

1  
2  
3  
4  
5  
6  
7  
8  
9  
10  
11  
12  
13  
14  
15  
16  
17  
18  
19  
20  
21  
22  
23  
24  
25

# **Riparian Function Literature Review and Annotated Bibliography**

**Prepared for the Riparian Scientific Advisory Group  
(RSAG) of Washington State**

*Prepared by:*  
Benjamin Spei, Brandon Light, Mark Kimsey  
**March 2024**

26 Table of Contents

27 Background ..... 2

28 Focal Questions ..... 2

29 Methods ..... 3

30 Results/Summary of Review ..... 5

31 Discussion of findings relative to FPHCP objectives..... 0

32 Litter/Organic matter inputs/Nutrients..... 0

33 Large Wood (LW)/wood load/wood recruitment ..... 7

34 Bank Stability and Sediment ..... 16

35 Shade and stream temperature ..... 24

36 Results/discussion by focal question ..... 31

37 Focal Question 1..... 31

38 Focal Question 1a..... 42

39 Focal Question 1b ..... 46

40 Focal Question 1c..... 48

41 Focal Question 1d ..... 51

42 Focal Question 1e..... 51

43 Focal Question 2..... 52

44 Focal Question 3..... 59

45 Focal Question 4..... 72

46 Focal Question 5..... 73

47 Focal Question 6..... 77

48 Focal Question 7..... 81

49 References..... 83

50 Appendix I ..... 93

51 Appendix II ..... 124

52

53

54

55

56

57

## 58 Background

59 Washington State Forest Practices rules and management guidelines covered by the FPHCP  
60 (Forest Practices Habitat Conservation Plan, 2006) are strongly influenced by the science of  
61 riparian processes articulated in the FPHCP Environmental Impact Statement (EIS Chapter 6  
62 References, Appendix A Regional Summaries, Appendix B Riparian Modeling, 2005). The EIS  
63 references include the Forest Ecosystem Management Assessment Team (FEMAT) report,  
64 “Forest Ecosystem Management: an ecological, economic, and social assessment. Section V:  
65 Aquatic Ecosystem Assessment (1993).” Although the Forests and Fish Report and FPHCP and  
66 the rules derived from it considered many sources, our scientific understanding of riparian  
67 processes has evolved based on additional research that has been completed since then. More  
68 recent science has affirmed some aspects of the then-current state of knowledge on riparian  
69 processes and the effects of timber harvest on them. Still, some of the scientific conclusions are  
70 changing. In addition, riparian management strategies have evolved to address resource  
71 objectives. This review will look at literature that has been completed since the FEMAT and  
72 Forests and Fish report, and the FPHCP EIS. It will inform the Adaptive Management Program  
73 (AMP) committees and the Forest Practices Board (FPB) regarding the effects of forest harvest  
74 and other management practices on riparian functions and processes.

75 This review will follow a similar but modified format of the riparian literature review developed  
76 by Schuett-Hames et al. (2015) for the Cooperative Monitoring Evaluation and Research  
77 Committee (CMER) under the Westside Type F Prescription Effectiveness Monitoring project.  
78 However, this review will not focus only on Type F (fish-bearing streams) but on the response of  
79 riparian functions following harvest in all forests adjacent to rivers and streams. Priority will be  
80 given to studies conducted in areas with similar habitat and landscape characteristics as those  
81 found in the state of Washington. Further, information extracted from these studies will include  
82 the experimental designs used, sampling programs, sampled covariates, the metrics used to  
83 quantify covariates, and analytical methods.

84 We summarized the overall findings by key riparian function, and related physical processes, and  
85 provide a review to support recommendations for future research. The riparian functions  
86 specified in the FPHCP include “large woody debris recruitment, sediment filtration, stream bank  
87 stability, shade, litterfall and nutrients, in addition to other processes important to riparian and  
88 aquatic systems.” (FPHCP, 2006).

89 This literature review will address specific questions (listed below) and identify appropriate  
90 variables and associated metrics that can be used to quantify and assess timber harvest effects on  
91 the riparian functions.

92

## 93 Focal Questions

- 94 1. What are the effects of timber harvest intensities and extent on the riparian functions,  
95 with an emphasis on the five key functions listed above, in comparison to conditions  
96 before harvest?

- 97 a. What are the effects of thinning (intensity, extent) on the riparian functions, over  
98 the short and long-term compared to untreated stands?  
99 b. How do buffer widths and adjacent upland timber harvest prescriptions influence  
100 impacts of riparian thinning treatments?  
101 c. What are the effects of clearcut gaps in riparian stands (intensity, extent) on the  
102 riparian functions, over the short and long-term, compared to untreated stands  
103 d. How do buffer widths and upland timber harvest influence impacts of clearcut  
104 gaps treatments?  
105 e. What are the effects of any combinations of the above treatments?  
106 2. How and to what degree do specific site conditions (e.g., topography, channel width and  
107 orientation, riparian stand age and composition) influence the response of the riparian  
108 functions?  
109 3. What is the frequency of weather-related effects (e.g., windthrow, ice storms, excessive  
110 heat, flood and drought events) on riparian areas? What are the weather-related effects  
111 (positive and negative) on the riparian functions, and how are they distinguished from  
112 harvest effects? How do these effects differ between treated and untreated riparian  
113 forests?  
114 4. How do various treatments within riparian buffers relate to forest health and resilience to  
115 fire, disease, and other forest disturbances?  
116 5. How do the functions provided by riparian stands change over time (e.g., large woody  
117 debris recruitment from farther away from the stream)?  
118 6. Are there feedback mechanisms (e.g., microclimate changes within the riparian buffer)  
119 related to forest management that affect the recovery rates of riparian functions?  
120 7. What major data gaps and uncertainties exist relative to effects of timber harvest (both  
121 riparian and adjacent upland) on the riparian functions?

## 122 Methods

123 The riparian function literature review includes literature pertinent to the effects of timber  
124 harvest, management, natural disturbances (e.g., fire, disease, insect infestation, etc.), and  
125 channel geomorphology in riparian areas on the “five key riparian functions” as defined in the  
126 Forest Practices Habitat Conservation Plan (FPHCP, 2006). Literature searches were primarily  
127 conducted using the Web of Science and Google Scholar. Sources were also gathered via  
128 personal communication with employees and members of the Washington State Department of  
129 Natural Resources’ Cooperative Monitoring Evaluation and Research (CMER) scientific  
130 advisory groups. Technical reports on the United States Forest Service website were also  
131 investigated for their potential use. Finally, we also considered studies and manuscripts  
132 unpublished in formal scientific journals available on ResearchGate and ProQuest, including  
133 Ph.D. dissertations and master's theses. Papers returned from the keyword searches were initially  
134 screened by title and abstract. Papers were deemed appropriate for inclusion if they fit 3 criteria:  
135 (1) utilize experimental designs such as before-after-control-impact (BACI), after-control-impact  
136 (ACI), before-after-impact (BAI), after-impact (AI), simulation modeling, or meta-analysis to  
137 quantify the effect of riparian forest treatment, harvest, disturbance, site characteristics and  
138 conditions, etc. on riparian functions with an emphasis on the five key functions. Observational

139 studies that that substituted space for time (e.g., difference between old-growth and young  
 140 regenerating forests) were also included. (2) have been published or completed since the Forest  
 141 and Fish report, i.e., 1999, (3) have been conducted in western North America including coastal  
 142 Alaska, southern and coastal British Columbia, southern Alberta, the Pacific Northwest, the  
 143 Intermountain West, and the Great Basin regions. Studies from outside these areas were included  
 144 if they contained generalizable information about riparian functions (e.g., the relationship of  
 145 canopy cover with shade and temperature).

146 A list of search terms was developed to capture any studies relevant to the topics of the seven  
 147 focal questions (Table. 1). A master list of all returned study titles and abstracts from Web of  
 148 Science was also compiled for further analysis of keyword popularity and combinations (Figure  
 149 1).

150 Table 1. List of terms used in search of keywords and titles of literature sourced from Web of  
 151 Science. Terms in **bold** were used in all searches. Terms were grouped by topic (e.g.,  
 152 management, physiography, disturbance, etc.). Results show the number of publications returned  
 153 for each combination of search terms.

Key Words/title	Results
<b>(Riparian OR stream OR headwater Or Watershed) AND</b>	
(Function OR sediment OR nutrient OR woody debris OR large wood OR LWD OR woody debris recruitment OR shade OR temperature OR light OR litter OR water quality OR diversity OR wood*) AND/OR	15,138
(Manag* OR harvest OR thin* OR forest* OR forest operation OR buffer OR buffer strips OR gap* OR treat* OR clearcut OR clearcut gap)	12,602
(Topograph* OR physiograph* OR channel width OR stream width OR bankfull width OR valley constraint OR morphology OR diversity OR distance to stream OR Parent material OR soil OR litholo* OR geolog*)	12,381
(Disturbance OR fire OR windthrow OR ice storms OR drought OR flood* OR resilience OR resistance OR microclimate OR site conditions)	12,725
(Climate)	12,588
(feedback OR long-term OR short-term OR time)	12,150
(Forest health OR recovery OR regeneration OR disease OR insect OR fung* OR patho*)	12,328
(Stand structure OR stand age OR composition OR density OR structure OR species OR species composition)	12,214

Total titles and abstracts searched, excluding duplicates	<b>16,750</b>
---	---------------

154

155 From the initial title and topic review of the 16,516 papers sourced in our search, we refined the  
 156 list to 528 papers for consideration based on the 3 criteria listed above (e.g., utilize experimental  
 157 design with results focusing on at least one of the five key functions; published after 1999; were  
 158 conducted in western North America). From these 528 papers we further refined our list to 105  
 159 articles based on information gleaned from the abstract, introduction and methods sections  
 160 regarding study design and relevant geography. Of these 105 articles 91 provided information on  
 161 at least one of the five key functions and were thoroughly read and used to develop an annotated  
 162 bibliography (Appendix). The other 14 articles provided information and experimental results  
 163 about fire frequency and fire behavior in riparian areas, or effects of fire on one of the five key  
 164 functions. These 14 papers about fire were not included in the literature review but were  
 165 reviewed and discussed in focal questions 3 and 7. Frequency of the top 8 keywords were  
 166 represented in a histogram to express the popularity of topics in the literature since the year 2000  
 167 (Figure 1). We organized our review of the relevant literature by (1) FPHCP objective and (2)  
 168 focal question. A table was submitted along with this report that gives a more thorough  
 169 description of details used to categorize publications in supplemental materials (supplemental  
 170 table of references; S1).

171 Table 2. Frequency of keywords in the original 16,516 publications sourced from Web of Science

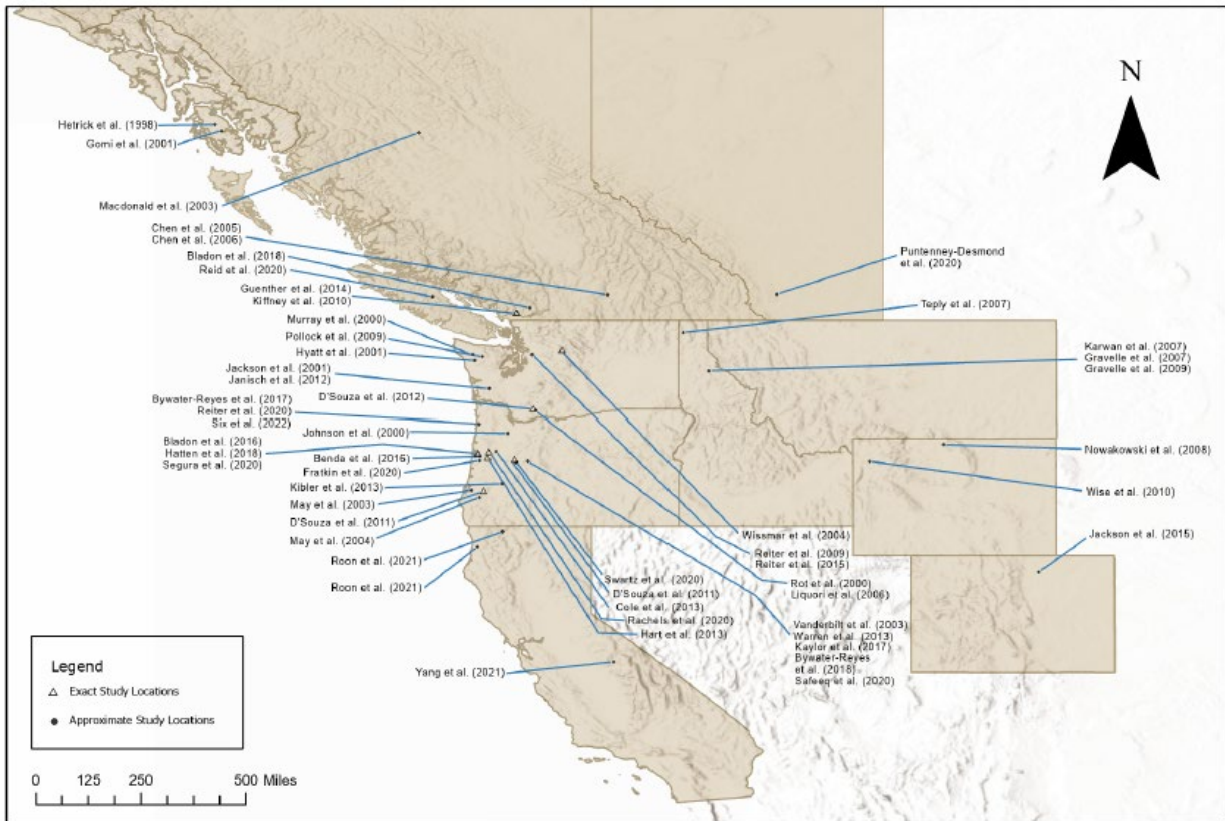
<b>Keywords</b>	<b>Count</b>
Water quality	1165
Streams	1004
Watershed	1000
Climate change	848
Watershed management	729
Riparian	652
Stream	604
Nitrogen	489

172

## 173 Results/Summary of Review

174 We conducted our review of the 72 relevant publications to (1) summarize the most current state  
 175 of knowledge of how timber harvest affects riparian function and related processes with a focus  
 176 on the five key riparian functions defined in the FPHCP, and (2) extract information that has the  
 177 potential to provide answers to, or methods and experimental designs that could be used to  
 178 answer the 7 focal questions. Our review focused primarily on peer-reviewed journal  
 179 publications but included 3 CMER reports and 1 report from the United States Forest Service  
 180 website. Of these 72 studies, 33 were conducted on headwater or non-fish-bearing streams, 16 on  
 181 fish-bearing streams, and 23 on a combination of fish and non-fish-bearing streams or

182 hypothetical streams in a model simulation (Table 3.). Most of the studies reviewed were  
 183 conducted in the Pacific Northwest region but several from just outside this region (British  
 184 Columbia, Alberta, Idaho, Montana, Wyoming, Colorado) were also included (Figure 2.). Few  
 185 studies could be found that quantify how riparian area treatments directly affect bank stability.  
 186 Several CMER studies, however, have investigated the effects of riparian timber management on  
 187 soil and streambank disturbance and erosion (Ehinger et al., 2021; McIntyre et al., 2018; Schuett-  
 188 Hames et al. 2011). In these studies, soil/bank disturbance and erosion were further analyzed for  
 189 their contribution to sediment export and delivery to streams. Because of this relationship  
 190 between bank erosion and sediment delivery, bank stability is discussed and reviewed in the  
 191 section with sediment. Further, because of the paucity of studies in the literature that provide  
 192 experimental evidence of how riparian area treatments affect bank stability, studies that  
 193 investigate bank stability or bank erosion based on other factors (e.g., vegetation type, vegetation  
 194 coverage) have been included and reviewed in question 7. These studies are provided as  
 195 recommendations for methods that could be used in an experimental design comparing changes  
 196 in bank stability before and after treatment or between treated and untreated riparian stands.



197  
 198 Figure 2. Locations where studies were conducted. References not listed include studies that  
 199 sourced data from multiple locations.

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Anderson & Meleason (2009)	The effects of buffer width in combination with thinned stands, patch openings, and unthinned stands on LW and vegetation cover.	5-6 years	2	LWD, vegetation	BACI experiment	Local, 6 reaches, 2 transects per reach	Oregon
Anderson et al. (2007).	The effects of forest mgmt. on stream shade and stream temperature.	3-6 years	2	SHD	BACI experiment	Local, 6 reaches, 2 transects per reach	Oregon
Bahuguna et al. (2010)	The effect of riparian buffer width on windthrow and LWD recruitment.	8 years	3	LWD	BACI experiment	Local, streams within 1 watershed	British Columbia
Benda et al. (2016).	The effects of forest management on LW recruitment	100 years in 5-year time steps	1	LWD	Simulation modelling from field data	Local, Alsea watershed	Oregon
Bilby & Heffner (2016)	Combination of literature and field experiments to determine factors contributing to litter delivery to streams.	1- year	4 mature sites, 3 young sites	LIT	Mixed effects modelling from field experiments	Local, data on windspeed collected from Humphrey Creek	Washington
Bladon et al. (2016)	Effects of buffers vs. no buffers on stream temperature.	6 years	6	SHD, stream temperature	BACI experiment	Local, Alsea watershed	Oregon



Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Bladon et al. (2018)	The effects of a variety of contemporary forest mgmt. prescriptions on small, headwater streams.	14 years	7	SHD	ACI experiment	Local, 3 watersheds: Alsea, Trask, and Hinkle watersheds	Oregon
Burton et al. (2016)	Instream wood loading at different buffer widths, basin geomorphologies, and harvest intensities.	15 years	6	LWD	BACI experiment	Regional, along Oregon coast and Cascade Range	Oregon
Bywater-Reyes et al. (2018)	Variability in suspended sediment yield over half-century.	60 years	10	SED	Modeling, regression analysis of historical data	Local, H.J. Andrews Experimental Watershed	Oregon
Bywater-Reyes et al. (2017)	Effect of forest mgmt. on stream sediment delivery.	6 years	10	SED	ACI experiment	Local, Trask River Watershed	Oregon
Chen et al. (2005)	Compares the LW biomass between 4 different mgmt. strategies.	1 year	4-5	LIT, LW, NUT	ACI experiment	Local, Okanagan Valley, Kelowna	British Columbia, Canada
Chen et al. (2006)	Assesses the amount, distribution, dynamics, and function of LW in forest streams	2 years	35	LW	ACI experiment	Local, Okanagan Valley, Kelowna	British Columbia, Canada
Cole & Newton (2013)	Effect of 3 different retention buffer prescriptions on stream temperature.	6-7 years	4	SHD	BACI experiment	Local, within a radius of 200 km of Corvallis	Oregon

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Deval et al. (2021)	Disturbance effects on stream chemistry.	13 years	7	NUT	BACI experiment	Local, Mica Creek Experimental Watershed	Idaho
Ehinger et al. (2021)	Effectiveness of riparian mgmt. in maintaining function in for non-fish-bearing headwater streams on incompetent lithologies.	9 years	11 (two treatments)	LW, NUT, SED, SHD, Stream Temperature	BACI experiment	Regional	southwestern Washington
Fox & Bolton (2007).	observational study that categorizes the effects of riparian site geomorphology on LW recruitment.	1 year, multiple age classes, covertypes and disturbance histories	150	LW	Descriptive, spatial modeling on historical data	Regional, Coastal, West and east Cascade Range of Washington State	Washington
Gomi et al. (2001)	LW recruitment in the short and long-term under 5 different mgmt. strategies.	1 year,	3	LW, SED	ACI experiment	Local, Maybeso Experimental Forests	Alaska
Gravelle & Link (2007)	The impacts of timber harvest practices on stream temperature.	13 years	3	SHD	BACI experiment	Local, Mica creek Experimental Watershed	Idaho
Gravelle et al. (2009).	The effects of contemporary forest practices on the chemical properties of headwater streams and downstream locations.	14 years	3	NUT, SED	BACI experiment	Local, Mica creek Experimental Watershed	Idaho

<b>Reference</b>	<b>Purpose</b>	<b>Sampling Period</b>	<b>Sample size (n)</b>	<b>Function / process</b>	<b>Study Type</b>	<b>Scale (local, regional, global)</b>	<b>State/Province</b>
Groom et al. (2011a)	The effect of forest management on stream shade and stream temperature under Oregon forest practice rules	7 years	Unbalanced (15 state-owned and 18 private-owned)	SHD, stream temperature	BACI experiment	Regional, Oregon Coast Range	Oregon
Groom et al. (2011b)	The efficacy of new riparian management protocols in preserving stream side shade and in-stream temperatures.	7 years	Unbalanced (15 state-owned and 18 private-owned)	SHD, stream temperature	BACI experiment	Regional, Oregon Coast Range	Oregon
Guenther et al. (2014)	Differences in surface/sub-surface variability as well as influences of partial retention harvesting on stream temperature.	2 years	3	SHD	BACI experiment	Local, Malcolm Knapp Research Forest	British Columbia, Canada
Hart et al. (2013)	What riparian forest characteristics influence litter input to streams.	2 years	5	LIT, NUT	ACI experiment	Local, 5 contiguous watersheds in Oregon Coast range	Oregon
Hatten et al. (2018)	The effect of contemporary and historical forest harvesting practices on suspended stream sediment.	12 years	3	SED	ACI experiment	Local, Central Oregon Coast Range	Oregon

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Hough-Snee et al. (2016)	Evaluates which riparian, geomorphic, and hydrologic attributes are most strongly correlated to instream wood loads.	2 years	7	LW, SHD	Modeling, corelative analysis	Regional, interior Columbia River basin	Canada, Oregon, Washington, Idaho
Hunter & Quinn (2009)	How differences in stream geomorphology affect water temperature.	2 years	2	stream temperature	AI experiment	Local, Olympic Peninsula	Washington
Hyatt & Naiman (2001)	The depletion rate of LW in streams by size and species.	1 year; Dendrochronology to estimate up to 50 years.	4	LW	AI	Local, Queets Ricer	Washington
Jackson & Wohl (2015)	Instream wood loads and geomorphic effects between streams draining montane forests of different ages.	1 year	10 sites > 200 years old, 23 young sites <200 years old	LW	CI, regression analysis	Local, Arapaho and Roosevelt National Forests	Colorado
Jackson et al. (2001)	Effect of forest mgmt. on stream temperature, LW, and stream sediment, between clearcut, thinned, and buffered treatments.	2 years	unbalanced: 4 -6	LW, SED	BACI experiment	Local, northwestern Washington Coast Range	Washington

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Janisch et al. (2012)	The response of stream temperature to forest harvest, testing differences in continuous vs. patch buffers.	4-5 years	unbalanced: 5-6	SHD	BACI experiment	Local, southwestern Washington Coast Range	Washington
Johnson & Jones (2000)	Short-term and long-term effects of forest harvest on stream temperatures.	Historical dataset 1959-1982	3	SHD	BACI experiment	Local, H.J. Andrews Experimental Watershed	Oregon
Karwan et al. (2007)	Effects of timber harvest on suspended sediments in streams following timber harvest.	3 years	2	SED	BACI experiment	Local, Mica creek Experimental Watershed	Idaho
Kaylor et al. (2017)	Examines the effects of riparian forest harvest and varying stages of stand recovery on light availability 50-60 years post treatment.	1 year, 50 - 60 years post treatment	14	SHD	AI	Local, H.J. Andrews Experimental Watershed	Oregon
Kibler et al. (2013)	Examined the effects of contemporary forest practices on warm-season stream temperature regimes in headwater streams.	3.5 years	8	SHD	BACI experiment	Local, Hinkle Creek	Oregon

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Kiffney & Richardson. (2010)	Evaluates the effects of forest mgmt. on organic matter/ litterfall recruitment.	8 years	Unbalanced: 2-3	LIT	ACI experiment	Local, southwestern British Columbia	British Columbia, Canada
Liquori (2006)	Examines differences in post-harvest ecological and geomorphic processes in buffered forest sites	1 year	Unbalanced: 4-9	Other processes, disturbance post-harvest	AI	Local, managed tree farm in Cascade Mountains of western Washington	Washington
Litschert & MacDonald (2009)	Assessed streamside management zones to understand characteristics of the sediment delivery pathways following upland harvest.	1 year	200	SED	AI	Regional, National Forests in the Sierra and Cascade mountains.	California
Macdonald et al. (2003b)	Examined the effects of three different variable retention harvesting prescriptions on stream temperature	7 years	5	SHD	ACI experiment	Local, Baptiste and Galuski watersheds	British Columbia, Canada
Macdonald et al. (2003a)	Evaluates the effects of 2 different harvest prescriptions on suspended sediment concentrations.	6 years	2	SED	BACI experiment	Local, Baptiste watershed	British Columbia, Canada

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Martin & Grotefendt (2007)	Compared site conditions between riparian buffer strips and unlogged riparian stands using aerial photography to determine mortality and LW recruitment	1 year	9	LW	ACI experiment	Regional, northern and southern portions of southeast Alaska	Alaska
May & Gresswell (2003)	Investigates the mechanisms responsible for LW recruitment into streams.	2 years	4	LW, SED	modeling, Regression analysis	Local, North Fork of Cherry Creek Research Natural Area	Oregon
McIntyre et al. (2018)	Effectiveness of forest mgmt. in maintaining function for non-fish-bearing headwater streams on competent lithologies.	6 years	17 (four treatments)	SHD, SED, NUT, LW, LIT, stream temperature	BACI experiment	Regional	western Washington
McIntyre et al. (2021)	Continuation of McIntyre et al., 2018 to assess changes over a longer time period (up to 9 years post-harvest).	12 years	17 (four treatments)	LW, NUT, SED, SHD, stream temperature	BACI experiment	Regional	western Washington
Meleason et al. (2003)	Evaluate of the potential effects of different riparian mgmt. strategies on the standing stock of wood.	Simulation modeling of 720 years	1	LW	Modeling	simulation of stream types common in PNW	PNW, hypothetical stream

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Mueller & Pitlick (2013)	Examines the relative importance of lithology as a driver of sediment delivery into streams.	multiple datasets ranging 5-90 years	83	SED	spatial modeling, correlative analysis of historical data	Regional, Northern Rocky Mountains	ID, WY, MT
Murray et al. (2000)	Examined the influence of partial harvesting on stream temperature, chemistry, and turbidity 10-15 years post treatment..	2 years, 10-15 years after treatment	1	NUT, SED, SHD	ACI experiment	Local, Rock and Tower Creek watersheds	Washington
Nowakowski & Wohl (2008)	Examined differences in wood load and valley/channel characteristics between managed and unmanaged riparian areas.	1 year	19	LW	ACI experiment	Local, Upper Tongue River and North Rock Creek watersheds	Wyoming
Pollock et al. (2009)	The influence of forest harvests on stream temperature.	2 months	33	SHD	ACI experiment	Local, Hoh river Basin, and Clearwater River Basin	Washington
Puntenney-Desmond et al. (2020)	The potential effect of climate change on sediment yield and concentrations in riparian area run-offs.	1 month	15	SED	BACI, simulated rainfall in field plots	Local, Star Creek headwater catchment	Alberta, Canada
Rachels et al. (2020)	Investigates the source of suspended sediment to a stream draining a recent harvested catchment.	1 summer	1	SED	ACI experiment	Local, Enos Creek	Oregon



Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Reid & Hassan (2020)	Combines a wood budget model and a 45-year record of LW to examine changes in LW characteristics.	Long-term dataset from 1973-2017, simulated 300 years	8	LW	Simulation Modeling for framework development	Local, Carnation Creek	BC, Canada
Reiter et al. (2009)	Effects of forest practices on sediment production at the watershed-scale with 30 years of water quality data.	Long-term dataset from 1975-2005	4	SED	AI	Local, Deschutes River watershed	Washington
Reiter et al. (2015)	Long-term combined effects of hydro-climatic factors and intensively managed forests with buffers on stream temperature.	Long-term dataset from 1975-2009	4	SHD	BAI experiment	Local, Deschutes River watershed	Washington
Reiter et al. (2020)	Effects of harvesting and variable buffer widths on stream temperature	10 years	Unbalanced: 3-7	SHD	BACI experiment	Local, Trask River Watershed	Oregon
Roon et al. (2021a)	Thinning effects of second growth redwood forests in northwestern California.	2 years	3	SHD	BACI experiment	Local, Tectah and Lost Man watersheds	California

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Roon et al. (2021b).	Investigation of how different thinning intensities affect stream temperature via loss of canopy cover at local and watershed scales.	2 years	3	SHD, stream temperature	BACI experiment	Local, Tectah and Lost Man watersheds	California
Safeeq et al. (2020)	Presents an approach at isolating the streamflow effect on sediment delivery post-harvest.	Long-term dataset, 1952-2016	2	SED	BACI experiment	Local, H.J. Andrews Experimental Watershed	Oregon
Schuett-Hames & Stewart (2019a)	comparison of LW inputs, tree fall, and stand structure 5 years post-harvest.	5 years	Unbalanced: 8-9	LW	ACI experiment	Regional, northeastern Washington, 1 site in East Cascades	Washington
Schuett-Hames & Stewart (BCIF), (2019b)	The study analyzes the changes in stand structure, buffer tree mortality, and riparian functions 10 years after upland timber harvest.	10 years	Unbalanced: 3-14	LW, SED, SHD	ACI experiment	Regional, western Washington Coast and Cascade Range	Washington
Schuett-Hames et al. (2011)	Evaluates the effects of forest mgmt. on stream shade, LW recruitment, and sediment delivery.	5 years	Unbalanced: 3-15	LW, SED, SHD	ACI experiment	Regional, western Washington Coast and Cascade Range	Washington

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Six et al. (2022)	Assessed differences in levels of riparian buffer retention at mitigating changes to organic matter dynamics.	2 years	3	LIT, LW	BACI experiment	Local, Trask River Watershed	Oregon
Sobota et al. (2006)	Study of riparian characteristics and their effects on tree fall direction and in-stream recruitment.	3 years	21	LW	model with field data	Regional, Pacific Northwest and Intermountain West	Idaho, Washington, Oregon, Montana
Sugden et al. (2019).	Assessed the efficacy of Montana SMZ guidelines for controlling stream temperature.	2 years	30	SHD	BACI experiment	Regional, Western Montana	Montana
Swartz, et al. (2020)	Assessed whether experimental canopy gaps meant to mimic natural disturbances affect stream temperature	2 years	6	SHD	BACI experiment	Local, Mckenzie River Basin	Oregon
Teply et al. (2007)	Compares the effects of mgmt. harvest prescriptions and no-harvest RMZs on LW recruitment in streams.	1 year data collection, 100 years simulated	58	LW	Simulation Modeling	Local, Priest Lake Watershed	Idaho
Vanderbilt et al. (2003)	Correlation of nutrient inputs with weather events (mainly precipitation).	long-term datasets, ranging from 20-30 years	6	NUT	ACI experiment	Local, H.J. Andrews Experimental Watershed	Oregon

Reference	Purpose	Sampling Period	Sample size (n)	Function / process	Study Type	Scale (local, regional, global)	State/Province
Warren et al. (2013)	Evaluates stand age and associated canopy structural differences on stream light in second-order streams.	1 year	2	SHD	ACI experiment	Local, H.J. Andrews Experimental Watershed	Oregon
Wing & Skaugset (2002)	Examines the relationship between channel characteristics and LW in streams.	Extensive spatial dataset from 1990-1996	3793	LW	modeling, regression analysis	Regional, Western Cascade and Coast Range of Oregon	Oregon
Wise et al. (2010)	Uses tree rings to augment previous records to reconstruct multi-century data for the Snake River.	Dendrochronology records from 1600-2005	3	Drought Frequency	Climate reconstruction from dendrochronology records	Local, 3 sites in western Wyoming	Wyoming
Yang et al. (2021)	Examined the temporal variation in response of downstream water chemistry to prolonged drought and forest thinning.	5 years	2	NUT	BACI experiment	Local, The Kings River Experimental Watershed	California
Yeung et al. (2019)	Modelled the post-harvest response of leaf litter coarse particulate organic matter quantity in a coastal stream	Published data spanning 4-5 years	Total n not reported	LIT	Heuristic modeling	CPOM data from local, streams in coastal BC	Model developed from multiple North American sites

## 204 Discussion of findings relative to FPHCP objectives

### 205 Litter/Organic matter inputs/Nutrients

206 Prior to the Forest and Fish Report (1999), studies that directly quantify the effects of timber  
207 harvest within riparian areas on litter and organic matter (OM) input into streams in managed  
208 watersheds of western north America are sparse. Two seminal studies, one from the H.J.  
209 Andrews experimental watershed studies (Gregory et al., 1987) and one from the Carnation  
210 Creek experimental watershed (Hartman & Scrivener, 1990) present results that estimate loss of  
211 litter input following harvest. Gregory et al., (1987) which was part of the Streamside  
212 Management: Forestry and Fishery Management collection produced by Salo & Cundy (1987)  
213 noted that removal of the forest canopy from timber harvesting resulted in decreases in annual  
214 litter fall from 300-400 g/m<sup>2</sup> in the mature forests to less than 100 g/m<sup>2</sup>. Further, they posit that  
215 decreased litter inputs after logging can persist for 10 – 20 years before recovering. Results from  
216 Hartman & Scrivener, (1990) showed that litter inputs post-logging were 25-50% of pre-logging  
217 levels with about 50% of the loss recovering within a decade (note: buffer widths varied from 1-  
218 70 m, litter input loss was not analyzed by buffer width).

219 Experimental studies published after 1999 that investigate the factors affecting litter and organic  
220 matter (OM) input (not including LW) into streams in western North America are still relatively  
221 few. In our search we found six papers that quantify the effects of timber harvest or the effects of  
222 site factors (e.g., topography, vegetation characteristics) Four of these studies focus on headwater  
223 streams and two of the studies reviewed here extend into larger fish-bearing streams (Bilby &  
224 Heffner, 2016; Hart et al., 2013; Kiffney & Richardson, 2010; McIntyre et al., 2018; Six et al.,  
225 2022; Yeung et al., 2019).

226 Studies specifically investigating controls on litter inputs used litter traps for sample collection  
227 and quantify changes in litter delivery from dry weight. Before litter quantification, it is  
228 commonly separated by type (e.g., leaves, twigs, cones, etc.), species (e.g., hardwood, conifer),  
229 season, and distance from stream. Litter weights are usually compared with treatment (e.g.,  
230 harvest intensities, buffer widths), site factors (e.g., slope, species composition, stand density,  
231 distance to stream), and local weather conditions (e.g., precipitation, wind speed) with statistical  
232 or simulation modeling.

233 In terms of site factors, Bilby & Heffner (2016) used a combination of field experiments,  
234 literature review, and modeling to estimate the relative importance of factors affecting litter  
235 delivery from riparian areas into streams of western Washington in the Cascade mountains at  
236 high and low elevations. Their results showed that under the wind conditions recorded at  
237 Humphrey Creek, most litter recruited into the stream originated from within 10 m of the stream  
238 regardless of litter or stand type. No difference was found in delivery distance and litter type  
239 (needles or broadleaf) at young sites. However, needles released at mature sites had a higher  
240 proportion of cumulative input from greater distances than needles or alder leaves released at  
241 younger sites. Litter travel distance was linearly related to wind speed ( $p < 0.0001$ ). Doubling  
242 wind speed at one site led to a 67-87% expansion of the riparian litter contribution zone in the

243 study area. The results also reveal a trend that suggests slope affects the width of the litter  
244 contributing area. However, the authors did not apply statistical analysis to these values and only  
245 speculate that increasing the slope from 0-45% would increase the width of the litter contributing  
246 area by up to 71% for needles and 95% for leaves. From these results, Bilby & Heffner (2016)  
247 suggest that wind speed has a strong effect on the width of litter delivery areas within riparian  
248 areas, but that relationship is also affected by stand age (suggesting that tree height was a factor)  
249 and litter type (deciduous vs. conifer). Other than stand structure and topography, another study  
250 shows evidence of species composition affecting litter delivery into streams. Hart et al. (2013)  
251 compared litter delivery into streams between riparian zones dominated by deciduous (red alder)  
252 and coniferous (Douglas-fir) tree species in western Oregon. Results from this study show that  
253 deciduous forests dominated by red alder delivered significantly greater vertical and lateral  
254 inputs ( $\text{g m}^{-2} \text{y}^{-1}$ ) to adjacent streams than did coniferous forests dominated by Douglas-fir.  
255 Deciduous-site vertical litter input (mean =  $504 \text{ g m}^{-2} \text{y}^{-1}$ ) exceeded that from coniferous sites  
256 ( $394 \text{ g m}^{-2} \text{y}^{-1}$ ) by  $110 \text{ g/m}^2$  over the full year. Annual lateral inputs at deciduous sites ( $109 \text{ g}$   
257  $\text{m}^{-2} \text{y}^{-1}$ ) were  $46 \text{ g m}^{-2} \text{y}^{-1}$  more than at coniferous sites ( $63 \text{ g m}^{-2} \text{y}^{-1}$ ). The timing of the  
258 inputs also differed, with the greatest differences occurring in November during autumn peak  
259 inputs for the deciduous forests. Further, annual lateral litter input increased with slope at  
260 deciduous sites ( $R^2 = 0.41$ ,  $p = 0.0771$ ), but showed no strong relationship at coniferous sites ( $R^2$   
261  $= 0.1863$ ,  $p = 0.2855$ ). These results were partially consistent with Bilby & Heffner (2016) in  
262 that they suggest litter type, and topography (slope) can affect the litter input rates. Lateral litter  
263 movement in the riparian area increased with slope for deciduous riparian forests throughout the  
264 year and for coniferous forests only in the spring and summer months.

265 In terms of the effects of timber harvest on litter and OM quantity in streams, 4 studies in our  
266 review were found that provide experimental results that have been conducted since 2000 and  
267 focus on western North America. Of these four studies, one used simulation modeling (Yeung et  
268 al., 2019), and the other 3 (Kiffney & Richardson, 2010; McIntyre et al., 2018; Six et al., 2022)  
269 used field-based experiments to estimate the effects of timber harvest within riparian forests on  
270 OM inputs and dynamics in streams. Yeung et al. (2019) simulated post-harvest responses to  
271 leaf-litter derived coarse particulate organic matter (CPOM) quantity in a coastal rainforest  
272 stream in British Columbia, Canada. For this study, Yeung et al. (2019) used published empirical  
273 data from representative small, forested streams in coastal British Columbia to calibrate and set  
274 parameters for their CPOM model. The model compared the effects litterfall reduction, increase  
275 in peak flows, and increase in stream temperature (estimated for 4 harvesting intensities based on  
276 available data) on in stream CPOM standing stocks. Results showed evidence that litterfall  
277 reductions from timber harvest was the strongest control on in-stream CPOM quantity for 4 years  
278 post-harvest. However, when litterfall reductions were below 30%, the effect size varied with  
279 relative changes to peak flows and stream temperature. Stream temperature increases specifically  
280 showed a significant interaction with litterfall reductions. The authors propose that the decreased  
281 activity of CPOM consumers caused by increasing stream temperatures by  $4 \text{ }^\circ\text{C}$  or more, may be  
282 enough to offset the loss of litterfall inputs of CPOM stocks. This speculation was made based on  
283 the temperature dependent function of leaf litter consumption by common shredder species and  
284 temperature ranges modeled by Stenroth et al. (2014). This model predicts shredder activity is  
285 optimized at  $\sim 15 \text{ }^\circ\text{C}$  (ranging between  $13.7 - 16.7^\circ\text{C}$ ) but begins to quickly decline at

286 temperatures above 16 °C. The caveat of this study is that it did not include LW dynamics in  
287 preserving CPOM post-harvest.

288 All four studies that applied an experimental design to assess the changes in litter and OM  
289 delivery into streams used a Before-After Impact-Control (BACI) design. Also, all these studies  
290 compared changes in litter and OM inputs into streams for two or more riparian forest harvest  
291 prescriptions (Table A1; Appendix I). Kiffney & Richardson (2010) compared changes in litter  
292 input between riparian harvest prescriptions that included clear-cut to stream edge, 10 m wide  
293 buffer reserve, 30 m buffer reserves, and an uncut control over the course of 8 years. No thinning  
294 was applied within the reserves. Upland treatment at all sites applied clearcut. Results showed  
295 differences in litter flux relative to riparian treatment persisted through year 7, while a positive  
296 trend between reserve width and litter flux remained through year 8. Needle inputs remained 6x  
297 higher in the buffer and control sites through year 7, and 3-6x higher in year 8 than in the  
298 clearcut sites. Twig inputs into the control and buffered sites were ~25x higher than in the  
299 clearcut sites in the first year after treatment. The linear relationship between reserve width and  
300 litter inputs was strongest in the first year after treatment, explaining ~57% of the variation, but  
301 the relationship could only explain ~17% of the variation in litter input by buffer width by year 8  
302 (i.e., the relationship degraded over time). The authors interpret these results as evidence that  
303 litter flux from riparian plants to streams, was affected by riparian reserve width and time since  
304 logging.

305 McIntyre et al. (2018) also experimentally assessed the difference in the changes in litterfall  
306 inputs into streams following four treatments: an unharvested control (Reference), current Forest  
307 Practices that apply a two-sided 50-ft riparian buffer along at least 50% of the stream (FP; with  
308 clearcut to stream's edge outside of the buffer), a two-sided 50-ft buffer along the entire stream  
309 (100%), and a clearcut to stream without a buffer (0%). The upland forests of all treatments were  
310 clearcut harvested. Results for litterfall input showed a significant decrease in total litterfall  
311 (includes leaves/needles, twigs, cones etc.) input in the FP and 0% treatments between pre- and  
312 post-treatment periods (2 years of pre-, and 2 years of post-harvest data). However, compared to  
313 the Reference streams, only the 0% treatment (unbuffered) showed a significantly lower litterfall  
314 input post-harvest and only for deciduous leaves, and combined total of deciduous leaves and  
315 conifer needles. The 100% buffer showed a non-significant increase in litterfall inputs relative to  
316 the reference streams. The authors interpret these results as evidence that the riparian vegetation  
317 community in the unbuffered treatment had not recovered by the end of year 2 post-harvest.

318 Six et al. (2022) also investigated the effects of timber harvest on litter inputs. However, this  
319 study had no replication in their design for each treatment and only 2 control sites (i.e., n = 1 for  
320 each treatment). The results are presented here because there is a general lack of studies available  
321 in the literature after 2000 that provide experimental evidence of the effects of riparian timber  
322 harvest on litterfall inputs into streams. Six et al. (2022) compared changes in litterfall pre- and  
323 post-treatment between sites with a complete clearcut to stream, a clear cut with leave trees  
324 (retention of 5 trees per hectare), clearcut with a 15 m no-cut retention buffer, and an uncut  
325 control. Because of the small sample sizes, no tests for significance could be applied. However,  
326 the authors interpreted the data with descriptive statistics and graphical summaries. Their results

327 showed post-harvest litter delivery decreased for the clearcut with no leave trees but increased  
328 for both the clearcut with leave tree and clear cut with retention buffer. These results are  
329 somewhat consistent with those of McIntyre et al., (2018) which showed significant decreases in  
330 litter delivery only in sites with no retention buffer.

331 The objective of the study from Wooton (2012) was to assess how riparian area treatments  
332 impact river food webs with an emphasis on economically important salmonid species in an  
333 Olympic Peninsula River in Washington state. However, they present results and statistical  
334 analysis for differences in litter inputs ( $\text{g m}^{-1} \text{hr}^{-1}$ ) between treated and untreated reaches.  
335 Because of the lack of litter input studies in literature, their results are presented here. Wooton  
336 (2012) removed the dominant tree species, red alder (*Alnus rubra*), from one bank along five  
337 treatment reaches ranging from 100-300 m long and replaced them with conifer seedlings. Paired  
338 control reaches were interspersed between treated reaches along the stream. Specific methods for  
339 tree removal or width of buffer in treatment reaches were not reported. Leaf litter decreased  
340 significantly ( $p = 0.04$ ) in the treatment reaches compared to the control reaches ( $4.92 + 2.55$  vs.  
341  $14.12 + 5.70 \text{ g m}^{-1} \text{hr}^{-1}$ ).

#### 342 *Nutrients*

343 Riparian timber management practices in the 1970s were developed for water quality standards  
344 with the development of the Clean Water Act of 1972, based on nutrient concentrations and  
345 water clarity. Before implementing these BMPs, timber harvest practices included clearcut to the  
346 stream edge, burning of slash, and application of pesticides which resulted in large and  
347 immediate increases in stream water nutrient concentrations that remained higher than pre-  
348 harvest or reference stream values for months and even years (Brown, 1973; Fredriksen, 1975).  
349 However, BMP development and implementation over the past several decades have shown  
350 evidence of their effectiveness in minimizing these effects both in magnitude and across time  
351 (Deval et al., 2021; Shah et al., 2022; Stednick, 2008). For example, Shah et al. (2022) in their  
352 global review of the effects of forest management on water quality under contemporary  
353 management practices concluded that the development of BMPs across the world has resulted in  
354 reduced or in some cases, undetectable impacts on water quality. However, they also report that  
355 harvest impacts on nutrient concentrations can be complex and depending on the management  
356 practices implemented, their effects may manifest many years after the work has been completed  
357 (e.g., slow decomposition of slash, regrowth of vegetation, changes in land use). Indeed,  
358 Sweeney & Newbold (2014) in their literature review and synthesis on the efficacy of forest  
359 buffers in protecting water quality based on buffer width, remark on the high variability of  
360 responses across studies. They report that removal of nitrogen from upland sources per unit  
361 width of a forested buffer varied inversely with subsurface water flux. This suggests factors that  
362 influence water flux through the buffer (e.g., hillslope gradient, soil porosity, vegetation type and  
363 composition, precipitation) also impact buffer efficacy in removing nutrients and pollutants.

364 Zhang et al. (2010) in a review and meta-analysis of the effectiveness of buffers in reducing  
365 nonpoint source pollution found comparable results. They reported slope (hillslope gradient) as  
366 having a linear relationship with buffer pollutant removal efficacy that switched from positive to  
367 negative when slope increased beyond 10% (i.e., hillslope gradients of ~10% were optimal for



368 buffer efficacy in removing pollutants). However, there may be some variation in these  
369 relationships based on the nutrient or pollutant observed (e.g. form of nitrogen, phosphorus, etc.).  
370 For example, Vanderbilt et al. (2003) analyzed long-term datasets (ranging 20-30 years for each  
371 watershed) to investigate patterns in dissolved organic nitrogen (DON) and dissolved inorganic  
372 nitrogen (DIN) export with watershed hydrology. Their results showed that total annual  
373 discharge was a positive predictor of annual DON export in all watersheds with  $R^2$  values  
374 ranging between 0.42 to 0.79. In contrast, relationships between total annual discharge and  
375 annual export of nitrate ( $\text{NO}_3\text{-N}$ ), ammonium ( $\text{NH}_4\text{-N}$ ), and particulate organic nitrogen (PON)  
376 were variable and inconsistent across watersheds. The authors speculate that different factors  
377 may control organic vs. inorganic N export.

378 In our search of the literature, four studies were found that provide experimental evidence of the  
379 effects of riparian timber harvest on nutrient flux in western north America and were published  
380 since 2000. Gravelle et al., 2009 compared the effects of contemporary forest harvesting  
381 practices in Idaho on nutrient cycling and in stream concentrations. This study followed the  
382 BACI design and featured a pre-treatment measurement phase (5 years), a post-road construction  
383 phase (5 years), and a post-harvest phase (5 years). Treatments imposed included a clearcut to  
384 stream with 30-foot equipment exclusion zone (non-fish-bearing), a target reduction of 50% of  
385 the canopy removal over 50% of the area, equating to 25% removal of existing shade (fish-  
386 bearing streams), and was compared to an uncut reference. Results for the post-road construction  
387 period showed no significant changes in any analyzed nutrient concentrations. Results for the  
388 post-harvest period showed significant increases in monthly mean nitrate and nitrite ( $\text{NO}^3$  and  
389  $\text{NO}^2$ ) at sites immediately downstream from the clearcut, the partial harvest, and at sites  
390 downstream from both treatments in the stream network (cumulative). The changes in monthly  
391 mean  $\text{NO}_3$  and  $\text{NO}^2$  during the five years post-harvest were greatest for the clearcut treatment  
392 ( $+0.29 \text{ mg L}^{-1}$ ), followed by the cumulative ( $+0.07$  and  $+0.05 \text{ mg L}^{-1}$ ) and partial harvest ( $+0.03$   
393  $\text{mg L}^{-1}$ ).  $\text{NO}^3$  showed progressively increasing monthly concentrations for 3 years after harvest  
394 before declining. None of the other nutrients analyzed in this study (Kjeldahl nitrogen (TKN),  
395 total phosphorus (TP), total ammonia nitrogen (TAN) consisting of un-ionized ( $\text{NH}^3$ ) and ionized  
396 ( $\text{NH}^{4+}$ ) ammonia, and unfiltered orthophosphate (OP) samples) showed significant changes  
397 during the post-harvest period.

398 In a follow up study, Deval et al. (2021) compared changes to nutrient concentrations 8 years  
399 after Gravelle et al. (2009) completed their study. During these 8 years (extended harvest period)  
400 the extent and frequency of harvest operations increased. Treatments consisted of additional road  
401 construction and timber harvest (clearcut), with site management operations including pile  
402 burning and competition release herbicide application. Following these treatments, streams in all  
403 harvested watersheds again experienced significant increases in  $\text{NO}^3 + \text{NO}^2$  concentrations of  
404 even higher magnitude than during the first post-harvest period. Further, there were also small  
405 but significant increases in mean monthly total phosphorus (TP) concentrations at all treatment  
406 sites, including the downstream cumulative site. Cumulative  $\text{NO}^3 + \text{NO}^2$  concentrations increased  
407 throughout the study but showed signs of recovery in one watershed approximately 3 years after  
408 the last treatment (clearcut, broadcast burn, herbicide). The authors attribute the increase in

409  $\text{NO}^3+\text{NO}^2$  and TP during the extended harvest periods (i.e., beyond what was observed in the  
410 first post-harvest period) to the application of herbicides and broadcast burning.

411 In general, the authors of both these studies (Deval et al 2021; Gravelle et al., 2009) concluded  
412 that Idaho BMPs for riparian forest harvest are effective in reducing sediment and pollutants into  
413 streams. While there were significant increases in nitrate and nitrite concentrations following  
414 management operations, levels never increased above acceptable values for water quality  
415 standards and there was evidence of nitrogen recovery to pre-harvest (or unharvested) levels  
416 after 3 years.

417 Considering the interaction between climate and forest harvest on nutrient transport, Yang et al.  
418 (2021) investigated the effects of drought and forest thinning operations (independently and  
419 combined) on stream and soil water chemistry in the Mediterranean climate headwater basins of  
420 the Sierra National Forest. Data on water chemistry were taken 2 years prior and 3 years  
421 following drought and thinning operations in two watersheds, each with thinned and control  
422 stands. Young stands with high shrub cover (> 50%) were masticated to < 10% shrub cover. The  
423 thinning prescription in mature stands removed trees across all diameter classes to a target basal  
424 area range of 27–55  $\text{m}^2 \text{ha}^{-1}$  with target basal areas varying based on tree density. Thinning  
425 extended into the riparian management zone. Trees within 15 m of the stream could be chainsaw-  
426 felled and skidded, but mechanical equipment was excluded within 30 m of the stream. Results  
427 showed that drought alone altered dissolved organic carbon (DOC) in stream water, as well as  
428 altered the proportion of dissolved organic carbon to nitrogen (DOC: DON) in soil solution in  
429 unthinned (control) watersheds. Volume-weighted concentration of DOC was 62% lower ( $p <$   
430  $0.01$ ) and DOC:DON was 82% lower ( $p = 0.004$ ) in stream water and soil solution, respectively,  
431 during years of drought than in years prior to drought. Drought combined with thinning altered  
432 DOC and dissolved inorganic nitrogen (DIN) in stream water, and DON and total dissolved  
433 nitrogen (TDN) in soil solution. For stream water, volume-weighted concentrations of DOC were  
434 66- 94% higher in thinned watersheds than in control watersheds for all three consecutive  
435 drought years following thinning. No differences in DOC concentrations were found between  
436 thinned and control watersheds before thinning. The authors conclude that their results provide  
437 evidence that the influences of drought and thinning are more pronounced for DOC than for  
438 nitrogen in streams. They also speculate that the periodic changes in climate (e.g., seasonal,  
439 drought) contribute to the high variability in carbon and nitrogen concentration in streams in  
440 Mediterranean climates following harvest.

441 Specific to Washington, the Hard Rock (McIntyre et al., 2021) and the Soft Rock (Ehinger et al.,  
442 2021) Studies reported on changes in nutrient concentrations and nutrient export in streams  
443 following riparian timber harvest along headwater streams of western Washington. Hard Rock  
444 treatments included a 50 ft buffer along both sides of the stream for the entire RMZ (“100%”),  
445 50 ft buffer along at least 50% of the RMZ (“FP”), clearcut to stream (“0%”), and an unharvested  
446 reference (Ref). Results for nitrogen and phosphorus concentrations in streams showed that post-  
447 harvest changes for total-N or total-P were not significant for any of the treatments relative to the  
448 Reference. The only significant difference detected post-harvest was for nitrate-N concentration  
449 between the 0% buffer treatment and all other treatments. However, for annual export ( $\text{kg ha}^{-1}$

450 yr-1), total-N and nitrate-N export increased post-harvest at all sites, with the smallest increase in  
451 the 100% treatment and the largest in the 0% treatment. Compared to the reference sites, analysis  
452 showed an increase in total-N export of 5.52 (P = 0.051), 11.52 (P = 0.0007), and 17.16 (P  
453 <0.0001) kg ha-1 yr-1 in the 100%, FP, and 0% treatments, respectively, in the first 2 years post-  
454 harvest. In the extended period (7-8 years post-harvest) export for total-N remained higher in all  
455 treatments compared to the reference by 6.20 (P = 0.095), 5.34 (P = 0.147), and 8.49 (P = 0.026)  
456 kg ha-1 yr-1 for the 100%, FP, and 0% treatments, respectively. Nitrate-N showed the same  
457 pattern with slightly lower values than total-N. The increase in total-N and nitrate-N export from  
458 the treatment watersheds post-harvest was strongly correlated with the increase in annual runoff  
459 (R<sup>2</sup> = 0.970 and 0.971; P = 0.001 and 0.001) and with the proportion of the basin harvested (R<sup>2</sup>  
460 = 0.854 and 0.852; P = 0.031 and 0.031). The authors note that there was high variability in the  
461 data for the extended period and nitrate-N export only returned to pre-harvest levels in one  
462 watershed. Total-P export increased post-harvest by a similar magnitude in all treatments: 0.10 (P  
463 = 0.006), 0.13 (P = 0.001), and 0.09 (P = 0.010) kg ha-1 yr-1 in the 100%, FP, and 0% treatments  
464 (only analyzed during the 2-year post-harvest period). The authors conclude that the 100%  
465 treatment was generally the most effective in minimizing changes from pre-harvest conditions,  
466 the FP was intermediate, and the 0% treatment was least effective. Thus, similar to the results of  
467 other studies reviewed, these results provide evidence that the effects of timber harvest on  
468 nutrient export is proportional to the intensity of the treatment (e.g. percent of basin harvested,  
469 presence of protective buffer).

#### 470 *Summary of Factors Impacting Nutrient Concentrations and Export*

471 Similar to instream sediment concentrations and export, there is evidence from the studies  
472 reviewed that nutrient dynamics are affected by the intensity of riparian timber harvest (e.g.,  
473 presence of buffer widths, percent of basin harvested), changes in streamflow (either seasonally  
474 or from harvest), climatic events (e.g., drought, heavy precipitation), physiography (e.g.,  
475 hillslope gradient), and soil disturbance. The Soft Rock study (Ehinger et al., 2021) did analyze  
476 changes in both sediment and nutrient flux following harvest for comparison with the Hard Rock  
477 study. While the authors of this study report that the softer lithologies were more erodible than  
478 the sites sampled for the Hard Rock study and that nutrient flux was within the range of results  
479 for the Hard Rock study, the inability to statistically detect a change in export after harvest was  
480 likely due to the extent of the riparian buffer (58% and 92% buffered), the unusually dry pre-  
481 harvest period, and small number of sites.

482 In contrast to the results for sediment, there is evidence that changes in nutrient flux following  
483 harvest can persist for considerably longer periods. This has been attributed to management  
484 operations such as slash burning, herbicide or fertilizer application that directly affect nutrient  
485 loads, and from decomposition of unburned downed wood and litter (Deval et al., 2021; Shah et  
486 al., 2022). Results showed that instream dissolved organic carbon (DOC) concentrations of un-  
487 thinned stands during drought years were lower, and aromatic DOC was higher than in non-  
488 drought years. In-stream DOC concentrations were higher for three consecutive years following  
489 thinning, than un-thinned stands.

## 490 Large Wood (LW)/wood load/wood recruitment

491 Large wood in streams is essential to create pools, regulate flow, and provide a slow pulse of  
492 nutrients that help create and maintain salmonid habitat (Harmon et al., 1986). Sievers et al.  
493 (2017), in a global meta-analysis of the effects of riparian alteration on trout populations, found  
494 the most positive response of trout populations was with increasing in-stream wood and livestock  
495 exclusion (+87.7% and +66.6%, respectively) from the riparian area. However, while most  
496 studies show a positive relationship between increasing LW and salmonid populations, few have  
497 examined long-term watershed-scale responses of increasing LW or studied a wide range of  
498 species (oni et al. 2014). Large wood production and recruitment into streams can vary between  
499 watersheds, and multiple studies have attempted to identify the drivers of LW production and  
500 recruitment with varying results. For example, Benda et al. (2003) present a wood budgeting  
501 framework, developed from 20 years of LW research based in the Pacific Northwest, for riparian  
502 zones that includes numerical expressions for punctuated forest mortality by important drivers  
503 they identify as fire, chronic mortality and tree fall, bank erosion and mass wasting, decay, and  
504 stream transport. This framework can be applied to different regions by adjusting parameter  
505 values to make predictions of the importance of landscape factors (e.g., climate, topography,  
506 basin size) on wood recruitment and abundance in streams for any area. Depending on the region  
507 or landscape for which the framework is being applied, less common but more locally important  
508 disturbances such as ice storms, ice breakage, and wind throw can also be incorporated. This  
509 study and the framework it developed illustrate the diversity of the wood recruitment, transport,  
510 and decay processes. The relative importance of each wood recruitment mechanism, and the fate  
511 and transport of the in-stream wood depends on the variation observed in the environmental,  
512 management, and vegetation factors of a site. Thus, frameworks such as the one developed by  
513 Benda et al. (2003) help identify the relative importance of these recruitment processes and their  
514 relationship with local landscape factors.

515 A Review of the Available Literature Related to Wood Loading Dynamics in and around Streams  
516 in Eastern Washington Forests, was developed for CMER in October of 2004 (CMER 03-308,  
517 2004). In this review, the researchers sourced 14 references with quantitative and descriptive  
518 information relating to the correlation between wood volume and pieces of wood in streams and  
519 the adjacent riparian community. The authors conclude that while the literature was incomplete,  
520 several significant correlations existed between LW in streams and riparian zone stand  
521 characteristics. For unmanaged (defined as unlogged and un-roaded) sites in Washington,  
522 researchers reported positive correlations between the volume of LW in streams with adjacent  
523 riparian zone mean tree height ( $P < 0.001$ ), mean tree diameter ( $P < 0.001$ ), and mean basal area  
524 ( $P < 0.001$ ). For numbers of LW pieces, positive correlations were found with the basal area  
525 ( $P < 0.007$ ) but no other vegetation characteristic of the adjacent riparian area. However,  
526 regression analysis showed a significant positive correlation of LW piece quantity with core zone  
527 trees/acre ( $P < 0.001$ ,  $R^2 = 0.45$ ) and core zone basal area/acre ( $p = 0.004$ ,  $R^2 = 0.29$ ). Relative to  
528 managed riparian areas, streams adjacent to unmanaged riparian areas had significantly higher  
529 LW volume. The most relevant sources of these results listed in this review were from Fox  
530 (2001), Chesney (2000), Camp et al. (1997), and Knight (1990). Two other studies named in this  
531 review (McDade et al., 1990; Fox, 2003) show evidence that as much as half of the wood found

532 in the streams could not be attributed to the adjacent designated riparian areas which indicates  
533 the importance of scale when investigating in stream LW source.

534 In the western United States, several notable studies since 2000 have continued to investigate  
535 and refine the factors important for LW recruitment. For example, Wing & Skaugset (2002)  
536 investigated the relationships between land use, land ownership, and channel and habitat  
537 characteristics with LW quantity and volume in stream reaches in western Oregon. The relevant  
538 results (those derived for forested streams only) showed that stream gradient was the most  
539 important explanatory variable for in-stream LW volume with the split in the regression analysis  
540 occurring at 4.7%. Stream reaches with gradients less than 4.7% had on average less than half  
541 the in-stream LW volume (11.3 m<sup>3</sup> vs. 25.2 m<sup>3</sup> per reach) than reaches with gradients >4.7%.  
542 Results for LW pieces (logs at least 0.15 m diameter, and 3 m long) per 100 m length showed  
543 bankfull width (BFW) as the most important explanatory variable with a split in the regression  
544 analysis occurring at 12.2 m BFW. Reaches with a BFW <12.2 m averaged 11.1 LW pieces per  
545 100 m compared to wider streams which averaged 4.9 pieces per 100 m. When the analysis was  
546 constrained to “key” LW pieces (logs at least 0.6 m diameter and 10 m long), stream gradient  
547 again emerged as the most important explanatory variable with the split in the regression  
548 occurring at 4.9% stream gradient (mean key pieces per 100 m were 0.5 and 0.9 for gradients <,  
549 and >4.9%, respectively). Following stream gradient and BFW, lithology was also an important  
550 explanatory variable showing splits for Mesozoic and sedimentary lithologies (in 3 out of 4  
551 analyses) grouped as containing half the LW quantity (pieces, key pieces, volume) on average  
552 than all other geologies (basalt, cascade, and marine sedimentary geologies). Wing & Skaugset  
553 (2002) suggests that geomorphic characteristics, in particular stream gradient and bankfull width,  
554 but also underlying lithology in forested areas correlate best with LW presence in headwater  
555 streams of western Oregon.

556 Another study from the Oregon Coast Range, May & Gresswell (2003), compared LW  
557 recruitment processes between small colluvial channels and larger alluvial channels. Results  
558 from this study showed that LW derived from local hillslopes and riparian areas accounted for  
559 the majority of pieces (63%) in small colluvial channels. In contrast, the larger alluvial channel  
560 received wood from a greater variety of sources, including recruitment from local hillslopes and  
561 riparian areas (36%), fluvial redistribution (9%), and debris flow transported wood (33%).  
562 Further, distributions of the source distance of wood pieces were significantly different between  
563 colluvial and alluvial channels. In colluvial streams, 80% of total wood and 80% of total wood  
564 volume recruited to colluvial streams originated from trees rooted within 50 m of the channel. In  
565 the alluvial channel, 80% of the pieces of wood and 50% of the total volume originated from  
566 trees which came from within 30 m of the channel. Considering the mechanisms responsible for  
567 recruitment, for both colluvial and alluvial stream channels, slope instability exhibited the  
568 longest source distance (median source distance = 40 m), followed by windthrow (median source  
569 distance = 20 m), then natural mortality (median source distance = 18 m), and for obvious  
570 reasons, bank erosion had the shortest median source distance (2 m). Compared between channel  
571 types (colluvial vs. alluvial), the median source distance of wood recruited by windthrow was  
572 significantly greater in colluvial channels than in the alluvial channel ( $p < 0.05$ ). Source  
573 distances for all other processes did not differ significantly between channel types. May &

574 Gresswell (2003) interpret these results as evidence that stream size and topographic position  
575 strongly influence processes that recruit and redistribute wood in channels. Processes of slope  
576 instability were shown to be important conveyors of wood from upland forests to small colluvial  
577 channels. In the larger alluvial channels, windthrow was found to be the dominant recruitment  
578 process from adjacent riparian area.

579 Three larger scale studies from Washington (Fox & Bolton, 2007), the northwestern United  
580 States (Sobota et al., 2006), and the Columbia River Basin (Hough-Snee et al., 2016) present  
581 results from simulation modeling or statistical modeling for site and physiographic factors  
582 influencing LW recruitment and in stream loading. Sobota et al. (2006), in a landscape-wide  
583 study of factors affecting tree fall direction and LW recruitment in watersheds of the Pacific  
584 Northwest (data sourced from Washington, Oregon, Idaho, and Montana), found valley  
585 constraint to have the strongest correlation with in-stream woody debris. Outputs from their  
586 model showed that riparian areas in channels with >40% valley side slopes had the highest  
587 tendency for tree fall towards streams; in these steep slope valleys, recruitment of large wood in  
588 streams was 1.5-2.4 times greater than on moderately sloped landforms (< 40%).

589  
590 Fox & Bolton (2007) modeled LW values from 150 stream segments located in unmanaged  
591 watersheds, across Washington, with landscape, reach, and stand characteristics to understand the  
592 central tendency of instream LW values in “natural” fish-bearing streams. Outputs from their  
593 models show evidence that in-stream wood volume (m<sup>3</sup> per 100 m stream length) and LW piece  
594 count for streams up to 20 m in bankfull width (BFW) increased with drainage area and as  
595 streams became less confined with BFW being a significantly better predictor of wood  
596 parameters than basin size. Also, in-stream wood volume increased with adjacent riparian timber  
597 age as determined by the last stand replacing fire. In this study (Fox & Bolton, 2007), the authors  
598 noted that other predictor variables (e.g., gradient, bedform) also showed some evidence of an  
599 effect but the variability of these variables were too great to evaluate with confidence.

600  
601 Hough-Snee et al. (2016) reported similar issues with their results using Random Forest (RF)  
602 models developed from field data to identify relationships between hydrogeomorphic and  
603 ecological attributes that influence instream wood accumulation. Final RF models explained  
604 43.5% of the variance in volume and 42.0% of the variance in frequency of in stream wood  
605 loads. Mean annual precipitation, riparian large tree cover, and watershed area were estimated as  
606 the most important predictors of in stream wood loads. However, so did individual watershed  
607 which showed there was an interaction with site (i.e., site conditions unaccounted for may be  
608 affecting the response). Given the heterogeneous results across all sub-basins studied, the authors  
609 conclude by emphasizing the importance of incorporating local data and context when building  
610 wood models to inform future management decisions.

611  
612 Multiple studies have also investigated the effects of timber harvest under varying riparian  
613 management zone prescriptions on LW recruitment. Specific to Washington, Schuett-Hames and  
614 Stewart (2019a) compared in stand structure, tree fall rates, and LW recruitment between riparian  
615 management zones harvested under the current standard Shade Rules (SR), the All-Available  
616 Shade Rule (AAS), and unharvested references for fish-bearing streams in the mixed conifer

617 habitat type (2500 - 5000 feet elevation) for eastern Washington. Both shade rules have a 30-ft  
618 no-cut buffer (core zone) immediately adjacent to the stream. The SR prescription allows  
619 thinning in the buffer zone 30-75 feet (inner zone) from the stream while the AAS prescription  
620 requires retention of all trees providing shade in this area. Results showed that cumulative wood  
621 recruitment from tree fall after the five-year post-harvest interval was highest in the SR group,  
622 lower in the AAS group and lowest in the REF group. The SR and AAS LW recruitment rates by  
623 volume were nearly 300% and 50% higher than the REF rates, respectively. Wood recruitment in  
624 the SR sites was significantly greater than in the AAS and reference sites. Conversely, wood  
625 recruitment did not differ significantly between the AAS and reference sites. Considering the  
626 source distance of post-harvest recruited LW, most recruited fallen trees originated in the core  
627 zone (76%, 72%, and 64% for the REF, AAS and SR groups, respectively), while the proportion  
628 from the inner zone (30–75 feet from the stream) was ~10% greater for the SR group compared  
629 to the AAS and REF groups. These results suggest that while treatment of SR sites is intended to  
630 increase resistance to disturbances such as fire and disease, it also provides evidence that these  
631 treatments increase the susceptibility to windthrow and thus increases mortality relative to  
632 reference sites five years post-harvest. Further, thinning treatments in the inner zone appeared to  
633 change the spatial pattern (source distance) of wood recruitment from fallen trees. It is important  
634 to note that this was a short-term study (5 years). The authors remark that LW recruitment is a  
635 process that can change over decadal time scales, and follow-up monitoring is recommended.

636 Four similar studies conducted for non-fish-bearing streams in western Washington compared  
637 changes in LW recruitment and stand mortality following harvest (Ehinger et al., 2021; McIntyre  
638 et al., 2021; Schuett-Hames et al., 2011; Schuett-Hames et al., 2019b. Schuett-Hames et al.,  
639 (2011) and Schuett-Hames & Stewart (2019b) investigated changes in riparian stand mortality  
640 and LW recruitment into the bankfull channel 5- and 10-years post-harvest, respectively.  
641 Treatments for riparian forests adjacent to non-fish-bearing streams evaluated in these studies  
642 include clearcut to stream edge, upland clearcut with a 50-foot no cut buffer, and these were  
643 compared to unharvested reference streams. Results showed that tree fall rates (annual fall rates  
644 of live and dead standing stems combined) was over 8 times and 5 times higher in the 50-foot  
645 buffers than in the reference buffers 3 years after treatment when compared as a percentage of  
646 standing trees and as trees/acre/yr, respectively. These differences were significant for both  
647 metrics ( $p \leq 0.001$ ). Total tree-fall rates in the period 4-5 years after treatment, while still higher  
648 in the 50-foot buffers was not significant.

649 Over the entire five-year period, the percentages of standing trees that were uprooted and broken  
650 (as well as the combined total) were significantly greater in the 50-foot buffer than in the  
651 reference. Differences in mortality followed a similar pattern to tree fall rates. In the 50-foot  
652 buffer sites, mortality rates were significantly higher (3.5 times higher) than in the reference sites  
653 for the first three years following harvest. However, in years 4-5 mortality rates increased in the  
654 reference buffers after high-intensity storms resulting in non-significant differences in mortality  
655 during this period. The cumulative percentage of live trees that died over the entire five-year  
656 period was 27.3% in the 50-ft buffers compared to 13.6% in the reference reaches, but the  
657 difference was not statistically significant. This was likely because of the high variability in  
658 mortality between sites in the 50-foot buffers. The data for mortality rates in the 50-foot buffers

659 had a bimodal distribution with most sites exhibiting less than 30% mortality, although three  
660 sites (of 13) exhibited mortality rates greater than 50%.

661 For LW recruitment into the bankfull channel, results showed during the first three years after  
662 treatment recruitment rates were 8 times and 14 times higher in the 50-foot buffers than in the  
663 reference buffers respectively. The differences in pieces/acre/year and volume/acre/year between  
664 reference and 50-foot buffers were significant. In years 4-5 after harvest LW recruitment  
665 decreased in the 50-ft buffers and increased in the reference patches, and the number of recruited  
666 LW pieces/acre/yr was greater in the reference patches, although the volume of LW recruited was  
667 greater in the 50-ft buffers. Differences in recruitment rates between the 50-foot buffer and the  
668 reference buffers for the 4–5-year period were not significant. For the entire first 5 years after  
669 harvest, the 50-ft buffers recruited about twice the number of LW pieces recruited in the  
670 reference patches, and over 3 times the volume; differences were marginally significant.

671 The results of the 10-year follow-up study for these sites (Schuett-Hames & Stewart, 2019b)  
672 showed that stand mortality in the 50-foot buffer sites had stabilized and showed a cumulative  
673 14.1% reduction in live basal area, while the reference stands showed a 2.7% increase in live  
674 basal area. The differences in these values were not significant. Cumulative LW recruited into the  
675 stream channel over the 10-period was double in the 50-ft treatment streams compared to the  
676 reference streams. However, the majority of the LW recruited in the 50-ft treatment streams came  
677 to rest above the streams, providing shade but not affecting streamflow, pool formation, or  
678 sediment storage. Further, while the 50-ft buffer treatment provided more LW recruitment in the  
679 short-term (10-years), the authors speculate there is a reduction in future LW recruitment  
680 potential given the removal of trees outside the 50-ft buffer.

681 Two other studies which evaluated changes in LW following riparian forest harvest along non-  
682 fish-bearing streams in western Washington were complimentary studies. Treatment sites in these  
683 studies were underlain by either competent (McIntyre et al., 2018, 2021; also referred to the  
684 “Hard Rock” study), or incompetent (easily eroded) marine sedimentary lithologies (Ehinger et  
685 al., 2021; also referred to as the “Soft Rock” study). The buffer treatments evaluated for these  
686 studies were compared against unharvested reference sites ("REF") and included a minimum  
687 two-sided 50-ft wide riparian buffer along the entire reach (“100%”) and the standard Forest  
688 Practices treatment (FP), a minimum two-sided 50-ft wide riparian buffer along at least 50% of  
689 the RMZ (buffered and unbuffered portions were analyzed separately; hereafter referred to as  
690 FPB for the buffered portion, and 0% for the unbuffered portion). However, because of unstable  
691 slopes in some of the sites in the Soft Rock study (Ehinger et al., 2021), many of the buffers were  
692 required to be wider than 50-feet (ranging from 18 –160% wider than 50-feet). Conversely, some  
693 of the sites had additional buffers that were narrower than 50 feet. Further, there was limited  
694 availability of sites that fit the criteria (marine sediment lithology, timing of treatment). Because  
695 of these limitations, statistical analysis and comparison of LW response between treatments and  
696 references was not performed as a part of the Soft Rock study. Thus, the results are only  
697 descriptive, but they provide an informative comparison to the Hard Rock study.

698 Results from the Soft Rock study showed mean cumulative post-harvest mortality during the 3-  
699 year post-harvest interval was only 6.5% of live density (trees/ha) in the reference sites. In



700 contrast, mean post-harvest mortality in the full buffer sites and the <50 ft buffer sites were 31  
701 and 25% of density, respectively. However, there was considerable variation in mortality among  
702 sites, exceeding 65% in two full buffer treatment sites. Windthrow and physical damage from  
703 falling trees accounted for ~75% of mortality in the full and <50 ft buffers. In contrast to the  
704 treated sites, <10% of trees died due to wind or physical damage in the reference sites. For LW  
705 recruitment, there was an increase in pieces of LW per 100 m length of stream in the full buffers  
706 (8%) and the unbuffered treatments (13%) and a decrease in the streams adjacent to buffers < 50  
707 feet wide (-15%) 3 years after harvest. The Hard Rock study did not require changes to the  
708 grouping of treatments (i.e., all treatment buffers were harvested as described above; e.g.,  
709 Reference, 100%, FPB, 0%). Also, the Hard Rock study collected up to 9 years of post-harvest  
710 data that allowed for the comparison of LW changes over time pre- to post-harvest, and between  
711 treatments.

712 Results for the Hard Rock study showed that by year 8 post-harvest mortality as a percentage of  
713 pre-harvest basal area was lower in the reference (16.1%) than in the 100% (24.3%) and FPB  
714 (50.8%) treatments. The FPB–Reference contrast in mortality was not significant 2 years post-  
715 harvest, but it was at 5- and 8-years post-harvest as mortality in FPB increased relative to the  
716 Reference over time. The contrast in mortality between the 100% and Reference were not  
717 significant for any time interval 8 years post-harvest. Wind/physical damage was the primary  
718 cause of mortality for all treatments, including the Reference. In the 100% treatment it accounted  
719 for 78% and 90% of the loss of basal area and density (trees/ha), respectively; in FPB it  
720 accounted for 78% and 65% of the loss. Wind accounted for a smaller proportion of mortality in  
721 the Reference RMZ (52% and 43%, respectively). Large wood recruitment to the channel was  
722 greater in the 100% and FPB RMZs than in the reference for each pre- to post-harvest time  
723 interval. Eight years post-harvest mean recruitment of large wood volume was two to nearly  
724 three times greater in 100% and FPB RMZs than in the references. Annual LW recruitment rates  
725 were greatest during the first two years, then decreased. However, there was a great deal of  
726 variability in recruitment rates within treatment sites and the differences between treatments  
727 were not significant. Mean LW loading into the channel (pieces/m of channel length) differed  
728 significantly between treatments in the magnitude of change over time. There was a 66%, 44%  
729 and 47% increase in mean large wood density in the 100%, FP and 0% treatments, respectively,  
730 in the first 2 years post-harvest compared with the pre-harvest period and after controlling for  
731 temporal changes in the references. By year 8, only the FP treatment showed a significantly  
732 higher proportional increase (41%) in wood loading when compared to the reference. In the time  
733 interval 2-8 years post-harvest wood loading in the 100% treatment stabilized and began to  
734 decrease in the 0% treatment.

735 The Hard Rock and Soft Rock studies showed similar results. Both studies showed an increase in  
736 stand mortality that resulted in an increase in LW recruitment into the channels adjacent to  
737 riparian buffers relative to unharvested reference sites. However, the longer period of study in the  
738 Hard Rock study demonstrated that both mortality and LW recruitment began to stabilize five  
739 years post-harvest. The results presented by Schuett-Hames (2012, 2019b) showed a similar  
740 pattern of an initial increase in mortality rates and LW recruitment rates in treated stands relative  
741 to untreated stands within three years of treatment, but stabilization within 5-10 years.

742 Unfortunately, because of the limitations in sample size and buffer width consistency in the Soft  
743 Rock study, confident conclusions on the effects of lithological competency on LW recruitment  
744 post-harvest cannot be drawn.

745 All studies reviewed above, which investigate the effect of timber harvest with riparian buffers  
746 on tree mortality and large wood recruitment, show that the initial increase in mortality within  
747 treatment buffers relative to reference buffers is primarily a result of increased windthrow  
748 mortality. Liquori (2006) found similar results in an investigation of treefall characteristics  
749 within riparian buffer sites ranging in width from 25-100 feet along non-fish-bearing and fish-  
750 bearing streams. Within no-cut buffers, windthrow caused mortality was up to 3 times greater  
751 than competition induced mortality for 3 years following treatment with tree fall probability  
752 highest in the outer areas (closest to upland clearcuts) of the buffers. Their results showed that  
753 treefall was generally highest at the outside edges of buffers (50+ feet), representing about 60%  
754 of the total observed treefall, while the 0–25-foot zone represented ~18%, and the 25–50-foot  
755 zone represented ~22%. This suggests an increase in windthrow susceptibility within riparian  
756 buffers with increasing distance from the stream. Liquori (2006), however, did not differentiate  
757 thinning treatments applied to the outer zones of the buffer in their analysis mentioning “very  
758 modest” thinning was applied to some buffers. They suggest in their interpretation of the results  
759 that buffer thinning may influence the depth to which wind forces can penetrate into the buffer.  
760 The results from Schuett-Hames & Stewart (2019a), discussed above, show evidence that  
761 thinning in the outer area (30-75 feet from bankfull width) changed the source distance curve of  
762 wood recruitment from fallen trees with thinned buffers (SR treatments). The results exhibited  
763 statistically higher overall treefall rates with a larger percentage coming from the outer area in  
764 the SR treatments than in the reference and more lightly thinned (AAS) treatment buffers.

765 Outside of Washington, but in areas with similar habitats (Oregon, British Columbia) several  
766 experimental studies that have investigated the effects of timber harvest on treefall, mortality,  
767 LW recruitment, and LW source distance have found comparable results to those conducted in  
768 Washington. For example, Martin & Grotefendt (2007) compared riparian stand mortality and in-  
769 stream LW recruitment characteristics between riparian buffer strips with upland timber harvest  
770 and riparian stands of unharvested watersheds using aerial photography in the northern and  
771 southern portions of Southeast Alaska. All buffer strips in this study were a minimum of 20 m  
772 wide and included selective harvest within the 20 m zone (thinning intensity not specified or  
773 included in the analyses as an effect). The results from this study showed significantly higher  
774 mortality (based on cumulative stand mortality: downed tree counts divided by standing tree  
775 counts + downed tree counts), significantly lower stand density (269 trees/ha in buffer units and  
776 328 trees/ha in reference units), and a significantly higher proportion of LW recruitment from the  
777 buffer zones of the treatment sites than in the reference sites. Also, results showed that mortality  
778 varied with distance to the stream. Differences in mortality for the treatment sites were similar to  
779 the reference sites for the first 0-10 m from the stream (only a 22% increase in the treated sites).  
780 However, mortality in the outer half of the buffers (10-20 m) from the stream in the treatment  
781 sites was more than double (120% increase) what was observed in the reference sites. The  
782 authors attribute the difference in cumulative stand mortality to the increase in windthrow

783 susceptibility. Mortality attributed to windthrow was twofold and fivefold greater in the inner  
784 and outer halves of the treatment buffers than in the reference buffers, respectively.

785

786 Bahuguna et al. (2010) evaluated the difference in windthrow caused mortality between 10 m, 30  
787 m buffer widths (neither had thinning within the buffer and both had upland clear-cuts) and  
788 unharvested controls in the Coast Mountains, British Columbia. Following harvest, 11% of  
789 initially standing timber was blown down in the first and second years in the 10 m buffer,  
790 compared to 4% in the 30 m buffer, and 1% in the unharvested controls. However, after 8 years  
791 post-harvest, a significant amount of annual mortality occurred when winter storms brought  
792 down multiple trees in the unharvested control at 30%, compared to 15% in both 30 m and 10 m  
793 buffers. These results show evidence that timber harvest can increase windthrow caused  
794 mortality within protective buffers in the short term but can stabilize within a decade. Further,  
795 this study shows evidence that windthrow caused mortality is stochastic and large storm events  
796 can cause just as much if not higher mortality within untreated riparian forests.

797 Burton et al. (2016) examined the relationship between annual in-stream wood loading and  
798 riparian buffer widths adjacent to upland thinning operations. No-cut buffer widths were 6, 15, or  
799 70 meters, and upland thinning was to 200 trees per ha (tph), with a second thinning (~10 years  
800 later) to ~85 tph, alongside an unthinned reference stand ~400 tph. Their results showed that  
801 slightly higher volumes of wood were found in sites with a narrow 6-m buffer, as compared with  
802 the 15-m and 70-m buffer sites in the first 5 years after the first harvest and maintained through  
803 year 1 of the second harvest (end of study). The authors attributed this difference to a higher  
804 likelihood of logging debris and/or windthrow, but these factors were not analyzed. Considering  
805 source distance, the authors used a mixed modeling approach to assess the relationship between  
806 wood volume and source distance for in-stream wood with an identifiable source. This model  
807 was only applied to the 70-meter buffer. The results showed that 82-85% of the wood with  
808 discernable sources (90% for wood in early stages of decay; 45% of wood in late stages of  
809 decay) came from within 15 m of the stream, and the relative contribution of wood to streams  
810 declined rapidly with increasing distance. Still, these results are similar to those presented by  
811 Schuett-Hames & Stewart (2019a) which showed the majority of the LW recruited (72-76% for  
812 treated stands) into the channel were from within the first 30 feet (~9.1 m) of the stream even  
813 though upland harvest prescriptions in this study differed from those evaluated by Burton et al.  
814 (2016) (e.g., clearcut vs thinning).

815

### 816 *Summary of Factors Impacting LW Loads and Recruitment*

817 In general, the studies reviewed above show evidence that upland timber harvest with riparian  
818 retention buffers initially increases stand mortality within the buffers and increases LW  
819 recruitment relative to unharvested reference stands in the short-term. This increase in mortality  
820 and LW recruitment is attributed to an increase in the susceptibility to windthrow within the  
821 riparian buffers relative to the unharvested controls. Further, multiple studies (Liquori, 2006;  
822 Martin & Grotefendt, 2007, Schuett-Hames & Stewart 2019a) showed evidence that the increase

823 in windthrow caused mortality is highest in the outer area of the riparian buffers (area closest to  
824 upland treatments). There is some evidence that thinning within the buffer can also affect  
825 mortality rates, but these studies are few. In the three studies that collected post-harvest data for 8  
826 or more years (Bahuguna et al., 2010; McIntyre et al., 2021; Schuett-Hames & Stewart 2019b),  
827 there is indication that mortality in the riparian buffers and annual LW recruitment into adjacent  
828 streams stabilizes within 5-10 years. However, in the subsequent decades following treatments  
829 with upland clearcuts there is evidence that LW recruitment rates can continue to decrease and in  
830 stream wood loads may become depleted before recruitment rates can recover (Nowakowski &  
831 Wohl, 2008; Reid & Hassan, 2020) depending on applied management practices (e.g., buffer  
832 widths, road construction, etc.). For example, Teply et al. (2007) used simulation modeling to  
833 estimate the effectiveness of Idaho Forest Practices for riparian buffers and found no significant  
834 difference between predicted LW loads for harvested and unharvested sites 30-, 60-, or 100-years  
835 post-harvest.

836 While the general conclusions of short-term increase in LW and long-term reduction of LW  
837 following treatment are similar among studies it is more apparent that LW recruitment dynamics  
838 are complex and highly variable even within treatment groups; and local site and landscape  
839 factors may interact with treatments making it difficult to generalize the effectiveness of different  
840 protective buffer treatments on preserving LW recruitment and in-stream wood loads. Indeed, the  
841 LW budget framework created by Benda et al. (2003) emphasizes the importance of including  
842 local physiographic, site, and disturbance factors. Additionally, the studies reviewed above  
843 present results from experimental studies that vary greatly in their design. Buffer widths, riparian  
844 and upland treatment prescriptions differ by region, state, and local regulations that can differ  
845 further by stream type and size, and location within the landscape (e.g., elevation). Thus, general  
846 global conclusions about the effect of riparian forest treatment on LW dynamics are difficult to  
847 discern.

## 848 Bank Stability and Sediment

### 849 *Bank Stability*

850 Few studies could be found that quantify how riparian area harvest directly affects bank stability  
851 or bank erosion based on our search criteria. Many studies published since 1999 that investigate  
852 bank stability and bank erosion compare relative rates of erosion based on the presence/absence  
853 of vegetation, type of vegetation (e.g., grassland vs. forest cover), and soil types or lithology  
854 (Konsoer et al., 2015; Micheli et al., 2004; Simon & Collision, 2001; Wynn & Mostaghimi,  
855 2006). Also, many studies have investigated the relative effects of different types of land use  
856 (e.g., agricultural, urban, forested) as well as cattle grazing intensity (McInnis & McIver, 2009;  
857 Zaimes & Schultz, 2014). The only studies that could be found that provide some experimental  
858 evidence as to how timber harvest within the riparian area affects bank stability or erosion come  
859 from three CMER reports (Ehinger et al. 2021; McIntyre et al. 2018, Schuett-Hames et al., 2011;  
860 Schuett-Hames & Stewart, 2019).

861 Schuett-Hames et al. (2011) investigated how soils and streambanks were disturbed following  
862 harvest within the riparian area along perennial non-fish-bearing streams (Type Np) in western  
863 Washington. To evaluate post-harvest soil and stream bank disturbance, Schuett-Hames et al.  
864 (2011) first described a soil erosion feature as areas of exposed soil that (1) had a surface area of  
865 greater than 10 square feet, and (2) was caused by harvest practice (e.g., felling, bucking, or  
866 yarding). If both criteria were met, the length, width, and distance to stream were recorded, and  
867 evidence of sediment delivery to the stream was noted. The number of harvest related soil  
868 disturbances were grouped by 100 ft lengths of stream, as were the number of features delivering  
869 sediment to the stream. Disturbances along stream bank were quantified using the same methods.  
870 The surface area (mean width x length) of disturbance features were used to estimate the percent  
871 coverage of soil disturbance within 50-feet of bankfull width and in the equipment exclusion  
872 zone (ELZ; within 30 feet of the bankfull width). Finally, the percent of harvested patches with a  
873 greater than 10% coverage of soil disturbance features in the ELZ were also quantified  
874 (performance target for bank stability). These methods were used to collect data for all 3 harvest  
875 treatments. These harvest treatments included 1) a 50-foot wide no cut buffer, 2) clearcut, no  
876 buffer, and 3) a 56-foot radius no-cut buffer surrounding the perennial initiation point (PIP). A  
877 non-parametric, two-sample Mann-Whitney U test was used to test differences in mean soil and  
878 stream bank disturbance metrics between the 50-foot buffer patches and the clearcut (no buffer)  
879 patches. A Fisher's exact test was used to test for differences in the relative frequency of patches  
880 exceeding the performance target (more than 10% of ELZ area disturbed by management related  
881 activities) between 50-foot and the clearcut buffer prescriptions.

882 Results showed that the differences between the mean values of harvest related soil and  
883 streambank disturbances for clear-cut patches and the 50-ft buffers were significant for all  
884 metrics (e.g., # of bank disturbance features per 100 ft, # of soil disturbance features per 100 feet,  
885 # of soil disturbance features, # of soil disturbance features delivering sediment to stream, % of  
886 ELZ with soil disturbance;  $P \leq 0.082$ ). Results for soil disturbance performance targets showed  
887 that all of the 50-foot buffer and PIP prescriptions met the performance targets (i.e., maintained  
888 <10% harvest-related soil disturbance in the ELZ). One clearcut patch exceeded the 10%

889 coverage performance target. The difference between clearcut patches and 50-foot buffer patches  
890 was significant ( $p = 0.007$ ). The average size of harvest related soil disturbances that delivered  
891 sediment to streams was  $752 \text{ ft}^2$  (range:  $31\text{-}9060 \text{ ft}^2$ ). The average size of soil disturbance  
892 features that did not deliver sediment to streams was  $65 \text{ ft}^2$  (range:  $13 - 214 \text{ ft}^2$ ). Delivery of  
893 sediment to streams was best predicted by the horizontal distance between the soil disturbance  
894 and the stream channel ( $P < 0.0001$ ). The average distance to the stream for soil disturbance  
895 features that delivered sediment was 1 ft (max. = 7.7), while the average distance for non-  
896 delivering soil disturbance features was 14 ft (min 3.3). Using distance-to-stream alone, 96% of  
897 the observations were correctly predicted based on whether the horizontal distance to the stream  
898 was greater or less than 5.4 ft ( $R^2 U4 = 0.80$ ). The authors concluded there were more harvest-  
899 related soil disturbances following harvest in the clear-cut patches than the 50-ft buffers. Further,  
900 that the management practices for the 50-foot and PIP buffers were sufficient at maintaining  
901 bank stability performance targets. The clearcut patches were mostly sufficient at maintaining  
902 performance targets with the exception of one site.

903 Schuett-Hames et al. (2011) also collected data on soil disturbance associated with post-harvest  
904 root pits created from trees being uprooted by wind or other disturbances. Four metrics were  
905 used to evaluate soil disturbance associated with uprooted trees: *Root-pits per acre*. Root-  
906 pits/acre was calculated by tallying the number of root-pits in each patch and dividing by the  
907 patch acreage. *Root-pits per 100 ft of stream length*. Root-pits/100 ft of stream length was  
908 calculated by tallying the number of root-pits in each patch (both sides of the stream), dividing  
909 by the stream length, and multiplying by 100. *Root-pits with sediment delivery per acre*. Root-  
910 pits/acre with evidence of sediment delivery to the channel was calculated by tallying the number  
911 of root-pits where evidence of sediment delivery to the stream channel is observed in each patch  
912 and dividing by the patch acreage. *Root-pits with sediment delivery per 100 ft of stream length*.  
913 Root-pits with sediment delivery/100 ft of stream length were calculated by tallying the number  
914 of root-pits with evidence of sediment delivery in each patch (both sides of the stream), dividing  
915 by the stream length, and multiplying by 100. These metrics were measured 3 years and 5 years  
916 following harvest to give an annual rate of change for each metric at 3 years, from 3-5 years, and  
917 for the entire 5 years. These standardized annual rates were compared between each treatment  
918 patch type and a unharvested reference patch of the same size.

919 Results showed that in the first three years after harvest, the mean annual rate of total root-pit  
920 formation (all root-pits) in the 50-ft buffers was over 10 times higher than the reference rate. This  
921 difference was significant ( $p = 0.002$ ). A similar result was found in the difference between root  
922 pits delivering sediment to streams ( $p = 0.002$ ). The mean total root-pit formation rate in the  
923 clear-cut patches was much lower than the reference rate (likely because there were less trees to  
924 topple). This difference was significant ( $P \leq 0.001$ ). During the second time period (years 4-5  
925 after harvest) the greatest change in the root-pit formation rates was a large increase in the rate  
926 for the reference patches and a decrease in rates for the 50-ft buffers. The difference in rates  
927 between the reference and the 50-foot buffer were not significant for this time period. The clear-  
928 cut patches continued to have the lowest rate and were still significantly lower than the reference  
929 patches ( $P \leq 0.001$ ). Over the entire first five years, the rate of total root-pit formation for the 50-  
930 ft buffers was nearly double the reference rate, however, this difference was not significant. The

931 pattern was similar for root-pits with sediment delivery, however the difference between the  
932 reference and buffer patches was less pronounced due to the higher percentage of root-pits  
933 delivering sediment in the reference patches. The percentage of root-pits with evidence of  
934 sediment delivery was much higher in the clear-cut patches than in the 50-ft buffers (20.1%) and  
935 the reference (26.0%) patches but was not significantly different. Results for the PIP buffers  
936 showed a similar trend as the 50-foot buffers with an increase in root pits delivering sediment to  
937 the stream in the first three years, but a sharp decline after the third year. Over the course of the  
938 full five years Over the entire 5 year period, the percentage of root-pits with evidence of  
939 sediment delivery in the PIP buffers (17.6%) was similar to the percentage for the 50-ft buffers  
940 (19.8%). These values did not differ significantly from the references.

941 The authors also investigated the factors affecting whether the post-harvest root pits delivered  
942 sediment to streams for 2006 and 2008 (3 and 5 years post-harvest). In both years, sediment  
943 delivery to streams was best predicted by the distance of the root-pit from the stream ( $P <$   
944  $0.0001$ ). Mean horizontal distance to the stream for root-pits that delivered sediment was 8.2 ft  
945 compared to 28.0 ft for those that did not deliver. Using horizontal distance to stream, the  
946 proportion of the total uncertainty that was attributed to the model fit was 0.39, and 80% of the  
947 observations were correctly predicted based on whether the horizontal distance to stream was  
948 greater or less than 12.5 ft. Width of root pits delivering soil to the stream were also larger on  
949 average but its inclusion to the model did not increase fitness. The authors speculate from their  
950 observations that the higher tree-fall rates in the 50-foot buffer during the first 3 years after  
951 harvest was due to an increase in wind-throw. However, in the second time period the reference  
952 patches showed an increase in windthrow following stronger storms during the 2006-2008  
953 period. One of the two reference streams did show string evidence of mass wasting.

954 Ehinger et al. (2021; Soft Rock Study) in their investigation of sediment export following harvest  
955 along Type Np streams in western Washington (same prescriptions as described above for  
956 Schuett-Hames, 2011) also quantified bank erosion events to assess sediment source. To assess  
957 erosion events, the researchers placed two eye screws outside of the bank full width to attach a  
958 reel tape for measuring length and depth across the bank. No evidence of bank erosion events  
959 were found during the pre-harvest periods (1-2 years depending on site) for any stream reach. No  
960 erosion events were found at any of the treatment sites during the post-harvest period (3-4 years  
961 depending on site). However, there were observations of sediment being sourced from root-pits  
962 developed in 2 treatment sites during the post-harvest period, but these effects were not  
963 statistically analyzed. Because of the large mass wasting event in the reference the data collected  
964 does not support any strong conclusion about the effect of riparian timber harvest on bank  
965 stability.

966 McIntyre et al. (2018; Hard Rock Study) also investigated post-harvest surface erosion following  
967 harvest along Type Np streams (including the FP treatment as investigated by Schuett-Hames,  
968 2011) on competent lithologies in western Washington. They conducted visual surveys to  
969 identify recently eroded areas (source of erosion not discerned) in the treated riparian areas that  
970 were 10 m<sup>2</sup> or larger. Post-harvest stream-delivering surface erosion was documented at 11 of 17  
971 sites observed. The total erosion area exceeded 110 m<sup>2</sup> at 5 of the 17 sites: 2 reference sites, 2

972 50-foot buffer sites, and 1 clearcut sites. At these five sites, post-harvest surface erosion was  
973 evident adjacent to only 1.5 to 4.6% (average = 2.2%) of the total stream channel length  
974 (including both mainstem and tributaries). At the remaining study sites where stream-delivering  
975 erosion events occurred, the total eroded area was 60 m<sup>2</sup> or less and occurred adjacent to 0.3% to  
976 0.8% (average = 0.6%) of the stream channel length. There were no statistically significant  
977 differences in stream-delivering surface erosion among treatments ( $\alpha = 0.05$ ), and on average,  
978 reference and buffer treatments visually exhibited a similar amount of exposed bank.

979 The researchers also investigated the frequency of uprooted trees that developed root pits during  
980 the post-harvest period. The average rate of root pits developed in the 50-foot buffers was  
981 approximately 3 times higher (3.6 pits/ha/yr) than in the reference sites (1.2 pits/ha/yr) for 3  
982 years following harvest. However, year to year values were highly variable with reference sites  
983 showing higher numbers of root pits per acre than either buffer treatment in the first year  
984 following treatment (27.4 vs. 18.5 vs. 6.4 for reference, 50-foot, and clearcuts respectively).

985 The results of the above studies on bank and riparian surface erosion after harvest show some  
986 evidence that bank erosion and soil disturbance is generally higher in treated areas than in  
987 untreated areas. Further, that bank erosion is likely higher in clearcut treatments without buffers  
988 than in treatments with no-cut buffers. However, development of root-pits (with and without  
989 sediment delivery pathways to streams) are more likely in treatments with no-cut buffers which  
990 is likely because no trees were left in the clearcuts to be toppled. When compared to a reference,  
991 the trends of surface erosion and soil disturbance shows there is generally an increase in the  
992 treated buffers within the first few years. However, these differences appear to stabilize within  
993 five years. Finally, soil disturbance and bank erosion (especially when caused by windthrow) are  
994 highly variable and in many instances do not exceed the natural range of variability found in  
995 reference streams (e.g., Ehinger et al. 2021; McIntyre et al. 2018).

#### 996 Sediment

997 The function of riparian areas to regulate and filter the flow of sediments into streams is essential  
998 not only for water clarity and pool formation but also because of the ability of sediments to carry  
999 nutrients and pollutants (Cooper et al., 1987; Hoffman et al., 2009; Polyakov et al., 2005). .  
1000 Sediment flux into streams can be affected by landscape factors, streamflow, vegetation  
1001 composition, and disturbance including riparian and adjacent upland forest management  
1002 (Crandall et al., 2021; Devotta et al., 2021; Vanderbilt et al., 2003). The movement of sediment  
1003 into the active channel can, in turn, impact aquatic habitat and geomorphic processes, especially  
1004 in small, forested streams (Benda et al. 2005; Gomi et al., 2005; Hassan et al., 2005).

1005 The effects of riparian area timber harvest on sediment flux into streams has been documented,  
1006 investigated, and incorporated into riparian forest management plans in western North America  
1007 since the 1970s with the development of the Clean Water Act of 1972 (Bilby et al., 1989;  
1008 Gregory 1990; Gresswell et al., 1989; Naiman et al., 1998; Salo & Cundy, 1986; Swanson et al.,  
1009 1982; Swanson & Dyrness, 1975). Prior to the Forests and Fish Report (FFR 1999), several  
1010 studies from western North America investigated the effects of riparian zone timber harvest  
1011 practices on sediment flux into streams.



1012 Specific to Washington, Rashin et al. (2006) evaluated the effectiveness of Washington State best  
1013 management practices (BMPs) for controlling sediment related water quality impacts. Although  
1014 this study was published in 2006, the data analyzed in this study were collected between 1992  
1015 and 1995. In their evaluation, Rashin et al. (2006) assessed site erosion, sediment delivery,  
1016 channel disturbance, and aquatic habitat condition within the first two years of harvest along  
1017 fish- and non-fish-bearing streams across Washington state. From their results, the authors  
1018 concluded that the site-specific factors influencing the effectiveness of BMPs in preventing  
1019 chronic sediment delivery into streams were 1) the proximity of ground disturbance to the  
1020 stream, 2) presence of a stream buffer, 3) falling and yarding practices that minimized  
1021 disturbance to stream channel, and 4) timing of harvest activities for certain climate zones where  
1022 frozen ground or snow cover may be exploited. The landscape factors that influenced BMP  
1023 effectiveness were 1) the density (specific metric not reported) of unbuffered small streams at  
1024 harvest sites, and 2) steepness of stream valley slopes. The authors conclude with a  
1025 recommendation of excluding timber falling and yarding activities at least 10 m from streams  
1026 and outside of steep inner gorges.

1027 Similar results were reported by Lewis (1998) in their evaluation of logging activities' effect on  
1028 erosion and suspended sediment transport in the Caspar Creek Watersheds of northwestern  
1029 California. From their results the authors concluded that the dominant factors influencing the  
1030 difference in suspended sediment loads between watersheds was the difference in road  
1031 alignment, yarding methods, and presence of stream protection zones (i.e., buffers). Because of  
1032 studies like these reviewed, contemporary riparian forest management practices in the western  
1033 United States include rules that limit harvesting, use of equipment, and procedures that disturb  
1034 soil in areas closest to the stream or on steep and unstable slopes ([WAC 222-30-022](#); WAC 22-  
1035 30-021; 2022 [ODF](#); IDAPA 20.[02.01](#))

1036 Since 2000, many of the studies published that evaluate changes in sediment delivery or water  
1037 turbidity following riparian timber harvest show similar results in that contemporary BMPs are  
1038 effective in mitigating increases in sediment delivery to streams (Hatten et al., 2018; Reiter et al.,  
1039 2009). For example, the studies reviewed that report a significant change in sediment delivery  
1040 following harvest show evidence that these changes only persist for a short period of time (1-3  
1041 years) and that the magnitude of these changes are related to the intensity of the harvest  
1042 prescriptions (Karwan et al., 2007; Macdonald et al., 2003a).

1043  
1044 For example, Macdonald et al. (2003a) compared changes in stream discharge rates and in-  
1045 stream suspended sediment concentrations during spring snowmelt between two harvest  
1046 intensities and one unharvested control, for pre- and post-harvest in first order streams of interior  
1047 British Columbia. Both treated riparian areas received a harvest of 55% of the watershed; one  
1048 (low-retention) removed all merchantable timber >15 cm DBH for pine and > 20 cm DBH for  
1049 spruce within 20 m of the stream; the other (high-retention) removed all merchantable timber >  
1050 30 cm within 20 m of the stream. The results showed an increase in spring snowmelt discharge  
1051 for both treatments above predicted values for the study (5 years). However, increased in-stream  
1052 total suspended sediments (TSS) only persisted for two-years post-harvest in the high-retention  
1053 treatment, and for 3-years in the low-retention.

1054  
1055 Karwan et al. (2007) investigated the effects of riparian timber harvest and road construction on  
1056 TSS concentrations in the Mica Creek Experimental Watershed in northern Idaho. Treatments in  
1057 the paired-watershed experiment consisted of 1) commercial clearcut of the watershed area by  
1058 50%, and was broadcast burned and replanted, 2) partial cut in which half the canopy was  
1059 removed in 50% of the watershed area 3) a no-harvest control. All harvests were done according  
1060 to best management practices and the Idaho Forest Practices Act. This included equipment  
1061 exclusion zones of 50- and 30-feet for fish- and non-fish-bearing streams, respectively. On all  
1062 skid trails, drainage features, such as water bars, were installed for erosion control at the end of  
1063 the harvest period. Results showed that road construction in both watersheds did not result in  
1064 significant impacts on monthly sediment loads in either treated watershed during the immediate  
1065 (1-year post-harvest) or recovery (2-4 years post-harvest) time intervals. A significant and  
1066 immediate impact of harvest on monthly sediment loads in the clear-cut watershed ( $p = 0.00011$ ),  
1067 and a marginally significant impact of harvest on monthly sediment loads in the partial cut ( $p =$   
1068  $0.081$ ) were observed. However, after one year, the TSS loads in both treatments became  
1069 statistically indistinguishable from the control.

1070  
1071 Specific to Washington, McIntyre et al. (2021) evaluated the effectiveness of riparian buffers on  
1072 non-fish-bearing streams underlain by competent lithologies (“Hard Rock”) in western  
1073 Washington. Riparian management zones (RMZs) were treated with one of four treatments  
1074 including three harvest prescriptions ((1) a minimum two-sided 50-ft riparian buffer along the  
1075 entire RMZ, (2) a minimum two-sided 50-ft riparian buffer along at least 50% of the RMZ, and  
1076 (3) clearcut to stream edge (no buffer)), and an unharvested reference. Results for suspended  
1077 sediment export (SSE) following treatment showed episodic increases with storm events that  
1078 rapidly declined. However, changes in SSE were poorly correlated with discharge and exhibited  
1079 high variation between treatment sites. The authors suggest that these results show evidence that  
1080 changes in SSE magnitudes were not related to harvest. Further, they conclude that the sites were  
1081 likely sediment-limited considering the underlying lithology.

1082  
1083 Site factors such as underlying lithology and physiography can interact with the effect of timber  
1084 harvest operations on sediment delivery into streams. Bywater-Reyes et al. (2017) assessed the  
1085 influence of natural controls (basin lithology and physiography) and forest management on  
1086 suspended sediment yields in temperate headwater catchments in northeastern Oregon. Results  
1087 from this study indicate that site lithology was the first order control over suspended sediment  
1088 yield (SSY) with SSY varying by an order of magnitude across lithologies observed.  
1089 Specifically, SSY was greater in catchments underlain by Siletz Volcanics ( $r = 0.6$ ), the Trask  
1090 River Formation ( $r = 0.4$ ), and landslide deposits ( $r = 0.9$ ) and displayed an exponential  
1091 relationship when plotted against the percentage of watershed area underlain by these lithologies.  
1092 In contrast, lithology had a strong negative correlation with percent area underlain by diabase ( $r$   
1093  $= 0.7$ ), with the lowest SSY associated with 100% diabase. Following timber harvest, increases  
1094 in SSY occurred in all harvested catchments but returned to pre-harvest levels within 1 year  
1095 except for sites that were underlain by sedimentary formations and were clearcut without  
1096 protective buffers. The authors conclude that sites underlain with a friable lithology (e.g.,

1097 sedimentary formations) had, on average, SSYs an order of magnitude higher following harvest  
1098 than those on more resistant lithologies (intrusive rocks).

1099 Mueller & Pitlick, (2013) found similar results in their assessment of the relative effect of  
1100 lithology, basin relief, mean basin slope, and drainage density on in stream sediment supply for  
1101 83 drainage basins in Idaho and Wyoming. The strongest correlation of in stream sediment  
1102 supply was with lithology relative softness (based on grouping of rock types – granitic,  
1103 metasedimentary, volcanic, and sedimentary). Sediment concentrations at bankfull width  
1104 increased by as much as 100-fold as basin lithology became dominated by softer sedimentary  
1105 and volcanic rock compared to lithologies dominated by harder granitic and metasedimentary  
1106 rock. Finally, Wissmar et al. (2004), developed and field-tested erosion risk indices for  
1107 watersheds in western Washington based on land cover. These erosion risk indices used the  
1108 presence of unstable soils (determined by geological formation and underlying lithology), rain-  
1109 on-snow events, immature forest cover (stands <35 years old where open canopies and  
1110 undeveloped root systems could contribute to hillslope instability), presence and coverage of  
1111 roads, and critical slope (hillslope gradients >36%, for terrain with surficial deposits of coarse-  
1112 textured colluvial materials). Results of this study showed these variables could explain ~65% of  
1113 the variation associated with sediment input into channels. The lowest risk areas contained the  
1114 fewest of these variables (most commonly critical slope with either rain-on snow events or  
1115 immature forests), while higher risk areas contained a combination of 4 or more of these factors  
1116 indicating a compounding effect.

1117 Changes in sediment yield may also interact with increases in discharge rates caused by timber  
1118 harvest as well as physiographic site factors. For example, Bywater-Reyes et al. (2018)  
1119 quantified how sediment yields vary with catchment lithology and physiography, discharge, and  
1120 disturbance history over 60 years in the H.J. Andrews experimental watershed in the western  
1121 Cascade Range of Oregon. Methods for determining suspended sediment concentration involved  
1122 using either vertically integrated storm-based grab samples, or discharge-proportional composite  
1123 samples where composite samples were collected every three weeks at the outlet of each  
1124 catchment. Data sets were taken from 10 watersheds, 7 with a history of management (mixture of  
1125 selective canopy removal, patch-cut, 25-100% clearcut, broadcast burning, road building, and  
1126 thinning), and 3 with no history of management that were used as a reference. A linear mixed  
1127 effects model (log transformed to meet the normality assumption) was used to predict annual  
1128 sediment yield. In this model, site was treated as a random effect while discharge and  
1129 physiographic variables were treated as fixed variables. This allowed for the evaluation of the  
1130 relationships between sediment yield and physiographic features (slope, elevation, roughness,  
1131 and index of sediment connectivity) while accounting for site. To account for the effect of  
1132 disturbance history a variable was added to the model when the watershed had a history of  
1133 management or natural disturbances. If the models for the disturbed watersheds significantly  
1134 underpredicted the sediment discharge, the timing of the sudden increases were further examined  
1135 to assess whether it correlated with a disturbance event (e.g., harvesting, road building, and  
1136 slash-burning.) The results of this study show that watershed physiography combined with  
1137 cumulative annual discharge explains 67% of the variation in annual sediment yield across the  
1138 60-year data set regardless of lithology. Relative to other physiographic variables, watershed

1139 slope was the greatest predictor of annual suspended sediment yield. However, the results  
1140 showed that annual sediment yields also moderately correlated with many other physiographic  
1141 variables and caution that the strong relationship with watershed slope is likely a proxy for many  
1142 processes, encompassing multiple catchment characteristics.

1143 In contrast, Safeeq et al. (2020) compared instream and bedload sediment supply under multiple  
1144 harvesting treatments in watersheds of western Oregon that were paired with control watersheds  
1145 by size, aspect, and topography. The treatment watershed was 100% clearcut during the period  
1146 from 1962-1966, broadcast burned in 1966, and re-seeded in 1968. For this study 15-minute  
1147 streamflow data was recorded for both watersheds, and after large storm events. Sediment data  
1148 was collected from 1952 (pre-harvest) through 1988 for suspended sediment data, and 2016 for  
1149 sediment bedload. The control watershed was forested, and had no treatments (e.g., harvest)  
1150 during the study period. Their results estimate that following streamside harvest, increased  
1151 streamflow alone is estimated to be responsible for <10% of sediment transport into streams  
1152 while the increased sediment supply caused by harvest operations is responsible for >90% of the  
1153 sediment transported into streams.

1154 Puntteney-Desmond et al. (2020) found similar results in their assessment of differences in  
1155 instream sediment contributions from the buffer area, harvest area, and buffer-harvest interface.  
1156 Sediment concentration in the runoff was 15.8 times higher for the harvested area than in the  
1157 riparian buffer, and 4.2 times greater than in the harvest-buffer interface. Total sediment yields  
1158 ( $\text{mg m}^{-2} \text{min}^{-1}$ ) from the harvested area (sediment concentration x flow rate) were approximately  
1159 2 times greater than in the buffer areas, and 1.2 times greater in the harvest-buffer interface than  
1160 in the buffer area.

#### 1161 *Summary of Factors Impacting Sediment Delivery into Streams*

1162 From the studies reviewed there is evidence that sediment delivery into streams following timber  
1163 harvest is influenced by not only the intensity of the harvest operation (e.g., presence of retention  
1164 buffers, yarding and equipment use immediately adjacent to the stream, upland clearcut vs.  
1165 thinning), but also by physiography (e.g., hillslope gradient), lithology relative softness, and  
1166 climate (e.g., precipitation, frequency of large storm events). Thus, the change in magnitude of  
1167 sediment delivery following harvest is context dependent and these landscape factors can interact  
1168 with one another to compound these changes. However, from the studies reviewed above there is  
1169 evidence that the implementation of BMPs since the 1970s in the northwestern United States  
1170 lessen the impact and duration of these changes.

1171 [Shade and stream temperature](#)

1172 Canopy cover provides shade for streams that decreases the amount of incoming solar radiation  
1173 and thus influences stream temperatures, although that influence can be highly variable  
1174 depending on shade structure and density surrounding stream courses. Temperature regulation is  
1175 vital for sensitive salmonid fish species that require cooler waters, and shade is often the primary  
1176 function assessed when developing state regulations (Groom et al., 2011; Groom et al., 2018;  
1177 Teply et al., 2014). The importance of shade and cooler in-stream temperatures for fish habitat  
1178 has been thoroughly investigated (Bjornn & Reiser, 1991; Chapman & Bjornn, 1969; Ebersole et  
1179 al., 2001; Sullivan et al., 2000). The streamside shade will likely become even more critical with  
1180 the predicted increases in air temperature over the next century (Manuta et al., 2009). While  
1181 stream temperature is initially reflective of moisture source (e.g., snowmelt, liquid precipitation,  
1182 groundwater inputs) and watershed subsurface soil characteristics. As water flows downstream  
1183 and into higher-order streams, the net rate of temperature gain or loss is the sum of incident  
1184 radiation, evaporation, conduction, and advection (Brown, 1983; Bescheta et al., 1987).

1185 Bescheta et al. (1987) presented evidence that direct beam solar radiation inputs are of the  
1186 highest importance to the stream's net heat exchange rate per unit area compared to other factors.  
1187 Within the net heat exchange calculation, the heat released from evaporation generally cancels  
1188 out the heat gained from warm air temperatures (convective and advective heat transfer). Thus,  
1189 temperature fluctuations are expected to be more severe in less-shaded/more-exposed streams.  
1190 This has been supported by many experimental field and simulation studies showing evidence  
1191 that the reduction of effective shade can lead to considerable increases in peak summer stream  
1192 temperatures primarily due to the increase of incoming solar radiation. However, while increases  
1193 in solar radiation are accepted as the most important factor in stream temperature changes and  
1194 fluctuations following harvest, other factors are also important and may compound these effects.

1195 Guenther et al. (2014) investigated the relationship between changes in stream temperature and  
1196 changes in wind speed, vapor pressure, and evaporation following riparian thinning treatments  
1197 along headwater streams in southwestern British Columbia. Treatment involved reduction of  
1198 basal area by 50% (resulting in 14% reduction in canopy closure) in the upland and riparian  
1199 forests. Results showed a post-harvest increase in wind speed, vapor pressure deficit, air  
1200 temperature and evaporation above the stream, which coincided with increased stream  
1201 temperatures and lower stability. The authors report that prior to harvest, vapor pressure  
1202 gradients often favored condensation over evaporation. Further, they concluded that the  
1203 relationships between the riparian and microclimate variables after harvesting became more  
1204 strongly coupled to ambient climatic conditions due to increased ventilation. Contemporary  
1205 riparian management practices in western North America vary by state. However, all require  
1206 retention of protective buffers that preserve some percentage of shade or canopy cover to  
1207 maintain or mitigate changes in stream temperatures, especially along fish-bearing streams.  
1208 Many studies published in the last two decades report evidence that these practices have been  
1209 effective in mitigating stream temperature changes after harvest.

1210 For example, Bladon et al. (2016), assessed the effectiveness of riparian management  
1211 prescriptions developed for the Oregon Forest Practices Act (FPA). Oregon State requires a 15 m

1212 buffer on either side of small fish-bearing streams with a 6 m no-cut buffer, and a minimum  
1213 retention for conifer basal area of  $\sim 3.7 \text{ m}^2$  for every 300 m ( $\sim 1000$  ft) length of stream. This  
1214 resulted in a reduction of mean canopy closure from  $\sim 96\%$  in the pre-harvest period to  $\sim 89\%$  in  
1215 the post-harvest period in the treatment reaches. In contrast, mean canopy closure in the  
1216 reference reaches changed from  $\sim 92\%$  to  $\sim 91\%$  from pre- to post-treatment periods. Results  
1217 showed there was a significant increase in the 7-day moving maximum temperature from pre- to  
1218 post-harvest values when data was constrained to the period of July 15 – August 15 by  $0.6 \pm$   
1219  $0.2 \text{ }^\circ\text{C}$ . However, when analyzed by individually paired sites, and when interannual and site  
1220 variability was accounted for, no significant changes in stream temperature were observed for 3  
1221 years post-harvest (length of study).

1222 However, Groom et al., (2011a, b) showed evidence that the more stringent rules of the  
1223 Northwest Oregon State Forest Management Plan (FMP; applied to riparian management zones  
1224 on state owned land) was even more effective at maintaining stream temperatures post-harvest.  
1225 The FMP requires a 52 m wide buffer for all fish-bearing streams, with an 8 m no cut buffer  
1226 immediately adjacent to the stream. The results from Groom et al. (2011b) showed that FPA  
1227 (Oregon Forest Practices) post-harvest shade values differed from pre-harvest values (mean  
1228 change in Shade from 85% to 78%), while no difference was found for FMP shade values pre-  
1229 harvest to post-harvest (mean change in Shade from 90% to 89%). Following harvest, maximum  
1230 temperatures at FPA increased relative to FMP on average by  $0.71 \text{ }^\circ\text{C}$ . Similarly, mean  
1231 temperatures increased by  $0.37 \text{ }^\circ\text{C}$  (range: 0.24 - 0.50), minimum temperatures by  $0.13 \text{ }^\circ\text{C}$   
1232 (range: 0.03 - 0.23), and diel fluctuation increased by  $0.58 \text{ }^\circ\text{C}$  (range: 0.41 - 0.75) relative to  
1233 FMP sites.

1234 Groom et al (2011a) developed prediction models from this data to estimate the probability of  
1235 riparian harvest under each regulation causing an increase in stream temperatures  $>0.3 \text{ }^\circ\text{C}$  (the  
1236 Protecting Cold Water criterion developed by the Department of Environmental Quality). Results  
1237 indicate that sites harvested according to FPA standards exhibited a 40.1% probability of a  
1238 temperature change of  $> 0.3^\circ\text{C}$  from pre- to post harvest. Conversely, harvest to FMP standards  
1239 resulted in an 8.6% probability of exceedance that did not significantly differ from all other  
1240 comparisons.

1241 In Montana, Sugden et al. (2019) investigated the effectiveness of state regulation which requires  
1242 timber be retained within a minimum of 15.2 m (50 feet) of the stream. Within the riparian  
1243 management zone, no more than half the trees greater than 204 mm (8 in) diameter at breast  
1244 height (DBH) can be removed. In no case, however, can stocking levels of leave trees be reduced  
1245 to less than 217 trees per hectare. Data for canopy cover, stream temperature, and fish population  
1246 were collected for 30 harvest reaches in western Montana (northern Rocky Mountain Region),  
1247 for a minimum of one-year pre- and one-year post-harvest. Shade over the stream surface was  
1248 not directly measured in this study. Instead, canopy cover was used as proxy, using two  
1249 independent estimates of canopy cover (1) used cruise data to populate a canopy cover model  
1250 within Forest Vegetation Simulator, and (2) measured canopy cover in the harvested reach every  
1251 30 m, before and after harvest. Within harvest units, mean basal area was reduced by 13%  
1252 (range: 0 – 36%), and again further by a mean of 2% due to windthrow. Mean canopy cover

1253 within the riparian management area reduced from 77% (pre-treatment) to 74% (post-treatment),  
1254 and mean canopy cover over the stream changed from 66% (pre-treatment) to 67% (post-  
1255 treatment) based on densiometer measurements. Neither of these changes were significant.  
1256 Results for stream temperature also showed no significant changes in stream temperatures or fish  
1257 populations in one-year post treatment compared to pre-treatment values.

1258 Specific to Washington, Cupp & Lofgren (2014) conducted a study to test the effectiveness of  
1259 riparian timber harvest rules for eastern Washington in preserving shade and stream  
1260 temperatures. Regulations for fish-bearing streams in eastern Washington (in the mixed  
1261 conifer/mid elevation zone) includes an “All Available Shade Rule” (ASR) for streams in the bull  
1262 trout habitat zones, and a “Standard Shade Rule” (SR). Under the ASR it is required to retain all  
1263 available shade within 75 feet of the stream. Under SR some harvest of shade providing trees is  
1264 allowed within the 75-foot buffer depending on elevation and pre-harvest canopy cover.  
1265 Unharvested reference reaches were located upstream from treatment reaches. Prior to harvest  
1266 treatments, canopy closure measurements ranged from 89% to 97%, with a mean of 93%.  
1267 Results showed post-harvest shade values decreased in SR sites (mean effect of -2.8%,  $p =$   
1268 0.002), as did the canopy closure values (mean effect of -4.5%,  $p < 0.001$ ). Shade and canopy  
1269 closure values did not significantly change after treatment in the ASR sites. Post-harvest mean  
1270 daily maximum stream temperature increased 0.16 °C in the SR harvest reaches, whereas stream  
1271 temperatures in both the ASR sites and in the no-harvest reference reaches increased on average  
1272 by 0.02 °C. Sample period means of daily maximum temperature responses varied from -1.1 °C  
1273 to 0.7 °C in the first two years post-harvest for the ASR sites, from -0.5 to 0.8 °C, in the SR  
1274 sites, and -0.5 to 0.9 °C in the reference sites. While these values show a slight increase in mean  
1275 temperatures and temperature ranges with treatment, the authors interpret these results as  
1276 evidence that temperature effects of the SR and ASR were similar to reference conditions along  
1277 sampled reaches.

1278 Riparian harvest rules along non-fish-bearing streams tend to allow for narrower buffer widths  
1279 (sometimes with no retention buffers) or more intense thinning within the buffer than for fish-  
1280 bearing streams. For example, in western Washington the Forest Practices (FP) buffer  
1281 prescription requires a minimum two-sided 15 m (50 ft) wide buffer along a minimum of 50% of  
1282 the length of a non-fish-bearing perennial stream (i.e., up to 50% of the stream may have no  
1283 buffer) with a 9.1 m (30 ft) equipment exclusion zone along the entire stream. Two recent studies  
1284 (Ehinger et al., 2021 “Soft Rock”; McIntyre et al., 2021 “Hard Rock”) have compared these FP  
1285 buffers, and for Hard Rock two alternative buffer treatments (50 ft buffer along 100% of the  
1286 stream length (100%), and no buffer (0%) treatment), to unharvested reference (REF) sites. Hard  
1287 Rock sites were underlain by competent lithologies and Soft Rock by incompetent (easily  
1288 eroded) lithologies.

1289 Results from the Hard Rock study showed that riparian canopy cover declined after harvest in all  
1290 buffer treatments reaching a minimum around 4 years post-harvest (after mortality stabilized).  
1291 The treatments, ranked from least to most change, were REF, 100%, FP, and 0% for all metrics  
1292 across all years. Effective shade results showed decreases of 11, 36, and 74 percent in the 100%,  
1293 FP, and 0% treatments, respectively. These changes in shade were significant for all treatments.

1294 This led to changes in mean stream temperature from pre- to post-harvest in the 100% treatment  
1295 by 1.1°C in the first two years following treatment, but returned to pre-harvest levels by post-  
1296 harvest year 3. In contrast, the mean difference in pre- to post-harvest stream temperatures in the  
1297 FP ranged from 0.5°C to 1.1°C in the first year, and changed little over the entire post-harvest  
1298 period. Results for the 0% treatment showed a mean increase of 3.8°C immediately following  
1299 harvest and declined over time to 0.8°C by post-harvest year 9. These results suggest that the  
1300 100% treatment was most effective at preventing increases to stream temperature followed by the  
1301 FP and 0% treatments. A weak and nearly significant (P-value range: 0.008 - 0.108) negative  
1302 relationship between canopy cover and stream temperature for the first 4 years after treatment  
1303 was detected. These results provide evidence that the effectiveness of buffers in maintaining  
1304 stream temperatures post-harvest is relative to the intensity of the treatment (e.g., presence of  
1305 buffer, reduction in canopy cover). Further, post-treatment mortality within the buffer from  
1306 events such as windthrow can cause fluctuations in stream temperature response during the first  
1307 decade. Results from the Soft Rock Study showed similar trends in canopy cover reduction and  
1308 stream temperature increases. Authors of the Soft Rock study note that stream temperature  
1309 changes varied as a function of the proportion of the stream buffered and tree mortality.

1310 Outside of Washington, several studies conducted in western North America since 2000 have  
1311 shown results similar to the Hard Rock and Soft Rock studies. For example, Roon et al. (2021b)  
1312 compared stream temperature changes following variable riparian thinning intensities in the  
1313 redwood forests of northern California. Treatments to riparian stands included reduction of  
1314 canopy cover that resulted in reduction of effective shade by either (19-30%) or by (4-5%). Their  
1315 results showed that local changes in stream temperature were dependent on thinning intensity,  
1316 with higher levels of canopy cover reduction leading to increases in local stream temperatures. In  
1317 the reaches with higher reductions in shade (19-30%) there was accumulation of 45° to 115°C  
1318 additional degree days from pre- to post treatment years, while the reaches with lower reductions  
1319 in shade (4-5%) only accumulated 10° to 15°C additional degree days. Further, travel distance of  
1320 increased stream temperatures also appeared to be dependent on thinning intensity. The lower  
1321 shade reduction reaches had an increased temperature effect downstream with travel distance of  
1322 75-150 m, while the high shade reduction sites had a downstream travel distance of 300- ~1000  
1323 m. Roon et al. (2021a) reported changes in in average daily maximum, maximum weekly  
1324 average of the maximum (MWMT), average daily mean, or maximum weekly average of the  
1325 mean (MWAT) at these same sites under the same timeline. The lower thinning intensity (4-5%  
1326 effective shade reduction) showed no significant changes in any temperature metrics. However,  
1327 The more intensely thinned sites (19-30% reduction in effective shade) showed an increase  
1328 MWMT during spring by a mean of 1.7°C (0.9 - 2.5 °C), summer by a mean of 2.8°C (1.8, 3.8  
1329 °C), and fall by a mean of 1.0°C (0.5, 1.5 °C) and increased in downstream reaches during spring  
1330 by a mean of 1.0°C (0.0, 2.0 °C) and summer by a mean of 1.4°C (0.3, 2.6 °C). The authors  
1331 interpret their results as evidence that that changes in shade of 5% or less caused minimal  
1332 changes in temperature while reductions in shade of 20–30% resulted in much larger increases in  
1333 temperature.

1334 Reiter et al. (2020) compared the changes in stream temperatures following different harvest  
1335 treatments along headwater streams in the Trask River Watershed in the northwestern coast range



1336 of Oregon. Treatments included a clearcut to stream (no buffer but half of sites contained some  
1337 leave trees along stream bank), upland clearcut with a 10 m no-cut buffer, upland thinning (basal  
1338 area reduction to 30-50% of original stand) with a 10 m no-cut buffer, and an unharvested  
1339 reference. Results showed that post-harvest stream temperature increases were only significant in  
1340 the clear-cut treatments without buffers with a mean increase of 3.6°C (SE = 0.4°C) for four  
1341 years after the study. They note that temperature changes were more severe in the unbuffered  
1342 streams with no leave trees (4.2 and 4.4°C), however, this difference was not analyzed. No  
1343 significant changes in stream temperature were detected in either treatment with a 10 m no-cut  
1344 buffer. The authors speculate that 10 m wide buffers were sufficient in maintaining stream  
1345 temperatures post-harvest in small, forested headwater streams.

1346 In the sub-boreal forest ecosystems of British Columbia, Canada, Macdonald et al. (2003b)  
1347 compared pre- to post-harvest stream temperature changes in first-order headwater streams under  
1348 3 different riparian forest treatments. These treatments included 1) low-retention – removal of all  
1349 merchantable timber >15 or >20 cm DBH for pine or spruce respectively, within 20 m of the  
1350 stream 2) high-retention – removal of merchantable timber >30 cm DBH within 20-30 m of the  
1351 stream, and 3) patch-cut – high retention for the lower 60% of watershed approaching streams  
1352 and removal of all vegetation in the upper 40% of the watershed. Results showed significant  
1353 increase in stream temperatures ranging from 4 – 6 °C in the low-retention and patch cut in the  
1354 first three years following harvest. However, by year five, mortality in the high-retention buffer  
1355 (due to windthrow) resulted in canopy cover reduction and increases in stream temperatures that  
1356 became equivalent to the other treatments. The authors conclude that while the variation in  
1357 harvest intensity initially appeared to dictate stream temperature responses, site effects (e.g.,  
1358 windthrow susceptibility) can impact the effectiveness of the buffer. While the studies above all  
1359 show evidence that the impact of riparian forest harvest on stream temperatures are related to the  
1360 severity of the harvest prescription (e.g., buffer width, thinning intensity, canopy reduction) the  
1361 results are variable within treatments indicating other site factors are also important when  
1362 evaluating buffer effectiveness. For example, in their review of experimental studies conducted  
1363 in the Pacific Northwest of Canada and the United States, Martin et al. (2021) reported high  
1364 variability in temperature response to streamside buffers. They report a substantial variability and  
1365 overlap in the effect size of the mean 7-day maximum temperature metric with no-cut buffers,  
1366 no-cut plus variable retention buffers, and no-cut patch buffers  $\leq 20$  m wide. The largest  
1367 temperature response ( $> 3.4$  °C) occurred in the clearcut buffers while treatments with buffers  
1368 (i.e., no cut buffers without variable retention) had the smallest response ( $< 0$  °C). The variable  
1369 retention buffers  $< 20$  m showed variable response (0.6 – 1.4 °C). They conclude that the  
1370 variation in temperature response following riparian harvest may be associated with multiple  
1371 factors such as geology, hydrology, topography, latitude, and stream azimuth.

1372 Bladon et al. (2018) investigated the changes in stream temperatures following treatments that  
1373 varied from clearcuts to stream to buffers  $> 20$  m in western Oregon. They performed a  
1374 regression analysis to assess the relative relationship between catchment lithology and the  
1375 percentage catchment harvested with stream temperature at all sites. Their results showed that at  
1376 the upstream harvested sites there was a strong relationship between stream temperature  
1377 increases and catchment lithologies, but no statistically significant relationship between stream

1378 temperature changes and percent of catchment harvested. Sites downstream from harvested areas  
1379 showed a significant relationship with the interaction of percentage of catchment harvested and  
1380 the underlying lithologies ( $p = 0.01$ ). The greatest temperature increases at downstream sites  
1381 were in areas with a higher percentage of catchment harvested and were underlain by more  
1382 resistant lithologies. There was no evidence for increases in stream temperatures in catchments  
1383 with a high percentage of harvest that were underlain by permeable geology. The authors suggest  
1384 that this relationship may be due to the buffering effect of increases in summer low flows and  
1385 greater groundwater or hyporheic exchange. They conclude that the variability of rock  
1386 permeability and the relative contribution of groundwater during summer months, and their  
1387 effect on stream temperatures following harvest should be investigated further.

1388 There is evidence that geomorphology alone can impact stream temperature fluctuations  
1389 throughout the year. Hunter & Quinn, (2009) compared seasonal fluctuations in stream  
1390 temperatures between two watersheds in the Olympic Peninsula, Washington. Both watersheds  
1391 were similar in all characteristics except for bed substrate. One was underlain by alluvial bed  
1392 substrate while the other was underlain by bedrock. Results from this study show consistent  
1393 differences in stream temperature response in alluvial versus bedrock channels. Seasonal  
1394 maximum and minimum average daily temperatures varied less at the alluvial site compared to  
1395 the bedrock site. This, the authors suggest, may be due to hyporheic exchange in alluvial  
1396 channels helping to buffer surface water temperatures from gaining or losing heat. In addition,  
1397 groundwater may also contribute to the increased stability at the alluvial site. Aside from shade  
1398 reduction from timber harvest, there is evidence that light availability and canopy cover naturally  
1399 changes over time as riparian stands develop. For example, Warren et al. (2013) compared  
1400 canopy cover and stream light availability between old-growth-forests (>500 years old) and  
1401 young harvest-aged stands (~40-60 years old) in the H.J. Andrews Experimental Forest in the  
1402 Cascade mountains of Oregon. Streams were paired based on reach length and bankfull width,  
1403 and north ( $n=2$ ), and south ( $n=2$ ) facing watersheds. Canopy cover was estimated using a  
1404 convex spherical densiometer, and light reaching the stream bed was estimated using a  
1405 fluorescent dye that degrades overtime from light exposure. Overall, three of the four paired old-  
1406 growth reaches (2 south-facing, 1 north-facing) had significantly lower mean percent canopy  
1407 cover ( $p < 0.10$ ), and significantly higher mean decline in fluorescent dye concentrations ( $p <$   
1408  $0.01$ ). The authors interpret these results as evidence that old-growth forest canopies were more  
1409 complex and had more frequent gaps allowing for more light availability and lower mean canopy  
1410 cover, on average, than in adjacent young, second growth forests.

1411 Kaylor et al. (2017) presented similar results when they compared canopy cover and light  
1412 availability between small mountain streams adjacent to late-successional forests (dominant  
1413 canopy trees >300 years old) and second-growth forests that had been harvested to the stream  
1414 50-60 years prior to data collection. Like Warren et al. (2013), canopy cover was estimated with  
1415 a convex spherical densiometer; and light availability to streams was estimated with a  
1416 photodegrading fluorescent dye. However, for this study, fluorescent dye degradation was  
1417 converted to photosynthetically active radiation (PAR) by building a linear relationship between  
1418 the dye degradation and PAR sensors. Results showed that mean PAR reaching streams was 1.7  
1419 times greater, and canopy openness was 6.1% greater in >300-year-old forests than in 30-100-

1420 year-old forests. Of the 14 paired sites, differences in canopy openness and PAR were significant  
1421 for 6 sites. The authors compared and combined their data with published data from 10 other  
1422 similar studies. The combined datapoints for canopy openness (%) were plotted against stand age  
1423 and fit it with a negative exponential curve. From the slope of the curve, the authors estimate that  
1424 canopy openness reaches its minimum value in regenerating forests at ~30 years and maintains  
1425 with little variability until ~100 years.

1426 *Summary of Factors Affecting Shade and Stream Temperature*

1427 From the studies reviewed above, the results show evidence that changes in canopy cover and  
1428 effective shade are, not surprisingly, directly related to the intensity of harvest operation. Initial  
1429 reduction in canopy cover and shade from pre- to post-harvest are influenced by the basal area  
1430 removed and the width of the retention buffer. However, there is evidence that multiple site  
1431 factors can interact with harvest operations (e.g., target basal areas).

1432

1433 Results/discussion by focal question

1434 Focal Question 1

1435 1. *What are the effects of timber harvest intensities and extent on the riparian functions, with an*  
1436 *emphasis on the five key functions listed above, in comparison to conditions before harvest?*

1437 From the perspective of an experimental design, this question inquires how the values of the  
1438 metrics used to describe the five key functions (large wood recruitment, sediment filtration,  
1439 stream bank stability, shade, litterfall and nutrients) differ from pre- to post-harvest within  
1440 particular riparian areas of interest. An attempt to answer this question would require data  
1441 collection before and after treatment with or without a control site. Thus, only studies that used a  
1442 BACI or BAI approach are appropriate for discussing this question. From our review, 22 papers  
1443 report pre- to post-harvest changes in the magnitude of one or more of the key functions with the  
1444 majority of these papers focusing on changes in shade. No studies published since 2000 that  
1445 apply an experimental design in western North America to quantify changes in bank stability  
1446 could be found in the literature.

Function	Count
Shade	12
Litter	3
LW	2
Sediment	4
Nutrients	3
Bank Stability	0

1447

1448 *Shade*

1449 Table 4. Treatment and responses for selected publications investigating shade relevant to Q1.

Reference	Treatment	Response
Bladon et al. (2016)	Buffer width of 15 meters (~50 feet)	3 years post-harvest (n = 6) Mean canopy closure was reduced from ~96% (pre-harvest) to ~89% (post-harvest).
Cupp & Lofgren (2014)	Buffer width of 75 feet	<u>1-2 years post-harvest</u> *ASR: Of 16 sites, 13 showed a decrease in shade ranging from 1 to 4%. 2 sites showed no change and 1 site showed an increase in shade of 4% (mean decrease of 1%). **SR: Of 14 sites, 13 showed a decrease in shade ranging from 1 to 10%, and 1 site showed an increase of 1% (mean decrease of 4%).
Gravelle & Link (2007)	Clearcut to stream; Thinning to 50% canopy cover	<u>1- and 2-years post-harvest</u> Pre-harvest shade ranged from 56% to 88% with a mean of 70% in control reaches (n = 4), 63% in clearcut reaches (n = 2), and 74% in thinned reaches (n = 2). In the clearcut reaches, post-harvest shade was reduced to a mean of 52% and 41% for years one and 2, respectively, In the thinned reaches, post-harvest shade remained near 75% for years 1 and 2.

Groom et al. (2011b)	Buffer width of 21 meters (~69 feet; Private); 52 meters (~170 feet; State)	<u>1 year post-harvest</u> For private sites (n = 18) mean post-harvest shade values decreased significantly from 85% to 78%; No statistical difference was found for state site (n = 15) mean shade values from pre- to post-harvest (90% to 89%).
McIntyre et al. (2021)	Buffer widths of ≥50 feet (100%); ≥50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)	<u>9 years post-harvest</u> Riparian cover declined after harvest in all buffer treatments reaching a <u>minimum at 4 years</u> . 100% buffers (n=4) showed a change in mean shade ranging from +1 to -10 % over nine years post-harvest. FP buffers (n = 3) showed a change in mean shade ranging from -12 to -32% over nine years. The unbuffered sites (n =4) showed a change in shade ranging from -27 to -87% over nine years. The 100% buffer recovered to pre-harvest values by year 9.
Reiter et al. (2020)	Clearcut, no buffer (CC_NB), clearcut with 10-m no cut buffer (CC_B), thinning with 10 m no-cut buffer (TH_B).	<u>1 year post-harvest</u> The CC_NB (n = 3) showed a reduction in shade ranging from 1.8 to 3.2% (mean = 2.4%). The CC_B treatment (n =3) showed a reduction ranging from 18.6 to 76.6% (mean = 56.6%). The TH_B (n = 1) showed an increase in shade of 2.8%.
Roon et al. (2021a)	Buffer width of 45 meters (~150 feet)	<u>1 year post-harvest</u> Of the two watersheds surveyed one showed a significant reduction in mean shade and canopy closure of 18.7 and 23.0%, respectively. The second showed a non-significant reduction of mean shade and canopy closure by 4.1 and 1.9%.
Sugden et al. (2019)	Buffer width of 15.2 meters (~50 feet)	<u>1 year post-harvest</u> <u>Mean post-harvest canopy cover increased by 1% (n = 28; range = -48 to +17%) measured with a densiometer. ***FVS Modeling based on tree metrics estimated a mean reduction in shade of 4.1% from pre- to post-harvest</u>
Swartz et al. (2020)	Buffer width of ~20-meter (65 feet) diameter gaps along streambank	<u>1-2 years post-harvest</u> Treatment reach (n = 4) mean shading declined by only 4% (SD ± 0.02%) post-harvest.

1450

1451 Specific to fish-bearing streams of eastern Washington, Cupp & Lofgren (2014) reported changes  
1452 in canopy closure (quantified with handheld densiometer) and shade (quantified with fisheye lens  
1453 digital camera) within reaches adjacent to riparian forests harvested under the All Available  
1454 Shade Rule (ASR) and the Standard Shade Rule (SR). Both shade rules have a 30-ft no-cut  
1455 buffer (core zone) immediately adjacent to the stream. The SR prescription allows thinning in the  
1456 buffer zone 30-75 feet (inner zone) from the stream while the AAS prescription requires  
1457 retention of all shade providing trees in this area. Results showed post-harvest shade values  
1458 decreased in SR sites (mean effect of -2.8%, p = 0.002), as did the canopy closure values (mean  
1459 effect of -4.5%, p < 0.001). Shade and canopy closure values did not significantly change in the  
1460 treatment reaches of the ASR sites. Mean shade reduction in the SR treatment sites exceeded the

1461 mean shade reduction in the ASR sites by 3%. Canopy closure reduction was also greater in the  
1462 SR sites than in the ASR sites by a mean of 4%.

1463 For non-fish-bearing streams of western Washington, McIntyre et al. (2021) report changes in  
1464 canopy closure following three different riparian buffer prescriptions. Riparian buffer  
1465 prescriptions included a minimum two-sided 50-ft wide riparian buffer along the entire stream  
1466 (100%), a minimum two-sided 50-ft riparian buffer along at least 50% of the stream consistent  
1467 with the current Forest Practices buffer prescription (FP), and a clearcut to stream edge without a  
1468 buffer (0%). The canopy cover was estimated at mid-stream with a handheld densiometer and  
1469 was converted to effective shade values for the 5 years post-harvest. Hemispherical canopy  
1470 photos were also taken for 2 years pre-harvest and 5 years post-harvest and converted to Canopy  
1471 and Topographic Density (percentage of the photograph obscured by vegetation or topography).  
1472 Results for canopy cover showed that riparian cover declined after harvest in all buffer  
1473 treatments reaching a minimum around 4 years post-harvest. The treatments, ranked from least to  
1474 most change, were 100%, FP, and 0% for all metrics and across all years. Effective shade results  
1475 showed decreases of 11, 36, and 74 percent in the 100%, FP, and 0% treatments, respectively, by  
1476 3 years post-harvest. However, by post-harvest year 9, canopy closure returned to pre-harvest  
1477 levels in the 100% treatment but remained 15% and 27% below pre-harvest values at the FP and  
1478 0% treatments, respectively. Another study, Janisch et al. (2012) also compared the effects of  
1479 similar treatments (clearcut to stream, a full continuous buffer (10-15 m wide), and a patched  
1480 buffer (~50-110 m long were retained in distinct patches along some portion of the channel) to  
1481 canopy cover. Canopy cover in all streams averaged 95% (SE = 0.4) prior to harvest. Following  
1482 treatment, canopy cover in the clearcut catchments averaged 53%, (SE = 7.4) canopy cover in the  
1483 patch buffer treatment averaged 76%, (SE = 5.1) and canopy cover in the continuous buffer  
1484 treatment averaged 86% (SE = 1.7). The changes were significant in the clearcut and patch  
1485 buffers.

1486

1487 Outside of Washington, Bladon et al. (2016) assessed the effects of harvest treatments under the  
1488 Oregon Forest Practices Act (FPA) on shade reduction and stream temperature. This study took  
1489 place in the Siuslaw National Forest in the Oregon Coast Range in the Alsea Watershed.  
1490 Treatment under the FPA includes a 15 m riparian management area with a minimum of ~3.7 m<sup>2</sup>  
1491 conifer basal area retained for every 300 m length of stream and an additional 4-5 wildlife leave  
1492 trees per hectare. This resulted in a mean canopy closure reduction from ~96% (pre-harvest) to  
1493 ~89% (post-harvest) based on measurements from a densiometer along the stream channel for 3  
1494 years pre- and 3 years post-harvest. Unfortunately, the authors did not compare these changes  
1495 with statistical analysis. Groom et al. (2011b) compared changes in shade from pre- to post-  
1496 harvest under the FPA and under the Northwest Oregon State Forest Management Plan (FMP).  
1497 The FMP requires a 52 m wide buffer for all fish-bearing streams, with an 8 m no cut buffer  
1498 immediately adjacent to the stream.

1499 Results from Groom et al. (2011b) showed that FPA site post-harvest shade values differed from  
1500 pre-harvest values (mean change in Shade from 85% to 78%); While no difference was found for  
1501 FMP site shade values pre-harvest to post-harvest (mean change in Shade from 90% to 89%). In  
1502 the Trask Watershed of the northwestern Oregon Coast range, Reiter et al. (2020) compared three  
1503 riparian zone treatments: 1) clearcut, no buffer (CC\_NB; n = 4), 2) clearcut with 10-m no cut

1504 buffer (CC\_B; n = 3), 3) thinning with 10 m no-cut buffer (TH\_B; n=1) in small non-fish-  
 1505 bearing streams. Pre- to post-harvest values in shade were quantified with hemispherical analysis  
 1506 over the stream one-year prior and one-year post-treatment. However, post-harvest overstory  
 1507 buffer width varied within each treatment depending on landscape factors. For this reason, we  
 1508 will present the change in percent shade with residual buffer width (Table 6). Again, changes in  
 1509 shade were not statistically analyzed.

1510 In fish-bearing streams within the McKenzie River basin in the western Cascade Mountains of  
 1511 Oregon Swartz et al. (2020) assessed the effects of experimental canopy gap treatments on shade  
 1512 and light availability to the stream. In each treatment reach (n = 6), 20 m gaps were prescribed to  
 1513 mimic gap openings that naturally occur after individual large tree mortality or small-scale  
 1514 disturbance events in late successional forests. Shade was recorded in the year before and the  
 1515 year after treatment with hemispherical photos. Changes in effective shade (1 year post-harvest)  
 1516 were estimated in HemiView 2.1 software. Mean stream shading could not be evaluated in the  
 1517 full BACI analysis because post-treatment hemispherical photographs could not be taken at all  
 1518 sites due to fire impeding access in 2018. For the remaining sites, the areas beneath each gap had  
 1519 notable localized declines in shade, through the entirety of the treatment reach mean shading  
 1520 declined by only 4% (SD ± 0.02%).

1521 Table 5. Results for changes in shade following treatment for the Trask River Watershed Study  
 1522 headwaters. Reproduced from Reiter et al (2020).

<b>Treatment</b>	<b>Mean residual buffer width (2-sided)</b>	<b>Pre-harvest shade (%)</b>	<b>Post-harvest shade (%)</b>
CC_B	33.2	85.9	82.7
CC_B	22.6	91.3	89.1
CC_B	23.9	84.7	82.9
CC_NB	0.0	83.6	7.0
CC_NB	0.0	85.5	10.9
CC_NB	16.0	84.3	65.7
CC_NB	14.1	80.6	76.6
TH_B	*	81.2	84.0

1523 CC\_B = clearcut with 10 m buffer, CC\_NB = clearcut no buffer, TH\_B upland thinning with  
 1524 buffer. \*Unable to determine exact buffer width because adjacent to thinning

1525 Gravelle & Link (2007) compared changes in shade following treatment for non-fish-bearing  
 1526 streams in northern Idaho. For non-fish-bearing streams there is a 30 ft (9.1 m) equipment  
 1527 exclusion zone on each side of the ordinary high-water mark (definable bank). There are no  
 1528 shade requirements and no leave tree requirements, but skidding logs in or through streams is  
 1529 prohibited. Harvesting treatments included (1) clearcut and (2) thinning to a 50% shade removal.  
 1530 Canopy cover measurements were made using a concave spherical densiometer. Preharvest  
 1531 canopy measurements ranged from 56% to 88%, with an average of 63% in the clearcut reaches,  
 1532 and 74% in the partial cut reaches. In the clearcut reaches, canopy was reduced to 52% in 2002  
 1533 and 41% in 2003, immediately following broadcast burning and replanting. In 2004 and 2005,

1534 overall canopy was measured at 56% and 54%, respectively. Streamside shade recovery can be  
1535 attributed entirely to low-lying understory species, as evidenced by the increase in  
1536 understory/deciduous cover of 26% in 2003 to 39% and 37% in 2004 and 2005, respectively. In  
1537 the partial cut reaches, canopy shade remained near 75%.

1538 In fish-bearing streams of Montana, Sugden et al. (2019) assessed the effectiveness of state  
1539 riparian management harvest prescriptions in maintaining canopy cover. Montana state law  
1540 requires timber be retained within a minimum of 15.2 m of fish-bearing streams, with equipment  
1541 exclusion zones extended on steep slopes for up to 30.5 m. Within the riparian management  
1542 zone, no more than half the trees greater than 204 mm (8 in) diameter at breast height (DBH) can  
1543 be removed. In no case, however, can stocking levels of leave trees be reduced to less than 217  
1544 trees per hectare. Shade over the stream surface was not directly measured in this study. Rather,  
1545 canopy cover was used as a general proxy, with two independent estimates of canopy cover  
1546 employed. One method used the riparian cruise data to populate a canopy cover model within the  
1547 Forest Vegetation Simulator (FVS), which estimated canopy cover for each study site, pre- and  
1548 post-harvest. The second method measured canopy cover in the harvest reach every 30 m, both  
1549 before and after timber harvest, using a concave spherical forest densiometer. Mean canopy  
1550 cover in the SMZ, as modelled in FVS, decreased from 77% to 74% following timber harvest  
1551 and 73% when subtracting windthrow to differentiate between direct and indirect impacts of  
1552 management (Table 3). The mean canopy cover over the stream channel based on densiometer  
1553 measurements was 66% pre-harvest and 67% post-harvest. Neither of these changes was  
1554 statistically significant.

1555 Roon et al. (2021a) compared the effects of two experimental thinning treatments on shade in  
1556 second growth redwood stands (40-60 years old) of northern California. This study took place  
1557 between 2016 and 2018 with thinning treatments applied during 2017 giving 1-year pre-  
1558 treatment and 1-year of post-treatment data. Two study sites prescribed treatment on one side of  
1559 the stream of a 45 m buffer width with a 22.5 m inner zone with 85% canopy retention and a  
1560 22.5 m outer zone that retained 70% canopy cover (Tectah watershed). At the third treatment site,  
1561 thinning prescriptions included removal of up to 40% of the basal area within the riparian zone  
1562 on slopes less than 20% on both sides of the channel along a ~100–150 m reach (Lost Man  
1563 watershed, Redwood national park). Shade over streams was measured with hemispherical  
1564 photos and effective shade was calculated in HemiView Canopy Analysis Software version 2.1.  
1565 Results for the Tectah watershed showed a significant reduction in canopy closure by a mean of  
1566 18.7%, (95% CI: -21.0, -16.3) and a significant reduction of effective shade by a mean of 23.0%  
1567 (-25.8, -20.1) one-year post treatment. In the Lost man watershed, a non-significant reduction of  
1568 mean shade by 4.1% (-8.0, -0.5), and mean canopy closure by 1.9% was observed in 2018.  
1569 Results for below canopy light availability showed significant increases by a mean of 33% (27.3,  
1570 38.5) in the Tectah watershed, and non-significant increases in Lost man watershed of 2.5% (-  
1571 1.6, 5.6) by 2018.

1572 In general, the results from the studies reviewed above suggest changes in shade or canopy cover  
1573 from pre- to post-harvest are directly impacted by the intensity of the treatment prescription.  
1574 Buffer treatments vary between states and within states by stream type (e.g., fish-bearing or non-



1575 fish-bearing), For the studies that quantified pre- to post-changes in shade along fish-bearing  
 1576 streams (Cupp & Lofgren, 2014; Sugden et al. 2019), results show evidence that the application  
 1577 of best management practices (BMPs) cause minimal or non-significant changes in shade  
 1578 following harvest. For non-fish-bearing streams harvest prescriptions are much more variable.  
 1579 Further, there are many more examples of application and comparison of different experimental  
 1580 buffer treatments which vary by width or thinning targets.

1581 *Litter*

1582

1583 Table 7. Treatment and responses for selected publications investigating Litter relevant to Q1.

Reference	Treatment	Response
McIntyre et al. (2018)	Buffer width of 50 feet (100%); 50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)	<u>2-years post-harvest</u> Total litterfall input showed a significant decrease in the FP buffers (n = 2; $\Delta = -0.2711$ g) and the unbuffered (n = 2; $\Delta = -0.3823$ g) treatments. Total Leaf litterfall (deciduous and conifer leaves combined) also showed a significant decrease in the FP buffers (n = 2; $\Delta = -0.1255$ g) and the unbuffered (n = 2; $\Delta = -0.2779$ g). Conifer litterfall input significantly decreased in the FP (n = 2; $\Delta = -0.0437$ ) and unbuffered (n = 2; $\Delta = -0.1574$ g) treatments. Deciduous litterfall decreased significantly only in the unbuffered (n = 2; $\Delta = -0.1563$ g) treatment. Wood input (twigs and cones) decreased significantly in the FP (n = 2; $\Delta = -0.2665$ g) and unbuffered (n = 2; $\Delta = -0.2203$ g) treatments.
Kiffney & Richardson (2010)	Upland clearcuts with (1) no buffer, (2) 10 m buffer (~30 feet), (3) 30 m buffers	<u>8 years post-harvest</u> The no buffer treatment showed an ~91% reduction of litterfall in the first year with recovery to an ~11% reduction by year 8. The 10 m buffer treatment showed an initial reduction in litterfall by ~2%, but an increase of ~37% by year 8, compared to pre-harvest. The 30 m buffer treatment showed an initial increase in litterfall by ~11% in the first year which increased to ~74% by year 8 relative to pre-harvest levels.

1584

1585 Specific to western Washington, McIntyre et al. (2018) compared the change in litterfall inputs  
 1586 from pre- to post-harvest under three different riparian harvest treatments. Treatments included a  
 1587 minimum two-sided 50-ft riparian buffer along at least 50% of the stream (FP; with clearcut to  
 1588 stream's edge outside of the buffer), a minimum two-sided 50-ft buffer along the entire stream  
 1589 (100%), and a clearcut to stream without a buffer (0%). Litterfall was collected with litter traps  
 1590 placed along the mainstem channel of each site. Litter was dried and sorted by type (e.g.,  
 1591 deciduous, conifer, small wood) and ashed to compare weight. Results for litterfall input showed  
 1592 a decrease in total litterfall input in the FP (P = 0.0034) and 0% (P = 0.0001) treatments between  
 1593 pre- and post-treatment periods. Leaf litterfall (deciduous and conifer leaves combined) input  
 1594 decreased in the FP (P = 0.0114) and 0% (P < 0.0001) treatments in the post-treatment period. In  
 1595 addition, conifer (conifer needles and scales) litterfall input decreased in the FP (P = 0.0437) and  
 1596 0% (P < 0.0001) treatments, deciduous leaves in the 0% (P < 0.0001) treatment, wood (twigs and

1597 cones) in the FP (P = 0.0044) and 0% (P = 0.0153) treatments, and misc. (e.g., moss and flowers)  
 1598 in the 0% (P = 0.0422) treatment.

1599 In the Malcom Knapp Research Forests of British Columbia, Canada, Kiffney & Richardson  
 1600 (2010) compared changes in litter input between riparian harvest prescriptions that included  
 1601 clear-cut to stream edge, 10 m wide buffer reserve, and 30 m buffer reserves over the course of 8  
 1602 years. No thinning was applied within the reserves. Upland treatment at all sites used clearcutting  
 1603 methods. Vertical litter inputs were collected monthly and at approximately 6–8-week intervals  
 1604 during each season for years 1,2,6,7, and 8 years after harvest. Litter was separated into  
 1605 broadleaf deciduous, twig, needles, and other (seeds, cones, and moss) categories following  
 1606 collection and subsequently dried and weighed using a microbalance. Results for post-harvest  
 1607 changes in litterfall input by treatment per year are summarized in Table 7. Actual values of pre-  
 1608 to post-harvest changes in litterfall input by type, treatment, and year were not directly reported,  
 1609 however, the authors report that post-harvest inputs of needles, twigs, and total particulate matter  
 1610 were significantly lower for clearcuts compared to all other treatments.

1611 Table 7. Percent change in total litterfall percentage post-harvest by treatment per year from  
 1612 Kiffney & Richardson (2010). Table reproduced and modified from Yeung et al. (2019)  
 1613 supplementary materials Appendix C, Table C3.

Harvest type (% of watershed area harvested)	Change in litterfall (%)	Time after harvest (year)
Clearcut (33%) no buffer	~ -91	1
	~ -78	2
	~ -79	6
	~ -47	7
	~ -11	8
Clearcut (23%); with 10-m riparian buffers	~ -2	1
	~ 6	2
	~ -14	6
	~ 6	7
	~ 37	8
Clearcut (18%); with 30-m riparian buffers	~ 11	1
	~ 44	2
	~ 14	6
	~ -6	7

1614

1615 *Large Wood (LW) recruitment*

1616 Table 8. Treatment and responses for selected publications investigating Large Wood relevant to  
1617 Q1.

Reference	Treatment	Response
McIntyre et al. (2021)	Buffer width of 50 feet for full stream length (100%); 50 feet for 50% of the stream length (FP); Clearcut to stream (unbuffered)	<u>8 years post-harvest</u> Large wood recruitment rates were greatest during the first two years post-harvest, then decreased. Mean LW density increased by 66, 44, and 47% in the 100% (n = 4), FP (n =3), and unbuffered treatments (n = 4), respectively. LW density continued to increase in the FP treatment by 42 and 41%, respectively, in years 5 and 8 post-harvest.
Ehinger et al. (2021)	50 feet for 50% of the stream length (FP)	<u>3 years post-harvest</u> Treatments were analyzed by buffer type: mean LW piece counts increased in the 50 foot buffer (n = 8) and the unbuffered (n = 7) type by 8 and 13%, respectively, and decreased in the <50 foot (n = 6) buffer type by 15%.

1618

1619 Specific to western Washington, McIntyre et al. (2021) compared the change in mean in-stream  
1620 large wood from pre- to post-harvest under three different riparian harvest treatments in non-fish-  
1621 bearing streams. Treatments included a minimum two-sided 50-ft riparian buffer along at least  
1622 50% of the stream (FP; with clearcut to stream’s edge outside of the buffer), a two-sided  
1623 minimum 50-ft buffer along the entire stream (100%), and a clearcut to stream without a buffer  
1624 (0%). Results showed a 66% (P <0.001), 44% (P = 0.05) and 47% (P = 0.01) increase in mean  
1625 large wood density in the 100%, FP and 0% treatments, respectively, in the first 2 years post-  
1626 harvest compared with the pre-harvest period and after controlling for temporal changes in the  
1627 references. Five years post-treatment the mean LW density in the FP continued to increase 42%  
1628 (P = 0.08), and again 8 years post-treatment (41%; P = 0.09).

1629 Ehinger et al. (2021) also quantified changes in in-stream LW following the FP riparian harvest  
1630 prescription. Because of unstable slopes, total buffer area was 18 to 163% greater than the  
1631 prescribed 50-foot-buffer. This resulted in 3 different buffer types 1) buffers encompassing the  
1632 full width (50 feet), 2) <50ft buffers, and 3) unbuffered, harvested to the edge of the channel.  
1633 Because of the separation into multiple treatments, sample sizes became small and unbalanced.

1634 Thus, no statistical analyses were conducted, and only descriptive statistics were applied for  
 1635 changes in stand structure and wood loading. However, given the lack of studies presenting  
 1636 changes in LW recruitment from pre- to post-harvest, it is presented here for comparison. Results  
 1637 showed the full buffer sites and <50 ft buffer sites received an average of 23 and 10 pieces/100 m  
 1638 and 2.3 and 0.7 m<sup>3</sup>/100 m of large wood, respectively, post-harvest. The majority of recruited  
 1639 large wood pieces had stems with roots attached (SWRW); 70, and 100% in the full buffer, and  
 1640 <50 ft buffer types, respectively. Pre-harvest channel large wood loading ranged from 55.8 to 111  
 1641 pieces/100 m and from 9.8 to 25.2 m<sup>3</sup>/100 m among buffer types. Piece counts increased in the  
 1642 full buffer and unbuffered sites (8 and 13%, respectively), and decreased in the <50 ft buffers  
 1643 (15%).

1644 *Sediment*

1645 Table 9. Treatment and responses for selected publications investigating Sediment relevant to  
 1646 Q1.

Reference	Treatment	Response
Hatten et al. (2018)	Buffer width of 15 m (~ 50 feet), Oregon Forest practices	<u>1 year post-harvest after 2 harvest events (n = 3)</u> Mean suspended sediment concentrations (SSC) was 32 mg L <sup>-1</sup> (~63%) lower after the first harvest and 28.3 mg L <sup>-1</sup> (~55%) lower after the second harvest when compared to the pre-harvest concentrations.
Bywater-Reyes et al. (2017)	Unbuffered clearcuts; 50 ft buffers. Oregon Forest Practices	<u>3 years post-harvest</u> The first year following harvest suspended sediment yield (SSY) increased in the unbuffered (n = 2) and buffered (n = 1) catchments. By year 2, SSY returned to pre-harvest levels in the buffered, and one of the unbuffered catchments. In one unbuffered catchment, SSY continued to increase annual for all three years.
Karwan et al. (2007)	Buffer width of 75 foot (Idaho Forest Practices Act) with (1) upland clearcut and (2) 50% canopy removal	<u>4 years post-harvest</u> Total suspended sediment (TSS) load from the clearcut exceeded the predicted load by 152% (6,791 kg km <sup>-2</sup> ) in the first year following harvest. The 50% canopy removal showed a non-significant increase in TSS. Neither treatment showed a statistical difference in TSS during the recovery time 2-4 years after harvest compared to pre-harvest.

1647

1648 No studies from Washington published since 2000 provide changes in sediment concentration or  
 1649 transport from pre- to post-harvest. The Hard Rock study (McIntyre et al., 2021) reported their  
 1650 results for water turbidity and suspended sediment export (SSE) were stochastic in nature and the  
 1651 relationships between SSE export and treatment effects were not strong enough to confidently  
 1652 draw conclusions. The lack of SSE in some high discharge events suggests that the basins are  
 1653 likely to be supply limited. The Soft Rock study (Ehinger et al., 2021) similarly reported that  
 1654 their results for changes in sediment post-harvest were highly variable. The SSE data in the Soft

1655 Rock study indicated that the marine sedimentary lithologies were more erodible than then  
 1656 lithologies sampled in the Hard Rock Study. However, prediction equations could not be  
 1657 calculated to predict the response of the treatment sites after harvest. Thus, strong conclusions  
 1658 about the effectiveness of the Forest Practices harvest prescription rules on discharge and SSE  
 1659 could not be drawn.

1660 Hatten et al. (2018) compared pre- to post-harvest suspended sediment concentrations (SSC) in a  
 1661 western Oregon Alsea watershed. Treatments followed contemporary harvesting practices (no  
 1662 buffer in non-fish-bearing streams with equipment exclusion zones, and a 15 m no-cut-buffer in  
 1663 fish-bearing streams) resulted in non-significant changes in SSC at all treatment sites.  
 1664 Surprisingly, in the fish-bearing streams there was a decrease in SSC (~63% and ~55%, after first  
 1665 and second harvest, respectively) compared to pre-harvest values. Bywater-Reyes et al. (2017)  
 1666 compared pre- to post-harvest changes in suspended sediment yield (SSY) following harvest in  
 1667 the Trask River Watershed of western Oregon. Harvest treatments of study sub-watersheds  
 1668 consisted of clearcuts (UM2 and GC3) and a clearcut with buffers (50 ft; ~15 m; PH4).  
 1669 Following timber harvest, (water year 2013), increases in SSY occurred in all harvested  
 1670 catchments. The SSY in both PH4 (clearcut with buffers) and GC3 (clearcut without buffers)  
 1671 declined to pre-harvest levels by water year 2014. Interestingly, the SSY in UM2 (clearcut  
 1672 without buffers) increased annually throughout the post-harvest period, ultimately resulting in  
 1673 the highest SSY of all catchments during the final two years (2015-2016) of the study after  
 1674 producing the lowest SSY in the pre-harvest period. Actual values for SSY and significance were  
 1675 not reported.

1676 Karwan et al. (2007) compared changes in total suspended solids (TSS) in streams from pre- to  
 1677 post-harvest in northern Idaho. Treatments in the paired-watershed experiment consisted of 1)  
 1678 commercial clearcut of the watershed area of 50%, and was broadcast burned and replanted by  
 1679 the end of May 2003, and 2) partial cut in which a target of 50% the canopy was removed in 50%  
 1680 of the watershed in 2001, with final 10% of log processing and hauling in early summer of 2002.  
 1681 All harvests were carried out according to best management practices and in accordance with the  
 1682 Idaho Forest Practices Act. Results showed a significant and immediate impact of harvest on  
 1683 monthly sediment loads in the clear-cut watershed ( $p = 0.00011$ ), and a marginally significant  
 1684 impact of harvest on monthly sediment loads in the partial cut ( $p = 0.081$ ). Total sediment load  
 1685 from the clearcut over the immediate harvest interval exceeded predicted load by 152% (6,791  
 1686 kg km<sup>-2</sup>); however, individual monthly loads varied around this amount. The largest increases in  
 1687 percentage and magnitude occurred during snowmelt months, namely April 2002 (560%, 2,958  
 1688 kg km<sup>-2</sup>) and May 2002 (171%, 3,394 kg km<sup>-2</sup>). Neither treatment showed a statistical  
 1689 difference in TSS during the recovery time 2-4 years after harvest (clearcut:  $p = 0.2336$ ; partial  
 1690 cut:  $p = 0,1739$ ) compared to the calibration loads (pre-harvest).

1691 *Nutrients*

1692 Table 9. Treatment and responses for selected publications investigating Sediment relevant to  
 1693 Q1.

Reference	Treatment	Response
-----------	-----------	----------

McIntyre et al. (2021)	Buffer width of 50 feet (100%); 50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)	Increases in total-N export of 5.73, 10.85, and 15.94 kg/ha/yr post-harvest in the 100%, FP, and 0% treatments, respectively, were detected in the first <b>2 years post-harvest</b> ; and of 6.20, 5.34, and 8.49 kg/ha/yr in the extended period ( <b>7-8 years post-harvest</b> ). Results for nitrate-N export showed changes similar to but slightly less than those seen in the total-N analysis with a relative increase in nitrate-N export of 4.79, 9.63, and 14.41 kg/ha/yr post-harvest in the 100%, FP, and 0% treatments, respectively in the first 2 years. None of the changes in the extended period were significant.
Gravelle et al. (2009)	75 foot buffers (Idaho Forest Practices Act) with (1) upland clearcut and (2) 50% canopy removal	<u>4 years post-harvest</u> Significant increases in nitrogen were observed in the clearcut (n = 1) and partial cut treatments (n = 1). Increases at the clearcut site was greatest from 0.06 mg-N L <sup>-1</sup> (pre-harvest) to 0.35 mg-N L <sup>-1</sup> (post-harvest period, 4 years). There was also an observable seasonal effect on NO <sub>3</sub> + NO <sub>2</sub> concentrations with the peak concentration of 0.89 mg-N L <sup>-1</sup> , with mean monthly concentrations of 0.43 mg-N L <sup>-1</sup> and 0.59 mg-N L <sup>-1</sup> in water years 3 and 4 following harvest, respectively, in the clearcut. No significant changes of in-stream concentration of any other nutrient recorded were found between time periods and treatments.
Deval et al. (2021)	Clearcut to stream with 24-47% vegetation removal (Phase I); Clearcut with 36 – 50% vegetation removal (Phase II)	<u>6 years post-harvest (Phase I), 8 years post-harvest (Phase II); (n = 7)</u> Mean annual NO <sub>3</sub> + NO <sub>2</sub> concentrations increased significantly at all treatment sites during both treatment Phases with the greatest increases occurring during the Phase II period (increases ranging from 1.73 kg ha <sup>-1</sup> yr <sup>-1</sup> – 3.95 kg ha <sup>-1</sup> yr <sup>-1</sup> ). NO <sub>3</sub> + NO <sub>2</sub> concentrations followed an increasing trend throughout the post-harvest period with <b>evidence of recovery in year 8</b> indicated by the flattening of the cumulative load curve.

1694

1695 The “Hard Rock” study (McIntyre et al., 2021) results showed an increase in total-N export of  
1696 5.73 (P = 0.121), 10.85 (P = 0.006), and 15.94 (P = 0.000) kg/ha/yr post-harvest in the 100%, FP,  
1697 and 0% treatments, respectively, in the first 2 years; and of 6.20 (P = 0.095), 5.34 (P = 0.147),  
1698 and 8.49 (P = 0.026) kg/ha/yr in the extended period (7-8 years post-harvest). Results for nitrate-  
1699 N export showed changes similar to but slightly less than those seen in the total-N analysis with  
1700 a relative increase in nitrate-N export of 4.79 (P = 0.123), 9.63 (P = 0.004), and 14.41 (P <0.001)  
1701 kg/ha/yr post-harvest in the 100%, FP, and 0% treatments, respectively in the first 2 years. None  
1702 of the changes in the extended period were significant. However, the authors note that there was  
1703 high variability in the data for the extended period and nitrate-N export only returned to pre-  
1704 harvest levels in one watershed. Total phosphorus export increased post-harvest by a similar  
1705 magnitude in all treatments: 0.10 (P = 0.006), 0.13 (P = 0.001), and 0.09 (P = 0.010) kg/ha/yr in  
1706 the 100%, FP, and 0% treatments, respectively in the first 2 years post-harvest. Changes in  
1707 phosphorus were not reported in the extended period.

1708 Gravelle et al. (2009) compared pre- to post changes in NO<sup>3</sup> and NO<sup>2</sup> concentrations in  
1709 headwater streams following a clearcut and a partial cut (50% removal of canopy cover) in  
1710 northern Idaho. Riparian buffers and leave trees are not required for non-fish-bearing headwater

1711 streams in Idaho. Results showed statistically significant increases in  $\text{NO}^3$  and  $\text{NO}^2$   
1712 concentrations following clearcut and partial harvest cuts in headwater streams ( $p < 0.001$ ).  
1713 Increases at the clearcut treatment site were greatest, where mean monthly concentrations  
1714 increased from 0.06 mg-N L<sup>-1</sup> during the calibration period to 0.35 mg-N L<sup>-1</sup> in the post-  
1715 harvest period. Mean monthly concentrations in the partial cut increased from 0.04 mg-N L<sup>-1</sup> in  
1716 the pre-harvest period to 0.05 mg-N L<sup>-1</sup> in the post-harvest period. No significant changes of  
1717 in-stream concentration of any other nutrient recorded (total Kjeldahl nitrogen (TKN), TP, total  
1718 ammonia nitrogen (TAN) consisting of unionized ( $\text{NH}_3$ ) and ionized ( $\text{NH}_4^+$ ) ammonia, and  
1719 unfiltered orthophosphate (OP)) were found between time periods and treatments.

1720 Deval et al. (2021) compared changes in the same nutrient concentrations in the same area of  
1721 northern Idaho but with an additional harvest prescription several years later. For this analysis,  
1722 time periods were broken into four distinct phases: 1) pre-disturbance (1992–1997), 2) post-road  
1723 (1997–2001), 3) experimental-harvest Phase I (PH-I) (2001–2007), and 4) operational sequential  
1724 harvest Phase II (PH-II) when the extent and frequency of harvests increased (2007–2016). PH-I  
1725 represents an experimental treatment phase during which harvest activities were experimentally  
1726 controlled (only upstream headwater watersheds were harvested and mature vegetation (size or  
1727 age threshold for “mature” not reported) removal ranged between 24% and 47%) followed by  
1728 site management operations including broadcast burning and replanting. PH-II represents the  
1729 post-experimental phase where the study area transitioned to operational treatments that  
1730 consisted of additional road construction and timber harvest, with site management operations  
1731 including pile burning and competition release herbicide application. During this operational  
1732 phase, the mature vegetation (size or age threshold for “mature” not reported) removal in the  
1733 upstream watersheds ranged between 36% and 50%. The response in  $\text{NO}^3 + \text{NO}^2$  concentrations  
1734 was negligible at all treatment sites following the road construction activities. However,  $\text{NO}^3 +$   
1735  $\text{NO}^2$  concentrations during the PH-I period increased significantly ( $p < 0.001$ ) at all treatment  
1736 sites. Similar to the PH-I period, all watersheds experienced significant increases in  $\text{NO}^3 + \text{NO}^2$   
1737 concentration during the PH-II treatment period ( $p < 0.001$ ). Similar to Gravelle et al. (2009),  
1738 significant increases in all other nutrients recorded were not detected.

1739

#### 1740 Focal Question 1a

1741 *1a. What are the effects of thinning (intensity, extent) on the riparian functions, over the short*  
1742 *and long-term compared to untreated stands?*

1743 Based on the wording of this question, papers deemed appropriate are those that compare  
1744 changes in measurable data indicative of the riparian functions between harvested and  
1745 unharvested stands. Further, studies chosen for this question should compare the response of  
1746 these functions based on different thinning intensities. Thus, the design of the studies reviewed  
1747 for this review should be a BACI or ACI design with results reported for differences between  
1748 treatment and reference reaches. Also included are a few simulation modeling experiments that  
1749 follow these designs.

1750

Function	Count
Shade	2
Litter	0
LW	2
Sediment	1
Nutrients	1
Bank Stability	0

1751

1752 *Shade*

1753 Table 10. Treatment and responses for selected publications investigating Shade relevant to Q1a.

Reference	Treatment	Response
Anderson et. al. (2007)	69 m buffers (B1); variable width buffer averaging 22 m (VB); streamside retention buffer averaging 9 m (SR-T)	<u>2-5 years post-harvest</u> Adjacent upland to each buffer treatment was thinned to a range of 98 – 297 trees per hectare. Visible sky at stream center increased with decreasing buffer width. Untreated stands maintained ~4.2% visible sky at stream center. VB and B1 sites showed an increase of visible sky to ~9.3% and the SR-T sites showed an increase to ~9.6%.
Roon et. al. (2021a)	45 m buffer width with 70-85% canopy retention (CC); Up to 40% basal area removal along stream (BA)	<u>1-year post-harvest</u> The CC sites showed a mean canopy cover reduction of 18.7%, (-21.0, -16.3) and a significant reduction of effective shade by a mean of 23.0% (-25.8, -20.1). The BA sites showed a reduction of mean shade by 4.1% (-8.0, -0.5), and mean canopy closure by 1.9%.

1754

1755 Anderson et. al. (2007) compared changes in canopy cover at stream centers between sites  
 1756 adjacent to different riparian zone treatments and an untreated control. This study was conducted  
 1757 in young headwater forests of western Oregon. Treatments included three buffer widths: 1) one  
 1758 site-potential tree averaging 69 m (B1), 2) variable width buffer averaging 22 m (VB), or 3)  
 1759 streamside retention buffer averaging 9 m (SR-T). Adjacent upland to each buffer treatment was  
 1760 thinned to ~198 trees per hectare. Results showed that visible sky at stream center only differed  
 1761 significantly between SR-T (9.6%) and the untreated (4.2%) sites post-harvest. These results  
 1762 were reported for the period 2-5 years post-harvest.

1763 Roon et. al. (2021a) used a BACI analysis to evaluate significant changes in canopy cover  
 1764 relative to untreated reaches following 2 different thinning intensities in second growth redwood  
 1765 forests of northern California. One study site prescribed treatment on one side of the stream of a  
 1766 45 m buffer width with a 22.5 m inner zone with a target 85% canopy retention and a 22.5 m  
 1767 outer zone that retained 70% canopy cover (Green Diamond Resource Company, Tectah  
 1768 watershed). The treatment site, thinning prescriptions included removal of up to 40% of the basal  
 1769 area within the riparian zone on slopes less than 20% on both sides of the channel along a ~100–  
 1770 150 m reach (Lost Man watershed, Redwood national park). Control reaches were located



1771 upstream from treatment reaches. Data analysis was conducted separately for each experimental  
 1772 watershed (i.e., 1 Lost man site, 2 Tectah sites). Results for the Tectah watershed showed a  
 1773 significant reduction in canopy closure by a mean of 18.7%, (95% CI: -21.0, -16.3) and a  
 1774 significant reduction of effective shade by a mean of 23.0% (-25.8, -20.1) one-year post  
 1775 treatment. In the Lost Man watershed, a non-significant reduction of mean shade by 4.1% (-8.0, -  
 1776 0.5), and mean canopy closure by 1.9% was observed. Results for below canopy light availability  
 1777 showed significant increases by a mean of 33% (27.3, 38.5) in the Tectah watershed, and non-  
 1778 significant increases in Lost Man watershed of 2.5% (-1.6, 5.6). Data for canopy closure and  
 1779 effective shade were recorded for 1-year pre- and 1-year post-harvest.

1780 *LW*

1781 Table 10. Treatment and responses for selected publications investigating Large Wood relevant to  
 1782 Q1a.

Reference	Treatment	Response
Benda et al. (2016)	<b>simulation modeling</b> of single entry thinning with and without a 10 m width no-cut buffers; and a double entry thinning occurring 25 years after first with and without 10 m no-cut buffers	<u>Simulated 100-year post harvest results</u> The model output for single entry thinning treatments predicts a 33% or 66% reduction of in-stream wood over a century relative to the unharvested reference for harvest on one side or both sides of the stream, respectively. Double entry thinning treatments without a buffer predicted further reduction in wood recruitment over a century of simulation with 42 and 84% reduction of in stream wood relative to the reference stream when one side and both sides of the channel were harvested.
Schuett Hames and Stewart (2019a)	30-ft no-cut buffer width, and thinning 30-75 ft from the stream (SR); retention of all shade providing trees in this area (AAS)	<u>5-years post-harvest</u> The SR and AAS LW input rates by volume were nearly 300% and 50% higher than the reference stream rates, respectively. <b>Wood recruitment in the SR sites was significantly greater than in the AAS and reference sites.</b> Conversely, differences in wood recruitment did not differ significantly between the AAS and reference sites.

1783

1784 Benda et al. (2016) used simulation modeling to estimate the changes in in-stream LW volume  
 1785 over time between sites with thinning treatments and unharvested reference sites. They used  
 1786 ORGANON growth models to simulate forest growth and LW recruitment over a 100-year  
 1787 period. The model simulated treatments of single entry thinning from below (thinning from  
 1788 below removes the smallest trees to simulate suppression mortality) with and without a 10 m  
 1789 width no-cut buffers; and a double entry thinning from below with the second thinning occurring  
 1790 25 years after the first with and without 10 m no-cut buffers (results with 10 m buffer presented  
 1791 in question 1b). Each thinning treatment was also combined with some mechanical introduction  
 1792 of thinned trees into the stream encompassing a range between 5 and 20 % of the thinned trees.  
 1793 The single-entry thin reduces stand density to 225 tph in 2015 (-67 %) and declines further to  
 1794 160 tph by 2110 (-77 %). The double entry thinning resulted in 123 tph after the second thinning  
 1795 in 2040 (-82%) and maintained that density until 2110. Both thinning treatments resulted in a

1796 substantial reduction of dead trees that could contribute to in-stream wood loads. The model  
1797 output for single entry thinning treatments predicts a 33% or 66% reduction of in-stream wood  
1798 over a century relative to the unharvested reference for harvest on one side or both sides of the  
1799 stream, respectively. Including mechanical tipping of 5,10,15, and 20% of cut stems without a  
1800 buffer in the single-entry thinning treatment changes the relative in-stream percentages of wood  
1801 relative to the reference stream to -15, -6, +1, and +6%, respectively. Double entry thinning  
1802 treatments without a buffer predicted further reduction in wood recruitment over a century of  
1803 simulation with 42 and 84% reduction of in stream wood relative to the reference stream when  
1804 one side and both sides of the channel were harvested. To offset the predicted changes of in  
1805 stream wood volume following double entry harvest would require tipping of 10% of cut stems.  
1806 The authors conclude that thinning without some mitigation efforts resulted in large losses of in  
1807 stream wood over a century.

1808 Schuett Hames and Stewart (2019a) compared recruitment rates of LW and volume of in-stream  
1809 LW between different riparian buffer thinning treatments and unharvested reference sites.  
1810 Treatments evaluated included prescriptions for standard shade rule (a 30-ft no-cut buffer width,  
1811 and thinning 30-75 ft from the stream), and all available shade rule (requires retention of all  
1812 shade providing trees in this area) for eastern Washington. Results showed cumulative wood  
1813 recruitment from tree fall over the five-year post-harvest interval was highest in the standard  
1814 shade rule (SR) group, lower in the all-available-shade rule (AAS) group and lowest in the  
1815 reference (REF) group. The SR and AAS rates by volume were nearly 300% and 50% higher  
1816 than the REF rates, respectively. Wood recruitment in the SR sites was significantly greater than  
1817 in the AAS and reference sites ( $P < 0.05$ ). Conversely, differences in wood recruitment did not  
1818 differ significantly between the AAS and reference sites.

#### 1819 *Sediment*

1820 Karwan et al. (2007) used BACI analysis to compare changes in total suspended solid (TSS)  
1821 yields between thinned sites and unharvested reference sites. This study was conducted in the  
1822 Mica Creek Experimental watershed of northern Idaho and focused on non-fish-bearing  
1823 headwater streams. The thinning treatment included a target 50% canopy removal without no-cut  
1824 buffers. Results showed a marginally significant ( $P = 0.081$ ) increase in TSS relative to the  
1825 reference streams in the first year following treatment. However, differences in TSS between the  
1826 treatment streams and reference streams were not significant ( $p = 0.174$ ) in the period 2-4 years  
1827 post-harvest.

#### 1828 *Nutrients*

1829 Yang et al. (2021) compared changes in stream chemistry between streams along thinned stands  
1830 and unharvested reference stands in young mixed conifer headwater basins of the Sierra National  
1831 Forest. Thinning treatment included mastication of shrub cover to  $< 10\%$  and harvesting of trees  
1832 to a target basal area of  $27\text{--}55 \text{ m}^2 \text{ ha}^{-1}$ . Data for dissolved organic carbon (DOC) and dissolved  
1833 organic nitrogen (DON) were recorded for 2 years prior to and 3 years after treatment. For  
1834 stream water, volume-weighted concentrations of DOC were 66- 94% higher in thinned  
1835 watersheds than in control watersheds for all three consecutive drought years following thinning  
1836 ( $p = 0.06, 0.01, \text{ and } 0.05$  for years 1,2, and 3 post-harvest, respectively). No differences in DOC

1837 concentrations were found between thinned and control watersheds before thinning ( $p = 0.50$ ,  
 1838 and 0.74 for pre-harvest years 1 and 2, respectively). Volume-weighted concentrations of DIN  
 1839 were 24% higher in thinned than in control watersheds only in the third year following thinning  
 1840 ( $p = 0.04$ ). No differences in DIN were detected between treatment and reference streams in the  
 1841 2 pre-harvest years ( $P \geq 0.44$ ). Note: Drought occurred at both sites during the three post-harvest  
 1842 years which may have compounded these effects. This is discussed in more detail in question 3.

1843

1844 **Focal Question 1b**

1845 *1b. How do buffer widths and adjacent upland timber harvest prescriptions influence impacts of*  
 1846 *riparian thinning treatments?*

1847 An experimental design that could provide information useful in answering this question would  
 1848 involve a comparison of sites with different buffer widths, all with upland harvest, and data  
 1849 would need to be recorded before and after thinning, with or without a control site (BAI, BACI),  
 1850 or differences after thinning between treatment and control sites (ACI). Three papers include an  
 1851 experimental design that investigate different buffer widths or different upland treatments along  
 1852 with riparian thinning treatments.

1853 *Shade*

1854 Anderson et al. (2007) compared changes in canopy cover at stream centers between sites  
 1855 adjacent to different riparian zone treatments and an untreated control. This study was conducted  
 1856 in young headwater forests of western Oregon. Treatments included three buffer widths (1) one  
 1857 site-potential tree averaging 69 m (B1), (2) variable width buffer averaging 22 m (VB), or (3)  
 1858 streamside retention buffer averaging 9 m (SR-T); the adjacent upland to each buffer was thinned  
 1859 to ~198 trees per hectare. Results showed that visible sky at stream center only differed  
 1860 significantly between SR-T (9.6%) and the untreated (4.2%) sites post-harvest. These results  
 1861 were reported for the period 2-5 years post-harvest.

1862 *LW*

1863 Table 11. Treatment and responses for selected publications investigating Large Wood relevant to  
 1864 Q1b.

Reference	Treatment	Response
Burton et al. (2016)	Buffer widths were 6, 15, or 70 meters and upland thinning was to 200 trees per ha (tph); unthinned reference stand of ~400 tph.	<u>5 years post-harvest</u> slightly higher volumes of wood were found in sites with a narrow 6-m buffer (not significant), as compared with the 15-m and 70-m buffer sites 5 years after harvest.

Benda et al. (2016)	simulation modeling of thinning from below with and without a 10 m width no-cut buffers;	<u>Simulated 100-year post harvest results</u> Adding a 10 m buffer reduced total reduction of in stream wood to 11 and 22% for thinning on one and both sides of the channel, respectively, from the predicted 42 and 84% reduction without the 10 m buffer.
---------------------	--	--

1865

1866 Burton et al. (2016) examined the relationship between annual in-stream wood loading and  
 1867 riparian buffer widths adjacent to upland thinning operations. Buffer widths were 6, 15, or 70  
 1868 meters and upland thinning was to 200 trees per ha (tph), with a second thinning (~10 years later)  
 1869 to ~85 tph, alongside an unthinned reference stand of ~400 tph. Their results showed that slightly  
 1870 higher volumes of wood were found in sites with a narrow 6-m buffer (not significant), as  
 1871 compared with the 15-m and 70-m buffer sites in the first 5 years after the first harvest and  
 1872 maintained through year 1 of the second harvest (end of study). The authors attributed this  
 1873 difference to a higher likelihood of logging debris and/or windthrow, but these factors were not  
 1874 analyzed.

1875 Benda et al. (2016) used simulation modeling to estimate the changes in in-stream LW volume  
 1876 over time between sites with thinning treatments and unharvested reference sites. They used  
 1877 ORGANON growth models to simulate forest growth and LW recruitment over a 100-year  
 1878 period. The model simulated treatments of single entry thinning from below (thinning from  
 1879 below removes the smallest trees to simulate suppression mortality) with and without a 10 m  
 1880 width no-cut buffers; and a double entry thinning from below with the second thinning occurring  
 1881 25 years after the first with and without 10 m no-cut buffers. Each thinning treatment was also  
 1882 combined with some mechanical introduction of thinned trees into the stream encompassing a  
 1883 range between 5 and 20 % of the thinned trees. The single-entry thin reduces stand density to 225  
 1884 tph in 2015 (-67 %) and declines further to 160 tph by 2110 (-77 %). The double entry thinning  
 1885 resulted in 123 tph after the second thinning in 2040 (-82%) and maintained that density until  
 1886 2110. Both thinning treatments resulted in a substantial reduction of dead trees that could  
 1887 contribute to in-stream. The model output for single entry thinning treatments predicts a 33% or  
 1888 66% reduction of in-stream wood over a century relative to the unharvested reference for harvest  
 1889 on one side or both sides of the stream, respectively. Adding the 10-m no cut buffer reduced total  
 1890 loss to 7 and 14%. Including mechanical tipping of 5,10,15, and 20% of cut stems without a  
 1891 buffer in the single-entry, thinning treatment changed the relative in-stream percentages of wood  
 1892 relative to the reference stream to -15, -6, +1, and +6%, respectively. To completely offset the  
 1893 loss of in stream wood due to single entry thinning, mechanical tipping of 14 and 12% were  
 1894 required without and with buffers. Double entry thinning treatments without a buffer predicted  
 1895 further reduction in wood recruitment over a century of simulation with 42 and 84% reduction of  
 1896 in stream wood relative to the reference stream when one side and both sides of the channel were  
 1897 harvested. Adding a 10 m buffer reduced total reduction of in stream wood to 11 and 22% for  
 1898 thinning on one and both sides of the channel. To offset the predicted changes of in stream wood  
 1899 volume following double entry harvest would require tipping of 10 and 7% of cut stems without

1900 and with the 10-m buffer. The authors conclude that thinning without some mitigation efforts  
 1901 resulted in large losses of in stream wood over a century.

1902

1903 [Focal Question 1c](#)

1904 *1c. What are the effects of clearcut gaps in riparian stands (intensity, extent) on the riparian*  
 1905 *functions, over the short and long-term, compared to untreated stands?*

1906 This question uses the general term “clearcut gaps” as a treatment within the riparian area but  
 1907 does not define a minimum or maximum threshold for gap size. Thus, studies reviewed that used  
 1908 a “patch” treatment were included as having information useful in answering this question. The  
 1909 question also identifies a comparison with untreated stands. Therefore, any design with a control  
 1910 site (BACI, ACI) is appropriate.

1911 There appears to be a paucity of studies in the literature that investigate the effects of gaps or  
 1912 patch harvesting treatments on riparian function within riparian stands. Only 4 papers discussed  
 1913 the effects of prescribed gaps or patches in the riparian area on riparian function.

1914 The “Hard Rock” study from McIntyre et al. (2021) and the “Soft Rock” study from Ehinger et  
 1915 al. (2021) present the most relevant results useful for answering this question. Riparian buffer  
 1916 prescriptions for non-fish-bearing streams in western Washington use a gap design. In this  
 1917 design, a 50-foot buffer is required along at least 50% of the treated stream length. The  
 1918 remaining 50% or less of the treated riparian management zone can be clear cut to the stream  
 1919 edge. The Hard Rock study compared differences in shade, in-stream sediment and nutrient  
 1920 concentrations, and large wood recruitment between treated and unharvested basins for 8-9 years  
 1921 post-harvest. The Soft Rock study compared differences in the same functions between treated  
 1922 and unharvested basins, with 3-6 years of post-harvest sampling depending on the function under  
 1923 investigation. The first phase of the Hard Rock study (McIntyre et al. 2018) also compared  
 1924 differences in litter inputs following treatment for 2 years post-harvest between treatment and  
 1925 reference reaches.

1926 *Shade*

1927 Table 12. Treatment and responses for selected publications investigating Shade relevant to Q1c.

1928

Reference	Treatment	Response
McIntyre et al. (2021)	Buffer width of 50 feet (100%); 50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)	9-years post-harvest Effective shade results showed decreases of 11, 36, and 74 percent in the 100%, FP, and 0% treatments, respectively. Results for canopy cover showed that riparian cover declined after harvest in all buffer treatments reaching a minimum around 4 years post-harvest (after mortality stabilized).

Janisch et al. (2012)	Patched buffer: clearcut to stream with ~50-110 m patches retained; continuous buffer 10-15 m	<u>1-year post-harvest</u> After treatment, canopy cover in the clearcut catchments averaged 53%, canopy cover in the patch buffer treatment averaged 76%, and canopy cover in the continuous buffer treatment averaged 86%. The canopy cover of the clearcut and patch buffer treatments were significantly lower than in the reference streams.
Swartz et al. (2020)	20 m diameter clearcut gaps over stream at 30 m intervals.	<u>1-year post-harvest (n = 6)</u> Post-harvest significant increase in mean reach light to a mean of 3.91 (SD ± 1.63) moles of photons m <sup>-2</sup> day <sup>-1</sup> , overall resulting in a mean change in light of 2.93 (SD ± 1.50) moles of photons m <sup>-2</sup> day <sup>-1</sup> . The areas beneath each gap had notable localized declines in shade, though the entirety of the treatment reach (100 m) mean shading declined by only 4% (SD ± 0.02%).

1929

1930 The Hard Rock study reported significant decreases in canopy cover (measured at 1 meter above  
 1931 the stream surface with a spherical densiometer) for all treated sites immediately following  
 1932 harvest compared to the reference sites ( $p < 0.05$ ). The mean canopy cover decreased from 96%  
 1933 (pre-harvest) to 72% in the first-year post-harvest and continued to decline for four years  
 1934 reaching a minimum of 54%. After year four, mean canopy cover began to recover increasing  
 1935 annually until year 9 to 74%. In contrast, mean canopy cover in the reference sites was 95%  
 1936 before harvest and never fell below 85% for 9 years. In the Soft Rock study, mean canopy  
 1937 closure decreased in the treatment sites from 97% in the pre-harvest period to 75%, 68%, and  
 1938 69% in the first, second, and third post-harvest years, respectively; and was further related to the  
 1939 proportion of stream buffered and to post-harvest windthrow within the buffer. Canopy closure  
 1940 remained stable in the reference sites throughout the course of the study, ranging from 95 to  
 1941 99%.

1942 Janisch et al. (2012) compared canopy cover before and after application of a “patched buffer”  
 1943 treatment with unharvested control reaches in headwater streams of western Washington. The  
 1944 “patched buffer” treatment included retention of portions of the riparian forests ~50-110 m long  
 1945 in distinct patches along the channel with the remaining riparian area clearcut. There was no  
 1946 standard width for patched buffers, with buffers spanning the full width of the floodplain area  
 1947 and/or extending some undefined distance away from the stream. Canopy density was measured  
 1948 once in the summer prior to logging and once in the summer following logging. The percentage  
 1949 of visible sky was determined from digital photos taken with a fish-eye lens using Hemiview  
 1950 Canopy Analysis software. Canopy cover in all streams averaged 95% prior to harvest and did  
 1951 not differ between treatment and reference streams. Following treatment, canopy cover in the  
 1952 patch buffer treatment averaged 76% and differed significantly from reference reaches.

1953 Swartz et al. (2020) tested the effects of adding canopy gaps within young (40 – 60 years old),  
 1954 regenerating forests of western Oregon on stream light availability and stream temperatures.  
 1955 While light availability and stream temperature are not functions described in the FPHCP, they  
 1956 are directly related to shade (an FPHCP function) Also, they directly affect water quality and  
 1957 aquatic habitat productivity which are functional objectives within the FPHCP. Further,  
 1958 considering the paucity of studies available that investigate the effects of clearcut gaps, the

1959 results are presented here. The addition of gaps in the young regenerating forests were used to  
1960 theoretically mimic the natural disturbance regimes and the higher canopy complexity of late-  
1961 successional forests. The researchers used a BACI design on six replicated streams within the  
1962 McKenzie River Basin. In each treatment reach, gaps were designed to create openings in the  
1963 canopy that were approximately 20 m in diameter. Gaps were centered on a tree next to the  
1964 stream and spaced approximately 30 meters apart along each reach. The BACI analysis showed  
1965 strong evidence for significant increase in mean reach light ( $p < 0.01$ ) up to 3.91 (SD  $\pm$  1.63)  
1966 moles of photons  $m^{-2} day^{-1}$  and an overall mean change in light of 2.93 (SD  $\pm$  1.50) moles of  
1967 photons  $m^{-2} day^{-1}$ . Mean stream shading could not be evaluated in the full BACI analysis  
1968 because post-treatment hemispherical photographs could not be taken at all sites due to fire  
1969 impeding access. For the remaining sites, the areas beneath each gap had notable localized  
1970 declines in shade, though the entirety of the treatment reach mean shading declined by only 4%  
1971 (SD  $\pm$  0.02%).

#### 1972 *Litter*

1973 The Hard Rock study only quantified changes in litter input for 2 years after treatment (McIntyre  
1974 et al., 2018). While significant decreases in litter input were observed from pre- to post-harvest  
1975 in the treatment sites (described in focal question 1) these values were not significant when  
1976 compared to the changes in the reference sites. Litter input was not collected in the Soft Rock  
1977 study.

#### 1978 *LW*

1979 For the Hard Rock study, large wood recruitment and loading were only compared between the  
1980 reference reaches and the buffered portion of the treatment reaches. The authors report large  
1981 wood recruitment into the channel was 3 times greater on average in the treatment buffer than in  
1982 the reference over the 8-year post-treatment period. However, while considerable, these  
1983 differences were not significant for any analyzed post-harvest interval (e.g., 1-2 years post, 1-5  
1984 years post, or 1-8 years post). The lack of significance was attributed to the large variability in  
1985 recruitment values among treatment sites. The greatest increase in LW recruitment in the  
1986 treatment sites relative to the reference sites occurred in the first 2 years post-harvest. Large  
1987 wood loading (pieces/m of channel length) increased significantly ( $\alpha = 0.10$ ) in the treatment  
1988 reaches, relative to the reference sites in the first 2 years (47%;  $p = 0.05$ ), 5 years (42%;  $p =$   
1989 0.08), and 8 years (41%;  $p = 0.09$ ) post-harvest. For the Soft Rock study there was little post-  
1990 harvest large wood input in reference sites: an average of 4.3 pieces and 0.34  $m^3$  of combined in-  
1991 and over-channel volume per 100 m of channel. In contrast, the full buffer sites and <50 ft buffer  
1992 sites received an average of 23 and 10 pieces/100 m and 2.3 and 0.7  $m^3/100$  m of large wood,  
1993 respectively.

#### 1994 *Sediment*

1995 For the Hard Rock study, results for water turbidity and suspended sediment export (SSE) were  
1996 stochastic in nature and the relationships between SSE and treatment effects were not strong  
1997 enough to confidently draw conclusions. Water turbidity and SSE increased with stream  
1998 discharge during large storm events but rapidly declined. The Soft Rock study reported similar

1999 issues with the data for SSE in that it appeared to be driven by site and event specific factors and  
2000 strong conclusions could not be drawn. The authors report that the softer lithologies sampled as  
2001 part of this study were more erodible than the competent lithologies sampled in the companion  
2002 Hard Rock Study.

### 2003 *Nutrients*

2004 The Hard Rock study analyzed changes in total nitrogen and nitrate export in the gap buffers  
2005 relative to untreated reference streams. Results showed an increase in total nitrogen export in the  
2006 treatment sites of 10.85 kg/ha/yr ( $p = 0.006$ ) in the first two years post-harvest relative to the  
2007 reference sites. In the extended periods, total nitrogen export increased by 5.34 ( $p = 0.147$ )  
2008 kg/ha/yr relative to the reference streams. Results for  $\text{NO}_3$  export showed similar but slightly  
2009 lower increases than total nitrogen with a relative increase in  $\text{NO}_3$  export of 9.63 ( $p = 0.004$ )  
2010 kg/ha/yr for the first two years post-harvest relative to the reference. None of the changes in  
2011 nitrate exports in the extended period were significant. The Soft Rock study reported significant  
2012 increases in concentrations of total nitrogen ( $p < 0.05$ ) and  $\text{NO}_3$  ( $p < 0.05$ ) post-harvest in the  
2013 treatment sites relative to the reference sites. The change in export appeared related to the  
2014 proportion of stream buffered.

2015

### 2016 [Focal Question 1d](#)

2017 *1d. How do buffer widths and upland timber harvest influence impacts of clearcut gaps*  
2018 *treatments?*

2019 The wording of this question implies that the effects of clearcut gaps (discussed in focal question  
2020 1c) on riparian function could be impacted when paired with different buffer widths and upland  
2021 harvest prescriptions. Similar to the results of the search in literature for focal question 1c, there  
2022 was a paucity of riparian function studies that implemented a clearcut gap or patch cutting  
2023 method within the riparian area. The added layer of complexity in this question specifying  
2024 differences in buffer widths and upland harvests only further refined the selection of appropriate  
2025 papers. Of the studies reviewed above, none included the evaluation of different buffer widths or  
2026 different upland harvests in their experimental design. The Hard Rock study compared the  
2027 clearcut gap buffers to full retention buffer and unbuffered sites (discussed in the literature  
2028 review section), but different widths were not compared in the gap buffer treatments.

2029

### 2030 [Focal Question 1e](#)

2031 *1e. What are the effects of any combinations of the above treatments?*

2032 No studies found in our search compared the effects of combined treatments on one or more of  
2033 the five functions, likely because combining multiple treatments into one design has the potential  
2034 to confound results and are difficult to implement with sufficient sample sizes. The majority of  
2035 the studies listed in our review investigate the effects of buffer width, thinning treatments, and  
2036 upland treatments separately.



2037 The only papers with some extractable evidence of the compounding/ameliorating effects of  
 2038 combined treatments were focused on shade. One study, Reiter et al. (2020), compared the  
 2039 effects of thinned and unthinned buffers, and clearcut on changes in percent shade over adjacent  
 2040 streams (discussed in focal question 1). However, changes in shade were not statistically  
 2041 analyzed and the implementation of the upland thinning treatment only occurred at one site  
 2042 (Table 6).

2043

2044 [Focal Question 2](#)

2045 *2. How and to what degree do specific site conditions (e.g., topography, channel width and*  
 2046 *orientation, riparian stand age and composition) influence the response of the riparian*  
 2047 *functions?*

2048 Multiple studies have investigated the influences of site conditions on riparian function. Few  
 2049 studies reviewed (4) investigated the interaction between specific site conditions (e.g., slope,  
 2050 lithology, elevation) and harvest on the response of riparian function. However, if these specific  
 2051 site conditions influence the magnitude of riparian function in the absence of harvest, it is  
 2052 possible they can compound the effects of harvest on their response. Thus, studies that assess the  
 2053 relationship between site factors and riparian function may provide some useful insight for  
 2054 management and are presented below. Further, we also included studies that investigated the  
 2055 relationships between road development and sediment transport because road development is  
 2056 directly related to changes in local topography.

2057 *Litter*

2058 Table 13. Treatment and responses for selected publications investigating Litter relevant to Q2.

Reference	Treatment	Response
Hart et al. (2013)	Remove plants in a 5 x 8 m section adjacent to stream < 10 cm DBH and >12 cm height every 2 months. 5 m fence extending underground and parallel to the stream	<u>1-2 years post-treatment (n = 5)</u> . Deciduous-site vertical litter input (504 g m <sup>-1</sup> y <sup>-1</sup> ) exceeded that from coniferous sites (394 g m <sup>-1</sup> y <sup>-1</sup> , 336.4–451.7) by 110 g/m <sup>2</sup> (28.6–191.6) over the full year. Annual lateral inputs at deciduous sites (109 g m <sup>-1</sup> y <sup>-1</sup> ) were 46 g/m more than at coniferous sites (63 g m <sup>-1</sup> y <sup>-1</sup> ). Lateral inputs calculated for a 3-m-wide stream accounted for 9.6% (5.4–12.5) of total annual inputs at coniferous sites and 12.7% (10.2–14.5) of total inputs at deciduous sites. The strongest deciduous inputs to streams occurred in November. Annual lateral litter input <b>increased with slope at deciduous sites</b> (R <sup>2</sup> = 0.4073) but showed <b>no strong relationship at coniferous sites</b> (R <sup>2</sup> = 0.1863).
Bilby & Heffner (2016)	<b>Simulation modeling</b> and field sampling	<u>1-year of litterfall data</u> the majority of the litter recruited into the stream originated from within 10 m of the stream regardless of litter or stand type. No difference was found in delivery distance and litter type (needles or broadleaf) at young sites (ages not specified; canopy height mean = 32.4 m). However, needles released at mature (canopy height mean = 47 m) sites had a higher proportion of cumulative input from greater distances than needles or alder leaves released at younger sites. Litter travel distance was linearly

		related to wind speed ( $p < 0.0001$ ). Doubling wind speed at one site led to a 67-87% expansion of the riparian contribution zone in the study area.
--	--	--

2059

2060 Hart et al. (2013) compared litter delivery into streams between riparian zones dominated by  
 2061 deciduous (red alder) and coniferous (Douglas-fir) tree species in western Oregon. Results from  
 2062 this study show that deciduous forests dominated by red alder delivered significantly greater  
 2063 vertical and lateral inputs ( $\text{g m}^{-2} \text{y}^{-1}$ ) to adjacent streams than did coniferous forests dominated by  
 2064 Douglas-fir. Deciduous-site vertical litter input (mean =  $504 \text{ g m}^{-2} \text{y}^{-1}$ ) exceeded that from  
 2065 coniferous sites ( $394 \text{ g m}^{-2} \text{y}^{-1}$ ) by  $110 \text{ g/m}^2$  over the full year. Annual lateral inputs at  
 2066 deciduous sites ( $109 \text{ g m}^{-2} \text{y}^{-1}$ ) were  $46 \text{ g m}^{-2} \text{y}^{-1}$  more than at coniferous sites ( $63 \text{ g m}^{-2} \text{y}^{-1}$ ).  
 2067 The timing of the inputs also differed, with the greatest differences occurring in November  
 2068 during autumn peak inputs for the deciduous forests. Further, annual lateral litter input increased  
 2069 with slope at deciduous sites ( $R^2 = 0.4073$ ,  $p = 0.0771$ ), but showed no strong relationship at  
 2070 coniferous sites ( $R^2 = 0.1863$ ,  $p = 0.2855$ ). These results were partially consistent with Bilby &  
 2071 Heffner (2016) in that they suggest litter type, and topography (slope) can affect the litter input  
 2072 rates.

2073 Bilby & Heffner (2016) used a combination of field experiments, literature review, and modeling  
 2074 to estimate the relative importance of factors affecting litter delivery from riparian areas into  
 2075 streams of western Washington in the Cascade mountains at high and low elevations. Their  
 2076 results for conifer needles released at mature sites had a higher proportion of cumulative input  
 2077 from greater distances than needles or leaves released at younger sites. The authors suggest from  
 2078 their interpretation of the model that the width of the litter contributing area was  $\sim 35\%$  greater at  
 2079 mature sites than at young sites. The mean age of “mature” and “young” sites was not specified  
 2080 but the mean tree heights were  $47.0 \text{ m}$  and  $32.4 \text{ m}$  for the mature and young sites, respectively.  
 2081 Thus, tree height is related to the width of the litter contributing area for conifer needles. Litter  
 2082 travel distance was also linearly related to wind speed ( $p < 0.0001$ ). Doubling wind speed at one  
 2083 site led to a 67-87% expansion of the riparian litter contribution zone in the study area.  
 2084 Interpretation of the regression curves revealed a trend that suggests hillslope gradient affects the  
 2085 width of the litter contributing area as well. However, the authors did not apply statistical  
 2086 analysis to these values and only speculated that increasing the slope from 0-45% would increase  
 2087 the width of the litter contributing area by up to 70%.

2088 *LW*

2089

2090

2091 Table 14. Treatment and responses for selected publications investigating Large Wood relevant to  
 2092 Q2.

Reference	Treatment	Response
Wing & Skaugset (2002)	Relationships between channel and habitat characteristics with LW piece count and volume	<u>Observation data from in 3793 stream reaches in western Oregon State.</u> <b>LW volume:</b> reaches with < 2.3% gradient averaged 5.8 m <sup>3</sup> while higher <b>gradient</b> streams averaged 17.9 m <sup>3</sup> per reach for all land types (ownership, forested and non-forested). Reaches with gradients less than 4.7% averaged LW volume of 11.5 m <sup>3</sup> , while mean volume at higher gradient reaches was 25.2 m <sup>3</sup> . <b>LW pieces:</b> Streams <12.2 m <b>bank full width</b> averaged 11.1 LW pieces per reach while larger channels averaged 4.9 pieces per reach. For key LW pieces (logs at least 0.60 m in diameter and 10 m long), <b>stream gradient</b> was again most important. gradient < 4.9% averaged 0.5 key LW pieces per reach while streams with higher gradients averaged 0.9 key LW pieces per reach.
Sobota et al. (2006)	patterns of riparian tree fall directions	<u>Data was collected from 21 field sites</u> Projections of LW recruitment estimated that sites with uniform steep side slopes (>40%) produced between 1.5 to 2.4 times more in stream LW by number of tree boles than sites with uniform moderate side slopes (< 40%).

2093

2094 Wing & Skaugset (2002) investigated the relationships between channel and habitat  
 2095 characteristics with LW piece count and volume in stream reaches in western Oregon. This study  
 2096 analyzed an extensive spatial database of aquatic habitat conditions created for western Oregon  
 2097 using stream habitat classification techniques and a geographic information system (GIS).  
 2098 Regression tree analysis (an exploratory regression analysis that allows for the inclusion of  
 2099 multiple explanatory variables) was used to compare the relative strength of each variable in  
 2100 predicting LW volume. Explanatory variables used in this analysis included morphology of  
 2101 active channel (hillslope, terrace, terrace hillslope, unconstrained), and lithology (e.g., alluvium,  
 2102 basalt, etc.). Results for channel characteristics showed that stream gradient was the most  
 2103 important explanatory variable for LW volume. The split for stream gradient occurred for reaches  
 2104 with < 2.3% gradient (mean LW volume: 5.8 m<sup>3</sup> per reach) while higher gradient streams showed  
 2105 a mean LW volume of 17.9 m<sup>3</sup> per reach.  
 2106  
 2107 For LW pieces in forested stream reaches bankfull channel width was the most important  
 2108 explanatory variable with the split occurring for streams channels less than 12.2 m wide. LW  
 2109 pieces for streams <12.2 m wide averaged 11.1 LW pieces per reach while larger channels  
 2110 averaged 4.9 pieces per reach; in this model the BFW split explained 7% of the variation in LW  
 2111 pieces found in forested streams. For key LW pieces (logs at least 0.60 m in diameter and 10 m  
 2112 long) in forested reaches, stream gradient was again the most important explanatory variable

2113 with the split occurring at a slope of 4.9%. The streams with a gradient < 4.9% averaged 0.5 key  
 2114 LW pieces per reach while streams with higher gradients averaged 0.9 key LW pieces per reach;  
 2115 in this model stream gradient explained 8% of the variation in key LW pieces found in streams.  
 2116

2117 Lithology caused second, third or fourth level splits after stream gradient or BFW. Specifically,  
 2118 Mesozoic sedimentary and metamorphic geologies, located in southern Oregon stream reaches,  
 2119 were grouped and split from basalt, Cascade, and marine sedimentary geologies. In stream  
 2120 reaches with Mesozoic sedimentary and metamorphic geologies, the quantity of large wood was  
 2121 roughly half the amount found in other geologies. The only exception to this grouping was for  
 2122 LW volume in larger stream reaches, where basalt and marine sedimentary geologies contained  
 2123 more LW volume when grouped separately from all other geologies in a fourth-level split. The  
 2124 authors conclude that the geomorphic characteristic of stream reaches, in particular stream  
 2125 gradient and bankfull width, correlated best with LW presence.  
 2126

2127 Sobota et al. (2006), evaluated patterns of riparian tree fall directions in diverse environmental  
 2128 conditions and evaluate correlations with tree characteristics, forest structural variables, and  
 2129 topographic features. Specifically, the authors were interested in correlations between fall  
 2130 directionality and tree species type, tree size, riparian forest structure, and valley topography  
 2131 (side slope). Data was collected from 21 field sites located west of the Cascade Mountains crest  
 2132 (11 sites: Coast Range and west slopes of the Cascades), and in the interior Columbia Basin (10  
 2133 sites: east slopes of the Cascades, Blue Mountains, and Northern Rockies) of Oregon,  
 2134 Washington, Idaho, and Montana, USA. Streams were second- to fourth-order channels and had  
 2135 riparian forests that were approximately 40 to >200 years old. Model projections of LW  
 2136 recruitment estimated that sites with uniform steep side slopes (>40%) produced between 1.5 to  
 2137 2.4 times more in stream LW by number of tree boles than sites with uniform moderate side  
 2138 slopes (< 40%). The authors warn that while side slope categories (>40%, <40%) was the  
 2139 strongest predictor of tree fall direction in this study, they believe the differences in tree fall  
 2140 direction between these categories mainly characterized differences between fluvial (88% of  
 2141 moderate slope sites) and hillslope landforms (71% of steep slope sites). They suggest that the  
 2142 implications from this study are most applicable to small- to medium-size streams (second to  
 2143 fourth order) in mountainous regions where sustained large wood recruitment from riparian  
 2144 forest mortality is the significant management concern.

2145 *Sediment*

2146 Table 15. Treatment and responses for selected publications investigating Sediment relevant to  
 2147 Q2.

Reference	Treatment	Response
Bywater-Reyes et al. (2017)	basin lithology and physiography effects on sediment delivery	<u>6 years of data from the Trask River Watershed</u> <b>Site lithology</b> was the first order control over suspended sediment yield (SSY). SSY was greater in catchments underlain by Siletz Volcanics (r = 0.6), the Trask River Formation (r = 0.4), and landslide deposits (r = 0.9). There was a strong negative correlation of SSY with percent area underlain by diabase (r = -0.7), with the lowest SSY associated with 100% diabase

Bywater-Reyes et al. (2018)	catchment lithography, physiography, discharge, and disturbance history effects on sediment delivery	<u>60 years of data in the H.J. Andrews experimental watershed (n = 10)</u> Watershed <b>slope</b> variability combined with cumulative <b>annual discharge</b> explained 67% of the variation in annual sediment yield. When considering disturbance, the largest magnitude changes in sediment movement, were after <b>floods with a <math>\geq 30</math>-year return interval</b> .
Mueller & Pitlick (2013)	correlation analysis to assess the relative impact of lithology, basin relief, mean basin slope, and drainage density on in stream sediment supply.	<u>Data sets ranging 1-96 years for 83 basins</u> the strongest correlation of bankfull sediment concentration was with <b>basin lithology</b> , and showed little correlation strength with slope, relief and drainage density. As lithologies become dominated by softer parent materials (volcanic and sedimentary rocks), bankfull sediment concentrations increased by as much as 100-fold.
Litschert & MacDonald (2009)	Post-harvest stream sediment delivery pathway development frequency and characteristics.	<u>1-year post-harvest data (n = 200 harvest units)</u> 19 harvest units developed sediment delivery pathways. Pathway length and probability of connecting to stream was significantly correlated with <b>mean annual precipitation, cosine of the aspect, elevation, and hillslope gradient</b> .

2148  
2149

2150 Bywater-Reyes et al. (2017) assessed the influence of natural controls (basin lithology and  
2151 physiography) and forest management on suspended sediment yields in temperate headwater  
2152 catchments. This study analyzed 6 years of data from the Trask River Watershed in northeastern  
2153 Oregon and included data from harvested and unharvested sub-catchments underlain by  
2154 heterogenous lithologies. Results from this study indicate that site lithology was the first order  
2155 control over suspended sediment yield (SSY) with SSY varying by an order of magnitude across  
2156 lithologies observed. Specifically, SSY was greater in catchments underlain by Siletz Volcanics  
2157 ( $r = 0.6$ ), the Trask River Formation ( $r = 0.4$ ), and landslide deposits ( $r = 0.9$ ) and displayed an  
2158 exponential relationship when plotted against the percentage of watershed area underlain by  
2159 these lithologies. In contrast, site lithology had a strong negative correlation with percent area  
2160 underlain by diabase ( $r = 0.7$ ), with the lowest SSY associated with 100% diabase. Following  
2161 timber harvest, increases in SSY occurred in all harvested catchments but returned to pre-harvest  
2162 levels within 1 year except for sites that were underlain by sedimentary formations and were  
2163 clearcut without protective buffers. The authors conclude that sites underlain with a friable  
2164 lithology (e.g., sedimentary formations) had on average, SSYs an order of magnitude higher  
2165 following harvest than those on more resistant lithologies (intrusive rocks).

2166 Bywater-Reyes et al. (2018) quantified how sediment yields vary with catchment lithography and  
2167 physiography, discharge, and disturbance history (management or natural disturbances) over 60  
2168 years in the H.J. Andrews experimental watershed in the western Cascade Range of Oregon. A  
2169 linear mixed effects model (log transformed to meet the normality assumption) was used to  
2170 predict annual sediment yield. In this model, site was treated as a random effect while discharge  
2171 and physiographic variables were treated as fixed variables. This allowed for the evaluation of  
2172 the relationships between sediment yield and physiographic features (slope, elevation, roughness,

2173 and index of sediment connectivity) while accounting for site. To account for the effect of  
2174 disturbance history a variable was added to the model when the watershed had a history of  
2175 management or natural disturbances. If the models for the disturbed watersheds significantly  
2176 underpredicted the sediment discharge, the timing of the sudden increases were further examined  
2177 to assess whether it correlated with a disturbance event. The results showed that watershed  
2178 physiography combined with cumulative annual discharge explained 67% of the variation in  
2179 annual sediment yield across the 60-year data set. Relative to other physiographic variables,  
2180 watershed slope was the greatest predictor of annual suspended sediment yield. However, the  
2181 results showed that annual sediment yields also moderately correlated with many other  
2182 physiographic variables and caution that the strong relationship with watershed slope is likely a  
2183 proxy for many processes, encompassing multiple catchment characteristics.

2184 Mueller & Pitlick (2013) used correlation analysis to assess the relative impact of lithology,  
2185 basin relief, mean basin slope, and drainage density on in stream sediment supply defined by the  
2186 bankfull sediment concentration (bedload and suspended load). The study used sediment  
2187 concentration data from 83 drainage basins in Idaho and Wyoming. Lithologies of the study area  
2188 were divided into four categories ranging from hardest to softest- granitic, metasedimentary,  
2189 volcanic, and sedimentary. The results showed the strongest correlation of bankfull sediment  
2190 concentration was with basin lithology, and showed little correlation strength with slope, relief  
2191 and drainage density. As lithologies become dominated by softer parent materials (volcanic and  
2192 sedimentary rocks), bankfull sediment concentrations increased by as much as 100-fold. The  
2193 authors interpret these results as evidence that lithology can be more important in estimating  
2194 sediment supply than topography.

2195 Rachels et al. (2020) used sediment source fingerprinting techniques to quantify the proportional  
2196 relationship of sediment sources (hillslope, roads, streambanks) in harvested and un-harvested  
2197 watersheds of the Oregon Coast Range. The study included one catchment (Enos Creek) that was  
2198 partially clearcut harvested in the summer of 2016 and an unharvested reference catchment  
2199 (Scheele Creek) located ~3.5 km northwest of Enos Creek. The paired watersheds had similar  
2200 road networks, drainage areas, lithologies and topographies. The treatment watershed was  
2201 harvested with a skyline buffer technique in the summer of 2016 under the Oregon Forest  
2202 practices Act policy that requires a minimum 15 m no-cut buffer. The proportion of suspended  
2203 sediment sources were similar in the harvested ( $90.3 \pm 3.4\%$  from stream bank;  $7.1 \pm 3.1\%$  from  
2204 hillslope) and unharvest ( $93.1 \pm 1.8\%$  from streambank;  $6.9 \pm 1.8\%$  from hillslope) watersheds.  
2205 However, the harvested watershed contained a small portion of sediment from roads ( $3.6 \pm$   
2206  $3.6\%$ ), while the unharvested reference watershed suspended sediment contained no sediment  
2207 sourced from roads. In the harvested watersheds the sediment mass eroded from the general  
2208 harvest areas ( $96.5 \pm 57.0$  g) was approximately 10 times greater than the amount trapped in the  
2209 riparian buffer ( $9.1 \pm 1.9$  g), and 4.6 times greater than the amount of sediment collected from  
2210 the unharvested hillslope ( $21.0 \pm 3.3$  g). These results suggest that the riparian buffer was  
2211 efficient in reducing sediment erosion relative to the harvested area. The caveat of this study was  
2212 the limited sample size (1 treatment, 1 paired reference watershed) and does not incorporate the  
2213 effects of different watershed physiography on sediment erosion. However, it is presented here as

2214 evidence that the formation of roads within a riparian area may interact with timber harvest to  
2215 increase the potential flow of sediments from roads.

2216 Litschert & MacDonald, (2009) investigated the frequency of sediment delivery pathways in  
2217 riparian management areas and their physical characteristics and connectivity following harvest.  
2218 In this study the authors describe sediment delivery pathways (“features”) as rills, gullies, and  
2219 sediment plumes that form when excess sediment relative to overland flows transports sediment  
2220 from the hillslope to the stream. The authors surveyed 200 riparian management areas (RMA) in  
2221 four different National Forests of the Sierra Nevada and Cascade Mountains of California. USFS  
2222 policy requires 90-m wide RMA along each side of perennial streams and 45-m wide RMA along  
2223 each side of all ephemeral and intermittent streams. When features were found within an RMA,  
2224 data for years since harvest, soil depth, soil erodibility (K), feature length, feature gradient,  
2225 aspect, elevation, hillslope gradient, hillslope curvature, surface roughness, and connectivity  
2226 were recorded for analysis. Association between these variables were analyzed with a  
2227 Spearman’s rank correlation. The variables most strongly associated with feature length were  
2228 used to develop a multiple linear regression model to predict feature length. Only 19 of the 200  
2229 harvest units had sediment development pathways. Feature pathways ranged in age (time since  
2230 harvest) from 2 to 18 years, and in length from 10 m to 220 m. Of the 19 feature pathways, only  
2231 six were connected to streams, and five of those originated from skid trails. Feature pathway  
2232 length was significantly related to mean annual precipitation, cosine of the aspect, elevation, and  
2233 hillslope gradient ( $R^2 = 64\%$ ,  $p = 0.004$ ). These results suggest that within treated riparian areas  
2234 topographic characteristics such as aspect, elevation and hillslope gradient can affect delivery of  
2235 sediment into streams.

2236 Rashin et al. (2006) evaluated the effectiveness of Washington State best management practices  
2237 (BMPs) for controlling sediment related water quality impacts. Although this study was  
2238 published in 2006, the data analyzed in this study were collected between 1992 and 1995. In their  
2239 evaluation, Rashin et al. (2006) assessed site erosion, sediment delivery, channel disturbance,  
2240 and aquatic habitat condition within the first two years of harvest along fish- and non-fish-  
2241 bearing streams across Washington state. From their results, the authors concluded that the site-  
2242 specific factors influencing the effectiveness of BMPs in preventing chronic sediment delivery  
2243 into streams were 1) the proximity of ground disturbance to the stream, 2) presence of a stream  
2244 buffer, 3) falling and yarding practices that minimized disturbance to stream channel, and 4)  
2245 timing of harvest activities for certain climate zones where frozen ground or snow cover may be  
2246 exploited. The landscape factors that influenced BMP effectiveness were 1) the density (specific  
2247 metric not reported) of unbuffered small streams at harvest sites, and 2) steepness of stream  
2248 valley slopes. The authors conclude with a recommendation of excluding timber falling and  
2249 yarding activities at least 10 m from streams and outside of steep inner gorges.

2250 From the studies reviewed there is evidence that sediment delivery into streams following timber  
2251 harvest is influenced by not only the intensity of the harvest operation (e.g., presence of retention  
2252 buffers, yarding and equipment use immediately adjacent to the stream, upland clearcut vs.  
2253 thinning), but also by physiography (especially hillslope gradient), lithology relative softness,  
2254 and the presence of roads. Thus, the change in magnitude of sediment delivery following harvest

2255 is context dependent and these landscape factors can interact with one another to compound  
2256 these changes. However, from the studies reviewed in the sediment section of the literature  
2257 review, there is evidence that the implementation of BMPs since the 1970s in the northwestern  
2258 United States has lessened the impact and duration of these changes.

### 2259 *Nutrient*

2260 None of the studies published since 2000 and conducted in western North America provide  
2261 experimental evidence of the effects of site factors on nutrient flux into streams. However, Zhang  
2262 et al. (2010) conducted a global review and meta-analysis of the effectiveness of buffers in  
2263 reducing nonpoint source pollution. They reported slope (hillslope gradient) as having a linear  
2264 relationship with buffer pollutant removal efficacy that switched from positive to negative when  
2265 slope increased beyond 10% (i.e., hillslope gradients of ~10% were optimal for buffer efficacy in  
2266 removing pollutants).

2267

### 2268 *Focal Question 3*

2269 *3. What is the frequency of weather-related effects (e.g., windthrow, ice storms, excessive heat,*  
2270 *flood and drought events) on riparian areas? What are the weather-related effects (positive and*  
2271 *negative) on the riparian functions, and how are they distinguished from harvest effects? How do*  
2272 *these effects differ between treated and untreated riparian forests?*

2273 The first part of this question “What is the frequency of weather-related effects (e.g., windthrow,  
2274 ice storms, excessive heat, flood and drought events) on riparian areas?” is a generally worded  
2275 question asking how often weather events in riparian areas occur. The second part of this  
2276 question “What are the weather-related effects (positive and negative) on the riparian functions,  
2277 and how are they distinguished from harvest effects?” contains within it 2 parts 1) what the  
2278 effects on the riparian functions are, and 2) how they are distinguished from timber harvest  
2279 effect. Any study reviewed that answers one or more parts of this question have been included.

### 2280 *Shade*

2281 McIntyre et al. (2021), the “Hard Rock” study, compared changes in shade from pre- to post-  
2282 harvest between three riparian harvest treatments and a reference. Treatments included a two-  
2283 sided 50-ft riparian buffer along at least 50% of the stream (FP; with clearcut to stream’s edge  
2284 outside of the buffer), a two-sided 50-ft buffer along the entire stream (100%), and a clearcut to  
2285 stream without a buffer (0%). The canopy cover was measured 1 meter above the stream surface  
2286 with a spherical densiometer. The changes in canopy cover were distinguished from harvest  
2287 effects and compared to unharvested reference sites by using a BACI design. For the FP  
2288 treatment, mean canopy cover declined from 96% to 72% in the first-year post-harvest but  
2289 continued to decline for 4 years to a minimum of 54%. In the 100% treatment mean canopy  
2290 cover was more stable, decreasing from 94% to 88% in the first year and reaching a minimum of  
2291 82% also by year 4. Canopy cover began to increase after year 4 through year 9 in both  
2292 treatments. In contrast, the reference sites experienced much smaller reductions in canopy cover  
2293 from 95% to 89% in the first four years. The cause of mortality in the treatment sites was



2294 primarily attributed to windthrow. However, while post-harvest mortality in the treatment sites  
 2295 were higher on average than in the reference sites there was a high amount of variability between  
 2296 sites in both the treated and reference sites. For example, in the first 2 years following harvest  
 2297 mortality ranged from 1.8 to 34.6% (loss of basal area) between sites in the FP treatment. In  
 2298 contrast, mortality in the reference sites ranged from 1.1 to 20.4% (loss of basal area) during the  
 2299 same period.

2300 *Litter*

2301 Bilby & Heffner (2016) showed evidence that wind speed has a strong effect on the width of  
 2302 litter delivery areas within riparian areas. They used a combination of field experiments and  
 2303 simulation modeling to estimate the influence of different site factors (physiography, stand age,  
 2304 species composition, wind speed) on litter delivery into streams. Their results showed that litter  
 2305 travel distance was also linearly related to wind speed ( $p < 0.0001$ ). Doubling wind speed at one  
 2306 site led to a 67-87% expansion of the riparian litter contribution zone in the study area. However,  
 2307 this study does not compare the differences in the influence of wind speed on the width of the  
 2308 litter contributing area between harvested and unharvested sites.

2309 *LW*

2310 Table 16. Treatment and responses for selected publications investigating Large Wood relevant to  
 2311 Q3.

Reference	Treatment	Response
McIntyre et al. (2021)	50 feet (100%); 50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)	<u>8-years post-harvest data 100% (n = 4), FP (n = 3), and unbuffered treatments (n = 4)</u> The FP–Reference contrast in mortality was not significant 2 years post-harvest, but it was at 5- and 8-years post-harvest as mortality in FP increased relative to the Reference over time. <b>Wind/physical damage was the primary cause of mortality</b> for all treatments, including the Reference. In the 100% treatment it accounted for 78% and 90% of the loss of basal area and density (stem/ha), respectively; in the FP it accounted for 78% and 65%, in the reference it accounted for 52% and 43%.
Liquori (2006)	Buffer widths ranging from 25-100 feet	<u>3 years post-harvest (n = 20)</u> within no-cut buffers, <b>windthrow caused mortality</b> was up to 3 times greater than competition induced mortality <b>for 3 years following treatment</b> with tree fall probability highest in the outer areas (closest to upland clearcuts) of the buffers. highest at the outside edges of buffers (50+ feet), ~ 60% of total treefall, ~18% in the 0 -25-foot zone, and ~22% in the 25–50-foot zone.
Martin & Grotenfendt (2007)	Buffer widths 20 m or greater	Differences in mortality for the treatment sites were similar to the reference sites for the first 0-10 m from the stream (22% increase). However, <b>mortality in the outer half of the buffers (10-20 m) from the stream in the treatment sites was more than double (120% increase)</b> what was observed in the reference sites. The authors estimate that <b>windthrow mortality was twofold and fivefold greater in the inner and outer halves of the treatment buffers than in the reference buffers, respectively.</b>
Bahuguna et al. (2010)	Buffer widths 10 m, and 30 m	<u>7-years post-harvest (n = 3)</u> In <b>the first 2 years</b> , 11% of the timber was blown down in the 10 m buffer, compared to 4% in the 30 m buffer, and 1% in the unharvested controls.

		Following <b>8 years post-harvest</b> , a significant amount of annual <b>windthrow caused</b> mortality occurred in the unharvested control at 30%, compared to 15% in both 30 m and 10 m buffers.
Schuettt-Hames & Stewart (2011, 2019b)	Buffer widths 50 feet	<u>10-years post-harvest</u> <b>3 years</b> after treatment annual tree fall rates (live and dead) were over <b>8 times</b> (by % of standing trees) and <b>5 times</b> (by trees/acre/yr) <b>higher in the 50-foot buffers</b> than in the reference. <b>4-5 years</b> after treatment mortality was still higher in the treated sites (27.3%) than in the reference (13.6%), but the difference was not significant. <b>10 years</b> after treatment stand mortality in the 50-ft buffer treatment stabilized.

2312

2313 Chapter 3 of the Hard Rock study compared changes in stand mortality and LW input from pre-  
 2314 to post-harvest and between treated and untreated reference sites. Results showed that by year 8,  
 2315 post-harvest mortality as a percentage of pre-harvest basal area was lower in the reference  
 2316 (16.1%) than in the 100% (24.3%) and FP (50.8%) treatments. The FP–Reference contrast in  
 2317 mortality was not significant 2 years post-harvest, but it was at 5- and 8-years post-harvest as  
 2318 mortality in FP increased relative to the Reference over time. The contrast in mortality between  
 2319 the 100% and Reference were not significant for any time interval 8 years post-harvest.  
 2320 Wind/physical damage was the primary cause of mortality for all treatments, including the  
 2321 Reference. In the 100% treatment it accounted for 78% and 90% of the loss of basal area and  
 2322 density (stem/ha), respectively; in FP it accounted for 78% and 65% of the loss. Wind accounted  
 2323 for a smaller proportion of mortality in the reference (52% and 43%, respectively).

2324  
 2325 LW recruitment to the channel was greater in the 100% and FP treatment than in the reference for  
 2326 each pre- to post-harvest time interval. Eight years post-harvest mean recruitment of large wood  
 2327 volume was two to nearly three times greater in 100% and FPB RMZs than in the references.  
 2328 Annual LW recruitment rates were greatest during the first two years, then decreased. However,  
 2329 there was a great deal of variability in recruitment rates within treatment sites and the differences  
 2330 between treatments were not significant. Mean LW loading into the channel (pieces/m of channel  
 2331 length) differed significantly between treatments in the magnitude of change over time. There  
 2332 was a 66%, 44% and 47% increase in mean large wood density in the 100%, FP and 0%  
 2333 treatments, respectively, in the first 2 years post-harvest compared with the pre-harvest period  
 2334 and after controlling for temporal changes in the references. By year 8, only the FP treatment  
 2335 showed a significantly higher proportional increase (41%) in wood loading when compared to  
 2336 the reference. In the time interval 2-8 years post-harvest wood loading in the 100% treatment  
 2337 stabilized.

2338  
 2339 Liquori (2006) investigated treefall characteristics within riparian buffer sites in a managed tree  
 2340 farm in the Cascade Mountains of western Washington. Buffer widths ranged between 25-100  
 2341 feet along non-fish-bearing and fish-bearing streams. Results showed that within no-cut buffers,  
 2342 windthrow caused mortality was up to 3 times greater than competition induced mortality for 3  
 2343 years following treatment with tree fall probability highest in the outer areas (closest to upland  
 2344 clearcuts) of the buffers. Their results showed that treefall was generally highest at the outside  
 2345 edges of buffers (50+ feet), representing about 60% of the total observed treefall, while the 0–25-

2346 foot zone represented ~18%, and the 25–50-foot zone represented ~22%. The researchers  
2347 interpret these results as evidence that windthrow susceptibility within riparian buffers increases  
2348 with increasing distance from the stream.

2349  
2350 Martin & Grotenfendt (2007) compared riparian stand mortality and in-stream LW recruitment  
2351 characteristics between riparian buffer strips with upland timber harvest and riparian stands of  
2352 unharvested watersheds using aerial photography in the northern and southern portions of  
2353 Southeast Alaska. All buffer strips in this study were a minimum of 20 m wide and included  
2354 selective harvest within the 20 m zone (thinning intensity not specified or included in the  
2355 analyses as an effect). The results from this study showed significantly higher mortality (based  
2356 on cumulative stand mortality: downed tree counts divided by standing tree counts + downed tree  
2357 counts by number/ha), significantly lower stand density (269 trees/ha in buffer units and 328  
2358 trees/ha in reference units), and a significantly higher proportion of LW recruitment from the  
2359 buffer zones of the treatment sites than in the reference sites. Also, results showed that mortality  
2360 varied with distance to the stream. Differences in mortality for the treatment sites were similar to  
2361 the reference sites for the first 0-10 m from the stream (only a 22% increase in the treated sites).  
2362 However, mortality in the outer half of the stream buffers (10-20 m) across treatment sites was  
2363 more than double (120% increase) that observed within the reference sites. The authors estimate  
2364 that windthrow mortality was twofold and fivefold greater in the inner and outer halves of the  
2365 treatment buffers than in the reference buffers, respectively.

2366  
2367 Bahuguna et al. (2010) evaluated the difference in windthrow caused mortality between 10 m, 30  
2368 m buffer widths (neither had thinning within the buffer and both had upland clear-cuts) and  
2369 unharvested controls in the Coast Mountains, British Columbia. Following harvest, 11% of  
2370 initially standing timber was blown down in the first and second years in the 10 m buffer,  
2371 compared to 4% in the 30 m buffer, and 1% in the unharvested controls. However, after 8 years  
2372 post-harvest, a significant amount of annual mortality occurred when winter storms brought  
2373 down multiple trees in the unharvested control at 30%, compared to 15% in both 30 m and 10 m  
2374 buffers. These results show evidence that timber harvest can increase windthrow caused  
2375 mortality within protective buffers in the short term but can stabilize within a decade. Further,  
2376 this study shows evidence that windthrow caused mortality is stochastic and large storm events  
2377 can cause significant mortality within untreated riparian forests.

2378  
2379 Schuett-Hames and Stewart (2019a) compared changes in stand mortality and LW recruitment  
2380 between treated and untreated riparian areas along fish-bearing streams in eastern Washington.  
2381 Treatments were prescribed under the Standard Shade Rule (SR), under the All-Available Shade  
2382 rule (AAS), and unharvested reference sites. Both shade rules have a 30-ft no-cut buffer (core  
2383 zone) immediately adjacent to the stream. The SR prescription allows thinning in the buffer zone  
2384 30-75 feet (inner zone) from the stream while the AAS prescription requires retention of all  
2385 shade providing trees in this area. Thinning non-shade providing trees within the inner zone is  
2386 allowed under the AAS rule. Results from a mixed model comparison showed that the frequency  
2387 of wood input from fallen trees was significantly greater in SR group compared to both the  
2388 reference and AAS groups ( $p < 0.001$ ), while the difference between reference and AAS groups

2389 was not significant. Over 60% of pieces recruited from AAS and SR fallen trees consisted of  
2390 stems with attached rootwads (SWAR), double the proportion in the reference sites. The  
2391 reference-AAS and reference-SR differences in recruitment of SWAR pieces were significant ( $p$   
2392  $<0.001$ ). The authors comment that the higher mortality and recruitment of LW in the SR sites  
2393 was primarily due to windthrow.

2394 Schuett-Hames et al, (2011) compared tree mortality and LW recruitment between treated and  
2395 untreated riparian stands along non-fish-bearing streams in western Washington. Treated sites  
2396 were prescribed a 50-foot-wide no-cut buffer. Annual fall rates of live and dead standing stems  
2397 combined were over 8 times (by % of standing trees) and 5 times (by trees/acre/yr) higher in the  
2398 50-foot buffers than in the reference buffers 3 years after treatment. These differences were  
2399 significant for both metrics ( $p < 0.001$ ). Over the entire five-year period, the percentages of  
2400 standing trees that were uprooted and broken (as well as the combined total) were significantly  
2401 greater in the 50-foot buffer. Wind was the dominant tree fall process, accounting for nearly 75%  
2402 of combined fallen trees, 11% fell from other trees falling against them and 1.8% of fallen trees  
2403 fell from bank erosion. Differences in mortality followed a similar pattern to tree fall rates. In the  
2404 50-foot buffer sites mortality rates were significantly higher (3.5 times higher) than in the  
2405 reference sites for the first three years following harvest. However, in years 4-5 mortality rates  
2406 increased in the reference buffers after high-intensity storms resulting in non-significant  
2407 differences in mortality during this period. The cumulative percentage of live trees that died over  
2408 the entire five-year period was 27.3% in the 50-ft buffers compared to 13.6% in the reference  
2409 reaches, but the difference was not statistically significant. The authors suggest that the lack of  
2410 significance was likely due to the high variability in mortality between sites in the 50-foot  
2411 buffers.

2412 In the follow-up study, Schuett-Hames & Stewart (2019b) reported that over a 10-year period,  
2413 stand mortality in the 50-ft buffer treatment stabilized and showed a cumulative 14.1% reduction  
2414 in live basal, while the reference stands showed a 2.7% increase in live basal area. The  
2415 differences in these values were not significant. Cumulative LW recruited into stream channel  
2416 over the 10-period was double in the 50-ft buffer treatment streams than in the reference streams.

2417 In general, the studies reviewed above show evidence that upland timber harvest with riparian  
2418 retention buffers initially increases stand mortality within the buffers and increases LW  
2419 recruitment relative to unharvested reference stands in the short-term. Hence, treated riparian  
2420 forests appear to have a higher susceptibility to windthrow caused mortality, at least in the short  
2421 term, compared to untreated stands. Depending on the streams in question, an increase in LW  
2422 could be considered a positive or negative impact This increase in mortality and LW recruitment  
2423 is attributed to an increase in the susceptibility to windthrow within the riparian buffers relative  
2424 to the unharvested controls. Further, multiple studies (Liquori, 2006; Martin & Grotefendt, 2007,  
2425 Schuett-Hames & Stewart 2019a) showed evidence that the increase in windthrow caused  
2426 mortality is highest in the outer area of the riparian buffers (area closest to upland treatments).  
2427 There is some evidence that thinning within the buffer can also affect mortality rates, but these  
2428 studies are few. In the three studies that collected post-harvest data for 8 or more years  
2429 (Bahuguna et al., 2010; McIntyre et al., 2021; Schuett-Hames & Stewart 2019b), there is  
2430 indication that mortality in the riparian buffers and annual LW recruitment into adjacent streams

2431 stabilizes within 5-10 years. However, in the subsequent decades following treatments with  
 2432 upland clearcuts there is evidence that LW recruitment rates can continue to decrease and in  
 2433 stream wood loads may become depleted before recruitment rates can recover (Nowakowski &  
 2434 Wohl, 2008; Reid & Hassan, 2020) depending on applied management practices (e.g., buffer  
 2435 widths, road construction, etc.). For example, Teply et al. (2007) used simulation modeling to  
 2436 estimate the effectiveness of Idaho Forest Practices for riparian buffers and found no significant  
 2437 difference between predicted LW loads for harvested and unharvested sites 30-, 60-, or 100-years  
 2438 post-harvest.

2439 *Nutrient*

2440 Table 17. Treatment and responses for selected publications investigating Nutrients relevant to  
 2441 Q3.

Reference	Treatment	Response
Vanderbilt et al. (2003)	long-term datasets from six watersheds in the H.J. Andrews Experimental Watershed	<u>20-30 years of historical data</u> <b>Total annual discharge</b> was a positive predictor of annual <b>dissolved organic nitrogen (DON)</b> export in all watersheds with R <sup>2</sup> values ranging between 0.42 to 0.79. No other nutrients nitrate (NO <sub>3</sub> -N), ammonium (NH <sub>4</sub> -N), and particulate organic nitrogen (PON) showed consistent patterns or relationships to any predictor variables. The increase in concentration began in July or August with <b>the earliest rain events</b> , and peak DON concentrations occurred in October through December before the peak in the hydrograph. DON concentrations then <b>declined during the winter months</b> .
Yang et al. (2021)	Mastication of riparian area shrubs to < 10% cover. Treatment effects compared with drought effects.	2 years pre-drought, 3 years following drought and treatment <b>Drought alone</b> altered the concentration of dissolved organic carbon (DOC) in stream water. Dissolved organic carbon (DOC) was 62% lower and the ratio of DOC to dissolved inorganic nitrogen (DIN) was 82% lower during drought years. <b>Drought combined with thinning</b> showed 66- 94% higher DOC than in unthinned watersheds.

2442  
 2443 Vanderbilt et al. (2003) analyzed long-term datasets (ranging 20-30 years for each watershed)  
 2444 from six watersheds in the H.J. Andrews Experimental Watershed in the west-central Cascade  
 2445 Mountains of Oregon to investigate patterns in dissolved organic nitrogen (DON) and dissolved  
 2446 inorganic nitrogen (DIN) export with watershed hydrology. The researchers used regression  
 2447 analysis of annual N inputs and outputs with annual precipitation and stream discharge to  
 2448 analyze patterns. Their results showed that total annual discharge was a positive predictor of  
 2449 annual DON export in all watersheds with R<sup>2</sup> values ranging between 0.42 to 0.79. In contrast,  
 2450 relationships between total annual discharge and annual export of nitrate (NO<sub>3</sub>-N), ammonium  
 2451 (NH<sub>4</sub>-N), and particulate organic nitrogen (PON) were variable and inconsistent across  
 2452 watersheds. The authors speculate that different factors may control organic vs. inorganic N  
 2453 export. The authors emphasize the importance of analyzing data from multiple watersheds in a  
 2454 single climactic zone to make inferences about stream chemistry.

2455 Yang et al. (2021) investigated the effects of drought and forest thinning operations  
2456 (independently and combined) on stream water chemistry in the Mediterranean climate  
2457 headwater basins of the Sierra National Forest. The effects of drought alone were examined by  
2458 comparing water samples collected from control watersheds for 2 years before and 3 years after  
2459 drought. The effects of drought and thinning combined were examined by comparing water  
2460 samples collected from treated sites to reference sites for three years post-harvest (all drought  
2461 years). Drought alone altered the concentration of dissolved organic carbon (DOC) in stream  
2462 water. Volume-weighted concentration of DOC was 62% lower ( $p < 0.01$ ) and the ratio of  
2463 dissolved organic carbon to dissolved inorganic nitrogen (DOC:DON) was 82% lower ( $p =$   
2464  $0.004$ ) in stream water in years during drought (WY 2013–2015) than in years prior to drought  
2465 (WY 2009 and 2010). Drought combined with thinning altered DOC and DIN concentrations in  
2466 stream. For stream water, volume-weighted concentrations of DOC were 66- 94% higher in  
2467 thinned watersheds than in control watersheds for all three consecutive drought years following  
2468 thinning. No differences in DOC concentrations were found between thinned and control  
2469 watersheds before thinning. The authors conclude that their results showed evidence that the  
2470 influences of drought and thinning are more pronounced for DOC than for DIN in streams.

#### 2471 *Drought Frequency*

2472 Wise (2010) used reconstructed newly collected tree-ring data augmented with existing  
2473 chronologies from sites at three headwater streams in the Snake River Basin to estimate  
2474 streamflow patterns for the 1600-2005 time-period. Streamflow patterns derived from  
2475 instrumental data and from reconstructed chronologies were compared with other streamflow  
2476 previously reconstructions of three other western rivers (the upper Colorado, the Sacramento,  
2477 and the Verde Rivers) in similar climates to examine synchronicity among the rivers and gain  
2478 insight into possible climatic controls on drought episodes. The reconstruction model developed  
2479 for the analysis explained 62% of the variance in the instrumental record after adjustment for  
2480 degrees of freedom. Results showed evidence that droughts of the recent past are not yet as  
2481 severe, in terms of overall magnitude, as a 30-year extended period of drought discovered in the  
2482 mid-1600s. However, in terms of number of individual years of  $< 60\%$  mean-flow (i.e., low-flow  
2483 years), the period from 1977-2001 were the most severe. Considering the frequency of  
2484 consecutive drought years, the longest (7-year-droughts), occurred in the early 17th and 18th  
2485 centuries. However, the 5-year drought period from 2000-2004 was the second driest period over  
2486 the 415-year period examined. The correlative analysis of the chronologies developed for the  
2487 upper Snake River with other rivers of the West showed mixed results with periods of positive  
2488 and negative correlations. The author interprets these results as evidence that drought frequency,  
2489 in general, in this area appears to be increasing in severity and that mean annual flow appears to  
2490 be reducing in the latter half of the 20th and the beginning of the 21st century. The exceptions  
2491 being the 1930's dustbowl, and an unusually long dry period in the early 1600s.

#### 2492 *Fire Frequency*

2493 Dwire & Kauffman (2003) in their reviewed and summarized the available conducted on fire  
2494 regimes in forested riparian areas relative to uplands in the western United States. They  
2495 summarized the distinctive features of riparian areas that can influence the properties of fire as

2496 (1) higher fuel loads because of higher net primary productivity, (2) higher fuel moisture content  
2497 due to proximity to water, shallow water tables, and dense shade, (3) active channels gravel bars  
2498 and wet meadows may act as fuel breaks, (4) topographic position (canyon bottoms, low point on  
2499 landscape) leads to higher relative humidity, fewer lightning strikes, but more human-caused  
2500 ignitions, (5) microclimate may lead to cooler temperatures and higher humidity that can lessen  
2501 fire intensity and spread. They highlight a need for more extensive research on the history and  
2502 ecological role of fire in the riparian areas of the western United States.

2503 There is a logical assumption that fire in riparian zones would be less frequent than in adjacent  
2504 uplands because of its proximity to water. However, several studies have been conducted which  
2505 reconstruct historical fire regimes in riparian areas relative to adjacent uplands and have  
2506 provided varying results. Everett et al. (2003) used fire-scar and stand-cohort records to estimate  
2507 the frequency and seasonality of fire in Douglas-fir dominated riparian areas and adjacent  
2508 uplands. They sampled sites along 49 stream segments on 24 different streams in the Wenatchee  
2509 (33 segments) and Okanogan (16 segments) National Forests. The data collected allowed for  
2510 reconstruction of fire occurrence back to 1896. Their results showed that the mean count of fire  
2511 scars was significantly fewer in riparian areas than in adjacent uplands regardless of valley type,  
2512 aspect, or plant association group. However, the difference between riparian and upland fire scars  
2513 was greatest for western aspects and least for northern aspects. Also, the differences were  
2514 greatest for the ‘warm mesic shrub/herb’ plant association group (e.g., common snowberry), and  
2515 least in the cool dry grass plant association group (e.g., pinegrass, or elk sedge).

2516 Prichard et al. (2020) evaluated drivers of fire severity and fuel treatment effectiveness at the  
2517 2014 Carlton Complex in north-central Washington State. While this study's objective does not  
2518 specifically evaluate differences in fire severity between riparian and upland forests, it did  
2519 evaluate differences in fire severity based on variations in topographic and vegetation type  
2520 variables. One vegetation variable was classified broadly as “riparian vegetation” from the  
2521 publicly available data set LANDFIRE. The authors used a combination of simultaneous  
2522 autoregression and random forests approaches to model drivers of fire severity. In the study  
2523 area's southern section (1 of 2 designated study areas), the results showed cover type was a  
2524 significant predictor with negative correlations with fire severity in non-forest types and riparian  
2525 forests.

2526 Conversely, Olson & Agee (2005) provide evidence that fire return intervals in the riparian areas  
2527 of the Umpqua National Forests, Oregon, may not have differed significantly from adjacent  
2528 upland forests. They reconstructed historical fire return intervals from fire scar cross sections  
2529 taken from 15 stream reaches and 13 paired upland forests. Sites were primarily dominated by  
2530 Douglas-fir, western red cedar, and western hemlock. The number of fires per plot, maximum  
2531 and minimum fire return intervals, and the Weibull median fire return interval (WMPIs) were  
2532 compared between riparian and upland stands using the Wilcoxon signed rank test, the Mann-  
2533 Whitney U-test for unmatched samples, and the Kruskal-Wallis one-way analysis of variance.  
2534 The results showed that between 1650 and 1900, 43 fire years occurred on 80 occasions. Of these  
2535 80 occasions, 33 were recorded in the riparian and adjacent upslope forest, 23 were recorded in  
2536 only the riparian area, and 24 were recorded only in the upland forests. The riparian WMPIs

2537 were somewhat longer (ranging from 35-39 years, with fire return intervals ranging from 4-167  
2538 years) than upslope WMPIs (ranging from 27-36 years, with fire return intervals ranging from 2-  
2539 110 years), but these differences were not significant. The authors, Olson & Agee (2005),  
2540 interpret these results as evidence that fires in this area were likely patchy and smaller in scale  
2541 with a high incidence of fires occurring only in the riparian area or only in the upland forests,  
2542 and less commonly in both. The authors also suggest that fire is a natural occurrence in the  
2543 riparian areas of this area and should be restored to protect riparian forest health.

2544 Another study from the Klamath Mountains in northern California showed evidence that fires in  
2545 riparian forests may have been more frequent than in adjacent upland forests (Skinner, 2003).  
2546 Skinner (2003) used dendrochronological methods to construct fire return intervals for 5 riparian  
2547 and adjacent upland forests sites, each between 1-2 hectares. Because of the small sample size,  
2548 statistical analysis was not conducted, and their results are only descriptive. The ranges of fire  
2549 return intervals (FRIs) were similar between riparian and upland forests. However, the median  
2550 FRI for the riparian forests was nearly double that in adjacent uplands. The authors conclude that  
2551 these limited data suggest fire in the riparian areas may be more variable than in the uplands in  
2552 frequency and intensity.

2553 Yet, another study from Harley et al. (2020) showed evidence that the differential fire occurrence  
2554 riparian and adjacent uplands may have been dependent on weather (i.e. drought). Harley et al.  
2555 (2020) reconstructed low-severity fire histories from tree rings in 38 1-ha plots. This data was  
2556 supplemented with existing fire histories from 104 adjacent upland plots. 2633 fire scars were  
2557 sampled from 454 (127 riparian; 329 upland) trees from two sites in the Blue Mountains in  
2558 north-eastern Oregon: One in the Wallowa-Whitman (WWNF) and one in the Malheur (MNF)  
2559 National Forests. Fire-scar dates were used to construct plot composite fire chronologies,  
2560 excluding fire dates recorded from only one tree. These were used to compute median fire  
2561 intervals for riparian and upland forests for each site and for both sites combined. A mixed linear  
2562 model with fire interval as a response and plot type (riparian vs. Upland) as a predictor was used  
2563 to check for statistical difference in fire frequency. The influence of climate on fire occurrence  
2564 was inferred by assessing whether the summer Palmer Drought Severity Index (PDSI) differed  
2565 significantly during the fire year or preceding or following years (-3 to +1 years) using  
2566 superimposed epoch analysis. Results showed that Fires burned synchronously in riparian and  
2567 upland plots during more than half of the fire years at both WWNF and MNF (55%and57%,  
2568 respectively). At WWNF, fires burned during 65 years of the analysis period (1650–1900); 36  
2569 burned in both riparian and upland plots, 7 burned only in riparian plots and 22 burned only in  
2570 upland plots. At MNF, fires burned during 74 years of the analysis period; 42 burned in both  
2571 riparian and upland plots, 3 burned only in riparian plots and 29 burned only in upland plots. At  
2572 both sites, average PDSI was significantly warm–dry during synchronous fire years. However,  
2573 climate was not significantly cool–wet during non-synchronous fire years at either site. The  
2574 authors interpret these results as evidence that historical synchronized fire occurrence was more  
2575 likely during excessively dry or drought years.

2576 There is also evidence that riparian forest fire regimes have been altered in many areas from pre-  
2577 Euro-American settlement due to fire suppression. Messier et al. (2012), used dendro-ecological



2578 methods to reconstruct pre-Euro-American settlement riparian forest structure and fire frequency  
 2579 for comparison of changes post-settlement in the Rouge River of southwestern Oregon. Fire  
 2580 events were dated from increment cores and fire-scar cross-sections back to the year 1600,  
 2581 approximately. Changes in annual radial growth rates were used to infer changes in stand density  
 2582 over time. Results showed the age distribution prior to 1850 followed a pulse pattern of  
 2583 recruitment with recruitment peaks occurring around 1850, 1800, and between 1740-1770  
 2584 (though this pulse was difficult to discern because the sample size of trees established prior to  
 2585 1740 were relatively few). After 1900, many mixed conifer sites showed a dramatic increase in  
 2586 the recruitment of more more-shade tolerant white fir (*Abies concolor*) compared to Douglas-fir  
 2587 (*Pseudotsuga menziesii*). White fir comprised 51% of the live trees recruited after 1900, but only  
 2588 18% of the live trees before 1900. Results from the 26 cross-dated fire scars spanned from 1748  
 2589 – 1919 with the highest number of detected fires occurring in the early-settlement period (1850-  
 2590 1900). The authors interpret these results as evidence that fire suppression over the last century  
 2591 has changed the successional pathway and stand structure of riparian forests in this area.

2592 Van de Water & North (2011) found similar results from their study in the northern Sierra  
 2593 Nevada. They compared current field data with reconstructed data to estimate changes in stand  
 2594 structure, fuel loads, and potential fire behavior over time. Additionally, they estimated how  
 2595 these conditions for riparian forests compared to adjacent upland forests during the reconstructed  
 2596 and current periods. Data for current forest structure, species composition, and fuel loads were  
 2597 collected from 36 adjacent riparian and upland sites (72 sites total). The reconstruction period  
 2598 was set at the year of the last fire (ranging from 1848 – 1990), determined from fire-scar records.  
 2599 Potential fire behavior, effects, and canopy bulk density were estimated for current and  
 2600 reconstructed stand conditions for riparian and upland sites using Forest vegetation Simulator  
 2601 (FVS). Stand structure (BA, stand density, snag volume, QMD, average canopy base height),  
 2602 species composition, fuel load, potential fire behavior, canopy bulk density, and mortality were  
 2603 compared between current and reconstructed periods for riparian and upland sites, and between  
 2604 sampling areas (riparian vs. Upland) with an analysis of variance (ANOVA). Results showed that  
 2605 under current conditions, riparian forests were significantly more fire prone than upland forests,  
 2606 with greater stand density (635 vs. 401 stems/ha), probability of torching (0.45 vs. 0.22),  
 2607 predicted mortality (31% vs. 16% BA), and lower quadratic mean diameter (46 vs. 55 cm),  
 2608 canopy base height (6.7 vs. 9.4 m), and frequency of fire tolerant species (13% vs. 36% BA).  
 2609 However, the reconstructed periods showed no significant difference between riparian and  
 2610 upland forests for fuels and structure. The authors suggest that these results provide evidence that  
 2611 the historic fire return intervals may not have differed significantly between riparian and upland  
 2612 forests in this area.

2613 *Fire Effects on Function*

2614 *Litter and Nutrients*

2615 Table 17. Treatment and responses for selected publications investigating the relationship  
 2616 between wildfire Litter and Nutrients relevant to Q3.

Reference	Treatment	Response
-----------	-----------	----------

Musetta-Lambert et al. (2017)	Buffer widths 30 m; wildfire	<u>Sampling began 7-17 years after harvest or 12 years after wildfire (n = 5 harvest, 7 fire, 6 reference)</u> <b>Total leaf litter input was significantly higher at fire sites than at harvested or reference sites.</b> Fire sites had significantly greater leaf litter inputs <b>by species</b> of willow spp., Atlantic ninebark, and speckled alder than in both reference and harvested sites
Rhoades et al. (2011)	Wildfire	<u>1- year pre- and 5-years following the 2002 Hayman Fire in Colorado</u> Cation concentrations and acid neutralizing capacity (ANC) increased immediately and significantly following fire that <b>peaked at 4 months</b> . Ca <sup>2+</sup> concentrations, ANC, and conductivity remained elevated in the burned streams <b>for 2 years compared</b> to pre-fire conditions, and unburned streams. Stream water nitrate and turbidity increased linearly with the proportion of a basin burned or burned at high severity.
Son et al. (2015)	Wildfire	<u>2-years pre- and immediately following wildfire</u> Results for turbidity showed no significant differences between pre- and post-fire ranges <b>immediately following fire</b> . After <b>first rainfall event</b> mean turbidity increased from 11.3 NTU to 641.62 NTU. <b>Post-fire</b> aqueous total phosphorus (TP) and nitrogen (TN) was significantly higher than pre-fire values (390 and 6 times higher than pre-fire values for TP and TN, respectively).

2617

2618 Musetta-Lambert et al. (2017) compared changes in leaf-litter inputs into streams following  
 2619 adjacent riparian forest harvesting or wildfire to reference sites. This study took place in the  
 2620 boreal forest of the White River Forest management Area in Ontario, Canada, ~75 km inland  
 2621 from the northern shore of Lake Superior. This study is outside of western North America (the  
 2622 focal area for this review), but it is the only study found that provides experimental evidence of  
 2623 wildfire's effects on litter inputs. The study sites consisted of ~50 m reaches in 25 catchments, 10  
 2624 that were harvested, 7 that experienced wildfire, and 8 references. Of these reaches a subset was  
 2625 used to riparian forest structure, leaf litter inputs, and water chemistry (5 harvest, 7 fire, 6  
 2626 reference). The harvested catchments were harvested 7-17 years prior to the study (minimum 30  
 2627 m riparian buffers; specific harvest rules/methods not described). The wildfire catchments had  
 2628 burned 12 years prior to the study and had no dead material removed. The reference catchments  
 2629 had no fire or harvesting for a minimum of 40 years. Water grab samples were collected in  
 2630 September, October and November 2010, and May, June and September of 2011 from the study  
 2631 reaches.

2632 Water samples were analyzed to obtain measurements for pH, conductivity, dissolved organic  
 2633 carbon (DOC) and dissolved inorganic carbon (DIC) concentrations, soluble reactive  
 2634 phosphorous (SRP), along with a suite of other major elements and nutrient measurements (total  
 2635 N, NH<sub>4</sub>, total P, Ca, K, Mg, etc.). Vertical leaf litter traps consisting of plastic bins were placed at  
 2636 10 locations along the bankfull width of each site. Lateral leaf fall was not collected or analyzed.  
 2637 Leaf litter inputs were focused on leaves from deciduous trees and shrubs. Leaves were separated  
 2638 to the lowest possible taxonomic level, dried and weighed for analysis.

2639 Univariate one-way ANOVA models were used to determine differences in water chemistry,  
 2640 riparian forest characteristics of juvenile tree and shrub communities (richness, Shannon's

2641 diversity index, relative occurrence of individual taxa), mature tree communities (total basal  
2642 area, stem density), and litter subsidies (richness, mass input). Results for water chemistry  
2643 showed that Conductivity, pH, and dissolved inorganic carbon were significantly higher at fire  
2644 sites than at reference sites ( $p = 0.02$ ,  $p = 0.04$ ,  $p = 0.03$ , respectively) but did not differ between  
2645 harvested and fire sites or harvested and reference sites.

2646 Results for stand structure showed there was significantly higher taxa richness in fire sites than  
2647 in reference sites or harvested sites ( $p = 0.04$ ). Taxa richness did not differ significantly between  
2648 reference and harvested sites. Reference sites had significantly higher total mean densities (# ha  
2649  $^{-1}$ ) of mature riparian trees (>10 cm DBH) than fire ( $p < 0.001$ ) and harvested sites ( $p = 0.036$ ).  
2650 Total mature tree densities in reference sites were 1.7x and 4x higher than in harvested and fire  
2651 sites, respectively. Taxa richness in leaf litter subsidies did not significantly differ among  
2652 disturbances ( $p = 0.477$ ). Total leaf litter input ( $\text{g m}^{-1}$ ) significantly higher at fire sites than at  
2653 harvest ( $p = 0.02$ ) or reference sites ( $p = 0.02$ ). Fire sites had significantly greater leaf litter  
2654 inputs of willow spp. ( $p = 0.0002$ ,  $0.006$ , respectively), Atlantic ninebark ( $p = 0.002$ ,  $0.003$ ,  
2655 respectively) and speckled alder ( $p = 0.02$ ,  $0.04$ , respectively) than in both reference and  
2656 harvested sites. The authors interpret these results as evidence that natural fire disturbance in  
2657 low-order boreal forest streams had higher leaf litter inputs, and different stand structures and  
2658 composition than harvested or untreated riparian stands. They suggest that while harvested  
2659 stands were more structurally similar to fire affected stands than reference stands, the future  
2660 implementation of these treatments should intend to emulate the patchy nature of wildfire  
2661 disturbance. This would enhance the diversity of riparian forest structure and increase litter  
2662 subsidies into streams.

### 2663 *Nutrients*

2664 Rhoades et al. (2011) monitored stream chemistry and sediment 1-year before and for 5-years  
2665 after the 2002 Hayman Fire in Colorado. Monthly water samples were collected from streams in  
2666 three burned and three unburned watersheds. Pre-fire and post-fire water nitrate, cation  
2667 concentration ( $\text{Ca}^{2+}$ ,  $\text{Mg}^{2+}$ ,  $\text{K}^{+}$ ), acid neutralizing capacity (ANC) and turbidity were compared  
2668 graphically and statistically between the three burned and unburned basins. Results for cation  
2669 concentrations and ANC showed an immediate and significant increase that peaked during the 4-  
2670 month period following the fire. The  $\text{Ca}^{2+}$  concentrations, ANC, and conductivity remained  
2671 elevated in the burned streams for 2 years compared to pre-fire conditions, and unburned  
2672 streams. Stream water nitrate and turbidity increased linearly with the proportion of a basin  
2673 burned or burned at high severity. No other chemical analyte showed a significant response to  
2674 fire severity or extent. Streams draining basins affected by extensive stand-replacement fires  
2675 showed a 3.3-fold higher ( $p = 0.000$ ) nitrate concentration than basins that burned less. Also,  
2676 turbidity was 2.4-fold ( $p = 0.000$ ) higher average turbidity compared to streams in basins burned  
2677 less severely or extensively. In the extensively burned basins, stream water nitrate concentrations  
2678 did not decline over the five years of the study and the mean concentrations of nitrate in the fifth  
2679 year did not differ from the fourth year. The authors conclude that wildfire can have immediate  
2680 and mid-term (up to 5 years) impacts on water chemistry and turbidity. Further, the magnitude

2681 and temporal increases of nitrate and turbidity, specifically, have a positive relationship with burn  
2682 severity and extent.

2683 Son et al. (2015) compared stream water samples before and after an intense wildfire in the  
2684 Cache la Poudre River basin in Colorado. Stream water samples for total phosphorus (TP) and  
2685 total nitrogen (TN) were collected over 2 years (2010 – May 2012) before the fire in June 2012.  
2686 Two post-fire water samples were taken: 1) immediately following containment of the fire (July  
2687 4, 2012) and 2) twelve days after the fire was contained (July 16, 2012). For each pre- and post-  
2688 fire sampling date water samples were collected at three randomly selected points at two sites.  
2689 Riverbed sediments were also collected at each site and sieved through a 2 mm sieve to capture  
2690 the geochemically reactive portion of the riverbed. The pre- and post-fire sediment and stream  
2691 water quality were compared with t-test. Correlations of sediment and stream water quality with  
2692 other factors (e.g., stream temperature, precipitation, streamflow) were evaluated with a  
2693 Pearson's correlation at 0.05 and 0.1 significance levels. Results for turbidity showed no  
2694 significant differences between pre- and post-fire ranges immediately following fire. However,  
2695 after the first post-fire rainfall (2.5 mm) nephelometric turbidity ranged from 113.6 - 2099.4  
2696 NTU (mean = 641.62 NTU), a considerable increase from pre-fire data (mean 11.3 NTU), and  
2697 post-fire data before rainfall (47.3 NTU). Post-fire aqueous TP and TN loads ranged from 30.5 -  
2698 56,086 and 45.4 - 1203 kg/day, respectively, and were significantly higher than pre-fire values  
2699 (390 and 6 times higher than pre-fire values for TP and TN, respectively). The authors note that  
2700 this is likely due to the transport and input of ash into the stream. After the first rainfall, all forms  
2701 of P were significantly higher than pre-fire concentrations, such as soluble reactive phosphorus  
2702 (SRP;  $p = 0.000$ ), dissolved organic phosphorus (DOP;  $p = 0.009$ ), and particulate phosphorus  
2703 (PP;  $p = 0.02$ ). Riverbed sediment equilibrium P concentrations increased significantly ( $p =$   
2704  $0.007$ ) from pre- to post-fire in all sites. The authors conclude that this study shows evidence that  
2705 stream TP and TN, and riverbed sediment TP all increased significantly after the first rainfall,  
2706 post-fire. They further suggest that the effects of wildfire on riverbed sorption mechanisms are  
2707 very complex but further research would be valuable because fire impacted sediments highly  
2708 concentrated P can become a long-term source of P.

2709 *LW*

2710 Bendix & Cowell (2010) investigated the effects of fire and flooding on LW input in two  
2711 tributaries of Sespe Creek (Potrero John Creek and Piedra Blanca Creek) in the Los Padres  
2712 national Forest in southern California. Both sites were located within the perimeter of the Wolf  
2713 Fire that burned in June of 2002. Extensive flooding in the area occurred during January and  
2714 February of 2005. The study area is characterized by chaparral dominated communities and a  
2715 Mediterranean-type climate. While there is a scarcity of trees in the uplands, the riparian areas  
2716 contained substantial growth of *Alnus rhombifolia* (white alder), *Populus fremontii* (Fremont  
2717 cottonwood), *Quercus agrifolia* (coast live oak), *Quercus dumosa* (scrub oak) and *Salix* sp.  
2718 (willows) on the valley floors. Thus, any change in in-stream or riparian area LW was sourced  
2719 exclusively from the riparian area. Data for LW and standing live and dead stems in the riparian  
2720 area were collected in July, of 2003 (1-year pre-fire) and again in July of 2005 (3-years post-fire,  
2721 5-6 months after flood events). This data was used to answer 4 questions: 1) How many of the

2722 burned snags fell during this time, and what was the species composition?, 2) Did snags differ by  
2723 species or size in the rate at which they fell?, 3) How did flooding after the fire affect the rate at  
2724 which snags fell?, 4) How did flooding affect the mobilization of fallen snags? Questions 1 was  
2725 analyzed by comparing descriptive data (i.e., no statistical analysis). A t-test was used to compare  
2726 mean diameter of standing and fallen stems (question 2). T-tests were also used to analyze  
2727 differences in mean flow depth for standing vs. fallen snags and for fallen snags still present vs.  
2728 snags that had been transported after flooding (questions 4 and 5). Results showed high post-fire  
2729 mortality (94%) with 339 of 362 stems killed. By 2005, 57 of the 339 snags had fallen (16.8%).  
2730 The majority of fallen stems were either *Alnus* or *Salix* species. Standing snags varied in size  
2731 from 3 cm to 69.2 cm, whereas those that had fallen ranged from 3 cm to 33 cm. Among the  
2732 fallen snags, those <10 cm were not proportionate to the overall numbers, whereas snags between  
2733 10 cm and 30 cm were disproportionately likely to fall. While fewer snags in the larger size  
2734 classes the mean diameter of fallen snags was larger than the mean diameter of standing snags  
2735 (11.4±10.9 cm vs. 11.0±8.0 cm) and did not differ significantly. The mean flood depth for fallen  
2736 snags (1.05±0.68 m) was significantly greater than those still standing (0.40±0.56 m;  $p < 0.0001$ ,  
2737  $n=339$ ). The three species experiencing no snagfall at all (*Abies glauca*, *Rhamnus californica* and  
2738 *Quercus agrifolia*) occurred only in higher quadrats, which had experienced virtually no  
2739 flooding. Of the 57 snags that had fallen by July 2005, 43 (75%) were gone from the quadrats in  
2740 which they had been recorded in 2003. The snags that had been mobilized were from quadrats  
2741 that had experienced deeper flood depths (1.14±0.69 m) than those that had remained. (0.80±0.62  
2742 m), but the difference is insignificant. The authors interpret these findings as an indication that  
2743 short-term rates of snagfall following wildfire are influenced by the species composition of  
2744 burned stems and by post-fire flood depth. Thus, although wildfire resulted in many burned snags  
2745 across the valley floor, the rate at which these stems are recruited into the fluvial system as  
2746 woody debris varies by the ecological characteristics and the geomorphic setting.

2747

#### 2748 Focal Question 4

2749 *4. How do various treatments within riparian buffers relate to forest health and resilience to fire,*  
2750 *disease, and other forest disturbances?*

2751 While there are several studies that discuss the frequency, dynamics, or potential for  
2752 disturbances, especially fire, in riparian areas of the western United States (Dwire & Kauffman,  
2753 2003; Everett et al., 2003; Merschel et al., 2014) there is a dearth of studies that investigate how  
2754 treatments within the riparian area or in riparian buffers relate to the riparian area's resilience to  
2755 disturbance. No studies found in our literature search and review were suitable for providing  
2756 direct experimental evidence of the effects of riparian buffer treatments on riparian health and  
2757 resilience to disturbance except for several studies that provide evidence that riparian harvest  
2758 treatments have the potential to increase susceptibility to windthrow caused mortality. Post-  
2759 harvest changes in windthrow susceptibility are discussed in focal question 3. One study used  
2760 simulation modeling to estimate changes in health and susceptibility to disturbance with and  
2761 without treatment.

2762 Ceder et al. (2018) used Forest Vegetation Simulator (FVS) to predict how treatment along fish-  
 2763 bearing streams of eastern Washington affects riparian stand health and susceptibility to insects,  
 2764 disease, and crown fire. The projected changes in susceptibility were produced for the low- and  
 2765 mid-elevation regulatory zones for timber harvest. Models were run for 50 years with and  
 2766 without application of prescribed treatments. Prescriptions for these zones include a buffer width  
 2767 of 75-130 ft depending on stream width category. For all treatments, no harvest is allowed within  
 2768 the first 30 feet from the bankfull channel. Timber harvest is allowed in the remaining width of  
 2769 the buffer but must meet a minimum basal area based on the regulatory zone. The authors report  
 2770 high variability in the data and the outputs of each modeling scenario. However, they report that  
 2771 overall, as riparian zone growth was simulated with and without management, tree size and stand  
 2772 density increased, along with some increases in insect and disease susceptibility and potential  
 2773 fire severity without management and decreases with management.

2774 [Focal Question 5](#)

2775 *5. How do the functions provided by riparian stands change over time (e.g., large woody debris*  
 2776 *recruitment from farther away from the stream)?*

2777 This question addresses a temporal and spatial component to changes in function. The question  
 2778 specifies “change over time” but provides an example with a spatial component. While harvest is  
 2779 not specified as a factor, studies that quantify changes to riparian function in harvested reaches  
 2780 have been included. Studies that compare differences in one or more functions between  
 2781 comparable sites in different successional stages (i.e., different mean age) are also included.  
 2782 Papers that investigate the changes in large wood source distance following harvest have been  
 2783 included because of the given example (*large wood recruitment from farther away from the*  
 2784 *stream*).

2785 *Shade*

2786 Table 18. Treatment and responses for selected publications investigating Shade relevant to Q5.

Reference	Treatment	Response
Kaylor et al. (2017)	old-growth (> 300 years old) and mid-successional (50-60 years old) Douglas-fir dominated forests	the authors estimate that <b>canopy openness reaches its minimum</b> value in regenerating forests <b>at ~30 years and maintains with little variability until ~100 years</b> . Mean canopy openness in stands 30-100 years old was 8.7% with a range from 1.2 to 32.0% (SD = 5.7). Canopy openness over streams in old-growth forests averaged 18.0% but was highly variable and ranged from 3.4 to 34.0% (SD= 5 7.9)
Warren et al. (2013)	old-growth-forests (>500 years old) and young second-growth stands (~40-60 years old)	Three of the four paired <b>old-growth reaches had significantly lower mean percent canopy cover</b>

2787

2788 Kaylor et al. (2017) compared canopy cover throughout stream networks adjacent to old-growth  
 2789 (> 300 years old) and mid-successional (50-60 years old) Douglas-fir dominated forests in the  
 2790 H.J. Andrews Experimental Forest in the Cascade Mountains of Oregon. Canopy openness was  
 2791 quantified with a handheld spherical densiometer. Data was supplemented with a review of  
 2792 literature studies conducted in the Pacific Northwest that reported stand age and canopy cover  
 2793 over the stream. The combined datapoints for canopy openness (%) were plotted against stand  
 2794 age and fit with a negative exponential curve. From the slope of the curve, the authors estimate  
 2795 that canopy openness reaches its minimum value in regenerating forests at ~30 years and  
 2796 maintains with little variability until ~100 years. Mean canopy openness in stands 30-100 years  
 2797 old was 8.7% with a range from 1.2 to 32.0% (standard deviation = 5.7). Canopy openness over  
 2798 streams in old-growth forests averaged 18.0% but was highly variable and ranged from 3.4 to  
 2799 34.0% (standard deviation = 5 7.9).

2800 Warren et al. (2013) compared canopy cover between old-growth-forests (>500 years old) and  
 2801 young second-growth stands (~40-60 years old) in the H.J. Andrews Experimental Forest in the  
 2802 Cascade Mountains of Oregon. Canopy cover was estimated using a convex spherical  
 2803 densiometer. Streams were paired based on reach length, bankfull width, and north (n =2), vs.  
 2804 south (n=2) facing watersheds. Results showed significant differences in percent forest cover  
 2805 between old-growth and second-growth reaches in both south-facing watersheds in mid-summer  
 2806 ( $p < 0.10$ ). For the north-facing watersheds, differences in canopy cover and light availability ( $p$   
 2807  $< 0.10$ ) were only significant at 1 of the two reaches. Overall, three of the four paired old-growth  
 2808 reaches had significantly lower mean percent canopy cover. The authors interpret these results as  
 2809 evidence that old-growth forest canopies were more complex and had more frequent gaps.

2810 *Litter*

2811 Table 19. Treatment and responses for selected publications investigating Litter relevant to Q5.

Reference	Treatment	Response
Kiffney & Richardson (2010)	Upland clearcuts with (1) no buffer, (2) 10 m buffer (~30 feet), (3) 30 m buffers	<u>8 years post-harvest data</u> Differences in litter flux relative to riparian treatment persisted through year 7, while <b>a positive trend between buffer width and litter flux remained through year 8</b> . The linear relationship between reserve width and litter inputs was <b>strongest in the first year</b> after treatment, explaining ~57% of the variation, but the relationship could only explain ~17% of the variation in litter input by buffer width by year 8.
Bilby & Heffner (2016)	Litter samples released from canopy height at one old-growth site and one young forest site.	<u>1-year of data actual age of stands not quantified, estimated by mean height (47.0 and 32.4 m)</u> Needles released at <b>mature sites</b> had a higher proportion of cumulative input from <b>greater distances</b> than needles or alder leaves released at younger sites. The model estimated that the <b>width of the contributing area for needles was ~35% greater at older sites</b> than at younger sites.

2812

2813 Kiffney & Richardson (2010) compared changes in litter input between riparian harvest  
 2814 prescriptions that included clear-cut to stream edge, 10 m wide buffer reserve, 30 m buffer  
 2815 reserves, and an uncut control over the course of 8 years. No thinning was applied within the  
 2816 reserves. Upland treatment at all sites applied clearcut. Results showed differences in litter flux  
 2817 relative to riparian treatment persisted through year 7, while a positive trend between reserve  
 2818 width and litter flux remained through year 8. Needle inputs remained 6x higher in the buffer and  
 2819 control sites through year 7, and 3-6x higher in year 8 than in the clearcut sites. Twig inputs into  
 2820 the control and buffered sites were ~25x higher than in the clearcut sites in the first year after  
 2821 treatment. The linear relationship between reserve width and litter inputs was strongest in the  
 2822 first year after treatment, explaining ~57% of the variation, but the relationship could only  
 2823 explain ~17% of the variation in litter input by buffer width by year 8 (i.e., the relationship  
 2824 degraded over time). The authors interpret these results as evidence that litter flux from riparian  
 2825 plants to streams, was affected by riparian reserve width and time since logging.

2826 Bilby & Heffner (2016) used linear mixed effects models developed for young and old-growth  
 2827 forests of western Washington to estimate controls on litter delivery. Litter samples were released  
 2828 from canopy height at one old-growth forest site and one young forest site. The mean age of  
 2829 “mature” and “young” sites was not specified but the mean tree heights were 47.0 m and 32.4 m  
 2830 for the mature and young sites, respectively. Results showed that needles released at mature sites  
 2831 had a higher proportion of cumulative input from greater distances than needles or alder leaves  
 2832 released at younger sites. The model estimated that the width of the contributing area for needles  
 2833 was ~35% greater at older sites than at younger sites.

2834 *Source distance curves for LW*

2835 Table 20. Treatment and responses for selected publications investigating LW source distance  
 2836 curves relevant to Q5.

Reference	Treatment	Response
Schuett-Hames & Stewart (2019a)	30-ft no-cut buffer width, and thinning 30-75 ft from the stream (SR); retention of all shade providing trees in this area (AAS)	<u>5-years post-harvest</u> Most recruited fallen trees originated in the core zone (0-30 feet; 76%, 72%, and 64% for the reference, AAS and SR groups, respectively), while the proportion from the inner zone ( <b>30–75 feet from the stream</b> ) was <b>~10% greater for the SR group</b> compared to the AAS and REF groups.
Burton et al. (2016)	Buffer widths of 6, 15, or 70 meters	<u>6 years post-harvest</u> <b>82-85%</b> of the wood with discernable sources came from <b>within 15 m of the stream</b> , and the relative contribution of wood to streams <b>declined rapidly with increasing distance</b> .
Martin & Grotenfendt (2007)	Minimum buffer width of 20 m	Recruitment from within 0-20 m of stream was only 17% greater in the treated sites than in the reference sites. However, recruitment from the <b>outer 10 – 20 m was more than double</b> in the buffered units than in the reference units. Estimate that <b>future supply of LW is diminished by ~10% in the treated sites</b> compared to the reference sites.



2837

2838 Schuett-Hames & Stewart (2019a) compared differences in LW recruitment between riparian  
2839 management zones harvested under the current standard Shade Rules (SR), the All-Available  
2840 Shade Rule (AAS), and unharvested references for fish-bearing streams in the mixed conifer  
2841 habitat type (2500 - 5000 feet elevation) for eastern Washington. Both shade rules have a 30-ft  
2842 no-cut buffer (core zone) immediately adjacent to the stream. The SR prescription allows  
2843 thinning in the buffer zone 30-75 feet (inner zone) from the stream while the AAS prescription  
2844 requires retention of all shade providing trees in this area. Results showed that cumulative wood  
2845 recruitment from tree fall after the five-year post-harvest interval was highest in the SR group,  
2846 lower in the AAS group and lowest in the REF group. The SR and AAS LW recruitment rates by  
2847 volume were nearly 300% and 50% higher than the REF rates, respectively. Wood recruitment in  
2848 the SR sites was significantly greater than in the AAS and reference sites. Conversely,  
2849 differences in wood recruitment did not differ significantly between the AAS and reference sites.  
2850 Considering the source distance of post-harvest recruited LW, most recruited fallen trees  
2851 originated in the core zone (76%, 72%, and 64% for the REF, AAS and SR groups, respectively),  
2852 while the proportion from the inner zone (30–75 feet from the stream) was ~10% greater for the  
2853 SR group compared to the AAS and REF groups. These results provide evidence that the  
2854 thinning treatments applied in the inner zone of the SR treatment changed the spatial pattern  
2855 (source distance) of wood recruitment from fallen trees within 5 years post-harvest.

2856 Burton et al. (2016) examined the relationship between annual in-stream wood loading and  
2857 riparian buffer widths adjacent to upland thinning operations. Buffer widths were 6, 15, or 70  
2858 meters and upland thinning was to 200 trees per ha (tph), with a second thinning (~10 years later)  
2859 to ~85 tph, alongside an unthinned reference stand ~400 tph. Data for LW in streams were  
2860 collected for 6 years (5 years after the first harvest and 1 additional year after the second  
2861 harvest). The results showed that between 82-85% of the wood with discernable sources (90%  
2862 for wood in early stages of decay; 45% of wood in late stages of decay) came from within 15 m  
2863 of the stream, and the relative contribution of wood to streams declined rapidly with increasing  
2864 distance.

2865 Martin & Grotenfendt (2007) compared riparian stand mortality and in-stream LW recruitment  
2866 characteristics between riparian buffer strips with upland timber harvest and riparian stands of  
2867 unharvested watersheds using aerial photography. All buffer strips in this study were a minimum  
2868 of 20 m wide and included selective harvest within the 20 m zone (thinning intensity not  
2869 specified or included in the analyses as an effect). The results showed significantly higher  
2870 mortality (based on cumulative stand mortality: downed tree counts divided by standing tree  
2871 counts + downed tree counts), significantly lower stand density (269 trees/ha in buffer units and  
2872 328 trees/ha in reference units), and a significantly higher proportion of LW recruitment from the  
2873 buffer zones of the treatment sites than in the reference sites. LW recruitment based on the  
2874 proportion of stand recruited (PSR) was significantly higher in the buffered units compared to  
2875 the reference units. However, PSR from the inner 0-20 m was only 17% greater in the buffer  
2876 units than in the reference units; while PSR of the outer unit (10 – 20 m) was more than double  
2877 in the buffered units than in the reference units. From their analysis they also estimate that future

2878 potential supply of LW is diminished by ~10% in the buffered sites compared to the reference  
2879 sites.

#### 2880 *LW and stand age*

2881 Jackson and Wohl (2015) compared in-stream wood loads between old-growth (> 200 years) and  
2882 young forests (age not reported). This study took place within the Arapaho and Roosevelt  
2883 National Forests in Colorado. In-stream wood loads (m<sup>3</sup>/ha) were recorded for reaches in 10 old-  
2884 growth forests and 23 young forests. Paired t- test or Kruskal-Wallis tests were used to check for  
2885 significant differences in wood load. Results indicated that channel wood load (OG = 304.4 +  
2886 161.1; Y = 197.8 + 245.5 m<sup>3</sup> /ha), floodplain wood load (OG = 109.4 + 80; Y = 47.1 + 52.8 m<sup>3</sup>  
2887 /ha), and total wood load (OG = 154.7 + 64.1; Y = 87.8 + 100.6 m<sup>3</sup> /ha) per 100 m length of  
2888 stream and were significantly higher in streams of old-growth forests than in young forests.  
2889 Streams in old-growth forests also had significantly more wood in jams, and more total wood  
2890 jams per unit length of channel than in younger forests (jam wood volume: OG = 7.10 +/- 6.9  
2891 m<sup>3</sup>; Y = 1.71 +/- 2.81 m<sup>3</sup>)

#### 2892 *Nutrient dynamics over time*

2893 Vanderbilt et al. (2003) investigated long-term datasets (ranging from 20-30 years) from six  
2894 watersheds in the H.J. Andrews Experimental Watershed (HJA) in the west-central Cascade  
2895 Mountains of Oregon. Their objective was to characterize long-term patterns of N dynamics in  
2896 precipitation and stream water at the HJA. Patterns between nitrogen with precipitation and  
2897 discharge were analyzed with logistic regression. Results showed that dissolved organic nitrogen  
2898 (DON) concentrations increased in the fall in every watershed. The increase in concentration  
2899 began in July or August with the earliest rain events, and peak DON concentrations occurred in  
2900 October through December before the peak in the hydrograph. DON concentrations then  
2901 declined during the winter months. However, other forms of N showed inconsistent patterns  
2902 across all other watersheds. The authors conclude that total annual stream discharge was a  
2903 positive predictor of DON output suggesting a relationship to precipitation. Also, DON had a  
2904 consistent seasonal concentration pattern. All other forms of N observed showed variability and  
2905 inconsistencies with annual and seasonal stream discharge. The authors speculate that different  
2906 factors may control organic vs. inorganic N export. Specifically, DIN may be strongly influenced  
2907 by terrestrial or in-stream biotic controls, while DON is more strongly influenced by climate.  
2908 Last, the authors suggest that DON in streams may be recalcitrant, and largely unavailable to  
2909 stream organisms.

2910

#### 2911 *Focal Question 6*

2912 *6. Are there feedback mechanisms (e.g., microclimate changes within the riparian buffer) related*  
2913 *to forest management that affect the recovery rates of riparian functions?*

2914 The studies considered appropriate for answering this question are those that quantify how forest  
2915 management practices impact one or more factors that can in-turn impact the rate of recovery of  
2916 riparian function. The regeneration, growth and development of vegetation within the riparian

2917 area following treatment can impact the rate of recovery of litter inputs, shade, sediment and  
2918 nutrient filtration. Reduction in shade may affect the amount of light reaching the forest  
2919 understory that then could impact productivity in the riparian area. Also, disturbance of soil and  
2920 removal of vegetation during riparian management operations can impact streamflow and  
2921 sediment supply, which in turn impacts sediment flux into streams. The studies summarized  
2922 below provide experimental evidence in how these factors (e.g., vegetation productivity,  
2923 streamflow discharge, sediment disturbance) are impacted by management.

2924 However, considering the second part of this question on how these feedback mechanisms affect  
2925 the recovery rates of riparian function can only be inferred. To properly answer the full question  
2926 a study would require an experimental design which 1) tracks the changes in site conditions (e.g.,  
2927 microclimate, light availability to groundcover, exposed soil...) after treatment relative to  
2928 untreated stands, 2) evaluates how these changes in site conditions lead to changes in stand  
2929 development that can then impact function (e.g., vegetation), and finally 3) how these changes in  
2930 development affect the recovery rates of function. This third step would require separating out  
2931 the effect of these “feedback mechanism” so that the differences in recovery rates in treated  
2932 stands with and without these effects (e.g., blocking newly available light to the understory) can  
2933 be compared quantitatively. No studies that specifically, and entirely address these 3 objectives  
2934 collectively could be found in the literature. Thus, the following reviewed studies provide  
2935 evidence of how feedback mechanisms can affect function (e.g., increased light = increased  
2936 primary productivity), but how these mechanisms affect the recovery rates of any particular  
2937 function (e.g., timing of recovery with and without the feedback mechanism) can only be  
2938 assumed.

### 2939 *Litter*

2940 Yeung et al. (2019) simulated post-harvest responses to leaf-litter derived coarse particulate  
2941 organic matter (CPOM) quantity in a coastal rainforest stream in British Columbia. This study  
2942 used a CPOM model that was calibrated using data from multiple published studies from,  
2943 primarily the Pacific Northwest region, and several other North American regions. Calibration  
2944 data included stream flow and temperature, and CPOM following different timber harvest  
2945 intensities within 4 years of harvest. The model used estimated litterfall decreases of (-10%, -  
2946 30%, -50%, -90%) for low, moderate, high, and very high basal area removal ; peak streamflow  
2947 increases of +20%, +40%, +100%, +300%); and stream temperature increases of +1°C, +2°C,  
2948 +4°C, and +6 °C. Treatment intensities in litterfall, peak flow, and stream temperature were  
2949 modeled and analyzed individually and cumulatively to estimate their relative and combined  
2950 effects on in-stream CPOM standing stocks. Results of the model showed that, in general, the  
2951 standing stocks of CPOM decreased under the independent effects of reduced litterfall and  
2952 elevated peak flows and increased with higher stream temperatures.

2953 Along the gradient of increasing timber removal, litterfall reductions on depleting CPOM  
2954 standing stocks were at least an order of magnitude greater than those of elevated peak flows.  
2955 The magnitude of CPOM changes induced by litterfall reductions was consistently greater than  
2956 stream temperature increases, but their differences in magnitude became smaller at higher levels  
2957 of disturbance severity. Only the effects of litterfall-temperature interactions on CPOM standing

2958 stocks were significant ( $p < 0.001$ ). The authors interpret these results as evidence that litterfall  
 2959 reduction from timber harvest was the strongest control on in-stream CPOM quantity for 4 years  
 2960 post-harvest. However, the authors propose that the decreased activity of CPOM consumers  
 2961 caused by increasing stream temperatures may be enough to offset the loss of litterfall inputs on  
 2962 standing CPOM stocks. The caveat of this study is that it did not include LW dynamics in  
 2963 preserving CPOM post-harvest. There is evidence that in-stream LW can act as a catchment for  
 2964 CPOM (May & Gresswell, 2003; Richardson et al. 2007).

2965 *Sediment*

2966 Table 21. Treatment and responses for selected publications investigating Sediment relevant to  
 2967 Q5.

Reference	Treatment	Response
Safeeq et al. (2020)	Long-term dataset with mixture of management, storm events, and	estimate that following harvest, <b>changes on streamflow alone was estimated in being responsible for &lt; 10%</b> of the resulting suspended sediment transported into streams, while the increase in sediment supply <b>due to harvest disturbance was responsible for &gt;90%</b> .
Litschert & MacDonald (2009)	Post-harvest stream sediment delivery pathway development frequency.	<u>1-year post-harvest data (n = 200 harvest units)</u> The authors conclude that in general, USFS riparian forest harvest practices are effective in reducing the development of sediment delivery pathways. They also interpret these results as <b>evidence that skid trails should be directed away from streams</b> , maintain surface roughness, and promptly decommissioned.

2968

2969 Safeeq et al. (2020) analyzed a long-term data set to changes in streamflow, and suspended  
 2970 sediment load and sediment bedload in streams between two watersheds; one with a history of  
 2971 timber management and one with no history of timber management. The two watersheds were  
 2972 located in the H.J. Andrews Experimental Forest and were paired by size, aspect, and  
 2973 topography. The treatment watershed was 100% clearcut during the period from 1962-1966,  
 2974 broadcast burned in 1966 and re-seeded in 1968. Streamflow and sediment data were taken  
 2975 intermittently; suspended sediment data after large storm events between 1952 (pre-harvest) and  
 2976 1988; and sediment bedload in 2016. The researchers used a reverse regression technique to  
 2977 evaluate the relative and absolute importance of changes in streamflow versus changes in  
 2978 sediment supply from timber harvest on sediment transport. There were no significant changes in  
 2979 precipitation patterns before or after harvest. The results for post-treatment sediment yields  
 2980 showed suspended load declined to pre-treatment levels in the first two decades following  
 2981 treatment and bedload remained elevated, causing the bedload proportion of the total load to  
 2982 increase through time. Changes in streamflow alone account for 477 Mg/km<sup>2</sup> (10%) of the  
 2983 suspended load and 113 Mg/km<sup>2</sup> (5%) of the bedload over the post-treatment period. Increase in  
 2984 suspended sediment yield due to increase in sediment supply from timber harvest activities was  
 2985 84% of the measured post-treatment total suspended sediment yield. The authors estimate that  
 2986 following harvest, changes on streamflow alone was estimated in being responsible for < 10% of

2987 the resulting suspended sediment transported into streams, while the increase in sediment supply  
2988 due to harvest disturbance was responsible for >90%. Thus, while timber harvest-induced  
2989 increases in streamflow does increase sediment transport, it is negligible compared to the  
2990 increase in sediment source created from management practices.

2991 Litschert & MacDonald (2009) investigated the frequency of sediment delivery pathways in  
2992 riparian management areas and their physical characteristics and connectivity following harvest.  
2993 In this study the authors describe sediment delivery pathways (“features”) as rills, gullies, and  
2994 sediment plumes that form when excess sediment relative to overland flows transports sediment  
2995 from the hillslope to the stream. The authors surveyed 200 riparian management areas (RMA) in  
2996 four different National Forests of the Sierra Nevada and Cascade Mountains of California. USFS  
2997 policy requires 90-m wide RMA along each side of perennial streams and 45-m wide RMA along  
2998 each side of all ephemeral and intermittent streams. When features were found within an RMA,  
2999 data for years since harvest, soil depth, soil erodibility (K), feature length, feature gradient,  
3000 aspect, elevation, hillslope gradient, hillslope curvature, surface roughness, and connectivity  
3001 were recorded for analysis. Association between these variables were analyzed with a  
3002 Spearman’s rank correlation. The variables most strongly associated with feature length were  
3003 used to develop a multiple linear regression model to predict feature length. Only 19 of the 200  
3004 harvest units had sediment development pathways. Feature pathways ranged in age (time since  
3005 harvest) from 2 to 18 years, and in length from 10 m to 220 m. Of the 19 feature pathways, only  
3006 six were connected to streams, and five of those originated from skid trails. Feature pathway  
3007 length was significantly related to mean annual precipitation, cosine of the aspect, elevation, and  
3008 hillslope gradient ( $R^2 = 64\%$ ,  $p = 0.004$ ). The authors conclude that in general, USFS riparian  
3009 forest harvest practices are effective in reducing the development of sediment delivery pathways.  
3010 They also interpret these results as evidence that skid trails should be directed away from  
3011 streams, maintain surface roughness, and promptly decommissioned.

3012

### 3013 *Impacts on Microclimate*

3014 Anderson et al. (2007) compared changes in understory microclimate above the stream, within  
3015 the channel, and within the riparian area between thinned and unthinned riparian stands. The  
3016 focus of this study was on second-growth (30- to 80-year-old) riparian Douglas-fir forests along  
3017 headwater streams in the western Oregon Coast and Cascade Range. Stands were either thinned  
3018 to approximately 198 trees per acre (TPA) or were left unthinned and ranged from 500-865 TPA.  
3019 Streams within treated stands were surrounded by buffers of either 1) one site-potential tree  
3020 averaging 69 m (B1, B1-T thinned and unthinned respectively), 2) variable width buffer  
3021 averaging 22 m (VB, and VB-T), or 3) streamside retention buffer averaging 9 m (SR, and SR-  
3022 T). Further, directly adjacent randomly selected B1-T and VB-T buffers patch openings (0.4 ha)  
3023 were created (B1-P, VB-P). Microsite and microclimate responses were repeat sampled for each  
3024 treatment and compared with untreated stands (UT). Within the riparian buffer zones, daily  
3025 maximum temperatures were higher in all treated stands when compared to UT stands. The  
3026 differences in daily maximum temperatures between treated and untreated stands ranged from  
3027 1.1°C (B1) to 4.0°C (SR-T), but the difference was only significant in one SR-T stand. Daily  
3028 maximum air temperature within buffer zones adjacent to patch openings were 3.5°C higher than

3029 in UT stands. Within patch openings daily maximum temperatures were on average 6 to 9°C  
3030 higher than in UT stands. Soil temperature changes were only evident within patch openings  
3031 ranging from 3.6 - 8.8°C higher than in UT stands. VB-T buffers that were 15 m wide or wider  
3032 exhibited changes in daily maximum air temperature above stream centers <1°C and daily  
3033 minimum relative humidity <5% lower than in untreated stands. The authors conclude that in  
3034 general, thinned stands are warmer and drier than unthinned stands. However, the results for  
3035 differences in microclimate were only significant in narrow (9 m) thinned buffers and patch  
3036 openings.

3037 Anderson & Meleason (2009) conducted a companion study to Anderson et al. (2007) and  
3038 compared changes in small (5-29 cm diameter) and large ( $\geq 30$  cm diameter) downed wood  
3039 abundance and understory vegetation between treated and untreated stands 5 years after harvest.  
3040 Treatments compared were the same as those described in Anderson et al. (2007) discussed  
3041 above. The results for small and large downed wood were highly variable between pre- and post-  
3042 harvest periods and between treatments but the authors speculate from trends in the data that  
3043 both wood and vegetation responses within buffers  $\geq 15$  m wide were insensitive to treatments.  
3044 The strongest contrast in rate of change in herb cover was between the SR-T and VB-T buffers  
3045 with higher herbaceous cover in the SR-T buffers and highest in SR-T buffers adjacent to patch  
3046 openings. The authors conclude that in general these thinning treatments only led to subtle  
3047 changes in understory vegetation cover and composition. Because of the high variability in  
3048 responses among and between treatments significance could not be confirmed. The authors  
3049 further conclude that a better functional understanding of the changes in ecological processes  
3050 associated with changes in habitat characteristics following changes in understory wood and  
3051 vegetation cover is needed to help discern ecological significance.

3052

### 3053 [Focal Question 7](#)

3054 *7. What major data gaps and uncertainties exist relative to effects of timber harvest (both*  
3055 *riparian and adjacent upland) on the riparian functions?*

3056 No studies that provide experimental evidence that quantifies how specific treatments within the  
3057 riparian area affect bank stability were found based on our search criteria (published after 2000,  
3058 conducted in western North America). However, this may be because bank erosion relates  
3059 directly to sediment transport and thus bank stability is inferred by the magnitude of change in  
3060 sediment export. Furthermore, the importance of vegetation retention and equipment exclusion in  
3061 areas closest to the stream for maintaining bank stability appears to be well understood  
3062 considering its prevalence in riparian forest management plans ([WAC 222-30-022](#); WAC 22-30-  
3063 [021](#); 2022 [ODF](#); IDAPA 20.[02.01](#)).

3064 Our search of the literature focused on how treatments within or adjacent to forested riparian  
3065 areas impact one or more of the riparian functions. Most of the studies found in our search focus  
3066 on the impacts of riparian treatment on LW and shade (commonly coupled with stream  
3067 temperature). There is also a significant body of research that considers the impact of harvest on

3068 nutrient and sediment flux into streams. Fewer studies could be found that quantify changes in  
3069 litter input following riparian management.

3070 While few studies could be found that provide direct experimental evidence of how bank  
3071 stability is affected by timber harvest, two studies were found that compared the relative  
3072 influence of different factors on bank stability. Both of which showed evidence that bank  
3073 stability is influenced by the type of vegetation dominating the riparian area. Rood et al. (2015)  
3074 compared the relative erosion resistance of riverbanks occupied by forests versus grassland along  
3075 the Elk River in British Columbia, Canada. This study used a combination of field sampling and  
3076 aerial photo analysis from 1995 to 2013 to estimate the differences in channel migration between  
3077 forest and grass dominated riparian areas. Relative tree cover was binned into 5 categories  
3078 ranging from (1) no trees to (5) completely treed. Relative channel change was binned into 2  
3079 categories as ‘moderate change’ for channels that migrated between 45 and 75 m, and as ‘major  
3080 change’ for channels that migrated more than 75 m. Chi square analysis was used to assess the  
3081 distributions of vegetation of channels with moderate and major changes. Results of the chi  
3082 square analysis showed that the distribution of the observed vegetation types differed  
3083 significantly ( $p < 0.05$ ) by channel change categories. Of the 15 sites assessed with moderate or  
3084 major erosion (changes), 7 were along banks dominated by grasslands without trees (‘1’), four  
3085 were assessed as a ‘2’, with some trees, and three were in a ‘3’ with a mixed zone of similar  
3086 proportions of trees and clearing. Only one site with a ‘4’ showed a moderate amount of change.  
3087 The authors interpret these results as evidence that trees are better than grass at stabilizing banks,  
3088 and that stability increases with tree cover.

3089 Outside of the U.S., Krzeminska et al. (2019), investigated the effect of different types of  
3090 riparian vegetation on stream bank stability in a small agricultural catchment in South-Eastern  
3091 Norway. The dominating soil type within the catchment is coarse moraine in the forested areas  
3092 and marine deposits with silt loam and silty clay loam texture in agriculture areas. The  
3093 researchers used a combination of field collected data with stream bank stability modeling using  
3094 Bank-Stability and Toe-Erosion Modeling (BSTEM). Three experimental plots were established,  
3095 one for each dominant vegetation type, grass dominated, shrub dominated, and tree dominated.  
3096 Investigations of in-situ undrained shear strength of the root-reinforced soil were done with a  
3097 Field Inspection Vane Tester. Additionally, potential changes in the bank profile were monitored  
3098 with a series of erosion pins, 6 pins per each plot. Changes in root cohesion and % cover over  
3099 time for each vegetation type were estimated using the RipRoots sub-model in BSTEM. Their  
3100 results showed a difference in bank stability based on vegetation type, that varied seasonally with  
3101 groundwater level and stream water level. The grass dominated and tree dominated plots,  
3102 specifically, showed the lowest estimated stability during spring (March to April) and early  
3103 autumn (September to November), and the highest estimated stability during the summer months  
3104 (May-June). This seasonal trend was also observed for the shrub plots but not as strongly.  
3105 Steeper slopes in the grass and shrub dominated plots showed a trend of reduced stability for  
3106 plots 54° slopes showing potential for failure. The tree dominated plots showed a trend of lower  
3107 stability for steeper slopes, however, it wasn’t as strong of a trend and the model did not predict  
3108 potential for failure or ‘instability’. Regardless of season, groundwater levels, or slope steepness  
3109 the tree plots showed the highest estimated bank stability overall.

3110 These two studies that investigate bank stability use methods which could be applied to an  
3111 experimental design that also considers differences in stability between treated (harvested) and  
3112 untreated stands. The combination of field observation and simulation modeling used by  
3113 Krzeminska et al. (2019), especially, could be used to estimate how timber harvest affects bank  
3114 stability (or erosion) while also accounting for geomorphic and hydrological differences.

3115 Considering the topics included in the focal questions, studies that investigate the effects of  
3116 clearcut gaps, and studies that quantify how treatment within the riparian zone relates to  
3117 resilience to fire had the fewest studies providing experimental evidence. Other than the Hard  
3118 Rock and Soft Rock studies, only 2 other studies (Janisch et al., 2012, Swartz et al., 2020) were  
3119 found that investigate the effects of similar buffer treatment designs (patched buffers and riparian  
3120 canopy gaps). For how treatments within the riparian zone relate to resilience to fire, there were  
3121 no studies that provide experimental evidence on this topic based on the search criteria. Some  
3122 studies were found to quantify the probability of fire or fire severity within riparian zones in  
3123 general (Reeves et al. 2006; Van de Water & North, 2011). However, none compares the  
3124 resilience of riparian stands between treated and untreated stands after fire. One study, Ceder et  
3125 al. (2018) used simulation modeling to compare fire susceptibility between managed and  
3126 unmanaged stands and has been included in focal question 4.

3127 Indeed, Stone et al. (2010) surveyed fire management officers from 55 national forests across 11  
3128 western states and found that fewer than half (43%) of them indicated that they were conducting  
3129 fuel reduction treatments in riparian areas. The primary objective for most of these treatments  
3130 involved some form of fuel reduction (83%), while others focused on multiple objectives such as  
3131 ecological restoration and habitat improvement. Most of these treatments (93%) were of small  
3132 extent (< 300 acres) and occurred in the wildland urban interface (73%). The authors conclude  
3133 that these results are promising, but that well-designed monitoring programs are needed to  
3134 estimate the consequences of these treatments on fire risk and other ecological effects.

3135 The study from Prichard et al. (2020), discussed in question 3, used a combination of  
3136 simultaneous autoregression (SAR) and random forest (RF) modeling approaches to model the  
3137 drivers of fire severity and the effectiveness of fuel treatments in mitigating fire severity in the  
3138 2014 Carlton Complex. Results from this study provided evidence on how vegetation (based on  
3139 broad LANDFIRE classifications), topography, and different fuel treatments (e.g., thinning only,  
3140 thin and pile burn, thin and broadcast burn, etc.) related to fire severity and fire spread. This  
3141 approach has potential to be used in riparian areas burned by wildfires. In terms of the topic of  
3142 how various treatments relate to riparian forest resistance and resilience to fire would require  
3143 using a dataset of riparian forest stand characteristics that includes information on fuel  
3144 treatments, time since last fire, and basin characteristics. This information could be used along  
3145 with spatial information of burn severity immediately following a fire.

## 3146 [References](#)

3147 Anderson, P. D., & Meleason, M. A. (2009). Discerning responses of down wood and understory  
3148 vegetation abundance to riparian buffer width and thinning treatments: an equivalence–inequivalence  
3149 approach. *Canadian Journal of Forest Research*, 39(12), 2470-2485.



- 3150 Anderson, P. D., Larson, D. J., & Chan, S. S. (2007). Riparian buffer and density management influences on  
3151 microclimate of young headwater forests of western Oregon. *Forest Science*, 53(2), 254-269.
- 3152 Arkle, R. S., & Pilliod, D. S. (2010). Prescribed fires as ecological surrogates for wildfires: a stream and  
3153 riparian perspective. *Forest Ecology and Management*, 259(5), 893-903.
- 3154 Bahuguna, D., Mitchell, S. J., & Miquelajauregui, Y. (2010). Windthrow and recruitment of large woody  
3155 debris in riparian stands. *Forest Ecology and Management*, 259(10), 2048-2055.
- 3156 Benda, L. E., Litschert, S. E., Reeves, G., & Pabst, R. (2016). Thinning and in-stream wood recruitment in  
3157 riparian second growth forests in coastal Oregon and the use of buffers and tree tipping as  
3158 mitigation. *Journal of forestry research*, 27(4), 821-836.
- 3159 Benda, L., Miller, D. A. N. I. E. L., Sias, J. O. A. N., Martin, D. O. U. G. L. A. S., Bilby, R., Veldhuisen, C., &  
3160 Dunne, T. (2003, January). Wood recruitment processes and wood budgeting. In *American Fisheries  
3161 Society Symposium* (pp. 49-74). American Fisheries Society.
- 3162 Beschta, R. L., Bilby, R.E.Brown, G.W., Holtby, L.B., and Hofstra., T.D., 1987. Stream Temperature and  
3163 Aquatic Habitat: Fisheries and Forestry Interactions. In: *Streamside Management: Forestry and Fisheries  
3164 Interactions*. E. O. Salo and T. W. Cundy (Editors). Contribution No. 57, University of Washington, Institute  
3165 of Forest Resources, 471 pp.
- 3166 Bilby, R. E., & Heffner, J. T. (2016). Factors influencing litter delivery to streams. *Forest Ecology and  
3167 Management*, 369, 29–37. <https://doi.org/10.1016/j.foreco.2016.03.031>
- 3168 Bilby, R. E., Sullivan, K., & Duncan, S. H. (1989). The generation and fate of road-surface sediment in  
3169 forested watersheds in southwestern Washington. *Forest Science*, 35(2), 453-468.
- 3170 Bjornn, T. C., & Reiser, D. W. (1991). Habitat requirements of salmonids in streams. *American Fisheries  
3171 Society Special Publication*, 19(837), 138.
- 3172 Bladon, K. D., Cook, N. A., Light, J. T., & Segura, C. (2016). A catchment-scale assessment of stream  
3173 temperature response to contemporary forest harvesting in the Oregon Coast Range. *Forest Ecology and  
3174 Management*, 379, 153-164.
- 3175 Bladon, K. D., Segura, C., Cook, N. A., Bywater-Reyes, S., & Reiter, M. (2018). A multicatchment analysis of  
3176 headwater and downstream temperature effects from contemporary forest harvesting. *Hydrological  
3177 Processes*, 32(2), 293-304.
- 3178 Brown, G. W. (1983). *Forestry and water quality*. *Forestry and water quality*, (Ed. 2).
- 3179 Brown, G. W., Gahler, A. R., & Marston, R. B. (1973). Nutrient losses after clear-cut logging and slash  
3180 burning in the Oregon Coast Range. *Water Resources Research*, 9(5), 1450-1453.
- 3181 Burton, J. I., Olson, D. H., & Puettmann, K. J. (2016). Effects of riparian buffer width on wood loading in  
3182 headwater streams after repeated forest thinning. *Forest Ecology and Management*, 372, 247–257.  
3183 <https://doi.org/10.1016/j.foreco.2016.03.053>
- 3184 Bywater-Reyes, S., Bladon, K. D., & Segura, C. (2018). Relative influence of landscape variables and  
3185 discharge on suspended sediment yields in temperate mountain catchments. *Water Resources  
3186 Research*, 54(7), 5126-5142.

- 3187 Bywater-Reyes, S., Segura, C., & Bladon, K. D. (2017). Geology and geomorphology control suspended  
3188 sediment yield and modulate increases following timber harvest in temperate headwater  
3189 streams. *Journal of Hydrology*, 548, 754-769.
- 3190 Camp, A., C. Oliver, P. Hessburg, and R. Everett. (1997). Predicting late-successional fire refugia pre-dating  
3191 European settlement in the Wenatchee Mountains. *Forest Ecology and Management* 95 63-77
- 3192 Chan, S., P. Anderson, J. Cissel, L. Lateen. and C. Thompson. 2004. Variable density management in  
3193 Riparian Reserves: lessons learned from an operational study in managed forests of western Oregon,  
3194 USA. *For. Snow Landsc. Res.* 78,1/2:151-172.
- 3195 Chapman, D. W., & Bjornn, T. C. (1969). Distribution of salmonids in streams. In *Symp. Salmon Trout*  
3196 *Streams*. Institute of Fisheries, University of British Columbia, Vancouver (pp. 153-176).
- 3197 Chen, X., Wei, X., & Scherer, R. (2005). Influence of wildfire and harvest on biomass, carbon pool, and  
3198 decomposition of large woody debris in forested streams of southern interior British Columbia. *Forest*  
3199 *Ecology and Management*, 208(1-3), 101-114.
- 3200 Chen, X., Wei, X., Scherer, R., Luider, C., & Darlington, W. (2006). A watershed scale assessment of in-  
3201 stream large woody debris patterns in the southern interior of British Columbia. *Forest Ecology and*  
3202 *Management*, 229(1-3), 50-62.
- 3203 Chesney, C. (2000). Functions of wood in small, steep streams in eastern Washington: Summary of  
3204 results for project activity in the Ahtanum, Cowiche, and Tieton basins. *Timber, Fish, Wildlife*.
- 3205 CMER 03-308 (2004) Review of the Available Literature Related to Wood Loading Dynamics in and  
3206 around Stream in eastern Washington Forests
- 3207 Cole, E., & Newton, M. (2013). Influence of streamside buffers on stream temperature response  
3208 following clear-cut harvesting in western Oregon. *Canadian journal of forest research*, 43(11), 993-1005.
- 3209 Cooper, J. R., Gilliam, J. W., Daniels, R. B., & Robarge, W. P. (1987). Riparian areas as filters for agricultural  
3210 sediment. *Soil science society of America journal*, 51(2), 416-420.
- 3211 Crandall, T., Jones, E., Greenhalgh, M., Frei, R. J., Griffin, N., Severe, E., ... & Abbott, B. W. (2021).  
3212 Megafire affects stream sediment flux and dissolved organic matter reactivity, but land use dominates  
3213 nutrient dynamics in semiarid watersheds. *PloS one*, 16(9), e0257733.
- 3214 Cupp, C.E. and T.J. Lofgren. 2014. Effectiveness of riparian management zone prescriptions in protecting  
3215 and maintaining shade and water temperature in forested streams of Eastern Washington. *Cooperative*  
3216 *Monitoring Evaluation and Research Report CMER 02-212*. Washington State Forest Practices Adaptive  
3217 *Management Program*. Washington Department of Natural Resources, Olympia, WA.
- 3218 Deval, C., Brooks, E. S., Gravelle, J. A., Link, T. E., Dobre, M., & Elliot, W. J. (2021). Long-term response in  
3219 nutrient load from commercial forest management operations in a mountainous watershed. *Forest*  
3220 *Ecology and Management*, 494, 119312.
- 3221 Devotta, D. A., Fraterrigo, J. M., Walsh, P. B., Lowe, S., Sewell, D. K., Schindler, D. E., & Hu, F. S. (2021).  
3222 Watershed *Alnus* cover alters N: P stoichiometry and intensifies P limitation in subarctic  
3223 streams. *Biogeochemistry*, 153(2), 155-176.

- 3224 Dwire, K. A., & Kauffman, J. B. (2003). Fire and riparian ecosystems in landscapes of the western USA.  
3225 *Forest Ecology and Management*, 178(1-2), 61-74.
- 3226 Ebersole, J. L., Liss, W. J., & Frissell, C. A. (2001). Relationship between stream temperature, thermal  
3227 refugia and rainbow trout *Oncorhynchus mykiss* abundance in arid-land streams in the northwestern  
3228 United States. *Ecology of freshwater fish*, 10(1), 1-10.
- 3229 Ehinger, W.J., W.D. Bretherton, S.M. Estrella, G. Stewart, D.E. Schuett-Hames, and S.A. Nelson. 2021.  
3230 Effectiveness of Forest Practices Buffer Prescriptions on Perennial Non-fish-bearing Streams on Marine  
3231 Sedimentary Lithologies in Western Washington. Cooperative Monitoring, Evaluation, and Research  
3232 Committee Report CMER 2021.08.24, Washington State Forest Practices Adaptive Management  
3233 Program, Washington Department of Natural Resources, Olympia, WA.
- 3234 Everett, R., Schellhaas, R., Ohlson, P., Spurbeck, D., & Keenum, D. (2003). Continuity in fire disturbance  
3235 between riparian and adjacent sideslope Douglas-fir forests. *Forest Ecology and Management*, 175(1-3),  
3236 31-47.
- 3237 Fox, M. J. (2001). A new look at the quantities and volumes of instream wood in forested basins within  
3238 Washington State (Doctoral dissertation, University of Washington).
- 3239 Fox, M., & Bolton, S. (2007). A regional and geomorphic reference for quantities and volumes of instream  
3240 wood in unmanaged forested basins of Washington State. *North American Journal of Fisheries*  
3241 *Management*, 27(1), 342-359.
- 3242 Fox, M., Bolton, S., & Conquest, L. (2003). 14Reference conditions for instream wood in western  
3243 Washington. University of Washington Press: Seattle, WA, 361-393.
- 3244 Fratkin, M. M., Segura, C., & Bywater-Reyes, S. (2020). The influence of lithology on channel geometry  
3245 and bed sediment organization in mountainous hillslope-coupled streams. *Earth Surface Processes and*  
3246 *Landforms*, 45(10), 2365-2379.
- 3247 Fredriksen, R. L. (1975). Nitrogen, phosphorus and particulate matter budgets of five coniferous forest  
3248 ecosystems in the western Cascades Range, Oregon.
- 3249 Gomi, T., Dan Moore, R., & Hassan, M. A. (2005). Suspended sediment dynamics in small forest streams  
3250 of the Pacific Northwest 1. *JAWRA Journal of the American Water Resources Association*, 41(4), 877-898.
- 3251 Gomi, T., Sidle, R. C., Bryant, M. D., & Woodsmith, R. D. (2001). The characteristics of woody debris and  
3252 sediment distribution in headwater streams, southeastern Alaska. *Canadian Journal of Forest*  
3253 *Research*, 31(8), 1386-1399.
- 3254 Gravelle, J. A., & Link, T. E. (2007). Influence of timber harvesting on headwater peak stream  
3255 temperatures in a northern Idaho watershed. *Forest Science*, 53(2), 189-205.
- 3256 Gravelle, J. A., Ice, G., Link, T. E., & Cook, D. L. (2009). Nutrient concentration dynamics in an inland  
3257 Pacific Northwest watershed before and after timber harvest. *Forest Ecology and Management*, 257(8),  
3258 1663-1675.
- 3259 Gregory, S. (1990). Riparian management guide: Willamette National Forest. US Department of  
3260 Agriculture, Forest Service, Pacific Northwest Region.

- 3261 Gresswell, R. E., Barton, B. A., & Kershner, J. L. (Eds.). (1989). *Practical approaches to riparian resource*  
3262 *management: an educational workshop: May 8-11, 1989, Billings, Montana*. US Bureau of Land  
3263 Management.
- 3264 Groom, J. D., Dent, L., Madsen, L. J., & Fleuret, J. (2011b). Response of western Oregon (USA) stream  
3265 temperatures to contemporary forest management. *Forest Ecology and Management*, 262(8), 1618-  
3266 1629.
- 3267 Groom, J. D., L. Dent, and L. J. Madsen. (2011a). Stream temperature change detection for state and  
3268 private forests in the Oregon Coast Range, *Water Resour. Res.* 47, W01501
- 3269 Guenther, S. M., Gomi, T., & Moore, R. D. (2014). Stream and bed temperature variability in a coastal  
3270 headwater catchment: influences of surface-subsurface interactions and partial-retention forest  
3271 harvesting. *Hydrological Processes*, 28(3), 1238-1249.
- 3272 Harmon, M. E., Franklin, J. F., Swanson, F. J., Sollins, P., Gregory, S. V., Lattin, J. D., ... & Cummins, K. W.  
3273 (1986). Ecology of coarse woody debris in temperate ecosystems. *Advances in ecological research*, 15,  
3274 133-302.
- 3275 Hart, Stephanie K., David E. Hibbs, and Steven S. Perakis. (2013) "Riparian litter inputs to streams in the  
3276 central Oregon Coast Range." *Freshwater Science* 32.1 (2013): 343-358.
- 3277 Hartman, G. F., & Scrivener, J. C. (1990). Impacts of forestry practices on a coastal stream ecosystem,  
3278 Carnation Creek, British Columbia.
- 3279 Hassan, M. A., Church, M., Lisle, T. E., Brardinoni, F., Benda, L., & Grant, G. E. (2005). Sediment transport  
3280 and channel morphology of small, forested streams 1. *Jawra journal of the american water resources*  
3281 *association*, 41(4), 853-876.
- 3282 Hatten, J. A., Segura, C., Bladon, K. D., Hale, V. C., Ice, G. G., & Stednick, J. D. (2018). Effects of  
3283 contemporary forest harvesting on suspended sediment in the Oregon Coast Range: Alsea Watershed  
3284 Study Revisited. *Forest Ecology and Management*, 408, 238-248.
- 3285 Hoffmann, C. C., Kjaergaard, C., Uusi-Kämppe, J., Hansen, H. C. B., & Kronvang, B. (2009). Phosphorus  
3286 Retention in Riparian Buffers: Review of Their Efficiency. *Journal of Environmental Quality*, 38(5), 1942-  
3287 1955. <https://doi.org/10.2134/jeq2008.0087>
- 3288 Hough-Snee, N., Kasprak, A., Rossi, R. K., Bouwes, N., Roper, B. B., & Wheaton, J. M. (2016).  
3289 Hydrogeomorphic and Biotic Drivers of Instream Wood Differ Across Sub-basins of the Columbia River  
3290 Basin, USA. *River Research and Applications*, 32(6), 1302-1315. <https://doi.org/10.1002/rra.2968>
- 3291 Hunter, M. A., & Quinn, T. (2009). Summer water temperatures in alluvial and bedrock channels of the  
3292 Olympic Peninsula. *Western Journal of Applied Forestry*, 24(2), 103-108.
- 3293 Hyatt, T. L., & Naiman, R. J. (2001). The residence time of large woody debris in the Queets River,  
3294 Washington, USA. *Ecological Applications*, 11(1), 191-202.
- 3295 Jackson, C. R., C. A. Sturm, and J. M. Ward. 2001. Timber harvest impacts on small headwater stream  
3296 channels in the coast ranges of Washington. *JAWRA Journal of the American Water Resources*  
3297 *Association* 37(6):1533-1549.

- 3298 Jackson, K. J., & Wohl, E. (2015). Instream wood loads in montane forest streams of the Colorado Front  
3299 Range, USA. *Geomorphology*, 234, 161-170.
- 3300 Janisch, J. E., Wondzell, S. M., & Ehinger, W. J. (2012). Headwater stream temperature: Interpreting  
3301 response after logging, with and without riparian buffers, Washington, USA. *Forest Ecology and*  
3302 *Management*, 270, 302-313.
- 3303 Johnson, S. L., & Jones, J. A. (2000). Stream temperature responses to forest harvest and debris flows in  
3304 western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences*, 57(S2), 30-39.
- 3305 Karwan, D. L., Gravelle, J. A., & Hubbart, J. A. (2007). Effects of timber harvest on suspended sediment  
3306 loads in Mica Creek, Idaho. *Forest Science*, 53(2), 181-188.
- 3307 Kaylor, M. J., Warren, D. R., & Kiffney, P. M. (2017). Long-term effects of riparian forest harvest on light in  
3308 Pacific Northwest (USA) streams. *Freshwater Science*, 36(1), 1-13.
- 3309 Kibler, K. M., Skaugset, A., Ganio, L. M., & Huso, M. M. (2013). Effect of contemporary forest harvesting  
3310 practices on headwater stream temperatures: Initial response of the Hinkle Creek catchment, Pacific  
3311 Northwest, USA. *Forest ecology and management*, 310, 680-691.
- 3312 Kiffney, P., and J. Richardson. 2010. Organic matter inputs into headwater streams of southwestern  
3313 British Columbia as a function of riparian reserves and time since harvesting. *Forest Ecology and*  
3314 *Management*. 260:1931-1942.
- 3315 Knight, S.M. (1990). Forest harvesting impacts on coarse woody debris and channel form in central  
3316 Oregon streams. M.S. Thesis. Oregon State University, Corvallis, Oregon.
- 3317 Lewis, D. D. (1998, April). Naive (Bayes) at forty: The independence assumption in information retrieval.  
3318 In *European conference on machine learning* (pp. 4-15). Berlin, Heidelberg: Springer Berlin Heidelberg.
- 3319 Liquori, M. K. (2006). POST-HARVEST RIPARIAN BUFFER RESPONSE: IMPLICATIONS FOR WOOD  
3320 RECRUITMENT MODELING AND BUFFER DESIGN 1. *JAWRA Journal of the American Water Resources*  
3321 *Association*, 42(1), 177-189.
- 3322 Litschert, S. E., & MacDonald, L. H. (2009). Frequency and characteristics of sediment delivery pathways  
3323 from forest harvest units to streams. *Forest Ecology and Management*, 259(2), 143-150.
- 3324 Macdonald, J. S., Beaudry, P. G., MacIsaac, E. A., & Herunter, H. E. (2003). The effects of forest harvesting  
3325 and best management practices on streamflow and suspended sediment concentrations during  
3326 snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada. *Canadian Journal of*  
3327 *Forest Research*, 33(8), 1397-1407.
- 3328 Macdonald, J. S., MacIsaac, E. A., & Herunter, H. E. (2003b). The effect of variable-retention riparian  
3329 buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of  
3330 British Columbia. *Canadian journal of forest research*, 33(8), 1371-1382.
- 3331 Martin, D. J., & Grotefendt, R. A. (2007). Stand mortality in buffer strips and the supply of woody debris  
3332 to streams in Southeast Alaska. *Canadian Journal of Forest Research*, 37(1), 36-49.

- 3333 Martin, D. J., Kroll, A. J., & Knoth, J. L. (2021). An evidence-based review of the effectiveness of riparian  
3334 buffers to maintain stream temperature and stream-associated amphibian populations in the Pacific  
3335 Northwest of Canada and the United States. *Forest Ecology and Management*, 491, 119190.
- 3336 May, C. L., & Gresswell, R. E. (2003). Large wood recruitment and redistribution in headwater streams in  
3337 the southern Oregon Coast Range, USA. *Canadian Journal of Forest Research*, 33(8), 1352-1362.
- 3338 McDade, M, F. Swanson, W. McKee, J. Franklin and J. Van Sickle. (1990). Source distances for coarse  
3339 woody debris entering small streams in western Oregon and Washington. *Canadian Journal of Forest*  
3340 *Resources* 20, 326-330
- 3341 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn (technical  
3342 coordinators). 2018. Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing  
3343 Streams on Competent Lithologies in Western Washington. Cooperative Monitoring, Evaluation and  
3344 Research Report CMER 18-100, Washington State Forest Practices Adaptive Management Program,  
3345 Washington Department of Natural Resources, Olympia, WA.
- 3346 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D.E. Schuett-Hames, R. Ojala-Barbour, G. Stewart  
3347 and T. Quinn (technical coordinators). 2021. Effectiveness of experimental riparian buffers on perennial  
3348 non-fish-bearing streams on competent lithologies in western Washington – Phase 2 (9 years after  
3349 harvest). Cooperative Monitoring, Evaluation and Research Report CMER 2021.07.27, Washington State  
3350 Forest Practices Adaptive Management Program, Washington Department of Natural Resources,  
3351 Olympia, WA.
- 3352 Meleason, M. A., Gregory, S. V., & Bolte, J. P. (2003). Implications of riparian management strategies on  
3353 wood in streams of the Pacific Northwest. *Ecological Applications*, 13(5), 1212-1221.
- 3354 Merschel, A. G., Spies, T. A., & Heyerdahl, E. K. (2014). Mixed-conifer forests of central Oregon: effects of  
3355 logging and fire exclusion vary with environment. *Ecological Applications*, 24(7), 1670-1688.
- 3356 Mueller, E. R., & Pitlick, J. (2013). Sediment supply and channel morphology in mountain river systems: 1.  
3357 Relative importance of lithology, topography, and climate. *Journal of Geophysical Research: Earth*  
3358 *Surface*, 118(4), 2325-2342.
- 3359 Murray, G. L. D., Edmonds, R. L., & Marra, J. L. (2000). Influence of partial harvesting on stream  
3360 temperatures, chemistry, and turbidity in forests on the western Olympic Peninsula,  
3361 Washington. *Northwest science.*, 74(2), 151-164.
- 3362 Naiman, R. J., Fetherston, K. L., McKay, S. J., & Chen, J. (1998). Riparian forests. *River ecology and*  
3363 *management: lessons from the Pacific Coastal Ecoregion*, 289-323.
- 3364 Nowakowski, A. L., & Wohl, E. (2008). Influences on wood load in mountain streams of the Bighorn  
3365 National Forest, Wyoming, USA. *Environmental Management*, 42(4), 557-571.
- 3366 Polyakov, V., Fares, A., & Ryder, M. H. (2005). Precision riparian buffers for the control of nonpoint source  
3367 pollutant loading into surface water: A review. *Environmental Reviews*, 13(3), 129-144.
- 3368 Puntteney-Desmond, K. C., Bladon, K. D., & Silins, U. (2020). Runoff and sediment production from  
3369 harvested hillslopes and the riparian area during high intensity rainfall events. *Journal of Hydrology*, 582,  
3370 124452.

- 3371 Quinn, T., G.F. Wilhere, and K.L. Krueger, technical editors. (2020). Riparian Ecosystems, Volume 1:  
3372 Science Synthesis and Management Implications. Habitat Program, Washington Department of Fish and  
3373 Wildlife, Olympia.
- 3374 Rachels, A. A., Bladon, K. D., Bywater-Reyes, S., & Hatten, J. A. (2020). Quantifying effects of forest  
3375 harvesting on sources of suspended sediment to an Oregon Coast Range headwater stream. *Forest*  
3376 *Ecology and Management*, 466, 118123.
- 3377 Reid, D. A., & Hassan, M. A. (2020). Response of in-stream wood to riparian timber harvesting: Field  
3378 observations and long-term projections. *Water Resources Research*, 56(8), e2020WR027077.
- 3379 Reiter, M., Bilby, R. E., Beech, S., & Heffner, J. (2015). Stream temperature patterns over 35 years in a  
3380 managed forest of western Washington. *JAWRA Journal of the American Water Resources*  
3381 *Association*, 51(5), 1418-1435.
- 3382 Reiter, M., Heffner, J. T., Beech, S., Turner, T., & Bilby, R. E. (2009). Temporal and Spatial Turbidity Patterns  
3383 Over 30 Years in a Managed Forest of Western Washington 1. *JAWRA Journal of the American Water*  
3384 *Resources Association*, 45(3), 793-808.
- 3385 Reiter, M., Johnson, S. L., Homyack, J., Jones, J. E., & James, P. L. (2020). Summer stream temperature  
3386 changes following forest harvest in the headwaters of the Trask River watershed, Oregon Coast  
3387 Range. *Ecohydrology*, 13(3), e2178.
- 3388 Roni, P., Beechie, T., Pess, G., & Hanson, K. (2015). Wood placement in river restoration: fact, fiction, and  
3389 future direction. *Canadian Journal of Fisheries and Aquatic Sciences*, 72(3), 466–478.  
3390 <https://doi.org/10.1139/cjfas-2014-0344>
- 3391 Roon, D. A., Dunham, J. B., & Groom, J. D. (2021a). Shade, light, and stream temperature responses to  
3392 riparian thinning in second-growth redwood forests of northern California. *PloS One*, 16(2), e0246822–  
3393 e0246822. <https://doi.org/10.1371/journal.pone.0246822>
- 3394 Roon, D. A., Dunham, J. B., & Torgersen, C. E. (2021b). A riverscape approach reveals downstream  
3395 propagation of stream thermal responses to riparian thinning at multiple scales. *Ecosphere*, 12(10),  
3396 e03775.
- 3397 Safeeq, M., Grant, G. E., Lewis, S. L., & Hayes, S. K. (2020). Disentangling effects of forest harvest on long-  
3398 term hydrologic and sediment dynamics, western Cascades, Oregon. *Journal of Hydrology*, 580, 124259.
- 3399 Salo, E. O., & Cundy, T. W. (1986, February). Streamside management: forestry and fishery interactions.  
3400 In *Proceedings of a conference sponsored by the College of Forest Resources, University of Washington*  
3401 *and others, and held at the University of Washington*.
- 3402 Schuett-Hames, D., & Stewart, G. (2019a). Post-Harvest Change in Stand Structure, Tree Mortality and  
3403 Tree Fall in Eastern Washington Riparian Buffers. Cooperative Monitoring Evaluation and Research  
3404 Report. Washington State Forest Practices Adaptive Management Program. Washington Department of  
3405 Natural Resources, Olympia, WA.
- 3406 Schuett-Hames, D. & Stewart, G. (2019b). Changes in stand structure, buffer tree mortality and riparian-  
3407 associated functions 10 years after timber harvest adjacent to non-fish-bearing perennial streams in

3408 western Washington. Cooperative Monitoring Evaluation and Research Report. Washington State Forest  
3409 Practices Adaptive Management Program. Washington Department of Natural Resources, Olympia, WA.

3410 Schuett-Hames, D., Martin, D., Mendoza, C., Flitcroft, R., & Haemmerle, H., (2015). Westside Type F  
3411 Riparian Prescription Monitoring Project Technical Writing and Implementation Group (TWIG).  
3412 Cooperative Monitoring Evaluation and Research Report. Washington State Forest Practices Adaptive  
3413 Management Program. Washington Department of Natural Resources, Olympia, WA.

3414 Schuett-Hames, D., Roorbach, A., & Conrad, R. (2011). Results of the Westside Type N Buffer  
3415 Characteristics, Integrity and Function Study Final Report. Cooperative Monitoring Evaluation and  
3416 Research Report, CMER 12-1201, Washington Department of Natural Resources, Olympia, WA.

3417 Shah, N. W., & Nisbet, T. R. (2019a). The effects of forest clearance for peatland restoration on water  
3418 quality. *Science of the Total Environment*, 693, 133617.

3419 Sievers, M., Hale, R., & Morrongiello, J. R. (2017). Do trout respond to riparian change? A meta-analysis  
3420 with implications for restoration and management. *Freshwater Biology*, 62(3), 445–457.  
3421 <https://doi.org/10.1111/fwb.12888>

3422 Six, L. J., Bilby, R. E., Reiter, M., James, P., & Villarin, L. (2022). Effects of current forest practices on  
3423 organic matter dynamics in headwater streams at the Trask river watershed, Oregon. *Trees, Forests and*  
3424 *People*, 8, 100233.

3425 Sobota, D. J., Gregory, S. V., & Sickle, J. V. (2006). Riparian tree fall directionality and modeling large  
3426 wood recruitment to streams. *Canadian Journal of Forest Research*, 36(5), 1243–1254.  
3427 <https://doi.org/10.1139/x06-022>

3428 Sugden, B. D., Steiner, R., & Jones, J. E. (2019). Streamside management zone effectiveness for water  
3429 temperature control in Western Montana. *International Journal of Forest Engineering*, 30(2), 87-98.

3430 Sullivan, P. F., Neale, M. C., & Kendler, K. S. (2000). Genetic epidemiology of major depression: review  
3431 and meta-analysis. *American journal of psychiatry*, 157(10), 1552-1562.

3432 Swanson, F. J., & Dyrness, C. T. (1975). Impact of clear-cutting and road construction on soil erosion by  
3433 landslides in the western Cascade Range, Oregon. *Geology*, 3(7), 393-396.

3434 Swanson, F. J., Gregory, S. V., Sedell, J. R., & Campbell, A. G. (1982). Land-water interactions: the riparian  
3435 zone.

3436 Swartz, A., Roon, D., Reiter, M., & Warren, D. (2020). Stream temperature responses to experimental  
3437 riparian canopy gaps along forested headwaters in western Oregon. *Forest Ecology and*  
3438 *Management*, 474, 118354.

3439 Sweeney, B. W., & Newbold, J. D. (2014). Streamside forest buffer width needed to protect stream water  
3440 quality, habitat, and organisms: a literature review. *JAWRA Journal of the American Water Resources*  
3441 *Association*, 50(3), 560-584.

3442 Teply, M., McGreer, D., & Ceder, K. (2014). Using simulation models to develop riparian buffer strip  
3443 prescriptions. *Journal of Forestry*, 112(3), 302-311.



3444 Teply, M., McGreer, D., Schult, D., & Seymour, P. (2007). Simulating the effects of forest management on  
3445 large woody debris in streams in northern Idaho. *Western Journal of Applied Forestry*, 22(2), 81–87.  
3446 <https://doi.org/10.1093/wjaf/22.2.81>

3447 Vanderbilt, K. L., Lajtha, K., & Swanson, F. J. (2003). Biogeochemistry of unpolluted forested watersheds  
3448 in the Oregon Cascades: temporal patterns of precipitation and stream nitrogen  
3449 fluxes. *Biogeochemistry*, 62(1), 87-117.

3450 Warren, D. R., Keeton, W. S., Bechtold, H. A., & Rosi-Marshall, E. J. (2013). Comparing streambed light  
3451 availability and canopy cover in streams with old-growth versus early-mature riparian forests in western  
3452 Oregon. *Aquatic sciences*, 75(4), 547-558.

3453 Wing, M. G., & Skaugset, A. (2002). Relationships of channel characteristics, land ownership, and land  
3454 use patterns to large woody debris in western Oregon streams. *Canadian Journal of Fisheries and*  
3455 *Aquatic Sciences*, 59(5), 796-807.

3456 Wise, E. K. (2010). Tree ring record of streamflow and drought in the upper Snake River. *Water Resources*  
3457 *Research*, 46(11).

3458 Wissmar, R.C., Beer, W.N. & Timm, R.K. Spatially explicit estimates of erosion-risk indices and variable  
3459 riparian buffer widths in watersheds. *Aquat. Sci.* 66, 446–455 (2004). [https://doi.org/10.1007/s00027-](https://doi.org/10.1007/s00027-004-0714-9)  
3460 [004-0714-9](https://doi.org/10.1007/s00027-004-0714-9)

3461 Yang, Y., Hart, S. C., McCorkle, E. P., Stacy, E. M., Barnes, M. E., Hunsaker, C. T., ... & Berhe, A. A. (2021).  
3462 Stream water chemistry in mixed-conifer headwater basins: role of water sources, seasonality,  
3463 watershed characteristics, and disturbances. *Ecosystems*, 24(8), 1853-1874.

3464 Yeung, A. C., Stenroth, K., & Richardson, J. S. (2019). Modelling biophysical controls on stream organic  
3465 matter standing stocks under a range of forest harvesting impacts. *Limnologica*, 78, 125714.

3466 Zhang, X., & Shu, C. W. (2010). On positivity-preserving high order discontinuous Galerkin schemes for  
3467 compressible Euler equations on rectangular meshes. *Journal of Computational Physics*, 229(23), 8918-  
3468 8934.

3469

3470

3471

3472

3473

3474

3475

3476

3477

3478 Appendix I

3479

3480 Table A-1. List of treatments, variables, metrics, and results from publications reviewed for information on litter, organic matter, and  
 3481 nutrient inputs.

Reference	Treatment	Variables	Metrics	Notes	Results
Anderson et al., 2007	Upland stands either thinned to 198 TPA or unthinned and ranged from 500-865 TPA. Within thinned stands, 10% of the area was harvested to create patch openings. streamside buffers ranged in width from <5 m to 150 m.	Microsite, microclimate, stand structure, canopy cover	Microsite and microclimate data (humidity, temperature sensors). Stand basal area. Canopy cover was estimated through photographic techniques.	Many of the reported differences in temperature and humidity were considerable but not significant. Results for changes in upland areas not reported here.	Subtle microclimatic changes as mean temperature maxima in treated stands were 1 to 4°C higher than in untreated stands. Buffer widths greater than or equal to 15 m experienced a daily maximum air temperature above stream center of less than 1°C greater than untreated stands. Daily minimum relative humidity for buffers 15 m or greater was less than 5 percent lower than for unthinned stands. Air temperatures were significantly higher in patch openings (+6 to +9°C), and within buffers adjacent to patch openings (+3.5°C), than in untreated stands.
Bilby & Heffner, 2016	Various wind speeds for young and old-growth conifer and deciduous forests. Distance of litter delivery.	Litter input	Models were developed with site characteristics and litter release experiments from sites along Humphrey Creek in the cascade mountains of western Washington. .	Wind speeds, direction, and litter release data were collected for only one year in one area of western Washington.	The results of the linear mixed model developed by the authors showed the strongest relationship for recruitment distance was with wind speed (p<0.0001). Using this relationship the authors estimated that the effective delivery area could be increased by 67-81% by doubling wind speed. The other significant relationship was with stand age for needles (not alder leaves). Needles released from mature stands traveled further distances. This is likely due to the higher height of the canopy in the mature stands.
Deval et al., 2021	clearcut to stream, 50% shade retention, with site management operations including pile burning and competition release herbicide application.	Changes in nitrogen and phosphorus compounds.	monthly grab samples from multiple flume sites pre- and post-harvest, laboratory chemical analysis	Data was compared from pre-harvest to post experimental harvest (PH-I), and post operational harvest (PH-II)	The response in NO3 + NO2 concentrations was negligible at all treatment sites following the road construction activities. However, NO3 + NO2 concentrations during the PH-I period increased significantly (p < 0.001) at all treatment sites. Similar to the PH-I period, all watersheds experienced significant increases in NO3 + NO2 concentration during the PH-II treatment period. Overall, the cumulative mean NO3 + NO2 load from all watersheds followed an increasing trend with initial signs of recovery in one treatment watershed after 2014. Mean monthly TP concentrations showed no significant changes in the concentrations

					during the post-road and PH-I treatment periods. However, a statistically significant increase in TP concentrations ( $p < 0.001$ ) occurred at all sites, including the downstream cumulative sites, during PH-II. Generally, OP concentrations throughout the study remained near the minimum detectable concentrations
Gravelle et al., 2009	clearcut to stream, 50% shade retention, uncut reference	Changes in nitrogen and phosphorus compounds.	monthly grab samples from multiple flume sites pre- and post-harvest, laboratory chemical analysis	Data was compared in three treatment periods: pre-harvest, under road construction, post-harvest.	Results showed significant increases in monthly mean NO <sub>3</sub> and NO <sub>2</sub> following clear-cut harvest treatments relative to the pre-harvest, and road construction periods. Monthly nitrate responses showed progressively increasing concentrations for 3 years after harvest before declining. Significant increases in NO <sub>3</sub> and NO <sub>2</sub> concentrations were also found further downstream but at values lower than those immediately downstream from harvest treatments. No significant changes of in-stream concentration of any other nutrient recorded were found between time periods and treatments except for one downstream site that showed a small increase in orthophosphate by 0.01 mg P L <sup>-1</sup> .
Hart et al., 2013	(1) a no cut or fence control; (2) cut and remove a 5 x 8 m section adjacent to stream for plants < 10 cm DBH and >12 cm; and (3) 5 m fence extending underground and parallel to the stream to block litter moving downslope from reaching stream	Litter inputs, vegetation composition, topography, litter chemistry	Litter collected with lateral and vertical traps. Litter was sorted by type, time of fall, spatial source, and quantified by weight. Vegetation, LW, and Site characteristics were quantified for each plot.	This study took place within 5 contiguous watersheds located in the central Coast Range of Oregon.	Deciduous forests dominated by red alder delivered greater vertical and lateral inputs to streams than did coniferous forests dominated by Douglas-fir by 110 g/m <sup>2</sup> (28.6–191.6) and 46 g/m (1.2–94.5), respectively. Annual lateral litter input increased with slope at deciduous sites ( $R^2 = 0.4073$ , $p = 0.0771$ ) but not at coniferous sites ( $R^2 = 0.1863$ , $p = 0.2855$ ). Total nitrogen flux to streams at deciduous sites was twice as much as recorded at coniferous sites. However, the nitrogen flux had a seasonal effect with the majority of N flux occurring in autumn at the deciduous sites. The authors of this study conclude by suggesting management in riparian areas consider utilizing deciduous species such as red alder for greater total N input to aquatic and terrestrial ecosystems with increased shade and large wood provided by coniferous species.
Kiffney & Richardson, 2010	clearcut to stream, 10 m buffer, 30 m buffer, uncut control	Litter inputs.	Litter was separated into broadleaf deciduous, twig, needles, and other (seeds, cones, and moss) categories following collection and subsequently dried and weighed using a microbalance.	Sites were measured over an 8-year period and included clear-cut (n=3), 10-m buffered reserve (n=3), 30-m buffered reserve (n=2), and uncut control (n=2) treatments.	Inputs consisting of needles and twigs were significantly lower adjacent to clearcuts compared to other treatments, while deciduous inputs were higher in clearcuts compared to other treatments. For example, one year post-treatment, needle inputs were 56x higher during the Fall into control and buffered treatments than into the clearcut. Needle inputs remained 6x higher in the buffer and control sites through year 7, and 3-6x higher in year 8 than in the clearcut sites. Twig inputs into the control and buffered sites were ~25x higher than in the clearcut sites in the first year after treatment. There was no significant difference in treatment for deciduous litter but a trend of increasing deciduous litter input in the clear cut was observed in the data. The linear relationship between

					reserve width and litter inputs was strongest in the first year after treatment, explaining ~57% of the variation, but the relationship could only explain ~17% of the variation in litter input by buffer width by year 8 (i.e., the relationship degraded over time).
McIntyre et al., 2018	(1) unharvested reference, (2) 100% treatment, a two-sided 50-ft riparian buffer along the entire Riparian Management Zone (RMZ), (3) FP treatment, a two-sided 50-ft riparian buffer along at least 50% of the RMZ (4) 0% treatment, clearcut to stream edge (no-buffer).	Litter inputs from litter traps situated along channel	Sorted by litter type (conifer needles, deciduous leaves, woody components, etc.). Compared between treatments by dry weight.	Authors of the study identify a lack of information on local meteorology as a primary limitation to the study. This, the authors suggest, would have allowed for a more detailed analysis including information on hydrologic mass balance.	Showed a decrease in TOTAL litterfall input in the FP (P = 0.0034) and 0% (P = 0.0001) treatments between pre- and post-treatment periods. LEAF litterfall (deciduous and conifer leaves combined) input decreased in the FP (P = 0.0114) and 0% (P <0.0001) treatments in the post-treatment period. In addition, CONIF (conifer needles and scales) litterfall input decreased in the FP (P = 0.0437) and 0% (P <0.0001) treatments, DECID (deciduous leaves) in the 0% (P <0.0001) treatment, WOOD (twigs and cones) in the FP (P = 0.0044) and 0% (P = 0.0153) treatments, and MISC (e.g., moss and flowers) in the 0% (P = 0.0422) treatment. Results for comparison of the post-harvest effects between treatments showed LEAF litterfall input decreased in the 0% treatment relative to the reference (P = 0.0040), 100% (P = 0.0008), and FP (P = 0.0267) treatments. Likewise, there was a decrease in DECID litterfall input in the 0% treatment relative to the Reference (P = 0.0001), 100% (P <0.0001), and FP (P = 0.0015) treatments. Statistical differences were only detected for deciduous inputs between the 0% treatment and the other treatments.
McIntyre et al., 2021	<u>1) unharvested reference, 2) 100% treatment, a two-sided 50-ft riparian buffer along the entire RMZ, 3) FP treatment a two-sided 50-ft riparian buffer along at least 50% of the RMZ, (4) 0% treatment, clearcut to stream edge (no-buffer).</u>	stream discharge, nitrogen export		Type N (non-fish-bearing streams). Hard-Rock study.	Discharge increased by 5-7% on average in the 100% treatments while increasing between 26-66% in the FP and 0% treatments Results for harvest effects on total Nitrogen export showed significant (P <0.05) treatment effects were present in the FP treatment and in the 0% treatment in the post-harvest (2-years immediately following harvest) and extended periods (7 and 8 years post-harvest) relative to the reference sites, Analysis showed an increase in total-N export of 5.73 (P = 0.121), 10.85 (P = 0.006), and 15.94 (P = 0.000) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments, respectively, and of 6.20 (P = 0.095), 5.34 (P = 0.147), and 8.49 (P = 0.026) kg/ha/yr in the extended period. The authors conclude that the 100% treatment was generally the most effective in minimizing changes in total-N from pre-harvest conditions, the FP was intermediate, and the 0% treatment was least effective. At the end of the study (8 years), only one site had recovered to pre-harvest nitrate-N levels.

Murray et al., 2000	7% and 33% watershed upland harvest. Harvest extended to stream channel.	stream chemistry, stream temperatures, sediment input	Chemistry and pH tested on water grab samples; Daily max, min, and average temperatures collected with Stowaway dataloggers; Sediment change detected with turbidity meters.	Results reflect differences in stream conditions 11-15 years post-harvest only. No data collected in first decade following treatment.	10-15 years post-harvest mean maximum daily summer temperatures were still significantly higher (15.4 °C) and mean maximum daily winter temperatures were lower (3.7 °C) than in the reference streams (12.1 °C and 6.0 °C) respectively. Also, winter minimum temperatures for one of the harvested watersheds reached 1.2 °C compared to a winter minimum of 6 °C There were no significant differences in stream chemistry with the exception of calcium and magnesium being consistently higher in the unharvested reference watersheds. No detectable difference in turbidity between treatment and reference watershed streams 10-5 years post-treatment. The stream temperature changes were significant but did not exceed the 16 °C threshold used as a standard for salmonid habitat.
Six et al., 2022	Clearcut with no leave trees or retention buffer (CC), clearcut with leave trees (CC w/LT; retention of 5 trees per hectare/2 trees per acre), and clearcut with 15 m wide retention buffer (CC c/B) and two uncut references (REF 1, and 2) along headwater streams	Litter input, LW recruitment	litter traps, In-stream LW volume, weight, and counts.	No replication of treatment sites. Data was analyzed with descriptive and graphical representation only.	Results showed a reduction of canopy cover from 91.4% to 34.4% in the clearcut treatment with no leave trees, from 89.8% to 76.1% in the clearcut treatment with leave trees, and from 89.5% to 86.9% in the clearcut treatment with the 15 m retention buffer. Post harvest litter delivery decreased for the clearcut with no leave trees but increased for both the clearcut with leave tree and clear cut with retention buffer.
Vanderbilt et al., 2003	Datasets (ranging from 20-30 years) from six watersheds in the H.J. Andrews Experimental Watershed.	Nitrogen concentration in streams, precipitation patterns	regression analysis of annual N inputs and outputs with annual precipitation and stream discharge to analyze patterns.	These results come from a coastal climate of western Oregon. The authors warn that the controls on in stream N concentrations will likely differ in different regions.	Total annual discharge was a positive predictor of annual DON export in all watersheds with r2 values ranging from 0.42 to 0.79. In contrast, significant relationships between total annual discharge and annual export of NO3-N, NH4-N, and PON were not found in all watersheds. DON concentrations increased in the fall in every watershed. The increase in concentration began in July or August with the earliest rain events, and peak DON concentrations occurred in October through December. DON concentrations then declined during the winter months. The authors conclude that total annual stream discharge was a positive predictor of DON output suggesting a relationship to precipitation.

Yang et al., 2021	Young stands with high shrub cover (> 50%) masticated to < 10% shrub cover. trees removed to a target basal area range of 27–55 m <sup>2</sup> ha <sup>-1</sup> .	Drought, nutrients, dissolved organic carbon	Stream water samples grab samples and chemical analysis	Because of difficulties with accessibility due to weather-related phenomena (particularly during winter months), snowmelt and soil samples were restricted to the lower elevation site.	Drought alone altered DOC in stream water, and DOC:DON in soil solution in unthinned (control) watersheds. The volume-weighted concentration of DOC was 62% lower, and DOC:DON was 82% lower in stream water in years during drought than in years prior to drought. Drought combined with thinning altered DOC and DIN in stream water, and DON and TDN in soil solution. For stream water, volume-weighted concentrations of DOC were 66- 94% higher in thinned watersheds than in control watersheds for all three consecutive drought years following thinning. No differences in DOC concentrations were found between thinned and control watersheds before thinning. Watershed characteristics inconsistently explained the variation in volume-weighted mean annual values of stream water chemistry among different watersheds
Yeung et al., 2019	Range of forest harvest intensities	Litter inputs, CPOM in streams	stream temperature, streamflow, litter traps, CPOM decay rates	Authors point out that model results are primarily applicable to stream reaches similar to those used in the study and may not be suitable for streams where large wood is a dominant structure retaining CPOM.	The simulation predicted that litter input reduction from timber harvest was the strongest control on CPOM in streams relative to streamflow and temperature variability. The effects of litterfall reduction were at least an order of magnitude higher than streamflow increases in depleting in-stream CPOM. Significant CPOM depletions were most likely when there was a 50% or greater reduction in litterfall following harvest. The caveat of this study is that it did not include LW dynamics in preserving CPOM post-harvest. As other studies have shown, harvest can increase in-stream LW, and in-stream LW can act as a catchment for CPOM.

3482

3483 Table A-2. List of treatments, variables, metrics, and results from publications reviewed for information on large wood (LW), wood loads, and wood recruitment.

Reference	Treatment	Variables	Metrics	Notes	Results
Anderson & Meleason, 2009	Buffer averaging 69 m adjacent to thinning and a 0.4 patch opening; variable width buffer averaging 22 m adjacent to thinning and a 0.4 patch opening.	Instream wood load, understory vegetation cover	Percent cover of LW in streams and in riparian area, %cover shrubs, herbs, moss.		LW changes were non-significant, decrease in treatment reaches with greatest pre-treatment values 5 years post-treatment caused homogenization of LW. Gaps (patch openings) showed the highest changes increase in herbaceous cover, decrease in shrub cover. Moss cover increased in thinned areas but decreased in gaps. LW and vegetation changes insensitive to treatment buffers > 15 m.

Bahuguna et al., 2010	Two buffer widths on each side of the stream (10 m and 30 m) with upland clearcuts, and an unharvested control.	LW, Stand Structure, mortality	Strip plot sampling method running parallel to the stream to collect data on stand metrics.	Experimental design included 3 replicates of each treatment. Data was collected annually for one year pre- and 8 years post-treatment. Vancouver, B.C.	Following harvest, 11% of initially standing timber was blown down in the first and second years in the 10 m buffer, compared to 4% in the 30 m buffer, and 1% in the unharvested controls. Small diameter trees were significantly more represented in streams - 77% of LW was in the 10 cm - 20 cm diameter class while the mean diameter of standing trees in riparian buffers was 30 cm. By 8 years post-harvest, a significant amount of annual mortality occurred in the unharvested control at 30%, compared to 15% in both 30 m and 10 m buffers.
Benda et al., 2016	Simulated treatments of single or double entry thinning with and without a 10-m no cut buffer, with and without mechanical tipping of stems into streams. Thinning encompassed 5-20 % thinning.	instream LW volume	ORGANON growth models simulated secondary forest growth. The model was run for 100 years in 5-year time steps.	used the reach scale wood model (RSWM) developed for the Alcea watershed in central coastal Oregon. Data was sourced from FIA.	Single entry thinning reduced in-stream wood by 33 and 66% after a century, relative to reference streams when one and both sides of the channel were harvested. Adding a 10 m buffer reduced total loss to 7 and 14%. Mechanical tipping of 14 and 12% of cut stems were sufficient in offsetting the loss of instream wood without and with buffers. Double entry thinning without a buffer resulted in 42 and 84% loss of in stream wood relative to the reference streams when one or both sides of the channel were harvested. Adding a 10 m buffer changed reductions of in stream wood to 11 and 22% for one- and two-sided channel harvest. To offset the total predicted reduction of in stream wood for the double entry thinning would require tipping of 10 and 7% of cut stems without and with 10 m buffers.
Burton et al., 2016	70-m buffer representative of one site potential tree, 15-m buffer, 6-m buffer. Outside of buffer, all treatment stands were thinned first to 200 trees per hectare (tph), then again to 85 tph ~ 10 years later. Uncut reference was ~400 tph.	LW recruitment, In-stream wood volume, biomass, and	LW volume, LW characteristics and source evidence, reach and stream characteristics.	Wood surveys were carried out at four times during the study: (1) prior to the first thinning, (2) five years after the first thinning, (3) 9-13 years after the first thinning and just prior to the second thinning, and (4) one year after the second thinning.	In-stream wood volume increased significantly with drainage basin area; for every 1-ha increase in drainage basin area, wood volume increased by 0.63%. LW volume was slightly higher in the streams adjacent to 6 m buffers than in streams bordered by 15 and 70 m buffers. The higher volume of wood in the 6 m buffers began 5 years after the first harvest and maintained through 1 year after the second harvest (end of study). . 82% to 85% of all wood inputs (early- and late-stage decay) were sourced from within 15 m of the streams (90% of early-stage decay wood could be sourced, only 45% of late-stage decay wood could be sourced).

Chen et al., 2005	All harvested streams were clearcut to stream edge. Wildfire streams had no post-fire harvest	Instream wood load, biomass, carbon pool	LW count, volume, decay class, size		LW volume, biomass, and carbon pools were significantly higher in streams adjacent to areas recently disturbed by timber harvest (~10 years) or wildfire (~40 years) than in streams passing through old-growth forests. There was no significant difference in in-stream LW between old-growth riparian areas and areas harvested > 30 years ago. The wildfire sites had significantly higher LW values than both the harvested sites. The authors conclude: (1) LWD input in old growth forested streams was relatively stable based on statistical significance. They also speculate: (1) timber harvesting activities would cause a short-term increase of LWD stocks and might greatly reduce LWD loadings over a long-term, and (2) wildfire disturbance would delay LWD recruitment because not all burnt trees would fall in the stream immediately after the wildfire, based on trends in, and extrapolation of the data.
Chen et al., 2006	A total of 35 sites with stream orders ranging from 1-5 (grouped into 4 stream size categories (I = first order; II = second to third order; III = third to fourth order; IV = fourth to fifth order) were selected to measure spatial distribution and variability of LW characteristics	LW, defined as having a diameter of > 0.1 m and a length > 1.0 m.	LW size, volume, density, and biomass. Multiple stream channel features obtained from readily available physiographic and forest cover data.	Study sites were selected based on the following criteria. (1) the streams were in areas of intact mature riparian forests (>80 years); (2) the stream side forests were not disturbed by human activities, such as harvesting, road building; (3) the streams were not salvaged.	Results from this study show that LW size, volume, and biomass generally increased with increasing stream size. For example, the mean LWD diameter in stream size I (16.4 cm) was lower than that in stream size III (20.6 cm) and IV (20.5 cm), respectively. Mean LW length also increases with stream size from 2.3 m in size I, 2.9 m in size II, 3.1 m in size III, and 3.9 m in size IV. Stream IV had the highest mean volume (0.18 m <sup>3</sup> ), significantly higher than stream size I (0.06 m <sup>3</sup> ). LW density (pieces per 100 m <sup>2</sup> of stream area), however, decreased as stream size increased. For example, LW density (defined as piece numbers per 100 m <sup>2</sup> ) numbers were 19, 17, 12, and 4 for stream size I, II, III, and IV respectively. Increases in channel bank full width (R <sup>2</sup> = 0.52) and stream area (R <sup>2</sup> = 0.58) was found to be strongly inversely correlated with LW density.
Ehinger et al., 2021	1) Buffers encompassing the full width (50 feet), 2) <50ft buffers, 3) Unbuffered, harvested to the edge of the			Soft Rock study. Only descriptive statistics were applied for changes in stand structure and wood loading.	There was little post-harvest large wood input in reference sites: an average of 4.3 pieces and 0.34 m <sup>3</sup> of combined in- and over-channel volume per 100 m of channel. In contrast, the full buffer sites and <50 ft buffer sites received an average of 23 and 10 pieces/100 m and 2.3 and 0.7 m <sup>3</sup> /100 m of large wood, respectively. Piece counts remained stable in the reference sites through year 3 post-harvest, increased in the



	channel, and 4) Reference sites in unharvested forests.			Small sample sizes.	full buffer and unbuffered sites (8 and 13%, respectively), and decreased in the <50 ft buffers (-15%).
<a href="#">Fox &amp; Bolton, 2007</a>	LW values from 150 stream segments located in unmanaged watersheds, across all of Washington State	Instream LW, geomorphology, forest zone, disturbance regimes	Descriptive statistics for LW volume and quantity, channel geomorphology, forest habitat type, disturbance regimes.	the authors warn that these values for reference conditions are only applicable to streams with bank-full widths 1-100 m, gradients 0.1%-47%, elevations 91-1,906 m, drainage areas 0.4-325 km <sup>2</sup> , glacial and rain- or snow-dominated origins, forest types common to the Pacific Northwest.	Results showed that in-stream wood volume increased with drainage area and as streams became less confined. Bank full width (BFW) was the single greatest predictor of in-stream wood volumes relative to other predictor variables. However, this result comes with the caveat that other processes and geomorphologies (e.g., channel bed form, gradient, confinement) are also important in the mechanisms for wood recruitment, modeling in this study showed too much inconsistency with these predictor variables to draw strong conclusions. In-stream wood volume also increased with adjacent riparian timber age as determined by the last stand replacing fire. The authors developed thresholds for expected "key piece volume (m <sup>3</sup> )" (pieces with independent stability) of wood for three BFW classes (20-30 m, >30 – 50 m, > 50 m width) per 100 m stream length for streams with BFW greater than 20 m. From percentile distributions the authors recommend minimum volumes, defined by the 25th percentiles, of approximately 9.7 m <sup>3</sup> for the 20- to 30-m BFW class, 10.5 m <sup>3</sup> for the 30- to 50-m BFW class, and 10.7 m <sup>3</sup> for channels greater than 50 m BFW per 100 m length of stream.
<a href="#">Gomi et al., 2001</a>	Five management or disturbance regimes: old growth (OG), recent clear-cut (CC; 3 years), young conifer forest (YC; 37 years after clear-cut), young alder (YA; 30 years after clear-cut), and recent landslide and debris flow channels (LS)	LW quantity and distribution, sediment quantity and distribution, landslide frequency, harvest intensities	LW counts, LW characteristics, stream characteristics.	Results are highly variable among treatments	in-channel numbers of LW pieces were significantly higher in YC and CC sites when compared to OG, YA, and LS sites. The number of LW pieces was highest in YC streams even though logging concluded 3 decades prior to sampling. LW volume per 100 m of stream length in YC was twice that in OG. The total volume of LW per 100 m associated with CC channels was half that in OG channels. The authors conclude (i) inputs of logging slash and unmerchantable logs significantly increase the abundance of in-channel woody debris; (ii) in the absence of landslides or debris flows, these woody materials remain in the channel 50–100 years after logging.

<p>Hough-Snee et al., 2016</p>	<p>In-stream wood volume and frequency were quantified across multiple sub basins.</p>	<p>LW frequency and volume, hydrologic and geomorphic attributes</p>	<p>Models were calibrated with site characteristics from multiple riparian stands in the Columbia River Basin.</p>	<p>Results show a high level of variability between sub basins studied. The overall model shows site (watershed) was an important predictor.</p>	<p>In stream wood volume and frequency were distinctly different across all seven sub-basins. According to random forest (RF) models, mean annual precipitation, riparian large tree cover, and individual watershed were the three most important predictors of wood volume and frequency, overall. Sinuosity and measures of streamflow and stream power were relatively weak predictors of wood volume and frequency. Final RF models explained 43.5% of the variance in volume and 42.0% of the variance in frequency of in stream wood loads. Depending on the sub basin wood volume and frequency was positively correlated with forest cover, watershed area, large tree cover, 25-year flood event stream power, riparian conifer cover, and precipitation. Negative correlations, depending on sub basin, of wood volume and frequency with baseflow discharge, riparian woody cover, watershed area, and large tree cover. Given the heterogeneous results across all sub-basins studied, the authors conclude by emphasizing the importance of incorporating local data and context when building wood models to inform future management decisions.</p>
<p>Hyatt &amp; Naiman, 2001</p>	<p>LW data was collected from multiple sites in the Queets River Watershed.</p>	<p>LW in stream and in riparian forests.</p>	<p>Increment cores from in-stream LW were cross-dated to estimate the time LW was recruited. LW pieces in decay were dated using carbon-dating. A depletion curve was fitted for LW recruited between 1599 and 1997.</p>	<p>The depletion constant was developed for a large, mostly alluvial river and should probably not be applied to smaller streams</p>	<p>Results from this study indicate that the half-life of stream LW to be approximately 20 years, suggesting that current LW will either be exported, broken down, or buried within 3 to 5 decades (for conifers). Hardwoods were better represented in riparian forests than as in-stream LW, and conversely, conifers were better represented as in-stream LW than in adjacent forests suggesting that LW originating from hardwoods is depleted faster than conifers.</p>
<p>Jackson &amp; Wohl, 2015</p>	<p>In-stream wood volume and frequency were quantified along 33 pool-riffle or plane-bed stream reaches in the Arapaho and Roosevelt National Forests in Colorado.</p>	<p>Sediment storage, channel geometry, in-stream wood load, and forest stand characteristics</p>	<p>Wood loads, wood jam volumes, log jam frequencies, residual pool volume, and fine sediment storage around wood, stand age, and</p>	<p>Old growth defined as forests <math>\geq 200</math> years. Age range of young forests not reported. Sample sizes include 10 old-growth and</p>	<p>Results indicated that channel wood load (OG = <math>304.4 + 161.1Y</math>; <math>Y = 197.8 + 245.5 \text{ m}^3/\text{ha}</math>), floodplain wood load (OG = <math>109.4 + 80Y</math>; <math>Y = 47.1 + 52.8 \text{ m}^3/\text{ha}</math>), and total wood load (OG = <math>154.7 + 64.1Y</math>; <math>Y = 87.8 + 100.6 \text{ m}^3/\text{ha}</math>) per 100 m length of stream and per unit surface area were significantly larger in streams of old-growth forests than in young forests. Streams in old-growth forests also had significantly more wood in jams, and more total wood jams per unit length of channel than in younger forests (jam wood volume: OG = <math>7.10 + 6.9Y \text{ m}^3</math>; <math>Y =</math></p>

			disturbance history.	23 younger forests.	1.71 + 2.81 m <sup>3</sup> ). Although wood load in streams draining from pine beetle infested forests did not differ significantly from healthy forests, best subset regression (following principal component analysis) indicated that elevation, stand age, and pine beetle infestation were the best predictors of wood load in channels and on floodplains.
<a href="#">Jackson et al., 2001</a>	3 unthinned riparian buffers; 1 with a partial buffer; 1 with a buffer of non-merchantable trees; and 6 were clearcut to the stream edge. Buffers ranged from 15 to 21 m wide, partial buffers were as thin as 2.3 m.	Instream LW, particle size, surface roughness	LW as functional and nonfunctional (not altering flow hydraulics). Particle size distributions.	Data collected for only 1-year pre- and 1-month post-harvest. These results only describe immediate effects of harvest on stream conditions.	Increased slash debris (LW) provided shade for the harvested streams but trapped sediments and prevented fluvial transport. The percentage of fine particles increased from 12 to 44% because of bank failure and increased surface roughness. This was a short-term study on small headwater streams. Sediment and LW conditions in the unharvested and buffered streams remained relatively unchanged during the study.
<a href="#">Liquori, 2006</a>	Data were collected from 20 riparian buffer sites that had all been clearcut within three years of sampling with standard no-cut 25 ft or 50-100 ft buffers for non-fish-bearing and fish-bearing streams, respectively.	Tree and tree fall characteristics, Site characteristics	Tree characteristic data estimated cause of mortality, and distance to the stream. Tree recruitment probability curves were developed as a function of tree height.		Within no-cut buffers windthrow caused mortality was up to 3 times greater than competition induced mortality for 3 years following treatment Tree fall direction was heavily biased towards the channel regardless of channel or buffer orientation and tree fall probability was highest in the outer areas of the buffers (adjacent to the harvest area). Tree fall rates and direction were also heavily biased by species with western hemlock and Pacific silver fir having the highest fall rates compared to Douglas-fir, western red cedar, and red alder.

<p>Martin &amp; Grotefendt, 2007</p>	<p>Buffer widths a minimum of 20 m. Multiple buffer widths and harvest intensities.</p>	<p>Instream wood load, stand mortality</p>	<p>Counts of downed wood, tree stumps, stand characteristics, instream wood from aerial photographs taken post-logging</p>	<p>Stand and stream characteristic, and LW data was surveyed from aerial photographs.</p>	<p>Results showed significantly higher mortality, significantly lower stand density, and a significantly higher proportion of LW recruitment from the buffer zones of the treatment sites than in the reference sites. Differences in mortality for the treatment sites were similar to the reference sites for the first 0-10 m from the stream (22% increase). However, mortality in the outer half of the buffers (10-20 m) from the stream in the treatment sites was more than double (120% increase) what was observed in the reference sites. This caused a change in the LW recruitment source distance curves, with a larger proportion of LW recruitment coming from greater distances in logged watersheds. LW recruitment based on the proportion of stand recruited (PSR) was significantly higher in the buffered units compared to the reference units. However, PSR from the inner 0-20 m was only 17% greater in the buffer units than in the reference units; while PSR of the outer unit (10 – 20 m) was more than double in the buffered units than in the reference units. The researchers conclude that the increase in mortality was caused by an increased susceptibility to windthrow. They estimate that future recruitment potential from the logged sites diminished by 10% relative to the unlogged reference sites.</p>
<p>May &amp; Gresswell, 2003</p>	<p>Survey of LW in three second-order streams and the mainstem of the North Fork of Cherry creek.</p>	<p>LW, delivery mechanism</p>	<p>LW &gt; 20 cm diameter, and &gt;2 m length was categorized by 4 delivery mechanisms, Delivery process, disturbance type, and channel characteristics.</p>	<p>Although mean age of Douglas-fir trees was identified to be excess of 300 years old, further information on differences in stand structure or development stage between sites are not included.</p>	<p>Processes of slope instability were shown to be important conveyors of wood from upland forests to small colluvial channels. In the larger alluvial channels, windthrow was found to be the dominant recruitment process from adjacent riparian area. 80% of total wood pieces and 80% of total wood volume recruited to colluvial streams originated from trees rooted within 50 m of the channel. In the alluvial channel, 80% of the pieces of wood and 50% of the total volume originated from trees which came from 30 m of the channel. The primary function of wood in colluvial channels was sediment storage (40%) and small wood storage (20%). The primary function of wood in alluvial channels is bank scour (26%), stream bed scour (26%), and sediment storage (14%).</p>
<p>McIntyre et al., 2021</p>	<p>(1) unharvested reference, (2) 100% treatment, a two-sided 50-ft riparian buffer along the entire Riparian Management Zone</p>			<p>Hard Rock Study Physical constraints such as a lack of suitable low gradient reaches and/or issues with</p>	<p>Large wood recruitment to the channel was greater in the 100% and FPB RMZs than in the reference for each pre- to post-harvest time interval. Eight years post-harvest mean recruitment of large wood volume was two to nearly three times greater in 100% and FPB RMZs than in the references. Annual LW recruitment rates were greatest during the first two years, then decreased. However, these differences were</p>

	(RMZ), (2) FP treatment a two-sided 50-ft riparian buffer along at least 50% of the RMZ, (3) 0% treatment, clearcut to stream edge (no-buffer).			accessibility related to weather limited downstream measurements of exports to just eight sites.	not significant between any treatment comparisons, likely due to the high variability in the data. Mean LW loading (pieces per meter of stream) differed significantly between treatments in the magnitude of change overtime. Results showed a 66% ( $P < 0.001$ ), 44% ( $P = 0.05$ ) and 47% ( $P = 0.01$ ) increase in mean large wood density in the 100%, FP and 0% treatments, respectively, in the first 2 years post-harvest compared with the pre-harvest period and after controlling for temporal changes in the references. Five years post-treatment the FP continued to increase 42% ( $P = 0.08$ ), and again 8 years post-treatment (41%; $P = 0.09$ ). From 2-8 years post-harvest LW density in the 100% treatment stabilized and began to decrease in the 0% treatment.
Meleason et al., 2003	Multiple buffer widths and upland harvest intensities	Change in instream wood load over time	Simulation metrics for forest growth, tree breakage, and in-channel process	A potential limitation of growth models in that they lack the ability to predict responses to novel climatic conditions different than those of the past.	Simulation results predicted clear-cut to stream accumulated little LW immediately following treatment and little change over time. Maximum in-stream LW loads were predicted for streams with no-cut buffers $>30$ m for 500-year-old forests (500 years post treatment). Streams with 6 m wide buffers predicted only 32% of pre-harvest standing LW loads after 240 years. Forest plantations with $> 10$ m buffer widths contributed minimal LW to the stream from outside the buffer zone.
Nowakowski & Wohl, 2008	History of regulated and unregulated timber harvest practices.	Instream wood volume	LW volume, LW characteristics source evidence, buffer widths, reach and stream characteristics.		In-stream LW was 2-3 times lower in a watershed with a history ( $>100$ years) of timber harvest (1.1 m <sup>3</sup> /100 m) when compared to unmanaged reference watersheds (3.3 m <sup>3</sup> /100 m). Valley characteristics (elevation, forest type, forest stand density, etc.) consistently explained more of the variability in wood load (42-80%) than channel characteristics (21-33%; reach gradient, channel width, etc.). Across all streams, the highest explanatory power of all models tested produced land use (managed vs unmanaged), and basal area as a significant predictor of wood loads ( $r^2 = 0.8048$ ). For the unmanaged watershed the model produced stream valley sideslope gradient as the single best predictor of wood load ( $r^2 = 0.5748$ ). Shear stress was the best predictor of wood load in the managed watersheds ( $r^2 = 0.2403$ ). When the significant valley and channel characteristics of the managed and unmanaged watersheds were controlled for, the significant difference in wood loads between managed and unmanaged watersheds were enhanced ( $p = 0.0006$ ). Managed watersheds

					(1.1 m <sup>3</sup> /100 m) had, on average, 2-3 times lower in-stream wood loads than unmanaged (3.3 m <sup>3</sup> /100 m) watersheds.
<a href="#">Reid &amp; Hassan, 2020</a>	Clearcut to stream and buffer widths that range from 1-70 m. Models were developed for 3 harvest scenarios (1: no-harvest; 2 partial loss of riparian forests; 3 intensive harvest in the riparian zone)	Instream LW	Models were calibrated with long-term data for site and LW characteristics in treatment reaches dating back to 1973.	One caveat of this model is it doesn't account for as much variability on stream configuration or valley morphologies that are likely to affect LW storage.	Results of the model show evidence that wood storage in streams of harvested reaches its minimum value in 50 years or more following loss of LW input, decay, and export of current stock. Recovery of LW volume in-streams following harvest is estimated to take approximately 150-200 years. The pattern and intensity of the harvesting operation had little effect on LW loss and recovery times but did affect the estimated magnitude of LW volume loss in the first 50 – 80 years. The authors conclude that the results show evidence that timber harvest has a long-term effect on LW storage and loading dynamics even with protective buffers. However, buffers can ameliorate the magnitude of LW loss during the recovery period.
<a href="#">Schuett-Hames &amp; Stewart, 2019a</a>	Buffer prescriptions for standard shade rule (a 30-ft no-cut buffer width, and thinning 30-75 ft from the stream), and all available shade rule (requires retention of all shade providing trees in this area) for eastern Washington.	LW recruitment, instream wood volume, mortality, stand structure	LW volume, LW characteristics, LW source evidence, reach and stream characteristics, basin metrics, stand metrics	Short-term study. Results only for 5 years post-harvest. The authors note that LW recruitment is a process that can change over decadal time scales.	Results showed cumulative wood recruitment from tree fall over the five-year post-harvest interval was highest in the standard shade rule (SR) group, lower in the all-available-shade rule (AAS) group and lowest in the reference (REF) group. The SR and AAS rates by volume were nearly 300% and 50% higher than the REF rates, respectively. Most recruiting fallen trees originated in the first 30 feet (76%, 72%, and 64% for the REF, AAS and SR groups, respectively), while the proportion from the inner zone (30–75 feet from the stream) was ~10% greater for the SR group compared to the AAS and REF groups.
<a href="#">Schuett-Hames et al., 2011; Schuett-Hames &amp; Stewart, 2019b</a>	Clearcut to stream with 30-foot equipment exclusion zone, and 50-foot no-cut buffers	LW, mortality, stand structure, canopy cover	QMD, basal area, tree fall rates, instream LW counts and volume, canopy percentage from densiometer.	1) Substantial variability among sites. 2) Due to scale of study, results only applicable to immediate vicinity of buffer treatment.	10 years post treatment, 50-foot buffer mortality stabilized, cumulative 14.1% reduction in basal area; Reference stands increased in basal area by 2.7% over the 10 years. 10-year cumulative LW recruitment into channels were double that of the reference stands 10-year canopy cover of the 50-foot buffer recovered to similar percentages as the reference stands 10-year cumulative canopy cover of CC was 71.5% due to ingrowth of dense shrubs, saplings and herbaceous plants.

Sobota et al., 2006	Data was collected at 15 riparian sites throughout the pacific northwest and the Intermountain West	Tree characteristics, forest structural variables and topographic features	Stand density, basal area, and dominant tree species by basal area; Active channel width and valley floor width.	Bias in landform types between slope categories. Effects of catastrophic disturbance regimes in large rivers not included in model.	The strongest correlations of tree fall direction were with valley constraint. When grouped by species, the individual trees showed a stronger tendency to fall towards the stream when hillslopes were >40%. When field data was integrated into the recruitment model, results showed that stream reaches with steep side slopes (>40%) were 1.5 to 2.4 times more likely to recruit LW into streams than in moderately sloped (< 40%) reaches. The authors warn that while side slope categories (>40%, <40%) was the strongest predictor of tree fall direction in this study, they believe the differences in tree fall direction between these categories mainly characterized differences between fluvial (88% of moderate slope sites) and hillslope landforms (71% of steep slope sites). They suggest that the Implications from this study are most applicable to small- to medium-size streams (second- to fourth-order) in mountainous regions where sustained large wood recruitment from riparian forest mortality is the significant management concern.
Teply et al., 2007	25-ft no-cut buffer, with additional 50-foot requiring 88 trees per acre.	Instream wood load	Simulation metrics for forest growth, tree breakage, and in-channel process	The simulation evaluated both a harvest and a no-harvest scenario to predict mean in-stream LW loads after 30, 60, and 100 years	Simulation results predict a 25-foot no-cut buffer, with an additional 50-foot (25 –75 feet from the high watermark) zone requiring retention of 88-trees-per-acre were sufficient in maintaining no significant change in in-stream LW loading relative to unharvested reference streams.
Wing & Skaugset, 2002	LW loads and site characteristics were collected from 3793 stream reaches in western Oregon State (west of Cascade crest).	LW pieces, LW key pieces, LW volume	LW abundance, land use history, land ownership, site level attributes	Results presented here are only for forested streams (“tree 3” in text). Landownership was the strongest predictor in some models, but this included multiple areas of unforested reaches.	For in stream LW volume, stream gradient was the most important explanatory variable with the split occurring for stream reaches with gradients less than 4.7% averaging 11.5 m <sup>3</sup> , which was less than half of the average found at higher gradient reaches (25.2 m <sup>3</sup> ); in this model the stream gradient split explained 11% of the variation observed of instream LW volume. For LW pieces in forested stream reaches, bankfull channel width was the most important explanatory variable with the split occurring for streams channels less than 12.2 m wide. LW pieces for streams <12.2 m wide averaged 11.1 LW pieces per reach while larger channels averaged 4.9 pieces per reach; in this model the BFW split explained 7% of the variation in LW pieces found in forested streams. For key LW pieces (logs at least 0.60 m in diameter and 10 m long) in forested reaches, stream gradient was again the most important explanatory variable with the split occurring at a

					gradient of 4.9%. The streams with a gradient < 4.9% averaged 0.5 key LW pieces per reach while streams with higher gradients averaged 0.9 key LW pieces per reach; in this model stream gradient explained 8% of the variation in key LW pieces found in streams. Lithology caused second, third or fourth level splits after stream gradient or BFW.
--	--	--	--	--	--

3484

3485 Table A-3 List of treatments, variables, metrics, and results from publications reviewed for information on sediment inputs and source.

Reference	Treatment	Variables	Metrics	Notes	Results
<a href="#">Bywater-Reyes et al., 2017</a>	Harvest had a mixture of intensities including clearcut to stream and clearcut with 15 m buffers.	Sediment concentration, basin lithology, geomorphology	Channel, stream, and riparian area characteristics sourced from a mixture of LiDAR and management data.	This study analyzed 6 years of data from the Trask River Watershed in Northeastern Oregon and included data from harvested and unharvested sub-catchments underlain by heterogeneous lithologies.	Results from this study indicate that site lithology was a first order control over suspended sediment yield (SSY) with SSY varying by an order of magnitude across lithologies observed. Specifically, SSY was greater in catchments underlain by Siletz Volcanics ( $r = 0.6$ ), the Trask River Formation ( $r = 0.4$ ), and landslide deposits. In contrast, the site effect had a strong negative correlation with percent area underlain by diabase ( $r = 0.7$ ), with the lowest SSY associated with 100% diabase independent of whether earthflow terrain was present. Sites with low SSY and underlain by more resistant lithologies were also resistant to harvest-related increases in SSY. The authors conclude that sites underlain with a friable lithology (e.g., sedimentary formations) had SSYs an order of magnitude higher, on average, following harvest than those on more resistant lithologies (intrusive rocks).
<a href="#">Bywater-Reyes et al., 2018</a>	long-term data (60 years) of sediment, discharge, weather, and disturbance.	Sediment yield, discharge history, physiography.	suspended sediment concentration involved using either vertically integrated storm-based grab samples, or discharge-proportional composite samples.	The authors caution that the high variability of sediment yield over space and time (~0.2 - ~953 t/km <sup>2</sup> ) indicates that the factors tested in this study should be tested more broadly to investigate their utility to forest managers.	The results of this study show that watershed slope variability combined with cumulative annual discharge explained 67% of the variation in annual sediment yield across the approximately 60-year data set. The results, however, show that annual sediment yields also moderately correlated with many other physiographic variables and the authors caution that the strong relationship with watershed slope variability is likely a proxy for many processes, encompassing multiple catchment For the relationships between disturbance and sediment yield the authors conclude that the few anomalous years of high sediment yield occurred in watersheds with high slope variability and within a decade of forest management and a large flood event.



<p>Hatten et al., 2018</p>	<p>Data from pre restriction and post Oregon BMPs prescriptions for non-fish-bearing streams. BMPs: no buffer in non-fish-bearing streams with equipment exclusion zones, and a 15 m no-cut-buffer in fish-bearing streams</p>	<p>suspended sediment concentrations (SSC)</p>	<p>suspended sediment, stream discharge, and daily precipitation</p>	<p>Phase I harvest: 2009 harvest of upper half of watershed. Phase II harvest: 2015 harvest of lower half of watershed.</p>	<p>Methods used in 1966 to harvest the same watershed (no buffer, road construction, broadcast burning) resulted in an approximate 2.8-fold increase in SSC from pre- to post-Harvest. In the contemporary study both the mean and maximum SSC were greater in the reference catchments (FCG and DCG) compared to the harvested catchment (NBLG) across all water years. In NBLG the mean SSC was 32 mg L<sup>-1</sup> (~63%) lower after the Phase I harvest and 28.3 mg L<sup>-1</sup> (~55%) lower after the Phase II harvest when compared to the pre-harvest concentrations. Compared to the reference watersheds, the mean SSC was 1.5-times greater in FCG (reference) compared to NBLG during the pre-harvest period. After Phase I harvest the mean SSC in FCG was 3.1-times greater and after Phase II harvest was 2.9-times greater when compared to the SSC in the harvested watershed. The authors conclude that contemporary harvesting practices (i.e., stream buffers, smaller harvest units, no broadcast burning, leaving material in channels) were shown to sufficiently mitigate sediment delivery to streams, especially when compared to historic practices.</p>
<p>Karwan et al., 2007</p>	<p>clearcut of the watershed area of by 50%, partial cut of 50% canopy removal, timber road construction Riparian zone harvest followed Idaho FPA rules.</p>	<p>Total suspended solid (TSS) yields</p>	<p>Monthly total suspended solid readings from multiple flume locations for pre-, and post-harvest, and pre- and post-road construction.</p>		<p>A significant and immediate impact of harvest on monthly sediment loads in the clear-cut watershed (<math>p = 0.00011</math>), and a marginally significant impact of harvest on monthly sediment loads in the partial-cut (<math>p = 0.081</math>) were observed. Total sediment load from the clearcut over the immediate harvest interval (1-year post-harvest) exceeded predicted load by 152%; however, individual monthly loads varied around this amount. The largest increases in percentage and magnitude occurred during snowmelt months, namely April 2002 (560%) and May 2002 (171%). Neither treatment showed a statistical difference in TSS during the recovery time, 2-4 years post-harvest (clearcut: <math>p = 0.2336</math>; partial-cut: <math>p = 0.1739</math>) compared to the control watersheds. Road construction in both watersheds did not result in statistically significant impacts on monthly sediment loads in either treated watershed during the immediate or recovery time intervals.</p>

Litschert & MacDonald, 2009	Data collected from 4 NF of Nort CA. ~200 harvest sites near riparian zones with 90 m and 45 m buffer widths.	Sediment delivery pathway frequency and characteristics.	Pathway length, width, origins, and connectivity of sediment delivery pathways to streams.	Authors mention a caveat to the results of the study in that there is a potential of underestimating the frequency of rills and sediment plumes as sites recover.	Only 19 of the 200 harvest units had sediment development pathways and only 6 of those were connected to streams and five of those originated from skid trails. Pathway length was significantly related to mean annual precipitation, cosine of the aspect, elevation, and hillslope gradient.
Macdonald et al., 2003a	low-retention = removed all timber >15 cm DBH for pine and > 20 cm DBH for spruce within 20 m of the stream; high-retention = removed all timber > 30 cm within 20 m of the stream.	suspended sediment yields, stream discharge	Discharge rate and total suspended sediments (TSS) collected using Parshall flumes	Only 1-year pre-harvest data was collected to generated predicted TSS and discharge values post-harvest.	Immediately following harvest, TSS concentrations and discharge rates increased above predicted values for both treatment streams. Increased TSS persisted for two-years post-harvest in the high-retention treatment, and for 3-years in the low-retention. This study shows evidence that harvest intensity (low vs. high retention) is proportional to the increase in stream discharge, TSS concentrations, and recovery time to pre-harvest levels. The authors speculate that the treatment areas may have accumulated more snow (e.g., more exposed area below canopy) than in the control reaches leading to the increase in discharge.
McIntyre et al., 2021	1) unharvested reference, 2) 100% treatment, a two-sided 50-ft riparian buffer along the entire RMZ, 3) FP treatment a two-sided 50-ft riparian buffer along at least 50% of the RMZ, (4) 0% treatment, clearcut to	stream discharge, turbidity, and suspended sediment export.		Type N (non-fish-bearing streams). Hard-Rock study.	Discharge increased by 5-7% on average in the 100% treatments while increasing between 26-66% in the FP and 0% treatments. Results for water turbidity and suspended sediment export (SSE) were stochastic in nature and the relationships between SSE export and treatment effects were not strong enough to confidently draw conclusions. The authors conclude that timber harvest did not change the magnitude of sediment export for any buffer treatment.

	stream edge (no-buffer).				
<a href="#">Mueller &amp; Pitlick, 2013</a>	The study used sediment concentration data from 83 drainage basins in Idaho and Wyoming.	Sediment concentration, basin lithology, geomorphology	Sediment concentration distribution, geomorphology, and weather data from multiple sources.		The strongest correlation of in stream sediment supply was with lithology relative softness. Bankfull sediment concentrations increased by as much as 100-fold as basin lithology became dominated by softer sedimentary and volcanic rock. Relief (elevation), basin sideslope, and drainage density showed little correlation strength with bankfull sediment supply.
<a href="#">Puntenney-Desmond et al., 2020</a>	Variable retention buffers with clearcut.	surface and subsurface runoff rates, sediment.	Simulation metrics calibrated with runoff and sediment samples from sample area. Precipitation calibrated for 100-year-rain events.	Differences in sediment yield not statistically significant.	Surface and shallow subsurface runoff rates were greatest in the buffer areas than in the harvested areas or in the harvest-buffer interfaces especially during dry conditions. The authors speculate this was likely due to the greater soil porosity in the disturbed, harvested areas. Sediment concentration in the runoff, however, was approximately 15.8 times higher for the harvested area than in the riparian buffer, and 4.2 times greater than in the harvest-buffer interface. Total sediment yields from the harvested area (runoff + sediment concentration) were approximately 2 times greater than in the buffer areas, and 1.2 times greater in the harvest-buffer interface, however this difference was not significant.
<a href="#">Rachels et al., 2020</a>	harvested following the current Oregon Forest Practices Act policies and BMPs	proportion of sediment from sources	Sediment collected in traps; sourced using chemical analysis	limited sample size (1 treatment, 1 paired reference watershed) and does not incorporate the effects of different watershed physiography on sediment erosion.	The proportion of suspended sediment sources were similar in the harvested (90.3 + 3.4% from stream bank; 7.1 + 3.1% from hillslope) and unharvest (93.1 + 1.8% from streambank; 6.9 + 1.8% from hillslope) watersheds. In the harvested watersheds the sediment mass eroded from the general harvest areas (96.5 + 57.0 g) was approximately 10 times greater than the amount trapped in the riparian buffer (9.1 + 1.9 g), and 4.6 times greater than the amount of sediment collected from the unharvested hillslope (21.0 + 3.3 g).

<p><a href="#">Safeeq et al., 2020</a></p>	<p>Long term (51 years) effects of clearcut to stream followed by broadcast burn.</p>	<p>streamflow, sediment transport</p>	<p>Historical streamflow data, precipitation data, sediment grab samples for bedload and suspended sediment.</p>	<p>Data compared one treatment watershed and one control watershed across 51+ years.</p>	<p>The results for post-treatment sediment yields showed suspended load declined to pre-treatment levels in the first two decades following treatment, bedload remained elevated, causing the bedload proportion of the total load to increase through time. Changes in streamflow alone account for 477 Mg/km<sup>2</sup> (10%) of the suspended load and 113 Mg/km<sup>2</sup> (5%) of the bedload over the post-treatment period. Increase in suspended sediment yield due to increase in sediment supply is 84% of the measured post-treatment total suspended sediment yield. In terms of bedload, 93% of the total measured bedload yield during the posttreatment period can be attributed to an increase in sediment supply. The authors conclude that Following harvest, changes on streamflow alone was estimated in being responsible for &lt; 10% of the resulting suspended sediment transported into streams, while the increase in sediment supply due to harvest disturbance was responsible for &gt;90%.</p>
<p><a href="#">Wise, 2010</a></p>	<p>Streamflow patterns derived from instrumental data and from reconstructed tree-ring chronologies were compared with other previously reconstructed rivers in similar climates.</p>	<p>Streamflow</p>	<p>Dendrochronology, historical data records, seasonal patterns</p>	<p>The reconstruction model developed for the analysis explained 62% of the variance in the instrumental record after adjustment for degrees of freedom.</p>	<p>Results showed evidence that droughts of the recent past are not yet as severe, in terms of overall magnitude, as a 30-year extended period of drought discovered in the mid-1600s. However, in terms of number of individual years of &lt; 60% mean-flow (i.e., low-flow years), the period from 1977-2001 were the most severe. Considering the frequency of consecutive drought years, the longest (7-year-droughts), occurred in the early 17th and 18th centuries. However, the 5-year drought period from 2000-2004 was the second driest period over the 415-year period examined.</p>
<p><a href="#">Wissmar et al., 2004</a></p>	<p>Data sourced from management records and geospatial data to identify high erosion-risk areas.</p>	<p>Sediment, weather, stand characteristics, landscape factors</p>	<p>unstable soils, immature forests, roads, critical slopes for land failure, and rain-on-snow events</p>		<p>The highest-risk areas contained a combination of all landscape cover factor combinations (rain-on-snow zone, critical failure slope, unstable soil, immature forests, and roaded areas). The lowest risk categories contained only rain-on-snow zones, and critical failure slopes. Roaded areas and unstable soils were only present in risk categories 3-6.</p>

3487 Table A-4. List of treatments, variables, metrics, and results from publications reviewed for information on shade and stream temperature.

Reference	Treatment	Variables	Metrics	Notes	Results
<a href="#">Bladon et al., 2016</a>	15 m buffer with a minimum of ~3.7 m <sup>2</sup> conifer basal area retained for every 300 m length of stream). Historical data with no streamside vegetation maintenance (i.e., no buffer) .	Stream temperature	7-day moving mean stream temperature, daily mean stream temperature, and diel stream temperature fluctuation. Data was recorded with Tidbit data loggers.	The authors caution that the streams in this study have potential for a muted stream temperature response following harvest relative to other regions because of the (1) north-south stream orientation (2) steep catchment and channel slopes, (3) potential increases in groundwater contributions after harvesting.	Under the contemporary Oregon Forest Practices Act there was no significant changes in the 7-day moving mean of daily maximum stream temperature, mean daily stream temperature, and diel stream temperature for 3 years following harvest when analyzed across all sites for all summer months (July – September). There was a significant increase in the 7-day moving maximum temperature from pre- to post-harvest values when data was constrained to the period of July 15 – August 15 by $0.6 \pm 0.2$ °C. However, when analyzed by individually paired sites and when interannual and site variability was accounted for, no significant changes in stream temperature were observed. The authors caution that these results should not be generalized to areas outside the Oregon coast or to riparian areas of different contexts (see notes).
<a href="#">Bladon et al., 2018</a>	Buffer widths at harvested sites varied but averaged 20 m on either side of streams.	Stream temperature, lithology	the 7-day moving average of daily maximum stream temperature adjacent to and downstream of harvest.	Conducted at 3 paired watershed studies on the coast and western Cascades of Oregon. The pre-harvest relationship in stream temperatures for paired sites were used to create predicted changes in stream temperatures post-harvest. Post-harvest stream temperatures exceeding the predictive temperature interval by more than 95% were reported as significant.	Results showed an increase in stream temperatures beyond the 95% predictive interval (PI) at 7 of the 8 sites within harvest areas. 4 of these 7 sites exceeded the PI between 22 and 100% of the time (all summer months for 3 years following harvest). In the remaining 3 sites, exceedance only occurred between 0 and 15% of the time. There was no evidence of elevated stream temperatures beyond the predicted intervals in any of the downstream sites following harvesting. At the harvested sites there was a strong relationship between stream temperature increases and catchment lithologies, but no statistically significant relationship between stream temperature changes and percent of catchment harvested. Downstream sites showed a strong relationship between stream temperatures and the interaction of harvest percentage and lithology. The greatest temperature increases at downstream sites were in areas with a higher percentage of catchment harvested and were underlain by more resistant lithologies. There was no evidence for increases in stream temperatures in catchments with a high percentage of harvest that were underlain by permeable geology

<p>Cole &amp; Newton, 2013</p>	<p>clearcut to stream, partial buffer (12 m width on predominant sun-side), Oregon state BMP (15-30 m no-cut buffer both sides)</p>	<p>Stream temperature</p>	<p>Controlled for yearly fluctuations in temperatures by analyzing the difference in stream temperature entering and exiting the reach with digital temperature data loggers</p>	<p>Stream temperature data collected for 2 – years prior and 4 to 5 years following harvest. Unharvested control sites were located downstream of treatment sites. Treatment applied to four small fish-bearing streams.</p>	<p>Results showed the most significant increases in daily maximum, and mean, and diel fluctuations in temperatures post-harvest for all no tree buffers. Changes to daily maxima ranged from -0.11 to 3.84 °C, and changes to daily minimum ranged from -1.12 to 0.49 °C. The no tree buffers also showed small but significant changes below predicted summer minima between -1.12 and -0.49 °C. The partial buffer units varied in their response to treatment exhibiting increases, decreases, and no change from preharvest trends.</p>
<p>Cupp &amp; Lofgren, 2014</p>	<p>the “all available shade” rule (ASR), and the standard rule (SR) in eastern WA. ASR: requires retention of all available shade within 75 feet of the stream. SR: some harvest is allowed within the 75-foot buffer depending on elevation and pre-harvest canopy cover.</p>	<p>Canopy closure, shade measurements, stream temperature</p>	<p>Hand-held densiometer (canopy closure), self-leveling fisheye lens digital camera (shade), temperature data loggers</p>	<p>Sites were between 65-100 years old and were situated along second to fourth order streams with harvest-regenerated or fire-regenerated forests. Reference reaches were located upstream from treatment reaches where harvest was applied.</p>	<p>Results showed post-harvest shade values decreased in SR sites (mean effect of -2.8%, <math>p = 0.002</math>), as did the canopy closure values (mean effect of -4.5%, <math>p &lt; 0.001</math>). Shade and canopy closure values did not significantly change in the ASR sites. Mean shade reduction in the SR treatment sites exceeded the mean shade reduction in the ASR sites by 3%. Canopy closure reduction was also greater in the SR sites than in the ASR sites by a mean of 4%. Site seasonal means of daily maximum stream temperature treatment responses in the first two years following harvest ranged from - 0.7 °C to 0.5 °C in the ASR reaches and from -0.3 to 0.6 in the SR reaches. Site seasonal mean post-harvest background responses in reference reaches ranged from - 0.5 °C to 0.6 °C in the first two years following harvest. Mean daily maximum stream temperature increased 0.16 °C in the SR harvest reaches, whereas stream temperatures in both the ASR sites and in the no-harvest reference reaches increased on average by 0.02 °C.</p>
<p>Ehinger et al., 2021</p>	<p>1) TRT: two-sided 50-ft riparian buffer along at least 50% of the RMZ 2) REF: Unharvested Reference.</p>	<p>Canopy closure estimated from densiometer, stream water temperature at 30-minute intervals using StowAway TidBit thermistors</p>		<p>Soft Rock study. Multiple Before-After Control-Impact (MBACI) study design. Because of unstable slopes, total buffer area was 18 to 163% greater than a simple 50-ft buffer along 50% of the stream length.</p>	<p>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. During and after harvest, mean monthly water temperatures were higher, but equaled or exceeded 16.0°C in 2 treatment sites by up to 1.8°C at one site (for 5 years post-harvest)</p>

					and by 0.1°C at another (at year 5 post-harvest). None of the three REF sites exceeded 16°C during the study.
<a href="#">Gravelle &amp; Link, 2007</a>	50% of the drainage area clearcut to stream edge, thinned to a 50% target shade removal in Fall 2001, and an unimpacted control. Riparian buffer zones were implemented according to Idaho Forest Practices.	stream temperatures at the headwater streams immediately adjacent to treatments, and downstream in larger fish-bearing streams.	Stream temperature data collected from digital sensors.	for the non-fish-bearing, headwater sites pre-treatment data was only collected one season prior to treatment.	In general, the downstream sites showed a cooling effect between -0.2 and -0.3°C. The estimated cooling effect could not be attributed to any cause (e.g., increase in water yield), but the authors conclude that there was no post-harvest increase in peak summer temperatures at the downstream sites. For streams immediately adjacent to the clearcut treatment (headwater streams) a significant increase in temperature was detected at 2 sites ranging between 0.4 and 1.9°C, while a marginally significant decrease in temperature was detected at the third site (-0.1°C, p = 0.06). At the sites located immediately adjacent to partial cuts, results showed mixed results with decreases in temperature (-0.1°C; non-significant) at one site and significant but minimal changes at another site (0.0-3.0°C) across the individual post-harvest years. Overall, there were minimal to no changes in stream peak temperatures following treatment in the partial-cut riparian areas. Despite slight increases in temperature in 2 of the headwater streams, no increase in stream temperature was detected in the larger downstream fish-bearing streams.
<a href="#">Groom et al., 2011a</a>	Private site FPA rules are 15 and 21 m wide on small and medium fish-bearing streams of limited entry. State sites followed a 52 m wide buffer of limited entry. FPA = 6 m no entry buffer, State = 8 m no entry buffer. Thinning intensity not specified.	Stream temperature	Stream temperature collected with digital temperature sensors within harvested areas before and after treatment.	Eighteen of the 33 sites were on privately owned lands, and the other 15 were on state-managed forest land. Treatment reaches were harvested according to the FPA or FMP and included 26 clear-cuts and 7 partial cuts. All private sites were clear-cut. Seventeen sites were	Pre harvest to post harvest comparison of 2 years of data will detect a temperature change of > 0.3°C. Conversely, harvest to state FMP standards resulted in an 8.6% probability of exceedance that did not significantly differ from all other comparisons. The a-priori and secondary post hoc multi-model comparisons did not indicate that timber harvest increased the probability of PCW exceedance at state sites. The authors point out that the 0.3°C change threshold still lies 1 or 2 orders of magnitude lower than previous findings from studies which took place prior to the enactment of the riparian protection standards. Note: PCW criterion is that anthropogenic activities are not permitted to increase

				harvested along one stream bank, of which 13 were state forest sites. The remaining 16 sites were harvested along both banks.	stream temperature by more than 0.3 °C above its ambient temperature
<a href="#">Groom et al., 2011b</a>	Private site FPA rules are 15 and 21 m wide on small and medium fish-bearing streams with a 6 m no-cut zone immediately adjacent to the stream. Harvesting is allowed in the remaining RMA to a minimum basal area of 10.0 (small streams) and 22.9 (medium streams) m <sup>2</sup> /ha. State sites followed a 52 m wide buffer with an 8 m no cut buffer. Limited harvest is allowed within 30 m of the stream only to create mature forest conditions.	Stream temperature, Shade, canopy cover	Stream temperature collected with digital temperature sensors. Stream temperature data was summarized to provide daily minimum, maximum, mean, and fluctuation for analysis. The temperature data was modeled using mixed-effects linear regression. Shade analysis included trees per hectare, basal area per hectare, vegetation plot blowdown, and tree height. a linear regression analysis of shade data (n = 33) was performed.	A comparison of within site changes in maximum temperatures pre-harvest to post-harvest showed an overall increase at private sites, but not all sites behaved the same and some had decreases in maximum temperatures.	Following harvest, maximum temperatures at private sites increased relative to state sites on average by 0.71 °C. Similarly, mean temperatures increased by 0.37 °C (0.24 - 0.50), minimum temperatures by 0.13 °C (0.03 - 0.23), and diel fluctuation increased by 0.58 °C (0.41 - 0.75) relative to state sites. The average of maximum state site temperature changes = 0.0 °C (range = -0.89 to 2.27 °C). Observed maximum temperature changes at private sites averaged 0.73 °C (range = -0.87 to 2.50 °C) and exhibit a greater frequency of post-harvest increases from 0.5 to 2.5 °C compared to state sites. Private site shade values also appeared to decrease pre-harvest to post-harvest. Private post-harvest shade values differed from pre-harvest values (mean change in Shade from 85% to 78%); however, no difference was found for state site shade values pre-harvest to post-harvest (mean change in Shade from 90% to 89%). Results from this study show that between 68% and 75% of variability in post-harvest shade may be accounted for by basal area within 30 m of the stream, tree height, and potentially blow down. The authors speculate that their results suggest sites with shorter trees have higher post-harvest shade and this may be due to the negative correlation between crown ratios and tree heights.
<a href="#">Guenther et al., 2014</a>	Partial retention (50% removal of basal area including riparian zone) methods resulting in approximately 14% reduction in canopy cover on average	Stream temperature, canopy cover, bed temperature	Bed temperatures, stream temperatures, and near stream shallow groundwater temperatures were collected with thermocouples.		Treated watersheds showed an increase of 1.6 - 3.0 °C in daily maximum stream temperatures during the summer months following harvest. Bed temperatures showed an overall increase in temperature but at lower magnitude averaging around 1 °C for up to 30 cm in depth. Bed temperature increases were higher in areas on downwelling flow than in areas of neutral and upwelling flows.



<p>Hunter &amp; Quinn, 2009</p>	<p>an alluvial study site and a bedrock study site whose overall characteristics were otherwise comparable apart from geomorphology.</p>	<p>Stream temperature, Alluvial depth</p>	<p>Water temperature was recorded at 75-m intervals along each channel during the summers of 2003 and 2004</p>	<p>Small sample sizes, results only from two sites for two summers. Actual numeric values not reported but shown in graphs.</p>	<p>Results from this study show consistent differences in stream temperature response in alluvial versus bedrock channels. Seasonal maximum and minimum average daily temperatures varied less at the alluvial site compared to the bedrock site. Two same-day measurements at each site showed the alluvial site gaining 8% of its flow, as compared to the bedrock site whose flow decreased by approximately 15%. Bedrock sites were shown to have the highest variation in reach-scale water temperatures during low flow.</p>
<p>Janisch et al., 2012</p>	<p>clearcut logging with two riparian buffer designs: a continuous buffer and a patched buffered stream. Buffers were 10-15 m wide.</p>	<p>Stream temperature</p>	<p>Channel and catchment attributes (e.g., BFW, Confinement, slope, FPA, etc.), Stream temperatures were recorded with a Tidbit datalogger in areas persistently submerged.</p>	<p>Separation of treatment streams into “clusters” based on year of treatment and an unbalanced experimental design resulted in small sample sizes. Thus, significant differences between treatments were not analyzed. Instead results presented as “significant” represent a significant increase in temperature different from zero.</p>	<p>In general, timber harvest with fixed-width continuous buffers, or patch buffers resulted in increased mean maximum daily summer stream temperatures in the first year following treatment by an average of 1.5 °C (range 0.2 – 3.6 °C). Mean maximum daily summer temperature increases were higher in the streams adjacent to continuous buffer (1.1 °C; range 0.0 to 2.8°C) than the patch buffered catchments (0.6 °C; range – 0.1 to 1.2°C). However, results were highly variable. Post-treatment temperature changes suggested that treatments (p=0.0019), the number of years after treatment (p=0.0090), and the day of the year (p=0.0007) were all significant effects explaining observed changes in temperature. Wetland area (0.96, p&lt;0.01) and length of surface flow (0.67, p=0.05) were strongly correlated with post-logging temperature changes.</p>
<p>Johnson &amp; Jones, 2000</p>	<p>clearcut to stream, patch cutting followed by debris flows (resulted in the removal of all streamside vegetation) , 450+ yo Doug-fir forest reference.</p>	<p>Stream temperature</p>	<p>long term monitoring of weekly stream temperature max, min, and average. Solar radiation data collected from digital sensors. Air and precipitation temperatures collected from local weather stations.</p>	<p>The experimental design used historic stream temperature data to examine changes in stream temperatures. This required conflating data from 2 different devices.</p>	<p>Removal of streamside vegetation whether by clearcut and burn (CCB), or patch-cut and debris (PCD) flow led to significant increases in mean weekly summer maximum and minimum stream temperatures relative to reference streams in the summer immediately following and for 3-4 years post treatment. The CCB’s summer mean weekly maximum stream temperatures ranged from 5.4-6.4°C higher than the reference stream for 4 years following treatment. The PCD’s summer mean weekly stream temperatures ranged from 3.5-5.2°C higher than the reference stream for 3 years following treatment. The diurnal fluctuations were significantly higher in both treatment streams (6-8 °C in CCB, and 5-6 °C in PCD) relative to reference stream (1-2°C). Pre-harvest temperatures recovered after 15 years of growth.</p>

					Differences in treatment streams and reference stream temperatures were less than 1.1°C pre-treatment and 30-years post-treatment.
<a href="#">Kaylor et al., 2017</a>	50 years post clearcut to streams, control stands were >300 years old	stream light availability, forest age	Stream bank-full width, wetted width, canopy openness, % red alder, and estimated photosynthetically active radiation (PAR) were quantified at 25-m intervals		PAR reaching streams was on average 1.7 times greater in >300-year-old forests than in 30–100-year-old forests. The greatest differences were in streams with both sides harvested. Mean canopy openness was higher in >300-year-old forests (18%) than in 30–100-year-old forests (8.7%). Space-for-time analysis with reviewed literature estimates that canopy closure and minimum light availability occurs at approximately 30 years and maintains until 100 years.
<a href="#">Kibler et al., 2013</a>	Clearcut to stream	Stream temperature, discharge rate,	Stream temperature and discharge rate were recorded with thermistor gauging stations. Canopy cover was recorded with a densiometer as portion of sky covered with vegetation	Post-harvest data was collected only during the summer and autumn immediately following harvest (i.e., 1 season of post-harvest data). Pre-harvest data was collected for 3 years.	Harvest in treatment watersheds resulted in a significant decrease in stream temperatures ranging from –1.9 to -2.8 °C relative to pre-treatment temperatures. The authors attribute the lack of increased temperatures to the shade provided by woody debris.
<a href="#">Macdonald et al., 2003b</a>	Low-retention – remove all timber >15 or >20 cm DBH for pine or spruce, 20 m of the stream 2) high-retention – remove timber >30 cm DBH 20-30m of stream, and 3) Patch-cut removal of all vegetation in the upper 40% of the watershed.	Stream temperature	Temperature data were recorded with Vemco dataloggers. Canopy cover was estimated with densiometers.		Significant increase in stream temperatures ranging from 4 – 6 °C at five years post-harvest, and increased ranges of diurnal temperature fluctuations for all treatment streams relative to the reference streams. Streams that had summer maximum mean weekly temperatures of 8°C before harvesting had maximum temperatures near 12°C or more following harvesting. Daily ranges of 1.0–1.3°C before harvesting became 2.0–3.0°C following harvesting, high-retention buffer treatment mitigated temperature increases for the first three years. Still, increased mortality (attributed to windthrow) caused a reduction in the canopy that, thus, led to increased stream temperatures equivalent to other treatment streams by year five.

<p>McIntyre et al., 2021</p>	<p>(1) unharvested reference, (2) 100% treatment, a two-sided 50-ft riparian buffer along the entire Riparian Management Zone (RMZ), (2) FP treatment a two-sided 50-ft riparian buffer along at least 50% of the RMZ, (3) 0% treatment, clearcut to stream edge (no-buffer).</p>			<p>Hard Rock Study.</p>	<p>Results for canopy cover showed that riparian cover declined after harvest in all buffer treatments reaching a minimum around 4 years post-harvest (after mortality stabilized). The treatments, ranked from least to most change, were REF, 100%, FP, and 0% for all metrics and across all years. Effective shade results showed decreases of 11, 36, and 74 percent in the 100%, FP, and 0% treatments, respectively. Significant post-harvest decreases in shade were noted for all treatments and all years. Results for stream temperature showed that within treatment mean post-pre-harvest difference in the REF treatment never exceeded 1.0°C. In contrast, mean within treatment difference in the 100% treatment was 2.4°C in 2009 (Post-harvest year 1) but never exceeded 1.0°C in later years. The mean difference in the FP treatment exceeded 1.0°C immediately after harvest then again in 2014–2016 (post-harvest years 6–9) while in the 0% treatment the mean difference was 5.3°C initially, then decreased over time to near, but never below, 0.9°C. Stream temperature increased post-harvest at most locations within all 12 harvested sites and remained elevated in the FP and 0% treatments over much of the nine years post-harvest.</p>
<p>Pollock et al., 2009</p>	<p>A range of harvest from 0 – 100%, &lt; 20 years old regrowth, ~ 40 years old regrowth . Unharvested sites were estimated as being &gt;150-years old</p>	<p>Stream temperature, time since harvest, percent of watershed and stream network harvested.</p>	<p>average daily maximum (ADM), average daily range, seasonal range, average, maximum, and minimum Stream temperatures collected with Tidbit data loggers. Stand age grouped by time since harvest.</p>	<p>tested 3 hypotheses: (1) the condition of the riparian forest immediately upstream of a site primarily controls stream temperature, (2) the condition of the entire riparian forest network affects stream temperature, and (3) the forest condition of the entire basin affects stream temperature.</p>	<p>Results of general temperature patterns showed that average daily maximum (ADM) were strongly correlated with average diurnal fluctuations (<math>r^2 = 0.87</math>, <math>p &lt; 0.001</math>, <math>n = 40</math>), indicating that cool streams also had more stable temperatures. For basin-level harvest effects on stream temperatures. The percentage of the basin harvested explained 39% of the variation in the ADM among subbasins (<math>r^2 = 0.39</math>, <math>p &lt; 0.001</math>, <math>n = 40</math>) and 32% of variation in the average daily range (ADR) (<math>r^2 = 0.32</math>, <math>p &lt; 0.001</math>, <math>n = 40</math>). The median ADM for the unharvested subbasins was 12.8 °C (mean = 12.1 °C), which was significantly lower than 14.5 °C, the median (and average) ADM for the harvested subbasins (<math>p &lt; 0.001</math>). Likewise, the median (and average) ADR for the unharvested subbasins was 0.9 °C, which was significantly lower than 1.6 °C, the median ADR (average = 1.7 °C) for the harvested subbasins (<math>p &lt; 0.001</math>). Results for the correlations between the riparian network scale forest harvest and stream temperature showed that the total</p>

				<p>percentage of the riparian forest network upstream of temperature loggers harvested explained 33% of the variation in the ADM among subbasins (<math>r^2 = 0.33</math>, <math>p &lt; 0.001</math>, <math>n = 40</math>) and 20% of variation in the ADR (<math>r^2 = 0.20</math>, <math>p = 0.003</math>, <math>n = 40</math>). However, the total percentage of upstream riparian forest harvested within the last 20 years was not significantly correlated to ADM or ADR. Results for near upstream riparian harvest and stream temperature showed either non-significant, or very weakly significant correlations. For example, there were no significant correlations between the percentage of near upstream riparian forest recently clear-cut and ADM temperature (<math>r^2 = 0.03</math>, <math>p = 0.79</math>, <math>n = 40</math>), the ADR of stream temperatures (<math>r^2 = 0.02</math>, <math>p = 0.61</math>, <math>n = 40</math>) or any other stream temperature parameters. The proportion of total harvested near upstream riparian forest (avg = 0.66, SD <math>\pm</math> 0.34, range = 0.0-1.0) was weakly correlated with ADM (<math>r^2 = 0.12</math>, <math>p = 0.02</math>, <math>n = 40</math>) and not significantly correlated with ADR (<math>r^2 = 0.07</math>, <math>p = 0.06</math>, <math>n = 40</math>). Even when the upstream riparian corridor length was shortened to 400 m and then to 200 m, and the definition of recently harvested was narrowed to &lt;10 year, no significant relationships between temperature and the condition of the near upstream riparian forest was found. for these models, the percentage of basin area harvested was the best predictor of variation in mean maximum stream temperatures. The probability of stream temperatures increasing beyond DOE standards (16 °C for seven-day average of maximum temperatures) increased with percent harvest. Nine of the 18 sites with 50-75% harvest and seven of the nine sites with &gt;75% harvest failed to meet these standards. The authors interpret these results as evidence that the total amount of forest harvested within a basin, and within a riparian stream network are the most important predictors of changes in summer stream temperatures. They conclude that watersheds with 25-100% of their total area harvested had higher stream temperatures than watersheds with little or no harvest.</p>
--	--	--	--	--

<p>Reiter et al., 2020</p>	<p>Clearcut, no buffer (CC_NB), clearcut with 10-m no cut buffer (CC_B), thinning with 10 m no-cut buffer (TH_B), and unharvested reference (REF) streams.</p>	<p>Stream temperature</p>	<p>Temperature data was separated into 5<sup>th</sup>, 25<sup>th</sup>, 50<sup>th</sup>, 75<sup>th</sup>, and 95<sup>th</sup> percentiles. the researchers also quantified the percentage of summer where temperatures where above 16 and 15 °C.</p>	<p>Sample sizes are relatively low for some treatments. (CC_NB; n = 4); (CC_B; n = 3); (TH_B; n =1); (REF; n = 7).</p>	<p>A 10 m buffer was sufficient in maintaining summer temperature changes compared to reference streams regardless of upland treatment (clear-cut, thinning). Unbuffered streams (Clear-cut to streams) showed significant increases in stream temperatures with an average of 3.6 °C (SE = 0.4) increase relative to reference streams. Unbuffered streams spent 1.3% and 4.7% of the recorded time above 16 °C and 15 °C respectively (habitat temperature thresholds for two local amphibian larvae, coastal tailed frog, coastal giant salamander). The authors conclude that while significant changes in mean and percentile changes in temperature were observed, the amount of time spent above critical temperature thresholds for important amphibian species was minimal.</p>
<p>Reiter et al., 2015</p>	<p>. Various buffer prescriptions as regulations changed over time. (mid1970s – 1980s = “nominal”; mid 1980s – mid 1990s = 23 m; 2001 – 2009 = 30 m buffers)</p>	<p>Stream temperature data from four permanent sampling stations in the Deschutes River Watershed from 1975- 2009. Results for this analysis are for 3 watersheds (1-large, 1-medium, 1-small)</p>	<p>Long term stream and air temperature collected from sampling stations. To detect correlations of stream and air temperature change with land management activity separately from climate changes the data was fit to a model that included the effects of climate.</p>	<p>Methods for stream temperature data collection varied at different periods resulting in a margin of error for monthly temperatures of 0.14°C for 1975 - 1983, 0.09°C for 1984 – 1999, and 0.02°C. for 2000 – 2009.</p>	<p>Results for trends in stream temperature over the 35-year study period without adjustment for climate change showed no statistically significant trend in water temperature changes for the large watershed, while the medium watershed (Thurston Creek) showed decreasing trends in TMAX_WAT for June, July, and August, ranging in magnitude from 0.05°C (August) to 0.08°C (July) per year. For the smaller watershed, Hard Creek (Ware Creek was not included in this analysis), had significant decreasing trends in TMAX_WAT for July, August, and September. The magnitude of these trends was yearly decreases of TMAX_WAT by 0.05, 0.08, and 0.05°C, for July, August, and September, respectively. Significant changes in trends for TMIN_WAT were only found for the large basin site with yearly increases of 0.04, 0.03, and 0.04°C for July, August, and September, respectively. Results for stream temperature trends after adjusting for changes in air temperature (climate) showed significant decreasing trends in TMAX_WAT for the large basin by 0.04, 0.03, and 0.04°C yearly, for July, August, and September, respectively. For the medium basin, trends showed yearly decreases in TMAX_WAT of 0.07, 0.08, 0.06, and 0.03 for June, July, August, and September, respectively. For the small basin, climate adjusted trends in TMAX_WAT showed significant decreases in yearly trends by 0.05, 0.08, and 0.05 for July, August, and September, respectively. When stream temperature was examined with its correlation with estimated annual</p>

					shade recovery from initial harvest (indexed by ACD). Significant correlations were found for monthly temperature metrics that were adjusted for climate, for all basins. The authors conclude that the results of this study show evidence that implementation of protection buffers in this area were sufficient in maintaining stream temperatures. Conversely, this study also shows evidence that despite these protections from land management induced stream temperature changes, these protections have been somewhat offset by the warming climate conditions.
<a href="#">Roon et al., 2021a</a>	Thinning treatments resulting in a mean shade reduction of <5% (-8.0 - -0.5) at one watershed and 23.0% at two watersheds (-25.8, -20.1)	Stream temperature, solar radiation, Shade	Stream temperature was collected using digital sensors; solar radiation was measured using silicon pyranometers; riparian shade was measured using hemispherical photography.	Only 1-year pre- and post-treatment data. Site selection and replication was not random and thus may not be applicable outside of the northern California redwood forests.	No significant changes in stream temperatures were detected in the low-intensity thinning treatment watersheds. For the higher intensity thinning treatments. Maximum weekly average of the maximum temperatures increased during spring by a mean of 1.7 °C (95% CI: 0.9, 2.5), summer by a mean of 2.8 °C (1.8, 3.8), and fall by a mean of 1.0 °C (0.5, 1.5) and increased in downstream reaches during spring by a mean of 1.0 °C (0.0, 2.0) and summer by a mean of 1.4 °C (0.3, 2.6). Thermal variability of streams were most pronounced during summer increasing the daily range by a mean of 2.5 °C (95% CI: 1.6, 3.4) and variance by a mean of 1.6 °C (0.7, 2.5), but also increased during spring (daily range: 0.5 °C; variance: 0.3 °C) and fall (daily range: 0.4 °C; variance: 0.1 °C). Increases in thermal variability in downstream reaches were limited to summer (daily range: 0.7 °C; variance: 0.5 °C). The authors interpret their results as evidence that that changes in shade of 5% or less caused minimal changes in temperature while reductions in shade of 20–30% resulted in much larger increases in temperature.
<a href="#">Roon et al., 2021b</a>	Effective shade reductions ranging between 19-30% along 200 m reach, or 4-5% along 100 m reach.	local and downstream temperature	Stream temperature collected with digital temperature sensors within harvest area and every 200 m downstream of stream network.	Stream temperature data was only collected for one-year pre- and one-year post-harvest.	In the reaches with higher reductions in shade (19-30%) there was accumulation of 45° to 115°C additional degree days from pre- to post treatment years, while the reaches with lower reductions in shade (4-5%) only accumulated 10° to 15°C additional degree days. Travel distance of increased stream temperatures also appeared to be dependent on thinning intensity. The lower shade reduction reaches had an increased temperature effect downstream with travel distance of 75-150 m, while the high shade reduction sites had a downstream travel

					distance of 300- ~1000 m. In the high shade reduction sites, treatment reaches that were further apart (> 400 m) showed dissipation in increased stream temperatures downstream, while in parts of the stream where treatments were <400 m apart, temperature increases did not always dissipate before entering another the next treatment reach.
<a href="#">Sugden et al., 2019</a>	Montana state law : 15.2 m wide buffers no more than half the trees greater than 204 mm (8 in) diameter at breast height (DBH). In no case, however, can stocking levels of leave trees be reduced to less than 217 trees per hectare. .	Stream temperature, fish population, Canopy cover	Daily max, min, and average stream temperatures collected with data loggers during summer months. The fish community was inventoried 100 m reaches using an electro-fishing pass of capture method. Canopy cover was estimated using a combination of simulation modeling and using a concave spherical densiometer.	Data only collected for one year pre-harvest and one year post-harvest.	The mean basal area (BA) declined from 30.2 m <sup>2</sup> /ha pre-harvest to 26.4 m <sup>2</sup> /ha post-harvest (mean = -13%, range from -32% to 0%). Windthrow further reduced the mean BA to 25.9 m <sup>2</sup> /ha (mean = -2%, range = -32% -0%). Change in mean canopy cover were not significant based on the simulation modeling (-3%), or densiometer readings (+1%). Results of the model for the effect of harvest on stream temperature showed no detectable increase in treatment streams relative to control streams. The estimated mean site level response in maximum weekly maximum temperatures (MWMT) varied from - 2.1 °C to +3.3 °C. Overall, 20 of 30 sites had estimated site level response within ±0.5 °C. There were five sites that had an estimated site-level response greater than 0.5 °C (i.e. warming) and five sites that had an estimated site level response less than -0.5 °C (i.e. cooling). Results for the fish population showed approximately 7% increase in trout population from pre-harvest to post-harvest, but this difference was not significant.
<a href="#">Swartz et al., 2020</a>	In the experimental reaches 30 m gaps were created, centered on a tree next to the stream and at least 30 m in from the beginning of the reach. Actual gap sizes varied across sites from approximately 514 m <sup>2</sup> to 1,374 m <sup>2</sup> with a mean of 962 m <sup>2</sup> .	Stream temperature, Light reaching stream, canopy cover	Riparian shade-hemispherical photos. Light reaching the stream- photodegradation of fluorescent dyes. Stream temperature - HOBO sensors for seven-day moving average of mean and maximum temperatures.	Data was collected for one year pre-harvest, during harvest year (harvest took place in late fall 2017), and one-year post-harvest.	Results showed that after gaps were cut, the BACI analysis showed strong evidence for significant increase in mean reach light (p < 0.01) to a mean of 3.91 (SD ± 1.63) moles of photons m <sup>-2</sup> day <sup>-1</sup> , overall resulting in a mean change in light of 2.93 (SD ± 1.50) moles of photons m <sup>-2</sup> day <sup>-1</sup> . Through the entirety of the treatment reach mean shading declined by only 4% (SD ± 0.02%). Overall, the gap treatments did not change summer T 7DayMax or T 7DayMean significantly across the 6 study sites. However, reaches showed a statistically significant effect of the gap for average daily maximums (p < 0.01) and for average daily means (p = 0.02). The regression comparison reveals there will be on average an additional 0.12 °C/°C increase in daily maximum temperature in the reach with a gap.

					Likewise, for the daily mean, for every degree increase in the shaded reference reach, an average additional increase of 0.05 °C in a reach with a small gap is expected. The regression comparison reveals there will be on average an additional 0.12 °C/°C increase in daily maximum temperature in the reach with a gap. Likewise, for the daily mean, for every degree increase in the shaded reference reach, an average additional increase of 0.05 °C in a reach with a small gap is expected.
Warren et al., 2013	Old-growth forests were estimated to be over 500 years old, and mature second growth forests were estimated to be between 31 and 59 years old.	Light reaching bottom of stream, canopy cover	The percent of canopy cover was estimated using a densiometer, the amount of light reaching the bottom of the stream was estimated using a fluorescent dye that degrades overtime from light exposure	Relatively small sample sizes (n = 4). Significant differences were only found in 3 of the four paired reaches.	Results showed that the differences in stream light availability and percent forest cover between old-growth and second-growth reaches were significant in both south-facing watersheds in mid-summer at an alpha of 0.01 for the dye results and 0.10 for the cover results. For the north-facing watersheds differences in canopy cover and light availability (alpha = 0.01, and 0.10 respectively) were only significant at 1 of the two reaches. Overall, three of the four paired old-growth reaches had significantly lower mean percent canopy cover, and significantly higher mean decline in fluorescent dye concentrations. The authors interpret these results as evidence that old-growth forest canopies were more complex and had more frequent gaps allowing for more light availability and lower mean canopy cover, on average, than in adjacent mature second- growth forests.



3489

3490 Appendix II

3491

3492 **Shade and LW**

3493

3494 Anderson & Meleason, 2009

3495

3496 Anderson, P.D., Meleason, M.A., 2009. Discerning responses of down wood and understory  
3497 vegetation abundance to riparian buffer width and thinning treatments: an equivalence-  
3498 inequivalence approach. *Can. J. For. Res.* 39, 2470–2485 <https://doi.org/10.1139/X09-151>

3499

3500 The purpose of this study was to determine the effect of buffer width on understory vegetation  
3501 and down woody responses both within the unthinned buffer and in the adjacent thinned stand. A  
3502 secondary objective of this study was to explore the ability of equivalence-nonequivalence  
3503 statistical tests at assessing the degree of similarity between stands. The focus of this study was  
3504 on second-growth stands dominated by Douglas-fir at multiple sites along the coast and Cascade  
3505 Range in western Oregon. Six combinations of buffer width and upslope density management  
3506 prescription were evaluated: one site potential tree height buffer averaging 69 m adjacent to  
3507 thinning and a 0.4 patch opening; variable width buffer averaging 22 m adjacent to thinning and  
3508 a 0.4 patch opening; streamside retention width averaging 9 m adjacent to thinning; and an  
3509 unthinned stand serving as a reference. Pearson correlation and multivariate analysis of variation  
3510 were used to examine data on percent cover of small and large down wood, and percent cover of  
3511 shrubs, herbs, and moss. Inferences on buffer performance were generated using linear mixed  
3512 model analysis, equivalence-inequivalence tests, and two post-hoc comparisons. The results from  
3513 this study show upland thinning led only to subtle changes in understory vegetation cover and  
3514 composition with vegetation responses most prevalent with narrow buffer widths and particularly  
3515 when adjacent to patch openings. There was a lack of significant change in down wood response  
3516 to treatments.

3517

3518 **Shade**

3519

3520 Anderson et al., 2007

3521

3522 Anderson, P.D., Larson, D.J., Chan, S.S., 2007. Riparian buffer and density management  
3523 influences on microclimate of young headwater forests of western Oregon. *For. Sci.* 53, 254–  
3524 269. <https://doi.org/10.1093/forestscience/53.2.254>

3525  
3526 The purpose of this study was to characterize variation in overstory density, canopy closure, and  
3527 microclimate as a function of distance from headwater streams, and (2) determine differences in the  
3528 ability of thinned stands and unthinned stands to maintain understory microclimate above the stream  
3529 channel and in the riparian zone. The focus of this study was on second-growth (30- to 80-year-old)  
3530 Douglas-fir forests characteristic of western Oregon. The study was located at four sites along the  
3531 Oregon coast and at one site on the western Oregon Cascade Range. Stands were either thinned to  
3532 approximately 198 trees per acre (TPA) or were left unthinned and ranged from 500-865 TPA. Within  
3533 thinned stands, 10% of the area was harvested to create patch openings and 10% was left as clusters of  
3534 “leave islands”. Streams within treated stands were surrounded by buffers of either (1) one site-potential  
3535 tree averaging 69 m (B1), (2) variable width buffer averaging 22 m (VB), or (3) streamside retention  
3536 buffer averaging 9 m (SR-T). These six combinations of buffer width and adjacent density management  
3537 were evaluated using univariate linear modeling and compared with untreated (UT) stands. Microsite  
3538 and microclimate data were obtained through repeated transect measurements extending laterally from  
3539 stream center and into the riparian zone and upland treated stand 2-5 years after treatment. The stand  
3540 basal area was determined through variable radius plot sampling. Canopy cover was estimated through  
3541 photographic techniques during the summer leaf-on period. The results from this study show that the  
3542 ability of narrow streamside buffers (SR-T) at moderating stream microclimate in treated stands was  
3543 questionable. Visible sky at stream center only differed significantly between SR-T (9.6%) and UT  
3544 (4.2%) stands. The SR-T stands showed a +4.5°C difference in daily maximum temperatures just above  
3545 stream center when compared to the UT stands. However, this difference was not statistically significant.  
3546 The researchers report that SR-T had a weak temperature gradient (tested at 0-10 m and 10-30 m  
3547 increments from stream center) indicating the stream center and buffer microclimates were nearly the  
3548 same as upslope in the thinned stand. Within the riparian buffer zones daily maximum temperatures  
3549 were higher in all treated stands when compared to UT stands. The differences in daily maximum  
3550 temperatures ranged from 1.1°C (B1) to 4.0°C (SR-T), but the difference was only significant in one  
3551 SR-T stand. The maximum air temperature within buffer zones adjacent to patch openings was 3.5°C  
3552 higher than in UT stands. Soil temperature changes were only evident within patch openings ranging  
3553 from 3.6 - 8.8°C higher than in UT stands. The researchers of this study conclude by saying that buffers  
3554 with widths defined by the transition of riparian to upslope vegetation or significant topographic slope  
3555 breaks appear sufficient at mitigating effects from upslope harvests on the above-stream microclimate.  
3556 Their suggestions for further study center around cross-disciplinary research into the relationships  
3557 between forest structure, microclimate, and habitat suitability on headwater riparian organisms.

3558

### 3559 **Stream Temperatures**

3560

3561 Cole & Newton, 2013

3562

3563 Cole, E., & Newton, M. (2013). Influence of streamside buffers on stream temperature response  
3564 following clear-cut harvesting in western Oregon. *Canadian journal of forest research*, 43(11), 993-1005.  
3565 <https://doi.org/10.1139/cjfr-2013-0138>

3566

3567 This study compares the changes in stream temperatures following a clearcut with three different buffer  
3568 treatments – no tree buffer, predominantly sun-sided 12 m wide partial buffer, and a two-sided 15-30 m  
3569 buffer (BMP for this area). The study was conducted on four small fish-bearing streams in the area  
3570 surrounding Corvallis, Oregon. Streams were dominated by both hardwood and conifers and were  
3571 located at low- and mid-elevations. Each treatment alternated with unharvested reference sections along  
3572 study reaches spanning 1800-2600 meters. Stream temperature data adjacent to treatment and  
3573 downstream of treatment were collected for 2 –years prior and 4 to 5 years following harvest. Time-  
3574 series regression analysis was used to evaluate the change in temperatures between pre- and post-  
3575 harvest. The researchers controlled for yearly fluctuations in temperatures by analyzing the difference in  
3576 stream temperature entering and exiting the experimental reaches. Results showed significant increases  
3577 in daily maximum, mean, and diel fluctuations in temperatures post-harvest for all no tree buffers (up to  
3578 3.8 °C). The no tree buffers also showed small but significant changes below predicted summer minima  
3579 by as much as 1.2°C. The partial buffer units varied in their response to treatment exhibiting increases,  
3580 decreases, and no change from preharvest trends. For example, at one site, there were no detectable  
3581 changes in means, minima, or diel fluctuations but significantly lower maximum temperatures post-  
3582 harvest ( $p = 0.0021$ ; actual temperatures not reported). Partial buffers at another site reported lower  
3583 trends in mean, maxima, and diel fluctuations in temperature post-harvest, and no difference in minima.  
3584 Only one partial buffer site showed increases in all recorded trends (mean, minima, maxima, diel  
3585 fluctuations). The BMP buffered treatment sites also showed variation in results. One site showed no  
3586 detectable changes, one site showed small but significant ( $p < 0.0350$ ; actual temperatures not reported)  
3587 decreases in downstream temperatures. Only two BMP buffered sites showed significant ( $p < 0.0499$ )  
3588 increases in mean, maxima, and diel fluctuations in temperatures. The highest increase in maxima for  
3589 any BMP buffered site was 5.3°C. Changes in temperature trends in uncut reference post-treatment were  
3590 minimal and attributed to downstream effects from the treatment reaches. However, when post-harvest  
3591 trends in upstream treated sites were higher than pre-harvest temperatures tended to fall below pre-  
3592 harvest values when passing through the unharvested downstream units. For within-unit trends,  
3593 unharvested units downstream from no tree and partial buffers showed trends of significantly decreasing  
3594 daily maximum temperatures. When the data was analyzed by 7-day moving mean maximum  
3595 temperatures, the no tree buffers showed significant increases after harvest. The authors report that most  
3596 partial and BMP buffers resulted in minimal increases or negligible changes to the 7-day moving mean  
3597 maximum temperatures (actual values not reported). Significant changes in one or more temperature  
3598 trends (mean, minima, maxima, diel fluctuations) were detected in all treatment stream post-harvest with  
3599 only one exception at a BMP buffered site This was a well planned and executed experimental design  
3600 that shows how changes in stream temperatures post-harvest are directly related to residual buffer  
3601 treatment while also showing evidence that many other factors such as stream features (orientation,

3602 topography, ground water source) can compound or ameliorate these effects (I.e., changes in temperature  
3603 were highly affected by site factors).

3604

### 3605 **Stream Temperature**

3606

3607 Johnson & Jones, 2000

3608

3609 Johnson, S. L., & Jones, J. A. (2000). Stream temperature responses to forest harvest and debris flows in  
3610 western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences*, 57(S2), 30-39.  
3611 <https://doi.org/10.1139/f00-109>

3612

3613 This paper is a study of the changes in mean stream temperature minimum, maximum, diurnal  
3614 fluctuation, and interannual and seasonal variability following harvest in three small basins of the  
3615 H.J. Andrews experimental watershed between 1962 and 1966. The experimental design used  
3616 historic stream temperature data to examine changes in stream temperature following clear-cut  
3617 (no buffer) and burning in one watershed; patch cutting and debris flows (resulted in the removal  
3618 of all streamside vegetation 3 years after cut) treatments in another watershed; and one old-  
3619 growth uncut reference watershed. All watersheds were dominated by 450-year-old Doug-fir  
3620 forests prior to harvest. Data was analyzed for the period 1959-1997. Mean weekly temperature  
3621 maximum, minimum, and annual fluctuations were compared between all three watersheds using  
3622 a complete factor analysis of variance (ANOVA). The experiment also involved long-term  
3623 monitoring to evaluate time until recovery of pre-treatment temperature fluctuations. Results  
3624 showed a significant increase in stream temperatures in both treatment watersheds after treatment  
3625 compared to the unharvested site. The unharvested watershed showed higher interannual  
3626 variability in maximum stream temperatures ranging from 15 to 19°C. The two treatment  
3627 watersheds, despite differences in disturbances, (clear-cut and burn vs. Patch cut and debris-  
3628 flow) followed similar trajectories from 1966-1982. Stream temperature summer maximums  
3629 reached 23.9°C and 21.7°C 1-2 years post-harvest (clear-cut/burn and patch-cut/debris flow  
3630 respectively) and returned to pre-harvest summer temperatures by 1980 (~15 years post-harvest).  
3631 Both treatment watersheds exhibited significant increases in mean weekly minimum and  
3632 maximum stream temperatures in the summer months immediately following harvest and for at  
3633 least 3 years compared to the unharvested reference. The clear-cut and burn watershed's  
3634 weekly maximum summer temperatures ranged between 5.4 and 6.4°C higher, and mean weekly  
3635 minimum ranged 1.6-2.0°C higher than the reference streams for 4 years post-harvest. The patch-  
3636 cut and debris-flow watershed exhibited mean weekly maximum stream temperatures 3.5-5.2°C  
3637 higher than in the reference stream for 3 years following harvest/disturbance. Prior to harvest and  
3638 30 years post-harvest the mean weekly maximum and minimum stream temperatures for both  
3639 treatment streams differed less than 1.1°C from the reference stream. These differences in stream  
3640 temperatures from treated and untreated sites were amplified during periods of high solar inputs

3641 and reduced during periods of cloud cover. Differences in stream temperatures were greatest  
3642 during the end of July and beginning of June. Diurnal fluctuations in stream temperatures were  
3643 also significantly higher in both treatment watersheds (6-8 °C in the clearcut, and 5-6 °C in the  
3644 patch-cut) relative to the reference stream (1-2 °C). Stream temperatures returned to pre-harvest  
3645 levels after 15 years of growth.

3646

### 3647 **Large Wood (LW)**

3648

3649 Bahuguna et al., 2010

3650

3651 Bahuguna, D., Mitchell, S.J., Miquelajauregui, Y., 2010. Windthrow and recruitment of large woody  
3652 debris in riparian stands. *Forest Ecology and Management* 259, 2048–2055.  
3653 <https://doi.org/10.1016/j.foreco.2010.02.015>

3654

3655 The purpose of this paper was to evaluate the effect of riparian buffer width on windthrow and LW  
3656 recruitment and to contrast data with unharvested controls. This paper also seeks to document the  
3657 geometry of post-harvest windthrow from buffers of varying widths and to develop a model framework  
3658 for incorporating supply of LW originating from windthrow to streams from riparian buffers. The focus  
3659 of this paper is on dense young conifer-dominated forests originating from harvest followed by wildfire.  
3660 This study is located in the Coast Mountains, approximately 60 km east of Vancouver, BC. Two buffer  
3661 widths on each side of the stream (10 m and 30 m) along with an unharvested control were each  
3662 replicated three times in the experiment. The researchers used a strip plot sampling method running  
3663 parallel to the stream to collect data on species, diameter, height, and status (standing live/dead)  
3664 beginning in the year prior to harvest and annually thereafter for seven years. A General Linear Model  
3665 Procedure was used to determine the significance of variables. The Pearson correlation coefficient was  
3666 used to assess correlations and potential predictor variables. Multiple linear regression was then used to  
3667 determine the utility of the variables at determining LW height above the stream. Following harvest,  
3668 11% of initially standing timber was blown down in the first and second years in the 10 m buffer,  
3669 compared to 4% in the 30 m buffer, and 1% in the unharvested controls. Following 8 years post-harvest,  
3670 a significant amount of annual mortality occurred in the unharvested control at 30%, compared to 15%  
3671 in both 30 m and 10 m buffers. 77% of LW was in the 10 cm - 20 cm diameter class while the mean  
3672 diameter of standing trees in riparian buffers was 30 cm indicating small diameter trees were  
3673 significantly more represented in streams. Only 3% of windthrown logs fell perpendicular to the stream  
3674 with the majority falling diagonal-perpendicular relative to the stream. The researchers of this study  
3675 conclude that recruitment of logs into streams lags behind the post-harvest pulse of windthrow by  
3676 several years. The lag depends on the size, species, and condition of logs, and their direction of fall  
3677 relative to stream valley geometry.

3678

3679 **Species Richness**

3680

3681 Baldwin et al., 2012 (Removed from focal list)

3682

3683 Baldwin, L.K., Petersen, C.L., Bradfield, G.E., Jones, W.M., Black, S.T., Karakatsoulis, J., 2012.  
3684 Bryophyte response to forest canopy treatments within the riparian zone of high-elevation small streams.  
3685 Can. J. For. Res. 42, 141–156. <https://doi.org/10.1139/x11-165>

3686

3687 The purpose of this study was to examine the influence of forest harvesting practices and distance from  
3688 the stream on riparian-bryophyte communities. The experiment was limited to the montane spruce forest  
3689 type which is considered moderately open and dominated by lodgepole pine in the uplands and by  
3690 hybrid spruce in well-developed riparian areas. The study took place at five different watersheds located  
3691 approximately 70 km from Kamloops, BC. Three primary treatments: clear-cut (n=7), two-sided buffer  
3692 averaging approximately 15 m on both sides (n=10), and a continuous forest (n=6) were used to sample  
3693 numerous environmental variables including elevation, aspect, slope, buffer width, and CWD decay  
3694 class. Bryophytes (classified into life history strategies), stand structure, and microhabitat were also  
3695 measured 1, 5, and 10 m from the streams edge. Additionally, the DBH of all conifer stems as well as  
3696 percent vegetation cover were measured along transects. All data were collected in July-August of 2007  
3697 and 2008. Minimum time since disturbance for clearcut sites was 13 years versus a minimum of 5 years  
3698 in buffered sites. An analysis of variance was used to compare environmental, stream, and stand  
3699 structure characteristics among canopy treatments. Mean values were calculated for stand structure and  
3700 substrate variables recording in transects. Bryophytes were analyzed within functional groups based on  
3701 growth form, substrate affiliations, and life history. Linear models were used to evaluate the effects of  
3702 distance to stream, forest canopy treatment, and their interaction on response variables. Overall CWD  
3703 did not differ significantly among treatments, although buffer treatment sites had significantly higher  
3704 volume of CWD in early decay classes compared to clearcut and continuous forests. The researchers  
3705 suggest the early decay class CWD in buffer treated sites was likely the result of increased stem  
3706 breakage. After accounting for distance from the stream, the richness and frequency of bryophyte  
3707 functional communities was intermediate to continuous and clearcut sites. Compared to continuous sites,  
3708 buffered sites featured significantly lower richness and frequency of many forest-associated groups.  
3709 Furthermore, buffered sites also did not support increased richness or frequency of disturbance-  
3710 associated species. Clearcut treatments featured higher levels of disturbance associated species including  
3711 colonists, canopy species, and species typically found on mineral soil. Data from this study also showed  
3712 bryophyte species richness and frequency decline with increasing distance from the stream. The authors  
3713 conclude by noting that while bryophyte communities in buffered sites are significantly more diverse  
3714 than communities in clearcut sites, reductions in forest-associated species as well as in the bryophyte  
3715 mat as a result of large-scale forestry indicate that the ecological function of buffer-dwelling bryophyte  
3716 communities may be hindered and could benefit alongside large uncut forest reserves.

3717

3718 **Sediment**

3719

3720 Mueller & Pitlick, 2013

3721

3722 Mueller, E. R., & Pitlick, J. (2013). Sediment supply and channel morphology in mountain river  
3723 systems: 1. Relative importance of lithology, topography, and climate. *Journal of Geophysical*  
3724 *Research: Earth Surface*, 118(4), 2325-2342. <https://doi.org/10.1002/2013JF002843>

3725

3726 This study used correlation analysis to assess the relative impact of lithology, basin relief, mean basin  
3727 slope, and drainage density on in stream sediment supply defined by the bankfull sediment concentration  
3728 (bedload and suspended load). The study used sediment concentration data from 83 drainage basins in  
3729 Idaho and Wyoming. Lithologies of the study area were divided into four categories ranging from  
3730 hardest to softest- granitic, metasedimentary, volcanic, and sedimentary. The results showed the  
3731 strongest correlation of bankfull sediment concentration was with basin lithology, and showed little  
3732 correlation strength with slope, relief and drainage density. As lithologies become dominated by softer  
3733 parent materials (volcanic and sedimentary rocks), bankfull sediment concentrations increased by as  
3734 much as 100-fold. These results suggest that lithology can be more important in estimating sediment  
3735 supply than topography. The authors discuss using a correlative analysis but give little description of  
3736 what that analysis was or how they compare the values of each correlation strength to see if the  
3737 differences were significant.

3738

3739 **CWD Modeling**

3740

3741 Benda et al., 2016

3742

3743 Benda, L.E., Litschert, S.E., Reeves, G., Pabst, R., 2016. Thinning and in-stream wood  
3744 recruitment in riparian second growth forests in coastal Oregon and the use of buffers and tree  
3745 tipping as mitigation. *J. For. Res.* 27, 821–836. <https://doi.org/10.1007/s11676-015-0173-2>

3746

3747 The purpose of this study was to develop a model which examines the effects of riparian thinning  
3748 on in-stream wood recruitment in second growth stands. A secondary objective of this study was  
3749 to model how manual felling of trees in no-harvest buffer zones impacts the effects of thinning.  
3750 The study site was located within the Alcea watershed in central coastal Oregon. Silvicultural  
3751 simulation treatments used the reach scale wood model (RSWM) and included: (1) no harvest  
3752 control; (2) single entry thinning from below (thinning from below removes the smallest trees to  
3753 simulate suppression mortality) with and without a 10 m width no-cut buffers; (3) double entry

3754 thinning from below with the second thinning occurring 25 years after the first with and without  
3755 10 m no-cut buffers (4) Each thinning treatment was also combined with some mechanical  
3756 introduction of thinned trees into the stream encompassing a range between 5 and 20 % of the  
3757 thinned trees. . The simulation model RSWM was run for 100 years in 5-year time steps. In the  
3758 no-harvest control, the model output shows the density of live trees declines from 687 trees-per-  
3759 hectare (tph) in 2015 to 266 tph in 2110 due to natural suppression mortality (-61 % from initial  
3760 conditions). The single-entry thin reduces stand density to 225 tph in 2015 (-67 %) and declines  
3761 further to 160 tph by 2110 (-77 %). The double entry thinning resulted in 123 tph after the  
3762 second thinning in 2040 (-82%) and maintained that density until 2110. Both thinning treatments  
3763 resulted in a substantial reduction of dead trees that could contribute to in-stream wood over  
3764 time. The model output for single entry thinning treatments predicts a 33% or 66% reduction of  
3765 in-stream wood over a century relative to the unharvested reference for harvest on one side or  
3766 both sides of the stream, respectively. Adding the 10-m no cut buffer reduced total loss to 7 and  
3767 14%. Including mechanical tipping of 5,10,15, and 20% of cut stems without a buffer in the  
3768 single entry thinning treatment changes the relative in-stream percentages of wood relative to the  
3769 reference stream to -15, -6, +1, and +6%, respectively. To completely offset the loss of in stream  
3770 wood due to single entry thinning mechanical tipping of 14 and 12% were required without and  
3771 with buffers. Double entry thinning treatments without a buffer predicted further reduction in  
3772 wood recruitment over a century of simulation with 42 and 84% reduction of in stream wood  
3773 relative to the reference stream when one side and both sides of the channel were harvested.  
3774 Adding a 10 m buffer reduced total reduction of in stream wood to 11 and 22% for thinning on  
3775 one and both sides of the channel. To offset the predicted changes of in stream wood volume  
3776 following double entry harvest would require tipping of 10 and 7% of cut stems without and with  
3777 the 10-m buffer. The authors conclude that thinning without some mitigation efforts resulted in  
3778 large losses of in stream wood over a century. However, by including a 10-m no cut buffer or a  
3779 practice of mechanical tipping can offset these losses Although predictions from this study  
3780 contribute to the in-stream wood recruitment conversation moving forward, the model contained  
3781 limitations such as utilizing data from FIA plots which only approximate riparian forest  
3782 conditions.

3783

### 3784 **Modeling Stream Litter Delivery**

3785

3786 Bilby & Heffner, 2016

3787

3788 Bilby, R.E., Heffner, J.T., 2016. Factors influencing litter delivery to streams. *Forest Ecology and*  
3789 *Management* 369, 29–37. <https://doi.org/10.1016/j.foreco.2016.03.031>

3790 The purpose of this study was to understand the relative influence of wind speed and direction,  
3791 topography, litter type, species, and stand conditions on the distance from which litter is  
3792 delivered to streams. This study utilized a combination of field experiments, literature, and



3793 simple models to estimate the width of a delivery areas. The effects of wind speed on litter  
3794 delivery distance were measured on litter samples from two common species of the Pacific  
3795 Northwest, Douglas-fir and red alder by releasing litter from a riparian tree canopy at various  
3796 wind speeds and recording the distances traveled for each litter type at each wind speed. The  
3797 relationship between distance of litter recruitment area and variables of interest (e.g., wind speed,  
3798 topography, litter type...) were determined with a linear mixed effects model Data for wind speed  
3799 and direction was recorded for one year in 30 min intervals along Humphrey Creek in the  
3800 Cascade Mountains of western Washington. Results showed that under the wind conditions  
3801 recorded at Humphrey Creek the majority of the litter recruited into the stream originated from  
3802 within 10 m of the stream regardless of litter or stand type. No difference was found in delivery  
3803 distance and litter type (needles or broadleaf) at young sites. However, needles released at mature  
3804 sites had a higher proportion of cumulative input from greater distances than needles or alder  
3805 leaves released at younger sites. This is likely due to the higher canopy and thus higher release  
3806 position. Litter travel distance was linearly related to wind speed ( $p < 0.0001$ ) Doubling wind  
3807 speed at one site led to a 67-87% expansion of the riparian contribution zone in the study area.  
3808 The results reveal a trend that suggests slope also contributes to the width of the litter  
3809 contributing area. However, the authors did not apply statistical analysis to these values and only  
3810 speculate that increasing the slope from 0-45% would increase the width of the litter contributing  
3811 area by 70%. Overall, the results of this study show evidence that wind speed has a strong effect  
3812 on the width of litter delivery areas within riparian areas, but that relationship is also affected  
3813 stand age and litter type. Trends in the data also suggest that topography is an important factor,  
3814 but it was not quantified.

3815

## 3816 **Stream Temperature**

3817

3818 Bladon et al., 2016

3819

3820 Bladon, K.D., Cook, N.A., Light, J.T., Segura, C., 2016. A catchment-scale assessment of stream  
3821 temperature response to contemporary forest harvesting in the Oregon Coast Range. *Forest  
3822 Ecology and Management* 379, 153–164. <http://dx.doi.org/10.1016/j.foreco.2016.08.021>

3823

3824 The purpose of this study was to compare the effects of contemporary riparian forest harvest  
3825 treatments under the Oregon Forest Practices Act (15 m riparian management area with a  
3826 minimum of  $\sim 3.7 \text{ m}^2$  conifer basal area retained for every 300 m length of stream) with historical  
3827 riparian forest harvest practices (no maintenance of streamside vegetation) on stream  
3828 temperatures. This study took place in the Siuslaw National Forest in the Oregon Coast Range  
3829 as part of the Alsea Watershed Study Revisited. Historical records of stream temperatures were  
3830 sourced from the original Alsea Watershed Study that monitored stream temperature changes  
3831 from 1958-1973, before and after streamside timber harvesting in 1966. Stream temperature data

3832 was collected for contemporary forest practices over a 6-year period (3 years pre- and 3 years  
3833 post-harvest; 2006-2012). Data for the contemporary harvest was also compared with stream  
3834 temperature changes in unharvested reference streams to support a Before-After-Control Impact  
3835 (BACI) design. Stream temperature thermistors were installed, and data was taken at 30-minute  
3836 intervals at three sections of both the harvested (2 within harvest boundary and 1 downstream)  
3837 and reference sites. Mean canopy closure, as measured with a densiometer, along the stream  
3838 channel in the harvested portion of Needle Branch was reduced from ~96% in the pre-harvest  
3839 period to ~89% in the post-harvest period. Comparatively, mean canopy closure along the stream  
3840 channel in the reference sites were ~92% in the pre-harvest period and 91% in the post-harvest  
3841 period. Data was analyzed to assess whether there were changes in the 7-day moving mean of  
3842 daily maximum stream temperature, mean daily stream temperature, and diel stream temperature  
3843 following harvest. The results showed no significant changes in any of the three parameters  
3844 measured following contemporary forest harvesting practices when analyzed across all  
3845 catchments for all summer months (July to September). When the mean 7-day moving maximum  
3846 temperature was constrained to the summer period between July 15 – August 15 across all sites  
3847 there was a significant increase in stream temperatures in the harvested sites by  $0.6 \pm 0.2^{\circ}\text{C}$   
3848 following harvest. However, when the data was arranged for individual pair-wise comparisons  
3849 with the unharvested sites, and intrinsic annual and site variability was accounted for, the  
3850 increases in stream temperature (ranging from  $0.3 \pm 0.3^{\circ}\text{C}$  to  $0.8 \pm 0.3^{\circ}\text{C}$ ) were not significant at  
3851 any site. The only comparison made in the study to the original Alsea Watershed study was with  
3852 the single day maximum stream temperatures for pre- and post-harvest. The contemporary  
3853 practices showed a change of single day maximum stream temperatures from  $15.7^{\circ}\text{C}$  to  $14.7^{\circ}\text{C}$   
3854 (a reduction) from pre- to post-harvest. In contrast, the historical stream temperature data showed  
3855 an increase in single day maximum stream temperatures from  $13.9^{\circ}\text{C}$  (pre-harvest) to as much  
3856 as  $29.4^{\circ}\text{C}$  (2-years post-harvest). The authors caution that while these results support the  
3857 conclusion that contemporary forest practices in Oregon are sufficient in maintaining stream  
3858 temperatures after riparian forest harvest, and much more efficient than historical practices; these  
3859 results should not be generalized to areas outside of coastal Oregon. The authors caution that the  
3860 streams in this study have potential for a muted stream temperature response following harvest  
3861 relative to other regions because of the (1) north-south stream orientation, which would  
3862 maximize RMA effectiveness (2) steep catchment and channel slopes that can increase stream  
3863 velocity and hyporheic exchange, (3) potential increases in groundwater contributions after  
3864 harvest.

3865

## 3866 **Stream temperature**

3867

3868 Bladon et al., 2018

3869

3870 Bladon, K.D., Segura, C., Cook, N.A., Bywater-Reyes, S., Reiter, M., 2018. A multicatchment  
3871 analysis of headwater and downstream temperature effects from contemporary forest harvesting.  
3872 *Hydrological Processes* 32, 293–304. <https://doi.org/10.1002/hyp.11415>

3873

3874 The purpose of this study was to (1) examine the effects of contemporary forest harvesting  
3875 practices on headwater stream temperature, (2) determine if increased temperatures from  
3876 harvesting was detectable in downstream fish-bearing streams, and (3) examine the relative role  
3877 of geology and forest management on influencing the differential stream temperature responses  
3878 in both headwater and downstream reaches. This study took place at three paired watershed  
3879 studies, of which two (Alesa, Trask) were located in the Oregon coast range, and one (Hinkle)  
3880 was located in the western Cascades of Oregon. This study featured pre- and post-harvest  
3881 measurements, as well as measurements within and downstream from harvested and reference  
3882 sites. Buffer widths at harvested sites varied but averaged 20 m on either side of streams.  
3883 Statistical models were generated which analyzed whether (a) the 7-day moving average of daily  
3884 maximum stream temperature (7daymax) changed between pre- and post-harvest sites, and (b)  
3885 whether post-harvest changes in 7daymax were detectable downstream. A regression analysis  
3886 was also performed to assess the relative relationship between catchment lithology and percent  
3887 catchment harvested on temperature at all sites. Statistical models were generated for each  
3888 harvest site and reference pair. The pre-harvest relationship in stream temperatures for paired  
3889 sites were used to create predicted changes in stream temperatures post-harvest. The post-harvest  
3890 stream temperatures were then compared to the predicted values and the 95% prediction  
3891 intervals. If post-harvest values of the 7daymax were outside the prediction interval the authors  
3892 referred to these observations as statistical “exceedances”. Results showed that the 7daymax  
3893 exceeded the predictive interval at 7 of the 8 harvested headwater sites (within the harvested  
3894 boundary) when analyzed across all harvest years. The exceedances were largest in the first year  
3895 after harvest but diminished in the second and third year at two treatment sites. However, at one  
3896 site, the elevated 7daymax continued for three years post-harvest. In 4 of the 7 harvested sites  
3897 with exceedances, the exceedances were recorded between 22 and 100% of the time. Smaller  
3898 increases in stream temperatures were detected in the other 3 streams with exceedances, the  
3899 exceedances occurred < 15% of the time. There was no evidence of elevated stream temperatures  
3900 beyond the predicted intervals in any of the downstream sites following harvesting. The  
3901 magnitude of change in stream temperature and transmission of warmer water downstream were  
3902 a function of percentage of catchment harvested and the underlying geology. Although, these  
3903 relationships were scale dependent. At the upstream, harvested sites there was a strong  
3904 relationship between stream temperature increases and catchment lithologies, but no statistically  
3905 significant relationship between stream temperature changes and percent of catchment harvested.  
3906 Sites downstream from harvested areas showed a strong relationship with the interaction of  
3907 percentage of catchment harvested and the underlying lithologies. The greatest temperature  
3908 increases at downstream sites were in areas with a higher percentage of catchment harvested and  
3909 were underlain by more resistant lithologies. There was no evidence for increases in stream  
3910 temperatures in catchments with a high percentage of harvest that were underlain by permeable  
3911 geology. The authors suggest that this relationship may be due to the buffering effect of increases

3912 in summer low flows and greater groundwater or hyporheic exchange. They conclude that the  
3913 variability of rock permeability and the relative contribution of groundwater during summer  
3914 months, and their effect on stream temperatures following harvest should be investigated further.

3915

## 3916 **Wood Loading**

3917

3918 Burton et al., 2016

3919

3920 Burton, J.I., Olson, D.H., Puettmann, K.J., 2016. Effects of riparian buffer width on wood  
3921 loading in headwater streams after repeated forest thinning. *Forest Ecology and Management*  
3922 372, 247–257. <https://doi.org/10.1016/j.foreco.2016.03.053>

3923

3924 The purpose of this study was to examine the relationship between in-stream wood loading and  
3925 riparian buffer width in thinned stands in conjunction with several stand, site, and stream  
3926 variables. This study is a part of a larger density management study which covered 6 sites along  
3927 the coastal and western Cascade Range of Oregon. The sites used for this study were dominated  
3928 by Douglas-fir and ranged in age from 30-70 years old. Two consecutive thinning treatments  
3929 took place on a portion of each site, while the other portions were designated as an unthinned  
3930 control. Treated sites featured one of four buffer width prescriptions: (1) ~ 70-m buffer  
3931 representative of one site potential tree, (2) ~15-m buffer, (3) a 6-m buffer representative of trees  
3932 immediately adjacent to the stream. Wood surveys were carried out at four times during the  
3933 study: (1) prior to the first thinning, (2) five years after the first thinning, (3) 9-13 years after the  
3934 first thinning and just prior to the second thinning, and (4) one year after the second thinning. At  
3935 each site, the first thinning was to 200 trees per ha (tph), the second thinning (~10 years later)  
3936 was to ~85 tph, alongside an unthinned reference stand ~400 tph. Spatial and geomorphic  
3937 characterization were measured using a combination of field and geospatial data. Hierarchical  
3938 linear mixed models were developed with repeated measures using a multi-step process to  
3939 examine relationships between large wood volume in headwater streams over time and in-stream  
3940 wood characteristics (decay stage, zone), buffer width, time since thinning, and reach and  
3941 geomorphology (drainage basin area, width:depth ratio, gradient). Wood volume was found to  
3942 increase exponentially with drainage basin area; for every 1-ha increase in drainage basin area,  
3943 wood volume increased by 0.63%. Slightly higher volumes of wood were found in sites with a  
3944 narrow 6-m buffer, as compared with the 15-m and 70-m buffer sites in the beginning 5 years  
3945 after the first harvest and maintained through year 1 of the second harvest (end of study). The  
3946 authors attributed this difference to a higher likelihood of logging debris and/or windthrow but  
3947 was not analyzed. Low volumes of wood from stands in the stem-exclusion phase were found to  
3948 contribute to overall in-stream wood. The results showed that between 82-85% of the wood with  
3949 discernable sources (90% for wood in early stages of decay; 45% of wood in late stages of  
3950 decay) came from within 15 m of the stream, and the relative contribution of wood to streams

3951 declined rapidly with increasing distance. The authors hypothesize that this finding in  
3952 conjunction with their results, which show a positive relationship between basin area and wood  
3953 volume suggests a greater role for other large wood recruitment processes such as creep,  
3954 landslides, and debris flow.

3955

## 3956 **Sediment**

3957

3958 Bywater-Reyes et al., 2018

3959

3960 Bywater-Reyes, S., Bladon, K.D., Segura, C., 2018. Relative Influence of Landscape Variables  
3961 and Discharge on Suspended Sediment Yields in Temperate Mountain Catchments. *Water*  
3962 *Resources Research* 54, 5126–5142. 10.1029/2017WR021728

3963

3964 The purpose of this paper was to improve our ability to predict suspended sediment yields by  
3965 quantifying how sediment yields vary with catchment lithography and physiography, discharge,  
3966 and disturbance history. This study took place at the HJ. Andrews Experimental Site in the  
3967 Western Cascade Range of Oregon. The questions this paper sought to answer were (1) What is  
3968 the relative association between discharge and catchment setting (i.e., lithology and  
3969 physiography) and suspended sediment yields over an ~60-year period (2) Is there an  
3970 association between historical forest management activities (i.e., forest harvesting and road  
3971 building) or extreme hydrologic events and the spatial and temporal trends in suspended  
3972 sediment yield Data was collected from 10 catchments, 8 within the Lookout Creek Watershed, 1  
3973 just below the Lookout Creek Watershed, and 1 that drains to the adjacent Blue River. The data  
3974 set spanned a 60-year period from 1955-2015 Methods for determining suspended sediment  
3975 concentration involved using either vertically integrated storm-based grab samples, or discharge-  
3976 proportional composite samples where composite samples were collected every three weeks at  
3977 the outlet of each catchment. A linear mixed effects model (log transformed to meet the  
3978 normality assumption) was used to predict annual sediment yield. In this model, site was treated  
3979 as a random effect while discharge and physiographic variables were treated as fixed variables.  
3980 This allowed for the evaluation of the relationships between sediment yield and physiographic  
3981 features (slope, elevation, roughness, and index of sediment connectivity) while accounting for  
3982 site. To account for the effect of disturbance history a variable was added to the model when the  
3983 watershed had a history of management or natural disturbances. If the models for the disturbed  
3984 watersheds significantly underpredicted the sediment discharge, the timing of the sudden  
3985 increases were further examined to assess whether it correlated with a disturbance event. Last,  
3986 the authors considered changes in stage derived from comparing measured historic stage values  
3987 to those predicted from current rating curves. Changes in stage were interpreted as a relative bed-  
3988 elevation change resulting from changes in scour and deposition of material likely moved as  
3989 bedload. The results of this study show that sediment yield varied greatly across space and time

3990 with the lowest annual yield occurring in 2001 ( $\sim 0.2$  t/km<sup>2</sup>) at one catchment, and the highest  
3991 annual yield ( $\sim 953$  t/km<sup>2</sup>) occurring in 1969 at another catchment. Annual suspended sediment  
3992 yield was most strongly correlated with the standard deviation of watershed slope ( $r = 0.72$ ), Only  
3993 moderately correlated with slope ( $r = 0.32$ ), and with drainage area ( $r = 0.38$ ). Standard deviation  
3994 of slope was also strongly correlated with TPI (a surface roughness index), and standard  
3995 deviation of index of connectivity. When considering disturbance, the largest magnitude changes  
3996 in bed-elevation (I.e., sediment movement), were after floods with a  $\geq 30$ -year return interval.  
3997 The authors conclude that variability in watershed slope was the best predictor of annual  
3998 suspended sediment yield relative to other physiographic variables. The authors report that the  
3999 variability in watershed slope combined with cumulative annual discharge explained 67% of the  
4000 variation in annual sediment yield across the 60-year data set. The results, however, show that  
4001 annual sediment yields also moderately correlated with many other physiographic variables and  
4002 caution that the strong relationship with watershed slope variability is likely a proxy for many  
4003 processes, encompassing multiple catchment characteristics. For example, the strong relationship  
4004 between watershed slope standard deviation and surface roughness. For the relationships  
4005 between disturbance and sediment yield the authors conclude that the few anomalous years of  
4006 high sediment yield occurred in watersheds with high slope variability and within a decade of  
4007 forest management and a large flood event. The authors further caution that the high variability  
4008 of sediment yield over space and time indicate that the factors tested in this study should be  
4009 tested more broadly to investigate their utility to forest managers.

4010

4011

## 4012 **LW, Wildfire**

4013

4014 Chen et al., 2005

4015

4016 Chen, X., Wei, X., Scherer, R., 2005. Influence of wildfire and harvest on biomass, carbon pool,  
4017 and decomposition of large woody debris in forested streams of southern interior British  
4018 Columbia. *Forest Ecology and Management* 208, 101–114. doi:10.1016/j.foreco.2004.11.018

4019

4020 The purpose of this study was to compare the components of in-stream LW features between  
4021 wildfire and forest harvesting disturbances. This study focuses particularly on the change in  
4022 biomass and carbon pool among LW under different disturbances. This study was located in the  
4023 central Okanagan Valley, Kelowna, British Columbia. A total of 19 forest streams, first and  
4024 second order, within the study area were divided into four categories based on disturbance  
4025 history of the adjacent upland forest and included: (1) riparian forest harvested 10 years ago; (2)  
4026 riparian forest harvested 30 years ago; (3) riparian forest burnt  $\sim 40$  years ago; and (4)  
4027 undisturbed old-growth riparian forests that had a mean forest age of 163 years.. All harvested  
4028 streams were clear-cut to the stream edge. New trees had established on these sites within 1-3

4029 years of harvest (planted or natural growth) and resulted in lodgepole pine being the dominant  
4030 species. The wildfire streams included those that had been burnt ~40 years ago with no post-fire  
4031 harvest or salvage logging. In stream LW was recorded for analysis if it had a minimum diameter  
4032 of 10 cm and length of 1.0 m and were situated within the bankfull width. LW biomass was  
4033 determined through the conversion of wood density and wood volume. LW was also categorized  
4034 by decay class (3 classes), species, orientation submergence, and distance from the beginning of  
4035 the study reach. Sampling took place during the period between July and October 2003 along a  
4036 150 m study reach for each stream. An analysis of variance was used to determine the  
4037 relationships between the chosen variables. When significant differences were found, the data  
4038 was further analyzed with the data was fitted with a linear regression model to obtain  
4039 correlations between the three variables (volume, biomass, and carbon). Results from this study  
4040 show that on average the riparian sites disturbed by wildfire had the highest biomass, volume,  
4041 and carbon content for individual LW pieces, followed by the 10-year harvest, then the old-  
4042 growth forest; the 30-year harvest had the lowest of all streams for all parameters. Mean LW  
4043 biomass of each individual piece of wood was significantly higher in sites which had been  
4044 burned than in harvested sites. Biomass values were, on average, 31 kg in the wildfire sites,  
4045 compared to 21 kg and 19 kg for sites harvested 10 years ago and 30 years ago, respectively. The  
4046 volume of individual pieces in wildfire sites was significantly higher than in old-growth sites,  
4047 and nearly significantly higher than in sites harvested 30 years ago. No statistical significance  
4048 was found comparing piece volume in wildfire sites to sites harvested 10 years ago. The average  
4049 carbon content of individual pieces of wood was also highest in the wildfire sites but the  
4050 differences were not significant. The authors present data that the LW found in the wildfire and  
4051 30-year harvest sites was mostly in the third decay class (most decayed), with less than 1% of  
4052 LW in the class 1 decay class. Statistical significance was not discussed in the results for  
4053 differences in decay class. The authors conclude that streams adjacent to wildfire disturbed and  
4054 recently harvested (10-years post-harvest) forests contained significantly higher LW individual  
4055 pieces and total volume than old-growth and 30-year post-harvest sites. Further because biomass,  
4056 volume, and carbon were significantly higher in the 10-year post harvest sites, but there was no  
4057 difference in the 30-year post-harvest sites and the old-growth sites; the authors speculate that  
4058 harvest can increase the abundance of LW in the short-term from leaving harvest residues but  
4059 reduces the abundance of LW over the long-term (~30 years post) due to a lack of recruitment  
4060 from the young forests, and loss of in-stream LW from decomposition. The three main takeaways  
4061 presented by the authors for this paper were (1) LWD input in old growth forested streams was  
4062 relatively stable, (2) timber harvesting activities would cause a short-term increase of LWD  
4063 stocks and might greatly reduce LWD loadings over a long-term, and (3) wildfire disturbance  
4064 would delay LWD recruitment because not all burnt trees would fall in the stream immediately  
4065 after the wildfire.

4066

4067

4068 **LW**

4069

4070 Chen et al., 2006

4071

4072 Chen, X., Wei, X., Scherer, R., Luider, C., Darlington, W., 2006. A watershed scale assessment of  
4073 in-stream large woody debris patterns in the southern interior of British Columbia. *Forest*  
4074 *Ecology and Management* 229, 50–62. <https://doi.org/10.1016/j.foreco.2006.03.010>

4075

4076 The purpose of this study was to (1) determine the spatial distribution and variation of LW  
4077 characteristics (size, amount, volume, mass, orientation, position) within different order streams  
4078 of forested watersheds; (2) to examine the relationship between LW characteristics and stream  
4079 features through channel networks; and (3) to estimate the total density, volume and mass of LW  
4080 at the watershed scale using a combination of field surveys and GIS data. This study took place  
4081 at three different watersheds located in the south-central interior of British Columbia near  
4082 Kelowna. A total of 35 study reaches with stream orders ranging from first- through fifth-order  
4083 were selected to measure spatial distribution and variability of LW characteristics. Data collected  
4084 for each reach was binned into 4 stream size categories (I = first order; II = second to third order;  
4085 III = third to fourth order; IV = fourth to fifth order). Study sites were selected based on the  
4086 following criteria. (1) the streams were in areas of intact mature riparian forests (>80 years); (2)  
4087 the stream side forests were not disturbed by human activities, such as harvesting, road building;  
4088 (3) the streams were not salvaged. Therefore, the results from this study provide a baseline of  
4089 LWD characteristics in intact mature riparian forests in the southern interior of British Columbia.  
4090 LW in this study is defined as having a diameter of > 0.1 m and a length > 1.0 m. LW  
4091 characteristics (decay class, orientation, position within channel, distance from downstream end  
4092 of channel) were recorded for any piece of LW that was within or above the bankfull width of the  
4093 channel. Watershed features and the distribution of stream orders were derived from remotely  
4094 sensed data. Mean values of LW density, volume, and biomass were compared between stream  
4095 size classes with an analysis of variance (ANOVA). Results from this study show that LW size,  
4096 volume, and biomass generally increased with increasing stream size. For example, the mean  
4097 LWD diameter in stream size I (16.4 cm) was lower than that in stream size III (20.6 cm) and  
4098 size IV (20.5 cm), respectively. Mean LW length also increases with stream size from 2.3 m in  
4099 size I, 2.9 m in size II, 3.1 m in size III, and 3.9 m in size IV. Stream IV had the highest mean  
4100 volume (0.18 m<sup>3</sup>), significantly higher than stream size I (0.06 m<sup>3</sup>). LW volume was also  
4101 significantly lower than in stream sizes II, and III. LW density (pieces per 100 m<sup>2</sup> of stream  
4102 area), however, decreased as stream size increased. For example, LW density (defined as piece  
4103 numbers per 100 m<sup>2</sup>) numbers were 19, 17, 12, and 4 for stream size I, II, III, and IV  
4104 respectively. Increases in channel bankfull width ( $R^2 = 0.52$ ) and stream area ( $R^2 = 0.58$ ) was  
4105 found to be strongly inversely correlated with LW density. Taken together, this study shows that  
4106 spatial variation and distribution of LW characteristics vary as a function of stream size. From  
4107 their results the authors conclude that in small sized streams, LW exhibit high density (number of  
4108 pieces per 100 m<sup>2</sup>), low volume and biomass per unit area of stream. While in large sized  
4109 streams, LW number, volume and biomass per unit of stream area are low but mean individual  
4110 LW size was high.



4111

4112 **Stream Temperature Response to Harvesting**

4113

4114 Gravelle & Link, 2007

4115

4116 Gravelle, J.A., Link, T., 2007. Influence of Timber Harvesting on Headwater Peak Stream  
4117 Temperatures in a Northern Idaho Watershed. *Forest Science* 53, 189–205.

4118

4119 The purpose of this study was to examine the effects of clearcutting and partial cutting on  
4120 summer peak water temperatures in downstream fish-bearing streams, and to measure direct  
4121 harvesting impacts on peak water temperature within headwater catchments. This study took  
4122 place at the Mica Creek Experimental Watershed in Northern Idaho. Three headwater drainages  
4123 were used to assess harvesting impacts on stream temperatures: (1) Watershed 1 which had 50%  
4124 of the drainage area clearcut in 2001; (2) Watershed 2 which was thinned to a 50% target shade  
4125 removal in Fall 2001; (3) and an unimpacted control. Riparian buffers were applied adjacent to  
4126 the streams under the Idaho Forest Practices Act. This means, for fish-bearing streams the  
4127 riparian management area must be at least 75 ft (22.9 m) wide on each side of the ordinary high-  
4128 water mark (definable bank). Harvesting is still permitted, but there is a restriction where 75% of  
4129 existing shade must be left. There are also leave tree requirements, which is a target number of  
4130 trees per 1,000 linear feet (305 m), depending on stream width. For non-fish-bearing streams  
4131 there is a 30 ft (9.1 m) equipment exclusion zone on each side of the ordinary high-water mark  
4132 (definable bank). There are no shade requirements and no leave tree requirements, but skidding  
4133 logs in or through streams is prohibited. Stream temperature data and canopy cover percentage  
4134 data were collected at multiple sites within and downstream of treatment areas between 1992-  
4135 2005. However, for the non-fish-bearing, headwater sites pre-treatment data was only collected  
4136 one season prior to treatment. Temperature data was summarized as maximum daily temperature  
4137 and was analyzed using simple linear regression to estimate changes in stream temperature  
4138 following harvest during the summer months (July 1 – September 1). Results from this study  
4139 show that there is no strong evidence of a posttreatment increase in stream temperature at long-  
4140 term downstream sampling points for each harvest treatment. In general, the downstream sites  
4141 showed a cooling effect between  $-0.2$  and  $-0.3^{\circ}\text{C}$ . The estimated cooling effect could not be  
4142 attributed to any cause (e.g., increase in water yield), but the authors conclude that there was no  
4143 post-harvest increase in peak summer temperatures at the downstream sites. For streams  
4144 immediately adjacent to the clearcut treatment (headwater streams) a significant increase in  
4145 temperature was detected at 2 sites ranging between  $0.4$  and  $1.9^{\circ}\text{C}$ , while a marginally  
4146 significant decrease in temperature was detected at the third site ( $-0.1^{\circ}\text{C}$ ,  $p = 0.06$ ). At the sites  
4147 located immediately adjacent to partial cuts, results showed mixed results with decreases in  
4148 temperature ( $-0.1^{\circ}\text{C}$ ; non-significant) at one site and significant but minimal changes at another  
4149 site ( $0.0$ - $3.0^{\circ}\text{C}$ ) across the individual post-harvest years. Overall, there were minimal to no  
4150 changes in stream peak temperatures following treatment in the partial-cut riparian areas. The

4151 authors go on to point out that headwater stream temperatures were highly variable, and that the  
4152 shade value of understory vegetation may be an important factor contributing to results.

4153

4154 **SED**

4155

4156 Bywater-Reyes et al., 2017

4157

4158 Bywater-Reyes, S., Segura, C., Bladon, K.D., 2017. Geology and geomorphology control  
4159 suspended sediment yield and modulate increases following timber harvest in temperate  
4160 headwater streams. *Journal of Hydrology* 548, 754–769.  
4161 <https://doi.org/10.1016/j.jhydrol.2017.03.048>

4162

4163 The purpose of this study was to assess the influence of natural controls (basin lithology and  
4164 physiography) and forest management on suspended sediment yields in temperate headwater  
4165 catchments. The study sought to achieve three objectives: (1) Quantify how suspended sediment  
4166 yield varies by catchment setting in forested headwater catchments, (2) Determine whether  
4167 contemporary forest management practices impact annual suspended sediment yield (SSY) in  
4168 forested headwater catchments (3) Determine whether there are natural catchment settings that  
4169 result in different levels of vulnerability or resilience to increases in suspended sediment yield  
4170 associated with disturbances (e.g., harvest activities). This study analyzed 6 years of data from  
4171 the Trask River Watershed in Northeastern Oregon and included data from harvested and  
4172 unharvested sub-catchments underlain by heterogeneous lithologies. Baseline SSY data collection  
4173 began in water year 2010 and continued through water year 2015, with road upgrades (July–  
4174 August 2011) and harvest (May–November 2012) occurring in the middle of the study period.  
4175 Generalized least square candidate models quantifying the parameters from each site were used  
4176 to test differences in the relationship between suspended sediment yield and catchment setting.  
4177 Results from this study indicate that site lithology was a first order control over SSY with SSY  
4178 varying by an order of magnitude across lithologies observed. Specifically, SSY was greater in  
4179 catchments underlain by Siletz Volcanics ( $r = 0.6$ ), the Trask River Formation ( $r = 0.4$ ), and  
4180 landslide deposits ( $r = 0.9$ ) and displayed an exponential relationship when plotted against  
4181 percent watershed area underlain by these lithologies, combined. In contrast, the site effect had a  
4182 strong negative correlation with percent area underlain by diabase ( $r = 0.7$ ), with the lowest SSY  
4183 associated with 100% diabase independent of whether or not earthflow terrain was present.  
4184 Following timber harvest (water year 2013), increases in SSY occurred in all harvested  
4185 catchments. The SSY in both PH4 (clearcut with buffers) and GC3 (clearcut without buffers)  
4186 declined to pre-harvest levels by water year 2014. Interestingly, the SSY in UM2 (clearcut  
4187 without buffers) increased annually throughout the post-harvest period, ultimately resulting in  
4188 the highest SSY of all catchments during the final two years of the study after producing the  
4189 lowest SSY in the pre-harvest period. Catchment physiographic variables (hypsoetry, slope,

4190 standardized topographic position index (SD TPI), and sediment connectivity (IC)) appeared to  
4191 be good indicators of the underlying lithology of each site. Principle component analysis  
4192 constructed from physiographic variables separated sites underlain by resistant diabase from  
4193 those underlain by mixed lithologies along the PC1 axis. While sites along the second axis (PC2)  
4194 were separated by relative values of earthflow terrain (high proportion vs. Little to none). Sites  
4195 with low SSY and underlain by more resistant lithologies were also resistant to harvest-related  
4196 increases in SSY. The authors conclude that sites underlain with a friable lithology (e.g.,  
4197 sedimentary formations) had SSYs an order of magnitude higher, on average, following harvest  
4198 than those on more resistant lithologies (intrusive rocks). In general, sites with higher SSY also  
4199 had 1) lower mean elevation and slope, 2) greater landscape roughness, and 3) lower sediment  
4200 connectivity (potential for sediment transport based on physiography). The authors suggest that  
4201 their research be undertaken in different regions with different disturbance types to broadly apply  
4202 their findings.

4203

## 4204 **Plant Communities**

4205

4206 D'Souza et al., 2012

4207

4208 D'Souza, L.E., Six, L.J., Bakker, J.D., Bilby, R.E., 2012. Spatial and temporal patterns of plant  
4209 communities near small mountain streams in managed forests. *Can. J. For. Res.* 42, 260–271.  
4210 <https://doi.org/10.1139/x11-17>

4211

4212 The purpose of this study was to examine spatial and temporal patterns in plant communities  
4213 along fish-bearing streams in western Washington. The focus of this study is on areas which were  
4214 harvested to the streambank within the last 100 years. The study took place in the western  
4215 Cascade Mountains of Washington. Sites were randomly selected using a geographic information  
4216 system. Stands that had been impacted by road development were excluded. Stands were  
4217 stratified into a chronosequence of age classes: young (31-51 years), mature (52-70 years), old  
4218 (>100 years). Due to availability, the sample sizes included 11 young stands, 10 mature stands,  
4219 but only 4 old stands. Vegetation characteristics were captured in each stand using 0.16 ha plots  
4220 located 30 m from stand edges to limit the influence of adjacent stands. Transects perpendicular  
4221 to the stream were used 10 m apart and extended 80 m upslope. Vegetation and physical features  
4222 along each transect were sampled using a series of subplots at 10 m intervals from the channel.  
4223 The authors found little variation in riparian landform type and or canopy cover and were not  
4224 included in the analysis for their effect on vegetation. Plant communities were examined  
4225 spatially as a function of distance to stream and temporally by using the chronosequence of stand  
4226 ages. Three distinct plant communities were observed in the shrub and herb layer (riparian: 0-9  
4227 m; transitional: 10-29 m; and upslope: 30-80 m) and their composition differed significantly  
4228 between communities. A total of 12 species were identified as indicators of these communities.

4229 For the shrub layer, community composition differed between old stands and young and mature  
4230 stands. In the herb layer, community composition differed between all age classes. The results  
4231 from this study suggest that plant communities along small fish-bearing streams have distinct  
4232 changes in community with distance to stream, but also reflect successional status in nearby  
4233 forests. The authors conclude by suggesting increased research in understanding the effects of  
4234 forest management on streamside vegetation.

4235

### 4236 **LW Residence Time**

4237

4238 Hyatt & Naiman, 2001

4239

4240 Hyatt, T.L., Naiman, R.J., 2001. The Residence Time of Large Woody Debris in the Queets  
4241 River, Washington, Usa. *Ecological Applications* 11, 191–202. [https://doi.org/10.1890/1051-  
4242 0761\(2001\)011\[0191:TRTOLW\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2001)011[0191:TRTOLW]2.0.CO;2)

4243

4244 The purpose of this study was to determine the depletion rate of LW by examining differences in  
4245 size and species composition in the Queets River compared to the adjacent forest. This study  
4246 took place in the Queets River Watershed located on the west slope of the Olympic Mountains in  
4247 Washington. Field sampling was carried out at 25 transects and four different sites. Increment  
4248 cores from in-stream LW were cross-dated against cores from riparian conifers to estimate the  
4249 time which LW was recruited into the channel. LW pieces which were in a heightened state of  
4250 decay were dated using carbon-dating techniques. the most common tree species (> 30 cm  
4251 diameter) in the riparian zone is red alder, followed by Sitka spruce and western hemlock,  
4252 whereas the most common species of LWD (> 30 cm diameter) is Sitka spruce, followed by red  
4253 alder and western hemlock. Each of the hardwood species is better represented among standing  
4254 trees than among LWD, and each of the conifers are better represented as LW than among trees  
4255 in the riparian zone. The depletion curve developed in the results was based only on conifer LW  
4256 because hardwood LW was either too small or too young to provide accurate estimates of  
4257 residence time in the stream. Based on the depletion curve developed for all available LW  
4258 showed that wood typically disappears from the active channel within the first 50 years, while  
4259 some pieces may remain for several hundred years. By cross-referencing the LW depletion  
4260 curves with field notes the authors suggest that the longer residence time, beyond 50 years, was  
4261 dependent on more than one process such as burial. Decay class was not an accurate predictor of  
4262 LW age. Also, Dependent vegetation on or around LWD was a poor and often misleading  
4263 indicator of residence time. Many LWD pieces that had 1–5 year old vegetation growing on  
4264 or around them were discovered to have died and presumably recruited to the channel 20 years  
4265 previous. The authors conclude that LW originating from hardwoods is depleted faster than  
4266 conifers. Considering the depletion rate curve, the authors speculate that the majority of LW is  
4267 transported out of the system within 50 years, while pieces of LW that are buried or jammed in

4268 the river floodplain may remain for hundreds of years. Overall, ~80% of LW residing in the  
4269 active channel were living within 50 years of the study. The authors explain there are several  
4270 caveats to the depletion curve created for this study (1) the depletion constant was developed for  
4271 a large, mostly alluvial river and should probably not be applied to smaller streams (mean  
4272 bankfull width at study transects on the Queets is 165 m and the range is 51–398 m; mean key  
4273 LWD length is 23.4 m, and the range is 5.3–69.0 m). Also, from the data the authors infer that  
4274 alluvial channel trap wood from upstream, and constrained channels export LWD downstream,  
4275 so it is not to be expected that the LWD resident in a channel was recruited from the riparian  
4276 zone in that reach. In general, the authors conclude that for this study the depletion curve shows  
4277 that the half-life of LW is ~20 years and thus all resident LW will be exported, buried, or broken  
4278 down within 3-5 decades. Also, hardwood LW will be depleted from the channel more rapidly  
4279 than conifers.

4280

## 4281 **Litter Input**

4282

4283 Hart et al., 2013

4284

4285 Hart, S.K., Hibbs, D.E., Perakis, S.S., 2013. Riparian litter inputs to streams in the central  
4286 Oregon Coast Range. *Freshwater Science* 32, 343–358. <https://doi.org/10.1899/12-074.1>

4287

4288 The purpose of this study was to understand how riparian vegetation composition, understory  
4289 density, and topography affect the quantity and quality of litter input to streams throughout the  
4290 annual cycle. This study took place within 5 contiguous watersheds located in the central Coast  
4291 Range of Oregon. At each of the study sites uniform areas along a  $\leq 300$  m stream reach, 3 plots  
4292 were delineated on 1 side of the stream, each 8x 25 m along the stream. Three treatments were  
4293 applied: (1) a no cut or fence control; (2) cut and remove a 5 x 8 m section adjacent to stream  
4294 plants < 10 cm DBH and >12 cm height every 2 months; and (3) 5 m fence extending  
4295 underground and parallel to the stream to block litter moving downslope from reaching stream.  
4296 Vertical and lateral litter traps were installed at each site and collected monthly between August  
4297 2003-August 2004. Variation of riparian vegetation and woody debris characteristics were  
4298 analyzed with a 3-way ANOVA using overstory, treatments, and sections and their interactions.  
4299 Two-way ANOVA with repeated measures was used to compare seasonal and monthly control  
4300 and treatment inputs for different overstory and litter types. 1-way ANOVA was used to test for  
4301 differences in nutrient concentration flux between overstory type. Results from this study show  
4302 that deciduous forests dominated by red alder delivered significantly greater vertical and lateral  
4303 inputs to stream than did coniferous forests dominated by Douglas-fir. Deciduous-site vertical  
4304 litter input (mean, 95% CI; 504 g m<sup>-1</sup> y<sup>-1</sup>, 446.6–561.9) exceeded that from coniferous sites  
4305 (394 g m<sup>-1</sup> y<sup>-1</sup>, 336.4–451.7) by 110 g/m<sup>2</sup> (28.6–191.6) over the full year. Annual lateral inputs  
4306 at deciduous sites (109 g m<sup>-1</sup> y<sup>-1</sup>, 75.6–143.3) were 46 g/m (1.2– 94.5) more than at coniferous

4307 sites (63 g m<sup>-1</sup> y<sup>-1</sup>, 28.9– 96.6). Lateral inputs calculated for a 3-m-wide stream accounted for  
4308 9.6% (5.4–12.5) of total annual inputs at coniferous sites and 12.7% (10.2–14.5) of total inputs at  
4309 deciduous sites. Composition of litter also differed significantly by overstory type. Annual lateral  
4310 inputs at coniferous sites were dominated by deciduous leaves (,33%), twigs (,23%), and leftover  
4311 (,18%) litter types, whereas annual lateral inputs at deciduous sites were deciduous leaves (,61%)  
4312 and leftover (,15%) litter types. Leftover litter types were defined as those that were too small or  
4313 decayed to identify, bark, moss, or lichens. Vertical litter inputs at deciduous sites were  
4314 dominated by deciduous leaves (,65%) and deciduous-other (,15%) litter types. While deciduous  
4315 leaves (,33%), coniferous needles (,24%), and twigs (,21%) composed the annual vertical litter  
4316 inputs at coniferous sites. The strongest deciduous inputs to streams occurred in November.  
4317 Annual lateral litter input increased with slope at deciduous sites ( $R^2 = 0.4073$ ,  $p = 0.0771$ ), but  
4318 showed no strong relationship at coniferous sites ( $R^2 = 0.1863$ ,  $p = 0.2855$ ). Total nitrogen flux  
4319 to streams at deciduous sites was twice as much as recorded at coniferous sites. However, there  
4320 was seasonal effect where the N fluxes in deciduous sites was only higher in autumn. The  
4321 authors of this study conclude by suggesting management in riparian areas consider utilizing  
4322 deciduous species such as red alder for greater total N input to aquatic and terrestrial ecosystems  
4323 along with the increased shade and large wood provided by coniferous species.

4324

## 4325 **Effect of Contemporary Management on Nutrient Concentration and Cycling**

4326

4327 Gravelle et al., 2009

4328

4329 Gravelle, J.A., Ice, G., Link, T.E., Cook, D.L., 2009. Nutrient concentration dynamics in an  
4330 inland Pacific Northwest watershed before and after timber harvest. *Forest Ecology and*  
4331 *Management* 257, 1663–1675. <https://doi.org/10.1016/j.foreco.2009.01.017>

4332

4333 The purpose of this study was to assess the effects of contemporary forest harvesting practices on  
4334 nutrient cycling and concentrations. This study took place at the Mica Creek Experimental  
4335 Watershed in Northern Idaho. Seven steel Parshall flumes were installed at select locations  
4336 within the watershed to assess the effects of clearcut to stream and partial cut (50% shade  
4337 retention) harvesting practices. All harvesting was conducted in compliance with the Idaho  
4338 Forest Practices Act. Within fish-bearing streams (Class I) Harvesting is permitted, but 75% of  
4339 existing shade must be retained. There are also leave tree requirements for a target number of  
4340 trees per 1000 linear feet (305 m), depending on stream width. In Mica Creek, this was roughly  
4341 200 trees in the 3–12 in. (8–30 cm) diameter class per 305 m of the riparian management zone  
4342 (RMZ). Along non-fish-bearing streams (Class II) the RMZ is 30 feet (9.1 m) of equipment  
4343 exclusion zone on each side of the ordinary high-water mark (definable bank); skidding logs in  
4344 or through streams is prohibited. There are no shade requirements and no requirements to leave  
4345 merchantable trees. Two-sided riparian buffers were left on all Class I streams during harvest

4346 operations. Timber was removed from both sides of the Class II streams. In the post-harvest and  
4347 post-burn conditions, Class II streams in clearcut treatments had only a small amount of green  
4348 tree retention within the riparian zone, while in partial cut treatments equal amounts of canopy  
4349 cover (approximately 50%) were removed from both sides of the stream. This study followed the  
4350 BACI design and featured a pre-treatment measurement phase (1992-1997), a post-road  
4351 construction phase (1997-2001), and a post-harvest phase (2001-2006). A students t-test was  
4352 used to analyze the data between the observed and predicted values of post-treatment sites for  
4353 several nitrogen and phosphorus compound concentrations (Kjeldahl nitrogen (TKN), nitrate +  
4354 nitrite (NO<sub>3</sub> + NO<sub>2</sub>), TP, total ammonia nitrogen (TAN) consisting of unionized (NH<sub>3</sub>) and  
4355 ionized (NH<sub>4</sub><sup>+</sup>) ammonia, and unfiltered orthophosphate (OP) samples). Results from the post-  
4356 road construction period showed no significant changes in concentrations of any nutrients  
4357 analyzed. Results from this study show statistically significant increases in NO<sub>3</sub> and NO<sub>2</sub>  
4358 concentrations following clearcut and partial harvest cuts in headwater streams. Increases at the  
4359 clearcut treatment site were greatest, where mean monthly concentrations increased from 0.06  
4360 mg-N L<sup>-1</sup> during the calibration and post-road periods to 0.35 mg-N L<sup>-1</sup>. There was also an  
4361 observable seasonal effect on NO<sub>3</sub> + NO<sub>2</sub> concentrations with the peak concentration of 0.89  
4362 mg-N L<sup>-1</sup> occurred at F1 in April 2004, with mean monthly concentrations of 0.43 mg-N L<sup>-1</sup>  
4363 and 0.59 mg-N L<sup>-1</sup> in water years (October–September) 2004 and 2005, respectively. Similar  
4364 results were also observed at sites further downstream although changes were smaller which, the  
4365 authors point out this may be due to in-stream uptake and/or dilution. No significant changes of  
4366 in-stream concentration of any other nutrient recorded were found between time periods and  
4367 treatments except for one downstream site that showed a small increase in orthophosphate by  
4368 0.01 mg P L<sup>-1</sup>. In general, the results of this study show that forest management influences in-  
4369 stream NO<sub>3</sub> + NO<sub>2</sub> immediately adjacent to treatment and downstream of treatment. The authors  
4370 conclude by suggesting future research in understanding variability in nutrient concentrations  
4371 and cycling as affected by seasons and storm runoff events.

4372

### 4373 **Organic Matter Inputs**

4374

4375 Kiffney & Richardson, 2010

4376

4377 Kiffney, P.M., Richardson, J.S., 2010. Organic matter inputs into headwater streams of  
4378 southwestern British Columbia as a function of riparian reserves and time since harvesting.  
4379 *Forest Ecology and Management* 260, 1931–1942. <https://doi.org/10.1016/j.foreco.2010.08.016>

4380

4381 The purpose of this paper was to assess how differences in riparian buffer width and timing since  
4382 harvest affect terrestrial particulate organic matter flux into streams. The focus of this paper was  
4383 on 1st and 2nd order headwater streams located approximately 45 km east of Vancouver in  
4384 British Columbia, Canada. Sites were measured over an 8-year period and included clear-cut

4385 (n=3), 10-m buffered reserve (n=3), 30-m buffered reserve (n=2), and uncut control (n=2)  
4386 treatments. For streams receiving a 10 or 30-m reserve, there was no logging on either side of the  
4387 stream within these reserves. Study reaches were approximately 200m long. Vertical litter inputs  
4388 were collected monthly and at approximately 6–8-week intervals during each season for years  
4389 1,2,6,7, and 8 years after harvest. Litter was separated into broadleaf deciduous, twig, needles,  
4390 and other (seeds, cones, and moss) categories following collection and subsequently dried and  
4391 weighed using a microbalance. A mixed-model analysis of covariance was used for Fall data  
4392 with riparian treatment as a fixed effect and year as a covariate. Secondly, ordinary least  
4393 squares regression was used to quantify the functional relationship between reserve width and  
4394 litter flux within each year. Results show riparian treatments having significant effects on the  
4395 quantity and composition of litter input into streams. Inputs consisting of needles and twigs were  
4396 significantly lower while deciduous inputs were higher in clearcuts compared to other  
4397 treatments. Differences in litter flux relative to riparian treatment persisted through year 7, while  
4398 a positive trend between reserve width and litter flux remained through year 8. For example, one-  
4399 year post-treatment, needle inputs were 56x higher during the Fall into control and buffered  
4400 treatments than into the clearcut. Needle inputs remained 6x higher in the buffer and control sites  
4401 through year 7, and 3-6x higher in year 8 than in the clearcut sites. Twig inputs into the control  
4402 and buffered sites were ~25x higher than in the clearcut sites in the first year after treatment.  
4403 There was no significant difference in treatment for deciduous litter but a trend of increasing  
4404 deciduous litter input in the clear cut was observed in the data. For example, one-year post-  
4405 treatment deciduous litter was lowest in the clearcut, but by year 8 deciduous litter was highest in  
4406 the clearcut sites relative to control and buffered sites. The linear relationship between reserve  
4407 width and litter inputs was strongest in the first year after treatment, explaining ~57% of the  
4408 variation, but the relationship could only explain ~17% of the variation in litter input by buffer  
4409 width by year 8 (i.e., the relationship degraded over time). The authors interpret these results as  
4410 evidence that riparian reserves showed a similar litter flux to streams when compared to uncut  
4411 controls. They also conclude that litter flux from riparian plants to streams, was affected by  
4412 riparian reserve width, time since logging, and potentially channel geomorphology.

4413

#### 4414 **In-stream Wood Loads**

4415

4416 Jackson & Wohl, 2015

4417

4418 Jackson, K.J., Wohl, E., 2015. Instream wood loads in montane forest streams of the Colorado  
4419 Front Range, USA. *Geomorphology* 234, 161–170.  
4420 <http://dx.doi.org/10.1016/j.geomorph.2015.01.022>

4421

4422 The purpose of this study was to examine in-stream wood loads and geomorphic effects between  
4423 stands of different ages and stands with different disturbance histories The first objective of this



4424 study was to determine whether instream wood and geomorphic effects differ significantly  
4425 among old-growth, younger, healthy, and beetle-infested forest stands. The second objective of  
4426 this study was to determine whether instream wood loads correlate with valley and channel  
4427 characteristics. The authors hypothesized that streams in old-growth montane forests have (1)  
4428 significantly larger in stream and floodplain wood loads than those in younger stands, (2) greater  
4429 frequency of volume of jams than those in younger forests, and (3) more wood created  
4430 geomorphic effects. They also hypothesized that instream wood loads in healthy montane forests  
4431 are significantly smaller than in beetle-infested forests. Last, they hypothesized that instream  
4432 wood load correlates with lateral valley confinement, with unconfined valleys having the greatest  
4433 in-stream and total wood loads. This study took place within the Arapaho and Roosevelt National  
4434 Forests in Colorado. Sediment storage, channel geometry, in-stream wood load, and forest stand  
4435 characteristics were measured along 33 pool-riffle or plane-bed stream reaches (10 located in  
4436 old-growth (> 200 years); 23 located in younger forests (age range not reported)). LW  
4437 characteristics were recorded for all in-stream wood  $\geq 10$  cm diameter and  $\geq 1$  m in length. Pair-  
4438 wise t-test or Kruskal-Wallis tests were used to check for significant differences in wood load,  
4439 logjam volume, and logjam frequencies. To test for significant differences in wood created  
4440 geomorphic effects a principal component analysis was used. Results indicated that channel  
4441 wood load (OG =  $304.4 \pm 161.1$ ; Y =  $197.8 \pm 245.5$  m<sup>3</sup> /ha), floodplain wood load (OG =  $109.4$   
4442  $\pm 80$ ; Y =  $47.1 \pm 52.8$  m<sup>3</sup> /ha), and total wood load (OG =  $154.7 \pm 64.1$ ; Y =  $87.8 \pm 100.6$  m<sup>3</sup>  
4443 /ha) per 100 m length of stream and per unit surface area were significantly larger in streams of  
4444 old-growth forests than in young forests. Streams in old-growth forests also had significantly  
4445 more wood in jams, and more total wood jams per unit length of channel than in younger forests  
4446 (jam wood volume: OG =  $7.10 \pm 6.9$  m<sup>3</sup>; Y =  $1.71 \pm 2.81$  m<sup>3</sup>). When standardized to stream  
4447 gradient, old-growth streams had significantly greater pool volume and significantly greater  
4448 sediment volume than younger stands. No significant difference was detected in in-stream wood  
4449 loads between healthy and beetle-infested stands. Although wood load in streams draining from  
4450 pine beetle infested forests did not differ significantly from healthy forests, best subset regression  
4451 (following principal component analysis) indicated that elevation, stand age, and pine beetle  
4452 infestation were the best predictors of wood load in channels and on floodplains. The authors  
4453 speculate that beetle infestation is affecting in-stream wood, but perhaps not enough time has  
4454 passed since the infestation for the affected trees to fall into the stream. Time since beetle-  
4455 infestation was not reported.

4456

## 4457 **LW Recruitment**

4458

4459 May & Gresswell, 2003

4460

4461 May, C.L., Gresswell, R.E., 2003. Large wood recruitment and redistribution in headwater  
4462 streams in the southern Oregon Coast Range, U.S.A. *Can. J. For. Res.* 33, 1352–1362.

4463 <https://doi.org/10.1139/x03-023>

4464

4465 The purpose of this study was to understand the relative influence of processes that recruit and  
4466 redistribute wood into channels and to understand how these processes vary spatially. Specific  
4467 research questions included the following:(i) Do processes that deliver and redistribute wood  
4468 differ in small colluvial channels compared with larger alluvial channels? (ii) Do proximal and  
4469 distal controls on wood delivery differ for colluvial and alluvial channels? (iii) How do input and  
4470 redistribution processes influence the functional role of wood in the channel? The focus of this  
4471 research is specifically on differences between small colluvial channels and large alluvial  
4472 channels in the southern Oregon Coast Range. All downed wood exceeding 20 cm mean  
4473 diameter and 2 m in length, and in contact with the bank-full channel were measured in three  
4474 second order and one third-order stream. Large wood was categorized based on the various  
4475 mechanisms delivering it to the stream channel. Categories included (i) direct delivery from local  
4476 hillslopes and riparian areas, (ii) fluvial redistribution, (iii) debris flow transported, or (iv) an  
4477 unidentified source. Results from this study show that stream size and topographic position  
4478 strongly influence processes that recruit and redistribute wood in channels. Processes of slope  
4479 instability were shown to be important conveyors of wood from upland forests to small colluvial  
4480 channels. In the larger alluvial channels, windthrow was found to be the dominant recruitment  
4481 process from adjacent riparian area. Results showed that Wood derived from local hillslopes and  
4482 riparian areas accounted for the majority of pieces (63%) in small colluvial channels. The larger  
4483 alluvial channel received wood from a greater variety of sources, including recruitment from  
4484 local hillslopes and riparian areas (36%), fluvial redistribution (9%), and debris flow transported  
4485 wood (33%). However, because pieces recruited from local sources (hillslope and riparian area)  
4486 were larger, these sources of wood had a disproportionately large contribution to volume of wood  
4487 in the stream. For example, wood recruited from the local hillslopes and riparian areas accounted  
4488 for 36% of wood pieces in the alluvial stream, which accounted for 74% of the total volume of  
4489 wood. Slope instability and windthrow were the dominant mechanisms for wood recruitment into  
4490 small colluvial channels. Windthrow was the dominant recruitment mechanism for wood  
4491 recruitment into larger alluvial channels. Distributions of the source distance of wood pieces  
4492 were significantly different between colluvial and alluvial channels. In colluvial streams, 80% of  
4493 total wood and 80% of total wood volume recruited originated from trees rooted within 50 m of  
4494 the channel. In the alluvial channel, 80% of the pieces of wood and 50% of the total volume  
4495 originated from trees which came from 30 m of the channel. The primary function of wood in  
4496 smaller colluvial channels was sediment storage (40%) and small wood storage (20%). The  
4497 primary function of wood in larger alluvial channels is bank scour (26%), stream bed scour  
4498 (26%), and sediment storage (14%). Recruitment and redistribution processes were shown to  
4499 affect the location of the piece relative to the channel/flow direction, thus influencing its  
4500 functional role. The authors conclude that wood recruited from local sources is variable by  
4501 position in the stream network because of differences in recruitment processes, degree of  
4502 hillslope constriction, and slope steepness.

4503

4504 **Sediment**

4505

4506 Macdonald et al., 2003a

4507

4508 Macdonald, J. S., Beaudry, P. G., MacIsaac, E. A., & Herunter, H. E. (2003). The effects of forest  
4509 harvesting and best management practices on streamflow and suspended sediment concentrations  
4510 during snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada.  
4511 *Canadian Journal of Forest Research*, 33(8), 1397-1407. <https://doi.org/10.1139/x03-110>

4512

4513 (BACI, only single year pre-harvest)

4514

4515 This study investigates the changes in suspended sediment concentration and stream discharge  
4516 during freshet (spring snowmelt) at two harvest intensities relative to each other and an  
4517 unharvested control watershed, pre- and post-harvest. The design included three small sub-  
4518 boreal, first order, forest streams (<1.5 m width) in the central interior of British Columbia  
4519 (Baptiste watershed). Both treatment streams received a 55% harvest treatment; one (low-  
4520 retention) removed all merchantable timber >15 cm DBH for pine and > 20 cm DBH for spruce  
4521 within 20 m of the stream; the other treatment (high-retention) removed all merchantable timber  
4522 > 30 cm within 20 m of the stream; and an un-harvested control. Data for stream flow and total  
4523 suspended sediments (TSS) was collected using Parshall flumes downstream from the treatment  
4524 and control sites for one-year pre- and four-years post-harvest during snowmelt periods.

4525 Regression analysis was used to analyze relationships between treatment and control reaches pre-  
4526 and post-treatment to estimate and compare predicted changes in TSS. The results showed an  
4527 increase in freshet discharge for both treatments above predicted values for the entirety of the  
4528 study. During the year prior to treatment, TSS relationships of both treatment watersheds during  
4529 freshet closely matched those of the control. Immediately following harvest TSS concentrations  
4530 increased above predicted values for both treatment streams. Increased TSS persisted for two-  
4531 years post-harvest in the high-retention treatment, and for 3-years in the low-retention. The  
4532 authors speculate that the treatment areas may have accumulated more snow (e.g., more exposed  
4533 area below canopy) than in the control reaches leading to the increase in discharge. This study  
4534 shows evidence that harvest intensity (low vs. high retention) is proportional to the increase in  
4535 stream discharge, TSS, and recovery time to pre-harvest levels.

4536

4537 **LW**

4538

4539 Fox & Bolton, 2007

4540

4541 Fox, M., & Bolton, S. (2007). A regional and geomorphic reference for quantities and volumes of  
4542 instream wood in unmanaged forested basins of Washington State. *North American Journal of*  
4543 *Fisheries Management*, 27(1), 342-359. <https://doi.org/10.1577/M05-024.1>

4544

4545 This study uses in-stream LW values from 150 stream segments located in unmanaged  
4546 watersheds, across all of Washington State, to investigate the relationships between  
4547 geomorphology, forest zone, and disturbance regimes with LW recruitment. The purpose of this  
4548 study was to create a base-line value of central tendency for in-stream LW values in “natural”  
4549 streams for which salmonids are theoretically adapted. The authors define natural and  
4550 unmanaged as streams that (1) had no part of the basin upstream of the survey site ever logged  
4551 using forest practices common after European settlement and (2) the basin upstream of the  
4552 survey site contains no roads or human modifications to the landscape that could affect the  
4553 hydrology, slope stability, or other natural processes of wood recruitment and transport in  
4554 streams. Sites were stratified to capture the variations in forest types, channel morphologies, and  
4555 hydrological origins. The authors used descriptive statistics to establish and evaluate correlations  
4556 between wood loading and watershed characteristics to reveal the highest valued variables  
4557 influencing wood loading. Following this analysis, the variables with the highest mechanistic  
4558 values in determining wood loading were evaluated and compared using simulation modeling.  
4559 Results showed that in-stream wood volume increased with drainage area and as streams became  
4560 less confined. However, bank full width (BFW) was a significantly better predictor of wood  
4561 parameters than basin size. There was observational evidence that alluvial channels contained  
4562 more wood volume on average than bedrock channels. However, due to limits in sample size  
4563 following stratification, statistical analysis could not be completed. Sample sizes for isolating  
4564 gradient and confinement were also too small to apply statistical analyses. Fire was found to  
4565 influence in-stream wood quantities and volumes west of the Cascade crest; In-stream wood  
4566 volume increased with adjacent riparian timber age as determined by the last stand replacing fire.  
4567 Other disturbances such as debris flow, snow avalanche, and flooding were too few in frequency  
4568 in the study area to be analyzed statistically. From these results the authors developed thresholds  
4569 for expected “key piece volume ( $m^3$ ) (pieces with independent stability) of wood for three BFW  
4570 classes (20-30 m, >30 – 50 m, > 50 m width) per 100 m stream length for streams with BFW  
4571 greater than 20 m. From percentile distributions the authors recommend minimum volumes,  
4572 defined by the 25<sup>th</sup> percentiles, of approximately 9.7  $m^3$  for the 20- to 30-m BFW class, 10.5  $m^3$   
4573 for the 30- to 50-m BFW class, and 10.7  $m^3$  for channels greater than 50 m BFW per 100 m  
4574 length of stream. The results of this study suggest that BFW is the single greatest predictor of in-  
4575 stream wood quantity and volume relative to other predictor variables. However, this result  
4576 comes with the caveat that other processes and geomorphologies (e.g., channel bed form,  
4577 gradient, confinement) are also important in the mechanisms for wood recruitment, modeling in  
4578 this study showed too much inconsistency with these predictor variables to draw strong  
4579 conclusions. Further the authors warn that these values for reference conditions are only  
4580 applicable to streams with bank-full widths between 1 and 100 m, gradients between 0.1% and  
4581 47%, elevations between 91 and 1,906 m, drainage areas between 0.4 and 325  $km^2$ , glacial and  
4582 rain- or snow-dominated origins, forest types common to the Pacific Northwest.

4583

4584 **LW and sediment**

4585

4586 Gomi et al., 2001

4587

4588 Gomi, T., Sidle, R. C., Bryant, M. D., & Woodsmith, R. D. (2001). The characteristics of woody  
4589 debris and sediment distribution in headwater streams, southeastern Alaska. *Canadian Journal of*  
4590 *Forest Research*, 31(8), 1386-1399. <https://doi.org/10.1139/x01-070>

4591

4592 This study investigated different riparian conditions related to harvest and disturbance  
4593 (landslides), their influence on woody debris and sediment distributions, and their related  
4594 functions in headwater streams. This study examined the effects of recent and past timber  
4595 harvests on woody debris abundance and distribution, landslides and debris flow on woody  
4596 debris abundance and sediment accumulations, and the function of in-stream woody debris on  
4597 sediment storage. The researchers examined 15 steep headwater streams in the Maybeso  
4598 Experimental Forest and Harris River basin in the Tongass National Forest, Prince of Wales  
4599 Island, southeastern Alaska. Treatments of headwater streams included five management or  
4600 disturbance regimes: old growth (OG), recent clear-cut (CC; 3 years), young growth conifer  
4601 forest (YC; 37 years after clear-cut), young growth alder (YA; 30 years after clear-cut), and  
4602 recent landslide and debris flow channels (LS). Three headwater streams were sampled for each  
4603 of the 5 treatments, 15 streams total. Analysis of covariance (ANCOVA) was used to compare  
4604 LW quantity and distribution, and sediment quantity and distribution, across plots nested within  
4605 each treatment site. Results showed in-channel numbers of LW pieces were significantly higher  
4606 in YC and CC sites when compared to OG, YA, and LS sites. The number of LW pieces was  
4607 highest in YC streams even though logging concluded 3 decades prior to sampling. No  
4608 significant differences in LW volume were found among OG, CC, and YC streams. However,  
4609 LW volume per 100 m of stream length in YC was twice that in OG. The total volume of LW per  
4610 100 m associated with CC channels was half that in OG channels. However, the majority of the  
4611 LW volume in OG systems was outside of the bank-full area. When the data was stratified by  
4612 channels that experienced landslides (LS and YA), the number of LW pieces among OG, YA, and  
4613 LS was not statistically significant. However, the in-channel volumes of LW in LS and YA  
4614 channels were significantly lower than in OG sites because individual LW pieces in the OG sites  
4615 were relatively larger than in the LS and YA sites. There was high variability among sites in the  
4616 amount of sediment stored within streams. The authors conclude that timber harvesting and  
4617 related landslides and debris flows affect the distribution and accumulation of LW and related  
4618 sediment accumulation in headwater streams. These effects are summarized as (i) inputs of  
4619 logging slash and unmerchantable logs significantly increase the abundance of in-channel woody  
4620 debris; (ii) in the absence of landslides or debris flows, these woody materials remain in the  
4621 channel 50–100 years after logging; (iii) relatively smaller woody debris initially stores  
4622 sediment; (iv) when landslides and debris flows occur 3–15 years after logging because of

4623 intensive rain and weakening of root strength (Sidle et al. 1985), woody debris is evacuated from  
4624 headwater streams and deposited in downstream reaches; (v) although less woody debris remains  
4625 in the scour zone, woody debris pieces and jams contribute to sediment storage in both the scour  
4626 and deposition zones of landslide and debris flow channels; (vi) red alder stands actively  
4627 recolonize riparian zones of headwater streams for 20–50 years after mass movement and recruit  
4628 woody debris and organic materials, which in turn provide sediment storage sites; and (vii)  
4629 subsequent sediment movement after landslides and debris flows are affected by residual woody  
4630 debris and newly introduced debris.

4631

## 4632 **LW and sediment**

4633

4634 Johnson et al., 2000 (removed from focal list)

4635

4636 Johnson, S. L., Swanson, F. J., Grant, G. E., & Wondzell, S. M. (2000). Riparian forest  
4637 disturbances by a mountain flood—the influence of floated wood. *Hydrological processes*,  
4638 14(16-17), 3031-3050. [https://doi.org/10.1002/1099-1085\(200011/12\)14:16/17<3031::AID-  
4639 HYP133>3.0.CO;2-6](https://doi.org/10.1002/1099-1085(200011/12)14:16/17<3031::AID-HYP133>3.0.CO;2-6)

4640

4641 This study examined the differences in riparian forest responses to a 100-year flood event along  
4642 eight third- to fifth-order streams in the Cascade Mountain Range of Oregon. Disturbance  
4643 intensities were grouped into three categories: purely fluvial (high water flow only), fluvial with  
4644 uncongested wood transport, and fluvial with congested wood transport. Riparian forest  
4645 responses were heavily influenced by pre-flood forest structure and disturbance/harvest history,  
4646 especially the characteristics of LW presence within streams and along channels. The quantity  
4647 and severity of toppled trees (fully uprooted vs. partially uprooted) during the flood event was  
4648 proportional to the quantity and congestion of LW already present (i.e., higher volumes of LW  
4649 already present during the flood event increased the frequency of toppled trees and newly  
4650 deposited LW in streams). Further, stands that experienced higher frequencies of toppled trees  
4651 also showed higher frequencies and magnitudes of debris flow. The authors concluded that the  
4652 land use practices, and disturbance histories influenced the age and structure of the riparian  
4653 forests, but also the availability of the agents of disturbance (presence of LW) during the 100-  
4654 year flood event. This paper is a good discussion of how pre-disturbance structure affects the  
4655 response of riparian forests to disturbances (in this case, flood), however, there is no statistical  
4656 analysis discussed in the methods. This is purely descriptive science that involves an intensive  
4657 survey of before and after riparian forest structures.

4658

## 4659 **Sediment**

4660

4661 Yang et al., 2022 (removed from focal list)

4662

4663 Yang, Y., Safeeq, M., Wagenbrenner, J. W., Asefaw Berhe, A., & Hart, S. C. (2022). Impacts of  
4664 climate and forest management on suspended sediment source and transport in montane  
4665 headwater catchments. *Hydrological Processes*, 36(9), e14684.  
4666 <https://doi.org/10.1002/hyp.14684>

4667

4668 This paper investigates the changes in annual hysteresis patterns for in-stream suspended  
4669 sediment in 10 headwater streams at 2 sites, Providence Creek (rain-snow-dominated,  
4670 transitional), and Kings River Experimental Watershed (snow-dominated). Aside from  
4671 precipitation pattern differences in the two catchments, the researchers also compared differences  
4672 in hysteresis patterns for forested riparian control, burn-only, thin-only, and thin-and-burn  
4673 combined areas. The differences in the proportion of clockwise-loop hysteresis patterns for  
4674 suspended sediments in the warmer rain-snow-transition sites compared to the colder snow-  
4675 dominated sites suggests that warming temperatures may cause the snow-dominated basins to  
4676 receive sediment from extended source areas and for longer periods if they transition to rain  
4677 dominated catchments. The results found no discernable difference in hysteresis loops between  
4678 the control, burn-only, thin-only, and thin-and-burn combined areas. Further, there seemed to be  
4679 little change in the hysteresis loops during drought, average, and excessively wet years. The  
4680 authors speculate that local conditions will be more important in understanding the impacts of  
4681 climate change than changes in precipitation patterns or average annual temperatures alone.  
4682 Mainly, there is evidence that if snow-dominated watersheds become warm enough to transition  
4683 to rain-dominated, there is potential for disruption to sediment discharge frequency, rates, and  
4684 source distance. The indiscernible difference in hysteresis loops for the different treatments also  
4685 suggests that management practices imposed to ameliorate these changes may not be completely  
4686 effective.

4687

## 4688 **Nutrients**

4689

4690 Vanderbilt et al., 2003

4691

4692 Vanderbilt, K. L., Lajtha, K., & Swanson, F. J. (2003). Biogeochemistry of unpolluted forested  
4693 watersheds in the Oregon Cascades: temporal patterns of precipitation and stream nitrogen  
4694 fluxes. *Biogeochemistry*, 62(1), 87-117. DOI:10.1023/A:1021171016945

4695

4696 This study uses long-term datasets (ranging from 20-30 years) from six watersheds in the H.J.  
4697 Andrews Experimental Watershed (HJA) in the west-central Cascade Mountains of Oregon to  
4698 investigate patterns in dissolved organic nitrogen (DON) and dissolved inorganic nitrogen (DIN)  
4699 export with watershed hydrology. The objectives of this study were to 1) characterize long-term  
4700 patterns of N dynamics in precipitation and stream water at the HJA, 2) analyze relationships  
4701 between annual output of N solutes and annual stream discharge, 3) analyze relationships  
4702 between seasonal stream water N solute concentrations and precipitation and stream discharge,  
4703 and 4) compare results with those from other forested watersheds. Precipitation data were  
4704 collected at three-week intervals from 10/1/1968 until 5/24/1988 and at one-week intervals  
4705 thereafter. Stream chemistry samples were collected weekly for the entirety of the study. Stream  
4706 discharge was measured continuously throughout the study. The researchers used regression  
4707 analysis of annual N inputs and outputs with annual precipitation and stream discharge to  
4708 analyze patterns. The results showed DON was the largest component of N input at the low-  
4709 elevation collector, followed by PON (particulate organic N), NO<sub>3</sub>-N, and NH<sub>4</sub>-N. At the high-  
4710 elevation collector, NO<sub>3</sub>-N input was higher than at low elevation and was the largest component  
4711 of N in bulk and wet-only inputs, followed by NH<sub>4</sub>-N, DON, and PON. For annual stream  
4712 outputs, DON was the largest fraction of annual N output, followed by PON, NH<sub>4</sub>-N and then  
4713 NO<sub>3</sub>-N. Total annual discharge was a positive predictor of annual DON export in all watersheds  
4714 with r<sup>2</sup> values ranging from 0.42 to 0.79. In contrast, significant relationships between total  
4715 annual discharge and annual export of NO<sub>3</sub>-N, NH<sub>4</sub>-N, and PON were not found in all  
4716 watersheds. No systematic long-term average seasonal trends were observed for NO<sub>3</sub>-N or PON  
4717 concentrations. Elevated concentrations of NH<sub>4</sub>-N occurred in spring and early summer in all  
4718 three watersheds, although they are not convincingly synchronous. DON concentrations  
4719 increased in the fall in every watershed. The increase in concentration began in July or August  
4720 with the earliest rain events, and peak DON concentrations occurred in October through  
4721 December before the peak in the hydrograph. DON concentrations then declined during the  
4722 winter months. The authors conclude that total annual stream discharge was a positive  
4723 predictor of DON output suggesting a relationship to precipitation. Also, DON had a consistent  
4724 seasonal concentration pattern. All other forms of N observed showed variability and  
4725 inconsistencies with annual and seasonal stream discharge. The authors speculate that different  
4726 factors may control organic vs. Inorganic N export. Also, DIN may be strongly influenced by  
4727 terrestrial or in-stream biotic controls, while DON is more strongly influenced by climate. Last,  
4728 the authors suggest that DON in streams may be recalcitrant, and largely unavailable to stream  
4729 organisms. The authors emphasize the importance of analyzing data from multiple watersheds in  
4730 a single climactic zone to make inferences about stream chemistry.

4731

### 4732 **Stream temperature**

4733

4734 Roon et al., 2021b

4735



4736 Roon, D. A., Dunham, J. B., & Torgersen, C. E. (2021). A riverscape approach reveals  
4737 downstream propagation of stream thermal responses to riparian thinning at multiple scales.  
4738 *Ecosphere*, 12(10), e03775. <https://doi.org/10.1002/ecs2.3775>

4739

4740 This study uses a riverscape approach to evaluate the effects of streamside forest thinning on  
4741 stream temperatures at multiple spatiotemporal scales. This study addresses the question of how  
4742 thinning second-growth riparian forests influences local and downstream temperatures at  
4743 watershed extents. This study attempts to answer this question by addressing four objectives: (1)  
4744 quantify pretreatment spatial and temporal variability in stream temperature conditions; (2)  
4745 evaluate local responses in stream temperature to riparian thinning; (3) assess the spatial extent  
4746 and temporal duration of downstream effects to local responses in temperature; and (4)  
4747 characterize local and downstream responses to thinning with a conceptual framework based on  
4748 waveforms. The researchers compared upstream, local, and downstream, stream temperature  
4749 fluctuations following different intensities of streamside forest thinning at 10 treatment reaches  
4750 across three watersheds in the redwood forests of northern California. Treatments varied by  
4751 landowners. In two watersheds thinning treatments were intended to reduce 50% of canopy  
4752 closure within the riparian zone along a 200 m reach on both sides of the active channel. This  
4753 treatment resulted in a reduction in effective shade over the stream between 19-30%. In the other  
4754 treatment watershed, thinning treatments reduced basal area by as much as 40% on both sides of  
4755 the active channel along a 100 m long reach. Reductions in effective shade over the stream in  
4756 these sites ranged from 4-5%. The analysis considered each reach both individually and  
4757 collectively to understand how site and treatment heterogeneity may affect thermal responses at  
4758 local and watershed extents. Temperature data were collected before, during, and after treatment  
4759 and in the thinned experimental reaches and in adjacent unthinned control reaches with digital  
4760 temperature sensors. Temperature data was collected for only 1-year pre-treatment and 1-year  
4761 post-treatment. For data analysis, semivariograms of summer degree days were used to  
4762 determine the presence of spatial autocorrelation. To control temporal variations in local and  
4763 downstream responses summer cumulative degree-days were plotted for pre- and post- treatment  
4764 temperatures and along a longitudinal gradient. A Lagrangian framework was used to track  
4765 changes in temperature through space and time. Results showed that increases in thermal  
4766 heterogeneity occurred in the treatment reaches, in the year following treatment (20° to 139°C),  
4767 compared to the pre-treatment year (66° to 112°C). Local changes in stream temperature were  
4768 dependent on thinning intensity, with higher levels of canopy cover reduction leading to higher  
4769 increases in local stream temperatures. In the reaches with higher reductions in shade (19-30%)  
4770 there was accumulation of 45° to 115°C additional degree days from pre- to post treatment years,  
4771 while the reaches with lower reductions in shade (4-5%) only accumulated 10° to 15°C  
4772 additional degree days. Travel distance of increased stream temperatures also appeared to be  
4773 dependent on thinning intensity. The lower shade reduction reaches had an increased temperature  
4774 effect downstream with travel distance of 75-150 m, while the high shade reduction sites had a  
4775 downstream travel distance of 300- ~1000 m. In the high shade reduction sites, treatment reaches  
4776 that were further apart (> 400 m) showed dissipation in increased stream temperatures  
4777 downstream, while in parts of the stream where treatments were <400 m apart, temperature

4778 increases did not always dissipate before entering another the next treatment reach. The analyses  
4779 with the conceptual framework based on waveforms showed there was no evidence of  
4780 cumulative watershed effects at the downstream extent. The authors conclude that their results  
4781 show evidence that riparian forest management impacts may extend beyond local stream  
4782 environments. Further, the authors propose that riparian forest management that uses a holistic  
4783 approach may be more effective in preserving some functions (e.g., shade).

4784

## 4785 **Sediment**

4786

4787 Wissmar et al., 2004

4788

4789 Wissmar, R.C., Beer, W.N. & Timm, R.K. (2004) Spatially explicit estimates of erosion-risk  
4790 indices and variable riparian buffer widths in watersheds. *Aquat. Sci.* 66, 446–455 . DOI:  
4791 10.1007/s00027-004-0714-9

4792

4793 The purpose of this study is to use management records, the spatial distribution, and the  
4794 variability of different landcover types that can contribute to unstable conditions to develop  
4795 erosion-risk indices and variable riparian buffer widths in watersheds of different drainages in  
4796 the State of Washington. The objectives of this study were to 1) define erosion risk indices based  
4797 on “different land cover types,” 2) evaluate erosion risk indices with sediment inputs into  
4798 streams, 3) use erosion risk categories to define locations of stream reaches that are susceptible  
4799 to different levels of erosion 4) use categories to identify distribution of channels requiring  
4800 variable width buffers for protection 5) Test procedure by applying ground-truthed data from the  
4801 upper Cedar River drainage near Seattle, Washington. The land cover types used to assess risk  
4802 included unstable soils, immature forests, roads, critical slopes for land failure, and rain-on-snow  
4803 events. Based on available data, the researchers developed a map of these land cover features  
4804 with sediment input values to define erosion risk indices. The indices were used to categorize the  
4805 landscape into 6 levels of erosion risk. Results of the mapped erosion risk categories explained  
4806 65% of the variation associated with sediment inputs. The highest-risk areas contained a  
4807 combination of all landscape cover factor combinations (rain-on-snow zone, critical failure  
4808 slope, unstable soil, immature forests, and roaded areas). The lowest risk categories contained  
4809 only rain-on-snow zones, and critical failure slopes. Roaded areas and unstable soils were only  
4810 present in risk categories 3-6. This paper shows the importance of investigating multiple factors  
4811 when evaluating the controls on sediment discharge and stream inputs. Further, when factors  
4812 influencing erosion combine in an area, their effects are compounded.

4813

## 4814 **Nutrient and forest structure**

4815

4816 Devotta et al., 2021 (removed)

4817

4818 Devotta, D. A., Fraterrigo, J. M., Walsh, P. B., Lowe, S., Sewell, D. K., Schindler, D. E., & Hu,  
4819 F. S. (2021). Watershed *Alnus* cover alters N: P stoichiometry and intensifies P limitation in  
4820 subarctic streams. *Biogeochemistry*, 153(2), 155-176. DOI:10.1007/s10533-021-00776-w

4821

4822 This study investigates how coverage of alder species affects the aquatic N and P availability  
4823 across a natural alder coverage gradient in 26 streams of southwestern Alaska. Alder coverage in  
4824 the Alaskan streams was inversely related to elevation (i.e., lower coverage at higher elevations).  
4825 To identify the presence of alder as the N and p contributing factor, the researchers analyzed  
4826 resin lysimeter samples from select watershed soils supporting variable percent coverages of  
4827 alder. Soils supporting alders leached, on average, three times more N and two times more P than  
4828 soils not containing alders. The relationship between alder coverage and N and P values was not  
4829 linear. Still, the authors identified 30% alder coverage as a transitional threshold from low to  
4830 markedly higher soil N and p availability. The higher soil N and P resulted in higher dissolved N  
4831 in streams, but the higher soil P under alder coverage did not translate to higher stream P  
4832 availability. The authors speculate that soil chemistry or local soil biota may be immobilizing the  
4833 soil P from transport into the streams. This led to a high N:P ratio in the spring and summer  
4834 stream chemistry of reaches supporting >30% alder coverage. As climate change causes  
4835 increasing temperatures, alder may begin to expand its range into higher elevations. This, in turn,  
4836 may lead to increased N availability, but higher P limitations in high-elevation montane streams.

4837

### 4838 **Sediment and lithology**

4839

4840 Fratkin et al., 2020 (removed from focal, scope and results not relevant to review)

4841

4842 Fratkin, M. M., Segura, C., & Bywater-Reyes, S. (2020). The influence of lithology on channel  
4843 geometry and bed sediment organization in mountainous hillslope-coupled streams. *Earth  
4844 Surface Processes and Landforms*, 45(10), 2365-2379. <https://doi.org/10.1002/esp.4885>

4845

4846 This study compares the differences in channel form patterns, sediment flow, grain size, and  
4847 shear stress thresholds between two gravel-bed streams, one on basalt and one on sandstone  
4848 parent material in the Oregon Coast Range. Study sites were in a region where widespread  
4849 landslides and debris flows occurred in 1996. The researchers compared channel  
4850 geomorphologies (e.g., slope, valley width, channel geometry, etc.) to evaluate thresholds and

4851 channel bed adjustments since the 1996 events. The results showed similar sediment coarsening  
4852 patterns in the first several kilometers indicating hillslope influence, but downstream fining was  
4853 lithology dependent. The authors hypothesized threshold channel conditions in the basalt basin,  
4854 and non-threshold conditions in the sandstone basin with a tendency to expose bedrock, based on  
4855 the relative competencies (i.e., basalt = high-competency, sandstone = low-competency).  
4856 However, results showed evidence of threshold conditions for over 60% of the streams in both  
4857 basins. The authors inferred a cycle adjustment to correct the assumed sediment delivery from  
4858 the 1996 flood season. The authors speculate that the basalt basins would act as threshold  
4859 channels over longer time periods despite a higher debris flow frequency. This paper provides  
4860 some evidence that lithologies impose control on channel adjustments driven by different rock  
4861 competencies. This difference in rock competency ultimately controls the grain size fining rates  
4862 and bed load transport (sediment availability).

4863

#### 4864 **Nutrient and species composition**

4865

4866 Whigham et al., 2017 (removed from focal)

4867

4868 Whigham, D. F., Walker, C. M., Maurer, J., King, R. S., Hauser, W., Baird, S., ... & Neale, P. J.  
4869 (2017). Watershed influences the structure and function of riparian wetlands associated with  
4870 headwater streams—Kenai Peninsula, Alaska. *Science of the Total Environment*, 599, 124-134.  
4871 <https://doi.org/10.1016/j.scitotenv.2017.03.290>

4872

4873 This field study was designed to test the hypothesis that alder cover in watersheds influences the  
4874 structure and function of riparian wetlands adjacent to headwater streams. The researchers  
4875 compared biomass production, biomass distribution (aboveground vs. belowground),  
4876 decomposition rates, and chemical characteristics of interstitial groundwater, between watersheds  
4877 with and without alder coverage. Study sites were located on two headwater streams located in  
4878 the Kenai Peninsula in south-central Alaska. The results showed that aboveground biomass was  
4879 higher in watersheds with alder cover, but the largest differences were in the litter layer and the  
4880 belowground biomass. Watersheds without alder had significantly higher belowground root  
4881 biomass. The litter overhanging the stream was higher in N content at the alder sites than in the  
4882 no-alder sites. The quantity of litter overhanging the stream was higher in the no-alder sites.  
4883 Interstitial groundwater was significantly higher in dissolved N at the alder sites. The results of  
4884 this study show that species composition within the riparian area can have a considerable effect  
4885 on nutrient concentrations which consequently affect stream chemistry, biomass production,  
4886 vegetation structure, and decomposition rates.

4887

4888 **LW**

4889

4890 Wing & Skaugset, 2002

4891

4892 Wing, M. G., & Skaugset, A. (2002). Relationships of channel characteristics, land ownership,  
4893 and land use patterns to large woody debris in western Oregon streams. *Canadian Journal of*  
4894 *Fisheries and Aquatic Sciences*, 59(5), 796-807. <https://doi.org/10.1139/f02-052>

4895

4896 This study investigated the relationships of land use, land ownership, and channel and habitat  
4897 characteristics with LW quantity and volume in 3793 stream reaches in western Oregon State  
4898 (west of Cascade crest). This study analyzed an extensive spatial database of aquatic habitat  
4899 conditions created for western Oregon using stream habitat classification techniques and a  
4900 geographic information system (GIS). The overall objectives of this study were to identify the  
4901 database factors most strongly related to LWD abundance and to determine whether ownership  
4902 and land use patterns are related to LWD abundance. Regression tree analysis is an exploratory  
4903 regression analysis that allows for the inclusion of multiple explanatory variables. LW counts (by  
4904 piece, and by key pieces (logs at least 0.60 m in diameter and 10 m long)) and volume were used  
4905 as the response variables and explanatory variables included morphology of active channel  
4906 (hillslope, terrace, terrace hillslope, unconstrained), lithology (e.g., alluvium, basalt, etc.), Land  
4907 use and land cover (e.g., young timber, old timber, rural resident, agriculture, etc.), ownership  
4908 (private industrial (PI), private non-industrial (PNI), state, federal (BLM, USFS)), vegetation  
4909 type, and other channel characteristics. The analysis was run at the reach scale. Results showed  
4910 that the most important predictor for LW volume was land ownership with PNI split from all  
4911 other ownership types. Mean LW volumes in stream reaches with PNI ownership were 3.1 m<sup>3</sup>  
4912 while mean volume of LW in reaches in all other ownerships (PI, state, BLM, USFS) were 17.9  
4913 m<sup>3</sup>. However, this was likely because the PNI lands held a disproportionately higher percentage of  
4914 unforested lands compared to all other ownership types. When the ownership and land use  
4915 variables were removed, stream gradient became the most important explanatory variable for LW  
4916 volume. The split for stream gradient occurred for reaches with < 2.3% gradient averaged 5.8 m<sup>3</sup>  
4917 while higher gradient streams averaged 17.9 m<sup>3</sup> per reach. When ownership and land use were  
4918 included but non-forested lands were removed, stream gradient again was the most important  
4919 predictor with the split occurring for stream reaches with gradients less than 4.7% averaging 11.5  
4920 m<sup>3</sup>, which was less than half of the average found at higher gradient reaches (25.2 m<sup>3</sup>); in this  
4921 model the stream gradient split explained 11% of the variation observed of instream LW volume.  
4922 For LW pieces in forested stream reaches bankfull channel width was the most important  
4923 explanatory variable with the split occurring for streams channels less than 12.2 m wide. LW  
4924 pieces for streams <12.2 m wide averaged 11.1 LW pieces per reach while larger channels  
4925 averaged 4.9 pieces per reach; in this model the BFW split explained 7% of the variation in LW  
4926 pieces found in forested streams. For key LW pieces (logs at least 0.60 m in diameter and 10 m  
4927 long) in forested reaches, stream gradient was again the most important explanatory variable  
4928 with the split occurring at a slope of 4.9%. The streams with a gradient < 4.9% averaged 0.5 key  
4929 LW pieces per reach while streams with higher gradients averaged 0.9 key LW pieces per reach;

4930 in this model stream gradient explained 8% of the variation in key LW pieces found in streams.  
4931 For forested streams, lithology caused second, third or fourth level splits after stream gradient or  
4932 BFW. In three of these four splits, Mesozoic sedimentary and metamorphic geologies, located in  
4933 southern Oregon stream reaches, were grouped and split from basalt, cascade, and marine  
4934 sedimentary geologies. In stream reaches in Mesozoic sedimentary and metamorphic geologies,  
4935 the quantity of LWD was roughly half the amount found in other geologies. The only exception  
4936 to this grouping was for LW volume in larger stream reaches, where basalt and marine  
4937 sedimentary geologies were grouped separately from all other geologies in a fourth-level split  
4938 and contained more LW volume. The authors conclude that the geomorphic characteristics of  
4939 stream reaches, in particular stream gradient and bankfull width, in forested areas correlated best  
4940 with LW presence.

4941

4942

### 4943 **LW and plant communities**

4944

4945 Rot et al., 2000 (removed from focal list)

4946

4947 Rot, B. W., Naiman, R. J., & Bilby, R. E. (2000). Stream channel configuration, landform, and  
4948 riparian forest structure in the Cascade Mountains, Washington. *Canadian Journal of Fisheries  
4949 and Aquatic Sciences*, 57(4), 699-707. <https://doi.org/10.1139/f00-002>

4950

4951 This study investigates the hierarchical relationships between the “five key elements”, valley  
4952 constraint, riparian landform, riparian plant community, channel type, and channel configuration.  
4953 for 21 sites in mature old-growth riparian forests of the western Cascade Mountains in  
4954 Washington State. The objective of this article is to expand this perspective over several spatial  
4955 scales and the temporal life span of a conifer by examining how channel configuration interacts  
4956 with valley constraint, streamside landform, channel bedform, and successional processes within  
4957 the riparian forest. Stepwise regression was used to examine the relationship between physical  
4958 and biological characteristics and the individual elements of channel configuration. Channel  
4959 configuration is the channel elements at the habitat unit scale, including channel units (total  
4960 number of pool–riffle habitat units per 100 m of channel length), LW pieces (per 100 m of  
4961 channel length), LW volume (cubic meters per 100 m of channel length), pool spacing, percent  
4962 pools, and percent LW-formed pools. Results showed that significantly more total LW pieces  
4963 were found in forced pool–riffle channels than in the bedrock and plane-bed channels (Kruskal–  
4964 Wallis,  $p < 0.05$ ). Forced pool–riffle channels averaged 16.4 pieces per 100 m, bedrock 10.8  
4965 pieces, and plane-bed 10.1 pieces. The volume of LW (cubic meters per 100 m) followed a  
4966 similar trend. The percentage of deep pools ( $>0.5$  m) formed by LW increased with stand age ( $r^2$   
4967 = 0.36). LW diameters were significantly smaller for ages 55–220 than for ages 333–727  
4968 (Kruskal–Wallis,  $p = 0.01$ ). The authors conclude that scale is an important consideration for

4969 management of aquatic habitat. At the largest spatial scale, results showed valley constraint  
4970 significantly influenced off-channel habitat (plant communities associations and landform  
4971 categories) and in-stream LW volume within forced pool-riffle channels. At the smallest scale,  
4972 channel type (bedrock, plane-bed, and forced pool-riffle) was most closely related to LW  
4973 volume, density, and the number of LW-formed pools. The diameter of the in-channel LW  
4974 increased with riparian forest stand age. Streams adjacent to old-growth forests in-channel LW  
4975 diameter were equivalent to or greater than the average standing riparian tree diameter at all  
4976 sites. In younger stands, the relationship of in-stream LW diameter had a mixed relationship with  
4977 riparian tree average diameters. The authors speculate this may be due to many in-stream LW  
4978 pieces being relics from previous old-growth communities. In this area, four landform classes  
4979 differentiated the riparian communities (floodplain, low terrace, high terrace, slope). Most were  
4980 dominated by conifers, except the floodplain landforms, which supported a higher density of  
4981 deciduous species, but a higher basal area of conifer species. The results of this study provide  
4982 more evidence, similar to other studies, that channel geomorphology and valley constraint are  
4983 important predictors of LW abundance (quantity and volume) in streams. The novelty in this  
4984 study is how the riparian area landforms lead to different riparian plant communities, which  
4985 consequently affect the input of LW.

4986

## 4987 **Nutrients**

4988

4989 Yang et al., 2021

4990

4991 Yang, Y., Hart, S. C., McCorkle, E. P., Stacy, E. M., Barnes, M. E., Hunsaker, C. T., ... & Berhe,  
4992 A. A. (2021). Stream water chemistry in mixed-conifer headwater basins: role of water sources,  
4993 seasonality, watershed characteristics, and disturbances. *Ecosystems*, 24(8), 1853-1874.  
4994 DOI:10.1007/s10021-021-00620-0

4995

4996 This study investigated the effects of drought and forest thinning operations (independently and  
4997 combined) on water chemistry from multiple basin water sources (snowmelt, soil solution,  
4998 stream water) in the Mediterranean climate headwater basins of the Sierra National Forest. Data  
4999 on water chemistry was taken 2 years prior and 3 years following drought and thinning  
5000 operations in two watersheds, each with thinned and control stands. This data was analyzed to  
5001 answer 3 questions: 1. How does the chemistry of different water sources (that is, snowmelt, soil  
5002 solution at two depths, stream water) vary monthly and interannually prior to drought and  
5003 thinning? 2. How does drought alone and drought combined with thinning impact water  
5004 chemistry? 3. Can watershed characteristics predict stream water chemistry over contrasting  
5005 water years? The authors used general linear models to analyze differences in chemistry by water  
5006 source, repeated measures analysis of variance for effects of drought and thinning on water  
5007 chemistry, and linear regression to predict water chemistry based on watershed characteristics.

5008 Results showed that monthly concentrations of dissolved C and N varied among different water  
5009 sources prior to drought and thinning. For dissolved organic carbon (DOC) soil solution at 13 cm  
5010 depth (mean  $\pm$  SE of  $25.97 \pm 2.75 \text{ mg l}^{-1}$ , across months for 2 years) had higher monthly  
5011 concentrations than soil solution collected at 26 cm depth ( $16.93 \pm 1.55 \text{ mg l}^{-1}$ ). Snowmelt ( $9.67$   
5012  $\pm 0.89 \text{ mg l}^{-1}$ ) and stream water ( $5.33 \pm 0.52 \text{ mg l}^{-1}$ ) had the lowest concentrations. For total  
5013 dissolved Nitrogen (TDN) and dissolved organic nitrogen (DON), soil solution at 13 cm depth  
5014 ( $1.72 \pm 0.57$  and  $1.66 \pm 0.57 \text{ mg l}^{-1}$ , respectively), soil solution at 26 cm depth ( $0.94 \pm 0.32$  and  
5015  $0.92 \pm 0.32 \text{ mg l}^{-1}$ ), and snowmelt ( $0.94 \pm 0.17$  and  $0.73 \pm 0.18 \text{ mg l}^{-1}$ ) had higher  
5016 concentrations than stream water ( $0.11 \pm 0.02$  and  $0.08 \pm 0.01 \text{ mg l}^{-1}$ ). For dissolved inorganic  
5017 nitrogen (DIN), snowmelt ( $0.25 \pm 0.05 \text{ mg l}^{-1}$ ) had the highest concentration followed by the soil  
5018 solution at 13 cm depth ( $0.06 \pm 0.01 \text{ mg l}^{-1}$ ). Soil solution at 26 cm depth ( $0.03 \pm 0.01 \text{ mg l}^{-1}$ )  
5019 and stream water had the lowest values ( $0.04 \pm 0.01 \text{ mg l}^{-1}$ ). For pH, snowmelt (pH  $6.09 \pm 0.06$ )  
5020 was more acidic than soil solutions at both depths ( $7.52 \pm 0.23$  at 13 cm depth and  $7.79 \pm 0.11$  at  
5021 26 cm depth) and stream water ( $7.37 \pm 0.07$ ). Drought alone altered DOC in stream water, and  
5022 DOC:DON in soil solution in unthinned (control) watersheds. Volume-weighted concentration of  
5023 DOC was 62% lower ( $p < 0.01$ ) and DOC:DON was 82% lower ( $p = 0.004$ ) in stream water in  
5024 years during drought (WY 2013–2015) than in years prior to drought (WY 2009 and 2010).  
5025 Drought combined with thinning altered DOC and DIN in stream water, and DON and TDN in  
5026 soil solution. For stream water, volume-weighted concentrations of DOC were 66- 94% higher in  
5027 thinned watersheds than in control watersheds for all three consecutive drought years following  
5028 thinning. No differences in DOC concentrations were found between thinned and control  
5029 watersheds before thinning. Watershed characteristics explained inconsistently the variation in  
5030 volume-weighted mean annual values of stream water chemistry among different watersheds.  
5031 The authors conclude that their results showed evidence that the influences of drought and  
5032 thinning are more pronounced for DOC than for N in streams.

5033

## 5034 **Geology**

5035

5036 Kusnierz and Sivers, 2018 (removed from focal)

5037

5038 Kusnierz, P.C., Sivers, E., 2018. How important is geology in evaluating stream habitat? *J Soils*  
5039 *Sediments* 18, 1176–1184. DOI:10.1007/s11368-017-1885-z

5040

5041 The purpose of this study was to assess the importance of considering geology when evaluating  
5042 stream habitat conditions. Stream habitat data were collected from 424 sites on federally  
5043 managed lands in western Montana, USA. These sites represented a variety of ecoregions, stream  
5044 types, management practices, and geologies. The importance of accounting for geology in data  
5045 analysis was evaluated using five sediment-related habitat variables and three analyses that  
5046 examined (1) differences across geology for the entire dataset and for sites in reference and



5047 managed watersheds; (2) differences between reference and managed sites within geologies; and  
5048 (3) the relative strength of geology as a factor when accounting for the effects of management,  
5049 stream type, and ecoregion. This objective was pursued by using five sediment-related habitat  
5050 variables (Log instability index, Log roughness-corrected index of relative bed stability, Median  
5051 substrate size, Percent pool tail fines < 6 mm, Percent stable banks). Five sediment-related  
5052 habitat variables were collected from 424 sites on federally managed lands between 2009-  
5053 2012. Factorial ANOVA on ranks was performed to evaluate the relative importance of geology  
5054 when other factors were taken into account. Results from this study show that differences in  
5055 sediment-related habitat variables did not differ significantly according to geology; however,  
5056 observed differences were typically drawn from managed sites. The authors conclude by  
5057 advising against using geology as the sole means of stratifying habitat data when attempting to  
5058 account for between-site variability.

5059

### 5060 **Stream Temperatures**

5061

5062 Leach et al., 2017 (removed from focal list)

5063

5064 Leach, J.A., Olson, D.H., Anderson, P.D., Eskelson, B.N.I., 2017. Spatial and seasonal variability  
5065 of forested headwater stream temperatures in western Oregon, USA. *Aquat Sci* 79, 291–307.  
5066 DOI:10.1007/s00027-016-0497-9

5067

5068 This study is a case study of thermal regimes for headwater streams in the Keel Mountain Study  
5069 area. This study examined (1) forested headwater stream temperature variability in space and  
5070 time; (2) relationships between stream temperature patterns and weather, above-stream canopy  
5071 cover, and geomorphic attributes; and (3) the predictive ability of a regional stream temperature  
5072 model to account for headwater stream temperature heterogeneity. Stream temperature data was  
5073 collected at 48 sites within a 128-ha watershed in western Oregon between 2012 and 2013.  
5074 Spatial statistical modeling was used to relate stream temperature patterns to site characteristics  
5075 (elevation, stream width, catchment area, slope, aspect, channel substrate, and terrain shading), a  
5076 cluster analysis was used to capture the full variability in annual stream temperatures. Results  
5077 from this study show considerable variability in stream temperature over relatively small areas,  
5078 and between seasons. The greatest spatial variability existed during summer (up to 10 Celsius)  
5079 and during cold and dry winter periods (up to 7.5 Celsius). Geomorphic attributes typically used  
5080 in stream temperature models were not good predictors of variability at headwater scales.

5081

### 5082 **Stream Temperatures**

5083

5084 Groom et al., 2011b

5085

5086 Groom, J.D., Dent, L., Madsen, L.J., Fleuret, J.(2011b). Response of western Oregon (USA)  
5087 stream temperatures to contemporary forest management. *Forest Ecology and Management* 262,  
5088 1618–1629. <https://doi.org/10.1016/j.foreco.2011.07.012>

5089

5090 The objective of this paper was to assess the riparian characteristics that best predict shade, and  
5091 to determine the stream temperature changes that result following harvest. This study took place  
5092 in the Oregon Coastal Range at 33 sites (15 state-owned and 18 private-owned). The 33 sites  
5093 studied were approximately 50-70 years old and predominately composed of Douglas-fir and red  
5094 alder. Private sites (n = 18) followed FPA rules whereby the riparian management area (RMA)s  
5095 are 15 and 21 m wide on small and medium fish-bearing streams, with a 6 m no-cut zone  
5096 immediately adjacent to the stream. Harvesting is allowed in the remaining RMA to a minimum  
5097 basal area of 10.0 (small streams) and 22.9 (medium streams) m<sup>2</sup>/ha. State sites (N = 15)  
5098 followed the state management plan whereby a 52 m wide buffer is required for all fish-bearing  
5099 streams, with an 8 m no cut buffer immediately adjacent to the stream. Limited harvest is  
5100 allowed within 30 m of the stream only to create mature forest conditions. Harvest operations  
5101 within this zone must maintain 124 trees per hectare and a 25% Stand Density Index. Additional  
5102 tree retentions of 25–111 conifer trees and snags/hectare are required between 30 and 52 m. A  
5103 site's control reach was located immediately upstream of its treatment reach. The control reaches  
5104 were continuously forested to a perpendicular slope distance of at least 60 m from the average  
5105 annual high-water level. Reach lengths varied from 137 m to 1,829 m with means of 276 m and  
5106 684 m for the control and treatment reaches, respectively. Temperature recording stations were  
5107 located upstream and downstream of both control and treatment sites. Stream temperature data  
5108 was summarized to provide daily minimum, maximum, mean, and fluctuation for analysis. The  
5109 temperature data was modeled using mixed-effects linear regression. Shade analysis included  
5110 trees per hectare, basal area per hectare, vegetation plot blowdown, and tree height. A linear  
5111 regression analysis of shade data (n = 33) was performed and compared small-sample AIC values  
5112 to determine relative model performance among 8 a priori models. Results showed that average,  
5113 minimum, and diel stream temperatures increased on private sites following harvest, suggesting a  
5114 relationship between decreased shade derived from buffer width and an increase in stream  
5115 temperature. Outputs from the model predicted an increase of ~2 °C for minimum shade  
5116 conditions and a decrease of ~ -1 °C for maximum shade conditions. For sites that exhibited an  
5117 absolute change of shade > 6% from pre-harvest to post-harvest experienced an increase in  
5118 maximum temperatures. Further, the model predicted an increase in stream temperature  
5119 proportional to treatment reach length. The authors estimate an increase in maximum and  
5120 minimum temperatures of 0.73 and 0.59 °C per km, respectively. Following harvest, maximum  
5121 temperatures at private sites increased relative to state sites on average by 0.71 °C. Similarly,  
5122 mean temperatures increased by 0.37 °C (0.24 - 0.50), minimum temperatures by 0.13 °C (0.03 -  
5123 0.23), and diel fluctuation increased by 0.58 °C (0.41 - 0.75) relative to state sites. A comparison  
5124 of within site changes in maximum temperatures pre-harvest to post-harvest showed an overall

5125 increase at private sites, but not all sites behaved the same and some had decreases in maximum  
5126 temperatures. The average of maximum state site temperature changes = 0.0 °C (range = -0.89 to  
5127 2.27 °C). Observed maximum temperature changes at private sites averaged 0.73 °C (range = -  
5128 0.87 to 2.50 °C) and exhibit a greater frequency of post-harvest increases from 0.5 to 2.5 °C  
5129 compared to state sites. Private site shade values also appeared to decrease pre-harvest to post-  
5130 harvest. Private post-harvest shade values differed from pre-harvest values (mean change in  
5131 Shade from 85% to 78%); however, no difference was found for state site shade values pre-  
5132 harvest to post-harvest (mean change in Shade from 90% to 89%). They did not find evidence  
5133 that shade differed if one or both banks were harvested for private sites although the sample size  
5134 for single sided harvests was low. Similarly, private site shade values did not appear to differ  
5135 between medium or small streams. Results from this study also show that between 68% and  
5136 75% of variability in post-harvest shade may be accounted for by basal area within 30 m of the  
5137 stream, tree height, and potentially blowdown. The authors speculate that their results suggest  
5138 sites with shorter trees have higher post-harvest shade and this may be due to the negative  
5139 correlation between crown ratios and tree heights. Overall, this study shows that buffers managed  
5140 by state sites were sufficient at mitigating the effects of upland harvesting on stream temperature.  
5141 Increases in stream temperature on private sites were related to decreases in shade, which were  
5142 related to decreases in basal area on sites with greater tree heights. The authors suggest that their  
5143 results are likely relevant to other high-rainfall low-order Douglas-fir dominated streams in the  
5144 Pacific Northwest that are subject to similar harvest practices.

5145

## 5146 **Litter**

5147

5148 Yeung et al., 2019

5149

5150 Yeung, A. C., Stenroth, K., & Richardson, J. S. (2019). Modelling biophysical controls on stream  
5151 organic matter standing stocks under a range of forest harvesting impacts. *Limnologia*, 78,  
5152 125714. <https://doi.org/10.1016/j.limno.2019.125714>

5153

5154 This study investigates the relative impact of major biophysical controls (stream temperature,  
5155 riparian litterfall, and stream discharge) on in-stream CPOM (coarse particulate organic matter)  
5156 quantity across a variety of streamside timber harvest intensities using simulation modeling. The  
5157 CPOM model used was developed by Stenroth et al., 2014, for similar stream types and  
5158 conditions of coastal rainforest streams of British Columbia. The model was calibrated using  
5159 data from multiple published studies from, primarily the Pacific Northwest region, and several  
5160 other North American regions, that quantified stream flow, temperature, and CPOM following  
5161 different timber harvest intensities within 4 years of harvest. The model used an estimated  
5162 response of low, moderate, high, and very high severity timber harvest for litterfall (-10%, -30%,  
5163 -50%, -90%), peak flows (+20%, +40%, +100%, +300%), and stream temperature (+1°C, +2°C,

5164 +4°C, +6 °C). These changes in litterfall, peak flow, and stream temperature were modeled and  
5165 analyzed individually and cumulatively to estimate their relative and combined effects on in  
5166 stream CPOM standing stocks. Results of the model showed that in general the standing stocks  
5167 of CPOM decreased under the independent effects of reduced litterfall and elevated peak flows  
5168 and increased with higher stream temperatures. Along the gradient of harvest severities, litterfall  
5169 reductions on depleting CPOM standing stocks were at least an order of magnitude greater than  
5170 those of elevated peak flows. At low severity, litterfall reductions led to a 13.5% reduction of  
5171 CPOM stocks while peak flow increases at high severity harvest only led to a 5% reduction in  
5172 CPOM stocks. The magnitude of CPOM changes induced by litterfall reductions was  
5173 consistently greater than stream temperature increases, but their differences in magnitude became  
5174 smaller at higher levels of disturbance severity. For example, at low severity, stream  
5175 temperatures only led to an increase on CPOM stocks by 1.1% while litter fall reductions led to a  
5176 reduction of CPOM by 13.5%. However, at the high intensity treatment CPOM stocks changed  
5177 by -90.24%, and +72.07% for litterfall, and stream temperature respectively. For scenarios  
5178 involving perturbations of multiple model drivers (combined effects), the effect size of  
5179 disturbance was significantly negative (indicating significantly lower CPOM standing stocks  
5180 than in undisturbed conditions) whenever litterfall reductions reached 50% or above (i.e., high  
5181 severity). When litterfall reductions were 30% or below, the effect size of disturbance varied with  
5182 the relative changes in peak flows and stream temperature. Only the effects of litterfall-  
5183 temperature interactions on CPOM standing stocks were significant ( $p < 0.001$ ). The authors  
5184 interpret these results as evidence that litterfall reduction from timber harvest was the strongest  
5185 control on in-stream CPOM quantity for 4 years post-harvest. Further, the authors propose that  
5186 the decreased activity of CPOM consumers caused by increasing stream temperatures may be  
5187 enough to offset the loss of litterfall inputs on CPOM stocks. The caveat of this study is that it  
5188 did not include LW dynamics in preserving CPOM post-harvest. As other studies have shown,  
5189 harvest can increase in-stream LW, and in-stream LW can act as a catchment for CPOM.

5190

## 5191 **Drought Frequency**

5192

5193 Wise, 2010

5194

5195 Wise, E. K. (2010). Tree ring record of streamflow and drought in the upper Snake River. *Water*  
5196 *Resources Research*, 46(11). <https://doi.org/10.1029/2010WR009282>

5197

5198 This study used newly collected tree-ring data augmented with existing chronologies from sites  
5199 at three headwater streams in the Snake River Basin to estimate streamflow patterns for the  
5200 1600-2005 time-period. The reconstructed chronologies were tested for significant correlations  
5201 with streamflow patterns during the 1911-2005 time period prior to extrapolation. Streamflow  
5202 patterns derived from instrumental data and from reconstructed chronologies were compared

5203 with other streamflow reconstructions of three other western rivers in similar climates to  
5204 examine synchronicity among the rivers and gain insight into possible climatic controls on  
5205 drought episodes. The reconstruction model developed for the analysis explained 62% of the  
5206 variance in the instrumental record after adjustment for degrees of freedom. Results showed  
5207 evidence that droughts of the recent past are not yet as severe, in terms of overall magnitude, as a  
5208 30-year extended period of drought discovered in the mid-1600s. However, in terms of number  
5209 of individual years of < 60% mean-flow (i.e., low-flow years), the period from 1977-2001 were  
5210 the most severe. Considering the frequency of consecutive drought years, the longest (7-year-  
5211 droughts), occurred in the early 17<sup>th</sup> and 18<sup>th</sup> centuries. However, the 5-year drought period from  
5212 2000-2004 was the second driest period over the 415-year period examined. The author explains  
5213 that the area has continued to experience a drought period, but its severity could not be  
5214 calculated as it hadn't ended by the time of the study (2010). The correlative analysis of the  
5215 chronologies developed for the upper Snake River with other rivers of the West (the upper  
5216 Colorado, the Sacramento, and the Verde Rivers) showed mixed results with periods of positive  
5217 and negative correlations. The author interprets these results as evidence that drought frequency  
5218 in general, in this area appears to be increasing in severity and that mean annual flow appears to  
5219 be reducing in the latter half of the 20<sup>th</sup> and the beginning of the 21<sup>st</sup> century. The exceptions  
5220 being the 1930's dustbowl, and an unusually long dry period in the early 1600s.

5221

## 5222 **Shade and structure**

5223

5224 Warren et al., 2013

5225

5226 Warren, D. R., Keeton, W. S., Bechtold, H. A., & Rosi-Marshall, E. J. (2013). Comparing  
5227 streambed light availability and canopy cover in streams with old-growth versus early-mature  
5228 riparian forests in western Oregon. *Aquatic sciences*, 75(4), 547-558. DOI:10.1007/s00027-013-  
5229 0299-2

5230

5231 This study investigates the differences in canopy cover and streambed light availability between  
5232 paired reaches in old-growth (> 500 years old) and secondary-growth (~40-60 years old) riparian  
5233 forests on canopy cover and streambed light exposure in four second order fish-bearing streams  
5234 in the H.J. Andrews Experimental Forest. Streams were paired based on reach length and  
5235 bankfull width and north (n=2), and south (n=2) facing watersheds. The overall mean percentage  
5236 of canopy cover was estimated using a convex spherical densiometer every five meters along the  
5237 thalweg of each stream reach. At each point densiometer readings were taken from four  
5238 directions (upstream, downstream, left bank, right bank) The amount of light reaching the bottom  
5239 of the stream was estimated every five meters using fluorescent dye that degrades overtime from  
5240 light exposure. Differences in light availability and canopy cover were analyzed separately for  
5241 each of the four reaches using a single factor ANOVA. To avoid the inclusion of overlapping

5242 canopy images from adjacent densiometer sampling locations, the canopy cover data from sites  
5243 every 15 m (rather than every 5 m) were used in the comparison of canopy cover between the  
5244 two age classes along each reach. Linear regression was used to compare values from mean  
5245 densiometer readings with mean dye photodegradation site (every 5 meters). To evaluate the  
5246 hypothesis that light availability in old-growth forested streams would be more variable than in  
5247 second-growth forested streams, the standard deviations of the mean densiometer readings and  
5248 mean photodegradation values were compared between old-growth and second-growth forested  
5249 streams with an ANOVA. Results showed that the differences in stream light availability and  
5250 percent forest cover between old-growth and second-growth reaches were significant in both of  
5251 the south-facing watersheds in mid-summer at an alpha of 0.01 for the dye results and 0.10 for  
5252 the cover results. For the north-facing watersheds differences in canopy cover and light  
5253 availability (alpha = 0.01, and 0.10 respectively) were only significant at 1 of the two reaches.  
5254 Overall, three of the four paired old-growth reaches had significantly lower mean percent canopy  
5255 cover, and significantly higher mean decline in fluorescent dye concentrations The authors  
5256 interpret these results as evidence that old-growth forest canopies were more complex and had  
5257 more frequent gaps allowing for more light availability and lower mean canopy cover, on  
5258 average, than in adjacent mature second-growth forests.

5259

5260 **LW**

5261

5262 Teply et al., 2007

5263

5264 Teply, M., McGreer, D., Schult, D., & Seymour, P. (2007). Simulating the effects of forest  
5265 management on large woody debris in streams in northern Idaho. *Western Journal of Applied*  
5266 *Forestry*, 22(2), 81–87. <https://doi.org/10.1093/wjaf/22.2.81>

5267

5268 This paper uses simulation modeling to estimate the effects of timber harvest, under the Idaho  
5269 Forest Plan (IFP), on in-stream LW loading for Class I streams (fish-bearing streams) of the  
5270 Priest Lake Watershed in northern Idaho relative to unharvested riparian forest streams. Under  
5271 the IFP, class one streams have a 25-foot no-cut-buffer that extends out from the high-watermark,  
5272 and an additional 50 feet beyond the edge of the no-cut-buffer where harvest requires retention of  
5273 88-trees-per-acre that are greater than 8-in diameter at breast height (DBH). This study used the  
5274 Riparian Aquatic Interaction Simulator (RAIS) to estimate the potential wood loading for 58  
5275 randomly selected north Idaho stream segments with and without harvest. Stream segments were  
5276 measured in the field along the stream centerline from the upstream starting point (0 ft) to a  
5277 downstream ending point (200 ft). Riparian stand conditions were measured within 75 ft-long by  
5278 10-ft-wide strips oriented perpendicular to the stream at 25, 75, 125, and 175 ft downstream of  
5279 the upstream starting point on each side of the stream segment to provide a total of eight strips  
5280 for each stream segment. Along each strip, live trees and snags greater than 8 in dbh within the

5281 strip were located and measured. Three circular subplots, each 10 ft in diameter, were located  
5282 along each 75-foot strip plot at 12.5, 37.5, and 62.5 ft from the stream edge. Within the subplots,  
5283 smaller live trees (less than 8-in. dbh) were tallied by 1-in. dbh classes. Instream LW loads were  
5284 surveyed along the same 200-ft stream segments located for measuring riparian stand conditions.  
5285 Qualifying LW (greater than 4-in diameter and longer than 6.6 ft) occurring within the high-  
5286 water mark along the entire extent of the segment was tallied. Observed instream LW loads  
5287 ranged from 10 to 710 pieces per 1,000 ft of stream. Stream size measured by bank full width  
5288 covered a wide range (1 ft to 190 ft), averaging 32.5 ft (SD = 28.1). The authors determined that  
5289 active streambank erosion was uncommon in the study area and did not include it as a LW  
5290 recruitment mechanism in their analysis. Simulation was based on a four-step process applied to  
5291 each riparian stand: 1) Harvest the stand according to riparian management prescriptions, 2)  
5292 Predict stand characteristics using growth and yield simulators, 3) Estimate the number of trees  
5293 that fall due to mortality in each time step, 4) Calculate the probability that a tree would deliver  
5294 LWD to the stream. The simulation evaluated both a harvest and a no-harvest scenario to predict  
5295 mean in-stream LW loads after 30, 60, and 100 years. The results predicted mean LW loads at 30  
5296 years for the 58 segments studied were 151.1 pieces per 1,000 ft for the no-harvest scenario (SD  
5297 = 76.2) and 145.1 pieces per 1,000 ft for the harvest scenario (SD = 75.6), which were not  
5298 significantly different ( $P = 0.67$ ). However, on a pairwise basis, loads predicted for these  
5299 segments using the harvest scenario were significantly lower by an average of about 6.0 pieces  
5300 per 1,000 ft than those predicted via the no-harvest scenario ( $P < 0.001$ ). Compared to the initial  
5301 surveyed LW loads, LW loads at 30 years predicted in the no-harvest scenario decreased by an  
5302 average of 19.5 pieces per 1,000 ft, representing a significant ( $P < 0.007$ ) downward shift in the  
5303 distribution. Predicted mean LW loads at 60 years were 136.1 pieces per 1,000 ft in the no-  
5304 harvest scenario (SD = 49.2) and 128.3 pieces per 1,000 ft under the harvest scenario (SD =  
5305 48.3). At 100 years, predicted mean LW loads were 122.5 (SD = 35.4) and 116.7 (SD = 35.8),  
5306 respectively. Based on 20-piece LW classes, the frequency distributions of predicted loads  
5307 between the scenarios were not significantly different at either time step. However, on a pairwise  
5308 basis, predicted loads for the harvest scenario were significantly lower than the no-harvest  
5309 scenario by an average of 7.8 ( $P < 0.001$ ) and 5.8 ( $P < 0.001$ ) pieces per 1,000 ft at 60 years and  
5310 100 years, respectively. Compared to LW loads predicted at 30 years and 60 years, LWD loads  
5311 decreased significantly on a pairwise basis by an average of 15.1 ( $P < 0.001$ ) and 13.6 ( $P <$   
5312  $0.001$ ) at 60 and 100 years, respectively. The authors note that the collective effect of the  
5313 assumptions made for the simulation is likely to underestimate the number and variability of LW  
5314 pieces recruited and retained in the streams sampled. The authors interpreted these results as  
5315 evidence that the IFP prescriptions for class I Idaho streams were sufficient in maintaining LW  
5316 recruitment potential.

5317

## 5318 **Shade**

5319

5320 Swartz et al., 2020

5321

5322 Swartz, A., Roon, D., Reiter, M., & Warren, D. (2020). Stream temperature responses to  
5323 experimental riparian canopy gaps along forested headwaters in western Oregon. *Forest Ecology*  
5324 *and Management*, 474, 118354. <https://doi.org/10.1016/j.foreco.2020.118354>

5325

5326 This study tested the effects of adding canopy gaps within young, regenerating forests of western  
5327 Oregon on stream light availability and stream temperatures. The addition of gaps in the young  
5328 regenerating forests were used to theoretically mimic the natural disturbance regimes and the  
5329 higher canopy complexity of late-successional forests. The researchers used a before-after-  
5330 control-impact design on six replicated streams within the McKenzie River Basin. In the  
5331 experimental reaches 30 m gaps were created, centered on a tree next to the stream and at least  
5332 30 m in from the beginning of the reach. The study reaches were located on second- and third-  
5333 order fish-bearing steep step-pool and cascade dominated headwater streams with boulder  
5334 substrate that ranged from 2.2 to 6.4 m in bankfull width and were lined by 40- to 60-year-old  
5335 riparian forests. Study sites in each stream encompassed two 120 m reaches with no large  
5336 tributary inputs within or between the study reaches, and reference and treatment reaches were  
5337 separated by a buffer section of 30–150 m. In each treatment reach, gaps were designed to create  
5338 openings in the canopy that were approximately 20 m in diameter. Gaps were centered on a tree  
5339 next to the stream at approximately meter 30 along each reach. The gap sizes were intended to  
5340 mimic naturally occurring gaps from an individual large tree mortality or small-scale disturbance  
5341 events found in these systems which range from 0.05 to 1.0 gap diameter to tree height ratio with  
5342 smaller gaps occurring more frequently. Using the Douglas-fir canopy height of 50 m, gaps were  
5343 created in the 0.4–1.0 gap diameter to tree height ratio range (approximately 314 m<sup>2</sup> – 1,963  
5344 m<sup>2</sup>). Actual gap sizes varied across sites from approximately 514 m<sup>2</sup> to 1,374 m<sup>2</sup> (0.45 – 0.74  
5345 gap ratios) with a mean of 962 m<sup>2</sup> (mean gap ratio 0.61). Riparian shade was quantified with  
5346 hemispherical photos. Light reaching the stream was quantified using photodegradation of  
5347 fluorescent dyes placed at 5 m intervals, over a 24 -hour period. Stream temperature was  
5348 recorded continuously, at 15-minute intervals, using HOBO sensors to quantify the seven-day  
5349 moving average of mean and maximum temperatures. Data was collected for one year pre-  
5350 harvest, during harvest year (harvest took place in late fall 2017), and one-year post-harvest. To  
5351 determine the effects of experimental canopy gaps on stream light as well as reach responses a  
5352 linear mixed-effects model was fit to the data. The results showed that after gaps were cut, the  
5353 BACI analysis showed strong evidence for significant increase in mean reach light ( $p < 0.01$ ) to  
5354 a mean of 3.91 (SD  $\pm$  1.63) moles of photons m<sup>-2</sup> day<sup>-1</sup>. overall resulting in a mean change in  
5355 light of 2.93 (SD  $\pm$  1.50) moles of photons m<sup>-2</sup> day<sup>-1</sup>. Mean stream shading could not be  
5356 evaluated in the full BACI analysis because post-treatment hemispherical photographs could not  
5357 be taken at all sites due to fire impeding access in 2018. For the remaining sites, the areas  
5358 beneath each gap had notable localized declines in shade, through the entirety of the treatment  
5359 reach mean shading declined by only 4% (SD  $\pm$  0.02%). Overall, the gap treatments did not  
5360 change summer T 7DayMax or T 7DayMean significantly across the 6 study sites. The mean  
5361 response (change in reach difference before and after the cut) indicated an increase on average  
5362 across the six sites in T7DayMax of 0.21 °C ( $\pm$ 0.12 °C) and in the T7DayMean of 0.15 °C ( $\pm$ 0.14  
5363 °C); however, there was not statistical support of the BACI effect for either metric. The light



5364 response was not correlated with T 7DayMax responses ( $r^2 < 0.01$ ,  $p = 0.69$ ), nor was gap area  
5365 ( $r^2 = 0.01$ ,  $p = 0.63$ ), but there was a significant relationship between discharge ( $r^2 = 0.73$ ,  $p =$   
5366  $0.03$ ), and bankfull width ( $r^2 = 0.93$ ,  $p < 0.01$ ) and the T7DayMax response. Wetted width was  
5367 also highly correlated with T 7DayMax responses, but the relationship was not as strong with  
5368 this stream size metric as with discharge or bankfull width ( $r^2 = 0.65$ ,  $p = 0.05$ ). In contrast to the  
5369 summary values, results from the analysis of individual days throughout the full 40-day summer  
5370 period identifying differences in the relationships of daily maximums and daily means between  
5371 reaches showed a statistically significant effect of the gap for average daily maximums ( $p < 0.01$ )  
5372 and for average daily means ( $p = 0.02$ ). The regression comparison reveals there will be on  
5373 average an additional  $0.12\text{ }^{\circ}\text{C}/^{\circ}\text{C}$  increase in daily maximum temperature in the reach with a gap.  
5374 Likewise, for the daily mean, for every degree increase in the shaded reference reach, an average  
5375 additional increase of  $0.05\text{ }^{\circ}\text{C}$  in a reach with a small gap is expected. The authors conclude that  
5376 adding gaps to young regenerating forests only minimally increases temperatures, dependent on  
5377 stream size, and that riparian canopy gaps may be a viable management strategy that can be  
5378 implemented with minimal effects on stream temperatures. This paper does not quantify changes  
5379 in stream productivity, also expected from the increase in available light.

5380

## 5381 **Shade**

5382

5383 Sugden et al., 2019

5384

5385 Sugden, B. D., Steiner, R., & Jones, J. E. (2019). Streamside management zone effectiveness for  
5386 water temperature control in Western Montana. *International Journal of Forest Engineering*,  
5387 30(2), 87-98. <https://doi.org/10.1080/14942119.2019.1571472>

5388

5389 This study investigates the effects of riparian forest timber harvest, under the Montana  
5390 Streamside Management Zone (SMZ) laws, on stream temperature in Class 1 streams (fish-  
5391 bearing, or flow more than 6 months per year and are connected to downstream waters).  
5392 Montana state law requires timber be retained within a minimum of 15.2 m of the class 1  
5393 streams, with equipment exclusion zones extended on steep slopes for up to 30.5 m. Within the  
5394 SMZ no more than half the trees greater than 204 mm (8 in) diameter at breast height (DBH) can  
5395 be removed, and trees retained must be representative of the pre-harvest stand. In no case,  
5396 however, can stocking levels of leave trees be reduced to less than 217 trees per hectare. The  
5397 objectives of this study were to fill the information gap in this region by: (1) evaluating the  
5398 performance of 15.2 m SMZs retained during harvest activities for protecting against adverse  
5399 changes in summer maximum stream temperatures, (2) quantifying the level of timber removal  
5400 occurring within operational SMZs that may help explain any observed changes, and (3)  
5401 Evaluating fish response that may be associated with a stream temperature change. Data for  
5402 stream temperature and fish population response was collected for 30 harvest reaches in western

5403 Montana (northern Rocky Mountain Region), for a minimum of one-year pre- and one-year post-  
5404 harvest. Data for stream temperatures and fish populations were also collected from unharvested  
5405 reaches upstream from the harvest sites as a control. Temperature data was collected  
5406 with Optic StowAway™ and StowAway TidBit™ digital temperature loggers manufactured by  
5407 Onset Computer Corporation. Shade over the stream surface was not directly measured in this  
5408 study. Canopy cover was estimated using a combination of simulation modeling and using a  
5409 concave spherical densiometer. Fish populations were estimated for 100 m reaches at study sites  
5410 using an electro-fishing pass of capture method. Linear mixed effects models were used to  
5411 analyze the relationship between year, stream position, harvest, fish populations and stream  
5412 temperatures. The results showed that within harvest areas, the mean basal area (BA) declined  
5413 from 30.2 m<sup>2</sup>/ha pre-harvest to 26.4 m<sup>2</sup>/ha post-harvest (mean = -13%, range from -32% to  
5414 0%). Windthrow further reduced the mean BA to 25.9 m<sup>2</sup>/ha (mean = -2%, range = -32% -0%).  
5415 Changes in mean canopy cover were not significant based on the simulation modeling (-3%), or  
5416 densiometer readings (+1%). Results of the model for the effect of harvest on stream  
5417 temperature showed no detectable increase in treatment streams relative to control streams. The  
5418 estimated mean site level response in maximum weekly maximum temperatures (MWMT)  
5419 varied from -2.1 °C to +3.3 °C. Overall, 20 of 30 sites had estimated site level response within  
5420 ±0.5 °C. There were five sites that had an estimated site-level response greater than 0.5 °C (i.e.  
5421 warming) and five sites that had an estimated site level response less than -0.5 °C (i.e. cooling).  
5422 Results for the fish population showed approximately 7% increase in trout population from pre-  
5423 harvest to post-harvest, but this difference was not significant. The authors conclude that the  
5424 results suggest that Montana's 15.2 m SMZs retained during timber harvest activities are highly  
5425 protective (change <0.5°C) of stream temperatures.

5426

5427 **LW**

5428

5429 Sobota et al., 2006

5430

5431 Sobota, D. J., Gregory, S. V., & Sickle, J. V. (2006). Riparian tree fall directionality and  
5432 modeling large wood recruitment to streams. *Canadian Journal of Forest Research*, 36(5), 1243–  
5433 1254. <https://doi.org/10.1139/x06-022>

5434

5435 The objectives of this study were to evaluate patterns of riparian tree fall directions in diverse  
5436 environmental conditions and evaluate correlations with tree characteristics, forest structural  
5437 variables, and topographic features. Specifically, the authors were interested in correlations  
5438 between fall directionality and tree species type, tree size, riparian forest structure, and valley  
5439 topography (side slope). Data was collected from 21 field sites located west of the Cascade  
5440 Mountains crest (11 sites: Coast Range and west slopes of the Cascades), and in the interior  
5441 Columbia Basin (10 sites: east slopes of the Cascades, Blue Mountains, and Northern Rockies)

5442 of Oregon, Washington, Idaho, and Montana, USA. Streams were second- to fourth-order  
5443 channels and had riparian forests that were approximately 40 to >200 years old. The location of  
5444 specific study reaches (200–300 m stream length) on each stream were selected randomly.  
5445 Minimum size criteria for a fallen tree in this study were diameter at breast height (DBH) of 0.1  
5446 m and height of 5 m. All fallen trees up to 50 m slope distance from stream or the first 100 trees  
5447 were measured at all sites. Tree fall direction was standardized among sites by streamside  
5448 location (upstream = 0° and 360°; toward stream = 90°; downstream = 180°; away from stream =  
5449 –90° and 270°). Spearman rank correlations were used to compare site level statistics of tree fall  
5450 directions with physical and riparian forest characteristics. Then trees were pooled among sites  
5451 and classified by species for analysis of species, tree size, and valley side slope effects. To avoid  
5452 small sample sizes species were grouped by side slope categories (< 40%, >40%). Average  
5453 direction of tree fall by site was significantly correlated with valley constraint (Spearman  $r = -$   
5454  $0.53$ ;  $P = 0.02$ ). Average direction of tree fall by site was weakly correlated with active channel  
5455 width, tree stem density, and basal area ( $P > 0.05$ ), with Spearman  $r$  coefficients of 0.22,  $-0.21$ ,  
5456 and 0.39, respectively. Trees on valley side slopes >40% for each species had a 95% CI that only  
5457 included falls directly towards the stream channel; trees on side slopes <40% had a 95% CI for  
5458 mean fall direction that included directly upstream, downstream, away from the stream, towards  
5459 the stream, or all four directions simultaneously (consistent with random fall directions),  
5460 depending on species. Tree size was only different between side slope categories for coastal  
5461 Douglas fir on >40% side slopes which had a median DBH 1.2 to 1.9 times greater than trees on  
5462 <40% side slopes. Also, red alder trees on side slopes > 40% had a median DBH 1.1 to 1.6 times  
5463 greater than on side slopes < 40%. Model projections of LW recruitment calibrated with the  
5464 results of the spearman rank correlations estimated that sites with uniform steep side slopes  
5465 (>40%) produced between 1.5 (first resolution) to 2.4 (second resolution) times more in stream  
5466 LW by number of tree boles than sites with uniform moderate side slopes (< 40%). The authors  
5467 interpret their results as evidence that edaphic, topographic, and hydrologic characteristics are  
5468 related to greater variability of tree fall directions on moderate slopes than on steep slopes. The  
5469 authors conclude that models that use tree fall directions in predictions of LW recruitment should  
5470 consider stream valley topography. The authors warn that while side slope categories (>40%,  
5471 <40%) was the strongest predictor of tree fall direction in this study, they believe the differences  
5472 in tree fall direction between these categories mainly characterized differences between fluvial  
5473 (88% of moderate slope sites) and hillslope landforms (71% of steep slope sites). They suggest  
5474 that the Implications from this study are most applicable to small- to medium-size streams  
5475 (second- to fourth-order) in mountainous regions where sustained large wood recruitment from  
5476 riparian forest mortality is the significant management concern.

5477

5478 **LW**

5479

5480 Schuett-Hames & Stewart, 2019a

5481

5482 Schuett-Hames, D., & Stewart, G. (2019a). Post-Harvest Change in Stand Structure, Tree  
5483 Mortality and Tree Fall in Eastern Washington Riparian Buffers: Comparison of the Standard  
5484 and All Available Shade Rules for the Fish-Bearing Streams in the Mixed Conifer Timber Habitat  
5485 Type Under Washington's Forest Practices Habitat Conservation Plan. Cooperative Monitoring  
5486 Evaluation and Research Report CMER. Washington State Forest Practices Adaptive  
5487 Management Program. Washington Department of Natural Resources, Olympia, WA.

5488

5489 This report is a comparative analysis of the differences in stand structure, tree fall, and LW  
5490 recruitment between riparian sites of eastern Washington harvested under the current Standard  
5491 Shade Rule (SR), under the All-Available Shade rule (AAS), and unharvested reference sites  
5492 (REF). Both shade rules have a 30-ft no-cut buffer (core zone) immediately adjacent to the  
5493 stream. The SR prescription allows thinning in the buffer zone 30-75 feet (inner zone) from the  
5494 stream while the AAS prescription requires retention of all shade providing trees in this area.  
5495 Post-harvest surveys were completed at each site one–two years and five years post-harvest. A  
5496 census was done of all standing trees  $\geq 4$  inches diameter at breast height (DBH) within 75 feet  
5497 (horizontal distance) of the channel on both sides of the stream in each treatment and reference  
5498 reach. The condition (live or dead), species, canopy class, and DBH were recorded for each tree.  
5499 Dead or fallen trees with a decay class of 1 or 2 were classified as post-harvest mortality and a  
5500 mortality agent was recorded (e.g. wind, erosion, suppression, fire, insects, disease, and physical  
5501 damage). Metrics were calculated separately for regulatory zones defined by horizontal distance  
5502 from the channel, including the core zone (0–30 feet) and inner zone (30–75 feet) and the  
5503 combined core and inner zone (the full RMZ). Mixed model analysis was used to evaluate  
5504 differences in treatment response. Results showed Cumulative wood recruitment from tree fall  
5505 over the five-year post-harvest interval was highest in the SR group, lower in the AAS group and  
5506 lowest in the REF group. The SR and AAS rates by volume were nearly 300% and 50% higher  
5507 than the REF rates, respectively. The mixed model comparisons indicated that the frequency of  
5508 wood input from fallen trees was significantly greater in SR group compared to both the REF  
5509 and AAS groups ( $p < 0.001$ ), while the difference between REF and AAS groups was not  
5510 significant. Over 60% of pieces recruited from AAS and SR fallen trees consisted of stems with  
5511 attached rootwads (SWAR), double the proportion in the REF sites. The REF-AAS and REF-SR  
5512 differences in recruitment of SWAR pieces were significant ( $p < 0.001$ ). Most recruiting fallen  
5513 trees originated in the core zone (76%, 72%, and 64% for the REF, AAS and SR groups,  
5514 respectively), while the proportion from the inner zone (30–75 feet from the stream) was ~10%  
5515 greater for the SR group compared to the AAS and REF groups. The authors interpret the results  
5516 and conclude that harvest of the adjacent stand outside the RMZ appeared to alter the spatial  
5517 pattern of wood recruitment from fallen trees, increasing recruitment from trees located farther  
5518 from the stream. Recruitment of fallen trees from the inner zone of the AAS and SR sites were  
5519 two and four times the rate for the inner zones of the unharvested reference sites due to increased  
5520 tree fall from wind disturbance in the buffers after harvest of the adjacent stand, as reported in  
5521 other studies. It is important to note that this was a short-term study (5 years). The authors note  
5522 that LW recruitment is a process that can change over decadal time scales. Adding that thinning

5523 and post-harvest mortality also reduced the standing stock of trees available for wood  
5524 recruitment in the SR and AAS RMZs compared to unharvested REF RMZs.

5525

5526 **Litter and LW**

5527

5528 Six et al., 2022

5529

5530 Six, L. J., Bilby, R. E., Reiter, M., James, P., & Villarin, L. (2022). Effects of current forest  
5531 practices on organic matter dynamics in headwater streams at the Trask River watershed,  
5532 Oregon. *Trees, Forests and People*, 8, 100233. <https://doi.org/10.1016/j.tfp.2022.100233>

5533

5534 This study investigates the effects of different riparian timber harvest intensities on changes in  
5535 canopy cover, and litter input into streams and litter transport downstream. The objective of this  
5536 study was to investigate whether differing levels of tree retention adjacent to the channel altered  
5537 coarse particulate organic matter (CPOM) delivery, retention, and transport. The authors  
5538 hypothesized an inverse relationship between tree removal and litter delivery (i.e., increase in  
5539 tree removal adjacent to the channel would result in a reduction of litter delivery). Data was  
5540 collected for leaf litter in streamside litter traps, canopy cover percentage using hemispherical  
5541 photos in-stream LW, and litter retention in stream flume litter traps pre- and post-treatment at  
5542 five watersheds of the Trask River in the northern Oregon Coast range. The experimental design  
5543 included three treatment watersheds: clearcut with no leave trees or retention buffer (CC),  
5544 clearcut with leave trees (CC w/LT; retention of 5 trees per hectare/2 trees per acre), and clearcut  
5545 with 15 m wide retention buffer (CC c/B) and two uncut references (REF 1, and 2) along  
5546 headwater streams. Because there were no replication sites for treatments, data was analyzed  
5547 using descriptive and graphical summaries of the data (i.e., no quantitative statistical analysis).  
5548 Results showed a reduction of canopy cover from 91.4% to 34.4% in the clearcut treatment with  
5549 no leave trees, from 89.8% to 76.1% in the clearcut treatment with leave trees, and from 89.5%  
5550 to 86.9% in the clearcut treatment with the 15 m retention buffer. Change in canopy cover in the  
5551 reference streams was < 1% for both reaches. Post harvest litter delivery decreased for the  
5552 clearcut with no leave trees but increased for both the clearcut with leave tree and clear cut with  
5553 retention buffer. The number of logjams, the total weight of logjams, and the volume of LW in  
5554 streams increased for all treatment sites. The results of this study were consistent with similar  
5555 studies and provide supporting evidence that riparian timber harvest can affect litter and LW  
5556 delivery into and retention in streams.

5557

5558 **Shade and LW**

5559

5560 Schuett-Hames et al., 2011

5561

5562 Dave Schuett-Hames, Ashley Roorbach, Robert Conrad. 2011. Results of the Westside Type N  
5563 Buffer Characteristics, Integrity and Function Study Final Report. Cooperative Monitoring  
5564 Evaluation and Research Report, CMER 12-1201. Washington Department of Natural Resources,  
5565 Olympia, WA.

5566

5567 This report presents the results from the Washington State Westside Type N Buffer  
5568 Characteristics, Integrity and Function (BCIF) study. The purpose of the study was to evaluate  
5569 the effects of westside riparian timber harvest prescriptions for Type Np (perennial non-fish-  
5570 bearing) streams on resource objectives (riparian stand tree mortality, wood recruitment, channel  
5571 debris, shade, and soil disturbance) described in the Forest and Fish Report of 1999. Three  
5572 treatment prescriptions were evaluated, 1) clearcut harvest to the edge of the stream (CC) at eight  
5573 sites, 50-foot-wide no-cut-buffers (50-ft) at 13 sites, and 56-foot radius circular no-cut-buffer at  
5574 the perennial initiation point (PIP) at three sites (not used in statistical analysis due to small  
5575 sample sizes). Each treatment site was paired with an uncut reference site as a control. The CC  
5576 and 50-ft treatments were compared with treatment sites at three time periods (the first 1-3 years,  
5577 years 4-5, and the whole 5-year period). Differences in variable mean values were checked for  
5578 statistical significance between treatment and reference streams using non-parametric Mann-  
5579 Whitney U tests. Tree fall rates (annual fall rates of live and dead standing stems combined) was  
5580 over 8 times and 5 times higher in the 50-foot buffers than in the reference buffers 3 years after  
5581 treatment when compared as a percentage of standing trees and as trees/acre/yr, respectively.  
5582 These differences were significant for both metrics ( $p \leq 0.001$ ). In the period 4-5 years post  
5583 treatment rate of tree uprooting decreased but rate of stem breakage increased in the 50-foot  
5584 buffer. For this period only the percentage of broken trees were significantly different (higher)  
5585 than what was observed in the reference buffers. Over the entire five-year period, the percentages  
5586 of standing trees that were uprooted and broken (as well as the combined total) were  
5587 significantly greater in the 50-foot buffer. Wind was the dominant tree fall process, accounting  
5588 for nearly 75% of combined fallen trees, 11% fell from other trees falling against them and 1.8%  
5589 of fallen trees fell from bank erosion. Differences in mortality followed a similar pattern to tree  
5590 fall rates. In the 50-foot buffer sites mortality rates were significantly higher (3.5 times higher)  
5591 than in the reference sites for the first three years following harvest. However, in years 4-5  
5592 mortality rates increased in the reference buffers after high-intensity storms resulting in non-  
5593 significant differences in mortality during this period. The cumulative percentage of live trees  
5594 that died over the entire five-year period was 27.3% in the 50-ft buffers compared to 13.6% in  
5595 the reference reaches, but the difference was not statistically significant. This was likely because  
5596 of the high variability in mortality between sites in the 50-foot buffers. LW recruitment into the  
5597 channel after treatment was higher in the 50-ft buffers than in the reference patches during the  
5598 first three years after harvest, over 8 times higher in pieces/acre/yr and over 14 times higher in  
5599 volume/acre/yr. In years 4-5 after harvest LW recruitment decreased in the 50-ft buffers and  
5600 increased in the reference patches, and the number of recruited LW pieces/acre/yr was greater in

5601 the reference patches, although the volume of LW recruited was greater in the 50-ft buffers. For  
5602 the entire first 5 years after harvest, the 50-ft buffers recruited about twice the number of LW  
5603 pieces recruited in the reference patches, and over 3 times the volume. The CC treatment,  
5604 unsurprisingly, had significantly lower LW recruitment following harvest relative to the reference  
5605 streams. Mean overhead shade (from trees and tall shrubs) was 13% lower in the 50-ft treatment,  
5606 and 77% lower in the CC treatment relative to reference streams. The CC treatment, however,  
5607 increased by 25% five years after harvest relative to values recorded 1-year following harvest.  
5608 The implications of these results suggest that immediate and direct changes in stand structure,  
5609 canopy cover, and LW are most severe for clear-cut treatments, but that the 50-foot buffer  
5610 treatment showed an increase in LW and stand mortality, and a decrease in shade over the five-  
5611 year period. Limitations of this study were the lack of pre-harvest data and the relatively short  
5612 time-period (5-years) in evaluating impacts that may last for several decades.

5613

5614 Schuett-Hames & Stewart, 2019b (BCIF)

5615

5616 Schuett-Hames, D & Stewart, G. (BCIF), (2019). Changes in stand structure, buffer tree  
5617 mortality and riparian-associated functions 10 years after timber harvest adjacent to non-fish-  
5618 bearing perennial streams in western Washington. Cooperative Monitoring Evaluation and  
5619 Research Report. Washington State Forest Practices Adaptive Management Program. Washington  
5620 Department of Natural Resources, Olympia, WA.

5621

5622 This paper presents a 10 -year follow-up to the results of the BCIF report (Schuett-Hames et al.,  
5623 2012) that originally presented 5-year post-treatment results. Over the 10-year period stand  
5624 mortality in the 50-ft buffer treatment stabilized and showed a cumulative 14.1% reduction in  
5625 live basal, while the reference stands showed a 2.7% increase in live basal area. The differences  
5626 in these values were not significant. Cumulative LW recruited into stream channel over the 10-  
5627 period was double in the 50-ft treatment streams than in the reference streams. However, the  
5628 majority of the LW recruited in the 50-ft treatment streams came to rest above the streams,  
5629 providing shade but not affecting streamflow, pool formation, or sediment storage. Further, while  
5630 the 50-ft buffer treatment provided more LW recruitment in the short-term (10-years), the authors  
5631 speculate there is a reduction in future LW recruitment potential given the removal of trees  
5632 outside the 50-ft buffer. Canopy cover in the 50-ft treatment streams recovered to similar  
5633 percentages as the reference's streams by the end of the 10-year period. The authors speculate  
5634 that the 50-ft buffer was better at maintaining resource objectives than the clearcut but propose  
5635 that the narrow buffers presented variable increases in mortality (specifically increased  
5636 susceptibility to windthrow) and recommend further research before drawing definitive  
5637 conclusions.

5638

5639 **Riparian thinning effects on shade, light, and temperature**

5640

5641 Roon et al., 2021a

5642

5643 Roon, D.A., Dunham, J.B., Groom, J.D., 2021. Shade, light, and stream temperature responses to  
5644 riparian thinning in second-growth redwood forests of northern California. PLOS ONE 16,  
5645 e0246822. <https://doi.org/10.1371/journal.pone.0246822>

5646

5647 The purpose of this study was to evaluate the effects of riparian thinning on shade, light, and  
5648 temperature in three watersheds located in second-growth redwood stands in northern California.  
5649 The objectives of this study were to evaluate: 1) the effects of experimental riparian thinning  
5650 treatments on shade and light conditions; 2) how changes in shade and light associated with  
5651 thinning affected stream temperatures at a reach-scale both locally and downstream; 3) how  
5652 thermal responses varied seasonally; and 4) how these thermal responses were expressed across  
5653 the broader thermal regime to gain a more complete understanding of thinning on stream  
5654 temperatures in these watersheds. This study took place between 2016 and 2018 with thinning  
5655 treatments applied during 2017 giving 1-year pre-treatment and 1-year of post-treatment data.  
5656 Two study sites prescribed treatment on one side of the stream of a 45 m buffer width with a 22.5  
5657 m inner zone with 85% canopy retention and a 22.5 m outer zone that retained 70% canopy  
5658 cover (Green Diamond Resource Company, Tectah watershed). At the third treatment site  
5659 thinning prescriptions included removal of up to 40% of the basal area within the riparian zone  
5660 on slopes less than 20% on both sides of the channel along a ~100–150 m reach (Lost Man  
5661 watershed, Redwood national park). Control reaches were located upstream from treatment  
5662 reaches. Data analysis was conducted separately for each experimental watershed (i.e., 1 Lost  
5663 man site, 2 Tectah sites). Stream temperature was collected using digital sensors; solar radiation  
5664 was measured using silicon pyranometers; riparian shade was measured using hemispherical  
5665 photography. A classical BACI analysis was performed to test the effects of riparian thinning on  
5666 shade, light, and stream temperature using linear-effects models. Results for the Tectah  
5667 watershed showed a significant reduction in canopy closure by a mean of 18.7%, (95% CI: -21.0,  
5668 -16.3) and a significant reduction of effective shade by a mean of 23.0% (-25.8, -20.1) one-year  
5669 post treatment. In the Lost man watershed, a non-significant reduction of mean shade by 4.1% (-  
5670 8.0, -0.5), and mean canopy closure by 1.9% was observed in 2018. Results for below canopy  
5671 light availability showed significant increases by a mean of 33% (27.3, 38.5) in the Tectah  
5672 watershed, and non-significant increases in Lost man watershed of 2.5% (-1.6, 5.6) by 2018.  
5673 Results for stream temperature changes showed variation seasonally and between watersheds.  
5674 The Lost Man watershed showed no significant changes in average daily maximum, maximum  
5675 weekly average of the maximum (MWMT), average daily mean, or maximum weekly average of  
5676 the mean (MWAT). In the Tectah watershed, MWMT increased during spring by a mean of 1.7°C  
5677 (95% CI: 0.9, 2.5), summer by a mean of 2.8°C (1.8, 3.8), and fall by a mean of 1.0°C (0.5, 1.5)  
5678 and increased in downstream reaches during spring by a mean of 1.0°C (0.0, 2.0) and summer by  
5679 a mean of 1.4°C (0.3, 2.6). Thermal variability of streams in the Tectah watershed were most  
5680 pronounced during summer increasing the daily range by a mean of 2.5°C (95% CI:



5681 1.6, 3.4) and variance by a mean of 1.6°C (0.7, 2.5), but also increased during spring (daily  
5682 range: 0.5°C; variance: 0.3°C) and fall (daily range: 0.4°C; variance: 0.1°C). Increases in thermal  
5683 variability in downstream reaches were limited to summer (daily range: 0.7°C; variance: 0.5°C).  
5684 Again, no significant changes in stream and downstream temperature variability were detected in  
5685 the Lost Man watershed. In the Techtah watersheds the frequency of days with temperatures  
5686 greater than 16°C increased in summer by a mean of 42.9 more days (95% CI: 31.5, 53.8) in  
5687 thinned reaches and a mean of 16.3 more days (6.1, 27.4) in downstream reaches. Temperatures  
5688 greater than 16°C persisted for a mean duration of 31.1 more consecutive days (21.0, 41.1) in  
5689 thinned reaches and 11.6 more consecutive days (3.9, 20.0) in downstream reaches under the  
5690 BACI analysis. The authors conclude that responses to the experimental riparian thinning  
5691 treatments we evaluated differed greatly depending on treatment intensity. For example, they  
5692 interpret their results as evidence that that changes in shade of 5% or less caused minimal  
5693 changes in temperature while reductions in shade of 20–30% resulted in much larger increases in  
5694 temperature. However, the authors warn that their data only evaluated immediate (1-year-post-  
5695 treatment) changes in stream shade and temperatures. Also, the study was conducted in relatively  
5696 small (< 10 km<sup>2</sup>) coastal watersheds and may not apply to larger watersheds of different regions.

5697

## 5698 **Sediment**

5699

5700 Safeeq et al., 2020

5701

5702 Safeeq, M., Grant, G.E., Lewis, S.L., Hayes, S.K., 2020. Disentangling effects of forest harvest  
5703 on long-term hydrologic and sediment dynamics, western Cascades, Oregon. *Journal of*  
5704 *Hydrology* 580, 124259. <https://doi.org/10.1016/j.jhydrol.2019.124259>

5705

5706 The purpose of this study was to separate and investigate the effects of changes in streamflow  
5707 and sediment supply due to disturbances (specifically timber harvest), on sediment transport into  
5708 streams. Timber harvest affects both streamflow and sediment supply simultaneously. The  
5709 researchers used a reverse regression technique to evaluate the relative and absolute importance  
5710 of changes in streamflow versus changes in sediment supply on sediment transport. The  
5711 technique was applied to long-term data collected from two paired experimental watersheds in  
5712 the H.J. Andrews Experimental Forest, Oregon. The two watersheds were paired by size, aspect,  
5713 and topography. The treatment watershed was 100% clearcut during the period from 1962-1966,  
5714 broadcast burned in 1966, and re-seeded in 1968. Streamflow, and sediment data were taken  
5715 intermittently, and after large storm events from 1952 (pre-harvest) through 1988 for suspended  
5716 sediment data, and 2016 for sediment bedload. The control watershed was forested, and had no  
5717 treatments (e.g., harvest) during the study period. The results that considered the effects of  
5718 harvest on streamflow alone showed an increase in annual water yield in the treatment watershed

5719 by 10% (136 mm/year) over the 51-year post-treatment period. There were no significant  
5720 changes in precipitation patterns before or after harvest. Further, the patterns of streamflow in the  
5721 control watershed showed diverging patterns in streamflow after the harvest period. The authors  
5722 state that these patterns strongly suggest that the increase in streamflow in the treatment  
5723 watershed was caused by timber harvest. The results for post-treatment sediment yields showed  
5724 suspended load declined to pre-treatment levels in the first two decades following treatment,  
5725 bedload remained elevated, causing the bedload proportion of the total load to increase through  
5726 time. Changes in streamflow alone account for 477 Mg/km<sup>2</sup> (10%) of the suspended load and  
5727 113 Mg/km<sup>2</sup> (5%) of the bedload over the post-treatment period. Increase in suspended sediment  
5728 yield due to increase in sediment supply is 84% of the measured post-treatment total suspended  
5729 sediment yield. In terms of bedload, 93% of the total measured bedload yield during the  
5730 posttreatment period can be attributed to an increase in sediment supply. The authors interpret  
5731 these results as evidence that while streamflow alone can cause a modest increase in sediment  
5732 transport, it is negligible compared to the increases in sediment transport following harvest.  
5733 Following harvest, changes on streamflow alone was estimated in being responsible for < 10% of  
5734 the resulting suspended sediment transported into streams, while the increase in sediment supply  
5735 due to harvest disturbance was responsible for >90%. The authors suggest these results provide  
5736 evidence for a need to investigate thresholds for specific watershed management regimes to  
5737 ameliorate these impacts following harvest, or thinning treatments. Also, the sharp increases in  
5738 sediment transport following logging can be confidently attributed to the increase in sediment  
5739 supply and delivery to streams due to the ground disturbances associated with logging rather than  
5740 increased streamflow.

5741

## 5742 **Stream Temperature**

5743

5744 Reiter et al., 2020

5745

5746 Reiter, M., Johnson, S. L., Homyack, J., Jones, J. E., & James, P. L. (2020). Summer stream  
5747 temperature changes following forest harvest in the headwaters of the Trask River watershed,  
5748 Oregon Coast Range. *Ecohydrology*, 13(3), e2178. <https://doi.org/10.1002/eco.2178>

5749

5750 This paper investigates the effects of different riparian forest harvest treatments on stream  
5751 temperature. Stream temperature data was collected from 2006 to 2016 for multiple small (<50  
5752 ha), non-fish-bearing headwater stream watersheds in the Trask River Watershed of the  
5753 northwestern Oregon Coast range. The experiment followed a BACI design with four treatments,  
5754 1) clearcut, no buffer (CC\_NB; n = 4), 2) clearcut with 10-m no cut buffer (CC\_B; n = 3), 3)  
5755 Thinning with 10 m no-cut buffer (TH\_B; n = 1), and 4) unharvested, reference streams (REF; n  
5756 = 7). Temperature data was collected at 30-minute increments for all streams using continuously  
5757 recording thermistors. Harvest operations occurred in the Summer of 2012 giving 6 summers of

5758 pre-treatment and 4 summers of post-treatment data collection. Temperature data was separated  
5759 into 5<sup>th</sup>, 25<sup>th</sup>, 50<sup>th</sup>, 75<sup>th</sup>, and 95<sup>th</sup> percentiles, with each percentile being treated as independent  
5760 response variables in a linear mixed model. Treatments were compared to reference watersheds  
5761 to check for significant differences in temperature percentiles. For ecological context, the  
5762 researchers also quantified the percentage of summer where temperatures were above 16 and 15  
5763 °C, the preferred thermal regime limits for two local amphibian larvae (coastal tailed frog,  
5764 coastal giant salamander). Results showed that even the small (10 m buffer; CC\_B, TH\_B) buffer  
5765 was efficient in maintaining similar temperature changes throughout the summers compared to  
5766 reference streams. There were no significant changes in the buffered watersheds with  
5767 temperature responses in these watersheds ranging from negative values to negative values close  
5768 to zero. The treatments with no buffer (CC\_NB), however, showed significant increases in  
5769 temperature for all percentiles with the greatest increases occurring in the 95<sup>th</sup> percentile,  
5770 showing a mean increase of 3.6 °C (SE = 0.4). For the 5<sup>th</sup> percentile, the CC\_NB also showed a  
5771 mean temperature response 1.7°C (SE = 0.3; range from 1.5 - 2.8°C). Temperature changes were  
5772 more severe in the CC\_NB watersheds with no leave trees (4.2 and 4.4°C), however, this  
5773 difference was not analyzed. The percentage of time the post-harvest, no-buffer treatments spent  
5774 above the 16 and 15 °C thresholds were 1.3% and 4.7%, respectively. This was an increase from  
5775 pre-harvest values that showed no instances of temperatures above 16°C, and only 0.2% of the  
5776 recorded time above 15°C. The authors conclude that their evaluation of temperature responses  
5777 as potential biologically significant changes adds context to the changes and fluctuations  
5778 observed in each harvest design. While significant changes in mean and percentile changes in  
5779 temperature were observed, the amount of time spent above critical temperature thresholds for  
5780 important amphibian species was minimal.

5781

## 5782 **SHD, Stream temperature**

5783

5784 Chan et al., 2004 (Removed from focal list, significant results only apply to fauna)

5785

5786 Chan, S.S., Anderson, P.D., Cissel, J., Larsen, L., Thompson, C., 2004. Variable density  
5787 management in Riparian Reserves: lessons learned from an operational study in managed forests  
5788 of western Oregon, USA. USDA Forest Service. <https://doi.org/10.1016/j.foreco.2013.06.055>

5789

5790 The purpose of this study was to assess the ability of variable retention thinning and riparian  
5791 buffers at accelerating late-seral habitat, facilitating rare species management, and maintaining  
5792 ecological functions within riparian zones of 40–70-year-old headwater forests in western  
5793 Oregon. This study evaluated 13 separate sites each averaging ~ 100 ha whereby 4 buffer width  
5794 treatments adjacent to variable retention thinning prescriptions were assessed. Buffer treatments  
5795 include: (1) one site potential tree; (2) two-site potential trees; (3) variable buffer width based on  
5796 vegetation and/or topographic site factors; (4) streamside buffer of only the first tree whereby

5797 thinning treatments applied up to 6 m of stream. Thinning treatments included: (1) Unthinned  
5798 control - 500-750 trees per hectare; (2) High density retention - 70-75% of area thinned to 300  
5799 TPH, 25-30% unthinned riparian reserves or leave islands; (3) Moderate density retention - 60-  
5800 65% area thinned to 200 TPH, 25-30% unthinned riparian reserves or leave islands with 10%  
5801 circular patch openings; (4) Variable density retention - 10% area thinned to 100 TPH, 25-30%  
5802 thinned to 200 TPH, 25-30% thinned to 300 TPH, 20-30% unthinned riparian reserves or leave  
5803 islands with 10% circular patch openings. Variables measured include stand development  
5804 metrics, understory vegetation, microclimate, aquatic ecology, invertebrates, lichens, and  
5805 bryophytes. Early findings from this study show that relatively small changes in the riparian  
5806 environment are attributed to different residual thinning densities and different buffer widths.  
5807 According to the results, the most suitable habitat for many species of fauna is consistently found  
5808 within 5 m of the stream. The largest changes in relative humidity in warm and dry summer  
5809 conditions occur within 15 m of the stream channel and begin to stabilize at 25 m. In summary,  
5810 the early findings of this study indicate the near-stream riparian environment provides critical  
5811 functions and habitat for a wide variety of organisms.

5812

## 5813 **Sediment**

5814

5815 Reiter et al., 2009

5816

5817 Reiter, M., Heffner, J. T., Beech, S., Turner, T., & Bilby, R. E. (2009). Temporal and Spatial  
5818 Turbidity Patterns Over 30 Years in a Managed Forest of Western Washington 1. *JAWRA Journal*  
5819 *of the American Water Resources Association*, 45(3), 793-808. [https://doi.org/10.1111/j.1752-](https://doi.org/10.1111/j.1752-1688.2009.00323.x)  
5820 [1688.2009.00323.x](https://doi.org/10.1111/j.1752-1688.2009.00323.x)

5821

5822 This study evaluates the efficacy of the changes in a forest practices plan developed in 1974 to  
5823 reduce sediment inputs into streams in the Deschutes River watershed of western Washington. To  
5824 test this, the researchers analyzed 30 years of data (1975-2005) on water levels, discharge,  
5825 suspended sediment, turbidity, and water and air temperature from four permanent sampling sites  
5826 representing a range of basin sizes from small tributary headwaters to the mainstem of the  
5827 Deschutes River. In the 1970s roughly 30% of the watershed had been harvested and  
5828 approximately 63% of the existing road network had been constructed. Timber harvest continued  
5829 until the early 1990s and the road network was completed in the late 1970s but updated to  
5830 include culverts and sediment traps in the early 2000s. The researchers used turbidity as a proxy  
5831 for suspended sediment correlation and corrected for typical seasonal increases in streamflow.  
5832 The results showed a declining trend in turbidity at all permanent sampling sites during the study  
5833 period even with active forest management. Following the road construction and harvest  
5834 activities of the 1980s turbidity levels continued to decline until the year 2000 when they

5835 returned to pre-logging levels. The authors interpret these results as evidence that management's  
5836 increased attention to reducing sediment is responsible for the reduction in sediment transport.

5837

5838 **Effect of debris torrents on shade, vegetation, and stream temperature**

5839

5840 D'Souza et al., 2011

5841

5842 D'Souza, L.E., Reiter, M., Six, L.J., Bilby, R.E., 2011. Response of vegetation, shade and stream  
5843 temperature to debris torrents in two western Oregon watersheds. *Forest Ecology and*  
5844 *Management* 261, 2157–2167. <https://doi.org/10.1016/j.foreco.2011.03.015>

5845

5846 The purpose of this study was to examine the effects of debris torrents on vegetation, shade, and  
5847 stream temperature eight years after an extreme storm-related disturbance. This study examined  
5848 two separate managed watersheds which were affected by storm-related debris torrents in 1996.  
5849 This study addressed several questions regarding the patterns and rate of vegetation, shade and  
5850 water temperature change post-disturbance: (1) What is the relationship between vegetation and  
5851 local landform and substrate types along the study streams? (2) Does vegetation composition and  
5852 structure, stream shade and water temperature in debris torrented streams differ between the two  
5853 watersheds? and (3) How does recovery of stream temperature relate to vegetation and shade  
5854 recovery and does this differ through time between watersheds? Data was gathered from  
5855 multiple headwater streams following the disturbance in 1996 at 2 managed watersheds: the  
5856 Williams River watershed (WRW), and the Calapooia River watershed (CRW). Data for stream  
5857 temperature, to analyze stream temperature recovery, was collected immediately following the  
5858 disturbance event in 5 streams, 3 at the CRW (2 disturbed; 1 reference), and 3 at the WRW (1  
5859 disturbed, 1 reference) and for 8 years through the summer of 2004. Eight years post-disturbance  
5860 12 disturbed streams (n = 6 for each watershed) were selected for data collection to examine the  
5861 relationships between riparian vegetation, shade, and stream temperatures. Data on landform,  
5862 substrate, and vegetation (density, species, and seedlings) were collected at each stream. Stream  
5863 shade was estimated using hemispherical photographs taken 1 m above the stream center during  
5864 summer and winter months and compared using t-tests. Stream temperature data was collected  
5865 using continuously recording thermistors. Data were averaged and analyzed using t-tests, chi-  
5866 square tests, simple linear regression, Pearson's correlation coefficient, and analysis of  
5867 covariance. Results from this study show early successional species red alder and willow species  
5868 dominated areas affected by debris torrents. All red alder variables (density, basal area, and  
5869 height) showed a significant relationship with vegetation-related shade. Red alder showed a  
5870 significantly higher density ( $p = 0.0277$ ) and basal area ( $p = 0.0367$ ) in the WRW sites. While  
5871 stem density of red alder was similar in both watersheds, the size of the trees differed suggesting  
5872 that colonization and/or growth of red alder in the WRW occurred more rapidly than in the CRW.  
5873 However, there was no statistical difference in landforms or site factors between watersheds that

5874 explained these differences. The only correlations found were a negative relationship between  
5875 alder density and rock; and a positive relationship between alder basal area and moss suggesting  
5876 a relationship between moisture availability and red alder establishment and growth. The authors  
5877 note that the WRW sites experienced greater precipitation in the years following disturbance and  
5878 may have contributed to the greater growth rates of red alder, but no analysis was conducted.  
5879 Total shade was also significantly higher in the WRW ( $p = 0.0049$ ). Mean maximum daily  
5880 temperature fluctuations ( $p = 0.0483$ ), and 7-day maximum temperatures ( $p=0,0483$ ) were also  
5881 significantly lower in the WRW streams. Mean max daily stream temperatures were lower in the  
5882 WRW streams but the difference was not significant ( $p = 0.0779$ ). The authors conclude that  
5883 even though the debris torrents resulted in poor soil conditions, the ability of red alder to thrive  
5884 in these conditions resulted in rapid recovery of shade and thermal control.

5885

### 5886 **Stream temperature, shade and climate**

5887

5888 Reiter et al., 2015

5889

5890 Reiter, M., Bilby, R. E., Beech, S., & Heffner, J. (2015). Stream temperature patterns over 35  
5891 years in a managed forest of western Washington. *JAWRA Journal of the American Water*  
5892 *Resources Association*, 51(5), 1418-1435. <https://doi.org/10.1111/1752-1688.12324>

5893

5894 This study was an analysis of long-term stream temperature data in a western Washington  
5895 watershed to evaluate the effects of forest management, before and after implementation of  
5896 riparian forest best management practices, and climate change on stream temperatures. Stream  
5897 temperature data from four permanent sampling stations in the Deschutes River Watershed.  
5898 Stream and air temperature data was analyzed on a monthly basis from 1975-2009. This long-  
5899 term dataset allowed for the examination of changes in stream temperature in four basins of  
5900 varying size across a period from before stream buffers were implemented, during their  
5901 implementation, and several instances of buffer expansion. Because the study period covered  
5902 such a long time the changes in stream temperature based on climate change needed to be  
5903 accounted for as well. The recovery of shade was estimated using the shade recovery function  
5904 developed by R. Summers of Oregon State University (1983), whereby stream shade is estimated  
5905 by angular canopy density (ACD) as a function of the age of stream-adjacent harvest units. To  
5906 detect correlations of stream and air temperature change with land management activity  
5907 separately from climate changes the data was fit to a model that included the effects of climate.  
5908 The researchers accomplished this with a technique for deriving the residuals between stream  
5909 temperature and climate called locally weighted scatterplot smoothing (LOWESS). The four  
5910 watersheds varied in size from small (2 sites: Hard Creek, 2.4 km<sup>2</sup>; Ware Creek, 2.9 km<sup>2</sup>),  
5911 medium (1 site: Thurston Creek, 9.3 km<sup>2</sup>), and large (1 site: The Deschutes River Station, 150  
5912 km<sup>2</sup>). In the 1970s nominal buffer widths were required along fish-bearing streams, which

5913 expanded in the 1980s (requirements not listed), again in the mid-1990s to 23 m, and again to 30  
5914 m in 2001. Methods for stream temperature data collection varied at different periods resulting in  
5915 a margin of error for monthly temperatures of 0.14°C for 1975 - 1983, 0.09°C for 1984 – 1999,  
5916 and 0.02°C. for 2000 – 2009. Because these margins of error were smaller than what the authors  
5917 expected from climate and management, they were not accounted for in confidence intervals and  
5918 p-values. The results for air temperature changes showed a statistically significant ( $p \leq 0.05$ )  
5919 increasing trend in regional air temperatures for July TMAX\_AIR and June and July  
5920 TMIN\_AIR. The trend for TMAX\_AIR for July resulted in a trend magnitude of +0.07°C per  
5921 year, for a total increase of 2.45°C over the 35-year record. For minimum air temperatures the  
5922 magnitude of the June trend was +0.03°C per year while July TMIN\_AIR had a trend magnitude  
5923 of +0.04°C per year. The resulting increases in minimum temperatures for the period of record  
5924 are 1.05°C and 1.40°C for June and July TMIN\_AIR, respectively. Results for trends in stream  
5925 temperature over the 35-year study period without adjustment for climate change showed no  
5926 statistically significant trend in water temperature changes for the large watershed, while the  
5927 medium watershed (Thurston Creek) showed decreasing trends in TMAX\_WAT for June, July,  
5928 and August, ranging in magnitude from 0.05°C (August) to 0.08°C (July) per year. For the  
5929 smaller watershed, Hard Creek (Ware Creek was not included in this analysis), had significant  
5930 decreasing trends in TMAX\_WAT for July, August, and September. The magnitude of these  
5931 trends was yearly decreases of TMAX\_WAT by 0.05, 0.08, and 0.05°C, for July, August, and  
5932 September, respectively. Significant changes in trends for TMIN\_WAT were only found for the  
5933 large basin site with yearly increases of 0.04, 0.03, and 0.04°C for July, August, and September,  
5934 respectively. Results for stream temperature trends after adjusting for changes in air temperature  
5935 (climate) showed significant decreasing trends in TMAX\_WAT for the large basin by 0.04, 0.03,  
5936 and 0.04°C yearly, for July, August, and September, respectively. For the medium basin, trends  
5937 showed yearly decreases in TMAX\_WAT of 0.07, 0.08, 0.06, and 0.03 for June, July, August,  
5938 and September, respectively. For the small basin, climate adjusted trends in TMAX\_WAT  
5939 showed significant decreases in yearly trends by 0.05, 0.08, and 0.05 for July, August, and  
5940 September, respectively. When stream temperature was examined with its correlation with  
5941 estimated annual shade recovery from initial harvest (indexed by ACD). Significant correlations  
5942 were found for monthly temperature metrics that were adjusted for climate, for all basins. The  
5943 strongest correlations were for the smallest basin (Ware Creek) with correlation coefficients for  
5944 climate adjusted maximum water temperatures (CTMAX\_WAT) with ACD valuing -0.66, -.078,  
5945 -0.65, and - 0.69 for June, July, August, and September, respectively. Correlation coefficients for  
5946 Ware Creek CTMIN\_WAT with ACD were -0.46, -0.64, -0.71, and -0.52 for June July, August,  
5947 and September respectively. The largest basin (The Deschutes River) only showed significant  
5948 correlations of CTMAX\_WAT with ACD with July (-0.39) and August (-0.25); and only showed  
5949 significant correlations of CTMIN\_WAT with ACD for the months of August (+0.27), and  
5950 September (+0.37). The authors interpret their results as evidence that following canopy  
5951 recovery after implementation of riparian harvest rules the larger mainstem of the Deschutes  
5952 River decreased in average maximum temperatures by approximately 1.3 °C when accounting for  
5953 climate driven changes. The effects of canopy closure cooling were even more dramatic in the  
5954 smaller headwater streams by 2.67 and 1.6 °C during the study period when accounting for  
5955 climate driven changes (this includes a 0.5 °C correction based on climate warming). However,

5956 following re-initiation of timber harvest in 2001 for the area, when riparian protection buffers of  
5957 30 m minimum were required, there was no detectable change in stream temperatures. The  
5958 authors conclude that the results of this study show evidence that implementation of protection  
5959 buffers in this area were sufficient in maintaining stream temperatures. Conversely, this study  
5960 also shows evidence that despite these protections from land management induced stream  
5961 temperature changes, these protections have been somewhat offset by the warming climate  
5962 conditions.

5963

#### 5964 **Overstory structure effects on understory light and vegetation**

5965

5966 Giesbrecht et al., 2017 (removed from focal, not relevant to questions, essentially a case study)

5967

5968 Giesbrecht, I.J.W., Saunders, S.C., MacKinnon, A., Lertzman, K.P., 2017. Overstory structure  
5969 drives fine-scale coupling of understory light and vegetation in two temperate rainforest  
5970 floodplains. *Can. J. For. Res.* 47, 1244–1256. [dx.doi.org/10.1139/cjfr-2016-0466](https://doi.org/10.1139/cjfr-2016-0466)

5971

5972 The purpose of this paper was to characterize the overstory structure and understory light  
5973 regimes of temperate rainforest floodplains, and to assess the role of light and other site variables  
5974 in driving stand vegetation patterns and processes. This study took place along two 1-ha coastal  
5975 BC, Canada floodplain sites. These sites were selected as representative examples of floodplain  
5976 forests in the Coastal Temperate Rainforest (CTR) as part of a larger network of long-term, old-  
5977 growth monitoring plots. These sites were in the submontane variant of the very wet maritime  
5978 subzone of the Coastal Western Hemlock zone (CWHvm1) of the B.C. coast. In each stand, the  
5979 largest overstory trees are *Picea sitchensis* (Bong.) Carr., with several individuals taller than 60 m  
5980 in height (maximum of 62 to 93 m). Based on coring a sample of main canopy trees, stand age at  
5981 Kitlope is at least 95 years. Stand age at Carmanah is at least 350 years, based on a core from a  
5982 50 m tall *P. sitchensis*. All trees  $\geq 5$  cm were measured along with all understory vegetation  
5983 within 25 2m x 2m subplots. Stand characteristics were recorded as well as information on gap  
5984 origins. Hemispheric canopy photographs were taken to estimate understory light penetration.  
5985 Visual estimations of organic material, mineral layer, CWD, and other substrates were taken in  
5986 each vegetation subplot. Relationships among measures of light transmission, vegetation  
5987 structure, and diversity were analyzed with linear correlation analysis. Nonmetric  
5988 multidimensional scaling was used to describe variation in species composition on multivariate  
5989 axes. Results from this study show both sites as having a relatively high degree of canopy  
5990 openness (11-11.6%) and light transmission (median 18% full sun) compared to many other  
5991 tropical and temperate forests. Light transmission at both sites is however significantly lower  
5992 than a number of old-growth sites in Quebec and northern BC. The origins of canopy openness  
5993 and stand shade differ between both sites indicating distinct stand processes and different stages  
5994 of stand development. Further, light levels vary substantially within short distances at each site



5995 reflecting a complex overstory structure. Although results from this study are reflective  
5996 specifically of the coastal temperate rainforests of BC, the descriptive assessment of these two  
5997 separate floodplain forests reveal a natural disturbance history which fostered a high degree of  
5998 canopy openness and structural heterogeneity which may ultimately aid in informing future  
5999 temperate rainforest floodplain restoration efforts.

6000

6001 **LW**

6002

6003 Reid & Hassan, 2020

6004

6005 Reid, D. A., & Hassan, M. A. (2020). Response of in-stream wood to riparian timber harvesting:  
6006 Field observations and long-term projections. *Water Resources Research*, 56(8),  
6007 e2020WR027077. <https://doi.org/10.1029/2020WR027077>

6008

6009 This paper proposes a conceptual model of wood storage response to different harvesting  
6010 intensities. The model predicts how LW in streams is expected to change spatially and  
6011 temporally following three different harvest patterns. The model was developed with 45 years of  
6012 LW data retrieved from the Pacific coastal region of Vancouver Island, British Columbia. The  
6013 Carnation Creek watershed, which supports gravel bed forested streams, contains riparian forests  
6014 that have received a wide range of harvest plans implemented. During logging in the 1970s and  
6015 '80s riparian forests of one region were harvested with buffer widths ranging from 1 – 70 meters  
6016 in upstream reaches, and another region with near complete or complete removal of vegetation to  
6017 the streams edge in downstream reaches. In-stream wood volume and characteristics data has  
6018 been collected in eight of these study reaches since 1973 (pre-harvest). The researchers used this  
6019 data with simulation modelling to develop a reach-scale wood budget model that predicts wood  
6020 loss and recover patterns for 300 years (1900-2200). This paper has two objectives: (i) to use this  
6021 field data and modeling approach to examine LW storage changes, the time to minimum wood  
6022 load, and wood load recovery times as a result of riparian timber harvesting and forest  
6023 regeneration, and (ii) to describe the characteristics of in stream wood, with particular focus to  
6024 spatial and temporal patterns in wood storage over the multidecade scale following harvesting in  
6025 riparian areas. The model was based upon the proposed response outlined by Murphy and Koski  
6026 (1989). Wood budget responses were estimated using three management scenarios. Scenario 1 is  
6027 a no harvest scenario, in this configuration, the loss of wood supply from the landscape has little  
6028 to no impact on input from wood mortality or bank erosion, and therefore in-stream storage,  
6029 decay, and transport of wood is not affected. Scenario 2 represents partial loss of forested area in  
6030 the riparian zone, which will lead to a near-immediate reduction in wood recruitment to the  
6031 channel from mortality and bank erosion along harvested areas. Wood decay and other  
6032 components of wood loss will exceed rates of input, leading to a reduction in storage until time  
6033  $T_{min}$ , the point where wood recruitment equals losses as the forest regrows in riparian areas and

6034 the greatest overall reduction in storage has occurred ( $\Delta S_{max}$ ). Wood storage increases  
6035 thereafter, eventually recovering to preharvest levels after time  $T_{rec}$ . Scenario 3 represents an  
6036 intensive harvest scenario where most of the riparian area has undergone harvesting over a short  
6037 period of time, a major reduction of input from bank erosion and mortality occurs. This greater  
6038 reduction leads to a much larger  $\Delta S_{max}$  than in Figure 1b as wood losses exceed recruitment.  
6039 However, as the dominant wood sources recover at the same rate, the time to  $T_{min}$  and  $T_{rec}$  is  
6040 similar under both the moderate and intensive harvest scenarios. Results of the model show  
6041 evidence that wood storage in streams of harvested reaches, hits its minimum value in 50 years  
6042 or more following loss of LW input, decay, and export of current stock. Recovery of LW volume  
6043 in-streams following harvest is estimated to take approximately 150-200 years. The pattern and  
6044 intensity of the harvesting operation had little effect on LW loss and recovery times but did affect  
6045 the estimated magnitude of LW volume loss in the first 50 – 80 years. These results show  
6046 evidence that timber harvest has a long-term effect on LW storage and loading dynamics even  
6047 with protective buffers. However, buffers can ameliorate the magnitude of LW loss during the  
6048 recovery period. The one caveat of this model is it doesn't account for as much variability on  
6049 stream configuration or valley morphologies that are likely to affect LW storage.

6050

#### 6051 **Buffers and LW Recruitment**

6052

6053 Grizzel et al., 2000 (Removed)

6054

6055 Grizzel, J., McGowan, M., Smith, D., Beechie, T., 2000. STREAMSIDE BUFFERS AND  
6056 LARGE WOODY DEBRIS RECRUITMENT: EVALUATING THE EFFECTIVENESS OF  
6057 WATERSHED ANALYSIS PRESCRIPTIONS IN THE NORTH CASCADES REGION  
6058 (Timber/Fish/Wildlife Monitoring Advisory Group and the Northwest Indian Fisheries  
6059 Commission). fp\_tfw\_mag1\_00\_003

6060

6061 This study analyzed the effectiveness of the Washington Watershed Analysis (WWA)  
6062 prescriptions at recruiting large wood. This study took place at 10 riparian sites distributed across  
6063 5 watershed administrative units in the Northern Cascades of Washington. Ten sites were  
6064 randomly chosen with gradients and buffer width classes in compliance with WWA indices. To  
6065 analyze WWA effectiveness, debris frequency and size at each site were compared to targets  
6066 derived from WWA. In addition, debris recruitment was compared between three buffer width  
6067 classes. Geometric mean diameter and geometric mean length of debris was calculated based on  
6068 measurements of midpoint diameter and total lengths. This data was then compared to targets  
6069 derived from a channel width-dependent regression. Results show post-harvest mortality  
6070 substantially decreasing stand density at several sites. In stream frequency targets were met at  
6071 most sites; however, debris categorized as "good" for habitat was only achieved at four out of ten  
6072 sites. At the time of data collection, a large portion of debris recruited from buffers was either

6073 above or outside the bankfull flow zone. The authors point out that the degree to which the debris  
6074 will influence fluvial processes in the future will depend on whether or not they are recruited into  
6075 the stream and will also depend on the size and state of decay. The size of debris recruited from  
6076 buffers was significantly smaller than recruited from unmanaged old-growth stands.  
6077 Interestingly, data shows recruitment occurring from the outermost margins of the widest buffers  
6078 (20-30 m, >30 m), suggesting narrow buffers may limit recruitment. The authors point out that  
6079 the large degree of variability in recruitment from site to site suggests windthrow as an important  
6080 causal factor. In channels oriented perpendicular to damaging winds (east-west), there was a  
6081 higher likelihood of potential recruitment as compared to channels oriented parallel to damaging  
6082 winds. The authors conclude with multiple recommendations for future study. First, they suggest  
6083 integrating habitat inventory with recruitment to achieve a better understanding of relationships.  
6084 Second, they suggest future study into the fate of debris suspended above channels given much  
6085 of our current understanding is based on assumptions of decay and breakage. Finally, they  
6086 recommend study into factors influencing windthrow in riparian buffers.

6087

## 6088 **Sediment**

6089

6090 Rachels et al., 2020

6091

6092 Rachels, A. A., Bladon, K. D., Bywater-Reyes, S., & Hatten, J. A. (2020). Quantifying effects of  
6093 forest harvesting on sources of suspended sediment to an Oregon Coast Range headwater stream.  
6094 *Forest Ecology and Management*, 466, 118123. <https://doi.org/10.1016/j.foreco.2020.118123>

6095

6096 This study uses sediment source fingerprinting techniques to quantify the proportional  
6097 relationship of sediment sources (hillslope, roads, streambanks) in harvested and un-harvested  
6098 watersheds of the Oregon Coast Range. The researchers used sediment traps, and chemical  
6099 analysis to estimate the origin of suspended sediment in the stream and to quantify magnitude of  
6100 sediment stored in protection buffers. The study included one catchment (Enos Creek) that was  
6101 partially clearcut harvested in the summer of 2016 and an unharvested reference catchment  
6102 (Scheele Creek) located ~3.5 km northwest of Enos Creek. The paired watersheds had similar  
6103 road networks, drainage areas, lithologies and topographies. The treatment watershed was  
6104 harvested with a skyline buffer technique in the summer of 2016 under the Oregon Forest  
6105 practices Act policy that requires a minimum 15 m no-cut buffer. The proportion of suspended  
6106 sediment sources were similar in the harvested ( $90.3 \pm 3.4\%$  from stream bank;  $7.1 \pm 3.1\%$  from  
6107 hillslope) and unharvest ( $93.1 \pm 1.8\%$  from streambank;  $6.9 \pm 1.8\%$  from hillslope) watersheds.  
6108 However, the harvested watershed contained a small portion of sediment from roads ( $3.6 \pm$   
6109  $3.6\%$ ), while the unharvested reference watershed suspended sediment contained no sediment  
6110 sourced from roads. In the harvested watersheds the sediment mass eroded from the general  
6111 harvest areas ( $96.5 \pm 57.0$  g) was approximately 10 times greater than the amount trapped in the

6112 riparian buffer ( $9.1 \pm 1.9$  g), and 4.6 times greater than the amount of sediment collected from  
6113 the unharvested hillslope ( $21.0 \pm 3.3$  g). These results suggest that the riparian buffer was  
6114 efficient in reducing sediment erosion relative to the harvested area. The caveat of this study was  
6115 the limited sample size (1 treatment, 1 paired reference watershed) and does not incorporate the  
6116 effects of different watershed physiography on sediment erosion.

6117

6118 **SED**

6119

6120 Puntteney-Desmond et al., 2020

6121

6122 Puntteney-Desmond, K. C., Bladon, K. D., & Silins, U. (2020). Runoff and sediment production  
6123 from harvested hillslopes and the riparian area during high intensity rainfall events. *Journal of*  
6124 *Hydrology*, 582, 124452. <https://doi.org/10.1016/j.jhydrol.2019.124452>

6125

6126 This study uses simulation modeling to evaluate the differences in run-off rates, sediment  
6127 concentrations, and sediment yields between watershed harvested areas, along the interface of  
6128 harvested areas and riparian buffers, and within riparian buffers during periods of high-intensity  
6129 rainfall events. The model simulations were calibrated with soil and watershed characteristic data  
6130 collected from the Star Creek catchment located in southeastern Alberta. Fifteen plots were  
6131 selected for rainfall simulations along three transects on a north facing hillslope (aspect:  $\sim 358^\circ$ )  
6132 and along two transects on a southeast facing hillslope (aspect:  $\sim 129^\circ$ ). Each transect consisted  
6133 of three plots that were spaced  $\sim 20$  m apart along the planar hillslopes. Each plot was one  
6134 square-meter, which was bounded by a three-sided steel frame that was inserted into the soil with  
6135 the open side facing down the slope. The plots were located either (a) within the general harvest  
6136 area, (b) along the edge of the riparian buffer at the interface with the harvested area, or (c)  
6137 within the riparian buffer. The high-intensity rainfall events were calibrated to mimic 100-year,  
6138 or greater, storm events of the Northern Rocky Mountains (1-hour high intensity rainfall). The  
6139 results showed runoff rates and surface and shallow subsurface were greatest in the buffer areas  
6140 than in the harvested areas or in the harvest-buffer interfaces especially during dry conditions.  
6141 During the dry condition rainfall simulations, the general pattern of runoff rates (surface/shallow  
6142 subsurface flow) was riparian buffer ( $175.6 \pm 17.3$  [SE]  $\text{ml min}^{-1}$ ) > harvest-riparian edge  
6143 ( $125.8 \pm 18.2$   $\text{ml min}^{-1}$ ) > general harvest area ( $37.2 \pm 8.5$   $\text{ml min}^{-1}$ ). Mean runoff rates within  
6144 the riparian buffer plots were greater than within the general harvest area plots ( $t = 2.90$ ,  $p = .03$ ).  
6145 Runoff ratios were only statistically greater in the riparian buffer plots ( $13.9 \pm 3.1\%$ ) relative to  
6146 the general harvest area ( $2.9 \pm 1.5\%$ ) during the dry conditions. All runoff ratios declined during  
6147 the wet condition rainfall simulations relative to the dry condition simulations with no evidence  
6148 for differences between any of the plot positions ( $p > .27$  for all pairwise comparisons). During  
6149 the dry condition rainfall simulations, the general patterns of sediment concentrations and  
6150 sediment yields were opposite of the runoff rates, with the general harvest area > harvest-riparian

6151 edge > riparian buffer. The sediment concentration was (a) 424.8 mg l<sup>-1</sup> (151.0–1195.3 mg l<sup>-1</sup>)  
6152 in the general harvest area, (b) 100.9 mg l<sup>-1</sup> (45.8–222.1 mg l<sup>-1</sup>) along the harvest riparian  
6153 edge, and (c) 26.9 mg l<sup>-1</sup> (12.2–59.1 mg l<sup>-1</sup>) in the riparian buffer. Statistically, there was  
6154 strong evidence for differences in sediment concentrations between the general harvest area and  
6155 along the harvest-riparian edge ( $t = 3.21$ ,  $p = .01$ ) and between the harvest area and the riparian  
6156 buffer ( $t = 6.17$ ,  $p < .001$ ). Statistically, there was no evidence for differences in sediment yields  
6157 between any of the plot positions. Sediment concentration among plot positions remained the  
6158 same during the wet rainfall simulations as the dry rainfall simulations—general harvest area >  
6159 harvest-riparian edge > riparian buffer. The geometric mean and 95% confidence intervals (back-  
6160 transformed) for the sediment concentration was (a) 285.7 mg l<sup>-1</sup> (67.9–1201.5 mg l<sup>-1</sup>) in the  
6161 general harvest area, (b) 79.6 mg l<sup>-1</sup> (36.5–173.5 mg l<sup>-1</sup>) along the harvest-riparian edge, and  
6162 (c) 22.3 mg l<sup>-1</sup> (3.5–141.7 mg l<sup>-1</sup>) in the riparian buffer. However, while sediment  
6163 concentrations differed most strongly between the general harvest area and the riparian buffer ( $t$   
6164 = 3.51,  $p = .01$ ), other pairwise comparisons were not significant ( $p > .20$ ). Statistically, there  
6165 was no evidence for differences in sediment yields between any of the plot positions for rainfall  
6166 simulations during wet conditions. The authors speculate this was likely due to the greater soil  
6167 porosity in the disturbed, harvested areas. Sediment concentration in the runoff, however, was  
6168 approximately 15.8 times higher for the harvested area than in the riparian buffer, and 4.2 times  
6169 greater than in the harvest-buffer interface. Total sediment yields from the harvested area (runoff  
6170 + sediment concentration) were approximately 2 times greater than in the buffer areas, and 1.2  
6171 times greater in the harvest-buffer interface (however, these proportions were not statistically  
6172 different). Replication of the model showed high levels of variability in total run off rate,  
6173 sediment concentrations, and sediment yields but the relationships between timing and relative  
6174 magnitudes between the three experimental areas were consistent. The authors speculate that  
6175 these results will become more relevant as climate change is expected to increase the frequency  
6176 of high-intensity rainfall events following dry periods in this area. They suggest expanding  
6177 similar methods to understand these effects in areas of different hydro-climatic settings.

6178

## 6179 **Stream Temperature**

6180

6181 Pollock et al., 2009

6182

6183 Pollock, M. M., Beechie, T. J., Liermann, M., & Bigley, R. E. (2009). Stream temperature  
6184 relationships to forest harvest in western Washington 1. *JAWRA Journal of the American Water*  
6185 *Resources Association*, 45(1), 141-156. <https://doi.org/10.1111/j.1752-1688.2008.00266.x>

6186

6187 This study investigates the effect of watershed harvest percentage, and time since harvest on  
6188 summer stream temperatures at different scales in the Olympic Peninsula, Washington. The  
6189 researchers examined recorded stream temperature data in 40 small watersheds that experienced

6190 a range of harvest from 0 – 100% (7 unharvested, 33 harvested between 25-100%), with  
6191 regrowth age groups binned for analysis as recently clear cut (< 20 years old) and less recently  
6192 clearcut (mostly < 40 years old). Unharvested sites were estimated as being >150-years old.  
6193 Clearcut is defined in this paper as removing any protective canopy cover for streams. This study  
6194 tested 3 hypotheses: (1) the condition of the riparian forest immediately upstream of a site  
6195 primarily controls stream temperature, (2) the condition of the entire riparian forest network  
6196 affects stream temperature, and (3) the forest condition of the entire basin affects stream  
6197 temperature. These hypotheses were test by examining correlations of stream temperature with  
6198 the condition of the immediate upstream riparian forest, or more correlated with forest conditions  
6199 more spatially distant and on a coarser scale, such as the entire upstream riparian forest network  
6200 or the forest condition of the entire basin. To avoid site effects in their analysis sites were chosen  
6201 from a narrow range of subbasin sizes (approximately 1-10 km<sup>2</sup>) and elevation (75-400 m).  
6202 Further, all sites were underlain by sedimentary rock and had perennial flow. Each hypothesis  
6203 was tested with linear regression to evaluate the correlations of each age group at each scale with  
6204 stream temperature data. The researchers also used AIC value comparisons for model selection to  
6205 assess the correlation of other physiographic features (elevation, basin area, aspect, slope, or  
6206 geologic composition) with stream temperatures. Results of general temperature patterns showed  
6207 that average daily maximum (ADM) were strongly correlated with average diurnal fluctuations  
6208 ( $r^2 = 0.87$ ,  $p < 0.001$ ,  $n = 40$ ), indicating that cool streams also had more stable temperatures. For  
6209 basin-level harvest effects on stream temperatures. The percentage of the basin harvested  
6210 explained 39% of the variation in the ADM among subbasins ( $r^2 = 0.39$ ,  $p < 0.001$ ,  $n = 40$ ) and  
6211 32% of variation in the average daily range (ADR) ( $r^2 = 0.32$ ,  $p < 0.001$ ,  $n = 40$ ). The median  
6212 ADM for the unharvested subbasins was 12.8 °C (mean = 12.1 °C), which was significantly  
6213 lower than 14.5 °C, the median (and average) ADM for the harvested subbasins ( $p < 0.001$ ).  
6214 Likewise, the median (and average) ADR for the unharvested subbasins was 0.9 °C, which was  
6215 significantly lower than 1.6 °C, the median ADR (average = 1.7 °C) for the harvested subbasins  
6216 ( $p < 0.001$ ). Results for the correlations between the riparian network scale forest harvest and  
6217 stream temperature showed that the total percentage of the riparian forest network upstream of  
6218 temperature loggers harvested explained 33% of the variation in the ADM among subbasins ( $r^2 =$   
6219  $0.33$ ,  $p < 0.001$ ,  $n = 40$ ) and 20% of variation in the ADR ( $r^2 = 0.20$ ,  $p = 0.003$ ,  $n = 40$ ).  
6220 However, the total percentage of upstream riparian forest harvested within the last 20 years was  
6221 not significantly correlated to ADM or ADR. Results for near upstream riparian harvest and  
6222 stream temperature showed either non-significant, or very weakly significant correlations. For  
6223 example, there were no significant correlations between the percentage of near upstream riparian  
6224 forest recently clear-cut and ADM temperature ( $r^2 = 0.03$ ,  $p = 0.79$ ,  $n = 40$ ), the ADR of stream  
6225 temperatures ( $r^2 = 0.02$ ,  $p = 0.61$ ,  $n = 40$ ) or any other stream temperature parameters. The  
6226 proportion of total harvested near upstream riparian forest (avg = 0.66, SD  $\pm$  0.34, range = 0.0-  
6227 1.0) was weakly correlated with ADM ( $r^2 = 0.12$ ,  $p = 0.02$ ,  $n = 40$ ) and not significantly  
6228 correlated with ADR ( $r^2 = 0.07$ ,  $p = 0.06$ ,  $n = 40$ ). Even when the upstream riparian corridor  
6229 length was shortened to 400 m and then to 200 m, and the definition of recently harvested was  
6230 narrowed to <10 year, no significant relationships between temperature and the condition of the  
6231 near upstream riparian forest was found. Results for the effect of physical landscape variables on  
6232 stream temperature found that the variables of elevation, slope, aspect, percent of the basin with

6233 a glacial surficial geology, upstream distance of the site to sedimentary (bedrock) geology, and  
6234 the percent of sedimentary surficial geology in the basin individually explain between 5% and  
6235 14% more of the variability relative to basin harvest. Adding any one of these variables to the  
6236 model increases the  $r^2$  from 0.40 up to between 0.48 and 0.51. However, the coefficient for  
6237 percent of basin harvested and its standard error stay essentially the same, thus the authors  
6238 concluded that adding additional variables to the model did not change the basic finding that  
6239 there is a strong relationship between ADM and total amount of harvest in a basin. Thus, for  
6240 these models, the percentage of basin area harvested was the best predictor of variation in mean  
6241 maximum stream temperatures. The probability of stream temperatures increasing beyond DOE  
6242 standards (16 °C for seven-day average of maximum temperatures) increased with percent  
6243 harvest. Nine of the 18 sites with 50-75% harvest and seven of the nine sites with >75% harvest  
6244 failed to meet these standards. The authors interpret these results as evidence that the total  
6245 amount of forest harvested within a basin, and within a riparian stream network are the most  
6246 important predictors of changes in summer stream temperatures. They conclude that watersheds  
6247 with 25-100% of their total area harvested had higher stream temperatures than watersheds with  
6248 little or no harvest. Furthermore, they speculate that past basin-wide timber management can  
6249 impact stream temperatures over long periods of time in a way that riparian buffer treatments  
6250 cannot entirely ameliorate.

6251

## 6252 **Stream Temperature**

6253

6254 Groom et al., 2011a

6255

6256 Groom, J.D., Dent, L., Madsen, L.J., 2011. Stream temperature change detection for state and  
6257 private forests in the Oregon Coast Range. *Water Resources Research* 47.  
6258 <https://doi.org/10.1029/2009WR009061>

6259

6260 The purpose of this study was to evaluate the effectiveness of private and state forest buffer rules  
6261 on state water quality stream temperature antidegradation standards in the Oregon Coast Range.  
6262 According to the Department of Environmental Quality (DEQ), under the Protecting Cold Water  
6263 (PCW) criterion, anthropogenic activities are not permitted to increase stream temperature by  
6264 more than 0.3 °C above its ambient temperature. In addition, the cumulative amount of  
6265 anthropogenic temperature increase allowed in streams with temperature total maximum daily  
6266 loads (TMDLs) is 0.3 °C for all sources combined. Stream temperature and riparian stand  
6267 conditions were measured pre- and post-harvest between 2002 and 2008 at 33 sites (18 private-  
6268 owned, 15 state-managed). Treatment stands included 26 clear-cuts and 7 partial cuts (leave tree  
6269 requirements not specified), all of which were harvested in adherence to FPA (private) and FMP  
6270 (state) standards. Private sites followed FPA rules whereby the riparian management area  
6271 (RMA)s are 15 and 21 m wide on small and medium fish-bearing streams, respectively, with a 6

6272 m no-cut zone immediately adjacent to the stream. State sites followed the state management  
6273 plan whereby a 52 m wide buffer is required for all fish-bearing streams, with an 8 m no cut  
6274 buffer immediately adjacent to the stream. Stream temperature data was collected for at least 2  
6275 years prior to harvest. Reference reaches were located immediately upstream from the harvested  
6276 reaches. Generalized least square regression was used to model ambient conditions while  
6277 accounting for temporal autocorrelation. The authors examined prediction intervals to assess the  
6278 rule exceedance ( $>0.3$  °C increase in temperature). Results indicate that sites harvested according  
6279 to FPA standards exhibited a 40.1% probability that a pre harvest to post harvest comparison of  
6280 2 years of data will detect a temperature change of  $> 0.3$ °C. Conversely, harvest to state FMP  
6281 standards resulted in an 8.6% probability of exceedance that did not significantly differ from all  
6282 other comparisons. The a priori and secondary post hoc multimodel comparisons did not indicate  
6283 that timber harvest increased the probability of PCW exceedance at state sites. The authors point  
6284 out that the  $0.3$ °C change threshold still lies 1 or 2 orders of magnitude lower than previous  
6285 findings from studies which took place prior to the enactment of the riparian protection  
6286 standards. The authors recommend further research looking into the potential persistence of  
6287 stream temperature change downstream after harvest. In addition, they recommend looking into  
6288 the biological significance of increases in stream temperature change particularly to aquatic life.

6289

## 6290 **Stream and subsurface water temperature**

6291

6292 Guenther et al., 2014

6293

6294 Guenther, S.M., Gomi, T., Moore, R.D., 2014. Stream and bed temperature variability in a  
6295 coastal headwater catchment: influences of surface-subsurface interactions and partial-retention  
6296 forest harvesting. *Hydrological Processes* 28, 1238–1249. <https://doi.org/10.1002/hyp.9673>

6297

6298 This study documented changes in stream and subsurface water temperature in response to forest  
6299 harvesting in two paired headwater catchments. Specifically, the researchers hypothesized that  
6300 post-logging changes in bed temperatures should be greatest in locations experiencing hyporheic  
6301 downwelling (DW) and least in areas with lateral inflow/groundwater discharge. This study took  
6302 place in the University of British Columbia Malcolm Knapp Research Forest near Vancouver,  
6303 Canada. As a part of an ongoing study into the effects of riparian buffers on stream ecology. The  
6304 catchments of 3 southerly-aspect first order streams were harvested using partial retention (50%  
6305 removal of basal area including riparian zone) methods resulting in approximately 14% reduction  
6306 in canopy cover on average; 3 other southerly-aspect streams served as unharvested controls.  
6307 Before thinning treatments, the harvested riparian forests were dominated by western hemlock,  
6308 (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), and Douglas-fir (*Pseudotsuga*  
6309 *menziesii*). The forests were mature second growth forests with trees approximately 30-40 m tall,  
6310 and canopy closure than 90%. Harvest operations began in September 2004 and completed in



6311 November of 2004. Temperature data was summarized from 10-minute intervals to daily  
6312 minimum, maximum, and mean temperatures for stream and bed temperatures for one-year prior  
6313 to, and one year following harvest. An analysis of the post-harvesting effects was conducted  
6314 using a paired-catchment analysis. Results from this study show treatment sites resulted in higher  
6315 daily maximum stream and bed temperatures after harvest but smaller changes in daily minima.  
6316 Daily maximum post-harvest stream temperatures averaged over July and August ranged from  
6317 1.6°C to 3°C at different locations. Post harvest changes in bed temperature at the lower reaches  
6318 were smaller than changes in stream temperature, but was greater at sites with downwelling (DF)  
6319 flow, and decreased with depth at upwelling (UW) and DF sites dropping to approximately 1°C  
6320 at a depth of 30 cm. Changes did not vary significantly with depth at the middle reach, and  
6321 averaged approximately 1°C change in daily maximum bed temperature over July and August. In  
6322 summary, stream temperature responses differed at different locations within the cutblock. Bed  
6323 temperatures also differed between UW and DW zones as well as between reaches with different  
6324 contributions of lateral inflow. Given evidence that stream/bed temperature is shown to change  
6325 spatially and with differences in hyporheic exchange and lateral inflow, the authors conclude by  
6326 suggesting further research into the how these results might impact biological and ecological  
6327 processes.

6328

#### 6329 **Stream Temperature and evaporation/wind speed**

6330

6331 Guenther et al., 2012 (not in focal, does not separate the effects of shade reduction from wind  
6332 speed/)

6333

6334 Guenther, S. M., Moore, R. D., & Gomi, T. (2012). Riparian microclimate and evaporation from  
6335 a coastal headwater stream, and their response to partial-retention forest harvesting. *Agricultural  
6336 and Forest Meteorology*, 164, 1-9.

6337

6338 The purpose of this study was to (1) develop and test an evaporimeter designed specifically to  
6339 measure stream surface evaporation from headwater streams; (2) fit a wind function for  
6340 computing evaporation from meteorological observations, and to compare it to previously  
6341 published wind functions for evaporation from streams; and (3) quantify the influence of partial-  
6342 retention forest harvesting on riparian microclimate and evaporation. This study was conducted  
6343 in the University of British Columbia Malcom Knapp Research Forest (MKRF), approximately  
6344 60 miles east of Vancouver, Canada and focused on the headwater stream of Griffith Creek. The  
6345 harvesting treatment involved removal of 50% of the basal area from within the cut block,  
6346 including the riparian zone. Smaller stems were removed, leaving the larger stems for harvest at  
6347 a later date. creek. Analysis of paired pre- and post-logging hemispherical photographs indicated  
6348 that canopy closure decreased by about 14% due to the logging treatment. Air temperature and  
6349 relative humidity were measured by a Campbell Scientific CS500 sensor with stated accuracies

6350 of  $\pm 0.5$  °C for temperature and  $\pm 3$ –6% for relative humidity. Wind speed was measured with a  
6351 Met One anemometer with a stall speed of  $0.447 \text{ m s}^{-1}$ . Instruments were scanned every 10 s by  
6352 a Campbell Scientific CR10x data logger; observations were averaged and stored every 10  
6353 minutes. Evaporation was measured using four specially designed evaporimeters comprising an  
6354 evaporation pan connected to a Mariotte cylinder. Results showed that Daily mean wind speeds  
6355 increased following harvest, but were still consistently lower than wind speeds at the control site,  
6356 with a maximum of  $1.09 \text{ m s}^{-1}$ . Vapor pressure was generally lower after harvesting. Vapor  
6357 pressure deficit (vpd) increased following harvesting, but tended to remain lower than vpd  
6358 measured at the control site. After harvesting, the relatively high wind speeds in the afternoon  
6359 generally coincided with higher water temperatures, which in turn are associated with higher vpd  
6360 at the water surface and a stronger vapor pressure gradient to drive evaporation. After harvest,  
6361 wind speeds and vapor pressure gradients were higher and stability was weaker, consistent with  
6362 the observed increase in evaporation. The authors conclude that the generally stronger relations  
6363 between riparian and open microclimate variables after harvesting suggest that the riparian zone  
6364 became more strongly coupled to ambient climatic conditions after harvesting as a result of  
6365 increased ventilation. Further, that stream evaporation increased markedly as a result of partial  
6366 retention harvest, consistent with the decrease in atmospheric vapor pressure, the increase in  
6367 stream vapor pressure, the increase in wind speed and the decreased stability. In fact, prior to  
6368 harvest, vapor pressure gradients often favored condensation rather than evaporation.

6369

6370 **LW**

6371

6372 Opperman, 2005 (Not in focal)

6373

6374 Opperman, J. J. (2005). Large woody debris and land management in California's hardwood-  
6375 dominated watersheds. *Environmental Management*, 35(3), 266-277. DOI:10.1007/s00267-004-  
6376 0068-z

6377

6378 The purpose of this paper was to evaluate the effects of stream and riparian area characteristics  
6379 (bankfull width, gradient, basal area), and land ownership (public vs. private) on LW loading,  
6380 and frequency, and debris jam frequency (response variables) in 21 hardwood-dominated forests  
6381 of a Mediterranean climate region of northern California. The relationship between the stream  
6382 and riparian area characteristics (explanatory variables: basal area of riparian trees, bankfull  
6383 width, and gradient), and the response variables (woody debris loading and frequency, and  
6384 debris-jam frequency) were evaluated with linear regression. The characteristics were then  
6385 combined with ownership categories and their relative weight in explaining LW loading,  
6386 frequency and pool frequency were assessed with a multi-variate analysis. Debris jam frequency  
6387 was also analyzed by channel position with a chi-square. Results showed that debris jam  
6388 frequency in the 21 reaches analyzed were strongly influenced by living standing trees rooted at

6389 the margins of the bank, especially in channel positions near the stream bank, but also spanning  
6390 the channel partially, or completely. In general, LW loading was significantly higher in reaches  
6391 adjacent to public lands ( $104 \pm 13$  m<sup>3</sup>/ha) than in those adjacent to private lands ( $46 \pm 8$  m<sup>3</sup>/ha;  
6392  $P = 0.0015$ ). The strongest relationship for LW loading was with bankfull width ( $r^2 = 0.32$ ;  $p =$   
6393  $0.0006$ ), and riparian basal area ( $r^2 = 0.22$ ;  $p = 0.006$ ) riparian basal area. This is likely the cause  
6394 of the difference in public vs. private, as the public lands had significantly higher basal area in  
6395 the riparian areas at distances  $>5$  m from the stream, than the private lands. Debris jam frequency  
6396 was also significantly influenced by riparian area gradient ( $r^2 = 0.14$ ;  $p = 0.03$ ) and basal area ( $r^2$   
6397  $= 0.11$ ;  $p = 0.05$ ). The author concludes that landownership, and thus, land-management  
6398 practices are driving factors in LW dynamics in this region.

6399

6400 **LW**

6401

6402 Nowakowski & Wohl, 2008

6403

6404 Nowakowski, A. L., & Wohl, E. (2008). Influences on wood load in mountain streams of the  
6405 Bighorn National Forest, Wyoming, USA. *Environmental Management*, 42(4), 557-571.  
6406 DOI:10.1007/s00267-008-9140-4

6407

6408 The purpose of this paper is to evaluate the relationship between riparian area characteristics, and  
6409 land management practices with in-stream wood-loads in the Bighorn National Forest of  
6410 northern Wyoming. The authors hypothesized that 1) valley geometry correlates with wood load,  
6411 2) stream gradient correlates with wood load, 3) wood loads are significantly lower in managed  
6412 watersheds than in similar unmanaged watersheds. The study analyzed data from 19 conifer  
6413 dominated, forested headwater reaches in the bighorn mountains. Study reaches were separated  
6414 by two watersheds, managed and unmanaged, with similar drainages, elevation, and lithology.  
6415 Unmanaged watersheds were defined as having a history of minimal anthropogenic influences.  
6416 The managed watershed had a history of different harvest prescriptions from unregulated in the  
6417 late 1800s, clearcutting in the mid-1900s with tie floating practices. The relationship between in-  
6418 stream wood loads (m<sup>3</sup>/ha) was analyzed with 11 valley-scale (elevation, forest type, forest stand  
6419 density, etc.) and 13 channel-scale (reach gradient, channel width, etc.) variables with linear  
6420 regression. Results support the first and third hypotheses. Across all streams, the highest  
6421 explanatory power of all models tested produced land use (managed vs unmanaged), and basal  
6422 area as a significant predictor of wood loads ( $r^2 = 0.8048$ ). For the unmanaged watershed the  
6423 model produced stream valley sideslope gradient as the single best predictor of wood load ( $r^2 =$   
6424  $0.5748$ ) supporting the first hypothesis. Shear stress was the best predictor of wood load in the  
6425 managed watersheds ( $r^2 = 0.2403$ ), These results did not directly support the second hypothesis.  
6426 The authors suggest that while shear stress is a function of stream gradient (shear stress and  
6427 stream gradient were significantly correlated,  $r^2 = 0.9392$ ), gradient itself did not have the

6428 highest explanatory power of wood load in any of the models tested. Valley characteristics  
6429 consistently explained more of the variability in wood load (42-80%) than channel characteristics  
6430 (21-33%). When land use (managed vs. Unmanaged) effect on wood loads was analyzed the  
6431 number of wood pieces per 100 m of stream was marginally significant ( $p = 0.0565$ ), and the  
6432 difference in wood volume per channel was significant ( $p = 0.0200$ ) supporting the third  
6433 hypothesis. When the significant valley and channel characteristics of the managed and  
6434 unmanaged watersheds were controlled for, the significant difference in wood loads between  
6435 managed and unmanaged watersheds were enhanced ( $p = 0.0006$ ). Managed watersheds (1.1  
6436  $m^3/100 m$ ) had, on average, 2-3 times lower in-stream wood loads than unmanaged (3.3  $m^3/100$   
6437  $m$ ) watersheds. These results suggest watersheds with a history of timber harvest have a decrease  
6438 in stream wood loads than unmanaged watersheds, and that wood load dynamics can be driven  
6439 by valley morphology, specifically, slope.

6440

### 6441 **Harvesting Practices on Suspended Sediment Yields**

6442

6443 Hatten et al., 2018

6444

6445 Hatten, J.A., Segura, C., Bladon, K.D., Hale, V.C., Ice, G.G., Stednick, J.D., 2018. Effects of  
6446 contemporary forest harvesting on suspended sediment in the Oregon Coast Range: Alsea  
6447 Watershed Study Revisited. *Forest Ecology and Management* 408, 238–248.

6448 <https://doi.org/10.1016/j.foreco.2017.10.049>

6449

6450 The objectives of this study were to (1) determine the effects of contemporary harvesting  
6451 practices on suspended sediment yields and concentration, and (2) determine if contemporary  
6452 harvesting practices produce lower sediment yields than historic practices. This study took place  
6453 in the central Oregon Coast Range and consisted of a paired watershed study whereby Flynn  
6454 Creek (FC) served as a reference watershed and Needle Branch (NB) served as a treatment  
6455 watershed. A third watershed, Deer Creek (DC) served as a secondary control to compare  
6456 historical vs contemporary harvest practices. The upper section of the treatment watershed was  
6457 clearcut harvested using contemporary harvest practices (no buffer in non-fish-bearing streams  
6458 with equipment exclusion zones, and a 15 m no-cut-buffer in fish-bearing streams) adhering to  
6459 BMP's. Daily precipitation, discharge, and suspended sediment were collected at all three  
6460 watersheds from October 2005 to June 2016. The upper half of the treatment watershed, (35 ha;  
6461 measured at the Needle Branch Upper Gage or NBUG) was harvested in 2009 (Phase I) and the  
6462 lower half (NBLG) was harvested in the fall of 2014 and mid-summer 2015 (Phase II). A model  
6463 was developed using step wise linear regression to compare suspended sediment concentration  
6464 (SSC). Differences in SSC among downstream sites and across harvest entries were compared  
6465 utilizing an analysis of covariance. Results of the stepwise multiple linear regression showed  
6466 strong evidence ( $p < .001$ ) that all covariates (hydrograph limb, cumulative area discharge within

6467 water year, day of water year, daily precipitation, previous day's precipitation) were related to  
6468 SSC across all watersheds. Both the mean and maximum SSC were greater in the reference  
6469 catchments (FCG and DCG) compared to the harvested catchment (NBLG) across all water  
6470 years. In NBLG the mean SSC was 32 mg L<sup>-1</sup> (~63%) lower after the Phase I harvest and 28.3  
6471 mg L<sup>-1</sup> (~55%) lower after the Phase II harvest when compared to the pre-harvest  
6472 concentrations. Compared to the reference watersheds, the mean SSC was 1.5-times greater in  
6473 FCG (reference) compared to NBLG during the pre-harvest period. After the Phase I harvest the  
6474 mean SSC in FCG (reference) was 3.1-times greater and after the Phase II harvest was 2.9-times  
6475 greater when compared to the SSC in NBLG, the harvested watershed. Data from historical and  
6476 contemporary harvests indicate contemporary practices are more effective at mitigating  
6477 sedimentation. Historical data from the original study show harvesting without buffers, road  
6478 building, and slash burning resulted in ~2.8 times increase in annual sediment yields and aquatic  
6479 ecosystem degradation. The authors conclude that contemporary harvesting practices (i.e., stream  
6480 buffers, smaller harvest units, no broadcast burning, leaving material in channels) using buffers  
6481 were shown to sufficiently mitigate sediment delivery to streams, especially when compared to  
6482 historic practices.

6483

#### 6484 **Riparian Vegetation Removal Effects on Inputs and Production.**

6485

6486 Hetrick et al., 1998 (Removed, outside of timeline)

6487

6488 Hetrick, N.J., Brusven, M.A., Meehan, W.R., Bjornn, T.C., 1998. Changes in Solar Input, Water  
6489 Temperature, Periphyton Accumulation, and Allochthonous Input and Storage after Canopy  
6490 Removal along Two Small Salmon Streams in Southeast Alaska. *Transactions of the American*  
6491 *Fisheries Society* 127, 859–875. [https://doi.org/10.1577/1548-](https://doi.org/10.1577/1548-8659(1998)127<0859:CISIWT>2.0.CO;2)  
6492 [8659\(1998\)127<0859:CISIWT>2.0.CO;2](https://doi.org/10.1577/1548-8659(1998)127<0859:CISIWT>2.0.CO;2)

6493

6494 The purpose of this study was to assess whether or not the removal of second growth riparian  
6495 vegetation would affect the production of juvenile coho salmon. In addition, this study aims to  
6496 understand whether perceived effects are due to changes in habitat or food availability. This  
6497 study took place in the Tongas National Forest on Prince of Wales Island, Alaska. Experimental  
6498 reaches were divided into untreated and treated sections whereby treated sections had all  
6499 vegetation on both sides of the streambank 6-15 m back removed. Stream discharge, water  
6500 temperature, periphyton accumulation, allochthonous inputs, and storage of benthic organic  
6501 matter were assessed during the summer and fall of 1988-1989. Differences in measured  
6502 variables were assessed with a split-block analysis of variance. Results from this study show  
6503 average light intensities reaching the water surface was significantly greater ( $P < 0.01$ ) in the  
6504 open canopy block than in the closed canopy block and was influenced significantly by weather  
6505 conditions. Removal of riparian vegetation in both sections of the study significantly increased

6506 the accumulation of periphyton biomass and chlorophyll a ( $P < 0.01$ ), and significantly decreased  
6507 the amount of allochthonous organic inputs to streams ( $P < 0.01$ ). Average daily allochthonous  
6508 input rates for closed and open canopy conditions at Eleven creek were 789 and 6 mg AFDM/m<sup>2</sup>  
6509 respectively, while input rates for closed and open canopy conditions at Woodsy creek were 805  
6510 and 6 mg AFDM/m<sup>2</sup>. Average daily water temperatures in open and closed canopy blocks at  
6511 Eleven Creek were similar in 1988 but were significantly higher in the open blocks than in the  
6512 closed blocks in 1989 ( $P < 0.01$ ). The authors conclude by suggesting a thorough investigation  
6513 into the interactions and responses of higher trophic levels to increases in periphyton biomass  
6514 production and decreases in allochthonous inputs resulting from removal of riparian vegetation.  
6515 Furthermore, the authors point out that the ability of stream segments to retain organic inputs  
6516 through in-stream large wood may be a more important factor for allochthonous input processing  
6517 by stream biota than the amount of allochthonous inputs entering a stream.

6518

### 6519 **Wood Recruitment and Retention**

6520

6521 Hough-Snee et al., 2016

6522

6523 Hough-Snee, N., Kasprak, A., Rossi, R.K., Bouwes, N., Roper, B.B., Wheaton, J.M., 2016.  
6524 Hydrogeomorphic and Biotic Drivers of Instream Wood Differ Across Sub-basins of the  
6525 Columbia River Basin, USA. *River Research and Applications* 32, 1302–1315.  
6526 <https://doi.org/10.1002/rra.2968>

6527

6528 The purpose of this study was to understand the hydrogeomorphic and ecological processes  
6529 which lead to wood recruitment and retention in seven sub-basins of the interior Columbia River  
6530 Basin (CRB), USA. To achieve this, in-stream wood volume and frequency are quantified across  
6531 sub basins. Following this, the riparian, geomorphic, and hydrologic attributes which are most  
6532 strongly correlated to in-stream wood loads were determined. Random forest models were used  
6533 to identify relationships between ecological and hydrogeomorphic attributes that influence in-  
6534 stream wood within each sub-basin. Non-metric multidimensional scaling was performed on a  
6535 matrix of hydrogeomorphic and forest cover variables, excluding instream wood frequency and  
6536 volume to visualize reaches and sub-basins' relative similarity. To determine how wood  
6537 predictors differed between sub-basins, ordinary least squares regression models of wood volume  
6538 and frequency were built within each sub-basin. Results from this study show that in stream  
6539 wood volume and frequency were distinctly different across all seven sub-basins. Across the  
6540 CRB, wood frequency ranged from 0 to 2117.0 pieces km<sup>-1</sup>, while volume ranged from 0 to 539  
6541 m<sup>3</sup> km<sup>-1</sup>. Large wood volume (PERMANOVA  $F = 5.1$ ;  $p = 0.001$ ) and frequency  
6542 (PERMANOVA  $F = 5.4$ ;  $p = 0.001$ ) differed significantly between sub-basins. According to  
6543 random forest (RF) models, mean annual precipitation, riparian large tree cover, and individual  
6544 watershed were the three most important predictors of wood volume and frequency. Watershed

6545 area was the fourth strongest predictor of wood frequency, while catchment-scale and reach-scale  
6546 forest cover were the fourth and fifth strongest predictor of wood volume. In contrast, sinuosity  
6547 and measures of streamflow and stream power were relatively weak predictors of wood volume  
6548 and frequency. Taken together, wood volume and frequency increased with precipitation and  
6549 large riparian tree cover and decreased with watershed area. Final RF models explained 43.5% of  
6550 the variance in volume and 42.0% of the variance in frequency of in stream wood loads. Results  
6551 for drivers of wood frequency and volume between sub-basins were highly variable either  
6552 showing no relationship between candidate models and predictive power (e.g.,  $r^2 \leq 0.12$ ; Entiat  
6553 sub-basin). The highest predictive models for wood volume ( $r^2 > 0.55$ ) and wood frequency ( $r^2$   
6554  $\leq 0.45$ ) were for the John Day sub basin. Depending on the sub basin wood volume and  
6555 frequency was positively correlated with forest cover, watershed area, large tree cover, 25-year  
6556 flood event stream power, riparian conifer cover, and precipitation. Negative correlations,  
6557 depending on sub basin, of wood volume and frequency with baseflow discharge, riparian woody  
6558 cover, watershed area, and large tree cover. Given the heterogeneous results across all sub-basins  
6559 studied, the authors conclude by emphasizing the importance of incorporating local data and  
6560 context when building wood models to inform future management decisions.

6561

## 6562 **Stream Temperature**

6563

6564 Hunter, 2010 (not in focal, treatments and results not relevant to questions)

6565

6566 Hunter, M.A., 2010. Water Temperature Evaluation of Hardwood Conversion Treatment Sites  
6567 Data Collection Report (Data Collection Report). Cooperative Monitoring, Evaluation, and  
6568 Research (CMER). Fp\_cmer\_05\_513

6569

6570 The purpose of this study is to evaluate the response of stream temperature to changes in canopy  
6571 cover using a before-after-control-impact design. This study took place along nine hardwood-  
6572 dominated riparian stands in Western Washington. Variables measured among locations and years  
6573 include riparian conditions, canopy cover, channel dimensions, substrate, flow and stream  
6574 temperature. Results from this study show that hardwood conversion buffers (HCB -  
6575 approximately 15 m width) intended to convert hardwood-dominated riparian areas to conifer-  
6576 dominated riparian areas usually resulted in decreased canopy cover of streams. Mean Global  
6577 Site Factor (GSF - the proportion of global radiation under a plant canopy relative to the amount  
6578 in an open area) increased in most study sites with HCB's. However, mean GSF did not change  
6579 substantially at sites with buffers closer to standard (~ 18 – 45 m) non-hardwood conversion  
6580 buffers. Temperature was highly variable over time and among locations suggesting stream  
6581 temperature is affected by many factors that might differ among locations and throughout time.  
6582 Longitudinal patterns of warming and cooling were consistent at all sites indicating the potential

6583 importance of careful site selection to account for changes in the longitudinal distribution of  
6584 temperatures.

6585

6586 **Influence of Stream Geomorphology on Water Temperature**

6587

6588 Hunter & Quinn, 2009

6589

6590 Hunter, M.A., Quinn, T., 2009. Summer Water Temperatures in Alluvial and Bedrock Channels  
6591 of the Olympic Peninsula. *Western Journal of Applied Forestry* 24, 103–108.  
6592 <https://doi.org/10.1093/wjaf/24.2.103>

6593

6594 The purpose of this study was to understand how stream geomorphology influences water  
6595 temperature in managed stands on the Olympic Peninsula, Washington. Sites chosen for this  
6596 included an alluvial study site and a bedrock study site whose overall characteristics were  
6597 otherwise comparable apart from geomorphology. The alluvial study site was a 1.6-km reach of  
6598 Thorndyke Creek. The bedrock study site was a 1.4-km reach of the South Fork Pysht River.  
6599 Both channels were located in 35–50-year-old managed forests dominated by Douglas-fir  
6600 (*Pseudotsuga menziesii*) in the uplands and red alder (*Alnus rubra*) in the riparian zone. Surface  
6601 substrate at the alluvial channel was composed mostly of gravel, whereas the bedrock channel  
6602 was composed of mostly bedrock, boulder, and cobble. The mean solar input (GSF: global site  
6603 factor) did not differ between streams. Water temperature was recorded at 75-m intervals along  
6604 each channel during the summers of 2003 and 2004. Results from this study show consistent  
6605 differences in stream temperature response in alluvial versus bedrock channels. Seasonal  
6606 maximum and minimum average daily temperatures varied less at the alluvial site compared to  
6607 the bedrock site. This, the authors suggest may be due to hyporheic exchange in alluvial channels  
6608 helping to buffer surface water temperatures from gaining or losing heat. In addition,  
6609 groundwater may also contribute to the increased stability at the alluvial site. Two same-day  
6610 measurements at each site showed the alluvial site gaining 8% of its flow, as compared to the  
6611 bedrock site whose flow decreased by approximately 15%. The bedrock site was also shown to  
6612 have the highest variation in reach-scale water temperatures during low flow. The authors  
6613 conclude that stream geomorphology may have profound impacts on spatial and temporal  
6614 patterns of channel water temperature. The authors suggest temperature reading from a single  
6615 location may not accurately represent the entire channel. Additional research involving collection  
6616 of temporal and longitudinal data will be needed to tailor riparian buffers to channel type.

6617

6618 **Stream temperature, sediment, nutrient**

6619



6620 Murray et al., 2000

6621

6622 Murray, G. L. D., Edmonds, R. L., & Marra, J. L. (2000). Influence of partial harvesting on  
6623 stream temperatures, chemistry, and turbidity in forests on the western Olympic Peninsula,  
6624 Washington. *Northwest science.*, 74(2), 151-164. Handle: <https://hdl.handle.net/2376/1065>

6625

6626 This study investigates the effects of partial watershed harvest (7-33%) on stream temperature,  
6627 chemistry, and turbidity relative to an unharvested old-growth watershed in the western Olympic  
6628 Peninsula, Washington. Both harvested watersheds (Rock and Tower creeks) originally contained  
6629 old-growth forests. Rock Creek had 7% of its watershed harvested in 1981, and Tower Creek had  
6630 33% of its watershed harvested between 1985 and 1987. Logging extended to the stream edge  
6631 near the in-stream monitoring sites. Data for stream daily maximum, minimum, and mean  
6632 temperatures, chemistry, and turbidity was recorded and monitored from June 1996 to June 1998  
6633 (10-15 years post-harvest). Differences in variables between treatment and reference watersheds  
6634 were compared with a one-way ANOVA with a posthoc Tukey HSD test. Results showed higher  
6635 maximum summer stream temperatures (15.4 °C), and lower winter maximum stream  
6636 temperatures (3.7 °C) in the two treatment watersheds compared to the unharvested reference  
6637 watershed (12.1 °C and 6.0 °C for summer max, and winter max, respectively). Winter minimum  
6638 temperatures for one of the harvested watersheds reached 1.2 °C (Rock Creek) compared to a  
6639 winter minimum of 6 °C Thus, seasonal variation of stream maximum temperatures and winter  
6640 minimum temperatures were more extreme in the treatment watershed than in the control. There  
6641 were no seasonal patterns or significant differences between watersheds in stream chemistry  
6642 except for calcium and magnesium concentrations being consistently higher in the unharvested  
6643 watersheds. Turbidity was low and not significantly different between watersheds. The authors  
6644 interpret these results as evidence of partial harvest having minimal impact on stream  
6645 temperatures, chemistry, and turbidity long-term (after 10-15 years). The stream temperature  
6646 changes were significant but did not exceed the 16 °C threshold used as a standard for salmonid  
6647 habitat. However, there was no data collection during the first decade following harvest.

6648

6649 **Channel Habitat, Particle Size, Stream Temperature, and Woody Debris Response to**  
6650 **Harvest**

6651

6652 Jackson et al., 2001

6653

6654 Jackson, C.Rhett., Sturm, C.A., Ward, J.M., 2001. Timber Harvest Impacts on Small Headwater  
6655 Stream Channels in the Coast Ranges of Washington1. *JAWRA Journal of the American Water*  
6656 *Resources Association* 37, 1533–1549.

6657 <https://doi.org/10.1111/j.1752-1688.2001.tb03658.x>

6658

6659 The purpose of this study was to evaluate changes in stream temperature, particle size  
6660 distributions of bed material, and channel habitat distributions in 15 first- or second order  
6661 streams located on the Coast Range of Western Washington. Four of the fifteen stream basins  
6662 were not harvested and served as references; three streams were cut with unthinned riparian  
6663 buffers; one with a partial buffer; one with a buffer of non-merchantable trees; and six were  
6664 clearcut to the stream edge. Buffer widths varied by operation; the average buffer width varied  
6665 from 15 – 21 meters. The narrowest buffer measured on one side of the stream was 2.3 meters.  
6666 Data for woody debris, sediment concentrations, turbidity, and stream temperatures were  
6667 recorded for one-year prior to harvest (1998). Harvest was conducted in the spring and early  
6668 summer of 1999, and post-harvest data was collected for about a month after operations were  
6669 complete. Thus, the results presented in this study represent changes in stream attributes and  
6670 characteristics immediately following harvest. Results from this study show that logging without  
6671 buffers had immediate and dramatic effects on channel morphology. Without buffers, and the  
6672 relatively steep topography of the study sites logging debris tended to accumulate at the bottom  
6673 of slopes thereby burying or covering many headwater streams. Covered channels were defined  
6674 in this study as having flow completely obscured by organic debris, but a recognizable channel  
6675 still exists below the debris. Buried channel was defined as having so much organic detritus in  
6676 the flow cross-section that the channel was no longer definable. Needles, twigs, whole branches,  
6677 and logs buried headwater streams with a mean depth of 0.94 meters of organic debris (range:  
6678 0.5 - 2.0 meters). Of the clearcut streams the percent of stream buried with organic matter ranged  
6679 from 6 to 90%, and the percent covered by organic matter ranged from 8 to 85%. The sum of  
6680 buried and covered for each stream ranged from 72 to 100%. On the other hand, most buffered  
6681 streams had 0% covered or buried by organic matter post-harvest with the only exception being  
6682 one stream that experienced blowdown post-harvest that covered 29% of the stream. While  
6683 debris accumulation tended to protect streams from the effects of solar radiation, organic logging  
6684 debris was also shown to trap fine sediment in the channels which, in the near term, greatly  
6685 reduced downstream sediment movement. As a result of increased roughness and additional bank  
6686 failures within the clearcut sites, sediment size shifted towards finer particles growing from 12 to  
6687 44 percent. In contrast, particle size distributions continued nearly unchanged in buffered and  
6688 reference sites. In the first summer after logging, significant increases were detected in overall  
6689 macroinvertebrate densities, collector densities, shredder abundance and biomass, and organic  
6690 and inorganic matter accretion. However, these responses were not detected one year following  
6691 logging. For stream temperature changes, because the data collection was for such a short period  
6692 of time (1-year pre- and 1-month post-harvest), and because the summer of 1999 was much  
6693 cooler than 1998, the assessment of harvest effects on stream temperature changes was difficult.  
6694 Thus, to interpret significant changes in stream temperatures from pre- to post- harvest, daily  
6695 maximum temperatures were plotted against the appropriate reference stream, and a regression  
6696 equation was calculated. The slopes of the regression lines were compared with a student's t-test  
6697 to determine significant differences. Of the seven clearcut streams, three showed no significant  
6698 changes in temperature, one became cooler (-1.1 °C), one became slightly warmer (+0.8 °C), and

6699 the other 2 became warmer or colder depending on location with decreases in temperature  
6700 upstream (-2.2 and -1.7 °C) and increases in temperature downstream (+5.2 and +15.1 °C). The  
6701 buffered streams had significant but less dramatic changes in temperature with one decreasing in  
6702 temperature (-0.3 °C), and 2 increasing in temperature (+1.6 and +2.4 °C). The one site with the  
6703 non-merchantable buffer had much higher temperature increases (+3.7 and +6.6 °C). The authors  
6704 posit that sites which retained riparian buffers succeeded in keeping debris out of streams as well  
6705 as served to protect streambanks from failure or erosion. Some mature trees left within buffers  
6706 experienced blow down and spanned the channel. While the clearcut streams had nearly all  
6707 canopy cover removed, the buildup of slash and LW in the stream also provided shade and  
6708 insulation that caused reductions in stream temperatures, or slight increases with one exception  
6709 (+15.1 °C) The authors point out that this study only served to point out immediate effects of  
6710 logging on physical channel conditions. Although important, there are still many questions about  
6711 how channel conditions will evolve over time.

6712

6713 **LW**

6714

6715 Meleason et al., 2003

6716

6717 Meleason, M. A., Gregory, S. V., & Bolte, J. P. (2003). Implications of riparian management  
6718 strategies on wood in streams of the Pacific Northwest. *Ecological Applications*, 13(5), 1212-  
6719 1221. <https://doi.org/10.1890/02-5004>

6720

6721 This study used simulation modeling to evaluate the potential effects of three different riparian  
6722 and watershed harvest scenarios on the standing stock of large wood in a hypothetical stream in  
6723 the Pacific Northwest. The three scenarios involved harvest 1) clearcut to the streambank, 2)  
6724 riparian management buffer widths ranging from 6-75 m, and 3) riparian buffers of various  
6725 widths with upland forest plantation. The effects of each scenario on wood load dynamics were  
6726 simulated with OSU STREAMWOOD for four harvest rotation periods (no harvest, 60, 90, and  
6727 120 years) over the course of 720 years. Results for scenario one (clear-cut to stream) showed  
6728 minimal accumulation of wood into the stream with little change over time due to the lack of a  
6729 forested riparian management zone. Results for scenario two showed the maximum standing  
6730 stock of in-stream wood loads required  $\geq 30$  m no-cut buffer zones for 500-year-old forests.  
6731 Wood loads in streams with 6 m wide buffers showed 32% of standing wood load stocks after  
6732 240 years. Results from scenario three showed minimal amounts of wood contributed into  
6733 streams from forest plantations when  $> 10$  m wide buffers were used. The authors interpret these  
6734 results as evidence that riparian buffer widths and forest age are more important for estimating  
6735 changes in wood loads over time than the harvest rotation age of plantation forests.

6736

6737 **LW**

6738

6739 Martin & Grotefendt, 2007

6740

6741 Martin, D. J., & Grotefendt, R. A. (2007). Stand mortality in buffer strips and the supply of  
6742 woody debris to streams in Southeast Alaska. *Canadian Journal of Forest Research*, 37(1), 36-49.  
6743 <https://doi.org/10.1139/x06-209>

6744

6745 This study compared riparian stand mortality and in-stream LW recruitment characteristics  
6746 between riparian buffer strips with upland timber harvest and riparian stands of unharvested  
6747 watersheds using aerial photography. This study was conducted in the northern and southern  
6748 portions of Southeast Alaska at multiple sites in nine timber harvest areas. All study sites were  
6749 along moderate- and low-gradient streams with channel widths ranging from 5 m to 30 m wide.  
6750 All buffer strips were conifer dominated and a minimum of 20 m wide that included selective  
6751 harvest within the 20 m zone. Reference sites were along unharvested reaches in the same area.  
6752 Stand mortality was estimated by the proportion of downed trees within a buffer strip.  
6753 Differences in downed tree proportions relative to reference streams were assumed to be caused  
6754 by timber harvest, accounting for selective in-buffer harvests. A one-tailed paired t-test or a  
6755 Wilcoxon signed rank test was used to check for statistical differences between treatment and  
6756 reference sites. Results showed significantly higher mortality (based on cumulative stand  
6757 mortality: downed tree counts divided by standing tree counts + downed tree counts),  
6758 significantly lower stand density (269 trees/ha in buffer units and 328 trees/ha in reference units),  
6759 and a significantly higher proportion of LW recruitment from the buffer zones of the treatment  
6760 sites than in the reference sites. Densities within all units ranged from 0 – 1334 trees/ha  
6761 depending on location. Overall, mean stand density in the buffer units was 18% lower than in the  
6762 reference units. Results also showed that mortality varied with distance to the stream.  
6763 Differences in mortality for the treatment sites were similar to the reference sites for the first 0-  
6764 10 m from the stream (only a 22% increase in the treated sites). However, mortality in the outer  
6765 half of the buffers (10-20 m) from the stream in the treatment sites was more than double (120%  
6766 increase) what was observed in the reference sites. This caused a change in the LW recruitment  
6767 source distance curves, with a larger proportion of LW recruitment coming from greater  
6768 distances in logged watersheds. LW recruitment based on the proportion of stand recruited (PSR)  
6769 was significantly higher in the buffered units compared to the reference units. However, PSR  
6770 from the inner 0-20 m was only 17% greater in the buffer units than in the reference units, while  
6771 PSR of the outer unit (10 – 20 m) was more than double in the buffered units than in the  
6772 reference units. The researchers conclude that the increase in mortality was caused by an  
6773 increased susceptibility to windthrow. They estimate that future recruitment potential from the  
6774 logged sites diminished by 10% relative to the unlogged reference sites.

6775

6776 **Stream temperatures**

6777

6778 Macdonald et al., 2003b

6779

6780 Macdonald, J. S., MacIsaac, E. A., & Herunter, H. E. (2003). The effect of variable-retention  
6781 riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest  
6782 ecosystems of British Columbia. *Canadian journal of forest research*, 33(8), 1371-1382.  
6783 <https://doi.org/10.1139/x03-015>

6784

6785 This study investigates the impacts of forest harvest on stream temperatures under three variable  
6786 retention buffer treatments in headwater streams of the interior sub-boreal forests of British  
6787 Columbia. Temperature data were recorded for two years pre- and five years post-harvest from  
6788 five harvested streams and two unharvested reference streams. Differences between pre- and  
6789 post-harvested stream temperatures were compared with the paired reference streams using  
6790 repeated measures ANOVA. Treatment riparian areas were harvested with the following  
6791 prescriptions: 1) low-retention – removal of all merchantable timber >15 or >20 cm DBH for  
6792 pine or spruce respectively, within 20 m of the stream 2) high-retention – removal of  
6793 merchantable timber >30 cm DBH within 20-30 m of the stream, and 3) Patch-cut – high  
6794 retention for the lower 60% of watershed approaching streams and removal of all vegetation in  
6795 the upper 60% of the watershed. Eight first-order streams were included in this study: two

6796 in the Gluskie Creek watershed (G5, G7) and six in the Baptiste Creek watershed (B1–B6). Five  
6797 of these streams were within the harvested boundaries (2 high-retention, 2 low-retention, and 1  
6798 patch cut), and 3 reaches outside of the harvest boundary served as controls. Results showed a  
6799 significant increase in stream temperatures ranging from 4 – 6 °C at five years post-harvest, and  
6800 increased ranges of diurnal temperature fluctuations for all treatment streams relative to the  
6801 reference streams. Streams that had summer maximum mean weekly temperatures of 8°C before  
6802 harvesting had maximum temperatures near 12°C or more following harvesting. Daily ranges of  
6803 1.0–1.3°C before harvesting became 2.0–3.0°C following harvesting, Greater temperature ranges  
6804 occurred in low-retention and patch treatments than the high-retention or control treatment  
6805 locations. The high-retention buffer treatment mitigated temperature increases for the first three  
6806 years. Still, increased mortality (windthrow) caused a reduction in the canopy that increased  
6807 stream temperatures equivalent to other treatment streams by year five. The results of this study  
6808 show evidence that high-retention buffers are no more effective in preserving stream temperature  
6809 changes than small retention buffers when treatment areas have a high susceptibility to  
6810 windthrow.

6811

6812 **Sediment delivery pathways**

6813

6814 Litschert & MacDonald, 2009

6815

6816 Litschert, S. E., & MacDonald, L. H. (2009). Frequency and characteristics of sediment delivery  
6817 pathways from forest harvest units to streams. *Forest Ecology and Management*, 259(2), 143-  
6818 150. <https://doi.org/10.1016/j.foreco.2009.09.038>

6819

6820 This study investigates the frequency of sediment delivery pathways (“features”) in riparian  
6821 management areas and measures the physical characteristics and connectivity of these pathways  
6822 following timber harvest. The results of this study were then used to develop models for  
6823 predicting the length and connectivity of pathways formed from harvest units. Data was collected  
6824 from over 200 harvest units with riparian management areas in the Eldorado, Lassen, Plumas,  
6825 and Tahoe National Forests in the Sierra and Cascade mountains of northern California. Riparian  
6826 buffer widths for this area are 90 m and 45 m for perennial and annual streams respectively. No  
6827 machinery is allowed in the riparian management areas. Data collected and analyzed for the  
6828 pathways included years since harvest, mean annual precipitation, soil depth, soil erodibility,  
6829 hillslope gradient, aspect, and elevation. Characteristics of pathway length, gradient, and  
6830 roughness were also collected. Relationships between site variables and pathway variables were  
6831 assessed using linear regression. The site variables with the most significant relationships with  
6832 the pathway variables were used in a multivariate regression model to predict pathway length.  
6833 Only 19 of the 200 harvest units had sediment development pathways. Pathways ranged in age  
6834 (time since harvest) from 2 to 18 years, and in length from 10 m to 220 m. Of the 19 pathways,  
6835 only six were connected to streams, and five of those originated from skid trails. Pathway length  
6836 was significantly related to mean annual precipitation, cosine of the aspect, elevation, and  
6837 hillslope gradient. The authors conclude that timber prescription practices for these National  
6838 Forests are effective in reducing sediment delivery pathways. The authors interpret these results  
6839 as evidence that skid trails should be directed away from streams, maintaining surface roughness,  
6840 and promptly decommissioning skid trails.

6841

6842 **LW**

6843

6844 Liquori, 2006

6845

6846 Liquori, M. K. (2006). POST-HARVEST RIPARIAN BUFFER RESPONSE: IMPLICATIONS  
6847 FOR WOOD RECRUITMENT MODELING AND BUFFER DESIGN 1. *JAWRA Journal of the*  
6848 *American Water Resources Association*, 42(1), 177-189. [https://doi.org/10.1111/j.1752-](https://doi.org/10.1111/j.1752-1688.2006.tb03832.x)  
6849 [1688.2006.tb03832.x](https://doi.org/10.1111/j.1752-1688.2006.tb03832.x)

6850

6851 This study investigates the differences in treefall characteristics in riparian management areas  
6852 based on ecological and physiographic variables to give insight on the variables important for  
6853 wood recruitment modeling. Data were collected from 20 riparian buffer sites that had all been  
6854 clearcut within three years of sampling with standard no-cut buffers 25 ft. An additional 50-100  
6855 ft buffer was applied to fish-bearing streams depending on stream type, in a managed tree farm in  
6856 the Cascade Mountains of western Washington. These riparian buffers generally consisted of  
6857 naturally regenerated, second-growth conifer stands about 45 to 70 years old. “Very modest”  
6858 thinning was applied to some stands to meet wildlife objectives and any downed wood not  
6859 affecting the channel was removed. Tree characteristic data collected included tree size (DBH  
6860 and height), species, fall direction, tree fall angles, estimated cause of mortality, and distance to  
6861 the stream. Site characteristics included stream gradient, valley morphology, and time since  
6862 harvest. Tree recruitment probability curves were developed as a function of tree height using  
6863 methods described by Beschta, (1990). Results showed that wind-caused mortality and tree fall  
6864 rates were significantly higher, up to three times higher, than competition-induced mortality  
6865 within buffers for three years following treatment. The median observed treefall per site was  
6866 15% of all trees in each buffer, ranging from 1 to 57%. total treefall at each site for one, two, and  
6867 three years since harvest was  $16 \pm 10\%$ ,  $28 \pm 21\%$ , and  $10 \pm 10\%$ , respectively. Total treefall  
6868 percentage for each site was not correlated to years since harvest (Spearman  $R = 0.11$ ;  $p = 0.34$ ).  
6869 The mean and standard deviation of the total normalized treefall for one-year old sites was  $405 \pm$   
6870  $394$  trees/km ( $n = 9$ ), for two-year old sites was  $264 \pm 280$  trees/km ( $n = 7$ ), and for three-year  
6871 old sites was  $556 \pm 316$  trees/km ( $n = 4$ ). Treefall varied significantly by species. Downed red  
6872 alder (*Alnus rubra*), western red cedar (*Thuja plicata*), and Douglas-fir (*Pseudotsuga menziesii*)  
6873 comprised 3 percent to 8 percent of all downed trees; these species had treefall rates ranging  
6874 from 5 percent to 9 percent of the total number of trees of the same species. By contrast, treefall  
6875 rates for western hemlock (*Tsuga heterophylla*) and Pacific silver fir (*Abies amabilis*) ranged  
6876 from 23 percent to 26 percent. Treefall rates also varied somewhat by size, with the 31 to 41 cm  
6877 (12 to 16 in) diameter class having the greatest treefall rates (All trees were grouped into size  
6878 classes based on diameter at breast height: 1 to 8 in; 8 to 12 in; 12 to 16 in; 16 to 20 in; and more  
6879 than 20 in). Treefall following harvest greatly exceeded the expected competition induced  
6880 mortality rates (posited by Franklin, 1970) of 0.5%, and the model of average competition  
6881 mortality used in Rainville et al. (1985), which ranged from 0.7 - 1.6%, and 2% per year for bank  
6882 undercutting. Treefall direction was heavily biased towards the channel regardless of channel or  
6883 buffer orientation and tree fall probability was highest in the outer areas of the buffers (adjacent  
6884 to the harvest area). Fall direction bias increased significantly in the inner portions of the buffer.  
6885 Within the 0 to 7 m zone and 7 to 15 m zone, 68% and 67% of the trees, respectively, fell toward  
6886 the channel ( $n = 125$  and  $153$ , respectively). Only 44% of the outer zone ( $> 15$  m) downed trees  
6887 fell toward the channel ( $n = 403$ ). Generally, recruitment was negatively correlated to buffer  
6888 width ( $r^2 = 0.40$ ). Treefall was generally highest at the outside edges of buffers (50+ feet),  
6889 representing about 60% of the total observed treefall, while the 0–25-foot zone represented  
6890 ~18%, and the 25–50-foot zone represented ~22%. The authors interpret their results as evidence  
6891 that tree fall models that use a random fall direction may underrepresent the probability of LW  
6892 recruitment into streams. Further, they suggest that the increase in windthrow mortality and the  
6893 probability of tree fall with increasing distance from the stream should be considered.

6894

6895 **LW**

6896

6897 Lininger et al., 2021 (removed from focal list, this is a case study)

6898

6899 Lininger, K. B., Scamardo, J. E., & Guiney, M. R. (2021). Floodplain large wood and organic  
6900 matter jam formation after a large flood: Investigating the influence of floodplain forest stand  
6901 characteristics and river corridor morphology. *Journal of Geophysical Research: Earth Surface*,  
6902 126(6), e2020JF006011. <https://doi.org/10.1029/2020JF006011>

6903

6904 This study examines how river corridor morphology and forest stand density influence LW and  
6905 coarse particulate matter (CPOM) deposition patterns in the flood plain resulting from a 400-year  
6906 flood event in West Creek in the Colorado Front Range in 2013. The researchers tested the  
6907 hypothesis that if river corridor geomorphology affects LW and CPOM deposition then there  
6908 should be an inverse relationship between elevation above and distance from the stream's edge.  
6909 Further, that deposition frequency would be higher in unconfined portions of the corridor.  
6910 Considering forest stand structure, the researchers hypothesized that LW/CPOM jams would be  
6911 pinned by trees, higher in intermediate forest densities, and decrease in size with increasing  
6912 forest stand density. Field data of LW/CPOM jams were analyzed with non-parametric Spearman  
6913 correlation tests to determine the strength of their relationship with channel and stand  
6914 characteristics. Results showed support for most of the hypotheses. LW accumulations did  
6915 decrease in size with distance from the stream, but CPOM did not. Confined channels (steeper  
6916 reaches) contained fewer LW/CPOM loads per unit area. The authors speculate that these reaches  
6917 had higher flow rates and thus lower deposition during the flood. CPOM jams increased in  
6918 number per area with increasing stand density with most jams pinned against live trees. The  
6919 authors conclude that the effect of riparian forest stand density is evidence that riparian forests in  
6920 the floodplains should be preserved to increase LW and CPOM trapping probability.

6921

6922 **Stream Temperature**

6923

6924 Janisch et al., 2012

6925

6926 Janisch, J.E., Wondzell, S.M., Ehinger, W.J., 2012. Headwater stream temperature: Interpreting  
6927 response after logging, with and without riparian buffers, Washington, USA. *Forest Ecology and*  
6928 *Management* 270, 302–313. <https://doi.org/10.1016/j.foreco.2011.12.035>

6929



6930 The purpose of this study was to assess the stream temperature response to three different  
6931 harvesting treatments in small, forested headwater catchments in western Washington. The pre-  
6932 logging calibration period lasted 1–2 summers and stream temperatures were monitored for two  
6933 or more summers after logging. Harvest treatments occurred between September 2003 and July  
6934 2005; catchments were clustered by harvest year for analysis. A before-after-control-impact  
6935 study design was used to contrast stream temperature responses for three forest harvest  
6936 treatments: clearcut logging to the stream (n=5), a continuous buffer (n=6) with widths 10-15 m  
6937 on each side of the channel, and a patched buffered (n=5) where portions of the riparian forests  
6938 ~50-110 m long were retained in distinct patches along some portion of the channel with the  
6939 remaining riparian area clearcut. For the patch buffers there was no standard width, the buffer  
6940 spanned the full width of the floodplain area and extended well away from the stream. Upland  
6941 areas adjacent to buffers were clearcut. Regression relationships were developed between  
6942 temperatures measured in the treatments and corresponding reference catchments. A simple  
6943 ANOVA model was used that only included fixed effects for treatment, years since treatment,  
6944 and day of year. Because of the unbalanced experimental design and variation in time of harvest,  
6945 clustering of treatments caused the sample sizes to become too small to apply a more complex  
6946 nested, repeated measures ANOVA could not be used. Correlation analysis was conducted  
6947 between post-harvest stream temperatures and descriptive variables on a subset of catchments to  
6948 examine possible factors that might control post-harvest thermal responses. Results from this  
6949 study show significant increases in stream temperature in all treatments. Although temperature  
6950 responses were highly variable within treatments, July and August daily maximum temperatures  
6951 increased in clearcut catchments during the first year after logging by an average of 1.5°C (range  
6952 0.2 to 3.6°C), in patch-buffered catchments by 0.6°C (range – 0.1 to 1.2°C), and in continuously  
6953 buffered catchments by 1.1°C (range 0.0 to 2.8°C). Canopy cover in all streams averaged 95%  
6954 prior to harvest and did not differ between treatment and reference streams. Following treatment,  
6955 canopy cover in the clearcut catchments averaged 53%, canopy cover in the patch buffer  
6956 treatment averaged 76%, and canopy cover in the continuous buffer treatment averaged 86%.  
6957 Following treatment, the canopy cover of the clearcut and patch buffer treatments were  
6958 significantly lower than in the reference streams. The continuous buffer treatments did not differ  
6959 significantly from the reference streams for canopy cover. Further analyses which attempted to  
6960 identify variables responsible for controlling the extent of stream temperature responses showed  
6961 the amount of cover retained in the riparian buffer was not a strong explanatory variable. Post-  
6962 treatment temperature changes suggested that treatments ( $p = 0.0019$ ), the number of years after  
6963 treatment ( $p = 0.0090$ ), and the day of the year ( $p = 0.0007$ ) were all significant effects  
6964 explaining observed changes in temperature. Wetland area ( $r^2 = 0.96$ ,  $p < 0.01$ ) and length of  
6965 surface flow ( $r^2 = 0.67$ ,  $p = 0.05$ ) were strongly correlated with post-logging temperature  
6966 changes. Regression analysis of these variables showed streams with fine-textured substrates  
6967 responded differently than coarse textured substrates. The authors speculate this is possibly due  
6968 to groundwater interactions which can buffer thermal responses of small streams. In summary,  
6969 the authors conclude that their results suggest small headwater streams may be fundamentally  
6970 different than larger streams partly because factors other than canopy shade can greatly influence  
6971 stream energy budgets to moderate stream temperatures despite changes and/or removal of the  
6972 overstory canopy.

6973

6974 **Large wood**

6975

6976 Jones et al., 2011 (Removed from focal list, study not relevant to focal questions)

6977

6978 Jones, T.A., Daniels, L.D., Powell, S.R., 2011. Abundance and function of large woody debris in  
6979 small, headwater streams in the Rocky Mountain foothills of Alberta, Canada. *River Research*  
6980 *and Applications* 27, 297–311. <https://doi.org/10.1002/rra.1353>

6981

6982 The purpose of this study was to assess LW abundance in the upper foothills of the Rocky  
6983 Mountains in Alberta, Canada. This study also sought to understand key processes that underlie  
6984 changes in LW function. Finally, this study used results to develop a LW recruitment, decay and  
6985 interaction model. This research was conducted in 21 headwater streams spanning two  
6986 watersheds. At each site, all LW was sampled and was classified according to decay, orientation,  
6987 position and function. LW frequency, total volume, and total in-stream volume were calculated  
6988 and analyzed for differences using a one-way ANOVA followed by a Tukey post hoc test to  
6989 differentiate among significant classes. Results show LW frequency was greater in the Alberta  
6990 foothills ( $64.0 \pm 3.3$  LW 100 m<sup>1</sup>) than in many small, headwater streams in mountain ( $46.2 \pm 3.6$ ),  
6991 coastal ( $47.6 \pm 3.8$ ), mixed broad-leaf ( $47.0 \pm 4.2$ ) and boreal ( $31.0 \pm 3.0$ ) streams. This, the  
6992 authors suggest, is likely due to the narrow bankfull width channels characteristic of the Alberta  
6993 foothills which are less able to transport LW downstream. LW with  $\geq 20$  cm was more frequent in  
6994 coastal streams, and overall LW volume was also greatest in coastal streams ( $721.0 \pm 99.9$  m<sup>3</sup> ha<sup>-1</sup>).  
6995 The authors note that large LW volumes in coastal streams are likely due to geomorphic  
6996 disturbances alongside large, long-lived, decay resistant tree species. According to Harmon et al.  
6997 1986, much of the variation in LW recruitment is due to differences in species life history and  
6998 forest type which together govern log size and decay rates.

6999

7000 **Suspended Sediment**

7001

7002 Karwan et al., 2007

7003

7004 Karwan, D., Gravelle, J., Hubbart, J., 2007. Effects of timber harvest on suspended sediment  
7005 loads in Mica Creek, Idaho. *Forest Science* 53, 181–188.  
7006 <https://doi.org/10.1093/forestscience/53.2.181>

7007

7008 The purpose of this study was to examine the effects of forest road construction and timber  
7009 harvest on total suspended solids (TSS) in a forested watershed. This study took place at the  
7010 Mica Creek Experimental Watershed in northern Idaho. The study area consisted of dense,  
7011 naturally regenerated, even-aged stands ~65 years old and ~300 trees per acre. Timber harvesting  
7012 and heavy road use began in 2001. Treatments in the paired-watershed experiment consisted of  
7013 (1) commercial clearcut of the watershed area of 50%, and was broadcast burned and replanted  
7014 by the end of May 2003, (2) partial cut in which half the canopy was removed in 50% of the  
7015 watershed in 2001, with final 10% of log processing and hauling in early summer of 2002. and  
7016 (3) a no-harvest control. All harvests were carried out according to best management practices  
7017 and in accordance with the Idaho Forest Practices Act. At the time of the study this involved a  
7018 22.86 m (75 ft) stream protection zones (SPZs) on each side of fish-bearing (Class I) streams.  
7019 The inner 50 ft is an equipment exclusion zone where no ground-based skidding machinery is  
7020 allowed. Timber harvesting is allowed in Class I SPZs, but 75% percent of existing shade must  
7021 be retained. Along non-fish-bearing (Class II) streams, harvesting equipment was excluded from  
7022 entering within 9.14 m (30 ft) of definable stream channels and any cut trees were felled away  
7023 from the stream; however, there were no tree retention requirements. In the clearcut and partial  
7024 cut units, line skidding was used on slopes in the watershed exceeding approximately 20%, while  
7025 tractor skidding was used on the lower gradient slopes. On all skid trails, drainage features, such  
7026 as water bars, were installed for erosion control at the end of the harvest period. Time series data  
7027 were compiled for all measured TSS values from 1991 through 2004. Data was collected via  
7028 seven stream monitoring flumes located within the Mica Creek Watershed. Monthly TSS loads  
7029 were compared across watersheds for five time intervals: (1) pretreatment: ~6 years, (2)  
7030 immediate post-road construction: ~1 year, (3) recovery post-road construction: ~3 years, (4)  
7031 immediate post-harvest: ~1 year, and (5) recovery post-harvest: ~3 years. Trends in the  
7032 relationship between treatment and control watersheds were statistically examined for each of the  
7033 time intervals. Treatments in the paired-watershed experiment consisted of (1) commercial  
7034 clearcut of the watershed area of 50%, and was broadcast burned and replanted, (2) partial cut in  
7035 which half the canopy was removed in 50% of the watershed (3) a no-harvest control. All  
7036 harvests were done according to best management practices and the Idaho Forest Practices Act.  
7037 This included equipment exclusion zones of 50- and 30-feet for fish- and non-fish-bearing  
7038 streams, respectively. On all skid trails, drainage features, such as water bars, were installed for  
7039 erosion control at the end of the harvest period. Analysis of covariance was used for each  
7040 treatment-control watershed pair. Results show monthly TSS loads from watersheds 1 (clearcut),  
7041 2 (partial cut), and 3 (no-harvest) ranged from 0.4 kg km<sup>-2</sup> to above 10,000 kg km<sup>-2</sup>, with a  
7042 maximum in the spring months and minimum in the winter and late summer months similar to  
7043 intra-annual trends in water yield. Road construction in both watersheds did not result in  
7044 statistically significant impacts on monthly sediment loads in either treated watershed during the  
7045 immediate or recovery time intervals. A significant and immediate impact of harvest on monthly  
7046 sediment loads in the clear-cut watershed ( $p = 0.00011$ ), and a marginally significant impact of  
7047 harvest on monthly sediment loads in the partial-cut ( $p = 0.081$ ) were observed. Total sediment  
7048 load from the clearcut over the immediate harvest interval exceeded predicted load by 152%  
7049 (6,791 kg km<sup>-2</sup>); however, individual monthly loads varied around this amount. The largest  
7050 increases in percentage and magnitude occurred during snowmelt months, namely April 2002

7051 (560%, 2,958 kg km<sup>-2</sup>) and May 2002 (171%, 3,394 kg km<sup>-2</sup>). Neither treatment showed a  
7052 statistical difference in TSS during the recovery time (clearcut: p = 0.2336; partial-cut: p =  
7053 0,1739) compared to calibration loads (pre-treatments). The authors conclude that best  
7054 management practices for road construction, including improvement of existing roads, did not  
7055 produce significant changes in TSS. Significant changes in TSS only occurred immediately after  
7056 harvest. However, after one year, the TS load became statistically indistinguishable from the  
7057 control.

7058

### 7059 **Harvest effects on Instream light**

7060

7061 Kaylor et al., 2017

7062

7063 Kaylor, M.J., Warren, D.R., Kiffney, P.M., 2017. Long-term effects of riparian forest harvest on  
7064 light in Pacific Northwest (USA) streams. *Freshwater Science* 36, 1–13.  
7065 <https://doi.org/10.1086/690624>

7066

7067 The purpose of this study was to evaluate relationships between riparian forest stand age and  
7068 stream light availability. The specific goals dealt with evaluating characteristics of late-  
7069 successional forest light regimes, and whether canopy openness and light differed between  
7070 streams flowing through harvested units and late-successional forest units. This study took place  
7071 at the HJ Andrews Experimental Forest in the Cascade Mountain, Oregon. Approximately 11.5  
7072 km of stream length were sampled in the McCrae Basin which consists mostly of old-growth  
7073 forests Douglas-fir forests with small patch clear cuts. All treatment sites were harvested within  
7074 50 to 60 years before the study. Clearing up to both stream banks occurred at two of seven  
7075 treated sites and clearing up to one bank occurred on all other treated sites. Stream bank-full  
7076 width, wetted width, canopy openness, % red alder, and estimated photosynthetically active  
7077 radiation (PAR) were quantified at 25-m intervals to evaluate relationships between channel and  
7078 riparian characteristics and stream light. Results from this study show mean estimated PAR  
7079 reaching the streams was lower in the recovering harvested units (50-year post-treatment) than  
7080 in up and downstream reaches bordered by old growth for all comparisons (n=14), while only 6  
7081 were significant (p<0.05). All in all, old growth reaches averaged 1.7 times greater PAR values  
7082 than in nearby harvested units with the greatest differences occurring when harvest was  
7083 implemented on both banks. Mean canopy openness was higher in late-successional forests (>  
7084 300 years old) than in young second growth forests (30–100-year-old forests), 18% and 8.7%  
7085 respectively. Results also indicate the relationship between canopy openness and PAR was  
7086 stronger at the reach scale than at individual locations with mean canopy openness explaining  
7087 78% of the variance in mean PAR estimates. The researchers also conducted a review of  
7088 available literature of studies that contained information on the effects of Northwest Douglas-fir  
7089 forest growth dynamics on canopy cover and light availability. The researchers concluded from

7090 this review that canopy closure, and thus lower light availability, occurs approximately 30 years  
7091 after growth and maintained until after 100 years of growth when the canopy structure begins to  
7092 open and produce gaps. Altogether, this study suggests stream light regimes are affected by  
7093 initial canopy removal and subsequent recovery. Depending on forest type, dominant species and  
7094 the age of the stand, different stages of stand development may reflect complex overstory  
7095 structures allowing variable levels of light to the stream.

7096

## 7097 **Stream Temperatures**

7098

7099 Kibler et al., 2013

7100

7101 Kibler, K.M., Skaugset, A., Ganio, L.M., Huso, M.M., 2013. Effect of contemporary forest  
7102 harvesting practices on headwater stream temperatures: Initial response of the Hinkle Creek  
7103 catchment, Pacific Northwest, USA. *Forest Ecology and Management* 310, 680–691.

7104 <https://doi.org/10.1016/j.foreco.2013.09.009>

7105

7106 The purpose of this study was to investigate the effects of contemporary forest harvesting  
7107 practices on headwater stream temperatures using a BACI design. This study was conducted as  
7108 part of the Hinkle Creek paired Watershed Study (HCPWS). This study consisted of a nested,  
7109 paired watershed study in which harvesting treatments in accordance with the Oregon Forest  
7110 Practices Act (FPA) were applied to four headwater catchments in southern Oregon. Oregon FPA  
7111 does not require retention of fixed-width buffer strips adjacent to non-fish-bearing streams. Thus,  
7112 as a part of the harvest activities, fixed-width buffer strips containing merchantable overstory  
7113 conifers were not left adjacent to the non-fish-bearing streams. Clearcut harvest took place  
7114 between August 2005 and May 2006. Streamflow and temperature were measured at 8 locations  
7115 within the basin from autumn 2002 until autumn of 2006 giving 3 years of pre-harvest data and  
7116 <1 year of post-harvest data. Treatment and reference catchments were paired based on similarity  
7117 in catchment area, aspect, stream orientation, stream length, and discharge. Significant  
7118 differences between pre- and post-harvest daily max temperature measurements were detected  
7119 across all sites, however, magnitude and direction of changes were inconsistent. Results for daily  
7120 mean maximum stream temperatures show a variable response across all four harvested streams  
7121 ranging from 1.5°C cooler to 1.1°C warmer relative to pre-harvest years. No statistically  
7122 significant changes in max, mean, or minimum daily stream temperatures to timber harvest were  
7123 observed. The authors suggest possible explanations for lack of consistent temperature increases  
7124 to shading provided by logging slash. Interestingly, statistically significant changes to  
7125 relationship between treatment and reference site pairs with respect to minimum and mean  
7126 stream temperatures resulted in decreased minimum daily stream temperatures on days where  
7127 high temperatures were observed in reference streams. At one treatment site, mean minimum  
7128 temperatures across the warm season decreased 1.9°C relative to pre-harvest years, and the

7129 minimum temperature on the warmest day decreased by 2.8°C relative to pre-harvest years.  
7130 Except for one treatment-reference pair, highly significant changes to slope and intercept  
7131 parameters of minimum daily stream temperatures were detected for each stream pair ( $p < 0.001$ ).  
7132 The authors suggest decreases in daily minimum stream temperature is a likely consequence of  
7133 timber harvest.

7134

### 7135 **Shade and Stream temperature**

7136

7137 Cupp & Lofgren, 2014

7138

7139 Cupp, C.E. & Lofgren, T.J. (2014). Effectiveness of riparian management zone prescriptions in  
7140 protecting and maintaining shade and water temperature in forested streams of Eastern  
7141 Washington. Cooperative Monitoring Evaluation and Research Report CMER 02-212.  
7142 Washington State Forest Practices Adaptive Management Program. Washington Department of  
7143 Natural Resources, Olympia, WA.

7144

7145 The purpose of this study was to assess the percent reduction in canopy cover, and the response  
7146 in stream temperatures following riparian timber harvest under the “all available shade” rule  
7147 (ASR), and the standard rule (SR) in eastern Washington. The ASR is applied to areas in the Bull  
7148 Trout Habitat Overlay (BTO; map of bull trout habitat) that requires retention of all available  
7149 shade within 75 feet of the stream. Under the standard shade rule (SR) some harvest is allowed  
7150 within the 75-foot buffer depending on elevation and pre-harvest canopy cover. The primary  
7151 objectives of this study were to (1) Quantify and compare differences in post-harvest canopy  
7152 closure between the SR and the ASR riparian prescriptions of eastern Washington; and (2)  
7153 Quantify and compare differences in stream temperature effects of the two riparian prescriptions:  
7154 the SR and the ASR. This study was conducted at 30 sites in eastern Washington. Sites were  
7155 between 65-100 years old and were situated along second to fourth order streams with harvest-  
7156 regenerated or fire-regenerated forests. Reference reaches were located upstream from treatment  
7157 reaches where harvest was applied. Eighteen sites were located on state owned and managed  
7158 forests and 12 sites were located on private industrial forests. Prior to harvest treatments, canopy  
7159 closure measurements ranged from 89% to 97%, with a mean of 93%. The riparian management  
7160 zone (RMZ) consists of three zones: The core zone is nearest to the edge of the stream and  
7161 extends out 30 feet horizontally from the bankfull edge or outer edge of the channel migration  
7162 zone (CMZ), whichever is greater. The inner zone is situated immediately outside of the core  
7163 zone. For streams with a bankfull width of less than or equal to 15 feet wide, the inner zone  
7164 width is 45 feet wide. All streams assessed in this study were less than or equal to 15 feet wide.  
7165 The outer zone of the RMZ is the zone furthest from the water and its width varies according to  
7166 stream width and site class for the land. The specific site class (a measure of site productivity) at  
7167 each treatment site would vary the outer zone width from 0 to 55 feet wide. Seven sites had up to

7168 four years pre-harvest temperature data with only two years post-harvest data. Nine sites had  
7169 three years pre-harvest data and one site had only one year pre-harvest data. The remaining 13  
7170 sites had two years pre-harvest data. Following harvest treatments, all 30 sites had at least two  
7171 years post-harvest temperature data collection, although 21 of the 30 sites had at least three years  
7172 post-harvest monitoring. Data collection included twice hourly stream and air temperature data  
7173 during each sample period. Canopy, shade, riparian, and channel data were collected during the  
7174 first-year pre-harvest and the first year post-harvest. Stream temperature data were collected at  
7175 30-minute intervals between 1 July and 15 September for a total of 77 days each year a site was  
7176 investigated. Stream canopy closure and shade were quantified at 75-ft intervals within each  
7177 reach using a hand-held densiometer (for canopy closure measurements) and a self-leveling  
7178 fisheye lens digital camera (for shade measurements). A t-test was used to evaluate differences in  
7179 pre-harvest canopy cover between reference and treatment reaches, and between ASR and SR  
7180 sites. A correlation analysis between post-harvest change in shade and the descriptive riparian  
7181 and channel values (e.g., trees per acre, basal area, channel gradient, etc.) was also used to  
7182 examine possible factors that may control post-harvest changes in shade. A linear mixed effects  
7183 model was used to quantify and compare differences in daily max stream temperatures (DMAX)  
7184 between no harvest, ASR and SR prescriptions. Results showed post-harvest shade values  
7185 decreased in SR sites (mean effect of -2.8%,  $p = 0.002$ ), as did the canopy closure values (mean  
7186 effect of -4.5%,  $p < 0.001$ ). Shade and canopy closure values did not significantly change in the  
7187 treatment reaches of the ASR sites. Mean shade reduction in the SR treatment sites exceeded the  
7188 mean shade reduction in the ASR sites by 3%. Canopy closure reduction was also greater in the  
7189 SR sites than in the ASR sites by a mean of 4%. Specifically, the mean shade reduction in ASR  
7190 sites was 1% with a maximum reduction of 4%. The mean reduction of shade in the SR sites was  
7191 4% with a maximum reduction of 10%. Mean shade contribution of upland trees (trees outside of  
7192 the RMZ) per study site was calculated as  $< 1\%$ . Shade reduction levels did not differ between  
7193 the sites receiving RMZ-harvest only and the sites receiving standard operational upland harvest.  
7194 Site seasonal means of daily maximum stream temperature treatment responses in the first two  
7195 years following harvest ranged from  $-0.7\text{ }^{\circ}\text{C}$  to  $0.5\text{ }^{\circ}\text{C}$  in the ASR reaches and from  $-0.3$  to  $0.6$  in  
7196 the SR reaches. Site seasonal mean post-harvest background responses in reference reaches  
7197 ranged from  $-0.5\text{ }^{\circ}\text{C}$  to  $0.6\text{ }^{\circ}\text{C}$  in the first two years following harvest. Mean daily maximum  
7198 stream temperature increased  $0.16\text{ }^{\circ}\text{C}$  in the SR harvest reaches, whereas stream temperatures in  
7199 both the ASR sites and in the no-harvest reference reaches increased on average by  $0.02\text{ }^{\circ}\text{C}$ .  
7200 Seasonal mean stream temperature responses of up to  $0.5\text{ }^{\circ}\text{C}$  in the no-harvest references were  
7201 common during the post-harvest test period. Sample period means of daily maximum  
7202 temperature responses varied from  $-1.1\text{ }^{\circ}\text{C}$  to  $0.7\text{ }^{\circ}\text{C}$  in the first two years post-harvest for the  
7203 ASR sites, from  $-0.5$  to  $0.8\text{ }^{\circ}\text{C}$ , in the SR sites, and  $-0.5$  to  $0.9\text{ }^{\circ}\text{C}$  in the reference sites. The  
7204 authors interpret these results as evidence that temperature effects of the SR, and ASR were  
7205 similar to reference conditions along sampled reaches for small streams in the mixed fir zone  
7206 mid-successional forests of eastern Washington. Further, that processes not directly related to  
7207 canopy cover alteration over streams may be primarily responsible for the small variations  
7208 observed in stream temperatures following harvest.

7209

7210

7211 Ehinger et al., 2021

7212

7213 Ehinger, W.J., W.D. Bretherton, S.M. Estrella, G. Stewart, D.E. Schuett-Hames, and S.A. Nelson.  
7214 2021. Effectiveness of Forest Practices Buffer Prescriptions on Perennial Non-fish-bearing  
7215 Streams on Marine Sedimentary Lithologies in Western Washington. Cooperative Monitoring,  
7216 Evaluation, and Research Committee Report CMER 2021.08.24, Washington State Forest  
7217 Practices Adaptive Management Program, Washington Department of Natural Resources,  
7218 Olympia, WA.

7219

7220 The purpose of this study was to assess the effectiveness of riparian management zone  
7221 prescriptions in maintaining functions and processes in headwater perennial, non-fish-bearing  
7222 streams in incompetent (easily eroded) marine sedimentary lithologies in western Washington.  
7223 Specifically, this study used a multiple before after control impact (MBACI) design to compare  
7224 unharvested reference sites to sites harvested under the western Washington Forest Practices for  
7225 non-fish-bearing streams to assess the effects of these rules on riparian vegetation and wood  
7226 recruitment, canopy closure and stream temperature, stream discharge and downstream transport  
7227 of suspended sediment and nitrogen, and benthic macroinvertebrates. The Forest Practices rules  
7228 for non-fish-bearing streams in the study area includes clearcut harvest with a two-sided 50-foot-  
7229 wide riparian buffer along at least 50% of the riparian management zone, including buffers  
7230 prescribed for sensitive sites and unstable slopes. Due to the additional unstable slope buffers,  
7231 total buffer area was 18 to 163% greater than the current minimum buffer rule. Ten study sites  
7232 were chosen with first-, second-, and third-order non-fish-bearing streams. Data was collected  
7233 for 1-2 years of pre-harvest, during the harvest period (2012 – 2014), and at least 2 years post-  
7234 harvest at all sites.

7235

7236 The riparian stand and wood recruitment data was summarized by 4 different buffer types 1) FP  
7237 Buffers, encompassing the full width (50 feet), 2) <50ft buffers, 3) Unbuffered, harvested to the  
7238 edge of the channel, and 4) Reference sites in unharvested basins. No formal statistical analyses  
7239 were conducted, only descriptive statistics were applied for changes in stand structure and wood  
7240 loading due to the low replication of reference sites and differences in harvest timing and  
7241 distribution of buffer types at the treatment sites. Density decreased by 33 and 51% and basal  
7242 area by 26 and 49% in the full and <50ft buffers, respectively, with high variability among sites.  
7243 Nearly all trees were removed from Unbuffered areas during harvest (>99% of basal area). The  
7244 reference sites cumulative post-harvest mortality during the 3-year post-harvest interval was only  
7245 6.5% of live density. In contrast, mean post-harvest mortality of the FP buffers and the <50 ft  
7246 buffers were 31 and 25% of density, respectively. However, there was considerable variation in  
7247 mortality among sites exceeding 65% in the FP buffers at two of the treatment sites. Windthrow  
7248 and physical damage from falling trees accounted for ~75% of mortality in the FP and <50 ft



7249 buffers. In contrast to the treatment sites, <10% of trees died due to wind or physical damage in  
7250 the reference sites. There was little post-harvest large wood input in reference sites: an average  
7251 of 4.3 pieces and 0.34 m<sup>3</sup> of combined in- and over-channel volume per 100 m of channel. In  
7252 contrast, the FP and <50 ft buffers received an average of 23 and 10 pieces/100 m and 2.3 and  
7253 0.7 m<sup>3</sup>/100 m of large wood, respectively. The majority of recruited large wood pieces had stems  
7254 with roots attached (SWRW); 60, 70, and 100% in the reference, FP, and <50 ft buffer types,  
7255 respectively. Pre-harvest channel large wood loading ranged from 55.8 to 111 pieces/100 m and  
7256 from 9.8 to 25.2 m<sup>3</sup>/100 m among buffer types. Piece counts remained stable in the reference  
7257 sites through year 3 post-harvest, increased in the FP buffers and unbuffered areas (8 and 13%,  
7258 respectively), and decreased in the <50 ft buffers (15%). The authors noted confidence in  
7259 interpretation of the descriptive statistics due to the large differences between the post-harvest  
7260 treatments and the reference sites. Also, these results were of similar direction and magnitude  
7261 that was seen in the preceding Hard Rock and BCIF studies.

7262 For effects of treatment on shade, data was analyzed with generalized linear mixed-effects  
7263 models. For effects of treatment on stream temperature, data was analyzed for the seven-day  
7264 average in a linear-mixed-effects model analysis of variance. Mean canopy closure decreased in  
7265 the treatment sites from 97% in the pre-harvest period to 75%, 68%, and 69% in the first, second,  
7266 and third post-harvest years, respectively, and was related to the proportion of stream buffered  
7267 and to post-harvest windthrow within the buffer. The seven-day average temperature response  
7268 increased by 0.6°C, 0.6°C, and 0.3°C in the first, second, and third post-harvest years,  
7269 respectively. During and after harvest, mean monthly water temperatures were higher, but  
7270 equaled or exceeded 16.0°C only in 2 treatment sites by up to 1.8°C at one site and by 0.1°C at  
7271 another. None of the three REF sites exceeded 16°C during the study.

7272 Results for changes in nutrient concentrations post-harvest were highly variable. Harvest  
7273 treatment effects on nutrient concentrations, discharge, and suspended sediment export could not  
7274 be calculated because discharge prediction equations could not be developed.

7275

7276

7277 McIntyre et al., 2018

7278

7279 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn  
7280 (technical coordinators). 2018. Effectiveness of Experimental Riparian Buffers on Perennial  
7281 Non-fish-bearing Streams on Competent Lithologies in Western Washington. Cooperative  
7282 Monitoring, Evaluation and Research Report CMER 18-100, Washington State Forest Practices  
7283 Adaptive Management Program, Washington Department of Natural Resources, Olympia, WA.

7284

7285 The purpose of the study was to evaluate the effectiveness of forest management prescriptions in  
7286 maintaining aquatic conditions and processes for small non-fish-bearing (Type N) headwater

7287 stream basins underlain by competent “hard rock” lithologies (i.e., volcanic or igneous rock) in  
7288 western Washington. Specifically, this study quantified and compared the effects of timber  
7289 harvest adjacent to Type N streams on riparian stand structure and tree mortality, in stream wood  
7290 loading and recruitment, stream temperature and canopy cover, stream discharge, turbidity, and  
7291 suspended sediment export, nitrogen export, and response of stream-associated amphibians. This  
7292 study used a before-after control-impact (BACI) study design. This involved evaluation of four  
7293 experimental treatments: (1) unharvested reference (n = 6); (2) 100% treatment (n = 4), a  
7294 minimum two-sided 50-ft riparian buffer along the entire Riparian Management Zone (RMZ);  
7295 (3) FP treatment (n = 3), a minimum two-sided 50-ft riparian buffer along at least 50% of the  
7296 RMZ, consistent with the current Forest Practices buffer prescription for Type N streams and  
7297 including buffers protecting the uppermost points of perennial flow (PIP), headwall and side-  
7298 slope seeps, and tributary junctions; and (4) 0% treatment (n = 4), clearcut to stream edge (no  
7299 riparian buffer). Study sites comprised the entire non-fish-bearing (Type N) basin when possible  
7300 and upland forests of all treatments were clearcut harvested. The study design included at least  
7301 two years of pre-harvest (2006 –2008) and up to three years of post-harvest data collection (2009  
7302 – 2011). Results for stand structure and tree mortality showed that in the RMZs, the proportional  
7303 changes in stem count (dstems) and basal area (dBA) were similar for the reference (mean  
7304 dstems: -11.8, SE 5.3; dBA: -6.9, SE 5.4) and 100% (mean dstems: -3.8, SE 5.9; dBA -6.7, SE  
7305 6.0) treatment. In contrast, the magnitude of decrease was significantly greater in the FPB  
7306 (portion of FP containing trees; mean dstems: -29.6, SE 6.5; dBA 124.4, SE 6.7) treatment than  
7307 in either the reference or 100% treatment. The pattern was similar in the PIPs. 2 years post-  
7308 harvest tree mortality was mostly (70%) attributed to wind/mechanical agents (pre-harvest  
7309 wind/mechanical agent caused mortality was 70%). In the reference sites, trees that died post-  
7310 harvest had smaller diameters (mean 10.3 in) and fewer came from the overstory crown class  
7311 (59.0%) than the other treatments. In contrast, in the 100% and FPB treatments, ~70% of trees  
7312 that died were from the overstory crown class and their mean diameters were 1 (11.2 in) and 2  
7313 (12.2 in) in greater than those in the reference sites, respectively. Results for wood recruitment  
7314 and loading showed that tree fall rates were highly variable during the pre-harvest period  
7315 between sites ranging from 0 to 239.9 trees/ha/yr. Large wood (LW) recruitment rates in the pre-  
7316 harvest period were also highly variable ranging from 0 to 121.6 pieces/ha/yr, along with  
7317 recruitment volume (0-16.2 m<sup>3</sup>/ha/yr). 2 years post-harvest recruitment rates in the reference  
7318 riparian management zones (RMZs) were lower and less variable (5.9 to 37.3 trees/ha/yr) than in  
7319 buffer treatments. Tree fall rates for the 100% treatment ranged from 7.7 to 76.4 trees/ha/yr, and  
7320 for the FPB treatments tree fall rates ranged from 4.2 to 152.2 trees/ha/yr. Post-harvest LW  
7321 recruitment volumes in reference RMZs were relatively low, ranging from 0.7 to 2.2 m<sup>3</sup>/ha/yr.  
7322 Post-harvest LW recruitment volumes were generally higher and more variable in the 100% and  
7323 FPB RMZs, ranging from 0.3 to 14.0 m<sup>3</sup>/ha/yr in the 100% treatment and 0 to 7.6 m<sup>3</sup>/ha/yr in  
7324 the FPB. Because of the high variability between sites in all treatments the p values for  
7325 comparisons between treatments were generally high ( $p \geq 0.35$ ), except for the FPB vs. reference  
7326 comparison for piece count which was nearly significant ( $p = 0.13$ ). The only significant  
7327 differences were for the 0% treatments which had significantly lower LW recruitment by volume  
7328 than the Reference RMZ ( $P = 0.02$ ). For PIPs, LW recruitment in the 100% treatment was over  
7329 12 times the reference rate by piece count ( $P = 0.03$ ) and 30 times the reference rate by volume

7330 (P = 0.04). Recruitment in the FPB PIPs was also high, over nine times the reference rate by  
7331 piece count (P = 0.08) and 18 times the reference rate by volume (P = 0.11). The amount of  
7332 change in the number of LW pieces per meter from pre-harvest to post-harvest depended on  
7333 treatment (P <0.01). Analysis estimated the changes in 100%, FP and 0% treatments to be  
7334 different from the change in the reference (P <0.001, 0.03 and <0.01, respectively). The  
7335 percentage of the stream channel length covered by newly recruited wood in the second post-  
7336 harvest year ranged from 0 to 11% in the reference, 1 to 15% in the 100% treatment and 0 to  
7337 10% in the FP treatment and was 0% in all four of the 0% treatments. The percent of stream  
7338 channel covered by new wood differed between the 0% treatment and the reference (P = 0.03),  
7339 100% (P <0.01), and FP treatments (P = 0.03). Overall, the authors estimated a mean between-  
7340 treatment increase of 60% (95% CI: 0–150%), 70% (95% CI: 0–190%) and 170% (95% CI:  
7341 80–330%) in the number of SW pieces per stream meter in the 100%, FP and 0% treatments  
7342 compared with the reference, respectively. Also, a between-treatment increase of 60% (95% CI:  
7343 30–110%), 40% (95% CI: 0–100%) and 50% (95% CI: 10–90%) in the number of LW pieces  
7344 per stream meter in the 100%, FP and 0% treatments compared with the reference, respectively.  
7345 The authors conclude that windthrow was responsible for much of the increase in LW. However,  
7346 they also posit that the timing and magnitude of wood inputs was inconsistent, resulting in  
7347 considerable variability between and within sites, especially in the FP treatment. Results for  
7348 shade response to treatments post-harvest was greatest in the 0% treatment than in either the  
7349 100% or the FP treatment. Effective shade decreased to 77, 52, and 14% 2 years post-treatment,  
7350 in the 100%, FP, and 0% buffer treatments, respectively. Canopy and Topographic Density  
7351 (CTD), defined as the percentage of the photograph obscured by vegetation or topography  
7352 decreased from an average of 95% pre-harvest to 86, 71, and 43% 2 years post-harvest in the  
7353 100%, FP, and 0% buffer treatments, respectively. All were significantly lower than the reference  
7354 (92% 2 years post-treatment). Results for stream temperature showed maximum daily water  
7355 temperatures increased post-harvest in all but one of the harvested sites and was elevated over  
7356 much of the year at most of the sites. Daily temperature response (TR) increased in late winter or  
7357 early spring, reached a maximum in July–August and was still elevated well into the fall. This  
7358 pattern was observed at most of the sites. For the Buffer Treatment locations, 94 of the 131  
7359 calculated mean monthly temperature responses (MMTRs) were significant and 91 of these  
7360 significant responses were positive. In comparison, only 52 of 156 MMTR values calculated for  
7361 the reference sites were significant and these were nearly evenly split with 25 positive and 27  
7362 negative responses. This strongly suggests that the pattern of post-harvest increases in daily  
7363 maximum water temperature is real even though the magnitude of some of the individual  
7364 MMTRs is relatively small (<0.5°C). Warming tended to be greatest in July or August with  
7365 MMTR ranging from 0.5°C to 2.3°C in the 100%, –0.4°C to 1.8°C in the FP, and 1.0°C to 3.5°C  
7366 in the 0% treatments. Post-harvest, Max7D (seven-day-average maximum stream temperature)  
7367 was higher at 36 of the 40 locations within the harvest units across all 11 buffer treatment sites  
7368 regardless of presence or absence of a buffer, buffer width, and longitudinal location along the  
7369 stream. Relative to the unharvested sites, there were summertime temperature increases  
7370 throughout the stream length and across all buffer treatment sites. The authors conclude that none  
7371 of the buffer treatments were successful in preventing significant increases in maximum stream  
7372 temperature. The generalizable conclusions made by the authors from this portion of the study

7373 are that 1) Buffer widths greater than 50 ft (15.2 m) are needed to prevent shade loss and (2)  
7374 Maximum water temperature decreased below the harvest unit after flowing through  
7375 approximately 100 m of intact forest but was still elevated compared to pre-harvest conditions.  
7376 Results for nitrogen and phosphorus concentrations showed that post-harvest changes for total-N  
7377 or total-P were not significant for any of the treatments relative to the Reference. The only  
7378 significant difference detected within 2 years post-harvest was for nitrate-N concentration  
7379 between the 0% buffer treatment and all other treatments. However, for annual export, total-N  
7380 and nitrate-N export increased post-harvest at all sites, with the smallest increase in the 100%  
7381 treatment and the largest in the 0% treatment. Compared to the reference sites, the GLMM  
7382 analysis showed a relative increase in total-N export post-harvest of 5.52 ( $P = 0.051$ ), 11.52 ( $P =$   
7383  $0.0007$ ), and 17.16 ( $P < 0.0001$ )  $\text{kg ha}^{-1} \text{ yr}^{-1}$  in the 100%, FP, and 0% treatments. The GLMM  
7384 analysis showed a relative increase in nitrate-N export post-harvest of 4.83 ( $P = 0.048$ ), 10.24 ( $P =$   
7385  $0.001$ ), and 15.35 ( $P < 0.0001$ )  $\text{kg ha}^{-1} \text{ yr}^{-1}$  in the 100%, FP, and 0% treatments, respectively,  
7386 only slightly less than the changes in total-N. Total-P export increased post-harvest by a similar  
7387 magnitude in all treatments: 0.10 ( $P = 0.006$ ), 0.13 ( $P = 0.001$ ), and 0.09 ( $P = 0.010$ )  $\text{kg ha}^{-1} \text{ yr}^{-1}$   
7388 in the 100%, FP, and 0% treatments, respectively. The increase in N, total-N and nitrate-N, from  
7389 the treatment watersheds post-harvest was strongly correlated with the increase in annual runoff  
7390 ( $R^2 = 0.970$  and  $0.971$ ;  $P = 0.001$  and  $0.001$ , respectively) and with the proportion of the basin  
7391 harvested ( $R^2 = 0.854$  and  $0.852$ ;  $P = 0.031$  and  $0.031$ , respectively). The correlation with the  
7392 proportion of stream length buffered was weaker ( $R^2 = 0.761$  and  $0.772$ ;  $P < 0.079$  and  $0.072$ ,  
7393 respectively). In contrast, total-P export was uncorrelated with all three variables. Overall, the  
7394 authors concluded that mean flow-weighted concentration of total-N and nitrate-N increased at  
7395 all buffer treatment sites post-harvest, however the magnitude was variable and significant only  
7396 for the 0% treatment. However, the export of total-N increased in the FP and 0% treatments and  
7397 nitrate-N increased in all buffer treatments. Increases in N export was correlated with increased  
7398 stream discharge and the proportion of the site that was harvested. Pre-harvest total-P  
7399 concentration was low and remained so post-harvest, although P export increased slightly post-  
7400 harvest in all treatments due to the increase in discharge. Results for changes in water turbidity  
7401 and suspended sediment concentrations (SSC) showed both turbidity and SSC increased with  
7402 increasing discharge during storm events but then rapidly fell off. Analysis of treatment effects  
7403 revealed no significant effects of harvest and no clear pattern regarding the relative effectiveness  
7404 of buffer treatments at mitigating the effects of clearcut harvests on suspended sediment export  
7405 (SSE). The general conclusions made by the authors were that all sites appeared to be supply  
7406 limited both pre- and post-harvest. Results for litterfall input showed a decrease in TOTAL  
7407 litterfall input in the FP ( $P = 0.0034$ ) and 0% ( $P = 0.0001$ ) treatments between pre- and post-  
7408 treatment periods. LEAF litterfall (deciduous and conifer leaves combined) input decreased in  
7409 the FP ( $P = 0.0114$ ) and 0% ( $P < 0.0001$ ) treatments in the post-treatment period. In addition,  
7410 CONIF (conifer needles and scales) litterfall input decreased in the FP ( $P = 0.0437$ ) and 0% ( $P$   
7411  $< 0.0001$ ) treatments, DECID (deciduous leaves) in the 0% ( $P < 0.0001$ ) treatment, WOOD (twigs  
7412 and cones) in the FP ( $P = 0.0044$ ) and 0% ( $P = 0.0153$ ) treatments, and MISC (e.g., moss and  
7413 flowers) in the 0% ( $P = 0.0422$ ) treatment. Results for comparison of the post-harvest effects  
7414 between treatments showed LEAF litterfall input decreased in the 0% treatment relative to the  
7415 reference ( $P = 0.0040$ ), 100% ( $P = 0.0008$ ), and FP ( $P = 0.0267$ ) treatments. Likewise, there was

7416 a decrease in DECID litterfall input in the 0% treatment relative to the Reference (P = 0.0001),  
7417 100% (P <0.0001), and FP (P = 0.0015) treatments. Results for detritus with comparisons  
7418 between the pre- and post-treatment periods showed an increase in TOTAL detritus export in the  
7419 100% treatment (P = 0.0051) and a decrease in the 0% treatment (P = 0.0046; Table 12-9).  
7420 Likewise, there was an increase in CPOM, WOOD, MISC, and FPOM detritus export in the  
7421 100% treatment (P <0.05), but a decrease in the 0% treatment (P <0.05) The authors for this  
7422 portion of the study conclude that overall, total litterfall input was slightly higher after harvest in  
7423 the 100% treatment, lower in the FP treatment and lowest in the 0% treatment; however,  
7424 statistical differences were only detected for deciduous inputs between the 0% treatment and the  
7425 other treatments. Total detritus export decreased in the 0% treatment relative to the reference,  
7426 and in the FP and 0% treatments relative to the 100% treatment.

7427

7428

7429 McIntyre et al., 2021

7430

7431 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D.E. Schuett-Hames, R. Ojala-Barbour,  
7432 G. Stewart and T. Quinn (technical coordinators). 2021. Effectiveness of experimental riparian  
7433 buffers on perennial non-fish-bearing streams on competent lithologies in western Washington –  
7434 Phase 2 (9 years after harvest). Cooperative Monitoring, Evaluation and Research Report CMER  
7435 2021.07.27, Washington State Forest Practices Adaptive Management Program, Washington  
7436 Department of Natural Resources, Olympia, WA.

7437

7438 This study was a follow-up study to the hard-rock Phase 1 study (McIntyre et al., 2018) to assess  
7439 changes over longer time periods (up to 9 years post-harvest). The purpose of the study was to  
7440 evaluate the effectiveness of forest management prescriptions in maintaining aquatic conditions  
7441 and processes for small non-fish-bearing (Type N) headwater stream basins underlain by  
7442 competent “hard rock” lithologies (i.e., volcanic or igneous rock) in western Washington.  
7443 Specifically, this study quantified and compared the effects of timber harvest adjacent to Type N  
7444 streams on riparian stand structure and tree mortality, in stream wood loading and recruitment,  
7445 stream temperature and canopy cover, stream discharge, turbidity, and suspended sediment  
7446 export, nitrogen export, and response of stream-associated amphibians. This study used a before-  
7447 after control-impact (BACI) study design. This involved evaluation of four experimental  
7448 treatments: (1) unharvested reference (n = 6), (2) 100% treatment (n = 4), a two-sided 50-ft  
7449 riparian buffer along the entire Riparian Management Zone (RMZ), (2) FP treatment (n = 3), a  
7450 two-sided 50-ft riparian buffer along at least 50% of the RMZ, consistent with the current Forest  
7451 Practices buffer prescription for Type N streams, (3) 0% treatment (n = 4), clearcut to stream  
7452 edge (no-buffer). The upland forests of all treatments were clearcut harvested. The study design  
7453 included data collection for at least two years pre-harvest (2006 –2008), and up to nine years  
7454 post-harvest from 2009 (harvest began in 2008) until 2016 or 2017 depending on the variable

7455 (e.g., wood loading, shade, etc.). Results for stand structure showed that in the buffered portions  
7456 of the FP treatments (FPB) density, basal area and relative density (RD) decreased by 59%, 55%  
7457 and 54%, respectively, 8 years after harvest. For the same variables, reductions in the 100%  
7458 RMZs were 30%, 14%, and 17%, respectively. In contrast, stand structure in the reference RMZs  
7459 was more stable, with a 17% decrease in density and little change in basal area or RD. Change in  
7460 live basal area did not differ statistically between 100% and REF RMZs for any time interval  
7461 although the differences increased over time. The FPB–REF contrast was not significant in the  
7462 first interval (years 1 and 2 post-harvest), but it was in subsequent intervals (5- and 8-years post-  
7463 harvest) as the magnitude of change in FPB RMZs increased over time. The FPB–100% contrast  
7464 was not significant until the last interval when basal area stabilized in the 100% treatment but  
7465 continued to decline in FPB. Between treatment comparison of cumulative change in live basal  
7466 area (m<sup>2</sup>/ha) between the 100% treatment and the Reference was -2.9 (CI: -16.9, 11.0), -6.0 (CI:  
7467 -20.0, 8.0), and -6.8 (CI -20.8, 7.1) for the first-, second-, and third-time intervals respectively  
7468 (none were significant). Comparison between the FPB and Reference were -10.2 (CI: -25.5, 5.2),  
7469 -16.1 (CI: -31.4, -0.8), and -21.1 (CI: -36.4, -5.8) for the first-, second-, and third-time intervals  
7470 respectively (differences for intervals 2 and 3 were significant). For tree mortality, results  
7471 showed that by year 8 post-harvest mortality as a percentage of pre-harvest basal area was lower  
7472 in the reference (16.1%) than in the 100% (24.3%) and FPB (50.8%). The FPB–Reference  
7473 contrast was not significant 2 years post-harvest, but it was at 5- and 8-years post-harvest as  
7474 mortality in FPB increased relative to the reference. The contrast between the 100% and Ref  
7475 were not significant for any time interval 8 years post-harvest. The contrasts 100% vs. REF and  
7476 FPB vs. 100%—were not significant for any time interval. This may have been because of the  
7477 high variability in the data. There was a temporal pattern to mortality in 100% and FPB RMZs.  
7478 Annual rates of mortality as percentage of live basal area and density were highest in the first  
7479 two years after harvest, then decreased. Wind/physical damage was the primary cause of  
7480 mortality. In the 100% treatment it accounted for 78% and 90% of the loss of basal area and  
7481 density, respectively; in FPB it accounted for 78% and 65% of the loss. Wind accounted for a  
7482 smaller proportion of mortality in reference RMZ (52%). Large wood recruitment to the channel  
7483 was greater in the 100% and FPB RMZs than in the reference for each pre- to post-harvest time  
7484 interval. Eight years post-harvest mean recruitment of large wood volume was two to nearly  
7485 three times greater in 100% and FPB RMZs than in the references. Large wood recruitment rates  
7486 were greatest during the first two years, then decreased. However, these differences were not  
7487 significant between any treatment comparisons, again, likely due to the high variability in the  
7488 data. Mean large wood loading differed significantly between treatments in the magnitude of  
7489 change overtime. Results showed a 66% (P < 0.001), 44% (P = 0.05) and 47% (P = 0.01) increase  
7490 in mean large wood density in the 100%, FP and 0% treatments, respectively, in the first 2 years  
7491 post-harvest compared with the pre-harvest period and after controlling for temporal changes in  
7492 the references. Five years post-treatment the mean LW density in the FP continued to increase  
7493 42% (P = 0.08), and again 8 years post-treatment (41%; P = 0.09). Results for canopy cover  
7494 showed that riparian cover declined after harvest in all buffer treatments reaching a minimum  
7495 around 4 years post-harvest. The treatments, ranked from least to most change, were REF, 100%,  
7496 FP, and 0% for all metrics and across all years. Effective shade results showed decreases of 11,  
7497 36, and 74 percent in the 100%, FP, and 0% treatments, respectively. Significant post-harvest

7498 decreases were noted for all treatments and all years. Results for stream temperature showed that  
7499 within treatment mean post-pre-harvest difference in the REF treatment never exceeded 1.0°C.  
7500 In contrast, the mean within treatment difference in the 100% treatment was 2.4°C in 2009 (Post-  
7501 harvest year 1) but never exceeded 1.0°C in later years. The mean difference in the FP treatment  
7502 exceeded 1.0°C immediately after harvest then again in 2014–2016 (post-harvest years 6–9)  
7503 while in the 0% treatment the mean difference was 5.3°C initially, then decreased over time to  
7504 near, but never below, 0.9°C. Stream temperature increased post-harvest at most locations within  
7505 all 12 harvested sites and remained elevated in the FP and 0% treatments over much of the nine  
7506 years post-harvest. Temperature responses varied by treatment, by season, and over the years. In  
7507 three out of the first four post-harvest years there was, at least, a weak ( $r < -0.48$ ) negative  
7508 correlation between July monthly mean temperature response (MMTR) and the change in  
7509 riparian cover based on each of the four shade metrics. The correlation was generally weaker  
7510 ( $-0.4 < r$  and  $P > 0.10$ ) after post-harvest year 4, except for post-harvest year 9 ( $-0.6 < r < -0.4$ ).  
7511 However, there were only eight data pairs available for Post 9, compared to ten to twelve for the  
7512 other years, which affected the correlation coefficient and p-value. However, there was a great  
7513 deal of variability in the correlation coefficient of July MMTR with shade across post-harvest  
7514 years among sites and treatments with some sites showing negative correlations and others  
7515 positive for some treatments in some years. Considering site characteristics, aspect showed an  
7516 influence on stream temperature response. In the first five post-harvest years and in Post 7 the  
7517 highest MMTR in each treatment was nearly always the site with a southern (SE or SW) aspect.  
7518 No significant correlation between July MMTR and either mean July discharge or the post-  
7519 harvest difference in discharge was observed. For the effects of harvest on stream discharge,  
7520 cumulative results of regression analysis (forward and reverse regression approaches) indicated  
7521 that discharge did increase following harvest. In relative terms, discharge increased by 5-7% on  
7522 average in the 100% treatments while increasing between 26-66% in the FP and 0% treatments.  
7523 The change in discharge following harvest was also affected by climate, weather, and physical  
7524 hydrology of the watershed. In all basins, discharge varied with precipitation, but this was a  
7525 complex relationship showing lag time between precipitation events and discharge rate response  
7526 in some watersheds. This indicated a potential relationship with physical hydrology at some  
7527 watersheds. Results for water turbidity and suspended sediment export (SSE) were stochastic in  
7528 nature and the relationships between SSE export and treatment effects were not strong enough to  
7529 confidently draw conclusions. Results for harvest effects on total nitrogen export following a  
7530 generalized linear mixed effects model, however, showed significant ( $P < 0.05$ ) treatment effects  
7531 were present in the FP treatment post-harvest and in the 0% treatment in the post-harvest (2-  
7532 years immediately following harvest) and extended periods (2015 – 2017; 7 and 8 years post-  
7533 harvest) relative to the reference sites, but there were no significant differences in total-N export  
7534 between the treatments. Analysis showed an increase in total-N export of 5.73 ( $P = 0.121$ ), 10.85  
7535 ( $P = 0.006$ ), and 15.94 ( $P = 0.000$ ) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments,  
7536 respectively, and of 6.20 ( $P = 0.095$ ), 5.34 ( $P = 0.147$ ), and 8.49 ( $P = 0.026$ ) kg/ha/yr in the  
7537 extended period. Results for nitrate-N export showed changes similar to but slightly less than  
7538 those seen in the total-N analysis with a relative increase in nitrate-N export of 4.79 ( $P = 0.123$ ),  
7539 9.63 ( $P = 0.004$ ), and 14.41 ( $P < 0.001$ ) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments,  
7540 respectively. None of the changes in the extended period were significant. However, the authors

7541 note that there was high variability in the data for the extended period and nitrate-N export only  
7542 returned to pre-harvest levels in one watershed. The increase in total-N and nitrate-N export  
7543 tended to be highest during the high flow months in the fall and early winter. The authors  
7544 conclude that the 100% treatment was generally the most effective in minimizing changes from  
7545 pre-harvest conditions, the FP was intermediate, and the 0% treatment was least effective. The  
7546 collective effects of timber harvest were most apparent in the 0% treatment in the two years  
7547 immediately post-harvest.

7548

7549 **LW**

7550

7551 Johnston et al., 2011

7552

7553 Johnston, N. T., Bird, S. A., Hogan, D. L., & MacIsaac, E. A. (2011). Mechanisms and source  
7554 distances for the input of large woody debris to forested streams in British Columbia, Canada.  
7555 Canadian journal of forest research, 41(11), 2231-2246. <https://doi.org/10.1139/x11-110>

7556

7557 The purpose of this study was to determine whether the processes and source distances from  
7558 which LW entered streams differed among channel types and sizes, to describe LW source  
7559 distance curves for a wide range of undisturbed stream and forest types, and to characterize the  
7560 relationships between LW input mechanism, source distance, and piece size. Input processes,  
7561 source distances, and physical characteristics of approximately 2100 pieces of LW at 51  
7562 anthropogenically undisturbed stream reaches throughout south and central British Columbia  
7563 were determined. Large wood (LW) was defined in this study as pieces within or suspended  
7564 above the active channel, with a minimum length of 1 m. and capable of inducing sediment scour  
7565 or deposition. A delivery mechanism was assigned to each LW piece, when it could be  
7566 determined, as bank erosion, landslide, windthrow of live trees, stem snap, or standing dead tree  
7567 fall. Differences in the frequencies of count data among LW delivery mechanisms, LW positions,  
7568 or LWD functions were assessed using chi-square tests. The effects of channel (type, width) and  
7569 forest (maximum tree height) characteristics on the proportions of LWD pieces entering the  
7570 channel by a given input mechanism were examined using ANCOVA. Channel type for this  
7571 study was grouped into 3 categories; riffle-pool (RP), cascade-pool (CP), and step-pool (SP).  
7572 Results showed that tree mortality was the most common entry mechanism at all channel types  
7573 and width categories and accounted for 65% of all LW pieces sampled. Both channel and  
7574 riparian forest characteristics influenced the proportion of LW pieces that entered streams by tree  
7575 mortality ( $P < 0.05$ ) but did not vary significantly among channel types ( $P = 0.13$ ). The  
7576 proportion of LW pieces recruited by tree mortality decreased with increasing channel width and  
7577 with increasing maximum tree height. Bank erosion inputs accounted for 20%–25% of all LW  
7578 pieces at the lower-gradient RP and CP sites but were much less important at the SP channels.  
7579 Erosion inputs increased with increasing stream size within all channel types ( $P = 0.0004$ ). Wind-



7580 induced inputs (windthrow and stem snap) accounted for 13%–20% of inputs over the channel  
7581 types and generally increased in importance in the smaller channels. The proportion of LW  
7582 recruited to the stream by stem breakage increased with increasing tree height ( $P < 0.0001$ ) and  
7583 varied among channel types ( $P = 0.040$ ), being about twice as prevalent at SP channels as  
7584 elsewhere. Landslide inputs of LWD were a minor delivery mechanism. There was considerable  
7585 variability in distances from which LW entered the stream. However, based on the cumulative  
7586 distributions over sites, 90% of the LW pieces or volume entering the channels originated within  
7587 18 m of the stream in 90% of all cases (between 2 and 23 m in all cases). The distances from  
7588 which LW entered the streams differed significantly among the various input mechanisms ( $P <$   
7589  $0.001$ ), the rank ordering of the mean source distances being bank erosion  $<$  tree mortality  $<$   
7590 stem breakage  $<$  windthrow  $<$  landslides. Bank erosion and landslides delivered the largest LW  
7591 pieces and tree mortality and stem breakage the smallest. In general, source distances increased  
7592 with increasing tree height, with the effect being stronger in the steeper channel types and  
7593 weaker in the wider channels for LW pieces and volume. However, all two-way interactions  
7594 among variables were significant implying that the mechanisms through which vegetation and  
7595 stream geomorphology influenced LW source distance were complex. Maximum tree height in  
7596 the adjacent forest accounted for the greatest variance in in-stream LW source distance for all  
7597 models.

7598

## 7599 **Nutrient**

7600

7601 Deval et al., 2021

7602

7603 Deval, C., Brooks, E. S., Gravelle, J. A., Link, T. E., Dobre, M., & Elliot, W. J. (2021). Long-  
7604 term response in nutrient load from commercial forest management operations in a mountainous  
7605 watershed. *Forest Ecology and Management*, 494, 119312.  
7606 <https://doi.org/10.1016/j.geomorph.2013.11.028>

7607

7608 The purpose of this study was to quantify and compare the differences in nitrogen and  
7609 phosphorus concentrations and loads between pre-disturbance, post road construction (post-  
7610 road), post experimental harvest (PH-I), and post operational harvest (PH-II) from both a  
7611 hydrological yield and nutrient concentration perspective. This study was carried out in the Mica  
7612 Creek Experimental Watershed in Northern Idaho. For this analysis time periods have been  
7613 broken into four distinct phases: Pre-disturbance (1992–1997), Post-road (1997–2001),  
7614 experimental-harvest Phase I (PH-I) (2001–2007), and operational sequential harvest Phase II  
7615 (PH-II) when the extent and frequency of harvests increased (2007–2016). PH-I represents an  
7616 experimental treatment phase during which harvest activities were experimentally controlled  
7617 (only upstream headwater watersheds were harvested and mature vegetation removal ranged  
7618 between 24% and 47%) followed by site management operations including broadcast burning

7619 and replanting. PH-II represents the post-experimental phase where the study area transitioned to  
7620 operational treatments that consisted of additional road construction and timber harvest, with site  
7621 management operations including pile burning and competition release herbicide application.  
7622 During this operational phase, the mature vegetation removal in the upstream and cumulative  
7623 downstream watersheds ranged between 36% and 50% and 17–28%, respectively. Monthly  
7624 annual grab samples of stream water were collected from seven flumes over the course of 25  
7625 years (from pre- to post-treatments). The samples were analyzed for six parameters, specifically  
7626 nitrate + nitrite (NO<sub>3</sub> + NO<sub>2</sub>), total Kjeldhal nitrogen (TKN), total ammonia nitrogen (TAN)  
7627 containing un-ionized (NH<sub>3</sub>) and ionized (NH<sub>4</sub><sup>+</sup>) ammonia, total nitrogen (TN), total  
7628 phosphorus (TP), and orthophosphate (OP). This study used a before-after, control-impact paired  
7629 series design (BACIPS) to evaluate direct and cumulative effects of forest management practices  
7630 on stream nutrient concentrations in paired and nested watersheds. Results for long-term trends  
7631 in stream flow showed a statistically significant increasing trend in all the watersheds during the  
7632 fall and winter seasons. Significant increases in summer streamflow only occurred in the control  
7633 watersheds. There were minimal changes in TKN concentration with a slight observed reduction  
7634 in long-term TKN loads. Overall, the cumulative mean TAN loads from all watersheds did not  
7635 show large variations with sequential varying treatments over time. In contrast to TAN, there was  
7636 a significant response in NO<sub>3</sub> + NO<sub>2</sub> following timber harvest. The response in NO<sub>3</sub> + NO<sub>2</sub>  
7637 concentrations was negligible at all treatment sites following the road construction activities.  
7638 However, NO<sub>3</sub> + NO<sub>2</sub> concentrations during the PH-I period increased significantly ( $p < 0.001$ )  
7639 at all treatment sites. Similar to the PH-I period, all watersheds experienced significant increases  
7640 in NO<sub>3</sub> + NO<sub>2</sub> concentration during the PH-II treatment period. Overall, the cumulative mean  
7641 NO<sub>3</sub> + NO<sub>2</sub> load from all watersheds followed an increasing trend with initial signs of recovery  
7642 in one treatment watershed after 2014. Mean monthly TP concentrations showed no significant  
7643 changes in the concentrations during the post-road and PH-I treatment periods. However, a  
7644 statistically significant increase in TP concentrations ( $p < 0.001$ ) occurred at all sites, including  
7645 the downstream cumulative sites, during PH-II. Generally, OP concentrations throughout the  
7646 study remained near the minimum detectable concentrations. A statistically significant increase  
7647 in mean monthly OP concentrations occurred only at the cumulative downstream treatment site  
7648 during both Post-road ( $p$ -value = 0.021) and PH-I ( $p$ -value  $< 0.001$ ) treatment periods,  
7649 respectively. The largest cumulative increase in mean annual loads was largely attributed to  
7650 increased flow. The authors conclude that only relatively small increases in nutrient loads were  
7651 detected suggesting that Idaho Forest Practices Act regulations and BMPs are effective in  
7652 minimizing the delivery of particulate-bound pollutants. Forest management activities increased  
7653 stream NO<sub>3</sub> + NO<sub>2</sub> concentrations and loads following timber harvest activities, but these effects  
7654 were also attenuated in downstream reaches and reduced through time as vegetation regrowth  
7655 occurred.

7656

7657 Quinn, T., G.F. Wilhere, and K.L. Krueger, technical editors. 2020. Riparian Ecosystems, Volume  
7658 1: Science Synthesis and Management Implications. Habitat Program, Washington Department  
7659 of Fish and Wildlife, Olympia.

7660 This publication is a synthesis of scientific literature concerning riparian areas (function, process,  
7661 characteristics, etc.) for the purpose of informing management and the development of policies  
7662 related to management of riparian areas and watersheds of Washington State. The most relevant  
7663 information in the publication to this review are in chapters 3 (large wood), 4 (stream  
7664 temperature), and 6 (Nutrient dynamics in Riparian ecosystems).

7665 The main conclusions from chapter 3 (large wood) state that the successful conservation, or  
7666 restoration, of fish habitats in forested areas requires management practices that deliver adequate  
7667 wood into aquatic systems. They propose the main scientific uncertainties, from a management  
7668 perspective, is (1) the shape of the wood recruitment curves under different watershed and site-  
7669 level conditions. These curves describe the function of wood input into streams from greater  
7670 distances from the stream (source distance curves) based on stand structure (e.g., young-old, tree  
7671 height variability, density metrics, etc.), species compositions (especially conifer vs. hardwood),  
7672 and site conditions (e.g., slope, moisture availability, soils, site index). The second uncertainty is  
7673 the effects of the potential wood delivery mechanisms that occur outside of the riparian area  
7674 (e.g., landslides, debris flows). They posit that management objectives for large wood  
7675 recruitment potential should aim to restore site composition and structure that is similar to  
7676 unmanaged riparian forests. The authors suggest that much is known about large wood  
7677 recruitment potential from within the riparian forests based on site potential and stand structure.  
7678 Previous work and the development of source distance curve equations show the range of  
7679 “effective” tree heights (trees with the bulk of stem > 10 cm, functionally classified as large  
7680 wood) is between 85 and 230 feet. This means 100% of a sites wood recruitment potential is  
7681 within 85 -230 feet of the stream. However, these equations do not account for the presence of  
7682 smaller trees or the potential of tree recruitment from outside of the riparian area, or from  
7683 extreme channel migration events.

7684 The main conclusions from chapter 4 identify that the science surrounding stream thermal  
7685 regimes is uneven. Scientists are certain that stream temperatures and thermal regimes are  
7686 important to aquatic species, and thus it is important for management practices to restore and  
7687 conserve these conditions. However, while the general conclusions of most studies show that  
7688 land use changes (urbanization, and agriculture) and forest management within riparian areas  
7689 leads to warmer stream temperatures, the spatial and temporal effects of any specific riparian  
7690 management action remain uncertain. Recovery rates for stream temperature post-treatment vary  
7691 greatly based on site location because of differences in stream size (width and depth), and  
7692 physiography (climate, physical geography). Shade from the adjacent riparian area is widely  
7693 accepted as the most important, and most directly manageable, factor affecting stream  
7694 temperature. However, because of the variability in other factors affecting stream temperature,  
7695 predicting changes in stream temperature from shade removal will likely always suffer from  
7696 imprecision.

7697 The main conclusions from chapter 6 (Nutrient dynamics in Riparian ecosystems) list land use,  
7698 forest age and composition, Climate and seasonality, elevation and topography, hydrology,  
7699 nutrient concentrations, forms and inputs, soil properties and geology, and biota as the major  
7700 factors influencing nutrient dynamics in riparian ecosystems. Riparian areas that are structurally

7701 diverse in physiography and soil are most likely to support diverse biota (vegetation, animals,  
7702 and microbial communities. More diverse riparian communities are considered best in processing  
7703 and assimilating nutrient loads. The authors identify headwater streams as important zones for  
7704 active nutrient processing because they affect downstream nutrient loads. Also, maintaining the  
7705 connection between the aquatic and terrestrial environments via floodplain conservation and  
7706 restoration is important. While there is still a lot of uncertainty involved in the mechanisms  
7707 responsible for nutrient transport through the system, it is clear that riparian areas are vital for the  
7708 not only providing nutrients to stream, but also in filtering, processing, and storing nutrients in  
7709 the short and long term. The results of most studies indicate that the storage and filtering of  
7710 nitrogen is most effective in areas with wide vegetated buffers compared to narrower buffers or  
7711 unvegetated riparian areas regardless of the type of vegetation present. However, the type of  
7712 vegetation directly impacts the quality and quantity of nutrients available. Deciduous trees  
7713 generally provide more litter with higher nutrient content. Coniferous trees, on the other hand,  
7714 life longer and provide more shade and large wood input potential. Thus, the authors conclude  
7715 that riparian management should consider both the structural and food web roles of each species  
7716 present in a forested riparian area.

7717