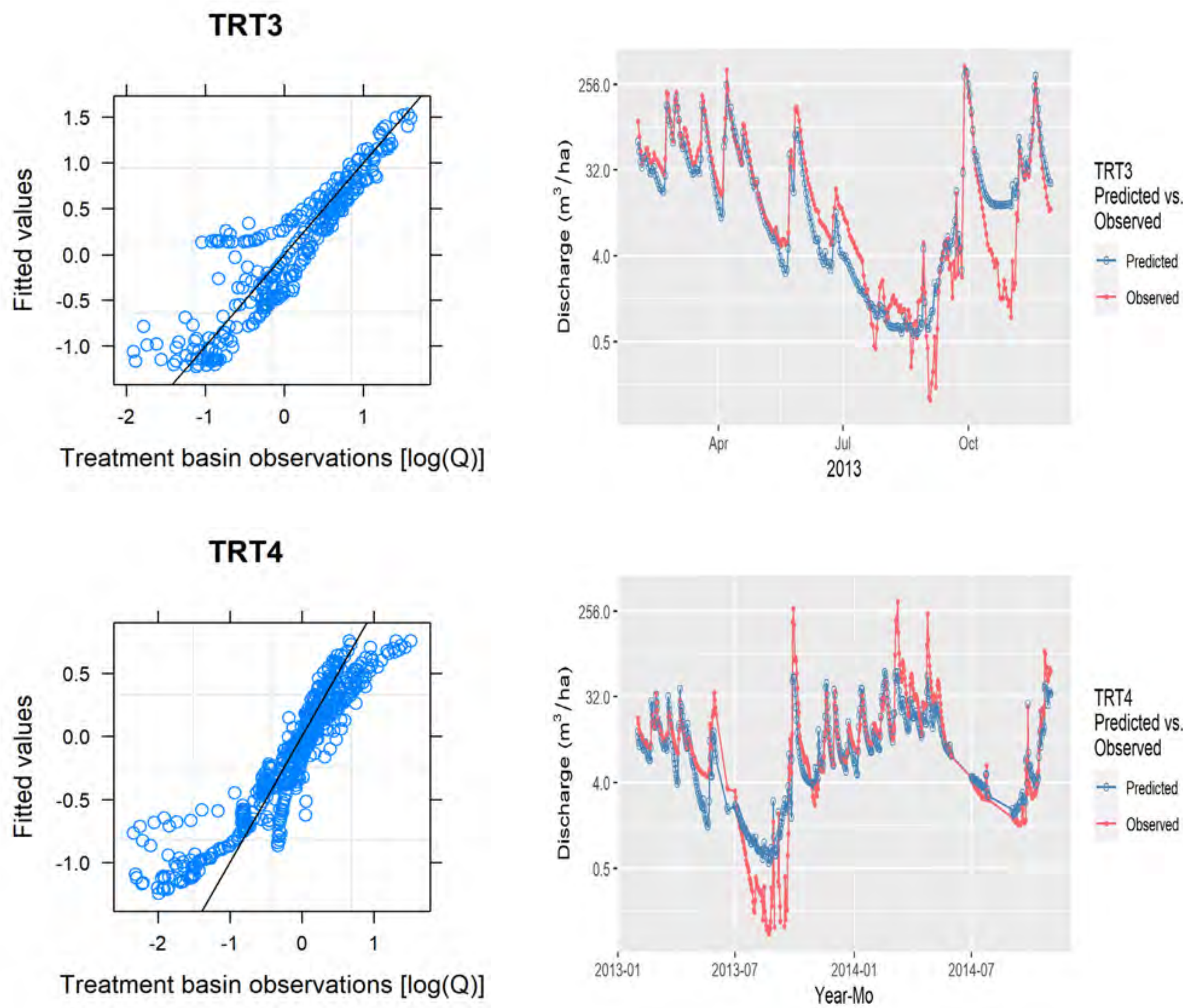

Figure 5-2. Fitted vs. observed (left) and predicted vs. observed (right) specific discharge.

5-4.2. SUSPENDED SEDIMENT EXPORT

At each 15-minute 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 in the data 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.

Each DTS-12 sensor was returned annually to the manufacturer for a calibration check at 11 standards ranging from 0 to 1600 NTUs followed by recalibration. We found persistent issues with drift in all four sensors as indicated by calibration check readings outside the stated tolerance at multiple calibration check standards. Drift outside the manufacturer's tolerance ranged from 0.3% to 11.7% for the 800 NTU standard to 1.3% to 34.3% for the 2 NTU standard.

Drift was worst in 2013, the pre-harvest period, appeared to improve in 2014–2016, only to reappear in 2017. Unfortunately, the observed drift was not consistent in direction nor could we identify when, within each year, it began, making correction impossible without making unsupported assumptions about the timing and rate of drift each year. As a result, turbidity values were not adjusted.

The final set of suspended sediment concentration estimates contained small but obvious (on a log scale) shifts in baseline concentration. The shifts were synchronized with, and clearly the result of, turbidity sensor recalibration. In most cases, the recalibration appeared to shift the lower end of SSC values higher so that estimated minimum concentrations increased by up to an order of magnitude over the course of the study, going from 0.3 mg/L to near 3 mg/L during baseline conditions. To account for this recalibration shift and keep it from affecting our estimates of cumulative suspended sediment export (SSE), we calculated the 20% percentile of SSC for each time period, zeroed out any concentration below the 20th percentile for that period, and then subtracted the 20th percentile from the remaining records in that period. This removed the influence of the lowest 20% of SSC estimates, and equalized the baseline condition across calibration periods. This adjustment was important because more than half of our pre-adjustment SSC estimates were less than 2.6 mg/L. It is important to note that while these shifts had a large effect on estimated SSC when SSC was very low, they affected our peak SSC by less than 0.04%.

The SSC values and the corresponding turbidity values 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. The analysis is based on a rating curve method similar to Bywater-Reyes and colleagues (2017). Under that method, we evaluated whether erosive severity of the streams differed from other nearby streams with a different geology and whether it changed with harvest.

The Soft Rock Study was intended to be a companion to the Hard Rock Study, but in softer lithologies where sediment transport processes might have a bigger effect. The Hard Rock Study measured SSE from two blocks: one in the western Olympic Mountains (OLYM) and one in the northern Willapa Hills (WIL1; McIntyre *et al.* 2018). All of the Hard Rock Study sites were underlain by basalt flows and flow breccias with the exception of the OLYM-100% site which was underlain by tectonic breccia and the WIL1-0% site which was underlain by terraced deposits. The Soft Rock Study sites, in contrast, were composed of marine sedimentary rocks of the Astoria or Lincoln Creek formations. When we plot unit suspended sediment yield against unit discharge, we see that the Soft Rock Study sites had higher yields per unit discharge than the Hard Rock Study WIL1 sites or the OLYM sites. When we fit a linear mixed model that predicts sediment export as a function of discharge by site (with a random effect for month), we found that the Soft Rock Study sites were more erodible on average (**Figure 5-3**).

When we examined cumulative SSE and discharge over the period of record, we found that with the exception of one storm event in March 2014, the top five sediment-producing storms in these sites all occurred during the post-harvest period even in the reference sites (**Figure 5-4**). In addition, we saw that the treatment sites produced less than 100 metric tons/km^2 over the course of the study while the references produced approximately 200 and 390 metric tons/km^2, respectively, with the 2016 water year having the greatest export in all sites except for TRT3

(**Figure 5-4**). Further, we found that SSE is punctuated in time, but those sediment exporting events are not synchronized in time across the sites. Instead, SSE appeared to be driven by site and event specific factors.

We conducted two surveys annually to identify sediment inputs to the streams above the hydrology stations. The first one involved checking for any source of sediment delivery from a major erosion event (e.g., root pit from upturned tree or a mass wasting event). Those surveys were conducted in the spring each year. We also conducted an annual survey of substrate conditions and channel profile at permanent transects every 40 m throughout the four sites. Each transect had two eye screws outside the bankfull channel that were used to attach a reel tape for length and depth measurements.

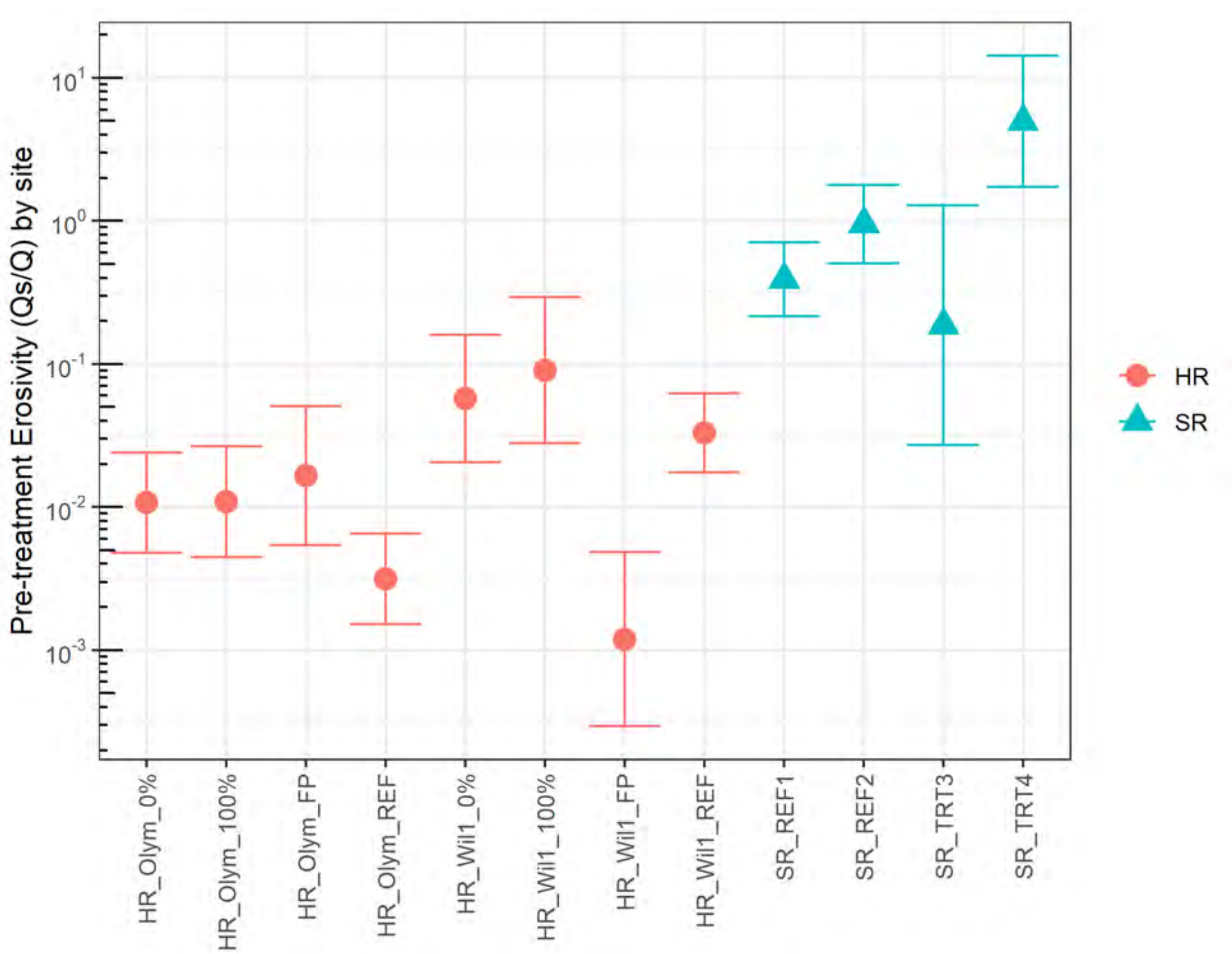

Figure 5-3. Pre-treatment erosivity (suspended sediment export per unit discharge) among Hard Rock and Soft Rock Study (this study) sites. HR = Hard Rock Study; SR = Soft Rock Study; Olym = Hard Rock Study Olympic block; Wil1 = Hard Rock Study Willapa 1 block; REF = reference; TRT = treatment; 100% = 100% treatment; FP = Forest Practices treatment; 0% = 0% treatment.

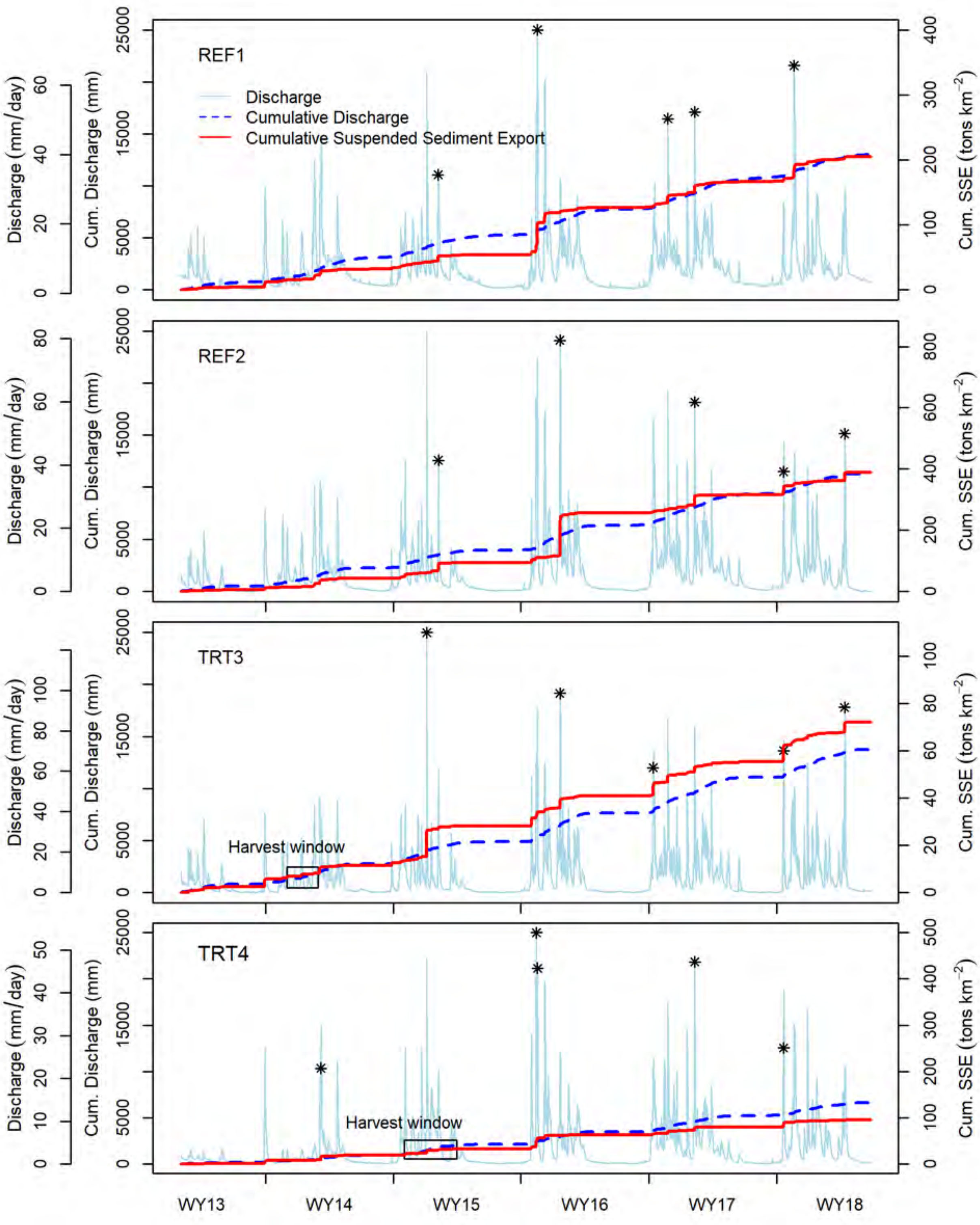

Figure 5-4. Discharge (background, light blue), cumulative discharge (dashed dark blue), and suspended sediment export (solid red) for the period of study. Asterisks denote the five days with the greatest suspended sediment export.

We saw no erosion events during the pre-harvest period but there was evidence of sediment entering the stream in TRT3 and TRT4 during the post-harvest period (5 to 20 m^3 annually). The sediment entering the treatment sites appeared to be from root pits created by fallen trees adjacent to the stream. REF1 had no location specific stream adjacent erosion events during the study, but we did see strong evidence of mass wasting in REF2. In 2015, 2016, and 2017 we estimated that 54, 145, and 264 m^3 of sediment was delivered to the REF2 stream channel in each of the respective years. These mass wasting events all originated along an area of unstable valley wall located 100 to 160 m upstream of the flume. Our sediment delivery estimates indicate that the amount of sediment that entered REF2 from a series of mass wasting events was significantly greater than sediment delivery from windthrow in the treatment sites (**Figure 5-5**).

We tried to measure aggradation and degradation of sediment in the channels at the channel profile transects throughout the flume sites. We were unable to see any trends in these data. The areas that had stable conditions showed little change whereas unstable areas (like windthrow or the mass wasting site in REF2) disrupted our permanent plots, preventing any measurements. We did see evidence of increased woody debris in the channel in the treatment sites after harvest (e.g., **Figure 5-6**) but we were unable to measure any sediment retention in the channels at the treatment sites due to the disruption of the channel profile and sediment transects.

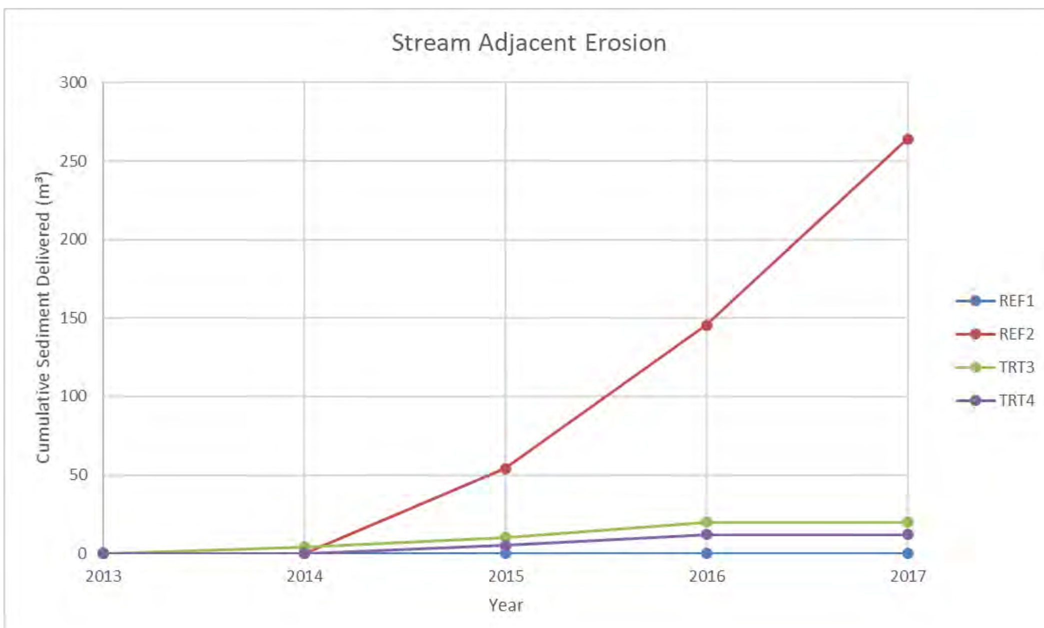

Figure 5-5. Estimated cumulative sediment delivery (m^3) per year from point sources.

Figure 5-6. Windthrow in TRT4.

5-5. CONCLUSIONS

It is difficult to draw any solid conclusions about the rule effectiveness with respect to discharge or suspended sediment export using the data collected in this study. Precipitation was not measured in any of the study sites and despite much effort we were unable to identify any specific reasons why we were unsuccessful in developing good discharge prediction equations for the treatment sites. However, it is likely that the relative lack of rain in the pre-treatment period, shorter than expected pre-treatment calibration periods, and distance between study sites contributed.

The suspended sediment data did show that the softer lithologies sampled as part of this study were more erodible than the competent lithologies sampled in the companion Hard Rock Study, but the data do not support any strong conclusions regarding Forest Practices rule effectiveness. As Arismendi and colleagues (2017) show, flow, turbidity, and SSC tend to be weakly correlated in low order streams like these, and SSC is highly influenced by local conditions and events. All the sites, including the references, exported more sediment in the post-treatment period. The two sites with the greatest suspended sediment export were the two references, with REF2 exporting twice as much as REF1 and approximately four-fold as much as either treatment site. The relatively large amount of suspended sediment export in REF2 was likely associated with streamside mass wasting. Windthrow-driven sediment delivery was observed in the two treatment sites, but the magnitude of that sediment delivery was estimated to be much less than the amount of sediment delivered by a single mass-wasting feature in REF2.

5-6. REFERENCES

Alila, Y., P.K. Kuras, M. Schnorbus, and R. Hudson. 2009. Forests and floods: A new paradigm sheds light on age-old controversies. *Water Resources Research* 45:W08416.

Araujo, H.A., A. Page, A.B. Cooper, J. Venditti, E. MacIsaac, M.A. Hassan, and D. Knowler. 2014. Modelling changes in suspended sediment from forest road surfaces in a coastal watershed of British Columbia. *Hydrological Processes* 28:4914-4927.

Arismendi, I., J.D. Groom, M. Reiter, S.L. Johnson, L. Dent, M. Meleason, A. Argerich, and A.E. Skaugset. 2017. Suspended sediment and turbidity after road construction/improvement and forest harvest in streams of the Trask River Watershed Study, Oregon. *Water Resources Research* 53:6763-6783.

Bosch, J.M., and J.D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology* 55:3-23.

Brown, A.E., L. Zhang, T.A. McMahon, A.W. Western, and R.A. Vertessy. 2005. A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. *Journal of Hydrology* 310:28-61.

Buffington, J.M., and D.R. Montgomery. 1999. Effects of sediment supply on surface textures of gravel-bed rivers. *Water Resources Research* 35:3523-3530.

Bywater-Reyes, S., C. Segura, and K.D. Bladon. 2017. Geology and geomorphology control suspended sediment yield and modulate increases following timber harvest in temperate headwater streams. *Journal of Hydrology* 548:754-769.

Gomi, T., R.D. Moore, and A.S. Dhakal. 2006. Headwater stream temperature response to clear-cut harvesting with different riparian treatments, coastal British Columbia, Canada. *Water Resources Research* 42:1-11.

Gomi, T., R.D. Moore, and M.A. Hassan. 2005. Suspended sediment dynamics in small forest streams of the Pacific Northwest. *Journal of the American Water Resources Association* 41:877-898.

Grizzel, J.D., and N. Wolff. 1998. Occurrence of windthrow in forest buffer strips and its effect on small streams in northwest Washington. *Northwest Science* 72:214-223.

Harr, R.D. 1983. Potential for augmenting water yield through forest practices in western Washington and western Oregon. *Journal of the American Water Resources Association* 19:383-393.

Harr, R.D. 1986. Effects of clearcutting on rain-on-snow runoff in western Oregon: A new look at old studies. *Water Resources Research* 22:1095-1100.

Harr, R.D., W.C. Harper, J.T. Krygier, and F.S. Hsieh. 1975. Changes in storm hydrographs after road building and clear-cutting in the Oregon Coast Range. *Water Resources Research* 11:436-444.

Harr, R.D., A. Levno, and R. Mersereau. 1982. Streamflow changes after logging 130-year-old Douglas fir in two small watersheds. *Water Resources Research* 18:637-644.

Hetherington, E.D. 1982. A first look at logging effects on the hydrologic regime of Carnation Creek Experimental Watershed. Pages 45-63 *in* G.F. Hartman (ed.). *Proceedings of the Carnation Creek Workshop: A Ten Year Review*. Malaspina College, Nanaimo, BC.

Hicks, B.J., R.L. Beschta, and R.D. Harr. 1991. Long-term changes in streamflow following logging in western Oregon and associated fisheries implications. *Water Resources Bulletin* 27:217-226.

Jackson, C.R., C.A. Sturm, and J.M. Ward. 2001. Timber harvest impacts on small headwater stream channels in the coast ranges of Washington. *Journal of the American Water Resources Association* 37:1533-1549.

Johnson, A.C., R.T. Edwards, and R. Erhardt. 2007. Ground-water response to forest harvest: Implications for hillslope stability. *Journal of the American Water Resources Association* 43:134-147.

Jones, J.A., and D.A. Post. 2004. Seasonal and successional streamflow response to forest cutting and regrowth in the northwest and eastern United States. *Water Resources Research* 40:W05203.

Kaufmann, P.R., D.P. Larsen, and J.M. Faustini. 2009. Bed stability and sedimentation associated with human disturbances in Pacific Northwest streams. *Journal of the American Water Resources Association* 45:434-459.

Keim, R.F., and A.E. Skaugset. 2003. Modelling effects of forest canopies on slope stability. *Hydrological Processes* 17:1457-1467.

Kemp, P., D. Sear, A. Collins, P. Naden, and I. Jones. 2011. The impacts of fine sediment on riverine fish. *Hydrological Processes* 25:1800-1821.

Klein, R.D., J. Lewis, and M.S. Buffleben. 2012. Logging and turbidity in the coastal watersheds of northern California. *Geomorphology* 139–140:136-144.

Lewis, J., and R. Eads. 2009. *Implementation Guide for Turbidity Threshold Sampling: Principles, Procedures, and Analysis.* General Technical Report PSW-GTR-212. US Forest Service, Pacific Southwest Research Station, Albany, CA. 87 p.

Lewis, J., S.R. Mori, E.T. Keppeler, and R.R. Ziemer. 2001. Impacts of logging on storm peak flows, flow volumes, and suspended sediment loads in Caspar Creek, California. Pages 85-125 *in* M.S. Wigmosta and S.J. Burges (eds.). *Land Use and Watersheds: Human Influence on Hydrology and Geomorphology in Urban and Forest Areas, Volume 2*. American Geophysical Union, Washington, DC.

MacDonald, J.S., P.G. Beaudry, E.A. MacIsaac, and H.E. Herunter. 2003. The effects of forest harvesting and best management practices on streamflow and suspended sediment concentrations during snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada. *Canadian Journal of Forest Research* 33:1397-1407.

Marks, D., J. Kimball, D. Tingey, and T. Link. 1998. The sensitivity of snowmelt processes to climate conditions and forest cover during rain-on-snow: A case study of the 1996 Pacific Northwest flood. *Hydrological Processes* 12:1569-1587.

McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn (technical coordinators). 2018. *Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington*. Cooperative Monitoring, Evaluation, and Research Report, CMER 18-100. Washington Department of Natural Resources, Olympia, WA. 890 p.

Moore, R.D., and S.M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: A review. *Journal of the American Water Resources Association* 41:763-784.

Reiter, M., J.T. Heffner, S. Beech, T. Turner, and R.E. Bilby. 2009, Temporal and spatial turbidity patterns over 30 years in a managed forest of western Washington. *Journal of the American Water Resources Association* 45:793-808.

Roberts, R.G., and M. Church. 1986. The sediment budget in severely disturbed watersheds, Queen Charlotte Ranges, British Columbia. *Canadian Journal of Forest Research* 16:1092-1106.

Stednick, J.D. 1996. Monitoring the effects of timber harvest on annual water yield. *Journal of Hydrology* 176:79-95.

Stoddard, M.A., and J.P. Hayes. 2005. The influence of forest management on headwater stream amphibians at multiple spatial scales. *Ecological Applications* 15:811-823.

Wemple, B.C., and J.A. Jones. 2003. Runoff production on forest roads in a steep, mountain catchment. *Water Resources Research* 39:1220.

Wemple, B.C., J.A. Jones, and G.E. Grant. 1996. Channel network extension by logging roads in two basins, western Cascades, Oregon. *Journal of the American Water Resources Association* 32:1195-1207.

Wilkins, R.N., and N.P. Peterson. 2000. Factors related to amphibian occurrence and abundance in headwater streams draining second-growth Douglas-fir forests in southwestern Washington. *Forest Ecology and Management* 139:79-91.

Winkler, R., D. Spittlehouse, and S. Boon. 2017. Streamflow response to clear-cut logging on British Columbia's Okanagan Plateau. *Ecohydrology* 10:e1836.

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CHAPTER 6 – NITROGEN EXPORT

William Ehinger, Stephanie Estrella, and Welles Bretherton

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

6-1. ABSTRACT

We monitored stream discharge and the concentrations of total nitrogen (N) and nitrate-N in four streams in a Before-After Control-Impact study intended to estimate nitrogen export from non-fish-bearing streams before and after timber harvest following current Forest Practices rules. All sites were located in southwestern Washington State and have been managed for timber production. Two streams were not harvested (REF) during the study and two were clearcut harvested (TRT) following the current Forest Practices rules that require leaving a two-sided, 50-ft wide unharvested riparian buffer along at least 50% of the length of a non-fish-bearing stream.

We observed higher concentrations of total-N ($P < 0.05$) and nitrate-N ($P < 0.02$) after harvest in the TRT sites relative to the REF sites and higher total discharge after harvest in all sites. Export of total-N and nitrate-N increased after harvest at all sites, but increased more at the TRT sites because of the higher post-harvest concentrations. The change in export in the two TRT sites varied and was related to the proportion of the stream buffered (less buffered = greater increase in export) and to the unusually dry weather and low stream discharge in the pre-harvest period. Pre-harvest nitrogen concentration and export was within the range measured in the Hard Rock Study as were the changes after harvest.

6-2. INTRODUCTION

Nitrogen export from streams draining into Puget Sound and coastal estuaries is of special interest to state environmental regulatory authorities. Excessive nitrogen loads can encourage high primary production in marine 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. Although targets for nitrogen are not specified within the Habitat Conservation Plan, better estimates of the effect of contemporary forest practices on N loads will be useful for managing the quality of Washington's coastal waters. This study was an opportunity to provide these estimates.

Forest practices may influence stream chemistry through changes in (1) geological weathering, (2) precipitation chemistry, hydrology, and temperature, (3) chemical uptake and transformation through terrestrial biological processes, (4) physical and chemical reactions in soils, and (5) processes within aquatic ecosystems (Feller 2005). Of these, the last three are the most important with respect to the effects on N concentrations in streams and subsequent export from the watershed. Clearcut harvest and vegetation control reduces canopy interception of rainfall and evapotranspiration of soil water, leading to an increase in runoff (Likens *et al.* 1970; Bosch and Hewlett 1982; Harr 1983; Stednick 1996; Feller 2005; Moore and Wondzell 2005). Nitrate concentrations in soil water and streams may increase with a decrease in uptake resulting from vegetation removal (Dahlgren 1998; Feller *et al.* 2000), an increase in microbial nitrification from warmer soil temperatures (Feller 2005; Boczulak *et al.* 2015), slash burning (Fredriksen

1971; Stark 1979; Feller and Kimmins 1984; Gravelle *et al.* 2009), and growth of nitrogen-fixing alder (Feller 2005). Forest harvest may also adversely affect soil mycorrhizae, at least temporarily (Harvey *et al.* 1980; Hagerman *et al.* 1999), which may further decrease uptake. An increase in nitrate concentration combined with an increase in runoff may intensify leaching of nutrients from the soil and export of nutrients downstream.

Numerous studies have reported increases in stream concentration of nitrate-N, especially during the first fall freshets, and increases in nitrate-N export post-harvest (Likens *et al.* 1970; Brown *et al.* 1973; Feller and Kimmins 1984; Harr and Fredricksen 1988; Dahlgren 1998; Gravelle *et al.* 2009; Schelker *et al.* 2016). Generally, the higher the proportion of a watershed harvested, the greater the increase in concentrations of soil and soil water nitrate (Feller *et al.* 2000) and in concentrations of stream nitrate (Stark 1979; Martin *et al.* 1984; Fowler *et al.* 1988; Tiedemann *et al.* 1988). Instream processing of N 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.

In the Hard Rock Study (McIntyre *et al.* 2018), total-N and nitrate-N export increased in all treatments in the two-year post-harvest period, with the greatest change in the sites with a higher proportion of watershed harvested. When the sites were revisited several years after harvest, there was no longer a relationship in nitrogen concentration and export to buffer treatment and only one site had recovered to pre-harvest export rates (McIntyre *et al.* 2021). The objective of the Soft Rock Study was to measure the response of total-N and nitrate-N concentration in and export from stream basins with incompetent lithologies before and after timber harvest.

6-3. METHODS

6-3.1. EXPERIMENTAL DESIGN

We used a Before-After Control-Impact design to evaluate post-harvest changes in treatment sites relative to unharvested 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).

6-3.2. STREAMFLOW

For cost and logistical reasons, we limited discharge monitoring to four Soft Rock Study sites, which included two treatments (TRT3 and TRT4) and two references (REF1 and REF2; **Figure 2-2**). At each site, stage height was recorded at 15-minute intervals using a system from Forest Technology Systems (www.ftsinc.com) consisting of a pressure transducer and Axiom H2 datalogger. Stage height was measured within a stilling well in an 18- or 24-in Montana-style Parshall flume. We measured stage height with a staff gauge on each site visit (approximately 6-week intervals), and used this measurement to correct for drift in the pressure transducer's stage height measurements prior to calculating flow.

Our intent was to collect at least two complete years of discharge data for the pre-harvest and the post-harvest periods. However, harvest timing determined the pre- and post-harvest periods and TRT3 was harvested in December 2013 after 11 months and TRT4 in November 2014 after 21 months. As a result, we have only one complete year or less of pre-harvest data and three complete post-harvest years.

6-3.3. WATER SAMPLE COLLECTION AND CHEMICAL ANALYSIS

We employed two concurrent water sample collection procedures to ensure we sampled a wide range of discharge across the year. We manually collected water samples at approximately eight-week intervals from February 2013 to September 2017, unless the site was inaccessible due to weather, road maintenance, or harvest activities. Water was collected at the flow gauging location into acid-washed Nalgene bottles containing concentrated sulfuric acid as a preservative. Sample bottles were cooled to ≤4°C and transported to the lab within 24 hours.

We were unable to manually sample high flow events regularly because of the long distances to and between sites. Instead, we implemented turbidity threshold sampling (TTS; Lewis and Eads 2009) to collect water samples during high flow events across the range of turbidity and flow values. 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 collected into acid-washed Nalgene bottles by an ISCO TM pump sampler when the turbidity value crossed a (rising or falling) threshold and flow exceeded approximately 10 to 20 L/s, depending upon the site. Water samples were retrieved from the pump sampler within one week of the triggering event and processed, as described above for the manually-collected samples.

We analyzed these samples for total-N and nitrate-N and used an approach similar to that used for suspended sediment concentration (see Chapter 5 – *Discharge and Suspended Sediment Export*) to predict total-N and nitrate-N concentrations using the continuous flow and turbidity data.

We were concerned about the effect of biological activity (uptake and transformation of N) while bottles were left in the pump sampler. We independently tested the effect of storing samples for one to four weeks at ambient air temperatures (daily mean 9.9 to 15.1°C) prior to adding preservative and cooling to ≤4°C (see Appendix 9-A in McIntyre *et al.* 2018). We collected four replicate water samples (four sample bottles filled in sequence from a single stream on a single visit) from our study sites in southwest Washington. 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 others 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. 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 or nitrate-N 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 samples were analyzed for total-N and nitrate-N (**Table 6-1**). Total-N and nitrate-N concentrations were determined from unfiltered water samples and represent both particulate and dissolved forms. However, nitrate-N is very soluble. In one study in western Washington, nitrate-N concentration in filtered samples on average 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. We believe our nitrate-N concentration and export estimates are comparable to estimates based on dissolved nitrate-N. All chemical analyses were done by the Washington State Department of Ecology’s Manchester Environmental Laboratory in Port Orchard, Washington.

Atmospheric deposition data were obtained from National Atmospheric Deposition Program site WA14 at the Hoh River Ranger station 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 2019).

Table 6-1. Nitrogen analytical methods.

Analyte	**Method**	**Reporting Limit** (µg/L)
Nitrate-N[a]	4500-NO3-I	10
Total-N[a]	4500-N B	25

[a] APHA (2016) *Standard Methods for the Examination of Water and Wastewater*, 22nd Edition.

6-3.4. NUTRIENT LOAD CALCULATIONS

We used a regression model to empirically estimate nitrogen concentration as a function of discharge. We calculated loads (product of estimated concentration and discharge) of total-N and nitrate-N following the methods of Helsel and Hirsch (2002) except we used discharge data collected at 15-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. We believe the predictive equations using the 15-minute data provided better temporal resolution than with daily mean data. We used **Equation 6-1** to calculate total-N and nitrate-N:

$$Log[N]_i = \beta_0 + \beta_1 LogQ_i + \beta_2 (LogQ)_i{}^2 + \beta_3 sin\frac{c\pi t}{365.25} + \beta_4 cos\frac{c\pi t}{365.25} + \varepsilon_i \qquad (6\text{-}1)$$

where: $Log[N]_i$ is base 10 logarithm of total-N or nitrate-N concentration of the ith sample,

$\beta_0 - \beta_5$ are regression coefficients,

$LogQ$ 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), and

ε_i is an error term.

We developed separate regression models for the pre- and post-harvest periods at both treatment sites (**Table 6-2**) because there was a substantial difference in the regression relationship between the pre- and post-harvest periods. One model was used for the entire period at each of the reference sites. Discharge and the seasonal terms were used in the total-N and nitrate-N models for all sites and all periods.

The adjusted r^2 of the regression models ranged from 0.514 to 0.750 for total-N and 0.456 to 0.683 for nitrate-N except at TRT3 during the pre-harvest period. The adjusted r^2 for these models, based on only 13 observations due to the short dry pre-harvest period, was 0.89. We examined the residuals of each regression model 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 load was calculated as the product of predicted nutrient concentration and flow for each 15-minute record. We assumed that each instantaneous load value applied to the entire preceding 15-minute interval so that the cumulative 15-minute load equaled 600 (seconds) times the instantaneous load (kg/sec). Annual export values were calculated as the sum of these cumulative 15-minute loads for each complete year immediately before the start of timber harvest and each complete year immediately after the end of harvest activities divided by the area of the drainage basin above the flume (units = kg/ha/yr).

All regressions and load calculations were done using SYSTAT 13 statistical software (SYSTAT Software, Inc. 2009).

Table 6-2. Regression models for estimating nutrient concentration. Separate models were developed for pre- and post-harvest periods for total-N and nitrate-N in both buffer treatment sites because the relationship between concentration and discharge changed post-harvest. A single regression model was used in both reference sites for total-N and nitrate-N. SE = standard error.

		Total-N			Nitrate-N		
Treatment	**Period**	**N**	r^2	**SE**	**N**	r^2	**SE**
REF1	All	193	0.533	0.179	193	0.456	0.247
REF2	All	232	0.521	0.187	231	0.470	0.240
TRT3	Pre-	13	0.890	0.078	13	0.887	0.093
	Post-	49	0.605	0.162	49	0.648	0.181
TRT4	Pre-	82	0.750	0.103	82	0.683	0.123
	Post-	65	0.514	0.096	65	0.617	0.090

6-3.5. STATISTICAL ANALYSIS

Our intent was to collect two years of pre-harvest data and at least two years of post-harvest data. However, a three-month delay in the delivery of the equipment and the unexpected early harvest of the TRT3 site in December 2013 resulted in only one year of pre-harvest monitoring and three years of post-harvest. Export from each watershed was highly dependent upon flow, which varied across years, both in quantity and timing. The variables analyzed below are the difference in mean concentration or annual export (harvested site minus reference) between each TRT site and a reference site over the same period. REF2 provided a much better match to runoff in both TRT3 and TRT4, so we used this as the reference site. However, N concentration and export values are presented for all four monitored sites.

We used a generalized linear mixed-effects model (GLMM) with site as a random effect and period (pre- versus post-harvest) as a fixed effect. We used SAS software version 9.4 for GLMM analyses (SAS 2013).

To inform the question:

> What is the magnitude of change in nitrogen (total-N and nitrate-N) concentration and annual export relative to an unharvested reference site following timber harvest?

We evaluated the hypothesis:

$$H_0: (TRT_{pre} - REF_{pre}) = (TRT_{post} - REF_{post}) \tag{6-2}$$

where: TRT is annual export from the treatment site,
REF is annual export from the reference site over the same period, and
pre and post denote pre- and post-harvest periods.

Estimates of the effect size and the associated 95% confidence intervals are presented.

6-4. RESULTS

6-4.1. NITROGEN CONCENTRATION

Total-N and nitrate-N concentrations varied seasonally with the highest concentrations coinciding with the first fall rain events (**Figures 6-1** and **6-2**). Peak nitrogen concentrations and export tended to be lower in calendar year 2013, when there were few high flow events, than in the following years. After harvest concentrations appeared to remain elevated longer into the spring, especially in TRT3. Harvest in TRT3 began in December 2013 so the entire pre-harvest year occurred during an unusually low discharge period. TRT4 was harvested in November 2014, but this low discharge period included the majority of the pre-harvest period at this site as well.

Pre-harvest total-N concentrations varied up to four-fold across sites with mean total-N ranging from 395 µg/L in TRT3 to 1465 µg/L in REF2 (**Table 6-3**). Post-harvest concentration was higher in both TRT sites but the response was variable in the REF sites. We observed a near doubling in concentration in TRT3, which had only 58% of the stream buffered. During this same period, concentration at both REF sites changed by -12% and -25%. Concentration in TRT4, with 92% of the stream buffered, increased less than TRT3, both absolutely (77 ug/L) and proportionally (9%), while response over the same period in the REF sites varied from +8% in REF1 to -11% in REF2.

Nitrate-N typically accounted for more than 75% of the total-N in each sample and so, not surprisingly, nitrate-N concentration and response was very similar to total-N. Mean pre-harvest concentration ranged from 336 µg/L in TRT3 to 1448 µg/L in REF2. After harvest mean concentration increased 337 µg/L in TRT3 and 13 µg/L in TRT4. Over the same time period, nitrate-N concentration varied from a 415 µg/L decrease to a 67 µg/L increase in the REF sites during the post-harvest periods.

The GLMM ANOVA estimated an increase (**Table 6-4**) of 495 µg/L in total-N ($P = 0.046$) and 569 µg/L in nitrate-N ($P = 0.013$) relative to the REF2 site after harvest (**Table 6-5**).

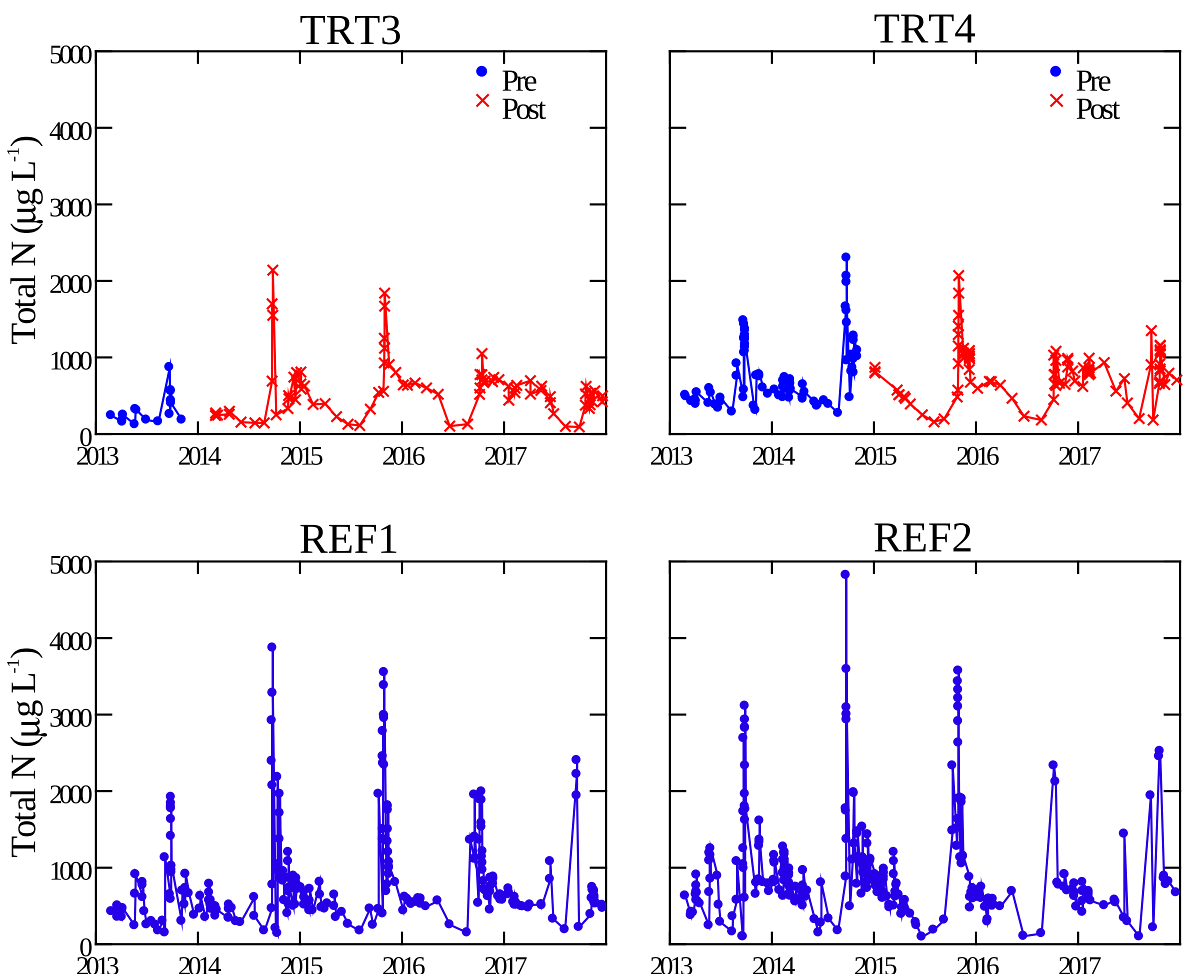

Figure 6-1. Measured total-N concentration over time. The post-harvest period is shown in red for TRT3 and TRT4 sites.

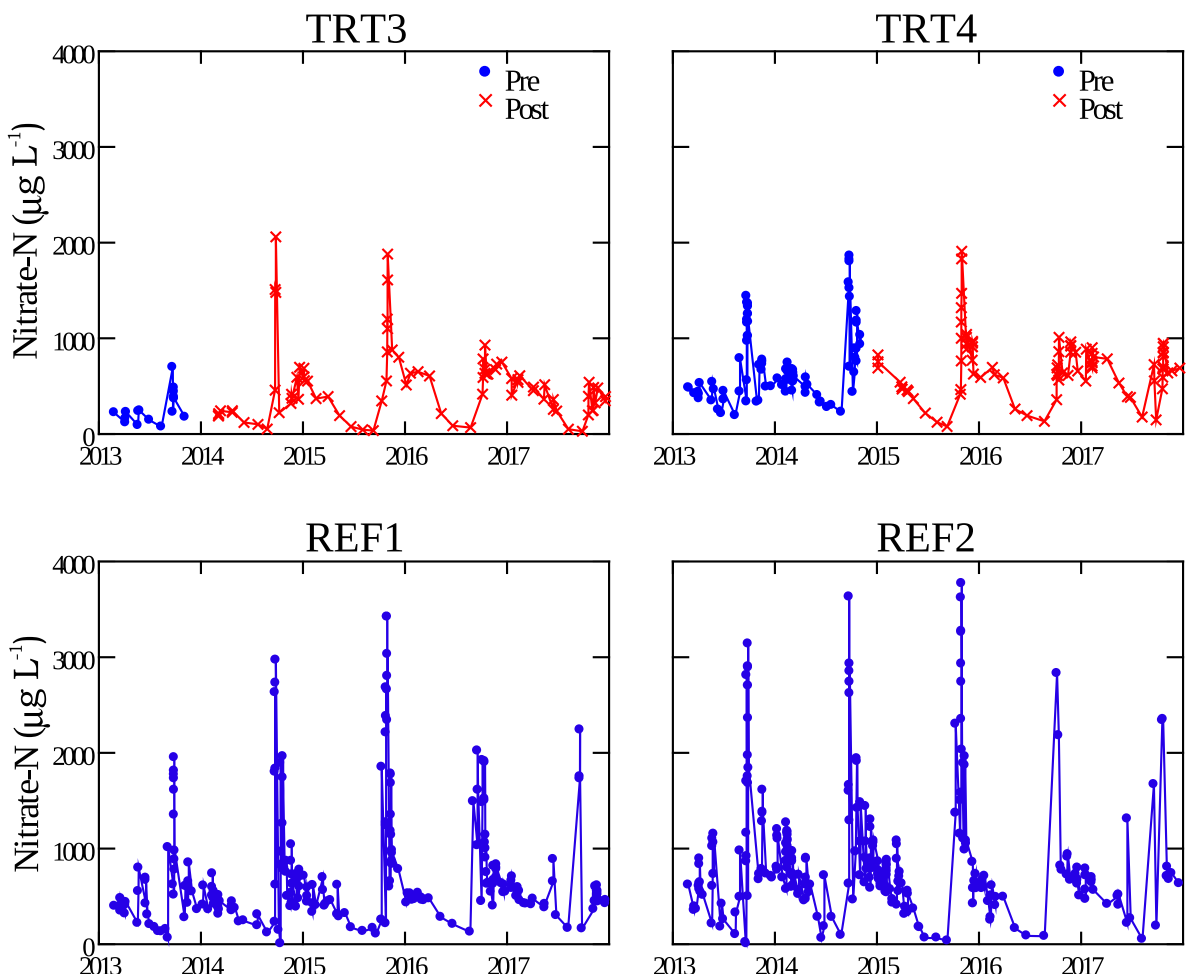

Figure 6-2. Measured nitrate-N concentration over time. The post-harvest period is shown in red for TRT3 and TRT4 sites.

Table 6-3. Mean flow-weighted concentrations (µg/L) for the pre- and post-harvest periods and the difference between periods for total-N and nitrate-N. Harvest timing determined the pre- and post-harvest periods and reference (REF) sites' concentrations are shown corresponding to each treatment (TRT) site's periods. TRT3 Pre = January-December 2013; TRT3 Post = March 2014-February 2017; TRT4 Pre = January 2013-October 2014; TRT4 Post = April 2014-March 2017.

		Treatment		REF1		REF2	
Site	**Period**	**Total-N**	**Nitrate-N**	**Total-N**	**Nitrate-N**	**Total-N**	**Nitrate-N**
TRT3	Pre	395	336	1014	948	1465	1448
	Post	744	673	890	809	1093	1033
	Difference	349	337	-124	-140	-372	-415
TRT4	Pre	894	860	849	774	1204	1165
	Post	971	873	915	840	1078	1022
	Difference	77	13	66	67	-127	-142

Table 6-4. Type 3 test of fixed effects for GLMM ANOVA for changes in concentration and export of total-N and nitrate-N. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Effect	Num DF	Den DF	F-value	P-value
Total-N Concentration				
Period	1	5	7.02	0.0455
Nitrate-N Concentration				
Period	1	5	14.06	0.0133
Total-N Export				
Period	1	5	0	0.9475
Nitrate-N Export				
Period	1	5	0.66	0.4546

Table 6-5. Estimate of post-harvest changes in total-N and nitrate-N concentration (µg/L). SE = standard error; DF = degrees of freedom; C.I. = confidence interval.

					95% C.I.	
Estimate	**SE**	**DF**	**T-value**	**P-value**	**Lower**	**Upper**
Total-N concentration						
495	187	5	-2.65	0.0455	14	976
Nitrate-N concentration						
569	152	5	-3.75	0.0133	179	959

6-4.2. NITROGEN EXPORT

6-4.2.1. Seasonal Patterns

There was a spike in total-N and nitrate-N export at all sites during the first fall freshets likely due to leeching of soluble nitrogen from the forest floor and litter entrainment during high fall flows. Although the monthly exports were higher post-harvest in both TRT sites, they were also higher in the REF sites, largely due to the higher flows in all sites during the post-harvest period (**Figures 6-3** and **6-4**).

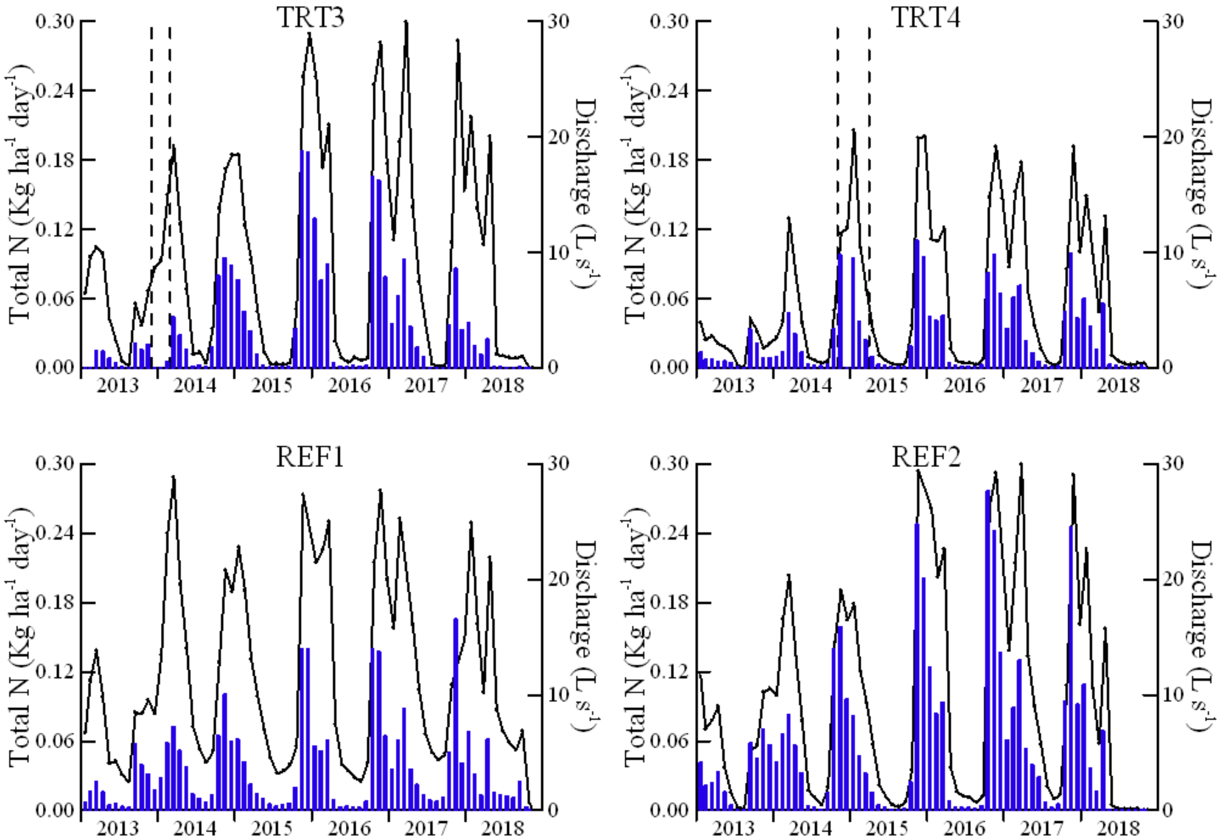

Figure 6-3. Total-N export (left axis; blue bars) and mean monthly stream flow (right axis; black line) from treatment (top) and reference (bottom) sites. The vertical dashed lines bracket the active harvest period.

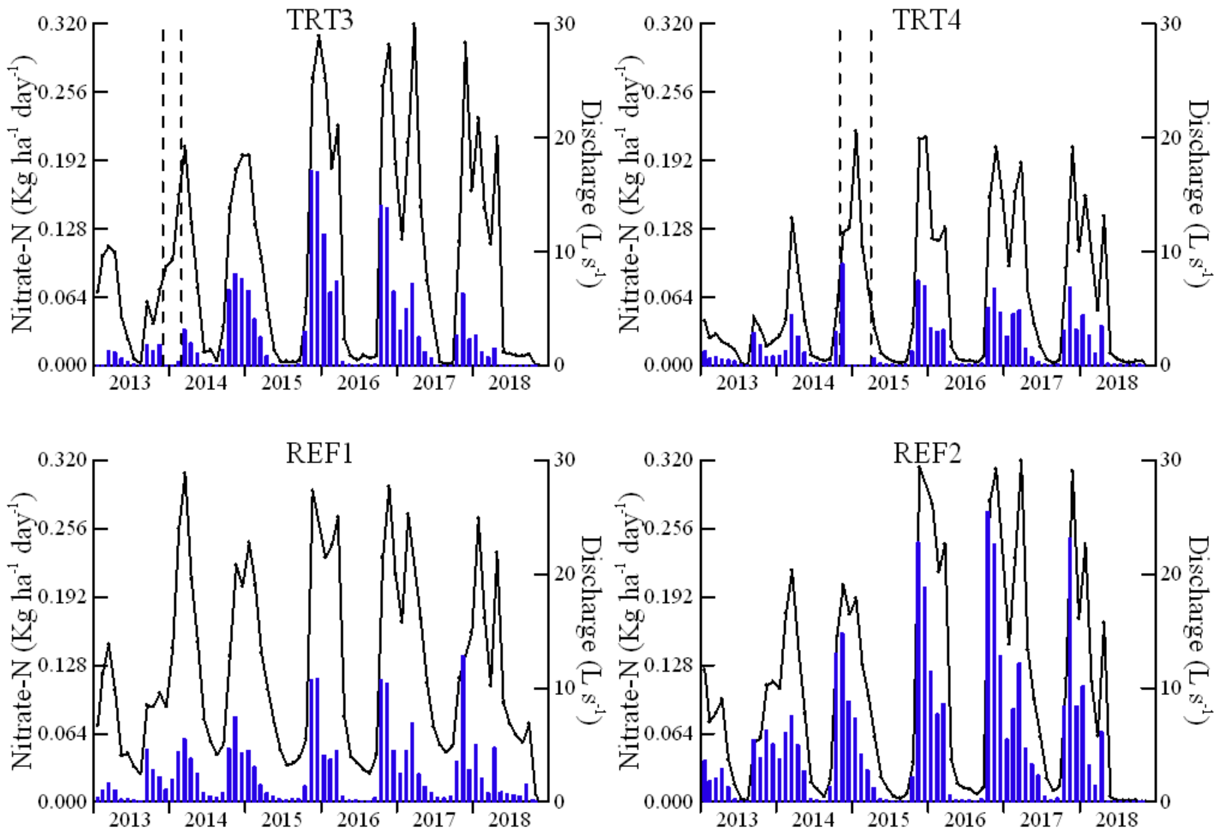

Figure 6-4. Nitrate-N export (left axis; blue bars) and mean monthly stream flow (right axis; black line) from treatment (top) and reference (bottom) sites. The vertical dashed lines bracket the active harvest period.

6-4.2.2. Annual Export

Annual pre-harvest total-N export varied widely among the sites. Export was 4.09 kg/ha/yr in TRT3 and 7.25 and 11.45 kg/ha/yr in REF1 and REF2, respectively, during the same period (**Table 6-6**). Total-N export was 5.28 kg/ha/yr in TRT4 and 12.44 and 17.45 kg/ha/yr in REF1 and REF2, respectively. Pre-harvest export from all sites corresponded to the proportion of hardwood trees in the riparian area (**Table 6-7**) with greater export from sites with a higher proportion of deciduous, largely red alder (*Alnus rubra*), trees.

Post-harvest total-N export more than quadrupled in TRT3 to 17.84 kg/ha/yr, while discharge increased less than two-fold. Over the same period export from the REF sites increased approximately two-fold, roughly in proportion to the change in discharge. Total-N export doubled in TRT4 to 11.52 kg/ha/yr, roughly proportional to the increase in discharge. Over the same period export increased by 24% and 44% in REF1 and REF2, respectively, roughly in proportion to the change in discharge at those sites over this period.

Nitrate-N comprised most of the total-N load in all streams, ranging from more than 73% in REF1 to more than 94% in REF2. As a result, the pattern of response in nitrate-N was very similar to total-N. Pre-harvest export ranged from 3.63 kg/ha/yr in TRT3 to 16.74 kg/ha/yr in REF2. Post-harvest nitrate-N export increased 345% and 68% in TRT3 and TRT4, respectively. Export increased in both REF sites post-harvest as well, 124% and 117% in REF1 and REF2, respectively, relative to TRT3 harvest, and 29% and 47% relative to TRT4 harvest.

The GLMM analysis showed no significant change in the export of total-N ($P = 0.948$) or in nitrate-N ($P = 0.455$) post-harvest relative to the REF sites (**Table 6-4**).

Table 6-6. Mean annual nutrient export (kg/ha/yr) and mean discharge for the pre- and post-harvest periods for each treatment site and the corresponding period in the unharvested reference sites.

		Treatment			REF1			REF2		
Site	**Period**	**Total-N**	**Nitrate-N**	**Discharge (mm/day)**	**Total-N**	**Nitrate-N**	**Discharge (mm/day)**	**Total-N**	**Nitrate-N**	**Discharge (mm/day)**
TRT3	Pre	4.09	3.63	4.09	7.25	5.33	4.07	11.45	10.8	3.31
	Post	17.84	16.15	7.09	15.52	11.94	7.13	24.05	23.47	6.24
TRT4	Pre	5.28	5.04	2.13	12.44	9.25	6.59	17.45	16.74	5.09
	Post	11.52	8.49	4.23	15.47	11.93	7.08	25.17	24.55	6.57

Table 6-7. Percent deciduous overstory vegetation (based on basal area) in riparian plots and pre-harvest total-N export (kg/ha/yr) for treatment (TRT) sites and both reference (REF) sites during the corresponding pre-harvest period.

Site	**% Deciduous**	**Total-N Export***	
		TRT3	**TRT4**
TRT3	1	4.09	
TRT4	9		5.28
REF1	18	7.25	12.44
REF2	28	11.45	17.45

* Harvest periods are defined by the harvest timing at each TRT site.

6-5. DISCUSSION

Pre-harvest total-N and nitrate-N concentration and export estimates were variable but 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; McIntyre *et al.* 2018), Oregon (Brown *et al.* 1973; Harr and Fredricksen 1988; Cairns and Lajtha 2005; Meininger 2011), and British Columbia (Feller and Kimmins 1984). Nitrate-N concentration 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).

However, it is unlikely that atmospheric deposition was a factor here. Mean annual atmospheric nitrogen deposition measured at the Hoh River Ranger Station (WA14) and near La Grande, Washington (WA21) was low, averaging 1.53 and 1.03 kg N/ha/yr, respectively, from 2013 to 2017 (NADP 2019). 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 concentration or 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 72 to 99% of total basal area within the 50-ft (15.2-m) riparian zone (see Chapter 3 – *Riparian Stand Structure and Wood Recruitment*). Pre-harvest N export was higher in the sites with a greater amount of deciduous, largely red alder, overstory measured in the riparian plots (**Table 6-7**), similar to Compton and colleagues (2003). However, it was not possible to determine whether this was a factor in the post-harvest export because both TRT sites had very little hardwood vegetation.

Mean post-harvest values of nitrate-N concentration and export increased in all of our sites, as expected, but remained as variable as the pre-harvest values, ranging from 8.49 kg/ha/yr in TRT4 to 24.55 kg/ha/yr in REF2. 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 red alder, at approximately 25 to 35 kg/ha/yr, 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/yr immediately after harvest. By 2006, nitrate-N export from Needle Branch had increased to 18 kg/ha/yr, which was attributed to the increase in red alder 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/yr. 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/yr following a clearcut harvest and an increase from 0.7 to 4.4 kg/ha/yr following a clearcut with slash burning. Sollins and McCorison (1981) reported an increase in N export of less than 2 kg/ha/yr after clearcutting an old-growth conifer forest in the Oregon Cascade Mountains.

The increase in post-harvest N export from TRT3 (**Table 6-6**) was due to higher concentrations and higher discharge (**Table 6-3**), similar to the large increases in total-N and nitrate-N export seen in McIntyre and colleagues (2018). In contrast, the relatively stable N concentrations in TRT4 and both REF sites suggest that it was primarily higher discharge during the post-harvest years that contributed to higher N export from these sites. Differences between TRT3 and TRT4 in the response of N export also may be due to differences in the application of the Forest Practices rules. Although 85% of each site was harvested, 92% of the stream length was buffered in TRT4 compared with only 58% in TRT3. The lower proportion of stream buffered in TRT3 may have resulted in less nitrogen uptake by riparian vegetation and more delivered to the stream. This is consistent with McIntyre and colleagues (2018) who estimated the increase in N export after harvest in fully buffered streams was approximately 5 kg/ha/yr less than in streams that were 58% to 73% buffered. Although both TRT sites exported more N after harvest relative to the REF sites, the differences in the application of the harvest treatments (buffer length) and the resulting differences in the magnitude of the response in N export, are likely partially responsible for not detecting a significant treatment effect.

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 (Lajtha *et al.* 1995; Pardo *et al.* 1995; Williams *et al.* 1996; Stottlemyer and Toczydlowski 1999; McIntyre *et al.* 2018) 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/yr), 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 2019) 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 leaching of accumulated soluble N from the forest floor as well as increased entrainment of instream organic matter.

The dominant form of stream nitrogen can vary. In this study, nitrate-N comprised 73 to 94% of total-N exported even though atmospheric N deposition at the two nearest NADP monitoring sites was low during the study (NADP 2019) and the riparian forests were conifer-dominated (1 to 28% deciduous; **Table 6-7**). Scott and colleagues (2007) found that 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) observed 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 1997). Nitrate-N dominated export in many hardwood-dominated streams in the Salmon River watershed of western Oregon, and 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% in individual samples.

Overall, we were able to estimate nitrogen exports pre- and post-harvest at our four sites. Estimates varied widely among sites but they were within the range of values observed for managed forest lands in the Pacific Northwest. Although nitrogen export increased after harvest at both TRT sites, there were marked differences in magnitude of the increase. Export from TRT3, with only 58% of the stream buffered, increased more than from TRT4, with 92% buffered. These differences in the extent of the riparian buffers, the unusually dry pre-harvest period, and small number of sites likely impeded our ability to statistically detect a change in export after harvest.

6-6. REFERENCES

Artigas-Alejo, J. 2008. *The Role of Fungi and Bacteria on the Organic Matter Decomposition Process in Streams: Interaction and Relevance in Biofilms*. Ph.D. dissertation. Universitat de Girona, Girona, Spain.

Bernhardt, E.S., G.E. Likens, D.C. Busco, and C.T. Driscoll. 2003. In-stream uptake dampens effects of major forest disturbance on watershed nitrogen export. *Proceedings of the National Academy of Sciences* 100:10304-10308.

Boczulak, S., B. Hawkins, D. Maynard, and R. Roy. 2015. Long-and short-term temperature differences affect organic and inorganic nitrogen availability in forest soils. *Canadian Journal of Soil Science* 95:77-86.

Bosch, J.M., and J.D. Hewlett. 1982. A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology* 55:3-23.

Brown, G.W., A.R. Gahler, and R.B. Marston. 1973. Nutrient losses after clear-cut logging and slash burning in the Oregon coast range. *Water Resources Research* 9:1450-1453.

Burke, P.M., S. Hill, N. Iricanin, C. Douglas, P. Essex, and D. Tharin. 2002. Evaluation of preservation methods for nutrient species collected by automatic samplers. *Environmental Monitoring and Assessment* 80:149-173.

Cairns, M.A., and K. Lajtha. 2005. Effects of succession on nitrogen export in the west-central Cascades, Oregon. *Ecosystems* 8:583-601.

Compton, J.D., M.R. Church, S.T. Larned, and W.E. Hogsett. 2003. Nitrogen export from forested watersheds in the Oregon coast range: The role of N_2-fixing red alder. *Ecosystems* 6:773-785.

Dahlgren, R.A. 1998. Effects of forest harvest on streamwater quality and nitrogen cycling in the Caspar Creek watershed. *In* R.R. Ziemer (technical coordinator). *Proceedings of the Conference on Coastal Watersheds: The Caspar Creek Story*. General Technical Report PSW-GTR-168. US Forest Service, Pacific Southwest Research Station, Albany, CA. http://www.fs.fed.us/psw/publications/documents/gtr-168/06-dahlgren.html.

Duan, N. 1983. Smearing estimate: A nonparametric retransformation method. *Journal of the American Statistical Association* 78:605-610.

Edmonds, R.L., R.D. Blew, J.L. Marra, J. Blew, A.K. Barg, G.L. Murray, and T.B. Thomas. 1998. *Vegetation Patterns, Hydrology, and Water Chemistry in Small Watersheds in the Hoh River Valley, Olympic National Park*. National Park Service Monograph NPSD/NRUSGS/NRSM-98/02. US Department of the Interior, National Park Service, Washington, DC. 131 p.

Edmonds R.L., T.B. Thomas, and R.D. Blew. 1995. Biogeochemistry of an old-growth forested watershed, Olympic National Park, Washington. *Water Resources Bulletin* 31:409-419.

Feller, M.C. 2005. Forest harvesting and streamwater inorganic chemistry in western North America: A review. *Journal of the American Water Resources Association* 41:785-811.

Feller, M.C., and J.P. Kimmins. 1984. Effects of clearcutting and slash burning on streamwater chemistry and watershed nutrient budgets in southwestern British Columbia. *Water Resources Research* 20:29-40.

Feller, M.C., R. Lehmann, and P. Olanski. 2000. Influence of forest harvesting intensity on nutrient leaching through soil in southwestern British Columbia. *Journal of Sustainable Forestry* 10:241-247.

Fowler, W.B., T.D. Anderson, and J.D. Helvey. 1988. *Changes in Water Quality and Climate After Forest Harvest in Central Washington State*. Research Paper PNW-RP-388. US Forest Service, Pacific Northwest Research Station, Portland, OR. 12 p.

Fredriksen, R.L. 1971. Comparative water quality—natural and disturbed streams. Pages 125-137 *in* J.T. Krygier and J.D. Hall (eds.). *Forest Land Uses and Stream Environment Symposium Proceedings*, Oregon State University, Corvallis, OR.

Gravelle, J.A., T.E. Link, J.R. Broglio, and J.H. Braatne. 2009. Effects of timber harvest on aquatic macroinvertebrate community composition in a northern Idaho watershed. *Forest Science* 55:352-366.

Hagerman, S.M., M.D. Jones, G.E. Bradfield, and S.M. Sakakibara. 1999. Ectomycorrhizal colonization of *Picea engelmannii* x *Picea glauca* seedlings planted across cut blocks of different sizes. *Canadian Journal of Forest Research* 29:1856-1870.

Hale, V.C. 2007. *A Physical and Chemical Characterization of Stream Water Draining Three Oregon Coast Range Catchments*. M.S. thesis. Oregon State University, Corvallis, OR.

Harr, R.D. 1983. Potential for augmenting water yield through forest practices in western Washington and western Oregon. *Journal of the American Water Resources Association* 19:383-393.

Harr, R.D., and R.L. Fredriksen. 1988. Water quality after logging small watersheds within the Bull Run watershed, Oregon. *Water Resources Bulletin* 24:1103-1111.

Harvey, A.E., M.F. Jurgensen, and M.J. Larsen. 1980. Clearcut harvesting and ectomycorrhizae: Survival of activity on residual roots and influence on a bordering forest stand in western Montana. *Canadian Journal of Forest Research* 10:300-303.

Hedin, L.O., J.J. Armesto, and A.H. Johnson. 1995. Patterns of nutrient loss from unpolluted, old-growth temperate forests: Evaluation of biogeochemical theory. *Ecology* 76:493-509.

Helsel, D.R., and R.M. Hirsch. 2002. *Statistical Methods in Water Resources*. Techniques of Water-Resources Investigations of the US Geological Survey Book 4, Hydrologic Analysis and Interpretation. http://water.usgs.gov/pubs/twri/twri4a3/.

Kiffney, P.M., J.S. Richardson, and J.P. Bull. 2003. Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. *Journal of Applied Ecology* 40:1060-1076.

Lajtha, K, B. Seely, and I. Valiela. 1995. Retention and leaching losses of atmospherically-derived nitrogen in the aggrading coastal watershed of Waquoit Bay, MA. *Biogeochemistry* 28:33-54.

Lewis, J., and R. Eads. 2009. *Implementation Guide for Turbidity Threshold Sampling: Principles, Procedures, and Analysis.* General Technical Report PSW-GTR-212. US Forest Service, Pacific Southwest Research Station, Albany, CA. 87 p.

Likens, G.E., F.H. Bormann, N.M. Johnson, D.W. Fisher, and R.S. Pierce. 1970. Effects of forest cutting and herbicide treatment on nutrient budgets in the Hubbard Brook watershed ecosystem. *Ecological Monographs* 40:23-47.

Liles, G.C. 2005. *Biogeochemistry of Managed Forest Headwater Streams in Low Elevation Western Washington*. M.S. thesis. University of Washington, Seattle, WA.

Martin C.W., and R.D. Harr. 1988. Precipitation and streamwater chemistry from undisturbed watersheds in the Cascade Mountains of Oregon. *Water, Air, and Soil Pollution* 42: 203-219.

Martin, C.W., D.S. Noel, and C.A. Federer. 1984. Effects of forest clearcutting in New England on stream chemistry. *Journal of Environmental Quality* 13:204-210.

McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn (technical coordinators). 2018. *Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington*. Cooperative Monitoring, Evaluation, and Research Report, CMER 18-100. Washington Department of Natural Resources, Olympia, WA. 890 p.

McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D.E. Schuett-Hames, R. Ojala-Barbour, G. Stewart, and T. Quinn (technical coordinators). 2021. *Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington—Phase 2 (9 Years after Harvest)*. Cooperative Monitoring, Evaluation, and Research Report CMER 2021.07.27, Washington State Forest Practices Adaptive Management Program, Washington Department of Natural Resources, Olympia, WA.

Meininger, W.S. 2011. *The Influence of Contemporary Forest Management on Stream Nutrient Concentrations in an Industrialized Forest in the Oregon Cascades*. M.S. thesis. Oregon State University, Corvallis, OR.

Mohamedali, T., M. Roberts., B. Sackmann, and A. Kolosseus. 2011. *Puget Sound Dissolved Oxygen Model Nutrient Load Summary for 1999–2008*. Publication No. 11-03-057. Washington Department of Ecology, Lacey, WA.

Moore, R.D., and S.M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: A review. *Journal of the American Water Resources Association* 41:763-784.

Murray, G.L.D., R.L. Edmonds, and J.L. Marra. 2000. Influence of partial harvesting on stream temperatures, chemistry, and turbidity in forests on the western Olympic Peninsula, Washington. *Northwest Science* 74:151-164.

NADP. 2019. National Atmospheric Deposition Program. Sites WA14 and WA21. NADP Program Office, Illinois State Water Survey, Champaign, IL. http://nadp.slh.wisc.edu/data/ntn/

Ohrui, K., and M.J. Mitchell. 1997. Nitrogen saturation in Japanese forested watersheds. *Ecological Applications* 7:391-401.

Pardo, L.H., C.T. Driscoll, and G.E. Likens. 1995. Patterns of nitrate loss from a chronosequence of clear-cut watersheds. *Water, Soil, and Air Pollution* 85:1659-1564.

Roberts, M., J. Bos, and S. Albertson. 2008. *South Puget Sound Dissolved Oxygen Study: Interim Data Report*. Publication No. 08-03-037. Washington State Department of Ecology, Lacey, WA.

Sackmann, B.S. 2011. *Deschutes River Continuous Nitrate Monitoring*. Publication No. 11-03-030. Washington State Department of Ecology, Lacey, WA.

SAS Institute, Inc. 2013. *SAS/STAT User's Guide*. SAS Statistical Institute, Cary, NC.

Schelker, J., R. Sponseller, E. Ring, L. Högbom, S. Löfgren, and H. Laudon. 2016. Nitrogen export from a boreal stream network following forest harvesting: seasonal nitrate removal and conservative export of organic forms. *Biogeosciences* 13:1-12.

Scott, D., J. Harvey, R. Alexander, and G. Schwarz. 2007. Dominance of organic nitrogen from headwater streams to large rivers across the conterminous United States. *Global Biogeochemical Cycles* 21, GB1003, doi:10.1029/2006GB002730.

Smith, E.P. 2002. BACI design. Pages 141-148 *in* A.H. El-Shaarawi and W.W. Piegorsch (eds.). *Encyclopedia of Environmetrics*. John Wiley & Sons, Ltd., Chichester, UK.

Sollins, P., C.C Grier, F.M. McCorison, K. Cromack, Jr., R. Fogel, and R.L. Fredriksen. 1980. The internal element cycles of an old growth Douglas-fir ecosystem in western Oregon. *Ecological Monographs* 50:261-85.

Sollins, P., and F.M. McCorison. 1981. Nitrogen and carbon solution chemistry of an old growth coniferous forest watershed before and after cutting. *Water Resources Research* 17:1409-1418.

Stark, N.M. 1979. *Nutrient Losses from Timber Harvesting in a Larch/Douglas-fir Forest*. Research Paper INT-231. US Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT. 41 p.

Stednick, J.D. 1996. Monitoring the effects of timber harvest on annual water yield. *Journal of Hydrology* 176:79-95.

Stoddard J.L. 1994. Long-term changes in watershed retention of nitrogen: Its causes and aquatic consequences. Pages 223-284 *in* L.A. Baker (ed.). *Environmental Chemistry of Lakes and Reservoirs. Advances in Chemistry*. Series Number 237. American Chemical Society, Washington, DC.

Stottlemyer, R., and D. Toczydlowski. 1999. Seasonal relationships between precipitation, forest floor, and streamwater nitrogen, Isle Royale, Michigan. *Soil Science Society of America Journal* 63:389-398.

Swank, W.T., and J.M. Vose. 1997. Long-term nitrogen dynamics of Coweeta forested watersheds in the southeastern United States of America. *Global Biogeochemical Cycles* 11:657-671.

SYSTAT Software, Inc. 2009. *SYSTAT Version 13. User's Manual*. Systat Software, Inc., Chicago, IL.

Taylor, J.C. 2008. *Effects of Riparian Buffers on Soil Nitrogen Mineralization and Stream Nitrogen Concentrations in Headwater Streams of Western Washington*. M.S. thesis. University of Washington, Seattle, WA.

Tiedemann, A.R., T.M. Quigley, and T.D. Anderson. 1988. Effects of timber harvest on stream chemistry and dissolved nutrient losses in northeast Oregon. *Forest Science* 34:344-358.

Triska, F.J., V.C. Kennedy, R.J. Avanzino, and B.N. Reilly. 1983. Effects of simulated canopy cover on regulation of nitrate uptake and primary production by natural periphyton assemblages. Pages 129-159 *in* F.D. Fontaine III and S.M. Bartell (eds.). *Dynamics of Lotic Ecosystems*. Ann Arbor Science, Ann Arbor, MI.

Vanderbilt, K.L., K. Lajtha, and F.J. Swanson. 2003. Biogeochemistry of unpolluted forested watersheds in the Oregon Cascades: Temporal patterns of precipitation and stream nitrogen fluxes. *Biogeochemistry* 62:85-117.

Wigington, P.J., M.R. Church, T.C. Strickland, K.N. Eshleman, and J. Van Sickle. 1998. Autumn chemistry of Oregon coast range streams. *Journal of the American Water Resources Association* 34:1035-1049.

Williams, M.W., R.C. Bales, A.D. Brown, and J.M. Melack. 1996. Fluxes and transformations of nitrogen in a high-elevation catchment, Sierra Nevada. *Biogeochemistry* 28:1-31.

CHAPTER 7 – BENTHIC MACROINVERTEBRATES

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF APPENDIX TABLES

7-1. ABSTRACT

Headwater streams comprise a significant proportion of the landscape and contribute a substantial proportion of macroinvertebrates to downstream fish-bearing waters. These streams receive most of their energy from organic matter inputs and consist predominantly of macroinvertebrate taxa able to use these resources. Timber harvest may change the energy balance of these streams through changes in organic matter inputs and primary production and cause a shift in the macroinvertebrate community. Also, changes in shade, temperature, stream flow, sediment, and wood inputs may also influence the assemblage. We assessed the response of benthic macroinvertebrates in non-fish-bearing, headwater streams to timber harvest using a before-after control-impact design. The study sites were treated with the current Washington State Forest Practices buffer or remained unharvested. We collected macroinvertebrates using a Surber sampler in the spring, summer, and fall for one year pre-harvest and one year post-harvest. Samples were sorted, identified, and analyzed for several metrics, functional feeding group, and major macroinvertebrate order using analysis of variance and ordination. Although we observed some changes after harvest, there were no major reductions in benthic macroinvertebrates or shifts in functional feeding groups associated with the treatment within the limitations of the study design and sampling methodology. We found a possible treatment × period interaction for Ephemeroptera-Plecoptera-Trichoptera richness ($P = 0.061$) and the Shannon H' diversity index ($P = 0.092$), but the metrics decreased in both the reference and treatment sites, which may indicate that broader environmental factors rather than a treatment effect were influencing macroinvertebrate assemblages. There was no treatment × period interaction for the other metrics, functional feeding groups, or major macroinvertebrate orders. The lack of major changes in the macroinvertebrate community may reflect the extensive buffers, increase in wood cover from logging slash and windthrow, and subsequent vegetation growth, which provided enough shade to inhibit primary production and instream structure to retain particulate organic matter.

7-2. INTRODUCTION

Headwater streams comprise a significant proportion of a stream network (Benda *et al.* 2005; Freeman *et al.* 2007; Richardson and Danehy 2007). Despite this, headwater streams are less studied than higher-order streams due to access difficulty and lack of key aquatic resources (e.g., fish) (Benda *et al.* 2005; Richardson and Danehy 2007). Productivity of these streams is typically lower than that of larger streams (Richardson *et al.* 2005), but headwater streams do contribute a substantial proportion of overall productivity to the stream network given their prevalence (Muchow and Richardson 2000; Gomi *et al.* 2002; Freeman *et al.* 2007). Because of their location in the stream network, headwater streams are susceptible to changes in the upland landscape, which may then influence downstream, fish-bearing waters.

In the Pacific Northwest, headwater streams receive most of their energy from organic matter inputs (Cummins *et al.* 1983; Gregory *et al.* 1991; Bilby and Bisson 1992) as shading from riparian vegetation generally limits autotrophic production (Richardson and Danehy 2007). Macroinvertebrate assemblages typically consist of those specialized in shredding and collecting particulate organic matter, which is retained in depositional areas upstream of wood dams (Bilby and Likens 1980; Bilby 1981; Gregory *et al.* 1991). Timber harvest has the potential to shift the energy balance of these streams through a reduction in organic matter inputs and increase in insolation and primary production (Bilby and Bisson 1992; Kiffney *et al.* 2003; Richardson and Danehy 2007; Warren *et al.* 2016). This shift may result in a decrease in macroinvertebrate taxa that consume organic matter and an increase in scrapers that graze on periphyton.

Timber harvest may affect other headwater stream processes, including temperature and shade (Brown and Krygier 1970; Gomi *et al.* 2006; McIntyre *et al.* 2018), channel morphology (Jackson *et al.* 2001), discharge (Macdonald *et al.* 2003; Moore and Wondzell 2005), nutrients (Feller and Kimmins 1984; Harr and Fredriksen 1988), and sediment (Beschta 1978; Waters 1995; Allan 2004; Karwan *et al.* 2007), which may influence macroinvertebrate assemblages. Stream temperature controls growth and metabolism of aquatic insects (Minshall 1984). An increase in temperature may result in earlier and asynchronous emergence in some species (Li *et al.* 2011) or cause declines in taxa less tolerant of higher temperatures while favoring tolerant species (Hawkins *et al.* 1997; Collier and Smith 2005; Dallas and Rivers-Moore 2012; Kroll *et al.* 2017). Inputs of large wood from logging slash (Jackson *et al.* 2001; Haggerty *et al.* 2004) and windthrow (Grizzel and Wolff 1998) combined with changes in stream flow may influence channel morphology and substrate composition, which in turn would determine available habitat and retention of food resources for macroinvertebrates (Minshall 1984; Hetrick *et al.* 1998; Richardson *et al.* 2005; Warren *et al.* 2016). An increase in peak flows following harvest may scour algal food resources from the streambed (Allan 1995), increase transport rates of particulate organic matter (Richardson *et al.* 2005), and dislodge macroinvertebrates from the substrate (Hershey and Lamberti 1998). Changes in the concentration of nitrogen and phosphorus may increase periphyton growth (Perrin and Richardson 1997; Francoeur *et al.* 1999; Kiffney and Richardson 2001). An increase in suspended sediment may affect invertebrate respiration and filtering apparatuses while sediment deposition may reduce interstitial habitat and movement of water and dissolved gases within the substrate (Minshall 1984; Waters 1995), initiating macroinvertebrate drift (Wiley and Kohler 1984; Culp *et al.* 1986; Waters 1995; Shaw and Richardson 2001; Suren and Jowett 2001). Previous studies have found 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). The response of functional feeding groups, however, has varied (e.g., Murphy and Hall 1981; Hawkins *et al*. 1982; Fuchs *et al*. 2003; Haggerty *et al*. 2004; Hernandez *et al*. 2005; Danehy *et al*. 2007; Gravelle *et al*. 2009).

In 2001, Washington State adopted riparian buffer prescriptions for Type N, or non-fish-bearing, headwater streams to minimize the impact of forest management activities. The Hard Rock Study, in part, evaluated the response of macroinvertebrate drift to the Type N riparian buffer prescriptions. While the authors observed some changes in drift after harvest, they found no major reductions in macroinvertebrate export and no major shifts in functional feeding groups (McIntyre *et al.* 2018). The Hard Rock Study sites, however, consisted of basaltic lithologies, which are typically more resistant to increases in suspended sediment associated with forest management activities (Bywater-Reyes *et al.* 2017). The Soft Rock Study sites with their easily eroded sedimentary lithologies may be more susceptible to higher sediment input.

The Soft Rock Study was designed to evaluate the effectiveness of the Type N riparian buffer prescriptions in incompetent lithologies. This component of the study examined the potential impact of harvest activities on downstream benthic macroinvertebrate communities. We used total richness, Ephemeroptera-Plecoptera-Trichoptera (EPT) richness, percent EPT, the Shannon H' diversity index, and the Fine Sediment Biotic Index (FSBI) to evaluate changes in the macroinvertebrate community. The EPT taxa consist of many intolerant and sensitive species (Hodkinson and Jackson 2005) and higher metric values generally indicate better habitat quality (Gravelle *et al.* 2009). The FSBI is a measure of the impact of fine sediment on benthic macroinvertebrate assemblages in the northwestern United States, with higher FSBI values indicating the presence of more sediment sensitive taxa and thus lower levels of fine sediment (Relyea *et al.* 2012). In addition to the macroinvertebrate metrics, we assessed the response of the macroinvertebrate functional feeding groups and major taxonomic orders to timber harvest.

Although the Type N riparian buffer prescriptions are intended to reduce or eliminate the impact of timber harvest on stream processes, we did measure changes in shade, stream temperature, discharge, nutrients, and wood inputs after harvest in the Hard Rock Study and there is the potential for changes in sediment inputs in the Soft Rock Study sites. We hypothesized that changes in shade, temperature, and sediment processes in the Soft Rock Study sites may result in a decrease in EPT taxa, diversity, and the FSBI while favoring less sensitive taxa. In addition, we hypothesized that a reduction in organic matter inputs and an increase in autotrophic production may cause a change in the energy balance of the streams that may shift the macroinvertebrate assemblage from collector and shredder taxa typical of headwater streams to scraper taxa.

7-3. METHODS

7-3.1. SAMPLE COLLECTION

We sampled macroinvertebrates from the 11 Soft Rock Study sites (see Chapter 2 – *Study Design* for site selection and site description details). We collected benthic macroinvertebrate samples three times annually, once in the spring, summer, and fall, for one year pre-harvest (2013) and one year post-harvest (2016). Each sample was a composite that consisted of eight randomly selected sampling areas located in the first 100 m of each study site. We numbered the sampling locations from 1 to 100, with each number representing a distance from the Type F/N break (at 0 m) to 100 m upstream, and then used a list of randomly generated numbers to select the sampling areas. Any replicate numbers were skipped and new numbers were chosen. We sampled starting from the furthest downstream location, then moved upstream to reduce substrate disturbance from wading and the number of macroinvertebrates in the water column.

We collected the samples using a 500-µm mesh Surber sampler with a 0.3-m by 0.3-m frame. We excluded pools with no flow velocity since we relied on flow to direct debris into the sampler, and excluded reaches where wood cover, cascades, or waterfalls prevented access to the streambed for sampler placement. The opening of the sampler net faced upstream, with the sampler positioned flush with the streambed, and included the thalweg. The frame of the sampler designated the sampling area (**Figure 7-1**). Debris and substrate larger than 5 cm in diameter was gently scrubbed by hand in front of the net, visually inspected for any remaining organisms, then set outside of the sampling area. We used a trowel to disturb the substrate down to a minimum depth of 4 to 5 cm for about 30 seconds in most cases, although we extended the substrate disturbance time to 120 seconds when needed in sampling areas with slow-flowing pools. This suspended debris and macroinvertebrates so that the stream flow would sweep them into the net. Any rocks, wood, or leaves that were collected were removed from the net, scrubbed, visually inspected for remaining invertebrates, and then placed outside of the sampling area.

Samples were preserved in the field with 95% ethanol stained with rose bengal (5 drops from a solution of 1 g rose bengal to 100 ml water) and transported to the laboratory for sorting.

Figure 7-1. Surber sampler placement for sampling benthic macroinvertebrates. We collected the sample from the area designated by the sampler frame.

7-3.2. SAMPLE PROCESSING

Samples were rinsed through a 250-µm sieve and spread across a gridded tray for subsampling. We randomly selected grids and then sorted macroinvertebrates from the debris. When we reached a count of 500 organisms, we finished sorting the remaining organisms in that grid and then recorded the number of grids sorted and the final sorted count. Samples with less than 500 organisms were sorted in their entirety. While the fixed count subsampling method has the potential to influence metrics (Courtemanch 1996), it is a commonly used method due to its ability to produce reliable results while remaining cost efficient (Barbour and Gerritson 1996). Sorted macroinvertebrates were sent to an off-site lab (Rhithron Associates, Inc.) for identification using the Washington State Department of Ecology’s Taxonomic Laboratory Protocols (Plotnikoff and White 1996).

7-3.3. STATISTICAL ANALYSIS

Several metrics were considered in our analysis of macroinvertebrate response to timber harvest. We assessed changes in total richness, EPT richness, percent EPT, and the Shannon H' diversity index using metric values provided by Rhithron Associates, Inc., and in the FSBI as determined from Relyea and colleagues (2012). We calculated percentages of the functional feeding groups and the major taxonomic orders for each sample. For family Chironomidae (Diptera), we also calculated percentages of individuals in subfamily Tanypodinae (primarily predators) and those in non-Tanypodinae subfamilies (primarily collector-gatherers).

Our analyses evaluated the generalized null hypothesis:

$$\Delta M_{REF} = \Delta M_{TRT} \quad (7\text{-}1)$$

where: ΔM_{REF} is the change in the macroinvertebrate metric in the references sites, and
ΔM_{TRT} is the change in the macroinvertebrate metric in the treatment sites.

7-3.3.1. Analysis of Variance

We used generalized linear mixed-effects models (GLMM) that incorporate both fixed and random effects for hypothesis testing. We conducted the statistical analyses using the GLIMMIX procedure in SAS 9.4 (SAS 2013) and estimated model parameters using the estimation technique, distribution, and link most suited for model convergence (**Table 7-1**). Fixed effects were treatment (REF and TRT), period (PRE and POST), and the treatment × period interaction, and random effects were site and season. To determine the fixed-effect parameter estimates and denominator degrees of freedom for *t* and *F* tests, we used either the containment, Kenward-Roger (Kenward and Roger 1997), or residual methods depending on the estimation technique most suited for model convergence (the Kenward-Rogers option is not available for the maximum likelihood with Laplace approximation) (**Table 7-1**). We ran standard diagnostics to check for non-normality and heteroscedasticity of residuals and found no evidence of either.

Pairwise comparisons were used to estimate the effect size for each treatment relative to the reference in the post-harvest period:

$$\text{Effect size} = (\text{TRT}_{\text{POST}} - \text{TRT}_{\text{PRE}}) - (\text{REF}_{\text{POST}} - \text{REF}_{\text{PRE}}) \quad (7\text{-}2)$$

where: REF = reference sites
TRT = treatment sites
PRE = pre-harvest period
POST = post-harvest period

We presented the estimates of the effect size and differences of least square means and associated 95% confidence intervals. Because the analyses required different distributions, the estimates, standard errors, and confidence intervals remain in their transformed space and are noted as such in the relevant tables. We did not adjust the P-values for multiple comparisons. Because of the unbalanced design and small sample size, we used a p-value threshold of 0.1, but also focused on the magnitude and pattern of the response.

Table 7-1. The response distribution, link function, estimation technique, and degrees of freedom method used in the analysis. DF = degrees of freedom; EPT = Ephemeroptera-Plecoptera-Trichoptera; FSBI = Fine Sediment Biotic Index; ML = maximum likelihood; ML (Laplace) = maximum likelihood with Laplace approximation; RML = restricted maximum likelihood; RPL = residual pseudo-likelihood.

	Response Distribution	Link Function	Estimation Technique	DF Method
Metric				
Total Richness	Gaussian	Identity	ML (Laplace)	Containment
EPT Richness	Gaussian	Identity	ML (Laplace)	Containment
% EPT	Beta	Logit	RPL	Kenward-Roger
Shannon H’	Gaussian	Identity	ML (Laplace)	Containment
FSBI	Gaussian	Identity	RML	Kenward-Roger
Feeding Group				
% Filterer	Beta	Logit	ML	Residual
% Gatherer	Beta	Logit	RPL	Kenward-Roger
% Parasite	Beta	Logit	ML	Residual
% Predator	Beta	Logit	RPL	Kenward-Roger
% Scraper	Beta	Logit	RPL	Kenward-Roger
% Shredder	Beta	Logit	RPL	Kenward-Roger
% Unknown	Beta	Logit	ML	Residual
Order				
% Coleoptera	Beta	Logit	ML	Residual
% Diptera	Beta	Logit	ML	Residual
% Tanypodinae	Beta	Logit	ML	Residual
% Non-Tanypodinae	Beta	Logit	ML	Residual
% Ephemeroptera	Beta	Logit	ML	Residual
% Plecoptera	Beta	Logit	ML	Residual
% Trichoptera	Beta	Logit	ML	Residual

7-3.3.2. Relationship of Metrics to Temperature and Canopy Cover

We correlated the pre- to post-harvest difference (POST minus PRE) of the macroinvertebrate metrics, functional feeding groups, and major taxonomic orders by site with the 2016 July mean monthly temperature response (MMTR) for the temperature station at the downstream end of the macroinvertebrate sampling reach and with mean canopy cover within the sampling reach. For this analysis, we used Pearson correlation coefficients and the associated, uncorrected probabilities using SYSTAT v13 statistical software (SYSTAT Software Inc. 2009).

7-3.3.3. Ordination

We used non-metric multidimensional scaling (NMDS) to examine potentially undetected changes in macroinvertebrate community structure in response to timber harvest. The NMDS used the Bray-Curtis distance measure and a minimum of 500 iterations to quantify the dissimilarity between the sites. We elected to use the full dataset of macroinvertebrate abundances for the analysis since merging taxa or omitting rare taxa did not improve the ordination. Taxa abundances were log transformed ($\log_{10}$ x+1), which gave a stronger ordination and lower stress value than using untransformed or relative abundances. We used Pearson correlation coefficients to examine the relationships of the metrics with the NMDS axes and permutational analysis of variance (PERMANOVA) to determine differences in community structure between sites stratified by basin. NMDS and PERMANOVA were conducted using the vegan Community Ecology package (Oksanen *et al.* 2020) in R (R Core Team 2020).

7-4. RESULTS

7-4.1. ANALYSIS OF VARIANCE

7-4.1.1. Metrics

The GLMM and pairwise comparisons suggested a treatment × period interaction for EPT richness ($P = 0.061$) and the Shannon H'diversity index ($P = 0.092$) (**Tables 7-2** and **7-3**), but no treatment × period interaction for total richness ($P = 0.311$), percent EPT ($P = 0.535$), or the FSBI ($P = 0.947$). Differences of least squares means showed that the REF and TRT sites changed in the same direction from pre- to post-harvest with a decrease in total richness, EPT richness, percent EPT, the Shannon H' diversity index, and the FSBI (**Table 7-4**). Total richness, EPT richness, and the Shannon H' diversity index had P values less than 0.05. Descriptive statistics for the macroinvertebrate metrics are provided in **Appendix Table 7-1**.

Table 7-2. Results of the generalized linear mixed-effects model for the macroinvertebrate metrics. The treatment × period interaction terms indicate pre- to post-harvest differences among treatments. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom; EPT = Ephemeroptera-Plecoptera-Trichoptera; FSBI = fine sediment biotic index.

Metric	Effect	Num DF	Den DF	F Value	Pr > F
Total Richness	Treatment	1	53.0	1.17	0.285
	Period	1	53.0	45.73	<0.001
	Treatment × Period	1	53.0	1.05	0.311
EPT Richness	Treatment	1	53.0	0.12	0.729
	Period	1	53.0	26.58	<0.001
	Treatment × Period	1	53.0	3.66	0.061
% EPT	Treatment	1	7.6	0.04	0.844
	Period	1	53.1	2.39	0.128
	Treatment × Period	1	53.1	0.39	0.535
Shannon H’	Treatment	1	53.0	5.40	0.024
	Period	1	53.0	23.40	<0.001
	Treatment × Period	1	53.0	2.94	0.092
FSBI	Treatment	1	9.0	0.04	0.848
	Period	1	53.0	1.22	0.274
	Treatment × Period	1	53.0	0.00	0.947

Table 7-3. Pairwise comparisons (reference vs. treatment) for the macroinvertebrate metrics using the generalized linear mixed-effects model analyses. SE = standard error; DF = degrees of freedom; C.I. = confidence interval; EPT = Ephemeroptera-Plecoptera-Trichoptera; FSBI = fine sediment biotic index.

						95% C.I.	
Metric	Estimate	SE	DF	T-value	P-value	Lower	Upper
Total Richness[a]	0.027	0.026	53.0	1.02	0.311	-0.025	0.079
EPT Richness[a]	0.063	0.033	53.0	1.91	0.061	-0.003	0.129
% EPT[b]	0.204	0.326	53.1	0.62	0.535	-0.451	0.858
Shannon H’	0.205	0.120	53.0	1.71	0.092	-0.035	0.445
FSBI	0.694	10.434	53.0	0.07	0.947	-20.234	21.623

[a] estimate, standard error, and confidence interval values in log space

[b] estimate, standard error, and confidence interval values in beta space

Table 7-4. Differences of least squares means between the pre- and post-harvest period for the macroinvertebrate metrics using the generalized linear mixed-effects model analyses. SE = standard error; DF = degrees of freedom; C.I. = confidence interval; REF = reference; TRT = treatment; PRE = pre-harvest period; POST = post-harvest period; EPT = Ephemeroptera-Plecoptera-Trichoptera; FSBI = fine sediment biotic index.

							95% C.I.	
Metric	**Comparison**	**Estimate**	**SE**	**DF**	**T-value**	**P-value**	**Lower**	**Upper**
Total Richness[a]	REF Post–Pre	-0.101	0.022	53.0	-4.57	<0.001	-0.145	-0.057
	TRT Post–Pre	-0.074	0.014	53.0	-5.49	<0.001	-0.102	-0.047
EPT Richness[a]	REF Post–Pre	-0.116	0.028	53.0	-4.15	<0.001	-0.173	-0.060
	TRT Post–Pre	-0.053	0.017	53.0	-3.10	0.003	-0.088	-0.019
% EPT[b]	REF Post–Pre	-0.354	0.279	53.1	-1.27	0.209	-0.914	0.205
	TRT Post–Pre	-0.151	0.170	53.1	-0.89	0.379	-0.491	0.190
Shannon H’	REF Post–Pre	-0.392	0.102	53.0	-3.84	<0.001	-0.596	-0.187
	TRT Post–Pre	-0.187	0.062	53.0	-2.99	0.004	-0.312	-0.062
FSBI	REF Post–Pre	-6.111	8.898	53.0	-0.69	0.495	-23.959	11.737
	TRT Post–Pre	-5.417	5.449	53.0	-0.99	0.325	-16.346	5.513

[a] estimate, standard error, and confidence interval values in log space

[b] estimate, standard error, and confidence interval values in beta space

7-4.1.2. Functional Feeding Groups

Seven functional feeding groups were represented in the samples. These included collector-filterers, collector-gatherers, parasites, predators, scrapers, shredders, and unknown. Taxa in the unknown feeding group were those represented by early instars that were not identifiable to family or genus.

The GLMM and pairwise comparisons suggested there was no treatment × period interaction for percent filterers ($P = 0.827$), gatherers ($P = 0.967$), parasites ($P = 0.511$), predators ($P = 0.165$), scrapers ($P = 0.651$), shredders ($P = 0.315$), or unknown ($P = 0.284$) (**Tables 7-5** and **7-6**). Differences of least squares means showed that percent filterers, parasites, and scrapers decreased in the REF and TRT sites from pre- to post-harvest, while percent gatherers, shredders, and unknown increased in the REF and TRT sites (**Table 7-7**). Percent predators, however, changed in the opposite direction from pre- to post- harvest with a decrease in the REF sites and an increase in the TRT sites. Percent filterers and gatherers in the TRT sites and percent parasites and scrapers in both the REF and TRT sites had P values less than 0.1.

Table 7-5. Results of the generalized linear mixed-effects model for the functional feeding groups. The treatment × period interaction terms indicate pre- to post-harvest differences among treatments. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Feeding Group	Effect	Num DF	Den DF	F Value	Pr > F
% Filterer	Treatment	1	62.0	0.06	0.801
	Period	1	62.0	5.84	0.019
	Treatment × Period	1	62.0	0.05	0.827
% Gatherer	Treatment	1	62.0	1.24	0.270
	Period	1	62.0	3.60	0.063
	Treatment × Period	1	62.0	0.00	0.967
% Parasite	Treatment	1	62.0	0.03	0.874
	Period	1	62.0	40.13	<0.001
	Treatment × Period	1	62.0	0.44	0.511
% Predator	Treatment	1	7.3	2.26	0.175
	Period	1	52.9	0.07	0.791
	Treatment × Period	1	52.9	1.98	0.165
% Scraper	Treatment	1	8.4	0.00	0.964
	Period	1	52.7	15.88	0.000
	Treatment × Period	1	52.7	0.21	0.651
% Shredder	Treatment	1	6.0	0.13	0.728
	Period	1	53.0	1.63	0.208
	Treatment × Period	1	53.0	1.03	0.315
% Unknown	Treatment	1	62.0	0.58	0.449
	Period	1	62.0	1.81	0.183
	Treatment × Period	1	62.0	1.17	0.284

Table 7-6. Pairwise comparisons (reference vs. treatment) for the functional feeding groups using the generalized linear mixed-effects model analyses. Estimate, standard error, and confidence interval values in beta space. SE = standard error; DF = degrees of freedom; C.I. = confidence interval.

						95% C.I.	
Feeding Group	**Estimate**	**SE**	**DF**	**T-value**	**P-value**	**Lower**	**Upper**
% Filterer	-0.092	0.420	62.0	-0.22	0.827	-0.931	0.746
% Gatherer	-0.011	0.276	62.0	-0.04	0.967	-0.564	0.541
% Parasite	-0.276	0.417	62.0	-0.66	0.511	-1.109	0.558
% Predator	0.341	0.242	52.9	1.41	0.165	-0.145	0.826
% Scraper	0.161	0.354	52.7	0.46	0.651	-0.549	0.872
% Shredder	-0.269	0.265	53.0	-1.01	0.315	-0.800	0.263
% Unknown	-0.400	0.370	62.0	-1.08	0.284	-1.141	0.340

Table 7-7. Differences of least squares means between the pre- and post-harvest period for the functional feeding groups using the generalized linear mixed-effects model analyses. Estimate, standard error, and confidence interval values in beta space. SE = standard error; DF = degrees of freedom; C.I. = confidence interval; REF = reference; TRT = treatment; PRE = pre-harvest period; POST = post-harvest period.

							95% C.I.	
Feeding Group	**Comparison**	**Estimate**	**SE**	**DF**	**T-value**	**P-value**	**Lower**	**Upper**
% Filterer	REF Post–Pre	-0.465	0.357	62.0	-1.30	0.198	-1.179	0.249
	TRT Post–Pre	-0.557	0.223	62.0	-2.49	0.015	-1.004	-0.111
% Gatherer	REF Post–Pre	0.268	0.237	62.0	1.13	0.263	-0.207	0.743
	TRT Post–Pre	0.257	0.142	62.0	1.81	0.075	-0.027	0.540
% Parasite	REF Post–Pre	-1.251	0.362	62.0	-3.46	0.001	-1.974	-0.528
	TRT Post–Pre	-1.527	0.229	62.0	-6.67	<0.001	-1.984	-1.069
% Predator	REF Post–Pre	-0.138	0.202	52.9	-0.68	0.497	-0.543	0.267
	TRT Post–Pre	0.203	0.134	52.9	1.52	0.135	-0.065	0.470
% Scraper	REF Post–Pre	-0.786	0.305	52.7	-2.58	0.013	-1.398	-0.175
	TRT Post–Pre	-0.625	0.181	52.8	-3.46	0.001	-0.987	-0.263
% Shredder	REF Post–Pre	0.303	0.227	53.0	1.34	0.187	-0.152	0.759
	TRT Post–Pre	0.035	0.137	53.0	0.25	0.801	-0.240	0.310
% Unknown	REF Post–Pre	0.450	0.312	62.0	1.44	0.155	-0.174	1.073
	TRT Post–Pre	0.049	0.200	62.0	0.25	0.806	-0.350	0.448

Descriptive statistics for the functional feeding groups are provided in **Appendix Table 7-1**. The relative proportion of functional feeding groups remained roughly similar between treatments and treatment periods (**Figure 7-2**). Collector-gatherers consistently made up the largest proportion of the feeding groups, and their proportion increased in the REF sites from 28.7 to 34.5% and in the TRT sites from 32.1 to 37.9% in the post-harvest period. The dominant collector-gatherers included *Micropsectra* and *Stempellinella* (Diptera: Chironomidae: Chironominae) and *Diphetor hageni* (Ephemeroptera: Baetidae).

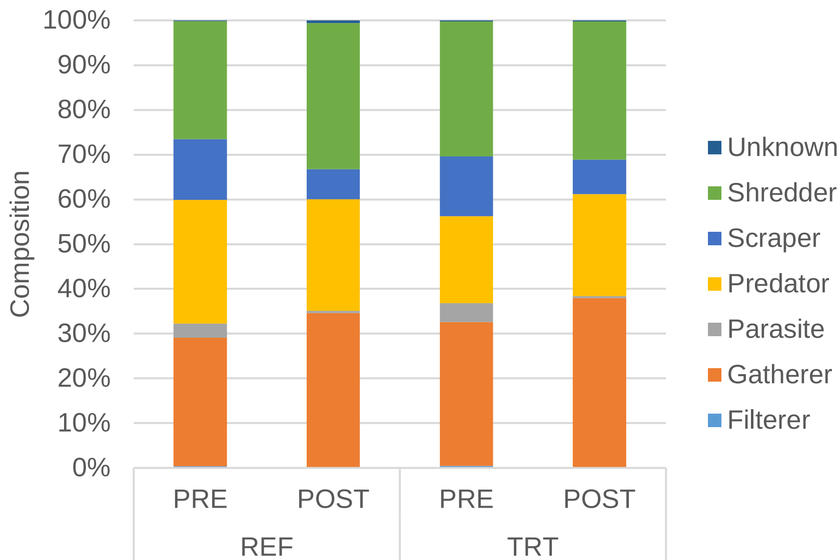


Figure 7-2. Percent composition of the benthic macroinvertebrate samples by functional feeding group, treatment, and treatment period. REF = reference; TRT = treatment; PRE = pre-harvest period; POST = post-harvest period.

Shredders also made up a large proportion of the feeding groups. The proportion of shredders increased in the REF sites from 26.4 to 32.6% and in the TRT sites from 30.1 to 30.9% in the post-harvest period. Shredders consisted predominantly of *Moselia infuscata* (Plecoptera: Leuctridae) and *Zapada cinctipes* and *Zapada columbiana* (Plecoptera: Nemouridae). Predators were abundant in the REF and TRT sites, although their proportion increased in the TRT sites from 19.5 to 22.8% and decreased in the REF sites from 27.6 to 25.0% in the post-harvest period. *Sweltsa* (Plecoptera: Chloroperlidae) was consistently the dominant predator across all treatments and treatment years.

Scrapers were less abundant and their proportion decreased substantially from 13.6 to 6.7% in the REF sites and from 13.4 to 7.7% in the TRT sites in the post-harvest period. Scrapers consisted predominantly of *Cinygma* (Ephemeroptera: Heptageniidae) across all treatments and treatment years. Collector-filterers, parasites, and taxa of the unknown feeding group were present in smaller proportions. Collector-filterers and parasites decreased in the REF and TRT sites in the post-harvest period while unknowns increased in the REF sites but remained unchanged in the TRT sites.

7-4.1.3. Taxonomic Composition

Macroinvertebrate taxa represented in the samples included members of the insect orders Coleoptera (beetles), Diptera (true flies), Ephemeroptera (mayflies), Lepidoptera (butterflies, moths), Megaloptera (alderflies, dobsonflies), Plecoptera (stoneflies), and Trichoptera (caddisflies), as well as arachnids (mites), crustaceans (amphipods, crayfish, isopods, ostracods), molluscs (clams, snails), hydrozoa, annelids (segmented worms), nematodes (roundworms), and platyhelminths (flatworms).

The GLMM and pairwise comparisons suggested there was no treatment × period interaction for the major taxonomic order metrics including percent Coleoptera ($P = 0.313$), Diptera ($P = 0.736$), Ephemeroptera ($P = 0.135$), Plecoptera ($P = 0.485$), and Trichoptera ($P = 0.343$), or the Chironomidae (Diptera) subfamily metrics percent Tanypodinae ($P = 0.961$) or Non-Tanypodinae ($P = 0.584$) (**Tables 7-8** and **7-9**). Differences of least squares means showed that percent Diptera, Tanypodinae, and Non-Tanypodinae increased in the REF and TRT sites from pre- to post-harvest, while percent Trichoptera decreased in the REF and TRT sites (**Table 7-10**). Percent Coleoptera and Plectoptera, however, increased in the REF sites but decreased in the TRT sites, and percent Ephemeroptera decreased in the REF sites but increased in the TRT sites. Percent Ephemeroptera in the REF sites, percent Tanypodinae in the TRT sites, and percent Diptera and Trichoptera in both the REF and TRT sites had P values less than 0.1.

Table 7-8. Results of the generalized linear mixed-effects model for the major taxonomic orders and chironomid (Diptera) subfamilies. The treatment × period interaction terms indicate pre- to post-harvest differences among treatments. Num DF = numerator degrees of freedom; Den DF = denominator degrees of freedom.

Order	Effect	Num DF	Den DF	F Value	Pr > F
% Coleoptera	Treatment	1	62.0	0.10	0.756
	Period	1	62.0	0.23	0.633
	Treatment × Period	1	62.0	1.03	0.313
% Diptera	Treatment	1	62.0	0.08	0.782
	Period	1	62.0	11.96	0.001
	Treatment × Period	1	62.0	0.11	0.736
% Tanypodinae	Treatment	1	62.0	0.82	0.368
	Period	1	62.0	5.24	0.026
	Treatment × Period	1	62.0	0.00	0.961
% Non-Tanypodinae	Treatment	1	62.0	0.05	0.828
	Period	1	62.0	4.42	0.040
	Treatment × Period	1	62.0	0.30	0.584
% Ephemeroptera	Treatment	1	62.0	3.58	0.063
	Period	1	62.0	2.26	0.138
	Treatment × Period	1	62.0	2.30	0.135
% Plecoptera	Treatment	1	62.0	8.19	0.006
	Period	1	62.0	0.20	0.656
	Treatment × Period	1	62.0	0.49	0.485
% Trichoptera	Treatment	1	62.0	1.80	0.185
	Period	1	62.0	16.62	0.000
	Treatment × Period	1	62.0	0.91	0.343

Table 7-9. Pairwise comparisons (reference vs. treatment) for the major taxonomic orders and chironomid (Diptera) subfamilies using the generalized linear mixed-effects model analyses. Estimate, standard error, and confidence interval values in beta space. SE = standard error; DF = degrees of freedom; C.I. = confidence interval.

						95% C.I.	
Order	**Estimate**	**SE**	**DF**	**T-value**	**P-value**	**Lower**	**Upper**
% Coleoptera	-0.411	0.404	62.0	-1.02	0.313	-1.219	0.397
% Diptera	-0.113	0.333	62.0	-0.34	0.736	-0.779	0.553
% Tanypodinae	-0.023	0.468	62.0	-0.05	0.961	-0.959	0.913
% Non-Tanypodinae	-0.193	0.351	62.0	-0.55	0.584	-0.894	0.508
% Ephemeroptera	0.539	0.356	62.0	1.52	0.135	-0.172	1.250
% Plecoptera	-0.188	0.268	62.0	-0.70	0.485	-0.725	0.348
% Trichoptera	0.310	0.325	62.0	0.95	0.343	-0.339	0.960

Table 7-10. Differences of least squares means between the pre- and post-harvest period for the major taxonomic orders and chironomid (Diptera) subfamilies using the generalized linear mixed-effects model analyses. Estimate, standard error, and confidence interval values in beta space. SE = standard error; DF = degrees of freedom; C.I. = confidence interval; REF = reference; TRT = treatment; PRE = pre-harvest period; POST = post-harvest period.

							95% C.I.	
Order	**Comparison**	**Estimate**	**SE**	**DF**	**T-value**	**P-value**	**Lower**	**Upper**
% Coleoptera	REF Post–Pre	0.109	0.346	62.0	0.31	0.755	-0.583	0.800
	TRT Post–Pre	-0.302	0.208	62.0	-1.45	0.152	-0.719	0.114
% Diptera	REF Post–Pre	0.634	0.284	62.0	2.23	0.029	0.066	1.202
	TRT Post–Pre	0.521	0.175	62.0	2.98	0.004	0.172	0.870
% Tanypodinae	REF Post–Pre	0.552	0.405	62.0	1.36	0.178	-0.258	1.362
	TRT Post–Pre	0.529	0.239	62.0	2.22	0.030	0.052	1.006
% Non-Tanypodinae	REF Post–Pre	0.465	0.299	62.0	1.56	0.125	-0.132	1.063
	TRT Post–Pre	0.272	0.184	62.0	1.48	0.144	-0.095	0.640
% Ephemeroptera	REF Post–Pre	-0.537	0.310	62.0	-1.73	0.088	-1.157	0.082
	TRT Post–Pre	0.002	0.174	62.0	0.01	0.991	-0.347	0.351
% Plecoptera	REF Post–Pre	0.154	0.225	62.0	0.69	0.496	-0.296	0.604
	TRT Post–Pre	-0.034	0.146	62.0	-0.23	0.816	-0.326	0.258
% Trichoptera	REF Post–Pre	-0.821	0.284	62.0	-2.89	0.005	-1.389	-0.254
	TRT Post–Pre	-0.511	0.160	62.0	-3.19	0.002	-0.830	-0.191

Descriptive statistics for the major taxonomic order metrics and the Chironomidae subfamily metrics are provided in **Appendix Table 7-1**. Percent composition of each macroinvertebrate taxon is shown in **Appendix Table 7-2**. Diptera and Plecoptera made up the largest proportion of the macroinvertebrate orders (**Figure 7-3**). The proportion of Diptera increased in the REF sites from 25.8 to 41.4% and in the TRT sites from 26.8 to 38.9% in the post-harvest period. About 62% of the Diptera taxa increased in proportion in the REF sites, 37% decreased in proportion, and 2% remained unchanged. The response was comparable in the TRT sites, with 54% of the taxa increasing in proportion, 38% decreasing, and 8% remaining changed (**Appendix Table 7-2**). The dominant dipterans included *Micropsectra* and *Stempellinella*. *Oreogeton* (Empididae), a moderately fine sediment sensitive taxon as scored by the FSBI, was present in the REF sites in the pre-harvest period and in the TRT sites in the pre- and post-harvest period, with the proportion decreasing in the post-harvest period. *Prosimulium* (Simuliidae) and *Rhabdomastix fascigera* (Tipulidae), both slightly sensitive taxa, were present in the REF and TRT sites pre- and post-harvest. The proportion of *Prosimulium* decreased post-harvest in the REF and TRT sites, while the proportion of *Rhabdomastix fascigera* decreased in the REF sites but did not change in the TRT sites.

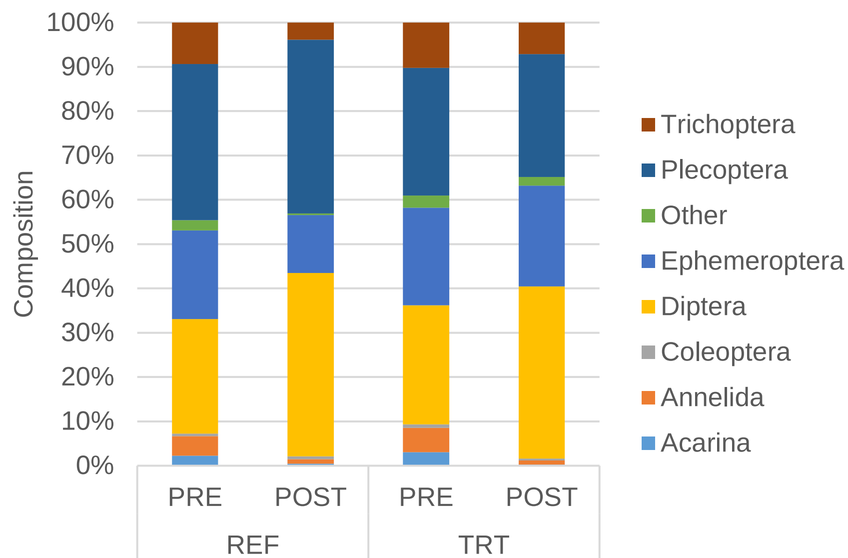


Figure 7-3. Percent composition of the benthic macroinvertebrate samples by dominant taxonomic group or order, treatment, and treatment period. REF = reference; TRT = treatment; PRE = pre-harvest period; POST = post-harvest period.

Plecoptera increased in proportion from 35.3 to 39.3% in the REF sites, but decreased slightly from 28.8 to 27.7% in the TRT sites in the post-harvest period (**Figure 7-3**). About 53% of the Plecoptera taxa increased in proportion and 47% decreased in the REF sites, while 42% increased in proportion, 54% decreased, and 4% remained unchanged in the TRT sites (**Appendix Table 7-2**). Plecopterans consisted predominantly of *Sweltsa*, *Moselia infuscata*, *Zapada cinctipes*, and *Zapada columbiana*. *Despaxia augusta* (Leuctridae), an extremely fine sediment sensitive taxon, was present in the REF sites in the post-harvest period and in the TRT sites in both periods, with the proportion increasing in the post-harvest period. *Doroneuria* (Perlidae), a very fine sediment sensitive taxon, was present in both treatments in both periods, decreasing in proportion in the REF sites post-harvest and increasing in proportion in the TRT sites in the post-harvest period. The response was variable for the moderately fine sediment sensitive taxa *Paraperla* (Chloroperlidae), *Calineuria californica* (Perlidae), and *Pteronarcys* (Pteronarcyidae). *Paraperla* was present in both treatments only in the pre-harvest period, *Calineuria californica* was present in both periods only in the TRT sites and increased in proportion after harvest, and *Pteronarcys* was present only in the TRT sites and only in the pre-harvest period. *Zapada oregonensis* (Nemouridae), a slightly fine sediment sensitive taxon, was present only in the REF sites and only in the post-harvest period. The other slightly fine sediment sensitive taxa, *Zapada frigida* (Nemouridae) and *Yoraperla* (Peltoperlidae), were present in both treatments pre- and post-harvest. *Zapada frigida* increased in proportion after harvest in both treatments, while *Yoraperla* decreased in proportion in both treatments.

Ephemeroptera were also abundant in the REF and TRT sites, although their proportion increased slightly in the TRT sites from 22.1 to 22.7% and decreased in the REF sites from 20.0 to 13.1% in the post-harvest period (**Figure 7-3**). About 57% of the Ephemeroptera taxa increased in proportion and 43% decreased in the REF sites, while 63% increased in proportion and 37% decreased in the TRT sites (**Appendix Table 7-2**). Dominant ephemeropterans included *Diphetor hageni*, *Cinygma*, and Leptophlebiidae and *Paraleptophlebia* (Leptophlebiidae). The very fine sediment sensitive taxa, *Drunella doddsii* (Ephemerellidae) and *Rhithrogena* (Heptageniidae), were both present in the TRT sites pre- and post-harvest, and their proportion increased slightly after harvest. *Drunella doddsii* was also present in the REF sites in the post-harvest period but not the pre-harvest period. The response was variable for the moderately fine sediment sensitive taxa *Drunella coloradensis* (Ephemerellidae) and *Epeorus* (Heptageniidae). *Drunella coloradensis* increased in proportion in the REF sites and decreased in proportion in the TRT sites, while *Epeorus* increased in proportion after harvest in both treatments. The slightly fine sediment sensitive taxa, *Baetis bicaudatus* (Baetidae) and *Ironodes* (Heptageniidae) decreased in proportion after harvest in both treatments, while *Cinygmula* (Heptageniidae) increased in proportion after harvest in both treatments. Of the other slightly fine sediment sensitive taxa, *Acentrella* (Baetidae) was present only in the TRT sites post-harvest and *Drunella* (Ephemerellidae) was present in both treatments only in the pre-harvest period.

Trichoptera were less abundant and their proportion decreased from 9.3 to 3.8% in the REF sites and from 10.3 to 7.1% in the TRT sites in the post-harvest period (**Figure 7-3**). About 10% of the Trichoptera taxa increased in proportion and 90% decreased in the REF sites, while 35% increased in proportion, 62% decreased, and 4% remained unchanged in the TRT sites (**Appendix Table 7-2**). Trichopterans consisted predominantly of *Micrasema* (Brachycentridae), *Lepidostoma* (Lepidostomatidae), and *Wormaldia* (Philopotamidae). The very fine sediment sensitive taxon *Rhyacophila angelita* group (Rhyacophilidae) was present only in the pre-harvest period in the TRT sites, while *Rhyacophila vofixa* group (Rhyacophilidae) was present in both periods and decreased in proportion in both treatments after harvest. *Dolophilodes* (Philopotamidae) and *Rhyacophila narvae* (Rhyacophilidae), moderately fine sediment sensitive taxa, were present only in the TRT sites and decreased in proportion after harvest. The response of other moderately sensitive taxa was more variable. Arctopsychinae (Hydropsychidae) was present only in the pre-harvest period in both treatments, *Rhyacophila betteni* group (Rhyacophilidae) decreased after harvest in the TRT sites and was present in the REF sites only in the pre-harvest period, *Glossosoma* (Glossosomatidae) decreased in proportion in the REF sites and increased in the TRT sites after harvest, and *Neophylax splendens* (Uenoidae) decreased in proportion in both treatments after harvest. The slightly fine sediment sensitive taxon *Anagapetus* (Glossosomatidae) was present only in the TRT sites and decreased in proportion after harvest, while *Parapsyche* (Hydropsychidae) decreased in proportion in the REF sites and increased in the TRT sites after harvest.

Acarina (mites), annelids, beetles, and all remaining taxonomic groups, were present in much smaller proportions and generally decreased in proportion after harvest. The beetle, *Narpus concolor* (Coleoptera: Elmidae), a slightly fine sediment sensitive taxon, increased in proportion in the REF sites, but was not present in the TRT sites.

7-4.2. RELATIONSHIP OF METRICS TO TEMPERATURE AND CANOPY COVER

The correlation of the pre- to post-harvest differences of the metrics, functional feeding groups, major taxonomic orders, and chironomid subfamilies with the July MMTR and mean canopy cover suggested no correlation between most of the metrics, temperature, and canopy cover (**Table 7-11**). The proportion of filterers (P = 0.003, r = 0.799) and non-Tanypodinae (P = 0.061, r = 0.581) increased in sites with a smaller change in canopy cover (**Figure 7-4**). In contrast, the proportion of Ephemeroptera decreased in sites with a smaller change in canopy cover (P = 0.037, r = -0.632) and increased in sites with a higher July MMTR (P = 0.039, r = 0.627). Of the Ephemeroptera taxa, *Baetis* (Baetidae; P = 0.001, r = -0.856), *Baetis tricaudatus* (Baetidae; P = 0.038, r = -0.629), *Diphetor hageni* (P = 0.02, r = -0.685), and *Paraleptophlebia* (P = 0.068, r = -0.569) decreased in sites with a smaller change in canopy cover, while *Baetis piscatoris* (Baetidae; P = 0.012, r = 0.72), *Baetis tricaudatus* (P = 0.005, r = 0.774), and Leptophlebiidae (P = 0.092, r = 0.532) increased in sites with a higher July MMTR. For all of the correlations, including those with P >0.1, there was separation between TRT1a and TRT1b and the other treatment and reference sites, with TRT1a experiencing the largest decrease in canopy cover and TRT1b the largest increase in July MMTR.

Table 7-11. Pearson correlation coefficients and p-values for the 2016 July mean monthly temperature response (MMTR) and the post-harvest change (post-harvest minus pre-harvest) in canopy cover and the macroinvertebrate metrics, functional feeding groups, major taxonomic orders, and chironomid (Diptera) subfamilies. EPT = Ephemeroptera-Plecoptera-Trichoptera; FSBI = fine sediment biotic index.

	% Canopy Cover		**July MMTR**	
	Pearson's *r*	**P-value**	**Pearson's *r***	**P-value**
Metric				
Total Richness	-0.057	0.868	-0.384	0.244
EPT Richness	-0.025	0.942	-0.255	0.450
% EPT	-0.492	0.124	0.298	0.374
Shannon H'	0.007	0.983	-0.435	0.181
FSBI	0.073	0.831	-0.017	0.960
Feeding Group				
% Filterer	0.799	0.003	-0.427	0.191
% Gatherer	0.043	0.900	0.210	0.536
% Parasite	0.035	0.918	-0.097	0.776
% Predator	-0.001	0.999	-0.032	0.925
% Scraper	-0.425	0.192	0.196	0.564
% Shredder	0.129	0.706	-0.222	0.511
% Unknown	0.103	0.762	0.043	0.901
Order				
% Coleoptera	0.402	0.220	-0.418	0.201
% Diptera	0.478	0.137	-0.127	0.711
% Tanypodinae	-0.039	0.909	0.308	0.357
% Non-Tanypodinae	0.581	0.061	-0.381	0.248
% Ephemeroptera	-0.632	0.037	0.627	0.039
% Plecoptera	0.313	0.348	-0.418	0.201
% Trichoptera	-0.486	0.130	0.064	0.852

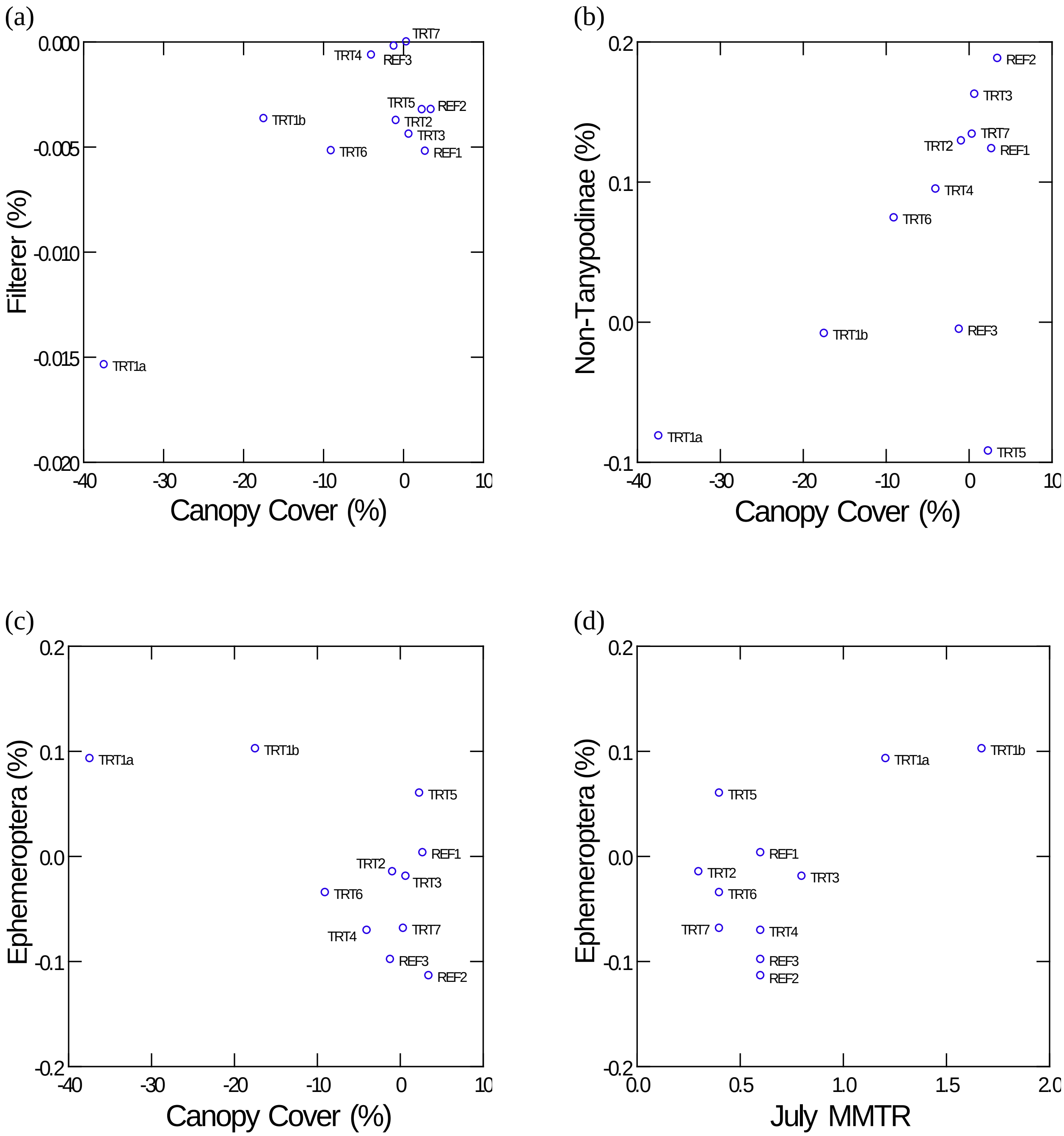


Figure 7-4. Correlation plots for proportion of (a) filterers, (b) non-Tanypodinae, and (c) Ephemeroptera by canopy cover and (d) Ephemeroptera by July mean monthly temperature response (MMTR).

7-4.3. ORDINATION

The NMDS ordination using the macroinvertebrate community data had a stress value of 0.16 and three axes. The results were consistent with those of the GLMM, where the PERMANOVA suggested no treatment × period interaction for the macroinvertebrate assemblage ($R^2 = 0.015$, $P = 0.233$). Samples clustered by treatment period (i.e., Pre vs. Post) and season but the REF and TRT did not differ from each other (**Figures 7-5** and **7-6**). There was separation between some of the variables, where percent filterer, scraper, Ephemeroptera, and Trichoptera correlated with total richness, EPT richness, percent EPT, the Shannon H' diversity index, and the FSBI, and percent gatherer, Diptera, Tanypodinae, and Non-Tanypodinae correlated with year (**Table 7-12**).

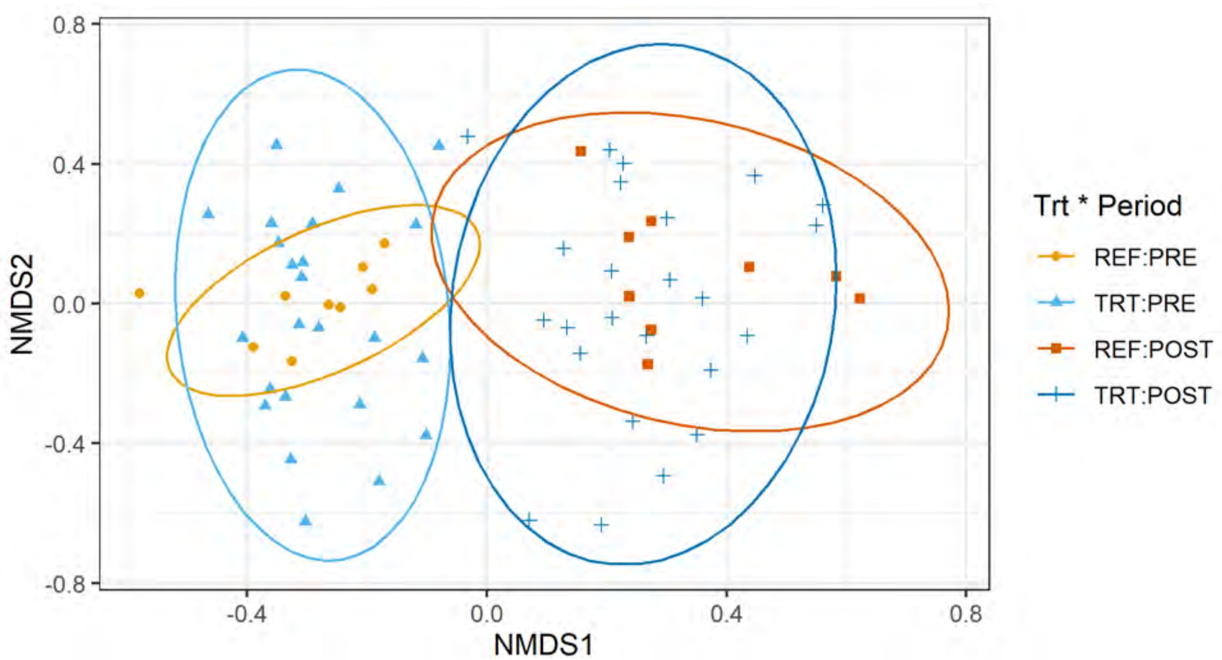


Figure 7-5. Ordination (NMDS axes 1 and 2) of the benthic macroinvertebrate community assemblage by treatment and treatment period. REF = reference; TRT = treatment; PRE = pre-harvest period; POST = post-harvest period.

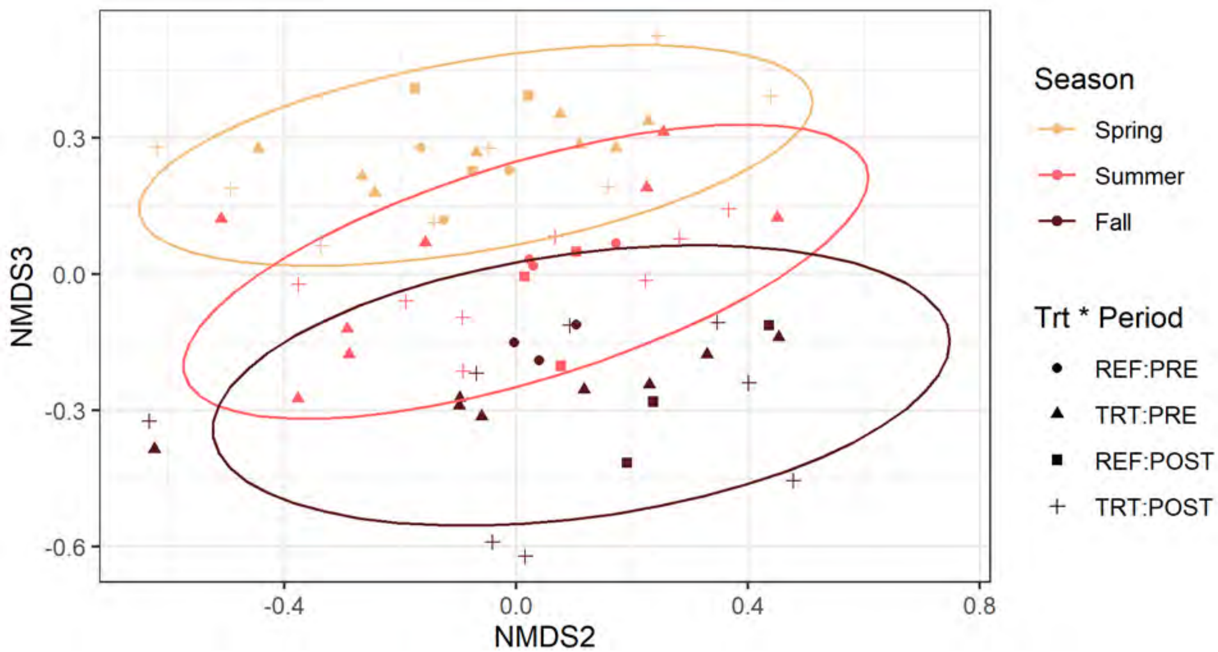


Figure 7-6. Ordination (NMDS axes 2 and 3) of the benthic macroinvertebrate community assemblage by treatment, treatment period, and season. REF = reference; TRT = treatment; PRE = pre-harvest period; POST = post-harvest period.

Table 7-12. Correlation coefficients (*r*; P <0.01) of macroinvertebrate metrics with ordination (NMDS) axes. EPT = Ephemeroptera-Plecoptera-Trichoptera; FSBI = fine sediment biotic index.

Metric	Axis 1 *r*	Axis 2 *r*	Axis 3 *r*
Year	1.000	0.000	0.000
Total Richness	-0.979	-0.197	0.059
EPT Richness	-0.354	-0.934	-0.050
% EPT	-0.319	-0.929	-0.189
Shannon H'	-0.807	-0.421	-0.413
FSBI	-0.048	-0.990	0.130
% Filterer	-0.772	0.022	0.636
% Gatherer	0.906	0.207	0.369
% Parasite	-0.897	0.438	-0.054
% Scraper	-0.538	-0.834	0.127
% Coleoptera	-0.349	-0.202	-0.915
% Diptera	0.691	0.712	0.127
% Tanypodinae	0.607	0.680	0.411
% Non-Tanypodinae	0.602	0.705	0.374
% Ephemeroptera	-0.132	-0.954	0.270
% Trichoptera	-0.766	-0.564	-0.308

We used NMDS to assess whether there was a relationship between FSBI and macroinvertebrate assemblages among sites and then compared the FSBI to stream slope calculated using LiDAR with a 1-m resolution and substrate sizes collected in 2012 at 40-m interval cross sections (unpublished data). We found that the sites with higher stream slopes and a higher proportion of boulder and cobble within the macroinvertebrate sampling reach had higher FSBI scores (**Table 7-13**; **Figure 7-7**). In contrast, the sites with lower stream slopes and a higher proportion of sand and fine sediments in the sampling reach had lower FSBI scores.

Table 7-13. Mean fine sediment biotic index (FSBI), stream slope, and substrate characteristics. Stream slopes were calculated using LiDAR with a 1-m resolution. Substrate sizes were collected in 2012 at 40-m interval cross sections. REF = reference; TRT = treatment.

Site	FSBI	Stream Slope (%)	Bedrock (%)	Boulder (%)	Cobble (%)	Gravel (%)	Sand (%)	Fine (%)
REF1	59	6.0	0.0	0.0	0.0	100.0	0.0	0.0
REF2	53	11.5	0.0	0.0	0.0	60.0	40.0	0.0
REF3	45	9.0	0.0	0.0	0.0	56.9	36.1	6.9
TRT1a	60	17.7	0.0	0.0	50.0	37.5	0.0	12.5
TRT1b	58	11.7	20.5	0.0	1.1	28.3	5.0	45.0
TRT2	18	9.3	0.0	0.0	0.0	46.6	15.4	38.0
TRT3	44	5.5	0.0	0.0	14.7	35.3	0.0	50.0
TRT4	64	19.1	0.0	4.2	22.5	60.7	8.3	4.3
TRT5	23	5.3	0.0	0.0	0.0	33.3	30.6	36.1
TRT6	62	14.0	0.0	0.0	0.0	89.2	8.3	2.5
TRT7	121	17.5	0.0	0.0	16.2	76.3	7.5	0.0

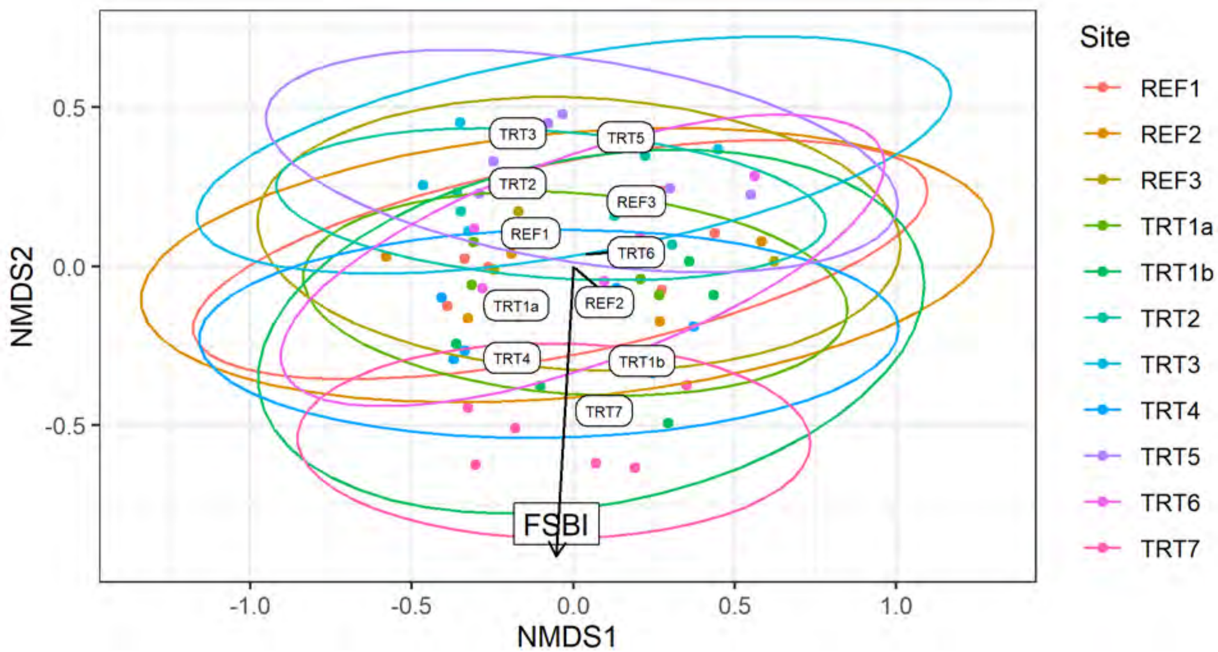


Figure 7-7. Ordination (NMDS axes 1 and 2) of the benthic macroinvertebrate community assemblage by site and fine sediment biotic index (FSBI). REF = reference; TRT = treatment.

7-5. DISCUSSION

We expected that changes in shade, temperature, and sediment processes may result in a decrease in EPT taxa, diversity, and the FSBI while favoring less sensitive species. Although the GLMM suggested a possible treatment × period interaction for EPT richness ($P = 0.061$) and the Shannon H' diversity index ($P = 0.092$), the metrics decreased in both the reference and treatment sites with a larger decrease in the references. Similarly, though there was no treatment × period interaction for total richness ($P = 0.311$), percent EPT ($P = 0.535$), or the FSBI ($P = 0.947$), these metrics decreased from pre- to post-harvest in both the reference and treatment sites with larger decreases in the references. The ordination and PERMANOVA results confirmed that there was no treatment × period interaction ($P = 0.233$) and that the reference and treatment sites moved in tandem. This may indicate that broader environmental factors, such as weather, rather than a treatment effect, were influencing macroinvertebrate assemblages.

The Soft Rock Study sites supported a mean total richness of 53 and mean EPT richness of 20 across all sites and periods. This is comparable with another study that found a mean total and EPT richness of 45 and 19, respectively, in the Oregon Coast Range (Cole *et al.* 2003). We did not detect a change in total richness, which is consistent with other studies that found no difference in total richness between their reference and harvested sites (Newbold *et al.* 1980; Danehy *et al.* 2007; Wilkerson *et al.* 2010; Richardson and Béraud 2014). In contrast, while we detected a decrease in EPT richness and the Shannon H' diversity index, other researchers found no difference in these metrics between their reference and treatment sites (Stone and Wallace 1998; Danehy *et al.* 2007; Gravelle *et al.* 2009). EPT richness typically decreases in response to increasing fine sediment (Angradi 1999; Zweig and Rabeni 2001), but our FSBI results did not indicate a change in sediment sensitive taxa post-harvest. As with other studies, we found no change in percent EPT between reference and harvested streams (Danehy *et al.* 2007; Kreutzweiser *et al.* 2008; Richardson and Béraud 2014).

We used the FSBI to assess the potential impact of fine sediment resulting from timber harvest on sediment sensitive taxa (Relyea *et al.* 2012). We found no change in the FSBI scores, and the means were comparable between the reference and treatment sites before and after harvest. Gravelle and colleagues (2009) also found no change in their FSBI scores after logging road construction and harvest. The Soft Rock Study sites had a mean FSBI score of 55 across all sites and periods, which is lower than Gravelle and colleagues' (2009) mean FSBI score of 218 for northern Idaho streams. Our FSBI scores fall slightly over the 10th percentile when compared with the 1025 streams used to develop the FSBI and is more comparable to the median FSBI score of 70 developed for the Coast Range ecoregion consisting of streams with the highest median percentage of fine sediments (Relyea *et al.* 2012). Despite our lower FSBI scores, *Despaxia augusta*, an extremely fine sediment sensitive taxon, and *Drunella doddsii*, *Doroneuria*, and *Rhyacophila* Vofixa group, all very fine sediment sensitive taxa (Reylea *et al.* 2012), were present in both the reference and treatment sites pre- and post- harvest. Interestingly, the ordination of FSBI and macroinvertebrate assemblages among sites showed that the sites with higher gradients and thus larger substrate, as measured at the onset of the study, had higher FSBI scores.

The use of richness metrics and the FSBI may not have been sufficient to detect changes in the macroinvertebrate community after harvest. Other studies have also indicated that richness metrics were not sensitive enough to detect the effects of forest harvest in their study streams (Stone and Wallace 1998; Herlihy *et al.* 2005; Coe *et al.* 2013). Richness does not detect species turnover and the loss of one species may be offset by gain in another (Coe *et al.* 2013). Chironomidae were the dominant taxa group in our samples pre- and post-harvest, but the FSBI excluded Chironomidae in the development of the index (Reylea *et al.* 2012). The FSBI may not be sensitive enough for a site-specific before-after control-impact study and the large standard error may indicate that the sample size was too small for use of the metric. In addition, large changes in taxa density may occur in response to an increase in fine sediments but taxa composition may remain unchanged. While we measured streambed substrate conditions at transects throughout the study sites at the onset of the study, we did not repeat this effort post-harvest or at the macroinvertebrate sampling sites. The presence of sediment sensitive taxa before and after harvest is consistent with the apparent lack of a treatment response we observed in the suspended sediment data (see Chapter 5 – *Discharge and Suspended Sediment Export*), but without a concurrent measure of substrate characteristics at our sampling sites we are unable to draw strong conclusions regarding the FSBI or presence or absence of sediment sensitive taxa at the microhabitat scale.

While we detected a possible decrease in EPT richness after harvest, there was no treatment × period interaction in the proportion of Ephemeroptera ($P = 0.135$), Plecoptera ($P = 0.485$), or Trichoptera ($P = 0.343$). The mean proportion of Ephemeroptera decreased in the reference sites and increased in the treatment sites after harvest, whereas the mean proportion of Plecoptera increased in the reference sites and decreased in the treatment sites. Trichoptera decreased in proportion in both the reference and treatment sites after harvest. EPT taxa tend to be more sensitive to high water temperatures than other macroinvertebrate groups (Stewart *et al.* 2013), but sensitivity also varies within families with some species more sensitive and others more tolerant (Dallas and Rivers-Moore 2012). Although 14 Plecoptera and 16 Trichoptera taxa decreased in the treatment sites after harvest, most of these taxa were present in small proportions in both the pre- and post-harvest periods and did not decrease by more than 0.5%. The taxa that decreased in proportion by more than 0.5% in the treatment sites included *Sweltsa*, Leuctridae, *Moselia infuscata*, *Zapada columbiana*, *Micrasema*, *Lepidostoma*, and *Rhyacophila* Vofixa group. Of these taxa, *Rhyacophila* Vofixa group is the only taxon included in the FSBI as sediment sensitive. All seven taxa prefer cold-cool water temperatures in the 0-15°C range (Vieira *et al.* 2006). Maximum daily temperatures increased above this range only in TRT1 in July and August of the post-harvest macroinvertebrate sampling period (see Chapter 4 – *Stream Temperature and Cover*). Temperatures were also higher in the reference sites in the post-harvest period and most of these taxa, with the exception of *Zapada columbiana* and *Micrasema*, also decreased in the reference sites. However, other taxa that prefer temperatures in the 0-15°C range, including *Despaxia augusta*, *Zapada cinctipes*, and *Parapsyche*, increased in proportion by more than 0.5% despite the warmer temperatures in the post-harvest period.

There was no treatment × period interaction in the proportion of Diptera ($P = 0.736$) or the Chironomidae subfamily metrics Tanypodinae ($P = 0.961$) or Non-Tanypodinae ($P = 0.584$). The mean proportion of Diptera increased in both the reference and treatment sites after harvest. Chironomidae comprised about 25.6% of all taxa and 75.5% of all Diptera taxa across all sites and periods, and the mean proportion of chironomids increased in both the reference and treatment sites after harvest. Of the chironomid subfamilies, Orthocladiinae and Chironominae made up 47.4% and 42.4% of all chironomid taxa across sites and periods, respectively, while Tanypodinae comprised 8.8%. Chironomids typically make up a large proportion of macroinvertebrate assemblages as they are fast colonizers, adaptable to frequent disturbance, and able to use a variety of food resources (Entrekin *et al.* 2007; Wilkerson *et al.* 2010). They also have short life cycles and are able to produce multiple generations in one year (i.e., multivoltine). Warmer temperatures increase the growth rate of some chironomid taxa and may influence voltinism (Huryn 1990). Other studies have documented a predominance of chironomids in samples from both reference and harvested streams (e.g., Noel *et al.* 1986; Wilkerson *et al.* 2010), and an increase in chironomid abundance after harvest (Collier and Smith 2005). Danehy and colleagues (2007) found a positive correlation between Chironomidae, fine sediment, and insolation in a comparison of macroinvertebrate assemblages between mature, thinned, and clearcut forest stands.

We predicted that the removal of trees may reduce organic matter inputs and increase autotrophic production, causing a shift in the energy balance of the streams that may influence macroinvertebrate assemblages. There was, however, no treatment × period interaction in the proportion of any of the macroinvertebrate functional feeding groups. Other studies also found no change in feeding group composition in streams following harvest (e.g., Danehy *et al.* 2007; Gravelle *et al.* 2009). Gatherers and shredders comprised the majority of the samples with a mean of 33.3% and 30.0%, respectively, across all sites and periods, and their proportion increased after harvest in the reference and treatment sites. Predators averaged 23.7%, scrapers 10.3%, and filterers 0.2%, across all sites and periods. Aside from percent predators, which trended downward in reference sites but upward in the treatment sites, the remaining feeding groups decreased in proportion after harvest in the reference and treatment sites. Other studies found an increase in numbers and/or biomass of gatherers in unbuffered streams following harvest (Hawkins *et al.* 1982; Haggerty *et al.* 2004; Hernandez *et al.* 2005), and noted that gatherers were dominant in all of the study sites before and after harvest (Gravelle *et al.* 2009). Danehy and colleagues (2007) also found that gatherers and shredders made up at least half of the invertebrate community in their streams and observed smaller proportions of filterers and scrapers.

The lack of major changes in the benthic community may reflect the larger buffers and wood inputs after harvest. The length of stream buffered varied from 40 to 100% with a mean width of 14 to 23 m (**Table 2-4**), and mean canopy closure decreased to 71% after harvest (see Chapter 4 – *Stream Temperature and Cover*). Wood and vegetation cover measured at 40-m interval cross sections increased from a mean of 75% in the pre-harvest period to 113% post-harvest in the treatment sites, a 55% increase compared to only a 9% increase in the reference sites (unpublished data). The extensive buffers combined with an increase in wood cover from logging slash and windthrow (see Chapter 3 – *Riparian Stand Structure and Wood Recruitment*) and subsequent vegetation growth may have provided enough shade to inhibit algal growth and thus scraper production. Instream wood may have increased retention of particulate organic matter (Hetrick *et al.* 1998; Brookshire and Dwire 2003) and warmer temperatures may have increased microbial activity associated with processing organic matter, improving food quality for gatherers and shredders (Merritt and Cummins 1996; McKie and Malmqvist 2009). On the other hand, wood debris may have trapped fine sediments (Jackson *et al.* 2001), which may have negatively affected filterers and scrapers. Both feeding groups are generally intolerant of fine sediment, as suspended sediment clogs filterer feeding structures and deposited sediment eliminates habitat for filterer attachment and inhibits algal growth and scraper grazing (Newcombe and MacDonald 1991; Rabeni *et al.* 2005). Although Danehy and colleagues (2007) found an increase in primary production in their clearcut sites, they did not find a change in scrapers, which they attributed to the dominance of a diatom resistant to grazing.

The correlation analysis found that filterers increased in proportion in sites with smaller changes in canopy cover ($P = 0.003$). The chironomid subfamilies in the non-Tanypodinae metric, which consist primarily of gatherers, exhibited a similar response ($P = 0.061$). While we did not collect microhabitat information at the sampling locations, we could assume that these sites experienced smaller changes in insolation and particulate organic matter, wood, and sediment inputs at the microhabitat scale may have created or maintained conditions favorable for filterers and non-Tanypodinae taxa. On the other hand, Ephemeroptera increased in proportion in sites with larger decreases in canopy cover ($P = 0.037$) and increases in July MMTR ($P = 0.039$). Four taxa from the family Baetidae and two from the family Leptophlebiidae were driving this response. Baetids and leptophlebiids are predominantly gatherers, and baetids are able to produce multiple generations in one year. Wallace and Gurtz (1986) noted that changes in stream temperatures and food availability following timber harvest led to higher growth rates of some mayflies, such as *Baetis*, whereas Stewart and colleagues (2013) found that mayflies were the most thermally sensitive of the taxa examined in their study. July MMTR may not be the best metric for assessing macroinvertebrate response to temperature, but mean monthly maximum daily temperatures were not that high and exceeded 15°C only in TRT1 in July and August of the post-harvest macroinvertebrate sampling period (see Chapter 4 – *Stream Temperature and Cover*). Temperatures were elevated in the spring and fall in all sites and the highest MMTRs occurred in the spring or fall months in treatments where the buffered stream length exceeded 90%. Elevated temperatures throughout most of the year could affect various life stages of some invertebrates through its influence on growth rates and metabolism (Minshall 1984; Merritt and Cummins 1996), voltinism (Huryn 1990), and emergence (Li *et al.* 2011).

An increase in stream flows and/or sediment inputs after harvest could dislodge macroinvertebrate from the substrate (Hershey and Lamberti 1998) or initiate drift behavior (Culp *et al.* 1986; Waters 1995; Shaw and Richardson 2001; Suren and Jowett 2001). While there was an increase in discharge in the post-harvest period, the pre-harvest period was very dry in comparison and the four sites monitored for stream flow responded differently, likely due to the high spatial variability of precipitation in the region (see Chapter 5 – *Discharge and Suspended Sediment Export*). Aspect may have also been a factor, given that the two reference sites monitored for flow were southwest facing while the two treatment sites were northwest facing. Our suspended sediment export results show that cumulative sediment export was greater in the reference sites than in the treatment sites but that export was more related to specific, stochastic events rather than to harvest. While there was site-specific variability in discharge, aspect, and erodibility in the four sites monitored for flow, the other seven sites sampled for macroinvertebrates were not monitored for flow, so we do not know if there were differences in flows or suspended sediment in those sites that may have influenced macroinvertebrate assemblages. In addition, we did not sample substrate at the macroinvertebrate sampling locations so we do not have a complete picture of sediment impacts on assemblages at the microhabitat scale. Headwater streams may have varying degrees of flow permanence and other studies have found changes in macroinvertebrate communities in response to flow permanence (Feminella 1996; DelRosario and Resh 2000; Price *et al.* 2003; Clarke *et al.* 2010; Stubbington *et al.* 2017). The Soft Rock Study streams were largely perennial and although some portions of the sites dried during the summer, there were no dry sections in the 100-m reach during macroinvertebrate sampling and most of the reaches with intermittent flow and drying were located higher in the stream basins.

Limitations of the sample methodology included a small number of samples and sample representativeness. Samples were collected from three reference sites and eight treatment sites in the spring, summer, and fall of one pre-harvest year and one post-harvest year. This left us with an uneven number of replicates for each treatment and only three samples in each treatment period, resulting in limited statistical power. An extra year of data from the pre- and post-harvest periods may have decreased variability and enabled us to differentiate between a treatment effect and environmental conditions. In addition, continued sampling in the post-harvest period would have given us insight into the response of the macroinvertebrate assemblage to post-harvest stand development (Warren *et al.* 2016) and to climate change as changes in temperature and flow regime may lead to changes in community composition and food webs (Poff *et al.* 2010, 2018; Chadd *et al.* 2017; Kroll *et al.* 2017; Stubbington *et al.* 2017; Aspin *et al.* 2018; Chessman 2018). Unfortunately, resampling the sites is no longer an option as some of the reference sites have been harvested.

We collected macroinvertebrate samples from the lower 100 m of each stream to maximize the treatment effect. The lower 100-m reach of each stream was buffered, however, and the buffer may have minimized changes in shade, temperature, and sediment delivery. In contrast, the upper reaches were largely unbuffered and may have had larger inputs of logging slash that contributed to sediment retention. Jackson and colleagues (2001) found that wood debris in harvested streams trapped sediment and increased the percentage of fine sediments.

Macroinvertebrate assemblages in the sampling reach thus may have differed from those in the upper reaches. In addition, flow in the upper reaches was often intermittent during the summer. Because macroinvertebrates drift short distances (less than 100 m) during periods of low flow (Danehy *et al.* 2011), invertebrates from the upper reaches were probably isolated and not represented in our samples. We collected invertebrates from riffles and slow moving pools, but did not sample pools where there was insufficient flow to direct invertebrates into the net. Riffles generally have higher taxa richness but lower densities than pools and taxa in riffles are adapted for attachment, clinging, or avoiding direct contact with the current while taxa in pools are adapted for sprawling, climbing, or burrowing (Minshall 1984). Although our methodology consistently sampled the same habitat types, we may have missed taxa that were limited to pools with no flow.

In conclusion, we observed some changes after harvest, but there were no major reductions in benthic macroinvertebrates or shifts in functional feeding groups associated with the treatment within the limitations of our study design and sampling methodology. Our study results are applicable to perennial, non-fish-bearing stream basins with marine sedimentary lithologies in western Washington. Because we were unable to include sites with freshwater sedimentary and glacial till lithologies, the reader should use caution when extrapolating these results to the other lithologies.

7-6. REFERENCES

Allan, J.D. 1995. *Stream Ecology: Structure and Function of Running Water*. Kluwer Academic, Boston, MA.

Allan, J.D. 2004. Landscapes and riverscapes: The influence of land use on stream ecosystems. *Annual Review of Ecology, Evolution, and Systematics*. 35:257-284.

Angradi, T.R. 1999. Fine sediment and macroinvertebrate assemblages in Appalachian streams: A field experiment with biomonitoring applications. *Journal of the North American Benthological Society* 18:49-66.

Aspin, T.W.H., K. Khamis, T.J. Matthews, A.M. Milner, M.J. O’Callaghan, M. Trimmer, G. Woodward, and M.E. Ledger. 2018. Extreme drought pushes stream invertebrate communities over functional thresholds. *Global Change Biology* 25:230-244.

Barbour, M.T., and J. Gerritson. 1996. Subsampling of benthic samples: A defense of the fixed-count method. *Journal of the North American Benthological Society* 15:386-391.

Benda, L.E., M.A. Hassan, M. Church, and C.L. May. 2005. Geomorphology of steepland headwaters: The transition from hillslopes to channels. *Journal of the American Water Resources Association* 41:835-851.

Beschta, R.L. 1978. Long term patterns of sediment production following road construction and logging in the Oregon coast range. *Water Resources Research* 14:1011-1016.

Bilby, R.E. 1981. Role of organic debris dams in regulating the export of dissolved and particulate matter from a forested watershed. *Ecology* 62:1234-1243.

Bilby, R.E., and P.A. Bisson. 1992. Allochthonous versus autochthonous organic matter contributions to the trophic support of fish populations in clear-cut and old-growth forested streams. *Canadian Journal of Fisheries and Aquatic Sciences* 49:540-551.

Bilby, R.E., and G.E. Likens. 1980. Importance of organic debris dams in the structure and function of stream ecosystems. *Ecology* 61:1107-1113.

Brookshire, E.N.J., and K.A. Dwire. 2003. Controls on patterns of coarse organic particle retention in headwater streams. *Journal of the North American Benthological Society* 22:17-34.

Brown, G.W., and J.T. Krygier. 1970. Effects of clear-cutting on stream temperature. *Water Resources Research* 6:1133-1139.

Bywater-Reyes, S., C. Segura, and K.D. Bladon. 2017. Geology and geomorphology control suspended sediment yield and modulate increases following timber harvest in temperate headwater streams. *Journal of Hydrology* 548:754-769.

Chadd, R.P., J.A. England, D. Constable, M.J. Dunbar, C.A. Extence, D.J. Leeming, J.A. Murray-Bligh, and P.J. Wood. 2017. An index to track the ecological effects of drought development and recovery on riverine invertebrate communities. *Ecological Indicators* 82:344-356.

Chessman, B.C. 2018. Dissolved-oxygen, current, and temperature preferences of stream invertebrates estimated from field distributions: Application to assemblage responses to drought. *Hydrobiologia* 809:141-153.

Clarke, A., R.M. Nally, N. Bond, and P.S. Lake. 2010. Flow permanence affects aquatic macroinvertebrate diversity and community structure in three headwater streams in a forested catchment. *Canadian Journal of Fisheries and Aquatic Sciences* 67:1649-1657.

Coe, H.J., X. Wei, and P.M. Kiffney. 2013. Linking forest harvest and landscape factors to benthic macroinvertebrate communities in the interior of British Columbia. *Hydrobiologia* 717:65-84.

Cole, M.B., K.R. Russell, and T.J. Mabee. 2003. Relation of headwater macroinvertebrate communities to in-stream and adjacent stand characteristics in managed second-growth forests of the Oregon coast range mountains. *Canadian Journal of Forest Research* 33:1433-1443.

Collier, K.J., and B.J. Smith. 2005. Effects of progressive catchment harvesting on stream invertebrates in two contrasting regions of New Zealand’s North Island. *Marine and Freshwater Research* 56:57-68.

Courtemanch, D.L. 1996. Commentary on the subsampling procedures used for rapid bioassessments. *Journal of the North American Benthological Society* 15:381-385.

Culp, J.M., F.J. Wrona, and R.W. Davies. 1986. Response of stream benthos and drift to fine sediment deposition versus transport. *Canadian Journal of Zoology* 64:1345-1351.

Cummins, K.W., J.R. Sedell, F.J. Swanson, G.W. Minshall, S.G. Fisher, C.E. Gushing, R.C. Petersen, and R.L. Vannote. 1983. Organic matter budgets for stream ecosystems: Problems in their evaluation. Pages 299-353 *in* R. Barnes and G.W. Minshall (eds.). *Stream Ecology: Application and Testing of General Ecological Theory*. Plenum Press, New York, NY.

Dallas, H.F., and Rivers-Moore, N.A. 2012. Critical thermal maxima of aquatic macroinvertebrates: Towards identifying bioindicators of thermal alteration. *Hydrobiologia* 679:61-76.

Danehy, R.J., S.S. Chan, G.T. Lester, R.B. Langshaw, and T.R. Turner. 2007. Periphyton and macroinvertebrate assemblage structure in headwaters bordered by mature, thinned, and clearcut Douglas-fir stands. *Forest Science* 53:294-307.

Danehy, R.J., R.B. Langshaw, S.D. Duke, and R.E. Bilby. 2011. Drift distance of macroinvertebrates throughout summer in headwater tributaries of the Calapooia River. *Fundamental and Applied Limnology/Archi fur Hydrobiologie* 178:111-120.

DelRosario, R.B., and V.H. Resh. 2000. Invertebrates in intermittent and perennial streams: Is the hyporheic zone a refuge from drying? *Journal of the North American Benthological Society* 19:680-696.

Entrekin, S.A., J.B. Wallace, and S.L. Eggert. 2007. The response of Chironomidae (Diptera) to a long-term exclusion of terrestrial organic matter. *Hydrobiologia* 575:401-413.

Feller, M.C., and J.P. Kimmins. 1984. Effects of clearcutting and slash burning on streamwater chemistry and watershed nutrient budgets in southwestern British Columbia. *Water Resources Research* 20:29-40.

Feminella, J.W. 1996. Comparison of benthic macroinvertebrate assemblages in small streams along a gradient of flow permanence. *Journal of the North American Benthological Society* 15:651-669.

Francoeur, S.N., B.J.F. Biggs, R.A. Smith, and R.L. Lowe. 1999. Nutrient limitation of algal biomass accrual in streams: Seasonal patterns and a comparison of methods. *Journal of the North American Benthological Society* 18:242-260.

Freeman, M.C., C.M. Pringle, and C.R. Jackson. 2007. Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. *Journal of the American Water Resources Association* 43:5-14.

Fuchs, S.A., S.G. Hinch, and E. Mellina. 2003. Effects of streamside logging on stream macroinvertebrate communities and habitat in the sub-boreal forests of British Columbia, Canada. *Canadian Journal of Forest Research* 33:1408-1415.

Gomi, T., R.D. Moore, and A.S. Dhakal. 2006. Headwater stream temperature response to clear-cut harvesting with different riparian treatments, coastal British Columbia, Canada. *Water Resources Research* 42:1-11.

Gomi, T., R.C. Sidle, and J.S. Richardson. 2002. Understanding processes and downstream linkages of headwater systems. *Bioscience* 52:905-916.

Gravelle, J.A., T.E. Link, J.R. Broglio, and J.H. Braatne. 2009. Effects of timber harvest on aquatic macroinvertebrate community composition in a northern Idaho watershed. *Forest Science* 44:352-366.

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones: Focus on links between land and water. *Bioscience* 41:540-551.

Grizzel, J.D., and N. Wolff. 1998. Occurrence of windthrow in forest buffer strips and its effect on small streams in northwest Washington. *Northwest Science* 72:214-223.

Haggerty, S.M., D.P. Batzer, and C.R. Jackson. 2004. Macroinvertebrate response to logging in coastal headwater streams of Washington, U.S.A. *Canadian Journal of Fisheries and Aquatic Sciences* 61:529-537.

Harr, R.D., and R.L. Fredriksen. 1988. Water quality after logging small watersheds within the Bull Run watershed, Oregon. *Water Resources Bulletin* 24:1103-1111.

Hawkins, C.P., J.N. Hogue, L.M. Decker, and J.W. Feminella. 1997. Channel morphology, water temperature, and assemblage structure of stream insects. *Journal of the North American Benthological Society* 16:728-749.

Hawkins, C.P., M.L. Murphy, and N.H. Anderson. 1982. Effects of canopy, substrate composition, and gradient on the structure of macroinvertebrate communities in Cascade range streams of Oregon. *Ecology* 63:1840-1856.

Herlihy, A.T., W.J. Gerth, J. Li, and J.L. Banks. 2005. Macroinvertebrate community response to natural and forest harvest gradients in western Oregon headwater streams. *Freshwater Biology* 50:905-919.

Hernandez, O., R.W. Merritt, and M.S. Wipfli. 2005. Benthic invertebrate community structure is influenced by forest succession after clearcut logging in southeastern Alaska. *Hydrobiologia* 533:45-59.

Hershey, A.E., and G.A. Lamberti. 1998. Stream macroinvertebrate communities. Pages 169-199 *in* R.J. Naiman and R.E. Bilby (eds.). *River Ecology and Management: Lessons from the Pacific Coastal Ecoregion*. Springer-Verlag, New York, NY.

Hetrick, N.J., M.A. Brusven, W.R. Meehan, and T.C. Bjornn. 1998. Changes in solar input, water temperature, periphyton accumulation, and allochthonous input and storage after canopy removal along two small salmon streams in southeast Alaska. *Transactions of the American Fisheries Society* 127:859-875.

Hodkinson, I.D., and J.K. Jackson. 2005. Terrestrial and aquatic invertebrates as bioindicators for environmental monitoring, with particular reference to mountain ecosystems. *Environmental Management* 35:649-666.

Huryn, A.D. 1990. Growth and voltinism of lotic midge larvae: Patterns across an Appalachian Mountain basin. *Limnology and Oceanography* 35:339-351.

Jackson, C.R., C.A. Sturm, and J.M. Ward. 2001. Timber harvest impacts on small headwater stream channels in the coast ranges of Washington. *Journal of the American Water Resources Association* 37:1533-1549.

Karwan, D.L., J.A. Gravelle, and J.A. Hubbart. 2007. Effects of timber harvest on suspended sediment loads in Mica Creek, Idaho. *Forest Science* 53:181-188.

Kenward, M.G., and J.H. Roger. 1997. Small sample inference for fixed effects from restricted maximum likelihood. *Biometrics* 53:983-997.

Kiffney, P.M., and J.S. Richardson. 2001. Interactions among nutrients, periphyton, and invertebrate and vertebrate (*Ascaphus truei*) grazers in experimental channels. *Copeia* 2001:422-429.

Kiffney, P.M., J.S. Richardson, and J.P. Bull. 2003. Responses of periphyton and insects to experimental manipulation of riparian buffer width along forest streams. *Journal of Applied Ecology* 40:1060-1076.

Kreutzweiser, D.P., K.P. Good, S.S. Capell, and S.B. Holmes. 2008. Leaf-litter decomposition and macroinvertebrate communities in boreal forest streams linked to upland logging disturbance. *Journal of the North American Benthological Society* 27:1-15.

Kroll, S.A., N.H. Ringler, M.D.L.C. Cano Costa, and J. De las Heras Ibanez. 2017. Macroinvertebrates on the front lines: Projected community response to temperature and precipitation changes in Mediterranean streams. *Journal of Freshwater Ecology* 32:513-528.

Li, J.L., S.L. Johnson, and J. Banks Sobota. 2011. Three responses to small changes in stream temperature by autumn-emerging aquatic insects. *Journal of the North American Benthological Society* 30:474-484.

MacDonald, J.S., P.G. Beaudry, E.A. MacIsaac, and H.E. Herunter. 2003. The effects of forest harvesting and best management practices on streamflow and suspended sediment concentrations during snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada. *Canadian Journal of Forest Research* 33:1397-1407.

McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn (technical coordinators). 2018. *Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing Streams on Competent Lithologies in Western Washington*. Cooperative Monitoring, Evaluation, and Research Report, CMER 18-100. Washington Department of Natural Resources, Olympia, WA. 890 p.

McKie, B.G., and B. Malmqvist. 2009. Assessing ecosystem functioning in streams affected by forest management: Increased leaf decomposition occurs without changes to the composition of benthic assemblages. *Freshwater Biology* 54: 2086-2100.

Merritt, R.W., and K.W. Cummins. 1996. *An Introduction to the Aquatic Insects of North America*. 3rd edition. Kendall Hunt Publishing Co., Dubuque, IA.

Minshall, G.W. 1984. Aquatic insect-substratum relationships. Pages 358-400 in V.H. Resh and D.M. Rosenberg (eds.). *The Ecology of Aquatic Insects*. Praeger Publishers, New York, NY.

Moore, R.D., and S.M. Wondzell. 2005. Physical hydrology and the effects of forest harvesting in the Pacific Northwest: A review. *Journal of the American Water Resources Association* 41:763-784.

Muchow, C.L., and J.S. Richardson. 2000. Unexplored diversity: Macroinvertebrates in coastal British Columbia headwater streams. Pages 15-19 *in* L.M. Darling (ed.). *Proceedings of Biology and Management of Species and Habitat at Risk*, February 1999. Kamloops, BC.

Murphy, M.L., and J.D. Hall. 1981. Varied effects of clear-cut logging on predators and their habitat in small streams of the Cascade Mountains, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences* 38:137-145.

Murphy, M.L., C.P. Hawkins, and N.H. Anderson. 1981. Effects of canopy modification and accumulated sediment on stream communities. *Transactions of the American Fisheries Society* 110:469-478.

Newbold, J.D., D.C. Erman, and K.B. Roby. 1980. Effects of logging on macroinvertebrates in streams with and without buffer strips. *Canadian Journal of Fisheries and Aquatic Sciences* 37:1076-1085.

Newcombe, C.P., and D.D. MacDonald. 1991. Effects of suspended sediments on aquatic ecosystems. *North American Journal of Fisheries Management* 11:72-82.

Noel, D.S., C.W. Martin, and C.A. Federer. 1986. Effects of forest clearcutting in New England on stream macroinvertebrates and periphyton. *Environmental Management* 10:661-670.

Oksanen, J., F.G. Blanchet, M. Friendly, R. Kindt, P. Legendre, D. McGlinn, P.R. Minchin, R.B. O'Hara, G.L. Simpson, P. Solymos, M.H.H. Stevens, E. Szoecs, and H. Wagner. 2020. Vegan: Community Ecology package. R package version 2.5-7. Available from https://CRAN.R-project.org/package=vegan.

Perrin, C.J., and J.S. Richardson. 1997. N and P limitation of benthos abundance in the Nechako River, British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences* 54:2574-2583.

Plotnikoff, R.W., and J.S. White. 1996. *Taxonomic Laboratory Protocol for Stream Macroinvertebrates Collected by the Washington State Department of Ecology*. Washington State Department of Ecology, Lacey, WA. 32 p.

Poff, N.L., E.I. Larson, P.E. Salerno, S.G. Morton, B.C. Kondratieff, A.S. Flecker, K.R. Zamudio, and W.C. Funk. 2018. Extreme streams: Species persistence and genomic change in montane insect populations across a flooding gradient. *Ecology Letters*. doi:10.1111/ele.12918.

Poff, N.L., M.I. Pyne, B.P. Bledsoe, C.C. Cuhaciyan, and D.M. Carlisle. 2010. Developing linkages between species traits and multiscaled environmental variation to explore vulnerability of stream benthic communities to climate change. *Journal of the North American Benthological Society* 29:1441-1458.

Price, K., A. Suski, J. McGarvie, B. Beasley, and J.S. Richardson. 2003. Communities of aquatic insects of old-growth and clearcut coastal headwater streams of varying flow persistence. *Canadian Journal of Forest Research* 33:1416-1432.

R Core Team. 2020. *R: A Language and Environment for Statistical Computing*. R Foundation for Statistical Computing, Vienna, Austria. https://www.R-project.org/.

Rabeni, C.F., K.E. Doisy, and L.D. Zweig. 2005. Stream invertebrate community functional responses to deposited sediment. *Aquatic Sciences* 67:395-402.

Relyea, C.D., G.W. Minshall, and R.J. Danehy. 2012. Development and validation of an aquatic fine sediment biotic index. *Environmental Management* 49:242-252.

Richardson, J.S., and S. Béraud. 2014. Effects of riparian forest harvest on streams: A meta-analysis. *Journal of Applied Ecology* 51:1712-1721.

Richardson, J.S., R.E. Bilby, and C.A. Bondar. 2005. Organic matter dynamics in small streams of the Pacific Northwest. *Journal of the American Water Resources Association* 41:921-934.

Richardson, J.S., and R.J. Danehy. 2007. A synthesis of the ecology of headwater streams and their riparian zones in temperate forests. *Forest Science* 53:131-147.

SAS Institute, Inc. 2013. *SAS/STAT User's Guide*. SAS Statistical Institute, Cary, NC.

Shaw, E.A., and J.S. Richardson. 2001. Direct and indirect effects of sediment pulse duration on stream invertebrate assemblages and rainbow trout (*Oncorhynchus mykiss*) growth and survival. *Canadian Journal of Fisheries and Aquatic Sciences* 58:2213-2221.

Stewart, B.A., P.G. Close, P.A. Cook, and P.M. Davies. 2013. Upper thermal tolerances of key taxonomic groups of stream invertebrates. *Hydrobiologia* 718:131-140.

Stone, M.K., and J.B. Wallace. 1998. Long-term recovery of a mountain stream from clear-cut logging: The effects of forest succession on benthic invertebrate community structure. *Freshwater Biology* 39:151-169.

Stubbington, R., M.T. Bogan, N. Bonada, A.J. Boulton, T. Datry, C. Leigh, and R.Vander Vorste. 2017. The biota of intermittent rivers and ephemeral streams: Aquatic invertebrates. Pages 217-243 *in* T. Datry, N. Bonada, and A.J. Boulton (eds.). *Intermittent Rivers and Ephemeral Streams: Ecology and Management*. Academic Press.

Suren, A.M., and I.G. Jowett. 2001. Effects of deposited sediment on invertebrate drift: An experimental study. *New Zealand Journal of Marine and Freshwater Research* 35:725-737.

SYSTAT Software, Inc. 2009. *SYSTAT Version 13. User's Manual*. Systat Software, Inc., Chicago, IL.

Vieira, N.K.M., N.L. Poff, D.M. Carlisle, S.R. Moulton, M.L. Koski, and B.C. Kondratieff. 2006. A database of lotic invertebrate traits for North America. US Geological Survey Data Series 187. http://pubs.water.usgs.gov/ds187.

Wallace, J.B., and M.E. Gurtz. 1986. Response of *Baetis* mayflies (Ephemeroptera) to catchment logging. *The American Midland Naturalist* 115:25-41.

Warren, D.R., W.S. Keeton, P.M. Kiffney, M.J. Kaylor, H.A. Bechtold, and J. Magee. 2016. Changing forests–changing streams: Riparian forest stand development and ecosystem function in temperature headwaters. *Ecosphere* 7(8):e01435. 10.1002/ecs2.1435.

Waters, T.F. 1995. *Sediment in Streams: Sources, Biological Effects, and Control*. American Fisheries Society Monograph 7.

Wiley, M.J., and S.L. Kohler. 1984. Behavioral adaptations of aquatic insects. Pages 101-133 *in* V. Resh and D. Rosenberg (eds.). *The Ecology of Aquatic Insects*. Praeger Publishers, New York, NY.

Wilkerson, E., J.M. Hagan, and A.A. Whitman. 2010. The effectiveness of different buffer widths for protecting water quality and macroinvertebrate and periphyton assemblages of headwater streams in Maine, USA. *Canadian Journal of Fisheries and Aquatic Sciences* 67:177-190.

Zweig, L.D., and C.F. Rabeni. 2001. Biomonitoring for deposited sediment using benthic invertebrates: A test on four Missouri streams. *Journal of the North American Benthological Society* 20:643-65.

7-7. APPENDIX TABLES

Appendix Table 7-1. Descriptive statistics for the macroinvertebrate metrics, functional feeding groups, and major taxonomic orders by treatment and treatment year. REF = reference; TRT = treatment; PRE = pre-harvest period; POST = post-harvest period; SE = standard error; EPT = Ephemeroptera-Plecoptera-Trichoptera; FSBI = fine sediment biotic index.

	Treatment	REF		TRT	
	Year	MEAN	SE	MEAN	SE
Metric					
Total Richness	PRE	56.89	1.84	57.63	1.26
	POST	45.11	1.61	48.67	1.34
EPT Richness	PRE	21.89	0.89	21.54	1.03
	POST	16.67	1.12	19.04	1.00
% EPT	PRE	64.62	3.20	61.14	3.20
	POST	56.18	7.29	57.53	2.78
Shannon H'	PRE	3.14	0.06	3.26	0.05
	POST	2.75	0.13	3.07	0.04
FSBI	PRE	55.56	6.43	58.96	7.64
	POST	49.44	7.19	53.54	6.35
Feeding Group					
% Filterer	PRE	0.36	0.13	0.48	0.17
	POST	0.07	0.05	0.03	0.03
% Gatherer	PRE	28.74	3.58	32.11	2.18
	POST	34.52	5.23	37.94	2.03
% Parasite	PRE	3.17	0.64	4.20	0.75
	POST	0.51	0.18	0.40	0.09
% Predator	PRE	27.62	2.11	19.45	1.28
	POST	24.95	3.84	22.81	2.10
% Scraper	PRE	13.56	1.64	13.38	1.80
	POST	6.68	1.54	7.70	1.23
% Shredder	PRE	26.36	3.87	30.12	1.79
	POST	32.65	4.29	30.86	2.09
% Unknown	PRE	0.20	0.05	0.26	0.06
	POST	0.62	0.29	0.28	0.06
Order					
% Coleoptera	PRE	0.58	0.22	0.75	0.13
	POST	0.70	0.26	0.46	0.09
% Diptera	PRE	25.80	3.06	26.85	2.69
	POST	41.37	6.97	38.87	2.84
% Tanypodinae	PRE	0.92	0.34	1.42	0.39
	POST	2.45	1.00	4.10	1.03
% Non-Tanypodinae	PRE	18.42	2.88	20.35	2.62
	POST	28.65	6.27	25.52	2.47
% Ephemeroptera	PRE	20.03	2.32	22.09	1.98
	POST	13.08	2.76	22.69	3.03
% Plecoptera	PRE	35.26	3.85	28.80	2.13
	POST	39.25	5.13	27.75	1.82
% Trichoptera	PRE	9.34	1.46	10.25	0.91
	POST	3.85	0.69	7.09	0.94

Appendix Table 7-2. Percent taxonomic composition of the benthic macroinvertebrate samples by treatment and treatment year. REF = reference; TRT = treatment; PRE = pre-harvest period; POST = post-harvest period. Asterisks indicate sediment sensitive taxa as scored by the fine sediment biotic index (Relyea *et al.* 2012). **** = extremely sensitive (0-20% fines); *** = very sensitive (20-30% fines); ** = moderately sensitive (30-40% fines); * = slightly sensitive (40-50% fines).

	REF		TRT	
Taxon	**PRE**	**POST**	**PRE**	**POST**
Annelida				
Oligochaeta	4.03	0.98	5.00	0.83
Arthropoda				
Chelicerata				
Arachnida				
Acarina	2.11	0.36	2.79	0.28
Crustacea				
Malacostraca				
Amphipoda	0.02	0.09	0.34	1.13
Decapoda		0.06	<0.01	0.01
Isopoda		0.01	0.02	<0.01
Ostracoda			0.01	
Hexapoda				
Coleoptera				
Dryopidae				0.01
Elmidae			0.01	
Bryelmis			0.15	0.02
Lara	0.29	0.19	0.21	0.39
Narpus concolor*	0.02	0.24		
Psephenidae	0.09		0.08	
Acneus		0.13	0.12	0.03
Ectopria			0.01	
Diptera				
Ceratopogonidae				
Ceratopogoninae	0.63	2.07	1.26	2.53
Forcipomyiinae		0.05	0.04	0.05
Chaoboridae				
Eucorethra underwoodi				0.01
Chironomidae				
Chironominae				
Chironomini			<0.01	
Microtendipes			0.03	
Polypedilum	0.17	1.31	1.22	0.99
Pseudochironomini				
Pseudochironomus			0.02	

Appendix Table 7-2 (continued).

Taxon	REF PRE	REF POST	TRT PRE	TRT POST
Tanytarsini				
Cladotanytarsus				0.02
Constempellina		0.56		0.04
Micropsectra	2.61	23.03	11.21	6.40
Rheotanytarsus				0.01
Stempellina	1.03		0.41	<0.01
Stempellinella	8.34	1.42	2.79	2.02
Tanytarsus	0.21	0.03	0.92	1.63
Diamesinae				
Pagastia			0.01	0.01
Potthastia gaedii		0.02		
Pseudodiamesa	0.01	0.01	0.03	0.02
Orthocladiinae	0.03	0.27	0.01	0.03
Brillia	2.83	4.93	2.65	5.34
Bryophaenocladius				0.01
Chaetocladius	0.14		0.16	0.05
Corynoneura	0.11	1.12	0.14	0.72
Cricotopus			0.41	0.22
Doithrix				0.01
Eretmoptera			0.02	
Eukiefferiella	0.19	0.44	0.92	1.04
Georthocladius				0.01
Gymnometriocnemus		0.15		0.02
Heleniella	1.33	1.44	0.68	0.51
Heterotanytarsus			0.02	0.36
Heterotrissocladius	0.02	0.09	0.69	0.28
Krenosmittia	0.02	0.23	0.10	0.21
Limnophyes	0.05	0.21	0.09	0.22
Metriocnemus	0.04		0.04	0.06
Nanocladius				0.06
Orthocladius		0.02	0.09	0.13
Orthocladius lignicola	0.05		0.10	0.13
Parachaetocladius	0.05	0.11	0.15	0.18
Parakiefferiella	0.10		0.55	0.60
Parametriocnemus	0.92	1.10	1.26	1.12
Paraphaenocladius	0.09	0.01	0.13	0.03
Parorthocladius		0.02	0.02	0.09
Pseudorthocladius				0.02
Pseudosmittia	0.03	0.01	0.01	
Psilometriocnemus triannulatus	0.10	0.73	0.11	0.44
Rheocricotopus	0.45	0.49	0.56	0.43
Stilocladius	0.18	0.08	0.39	0.13
Synorthocladius			0.03	0.03
Thienemanniella			0.11	0.37
Tvetenia	0.04	0.08	0.04	0.13
Tvetenia bavarica	0.82	1.56	0.56	1.26

Appendix Table 7-2 (continued).

	REF		TRT	
Taxon	**PRE**	**POST**	**PRE**	**POST**
Podonominae				
Boreochlus	0.10	0.14	0.02	0.58
Prodiamesinae				
Odontomesa			0.03	<0.01
Prodiamesa	0.03	0.09	0.06	0.06
Tanypodinae			0.11	0.05
Ablabesmyia			0.01	
Bilyjomyia algens	0.07		0.01	
Brundiniella eumorpha	0.16	0.88	0.15	0.54
Krenopelopia		0.04		0.06
Larsia	0.07	1.03	0.08	2.18
Macropelopia				0.07
Natarsia		0.02		0.06
Pentaneura			0.01	
Radotanypus		0.01		
Thienemannimyia	0.07	0.04	0.50	0.57
Zavrelimyia	0.51	0.89	1.13	0.76
Dixidae	0.02		0.01	0.03
Dixa	0.34	0.70	0.31	1.17
Meringodixa	0.22	0.43	0.16	0.72
Dolichopodidae		0.01		0.02
Empididae	0.04			0.04
Chelifera			0.01	
Clinocera			0.04	0.02
Neoplasta	0.05		0.05	0.06
Oreogeton**	0.11		0.05	0.02
Wiedemannia			<0.01	
Psychodidae			<0.01	<0.01
Pericoma/Telmatoscopus			<0.01	<0.01
Psychoda				0.04
Ptychopteridae				
Bittacomorpha			0.01	
Ptychoptera	0.43	2.18	0.09	0.54
Simuliidae				
Prosimulium*	2.21	0.19	0.52	0.09
Simulium	0.15	0.22	0.62	0.77
Stegopterna		0.02	0.03	0.01
Thaumaleidae			0.03	0.02

Appendix Table 7-2 (continued).

Taxon	REF PRE	REF POST	TRT PRE	TRT POST
Tipulidae	0.06		0.03	0.01
Austrolimnophila				0.01
Dicranota	0.80	1.66	0.58	0.91
Erioptera			0.01	
Gonomyia			<0.01	<0.01
Hexatoma	0.15	0.27	0.02	0.06
Holorusia hespera				0.01
Limnophila	0.38	0.28	0.13	0.22
Limonia			0.05	
Lipsothrix			0.02	0.02
Molophilus	0.10	0.01	0.01	0.03
Ormosia	0.03		0.02	
Pedicia	0.45	0.10	0.20	0.01
Rhabdomastix fascigera*	0.02	0.01	0.02	0.02
Tipula	0.08	0.23	0.05	0.10
Ulomorpha	0.21	0.05	0.16	0.13
Ephemeroptera				
Ameletidae				
Ameletus	1.24	1.67	0.52	1.49
Baetidae				
Acentrella*				0.01
Baetis			0.53	
Baetis bicaudatus*	2.42	0.19	1.99	0.21
Baetis piscatoris				0.49
Baetis tricaudatus		0.02	0.52	3.39
Diphetor hageni	1.12	1.11	1.41	6.43
Labiobaetis sonajuventus				0.01
Ephemerellidae				
Drunella*	0.05		0.02	
Drunella coloradensis**	0.04	0.11	0.48	0.12
Drunella doddsii***		0.02	<0.01	0.01
Heptageniidae				
Cinygma	9.72	3.09	7.27	2.85
Cinygmula*	0.22	0.36	1.05	1.85
Epeorus**		0.02	0.11	0.37
Ironodes*	1.51	0.23	1.18	0.78
Rhithrogena***			0.01	0.02
Leptophlebiidae		3.25		5.86
Neoleptophlebia		1.15		0.94
Paraleptophlebia	3.87		4.29	0.56
Lepidoptera				
Limacodidae				
Isochaetides	0.17			

Appendix Table 7-2 (continued).

	REF		TRT	
Taxon	**PRE**	**POST**	**PRE**	**POST**
Megaloptera				
Sialidae				
Sialis			0.04	0.05
Plecoptera				
Capniidae	0.07	0.19	0.26	0.39
Chloroperlidae				
Haploperla		0.05		
Kathroperla			0.02	0.10
Paraperla**	0.03		0.01	
Plumiperla			0.13	
Sweltsa	15.20	8.58	6.17	4.98
Leuctridae	3.83	1.39	3.42	0.89
Despaxia augusta****		3.18	0.63	2.87
Moselia infuscata	5.14	1.80	3.85	1.70
Nemouridae			0.03	
Malenka	0.70	0.55	1.32	1.19
Ostrocerca	0.54	0.14	0.21	0.05
Soyedina	0.76	2.07	1.23	1.48
Zapada cinctipes	0.66	2.93	1.38	6.37
Zapada columbiana	5.53	10.44	5.80	4.20
Zapada frigida*	0.12	0.16	0.16	0.39
Zapada oregonensis*		0.01		
Peltoperlidae				
Soliperla			0.03	0.02
Yoraperla*	0.63	0.25	1.22	0.73
Perlidae	0.04	0.09	0.07	0.04
Calineuria californica**			0.08	0.34
Doroneuria***	0.07	0.03	0.30	0.64
Perlodidae			0.13	0.03
Isoperla	0.60	0.09	0.13	0.18
Skwala		0.14		0.05
Pteronarcyidae			0.02	0.02
Pteronarcys**			0.02	
Pteronarcys princeps				0.04
Trichoptera				
Brachycentridae			0.01	
Micrasema	0.51	0.62	1.92	0.55
Glossosomatidae	0.01			0.01
Anagapetus*			0.08	0.01
Glossosoma**	0.16	0.07	0.14	0.48
Hydropsychidae	0.31		0.02	0.01
Arctopsychinae**	0.29		0.41	
Parapsyche*	0.22	0.11	0.22	1.05

Appendix Table 7-2 (continued).

	REF		TRT	
Taxon	PRE	POST	PRE	POST
Lepidostomatidae				
Lepidostoma	1.75	0.24	1.87	0.77
Limnephilidae	0.31	0.06	0.35	0.64
Chyrandra centralis	0.11		0.15	
Cryptochia	0.04		0.02	
Hydatophylax hesperus			0.03	
Psychoglypha	1.52	0.20	0.12	0.24
Philopotamidae	0.02		0.02	0.02
Dolophilodes**			0.05	0.03
Wormaldia	1.12	0.99	2.29	2.38
Polycentropodidae				
Polycentropus	0.03		0.03	0.10
Rhyacophilidae				
Rhyacophila		0.14	0.10	0.11
Rhyacophila angelita group***			0.04	
Rhyacophila betteni group**	0.05		0.05	0.02
Rhyacophila ecosa group	0.12			
Rhyacophila grandis	1.06	0.33	0.61	0.22
Rhyacophila narvae**			0.05	0.02
Rhyacophila rotunda group	0.03			
Rhyacophila viquaea group				<0.01
Rhyacophila vofixa group***	1.49	0.58	1.11	0.22
Uenoidae				
Neophylax splendens**	0.42	0.03	0.20	0.01
Cnidaria				
Hydrozoa	0.02			
Mollusca				
Bivalvia	0.32	0.03	0.50	0.06
Gastropoda			0.15	0.04
Nematoda	0.72	0.07	1.05	0.17
Platyhelminthes	1.11		0.27	0.01

APPENDIX A – SITE SELECTION PROCESS

Welles Bretherton

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

A-1. IDENTIFICATION OF SITES MEETING STUDY DESIGN CRITERIA

Site selection began in March 2011 and continued through July 2012, when the last reference site was selected. Identification of study sites began with an ArcMap (ESRI 2010) analysis using the Geographic Information System (GIS) criteria (**Table A-1**). This process selected 9,407 potential sites from the 98,138 Type Np basins located in western Washington (**Figure A-1**).

Table A-1. Original site selection criteria for the Soft Rock Study.

Criterion	Limit	Information Source
Geographic location	West of the crest of the Cascade Mountains in Washington State	GIS
Elevation	<1,067 m (3500 ft) for the Olympic Peninsula <1,219 m (4000 ft) for the Cascades	
Gradient	5–50% (3–27 degrees)	
Lithology	>80% of the basin with lithology classes of marine sedimentary or glacial till	
Basin area	~12–49 ha (~30–120 ac)	
Stand age	>70% of basin with tree ages between 30 and 80 years during harvest-treatment window	Landowner
Ownership	>80% owned by a single participating landowner	
Harvest timing	Treatment basins: harvest October 2013 to May 2015 Reference basins: no harvest before October 2016	
Landowner commitment	5 years	

Landowners were then contacted for participation in the study. However, the next two steps in the decision making process (**Figure A-1**) were not always linear. Landowners have different means of tracking harvest units. Some received the entire 9,407 basin shapefile, while others preferred a pre-selected version. For some landowners, we used the National Agriculture Imagery Program (NAIP) orthophotos to reject basins that had obvious recent harvest. Tax parcel data were also used to provide landowner-specific shapefiles as well as to find additional owners to participate in the study. Coordination with the landowners and NAIP analysis further reduced the pool to 150 potential study sites.

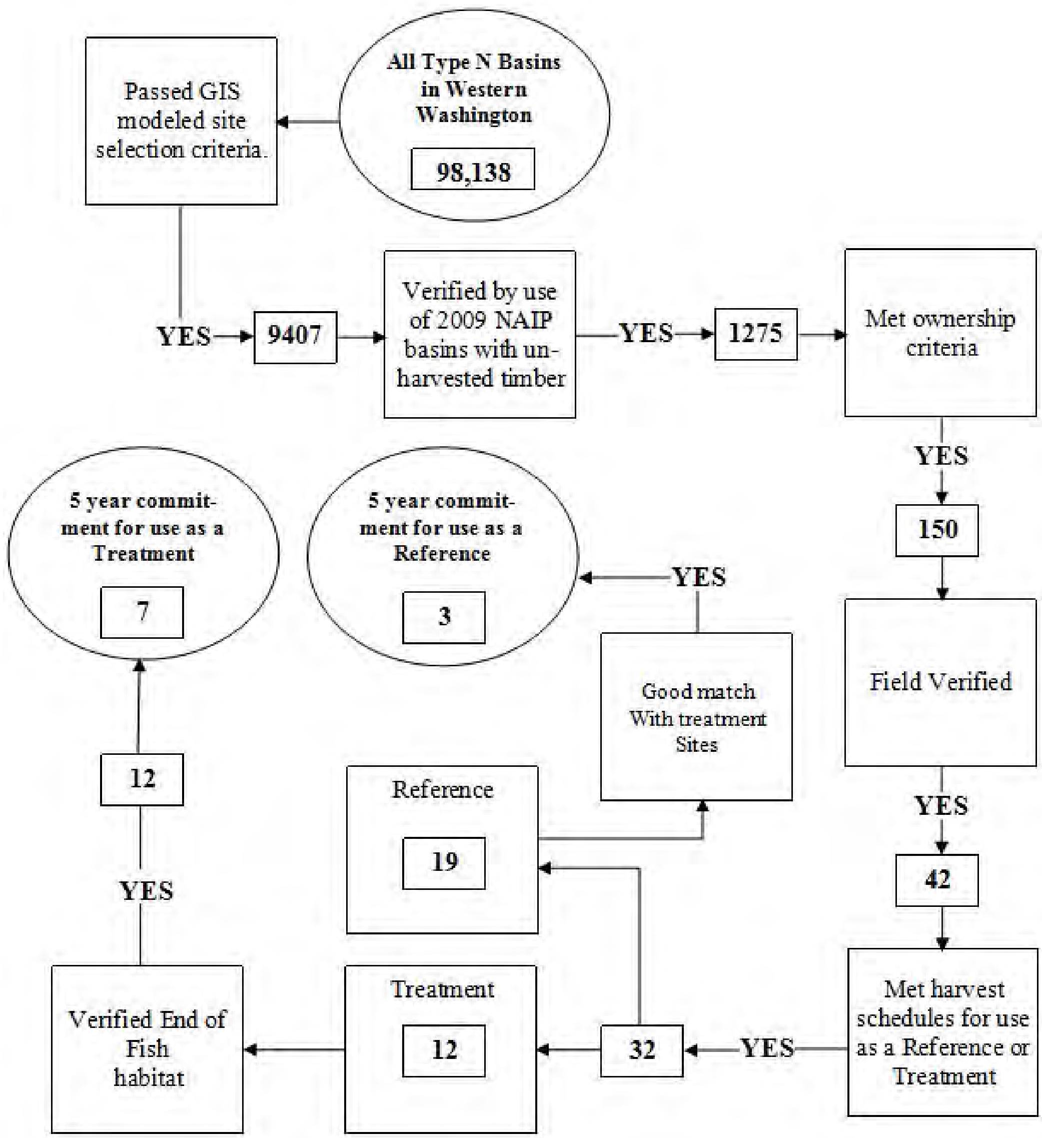


Figure A-1. Flow chart showing the decision process for selecting treatment and reference sites for the Soft Rock Study.

The large reduction in potential study sites stems from three factors, as illustrated by the NAIP and owner-verified maps in **Figure A-2**:

- Federal lands were not available to harvest on our schedule.
- The majority of the Puget Trough is dominated by residential and small private ownership.
- Large tree farms harvest smaller units on a quicker rotation, limiting the number of fully unharvested Type N stream basins.

A-2. SELECTION BY LITHOLOGY AND REGION

Initially, the goal of the study was to have two blocks of study sites representing glacial till and marine sedimentary lithologies. On-site verification focused on finding a sufficient number of sites to constitute each block. Since sites within each block needed to be in close proximity, we divided the sites into three distinct regions in western Washington: Puget Sound, Willapa Hills, and Olympic Peninsula. We divided each lithology (adding freshwater sedimentary lithologies later) into regions to identify sites that were morphologically and spatially similar (**Table A-2**).

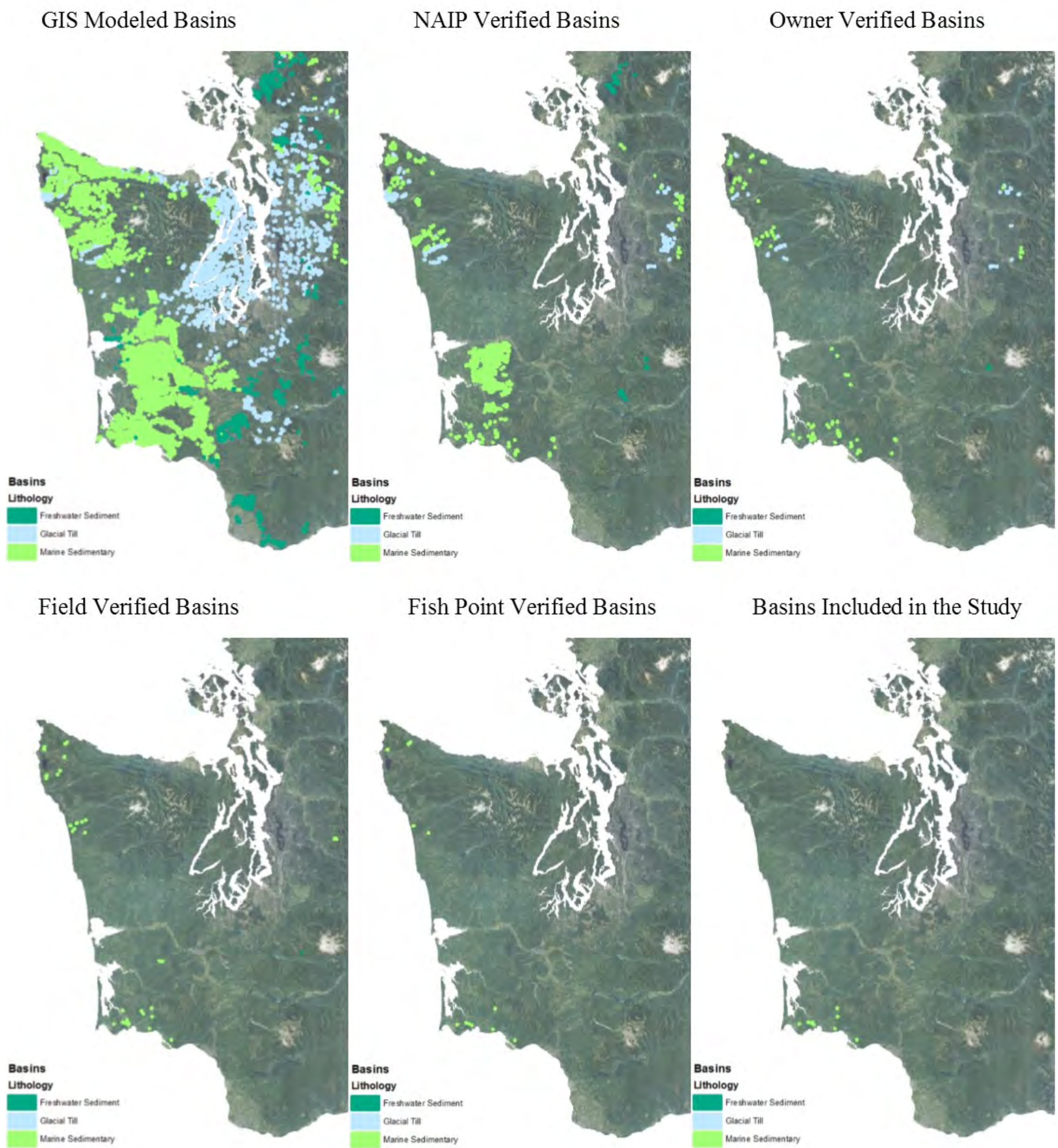


Figure A-2. Maps of potential study sites at key points in the site selection process.

Table A-2. Number of potential sites identified at each step of the site selection process by lithology and region.

Lithology	Marine Sediment			Glacial Till		Freshwater Sediment	Total
Region	Willapa Hills	Olympic Peninsula	Puget Sound	Olympic Peninsula	Puget Sound	Puget Sound	
All Type Np Basins	98,138						98,138
GIS-Modeled Criteria	6,927			1,744		736	9,407
NAIP Verified	876	154	19	128	61	37	1,275
Ownership Criteria	52	49	6	23	16	4	150
Field Verified	19	20	2	0	0	1	42
Fish Point Verified	7	5	0	0	0	0	12
Treatment Verified	7	2	0	0	0	0	9
Reference Verified	3	1	0	0	0	1	5
5-Year Commitment	10	0	0	0	0	0	10

GLACIAL TILL

Glacial till was located only in the Puget Sound and Olympic Peninsula regions. Using the 2011 NAIP, we identified 16 potential glacial till sites on state land in the Puget Sound region. On-site verification, however, removed the sites from consideration. Three sites were located behind miles of decommissioned roads that were not going to be reopened in the foreseeable future. Five sites were identified on public land as potential references, but they could not be used as treatments. Five sites were identified as potential treatment sites, but had wetlands and beaver ponds distributed through the stream networks and thus were rejected. Private landowners offered three sites as potential reference or treatment sites, but one had a very large wetland, one had houses in the basin, and one had already been partially harvested.

We conducted initial site visits of potential glacial till sites on the Olympic Peninsula in the summer of 2011. Many of the streams had low or discontinuous flows or none at all. We conducted a second site visit with the landowner in August 2011 to assess low flow conditions at the sites with the most potential. Of the six sites assessed, two had marine sediment outcrops in the channel and four were either mostly dry or had wetlands near the Type F/N break.

Because we were unable to find suitable glacial till study sites during the on-site verification process, we brought our findings before the Soft Rock Science Advisory Group (SRSAG) to recommend dropping the glacial till lithology from the project. The group agreed to this proposal and instead recommended finding sites dominated by marine sedimentary or freshwater sedimentary lithologies in the Puget Sound region.

FRESHWATER SEDIMENTARY

We identified potential freshwater sedimentary sites in the Puget Sound region, but on-site verification removed the sites from consideration. In the north Puget Sound, most of the 26 possible sites were in the Lake Whatcom watershed, which could not be harvested within our timeframe, so were only suitable as references. We found six possible treatment sites on private land and one on state land, but all were too young to be harvested within our time frame. In the south Puget Sound, seven potential sites had all been recently harvested and four were available only as references. Because no sites were available for harvest within our harvest window, the SRSAG and principal investigators decided to focus on marine sedimentary lithologies.

MARINE SEDIMENTARY

Marine sedimentary lithologies were found in the Puget Sound, Olympic Peninsula, and Willapa Hills regions. In the Puget Sound region, we identified 13 potential sites on public utilities district, state park, and federal lands, but the sites were available only as references. Of the two sites on state land, one consisted of a small seasonal stream and the other was not available for harvest within our time frame. Private landowners provided six potential sites, but two were partially harvested, one was not available for harvest within our time frame, and one had no flow. The remaining two sites were acceptable treatment options, but both had multiple mainline road crossings with evidence of mass wasting and were not suitable for flume installation because of high water volumes and velocity. The lack of sites suitable for harvest and flume installation forced the SRSAG and principal investigators to drop the Puget Sound region altogether and instead focus on the Olympic Peninsula and Willapa Hill regions where there were more options.

On the Olympic Peninsula, we selected 154 sites on state and private land as potential references or treatments. The Washington Department of Natural Resources (DNR) offered 33 reference sites and private landowners offered 11 reference sites. We visited 51 potential treatment sites from summer 2011 to spring 2012, and coordinated with landowners to determine commitment to harvest within our time frame. Two sites on private land, one north of Lake Ozette and the other east of Kalaloch, were available for harvest within our time frame. The Ozette site had a strong coastal influence (extensive/persistent fog bank) whereas the site east of Kalaloch had a more inland climate, resulting in very different temperature and precipitation regimes. We decided that replicates of each site would be required to use these as treatments in the study.

In early 2012, we planned on using three sites on state land, two near Lake Ozette and one near Kalaloch, as replicates of the two sites on private land. Some portion of each of these sites was scheduled for harvest within our time frame, although two were scheduled as a thinning. At the Washington State Department of Ecology's (Ecology) request, the DNR conducted a water-typing survey to determine the type break between the F and Np waters. The surveys resulted in moving the type break 600 to 1200 ft upstream of the modeled type break, which reduced the basin size to below the 30-acre harvest minimum. Ecology staff requested to use the modeled type break instead of the surveyed break and to have the buffer treatment applied to the entire study site. Ultimately, the DNR was unable to meet those requests. The SRSAG and principal investigators decided that the two remaining treatment sites on the Olympic Peninsula were too dissimilar to be paired, resulting in case studies at those sites. Ultimately, the sites were dropped due to the remote location and high cost-to-benefit ratio. The SRSAG decided to focus the study in the Willapa Hills region.

A-3. TREATMENT SITE SELECTION

In the Willapa Hills region, we identified 52 study sites on private land as potential treatment sites, and conducted on-site verification and landowner coordination during summer and fall 2011. We rejected nine of the sites because of massive inner gorges, wetlands, waterfalls, or no flow. An additional 24 sites were not available for harvest during our time frame and were set aside as potential references. From the remaining potential sites, we received preliminary commitments from landowners to harvest seven during our harvest window.

STREAM TYPING SURVEYS

We conducted stream typing surveys in spring 2012 to determine the physical limitation to fish passage (i.e., Type F/N break). This location was necessary to determine basin area of the non-fish-bearing portion of the stream. We found that two of the surveyed breaks remained in relatively the same location as the modeled break, while one moved downstream and four moved upstream. For three of the sites where the type break moved upstream, the basin area was reduced to less than 30 acres (too small for the study). To achieve replication, we requested and were granted an exemption (pilot rule) from the DNR to use the modeled type break for these three sites.

BUFFER LAYOUTS

It was originally conceived that all treatment sites would be buffered with a two-sided 50-foot buffer along 50% of the Np stream, as prescribed in the Forest Practices rules. However, the presence of unstable slopes (e.g., inner gorges, bedrock hollows) would greatly increase the percentage of stream buffered. We conducted buffer layouts in conjunction with foresters employed by the landowners. While three of the sites qualified for the 50% buffer, the other four would require a 90 to 100% buffer due to unstable slopes. The SRSAG and principal investigators decided that the unstable slope buffers were consistent with the design of harvest prescriptions in western Washington and that all seven sites should be included in the study. This resulted in a gradation of buffer lengths from 50% to 100%, with some buffers wider and some narrower than required under westside Forest Practices rules for Type N streams (**Figure A-3**).

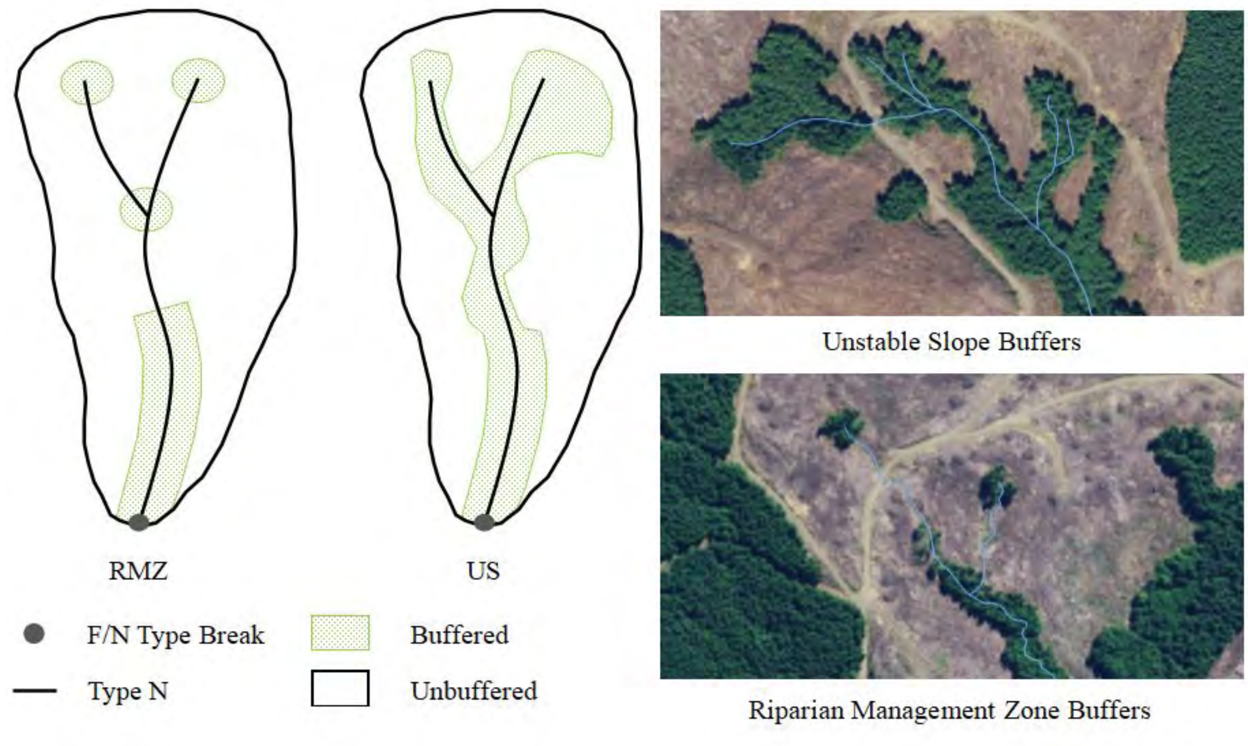


Figure A-3. Conceptual examples of Type Np buffers with only riparian management zone (RMZ) buffers and with additional unstable slope (US) buffers (left) and examples from two Soft Rock Study treatment sites (right).

A-4. REFERENCE SELECTION

We identified 37 sites on private and state land as potential references and conducted on-site verification and landowner coordination throughout the treatment site selection process. We selected two references near the four treatment sites in the Naselle area based on similarities in morphology as well as landowner commitment to withhold harvest in the reference sites until after October 2016. For sites in the Cathlamet area, we selected only one reference because most landowners were not able to commit to withholding harvest until after October 2016. Because having un-harvested control sites is critical to the Multiple Before-After/Control-Impact study design, this was the most important factor in selecting reference sites.

A-5. FLUME SITE SELECTION

We monitored four of the Soft Rock sites (two treatments and two references) for hydrologic conditions. Site selection was determined by geomorphic conditions at the F/N break. These sites required confined valley walls to funnel stream flow through the flume for the duration of the study. We selected the four sites that were least likely to have the flumes bypassed by channel migration. We were unable to monitor additional sites due to funding limitations.

A-6. REFERENCES

ESRI. 2010. ArcMap 10.0. Environmental Systems Resource Institute, Redlands, CA.

APPENDIX B – SITE LAYOUT

Welles Bretherton

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

B-1. CHANNEL MAPPING

Pre-harvest preparation and monitoring began in May 2012. We used a hip chain to map the entire stream network and establish transects beginning at the junction of fish-bearing (F) and non-fish-bearing (N) waters. We flagged and recorded a GPS point at every 10 m interval throughout the Np channel network, including tributaries. Transects were labeled sequentially working upstream from the F/N junction to the perennial initiation point of the main channel. Tributaries were labeled right bank (RB) or left bank (LB) based on which bank the tributary entered the main channel facing downstream. Tributaries along each stream bank were then numbered sequentially from the F/N junction. Finally, tributaries were labeled based on their distance from the confluence with the main channel. If there was a tributary off of a tributary, the tributary label was followed by a letter series based on how many tributaries entered that tributary (**Figure B-1**). This grid system was used to determine the location of all measurements taken during this study.

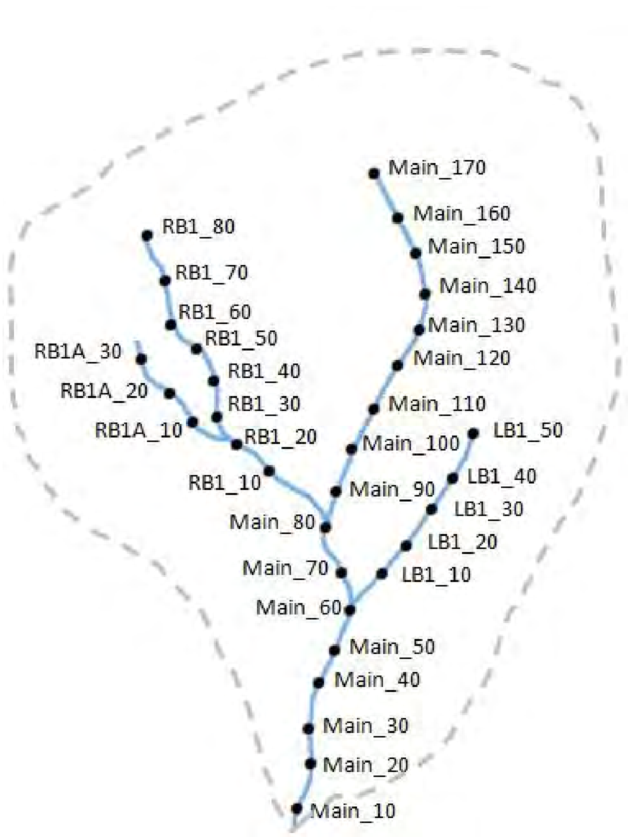


Figure B-1. Example of numbering scheme used for labeling transects in the Soft Rock Study sites.

B-2. SITE DESCRIPTIONS

Site characteristics based on the original site selection parameters are listed in **Table B-1**, along with other relevent measurements. These data were compiled from on-the-ground measurements, GIS-modeled criteria, and information from landowners.

Table B-1. Site characteristics for the Soft Rock Study sites including basin area, length of entire stream network, mean percent of stream with surface flow during summer low flow (2013-2017), elevation (from 1-m Digital Terrain Model [DTM] at F/N break), gradient (mean stream network slope derived from 1-m DTM), general basin aspect, percent of basin composed of marine sediment (MS), number of owners of the harvest area, approximate stand age (±2 or 3 years), year harvest occurred (or planned future harvest of reference sites), percent buffer length, and mean buffer width (calculated in ArcMap). TRT1a and TRT1b are sub-basins of TRT1 (see text for details).

Study Site	Basin Area (ha)	Stream Length (m)	Mean Wetted Channel (%)	Elevation (m)	Gradient (%)	Aspect	Lithology	Ownership	Stand Age	Harvest Timing	Buffer Length (%)	Mean Buffer Width (m)
REF1	16	1456	84	114	21	SW	100% MS	1 Owner	30	2020		
REF2	15	856	75	58	18	SW	100% MS	1 Owner	35	2020		
REF3	12	697	74	46	19	W	100% MS	1 Owner	35	2020		
TRT1	30	1827	98	73	20	N	87% MS	1 Owner	38	2014	53	17
TRT1a	13	797	99	73	22	N	87% MS	1 Owner	38	2014	40	15
TRT1b	17	930	97	74	18	N	87% MS	1 Owner	38	2014	63	20
TRT2	10	591	64	31	13	NW	100% MS	1 Owner	40	2014	54	15
TRT3	13	958	80	36	15	NW	100% MS	1 Owner	39	2013	58	14
TRT4	15	864	75	34	18	NW	100% MS	1 Owner	50	2014	92	17
TRT5	14	1049	69	63	19	SW	100% MS	1 Owner	37	2015	95	15
TRT6	12	992	84	46	22	NE	100% MS	1 Owner	37	2014	96	14
TRT7	24	940	89	289	30	S	99% MS	1 Owner	37	2015	100	23

This appendix includes maps of all ten Soft Rock Study sites (seven treatments, three references). These maps use a common legend detailed in **Figure B-2**. The stream maps show the location of the hydrology stations, temperature sensors, plots, and transects, and are variable in scale to show a close up of the stream network. The map insert has a fixed scale (1:11,000) to illustrate differences in basin area among the sites. The minimum required buffer area and the actual buffer area are also displayed in the map inserts. A table with descriptive statistics is included with each site map with details in **Table B-2**.

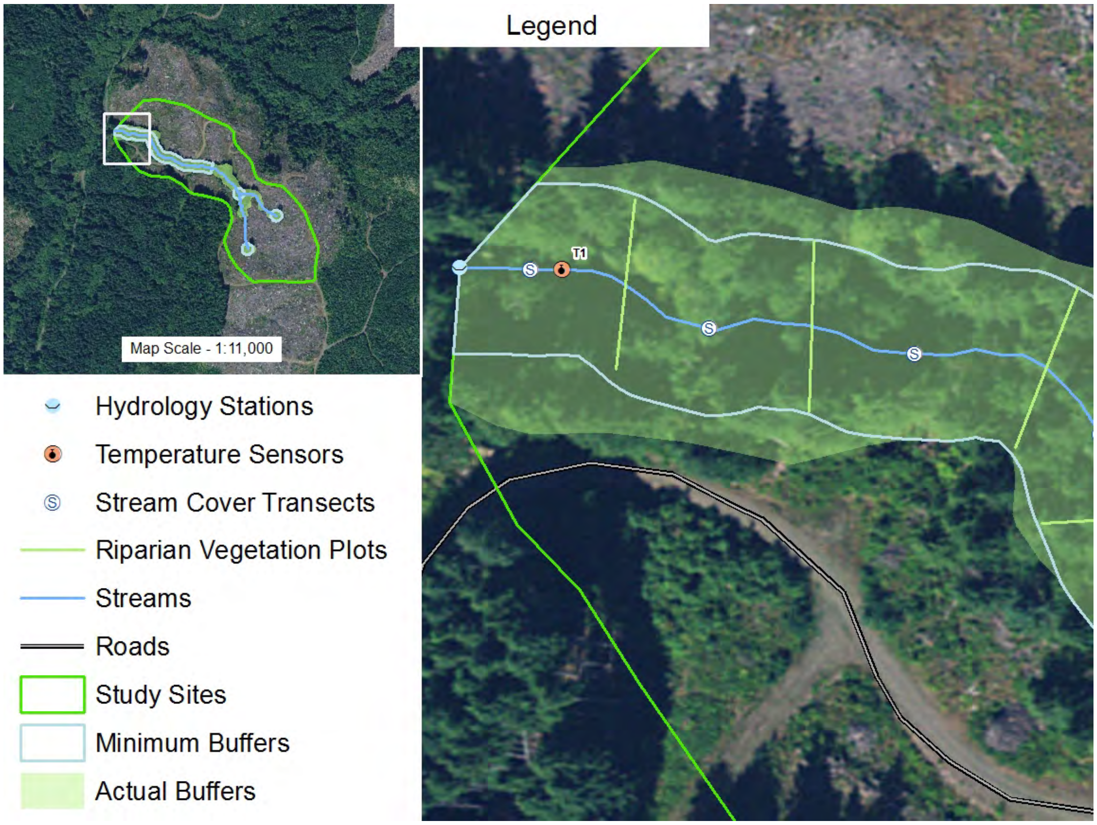
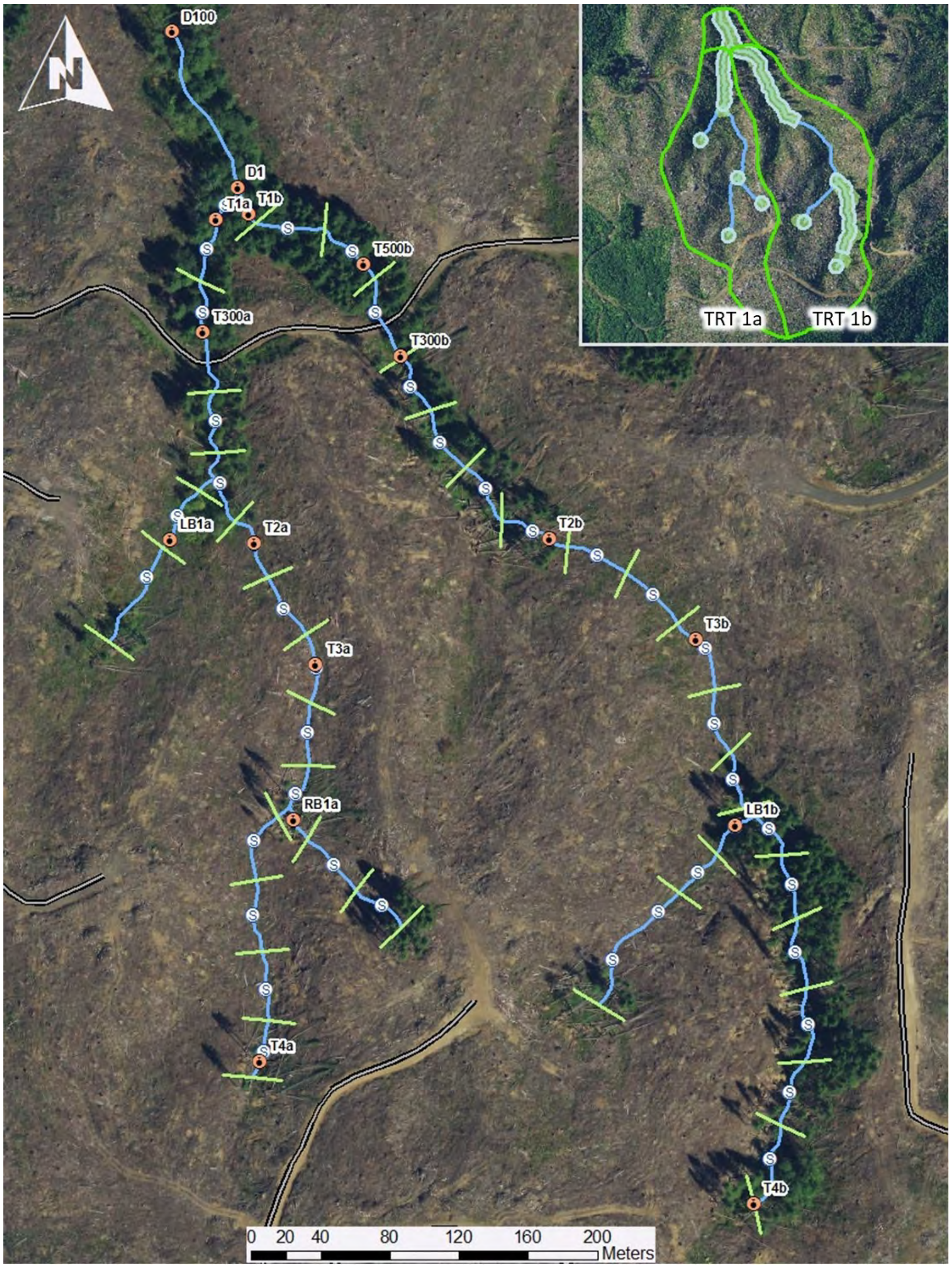


Figure B-2. Legend for the Soft Rock Study site maps with an example site (TRT4).

Table B-2. Description of tables included with each Soft Rock Study site map.

<table>
<tr><td>Basin Size</td><td>Total basin area</td><td>Stream Length</td><td>Total stream length</td></tr>
<tr><td rowspan="3">Bankfull Width
Wetted Width
Stream Slope
Valley Wall Slope</td><td rowspan="3">Mean across the study site</td><td colspan="2">Buffers</td></tr>
<tr><td>Length Buffered</td><td>Percent length buffered</td></tr>
<tr><td>Minimum Area
Actual Area
Increase in Area</td><td>Buffer areas calculated in ArcMap (ESRI 2010)</td></tr>
<tr><td>Wetted Channel</td><td>Mean percent wetted channel in August over the study period (2013-2017)</td><td rowspan="2">Mean Buffer Width</td><td rowspan="2">Mean buffer width calculated from the riparian vegetation plots</td></tr>
<tr><td>Average Precip</td><td>PRISM 30-yr average precipitation values</td></tr>
</table>

TRT 1



Basin Size (ha [ac])	32 (79)	Stream Length (m [ft])	1827 (5994)
Bankfull Width (cm)	152	**Buffers**	
Wetted Width (cm)	76	Length Buffered (%)	53
Stream Slope (%)	20	Minimum Area (ha [ac])	3.09 (7.64)
Valley Wall Slope (%)	44	Actual Area (ha [ac])	3.68 (9.10)
Wetted Channel (%)	98	Increase in Area (%)	19
Average Precip (cm [in])	201 (79)	Mean Buffer Width (m [ft])	18 (58)

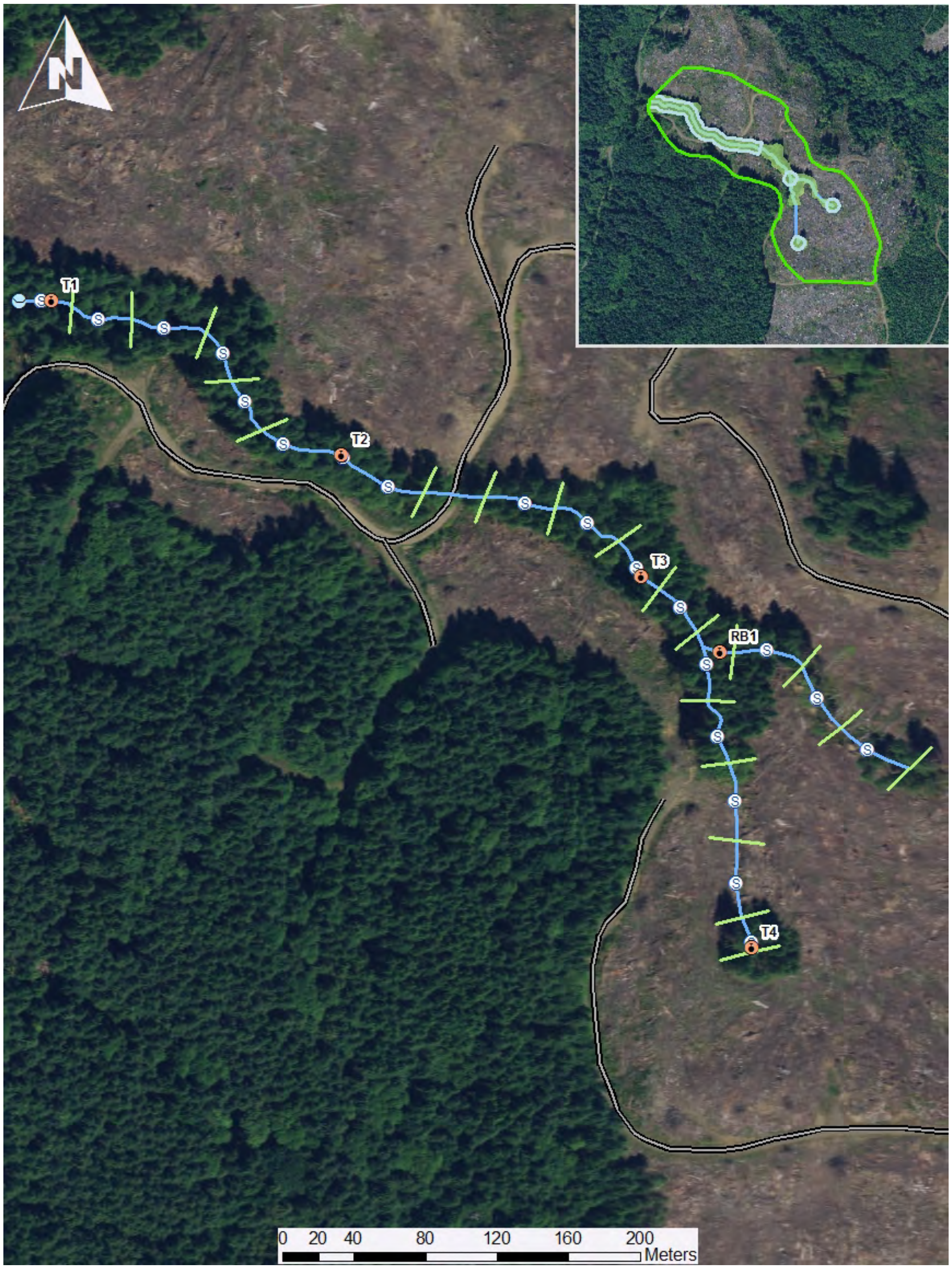
TRT2

Basin Size (ha [ac])	10 (25)	Stream Length (m [ft])	591 (1939)
Bankfull Width (cm)	114	**Buffers**	
Wetted Width (cm)	52	Length Buffered (%)	54
Stream Slope (%)	12	Minimum Area (ha [ac])	1.00 (2.47)
Valley Wall Slope (%)	58	Actual Area (ha [ac])	1.19 (2.93)
Wetted Channel (%)	65	Increase in Area (%)	18
Average Precip (cm [in])	262 (103)	Mean Buffer Width (m [ft])	15 (48)

TRT3

Basin Size (ha [ac])	13 (31)	Stream Length (m [ft])	958 (3143)
Bankfull Width (cm)	102	**Buffers**	
Wetted Width (cm)	54	Length Buffered (%)	58
Stream Slope (%)	20	Minimum Area (ha [ac])	1.53 (3.79)
Valley Wall Slope (%)	58	Actual Area (ha [ac])	1.89 (4.68)
Wetted Channel (%)	79	Increase in Area (%)	23
Average Precip (cm [in])	259 (102)	Mean Buffer Width (m [ft])	20 (42)

TRT4



Basin Size (ha [ac])	15 (38)	Stream Length (m [ft])	864 (2835)
Bankfull Width (cm)	126	**Buffers**	
Wetted Width (cm)	49	Length Buffered (%)	92
Stream Slope (%)	16	Minimum Area (ha [ac])	1.33 (3.28)
Valley Wall Slope (%)	60	Actual Area (ha [ac])	2.39 (5.92)
Wetted Channel (%)	76	Increase in Area (%)	80
Average Precip (cm [in])	246 (97)	Mean Buffer Width (m [ft])	15 (48)

TRT5

Basin Size (ha [ac])	14 (35)	Stream Length (m [ft])	1049 (3442)
Bankfull Width (cm)	79	**Buffers**	
Wetted Width (cm)	32	Length Buffered (%)	95
Stream Slope (%)	18	Minimum Area (ha [ac])	1.86 (4.59)
Valley Wall Slope (%)	64	Actual Area (ha [ac])	3.07 (7.59)
Wetted Channel (%)	70	Increase in Area (%)	65
Average Precip (cm [in])	272 (107)	Mean Buffer Width (m [ft])	15 (48)

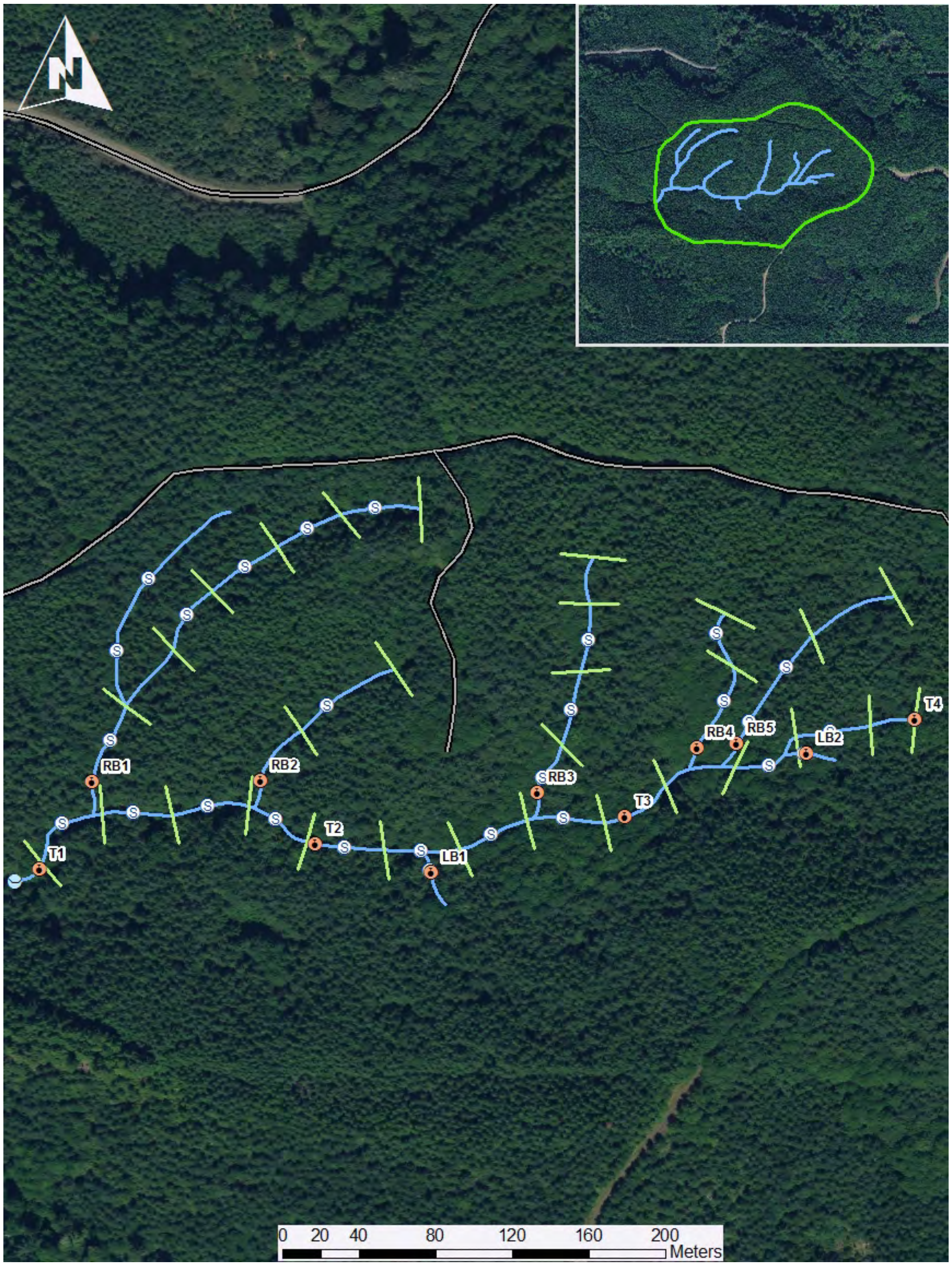
TRT6

Basin Size (ha [ac])	12 (29)	Stream Length (m [ft])	992 (3255)
Bankfull Width (cm)	119	**Buffers**	
Wetted Width (cm)	47	Length Buffered (%)	96
Stream Slope (%)	21	Minimum Area (ha [ac])	1.60 (3.96)
Valley Wall Slope (%)	82	Actual Area (ha [ac])	3.10 (7.67)
Wetted Channel (%)	84	Increase in Area (%)	94
Average Precip (cm [in])	231 (91)	Mean Buffer Width (m [ft])	17 (56)

TRT7

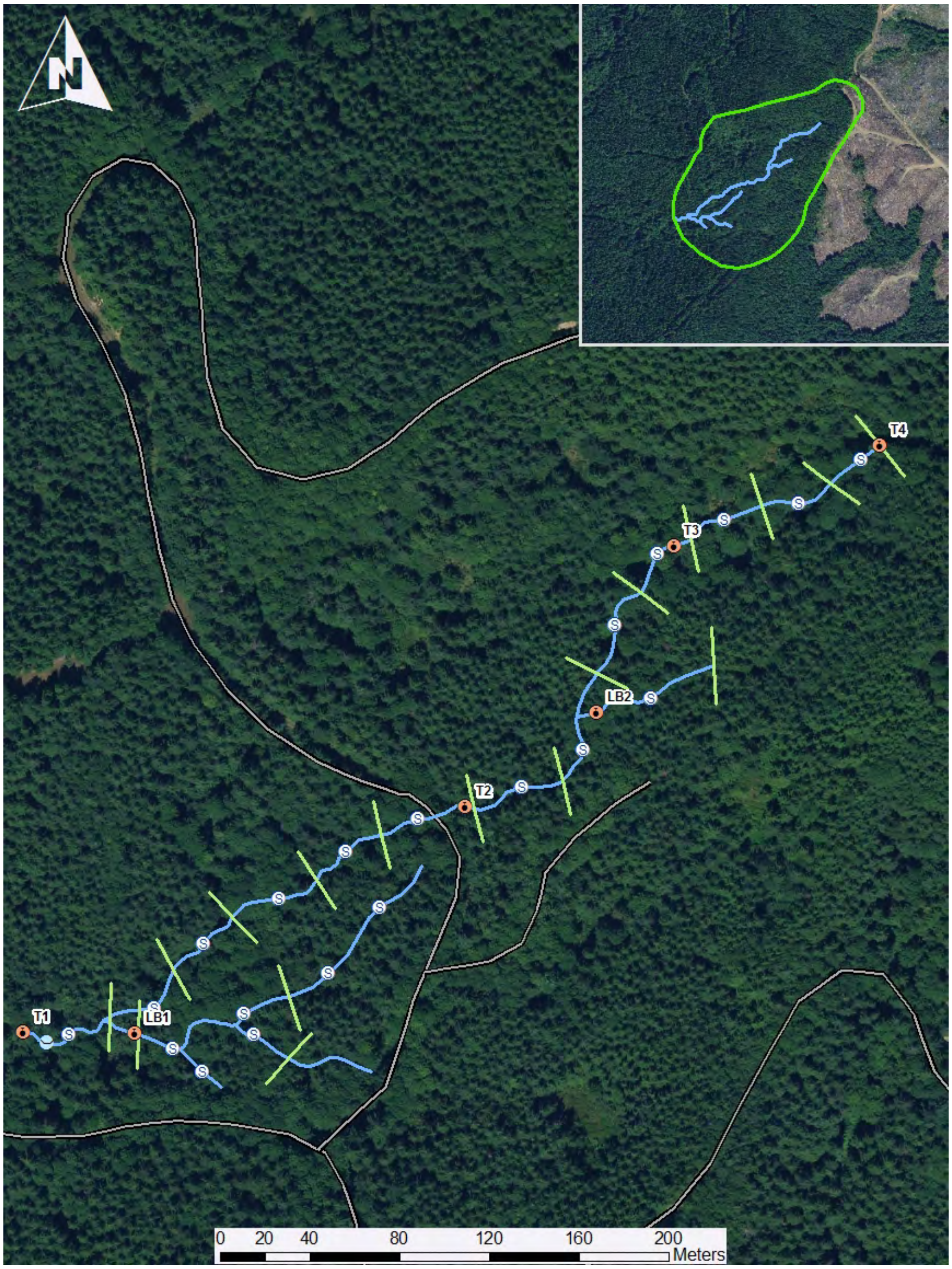
Basin Size (ha [ac])	24 (59)	Stream Length (m [ft])	940 (3084)
Bankfull Width (cm)	192	**Buffers**	
Wetted Width (cm)	68	Length Buffered (%)	100
Stream Slope (%)	30	Minimum Area (ha [ac])	1.59 (3.93)
Valley Wall Slope (%)	71	Actual Area (ha [ac])	4.19 (10.34)
Wetted Channel (%)	86	Increase in Area (%)	163
Average Precip (cm [in])	307 (121)	Mean Buffer Width (m [ft])	24 (79)

REF1



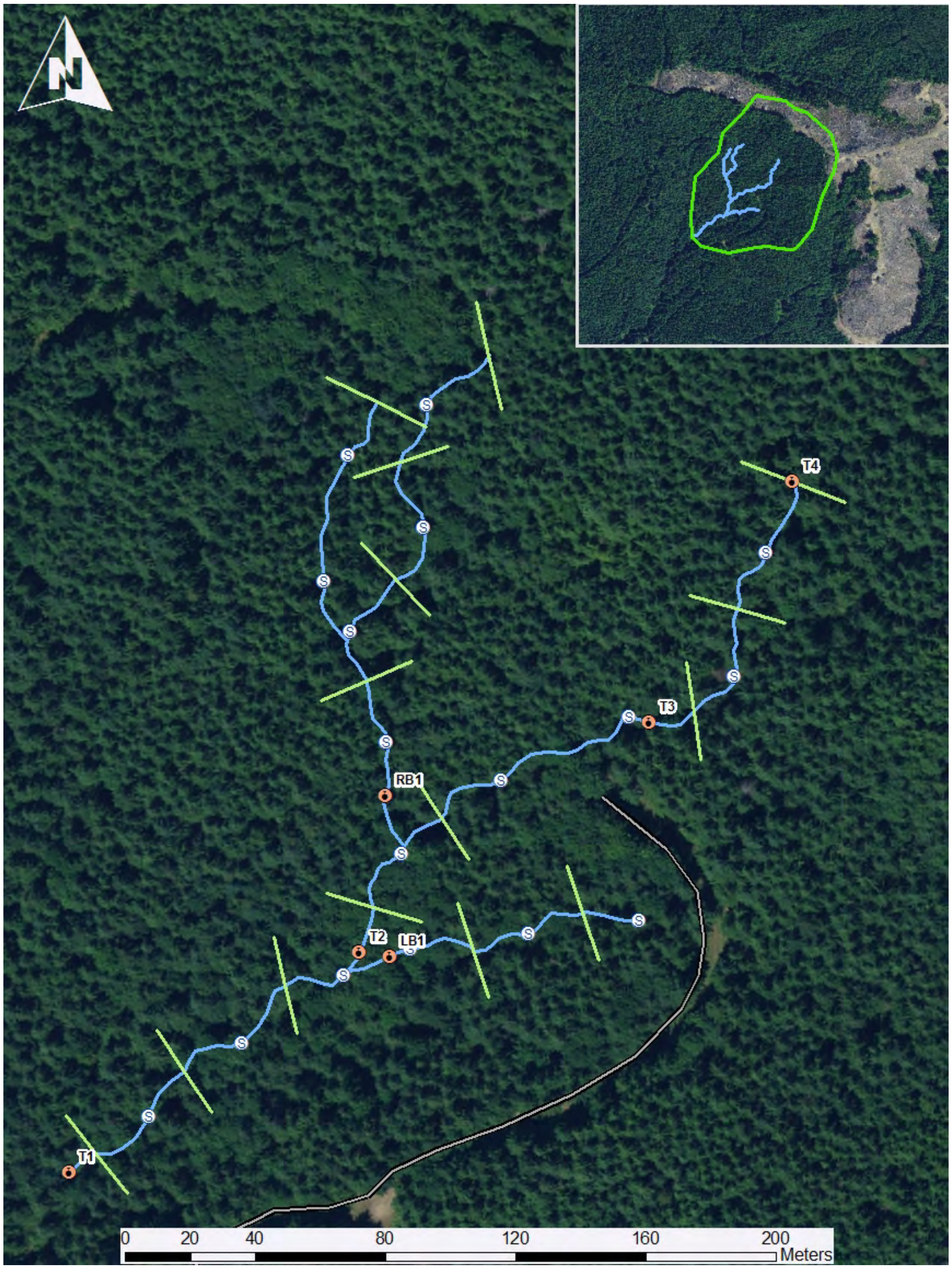
Basin Size (ha [ac])	16 (40)	Stream Length (m [ft])	1456 (4777)
Bankfull Width (cm)	120		
Wetted Width (cm)	61		
Stream Slope (%)	19		
Valley Wall Slope (%)	55		
Wetted Channel (%)	85		
Average Precip (cm [in])	269 (106)		

REF2



Basin Size (ha [ac])	15 (38)	Stream Length (m [ft])	856 (2808)
Bankfull Width (cm)	125		
Wetted Width (cm)	50		
Stream Slope (%)	20		
Valley Wall Slope (%)	53		
Wetted Channel (%)	76		
Average Precip (cm [in])	282 (111)		

REF3



Basin Size (ha [ac])	12 (29)	Stream Length (m [ft])	697 (2287)
Bankfull Width (cm)	94		
Wetted Width (cm)	35		
Stream Slope (%)	22		
Valley Wall Slope (%)	58		
Wetted Channel (%)	75		
Average Precip (cm [in])	292 (115)		

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