

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

Riparian Function Literature **Synthesis**

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

Commented [JK1]: There is a lot of information compiled and summarized in this project. I am having a difficult time digesting the information presented as a "synthesis" and find that it might be better presented as a Summary or annotated bibliography.

Overall, I appreciate the effort by the authors however, I think this document is still raw in terms of synthesizing the findings into a clear picture of how the collected studies answer the focal questions (or don't where there are gaps).

I agree with many of the comments from the other reviewers and have added only comments that are different.

Commented [JK2R1]: Do NOT address

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

25	Table of Contents	
26	Background	2
27	Focal Questions	3
28	Methods.....	3
29	Results/Summary of Review	6
30	Discussion of findings relative to FPHCP objectives	0
31	Litter/Organic matter inputs/Nutrients	0
32	Large Wood (LW)/wood load/wood recruitment.....	12
33	Bank Stability and Sediment	32
34	Shade and stream temperature	45
35	Results/discussion by focal question.....	64
36	Focal Question 1.....	64
37	Focal Question 1a	75
38	Focal Question 1b.....	79
39	Focal Question 1c	81
40	Focal Question 1d.....	84
41	Focal Question 1e.....	85
42	Focal Question 2.....	85
43	Focal Question 3.....	92
44	Focal Question 4.....	105
45	Focal Question 5.....	106
46	Focal Question 6.....	111
47	Focal Question 7.....	115
48	References.....	117
49	Appendix.....	160
50		
51		
52		
53		
54		
55		

56 Background

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

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

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

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

Commented [AJK3]: I have a major concern with the absence of a defined standard of evidence in this document.

The studies differ based on strength of experimental design and statistical power based on sample sizes. As a result, the conclusions from each study cannot be placed on equal footing.

I understand reviews have been conducted in this manner, but providing narrative summaries of individual studies and reporting conclusions at face value is not a consistent with contemporary standards of evidence.

Commented [AK4R3]: DO NOT ADDRESS

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?

Commented [AJK5]: If the Focal Questions are the main items of interest, then why include the Discussion of findings relative to FPHCP objectives?

Commented [AK6R5]: ADDRESS

Commented [JK7R5]: Concur - Address

Commented [bs8R5]: Because the scoping document presented to us requested that it be done this way “ A synthesis of the literature will also be produced that summarizes the overall findings by key riparian function, and related physical processes, that will provide recommendations for future research. “ It also provided Schuett-Hames et al. 2015 as an example of the format.

122 Methods

123 The riparian function literature synthesis 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
(Riparian OR stream OR headwater Or Watershed) AND	
(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

Commented [AJK9]: No observational studies were included? For example, no studies that substituted space for time to evaluate responses of interest?

Commented [AK10R9]: ADDRESS

Commented [bs11R9]: Yes, they were. Statement included.

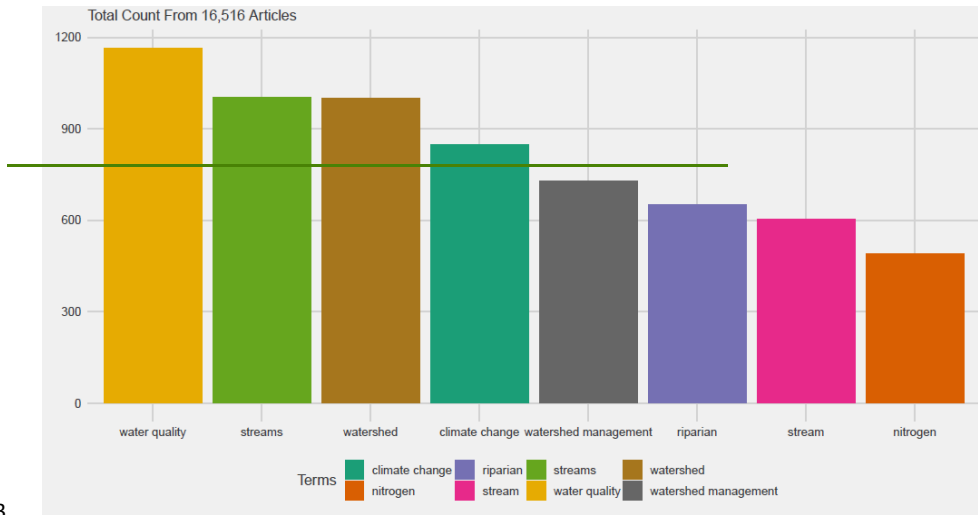
(Forest health OR recovery OR regeneration OR disease OR insect OR fung* OR patho*)	12,328
(Stand structure OR stand age OR composition OR density OR structure OR species OR species composition)	12,214
Total titles and abstracts searched, excluding duplicates	16,750

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

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

Formatted Table



173

174 Figure 1. Frequency of keywords in the original 16,516 publications sourced from Web of
 175 Science

176 **Results/Summary of Review**

177 We conducted our review of the 72 relevant publications to (1) summarize the most current state of
 178 knowledge of how timber harvest affects riparian function and related processes with a focus
 179 on the five key riparian functions defined in the FPHCP, and (2) extract information that has the
 180 potential to provide answers to, or methods and experimental designs that could be used to
 181 answer the 7 focal questions. Our review focused primarily on peer-reviewed journal
 182 publications but included 3 CMER reports and 1 report from the United States Forest Service
 183 website. Of these 72 studies, 33 were conducted on headwater or non-fish-bearing streams, 16 on
 184 fish-bearing streams, and 23 on a combination of fish and non-fish-bearing streams or
 185 hypothetical streams in a model simulation (Table 3.). Most of the studies reviewed were
 186 conducted in the Pacific Northwest region but several from just outside this region (British
 187 Columbia, Alberta, Idaho, Montana, Wyoming, Colorado) were also included (Figure 2.). Few
 188 studies could be found that quantify how riparian area treatments directly affect bank stability.
 189 Several CMER studies, however, have investigated the effects of riparian timber management on
 190 soil and streambank disturbance and erosion (Ehinger et al., 2021; McIntyre et al., 2018; Schuett-
 191 Hames et al. 2011). In these studies, soil/bank disturbance and erosion were further analyzed for
 192 their contribution to sediment export and delivery to streams. Because of this relationship
 193 between bank erosion and sediment delivery, bank stability is discussed and reviewed in the
 194 section with sediment. Further, because of the paucity of studies in the literature that provide
 195 experimental evidence of how riparian area treatments affect bank stability, studies that
 196 investigate bank stability or bank erosion based on other factors (e.g., vegetation type, vegetation
 197 coverage) have been included and reviewed in question 7. These studies are provided as

Commented [AJK12]: This information belongs in a table.

Commented [AK13R12]: ADDRESS

Commented [bs14R12]: Table included

Commented [JK15]: Red: Throughout the document there has been no synthesis of the findings from the collected studies and the same weight seems to be given to modeled/estimated results as with empirical data.

Commented [JK16R15]: Do NOT address

Commented [AJK17]: A table that describes characteristics of the individual studies would provide a helpful summary to readers.

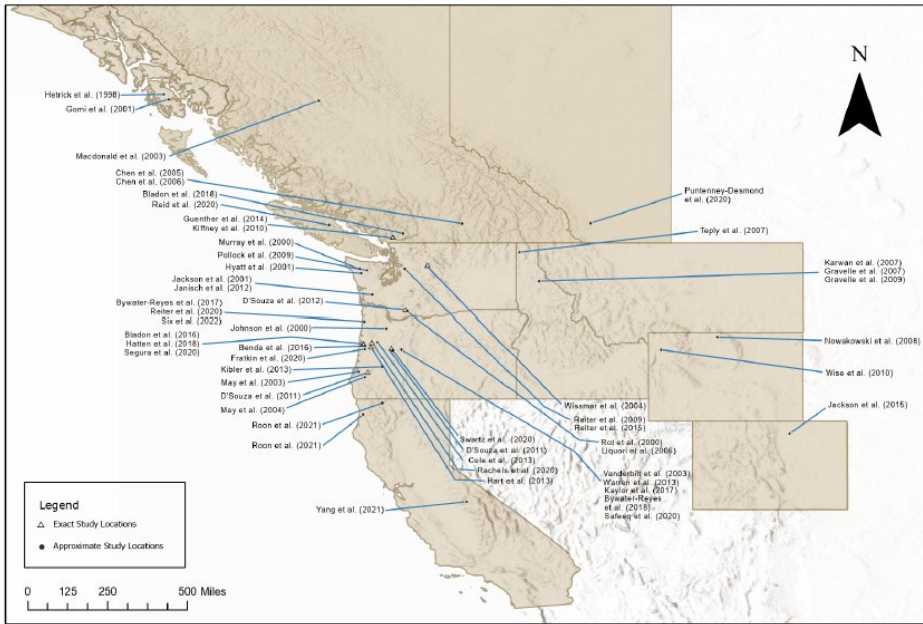
Each study could be characterized with regards to spatial and temporal scale of sampling, sample size, how responses were summarized, and whether measures of precision were included (among other characteristics).

Commented [AK18R17]: ADDRESS

Commented [JK19R17]: concur

Commented [bs20R17]: Included

198 recommendations for methods that could be used in an experimental design comparing changes
199 in bank stability before and after treatment or between treated and untreated riparian stands.



200

201 Figure 2. Locations where studies were conducted. References not listed include studies that
202 sourced data from multiple locations.

203

204

205

206

207

208

209

210

211

212

213

Table 3. Characteristics of 72 relevant publications

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

Bladon et al. (2018)	The effects of a variety of contemporary forest mgmt. prescriptions on small, headwater streams.	14 years	7	SHD	ACI	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	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	Local, Trask River Watershed	Oregon
Chen et al. (2005)	Compares the LWD biomass between different mgmt. strategies.	1 year data collection, 4 disturbance histories	4-5	LIT, LWD, NUT	ACI	Local, Okanagan Valley, Kelowna	British Columbia, Canada
Chen et al. (2006)	Assesses the amount, distribution, dynamics, and function of LWD in forest streams	2 years field data	35	LWD	ACI	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	Local, within a radius of 200 km of Corvallis	Oregon

<u>Fox & Bolton (2007).</u>	<u>observational study that categorizes the effects of riparian site geomorphology on LWD recruitment.</u>	<u>1 year data collection, multiple age classes, covertypes and disturbance histories</u>	<u>150</u>	<u>LWD</u>	<u>Descriptive, spatial modeling on historical data</u>	<u>Regional, Coastal, West and east Cascade Range of Washington State</u>	<u>Washington</u>
<u>Gomi et al. (2001)</u>	<u>LWD recruitment in the short and long-term under 5 different mgmt. strategies.</u>	<u>1 year data collection, 5 management histories</u>	<u>3</u>	<u>LWD, SED</u>	<u>ACI</u>	<u>Local, Maybeso Experimental Forests</u>	<u>Alaska</u>
<u>Gravelle & Link (2007)</u>	<u>The impacts of timber harvest practices on stream temperature.</u>	<u>13 years</u>	<u>3</u>	<u>SHD</u>	<u>BACI</u>	<u>Local, Mica creek Experimental Watershed</u>	<u>Idaho</u>
<u>Gravelle et al. (2009).</u>	<u>The effects of contemporary forest practices on the chemical properties of headwater streams and downstream locations.</u>	<u>14 years</u>	<u>3</u>	<u>NUT, SED</u>	<u>BACI</u>	<u>Local, Mica creek Experimental Watershed</u>	<u>Idaho</u>
<u>Groom et al. (2011b)</u>	<u>The efficacy of new riparian management protocols in preserving stream side shade and in-stream temperatures.</u>	<u>7 years</u>	<u>Unbalanced (15 state-owned and 18 private-owned)</u>	<u>SHD, stream temperature</u>	<u>BACI</u>	<u>Regional, Oregon Coast Range</u>	<u>Oregon</u>
<u>Groom et al. (2011a)</u>	<u>The effect of forest management on stream shade and stream temperature under Oregon forest practice rules</u>	<u>7 years</u>	<u>Unbalanced (15 state-owned and 18 private-owned)</u>	<u>SHD, stream temperature</u>	<u>BACI</u>	<u>Regional, Oregon Coast Range</u>	<u>Oregon</u>

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	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	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	Local, Central Oregon Coast Range	Oregon
Hough-Snee et al. (2016)	Evaluates which riparian, geomorphic, and hydrologic attributes are most strongly correlated to instream wood loads.	2 years of data	7	LWD, 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 of data	2	stream temperature	AI	Local, Olympic Peninsula	Washington
Hyatt & Naiman (2001)	The depletion rate of LWD in streams by size and species.	1 year of data collection. Dendrochronology to estimate up to 50 years.	4	LWD	AI	Local, Queets Ricer	Washington

<u>Jackson et al. (2001)</u>	<u>Effect of forest mgmt. on stream temperature, large woody debris, and stream sediment, between clearcut, thinned, and buffered treatments.</u>	<u>2 years</u>	<u>unbalanced: 4 -6</u>	<u>LWD, SED</u>	<u>BACI</u>	<u>Local, northwestern Washington Coast Range</u>	<u>Washington</u>
<u>Jackson & Wohl (2015)</u>	<u>Instream wood loads and geomorphic effects between streams draining montane forests of different ages.</u>	<u>1 year of data</u>	<u>10 sites > 200 years old, 23 young sites <200 years old</u>	<u>LWD</u>	<u>CI, regression analysis</u>	<u>Local, Arapaho and Roosevelt National Forests</u>	<u>Colorado</u>
<u>Janisch et al. (2012)</u>	<u>The response of stream temperature to forest harvest, testing differences in continuous vs. patch buffers.</u>	<u>4-5 years</u>	<u>unbalanced: 5-6</u>	<u>SHD</u>	<u>BACI</u>	<u>Local, southwestern Washington Coast Range</u>	<u>Washington</u>
<u>Johnson & Jones (2000)</u>	<u>Short-term and long-term effects of forest harvest on stream temperatures.</u>	<u>Historical dataset 1959-1982</u>	<u>3</u>	<u>SHD</u>	<u>BACI</u>	<u>Local, H.J. Andrews Experimental Watershed</u>	<u>Oregon</u>
<u>Karwan et al. (2007)</u>	<u>Effects of timber harvest on suspended sediments in streams following timber harvest.</u>	<u>3 years</u>	<u>2</u>	<u>SED</u>	<u>BACI</u>	<u>Local, Mica creek Experimental Watershed</u>	<u>Idaho</u>
<u>Kaylor et al. (2017)</u>	<u>Examines the effects of riparian forest harvest and varying stages of stand recovery on light availability.</u>	<u>1 year data collection, 50 - 60 years post treatment</u>	<u>14</u>	<u>SHD</u>	<u>AI</u>	<u>Local, H.J. Andrews Experimental Watershed</u>	<u>Oregon</u>

<u>Kibler et al. (2013)</u>	<u>Examined the effects of contemporary forest practices on warm-season stream temperature regimes in headwater streams.</u>	<u>3.5 years</u>	<u>8</u>	<u>SHD</u>	<u>BACI</u>	<u>Local, Hinkle Creek</u>	<u>Oregon</u>
<u>Kiffney & Richardson. (2010)</u>	<u>Evaluates the effects of forest mgmt. on organic matter/ litterfall recruitment.</u>	<u>8 years</u>	<u>Unbalanced: 2-3</u>	<u>LIT</u>	<u>ACI</u>	<u>Local, southwestern British Columbia</u>	<u>British Columbia, Canada</u>
<u>Liquori (2006)</u>	<u>Examines differences in post-harvest ecological and geomorphic processes in buffered forest sites</u>	<u>1 year data collection</u>	<u>Unbalanced: 4-9</u>	<u>Other processes, disturbance post-harvest</u>	<u>AI</u>	<u>Local, managed tree farm in Cascade Mountains of western Washington</u>	<u>Washington</u>
<u>Litschert & MacDonald (2009)</u>	<u>Assessed streamside management zones to understand characteristics of the sediment delivery pathways following upland harvest.</u>	<u>1 year data collection</u>	<u>200</u>	<u>SED</u>	<u>AI</u>	<u>Regional, National Forests in the Sierra and Cascade mountains.</u>	<u>California</u>
<u>Macdonald et al. (2003a)</u>	<u>Evaluates the effects of 2 different harvest prescriptions on suspended sediment concentrations.</u>	<u>6 years</u>	<u>2</u>	<u>SED</u>	<u>BACI</u>	<u>Local, Baptiste watershed</u>	<u>British Columbia, Canada</u>
<u>Macdonald et al. (2003b)</u>	<u>Examined the effects of three different variable retention harvesting prescriptions on stream temperature</u>	<u>7 years</u>	<u>5</u>	<u>SHD</u>	<u>ACI</u>	<u>Local, Baptiste and Galuski watersheds</u>	<u>British Columbia, Canada</u>

<u>Martin & Grotefendt (2007)</u>	<u>Compared site conditions between riparian buffer strips and unlogged riparian stands using aerial photography to determine mortality and LWD recruitment</u>	<u>1 year data collection</u>	<u>2</u>	<u>LWD</u>	<u>ACI</u>	<u>Regional, northern and southern portions of southeast Alaska</u>	<u>Alaska</u>
<u>May & Gresswell (2003)</u>	<u>Investigates the mechanisms responsible for LWD recruitment into streams.</u>	<u>2-year data collection</u>	<u>4</u>	<u>LWD, SED</u>	<u>modeling, Regression analysis</u>	<u>Local, North Fork of Cherry Creek Research Natural Area</u>	<u>Oregon</u>
<u>Meleason et al. (2003)</u>	<u>Evaluate of the potential effects of different riparian mgmt. strategies on the standing stock of wood.</u>	<u>Simulation modeling of 720 years</u>	<u>1</u>	<u>LWD</u>	<u>Modeling</u>	<u>simulation of stream types common in PNW</u>	<u>PNW, hypothetical stream</u>
<u>Mueller & Pitlick (2013)</u>	<u>Examines the relative importance of lithology as a driver of sediment delivery into streams.</u>	<u>multiple datasets ranging 5-90 years</u>	<u>83</u>	<u>SED</u>	<u>spatial modeling, correlative analysis of historical data</u>	<u>Regional, Northern Rocky Mountains</u>	<u>ID, WY, MT</u>
<u>Murray et al. (2000)</u>	<u>Examined the influence of partial harvesting on stream temperature, chemistry, and turbidity.</u>	<u>2 years data collection, 10-15 years after treatment</u>	<u>1</u>	<u>NUT, SED, SHD</u>	<u>ACI</u>	<u>Local, Rock and Tower Creek watersheds</u>	<u>Washington</u>

Nowakowski & Wohl (2008)	Examined differences in wood load and valley/channel characteristics between managed and unmanaged riparian areas.	1 year data collection	19	LWD	ACI	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	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	Local, Enos Creek	Oregon
Reid & Hassan (2020)	Combines a wood budget model and a 45-year record of LWD to examine changes in LWD characteristics.	Long-term dataset from 1973-2017, simulated 300 years	8	LWD	Simulation Modeling for framework development	Local, Carnation Creek	BC, Canada
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	Local, Deschutes River watershed	Washington

<u>Reiter et al. (2009)</u>	<u>Effects of forest practices on sediment production at the watershed-scale with 30 years of water quality data.</u>	<u>Long-term dataset from 1975-2005</u>	<u>4</u>	<u>SED</u>	<u>AI</u>	<u>Local, Deschutes River watershed</u>	<u>Washington</u>
<u>Reiter et al. (2020)</u>	<u>Effects of harvesting and variable buffer widths on stream temperature</u>	<u>10 years</u>	<u>Unbalanced: 3-7</u>	<u>SHD</u>	<u>BACI</u>	<u>Local, Trask River Watershed</u>	<u>Oregon</u>
<u>Roon et al. (2021a)</u>	<u>Thinning effects of second growth redwood forests in northwestern California.</u>	<u>2 years</u>	<u>3</u>	<u>SHD</u>	<u>BACI</u>	<u>Local, Tectah and Lost Man watersheds</u>	<u>California</u>
<u>Roon et al. (2021b).</u>	<u>Investigation of how different thinning intensities affect stream temperature via loss of canopy cover at local and watershed scales.</u>	<u>2 years</u>	<u>3</u>	<u>SHD, stream temperature</u>	<u>BACI</u>	<u>Local, Tectah and Lost Man watersheds</u>	<u>California</u>
<u>Safeeq et al. (2020)</u>	<u>Presents an approach at isolating the streamflow effect on sediment delivery post-harvest.</u>	<u>Long-term dataset, 1952-2016</u>	<u>2</u>	<u>SED</u>	<u>BACI</u>	<u>Local, H.J. Andrews Experimental Watershed</u>	<u>Oregon</u>
<u>Schuett-Hames & Stewart (BCIF), (2019b)</u>	<u>The study analyzes the changes in stand structure, buffer tree mortality, and riparian functions 10 years after upland timber harvest.</u>	<u>10 years</u>	<u>Unbalanced: 3-14</u>	<u>LWD, SED, SHD</u>	<u>ACI</u>	<u>Regional, western Washington Coast and Cascade Range</u>	<u>Washington</u>

<u>Schuett-Hames & Stewart (2019a)</u>	<u>comparison of LWD inputs, tree fall, and stand structure 5 years post-harvest.</u>	<u>5 years</u>	<u>Unbalanced: 8-9</u>	<u>LWD</u>	<u>ACI</u>	<u>Regional, northeastern Washington, 1 site in East Cascades</u>	<u>Washington</u>
<u>Schuett-Hames et al. (2011)</u>	<u>Evaluates the effects of forest mgmt. on stream shade, large woody debris recruitment, and sediment delivery.</u>	<u>5 years</u>	<u>Unbalanced: 3-15</u>	<u>LWD, SED, SHD</u>	<u>ACI</u>	<u>Regional, western Washington Coast and Cascade Range</u>	<u>Washington</u>
<u>Six et al. (2022)</u>	<u>Assessed differences in levels of riparian buffer retention at mitigating changes to organic matter dynamics.</u>	<u>2 years</u>	<u>3</u>	<u>LIT, LWD</u>	<u>BACI</u>	<u>Local, Trask River Watershed</u>	<u>Oregon</u>
<u>Sobota et al. (2006)</u>	<u>Study of riparian characteristics and their effects on tree fall direction and in-stream recruitment.</u>	<u>3 years</u>	<u>21</u>	<u>LWD</u>	<u>model with field data</u>	<u>Regional, Pacific Northwest and Intermountain West</u>	<u>Idaho, Washington, Oregon, Montana</u>
<u>Sugden et al. (2019).</u>	<u>Assessed the efficacy of Montana SMZ guidelines for controlling stream temperature.</u>	<u>2 years</u>	<u>30</u>	<u>SHD</u>	<u>BACI</u>	<u>Regional, Western Montana</u>	<u>Montana</u>
<u>Swartz, et al. (2020)</u>	<u>Assessed whether experimental canopy gaps meant to mimic natural disturbances affect stream temperature</u>	<u>2 years</u>	<u>6</u>	<u>SHD</u>	<u>BACI</u>	<u>Local, Mckenzie River Basin</u>	<u>Oregon</u>

<u>Teply et al. (2007)</u>	<u>Compares the effects of mgmt. harvest prescriptions and no-harvest RMZs on LWD recruitment in streams.</u>	<u>1 year data collection, 100 years simulated</u>	<u>58</u>	<u>LWD</u>	<u>Simulation Modeling</u>	<u>Local, Priest Lake Watershed</u>	<u>Idaho</u>
<u>Vanderbilt et al. (2003)</u>	<u>Correlation of nutrient inputs with weather events (mainly precipitation).</u>	<u>long-term datasets, ranging from 20-30 years</u>	<u>6</u>	<u>NUT</u>	<u>ACI</u>	<u>Local, H.J. Andrews Experimental Watershed</u>	<u>Oregon</u>
<u>Warren et al. (2013)</u>	<u>Evaluates stand age and associated canopy structural differences on stream light in second-order streams.</u>	<u>1 year data collection</u>	<u>2</u>	<u>SHD</u>	<u>ACI</u>	<u>Local, H.J. Andrews Experimental Watershed</u>	<u>Oregon</u>
<u>Wing & Skaugset (2002)</u>	<u>Examines the relationship between channel characteristics and LWD in streams.</u>	<u>Extensive spatial dataset from 1990-1996</u>	<u>3793</u>	<u>LWD</u>	<u>modeling, regression analysis</u>	<u>Regional, Western Cascad and Coast Range of Oregon</u>	<u>Oregon</u>
<u>Wise et al. (2010)</u>	<u>Uses tree rings to augment previous records to reconstruct multi-century data for the Snake River.</u>	<u>Dendrochronology records from 1600-2005</u>	<u>3</u>	<u>Drought Frequency</u>	<u>Climate reconstruction from dendrochronology records</u>	<u>Local, 3 sites in western Wyoming</u>	<u>Wyoming</u>
<u>Yang et al. (2021)</u>	<u>Examined the temporal variation in response of downstream water chemistry to prolonged drought and forest thinning.</u>	<u>5 years</u>	<u>2</u>	<u>NUT</u>	<u>BACI</u>	<u>Local, The Kings River Experimental Watershed</u>	<u>California</u>

<u>Yeung et al. (2019)</u>	<u>Modelled the post-harvest response of leaf litter coarse particulate organic matter quantity in a coastal stream</u>	<u>Published data spanning 4-5 years</u>	<u>Total n not reported</u>	<u>LIT</u>	<u>Heuristic modeling</u>	<u>CPOM data from local streams in coastal BC.</u>	<u>Model developed from multiple North American sites</u>
<u>Ehinger et al. (2021)</u>	<u>Effectiveness of riparian mgmt. in maintaining function in headwater streams on incompetent lithologies.</u>	<u>5-6 years</u>	<u>Unbalanced: 6-8</u>	<u>LWD, NUT, SED, SHD</u>	<u>BACI</u>	<u>Regional, southwestern Washington</u>	<u>western Washington</u>
<u>McIntyre et al. (2021)</u>	<u>Follow-up study to the McIntyre et al., 2018 to assess changes over longer time periods (up to 9 years post-harvest).</u>	<u>5 years</u>	<u>Unbalanced: 3-6</u>	<u>LWD, NUT, SED, SHD</u>	<u>BACI</u>	<u>Regional, western Washington</u>	<u>western Washington</u>
<u>McIntyre et al. (2018)</u>	<u>Effectiveness of forest mgmt. in maintaining function for small headwater streams on competent lithologies.</u>	<u>11 years</u>	<u>Unbalanced: 3-7</u>	<u>SHD, SED, NUT, LW, LIT</u>	<u>BACI</u>	<u>Regional, western Washington</u>	<u>western Washington</u>
<u>Deval et al. (2021)</u>	<u>Disturbance effects on stream chemistry.</u>	<u>13 years</u>	<u>7</u>	<u>NUT</u>	<u>BACI</u>	<u>Local, Mica creek Experimental Watershed</u>	<u>Idaho</u>

217 Discussion of findings relative to FPHCP objectives

218 Litter/Organic matter inputs/Nutrients

219 Prior to the Forest and Fish Report (1999), studies that directly quantify the effects of timber
220 harvest within riparian areas on litter and organic matter (OM) input into streams in managed
221 watersheds of western north America are sparse. Two seminal studies, one from the H.J.
222 Andrews experimental watershed studies (Gregory et al., 1987) and one from the Carnation
223 Creek experimental watershed (Hartman & Scrivener, 1990) present results that estimate loss of
224 litter input following harvest. Gregory et al., (1987) which was part of the Streamside
225 Management: Forestry and Fishery Management collection produced by Salo & Cundy (1987)
226 noted that removal of the forest canopy from timber harvesting resulted in decreases in annual
227 litter fall from 300-400 g/m² in the mature forests to less than 100 g/m². Further, they posit that
228 decreased litter inputs after logging can persist for 10 – 20 years before recovering. Results from
229 Hartman & Scrivener, (1990) showed that litter inputs post-logging were 25-50% of pre-logging
230 levels with about 50% of the loss recovering within a decade (note: buffer widths varied from 1-
231 70 m, litter input loss was not analyzed by buffer width).

232 Experimental studies published after 1999 that investigate the factors affecting litter and organic
233 matter (OM) input (not including LW) into streams in western North America are still relatively
234 few. In our search we found six papers that quantify the effects of timber harvest or the effects of
235 site factors (e.g., topography, vegetation characteristics) Four of these studies focus on headwater
236 streams and two of the studies reviewed here extend into larger fish-bearing streams (Bilby &
237 Heffner, 2016; Hart et al., 2013; Kiffney & Richardson, 2010; McIntyre et al., 2018; Six et al.,
238 2022; Yeung et al., 2019).

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

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

256 study area. The results also reveal a trend that suggests slope affects the width of the litter
257 contributing area. However, the authors did not apply statistical analysis to these values and only
258 speculate that increasing the slope from 0-45% would increase the width of the litter contributing
259 area by up to 71% for needles and 95% for leaves. From these results, Bilby & Heffner (2016)
260 suggest that wind speed has a strong effect on the width of litter delivery areas within riparian
261 areas, but that relationship is also affected by stand age (suggesting that tree height was a factor)
262 and litter type (deciduous vs. conifer). Other than stand structure and topography, another study
263 shows evidence of species composition affecting litter delivery into streams. Hart et al. (2013)
264 compared litter delivery into streams between riparian zones dominated by deciduous (red alder)
265 and coniferous (Douglas-fir) tree species in western Oregon. Results from this study show that
266 deciduous forests dominated by red alder delivered significantly greater vertical and lateral
267 inputs ($\text{g m}^{-2} \text{y}^{-1}$) to adjacent streams than did coniferous forests dominated by Douglas-fir.
268 Deciduous-site vertical litter input (mean = 504 $\text{g m}^{-2} \text{y}^{-1}$) exceeded that from coniferous sites
269 (394 $\text{g m}^{-2} \text{y}^{-1}$) by 110 g/m^2 over the full year. Annual lateral inputs at deciduous sites (109 g
270 $\text{m}^{-2} \text{y}^{-1}$) 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
271 inputs also differed, with the greatest differences occurring in November during autumn peak
272 inputs for the deciduous forests. Further, annual lateral litter input increased with slope at
273 deciduous sites ($R^2 = 0.41073$, $p = 0.0771$), but showed no strong relationship at coniferous sites
274 ($R^2 = 0.1863$, $p = 0.2855$). These results were partially consistent with Bilby & Heffner (2016)
275 in that they suggest litter type, and topography (slope) can affect the litter input rates. Lateral
276 litter movement in the riparian area increased with slope for deciduous riparian forests
277 throughout the year and for coniferous forests only in the spring and summer months.

278 In terms of the effects of timber harvest on litter and OM quantity in streams, 4 studies in our
279 review were found that provide experimental results that have been conducted since 2000 and
280 focus on western North America. Of these 4 studies, 1 used simulation modeling (Yeung et al.,
281 2019), and the other 3 (Kiffney & Richardson, 2010; McIntyre et al., 2018; Six et al., 2022) used
282 field-based experiments to estimate the effects of timber harvest within riparian forests on OM
283 inputs and dynamics in streams. Yeung et al. (2019) simulated post-harvest responses to leaf-
284 litter derived coarse particulate organic matter (CPOM) quantity in a coastal rainforest stream in
285 British Columbia, Canada. For this study, Yeung et al. (2019) used published empirical data from
286 representative small, forested streams in coastal British Columbia to calibrate and set parameters
287 for their CPOM model. The model compared the effects litterfall reduction, increase in peak
288 flows, and increase in stream temperature (estimated for 4 harvesting intensities based on
289 available data) on in stream CPOM standing stocks. Results showed evidence that litterfall
290 reductions from timber harvest was the strongest control on in-stream CPOM quantity for 4 years
291 post-harvest. However, when litterfall reductions were below 30%, the effect size varied with
292 relative changes to peak flows and stream temperature. Stream temperature increases specifically
293 showed a significant interaction with litterfall reductions. The authors propose that the decreased
294 activity of CPOM consumers caused by increasing stream temperatures by 4 °C or more, may be
295 enough to offset the loss of litterfall inputs of CPOM stocks. This speculation was made based on
296 the temperature dependent function of leaf litter consumption by common shredder species and
297 temperature ranges modeled by Stenroth et al. (2014). [This model predicts shredder activity is
298 optimized at ~15 °C (ranging between 13.7 – 16.7°C) but begins to quickly decline at

Commented [AJK21]: r-squared?

Commented [AK22R21]: ADDRESS

Commented [bs23R21]: corrected

Commented [AJK24]: Pearson's correlation coefficients do not need to be reported beyond 2 decimal places.

Commented [AK25R24]: ADDRESS

Commented [bs26R24]: corrected

Commented [AJK27]: This result is a weak one.

A correlation coefficient of 0.41 doesn't suggest much correlation at all (and I will assume the relationship was approximately linear). Also, a p-value of 0.077 shows only a moderate relationship at best (assuming one is interested at all in p-values in 2024).

Commented [AK28R27]: DO NOT ADDRESS

Formatted: Superscript

Formatted: Superscript

Formatted: Superscript

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

301 All four studies that applied an experimental design to assess the changes in litter and OM
302 delivery into streams used a Before-After Impact-Control (BACI) design. Also, all these studies
303 compared changes in litter and OM inputs into streams for two or more riparian forest harvest
304 prescriptions (Table A1; Appendix I2). Kiffney & Richardson (2010) compared changes in litter
305 input between riparian harvest prescriptions that included clear-cut to stream edge, 10 m wide
306 buffer reserve, 30 m buffer reserves, and an uncut control over the course of 8 years. No thinning
307 was applied within the reserves. Upland treatment at all sites applied clearcut. Results showed
308 differences in litter flux relative to riparian treatment persisted through year 7, while a positive
309 trend between reserve width and litter flux remained through year 8. Needle inputs remained 6x
310 higher in the buffer and control sites through year 7, and 3-6x higher in year 8 than in the
311 clearcut sites. Twig inputs into the control and buffered sites were ~25x higher than in the
312 clearcut sites in the first year after treatment. The linear relationship between reserve width and
313 litter inputs was strongest in the first year after treatment, explaining ~57% of the variation, but
314 the relationship could only explain ~17% of the variation in litter input by buffer width by year 8
315 (i.e., the relationship degraded over time). The authors interpret these results as evidence that
316 litter flux from riparian plants to streams, was affected by riparian reserve width and time since
317 logging.

318 McIntyre et al. (2018) also assessed the difference in the changes in litterfall inputs into streams
319 following three experimental treatments: an unharvested control (Reference), current Forest
320 Practices that apply a two-sided 50-ft riparian buffer along at least 50% of the stream (FP; with
321 clearcut to stream's edge outside of the buffer), a two sided 50-ft buffer along the entire stream
322 (100%), and a clearcut to stream without a buffer (0%). The upland forests of all treatments were
323 clearcut harvested. Results for litterfall input showed a significant decrease in total litterfall
324 (includes leaves/needles, twigs, cones etc.) input in the FP and 0% treatments between pre- and
325 post-treatment periods (2 years of pre-, and 2 years of post-harvest data). However, compared to
326 the Reference streams, only the 0% treatment (unbuffered) showed a significantly lower litterfall
327 input post-harvest and only for deciduous leaves, and combined total of deciduous leaves and
328 conifer needles. The 100% buffer showed a non-significant increase in litterfall inputs relative to
329 the reference streams. The authors interpret these results as evidence that the riparian vegetation
330 community in the unbuffered treatment had not recovered by the end of year 2 post-harvest.

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

Commented [AJK29]: Was a confidence interval provided with the prediction?

Commented [AK30R29]: ADDRESS

Commented [bs31R29]: A range of optimum temperatures included

Commented [AJK32]: More generally, I urge you not to report summary statistics from studies without standard deviations, standard errors, confidence intervals, or prediction intervals. If the authors did not provide any summary measures of precision, that should be reported your summary. At the very least, the range of responses should be reported.

Commented [AK33R32]: DO NOT ADDRESS

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

344 The objective of the study from Wooton (2012) was to assess how riparian area treatments
345 impact river food webs with an emphasis on economically important salmonid species in an
346 Olympic Peninsula River in Washington state. However, they present results and statistical
347 analysis for differences in litter inputs ($\text{g m}^{-1} \text{hr}^{-1}$) between treated and untreated reaches.
348 Because of the lack of litter input studies in literature, their results are presented here. Wooton
349 (2012) removed the dominant tree species, red alder (*Alnus rubra*), from one bank along five
350 treatment reaches ranging from 100-300 m long and replaced them with conifer seedlings. Paired
351 control reaches were interspersed between treated reaches along the stream. Specific methods for
352 tree removal or width of buffer in treatment reaches were not reported. Leaf litter decreased
353 significantly ($p = 0.04$) in the treatment reaches compared to the control reaches ($4.92 + 2.55$ vs.
354 $14.12 + 5.70 \text{ g m}^{-1} \text{hr}^{-1}$).

355 *Nutrients*

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

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

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

391 In our search of the literature, four studies were found that provide experimental evidence of the
392 effects of riparian timber harvest on nutrient flux in western north America and were published
393 since 2000. Gravelle et al., 2009 compared the effects of contemporary forest harvesting
394 practices in Idaho on nutrient cycling and in stream concentrations. This study followed the
395 BACI design and featured a pre-treatment measurement phase (5 years), a post-road construction
396 phase (5 years), and a post-harvest phase (5 years). Treatments imposed included a clearcut to
397 stream with 30-foot equipment exclusion zone (non-fish-bearing), a target reduction of 50% of
398 the canopy removal over 50% of the area, equating to 25% removal of existing shade (fish-
399 bearing streams), and was compared to an uncut reference. Results for the post-road construction
400 period showed no significant changes in any analyzed nutrient concentrations. Results for the
401 post-harvest period showed significant increases in monthly mean nitrate and nitrite (NO_3 and
402 NO_2) at sites immediately downstream from the clearcut, the partial harvest, and at sites
403 downstream from both treatments in the stream network (cumulative). The changes in monthly
404 mean NO_3 and NO_2 during the five years post-harvest were greatest for the clearcut treatment
405 ($+0.29 \text{ mg L}^{-1}$), followed by the cumulative ($+0.07$ and $+0.05 \text{ mg L}^{-1}$) and partial harvest ($+0.03$
406 mg L^{-1}). NO_3 showed progressively increasing monthly concentrations for 3 years after harvest
407 before declining. None of the other nutrients analyzed in this study (Kjeldahl nitrogen (TKN),
408 total phosphorus (TP), total ammonia nitrogen (TAN) consisting of un-ionized (NH_3) and ionized
409 (NH_4^+) ammonia, and unfiltered orthophosphate (OP) samples) showed significant changes
410 during the post-harvest period.

411 In a follow up study, Deval et al. (2021) compared changes to nutrient concentrations 8 years
412 after Gravelle et al. (2009) completed their study. During these 8 years (extended harvest period)
413 the extent and frequency of harvest operations increased. Treatments consisted of additional road
414 construction and timber harvest (clearcut), with site management operations including pile
415 burning and competition release herbicide application. Following these treatments, streams in all
416 harvested watersheds again experienced significant increases in $\text{NO}_3 + \text{NO}_2$ concentrations of
417 even higher magnitude than during the first post-harvest period. Further, there were also small
418 but significant increases in mean monthly total phosphorus (TP) concentrations at all treatment
419 sites, including the downstream cumulative site. Cumulative $\text{NO}_3 + \text{NO}_2$ concentrations increased
420 throughout the study but showed signs of recovery in one watershed approximately 3 years after
421 the last treatment (clearcut, broadcast burn, herbicide). The authors attribute the increase in

422 NO³+NO² and TP during the extended harvest periods (i.e., beyond what was observed in the
423 first post-harvest period) to the application of herbicides and broadcast burning.

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

430 Considering the interaction between climate and forest harvest on nutrient transport, Yang et al.
431 (2021) investigated the effects of drought and forest thinning operations (independently and
432 combined) on stream and soil water chemistry in the Mediterranean climate headwater basins of
433 the Sierra National Forest. Data on water chemistry were taken 2 years prior and 3 years
434 following drought and thinning operations in two watersheds, each with thinned and control
435 stands. Young stands with high shrub cover (> 50%) were masticated to < 10% shrub cover. The
436 thinning prescription in mature stands removed trees across all diameter classes to a target basal
437 area range of 27–55 m² ha⁻¹ with target basal areas varying based on tree density. Thinning
438 extended into the riparian management zone. Trees within 15 m of the stream could be chainsaw-
439 felled and skidded, but mechanical equipment was excluded within 30 m of the stream. Results
440 showed that drought alone altered dissolved organic carbon (DOC) in stream water, as well as
441 altered the proportion of dissolved organic carbon to nitrogen (DOC: DON) in soil solution in
442 unthinned (control) watersheds. Volume-weighted concentration of DOC was 62% lower ($p <$
443 0.01) and DOC:DON was 82% lower ($p = 0.004$) in stream water and soil solution, respectively,
444 during years of drought than in years prior to drought. Drought combined with thinning altered
445 DOC and dissolved inorganic nitrogen (DIN) in stream water, and DON and total dissolved
446 nitrogen (TDN) in soil solution. For stream water, volume-weighted concentrations of DOC were
447 66–94% higher in thinned watersheds than in control watersheds for all three consecutive
448 drought years following thinning. No differences in DOC concentrations were found between
449 thinned and control watersheds before thinning. The authors conclude that their results provide
450 evidence that the influences of drought and thinning are more pronounced for DOC than for
451 nitrogen in streams. They also speculate that the periodic changes in climate (e.g., seasonal,
452 drought) contribute to the high variability in carbon and nitrogen concentration in streams in
453 Mediterranean climates following harvest.

454 Specific to Washington, the Hard Rock (McIntyre et al., 2021) and the Soft Rock (Ehinger et al.,
455 2021) studies also reported on changes in nutrient concentrations and nutrient export in streams
456 following riparian timber harvest along headwater streams of western Washington. Treatments
457 included a 50 ft buffer along both sides of the stream for the entire RMZ (“100%”), 50 ft buffer
458 along at least 50% of the RMZ (“FP”), clearcut to stream (“0%”), and an unharvested reference
459 (Ref). Results for nitrogen and phosphorus concentrations in streams showed that post-harvest
460 changes for total-N or total-P were not significant for any of the treatments relative to the
461 Reference. The only significant difference detected post-harvest was for nitrate-N concentration
462 between the 0% buffer treatment and all other treatments. However, for annual export (kg ha⁻¹

463 yr-1), total-N and nitrate-N export increased post-harvest at all sites, with the smallest increase in
464 the 100% treatment and the largest in the 0% treatment. Compared to the reference sites, analysis
465 showed an increase in total-N export of 5.52 (P = 0.051), 11.52 (P = 0.0007), and 17.16 (P
466 <0.0001) kg ha-1 yr-1 in the 100%, FP, and 0% treatments, respectively, in the first 2 years post-
467 harvest. In the extended period (7-8 years post-harvest) export for total-N remained higher in all
468 treatments compared to the reference by 6.20 (P = 0.095), 5.34 (P = 0.147), and 8.49 (P = 0.026)
469 kg ha-1 yr-1 for the 100%, FP, and 0% treatments, respectively. Nitrate-N showed the same
470 pattern with slightly lower values than total-N. The increase in total-N and nitrate-N export from
471 the treatment watersheds post-harvest was strongly correlated with the increase in annual runoff
472 (R2 = 0.970 and 0.971; P = 0.001 and 0.001) and with the proportion of the basin harvested (R2
473 = 0.854 and 0.852; P = 0.031 and 0.031). The authors note that there was high variability in the
474 data for the extended period and nitrate-N export only returned to pre-harvest levels in one
475 watershed. Total-P export increased post-harvest by a similar magnitude in all treatments: 0.10 (P
476 = 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
477 (only analyzed during the 2-year post-harvest period). The authors conclude that the 100%
478 treatment was generally the most effective in minimizing changes from pre-harvest conditions,
479 the FP was intermediate, and the 0% treatment was least effective. Thus, similar to the results of
480 other studies reviewed, these results provide evidence that the effects of timber harvest on
481 nutrient export is proportional to the intensity of the treatment (e.g. percent of basin harvested,
482 presence of protective buffer).

483 *Summary of Factors Impacting Nutrient Concentrations and Export*

484 Similar to instream sediment concentrations and export, there is evidence from the studies
485 reviewed that nutrient dynamics are affected by the intensity of riparian timber harvest (e.g.,
486 presence of buffer widths, percent of basin harvested), changes in streamflow (either seasonally
487 or from harvest), climatic events (e.g., drought, heavy precipitation), physiography (e.g.,
488 hillslope gradient), and soil disturbance. The Soft Rock study (Ehinger et al., 2021) did analyze
489 changes in both sediment and nutrient flux following harvest for comparison with the Hard Rock
490 study. While the authors of this study report that the softer lithologies were more erodible than
491 the sites sampled for the Hard Rock study and that nutrient flux was within the range of results
492 for the Hard Rock study, effects of treatment and significant differences between studies could
493 not be detected because of limited sample sizes, inconsistent buffer widths, and timing of
494 harvest.

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

503

Commented [AJK34]: Throughout the document, this type of comment must be supported with statistical summaries of evidence.

Commented [AK35R34]: DO NOT ADDRESS

504 Table 2. List of treatments, variables, metrics, and results from publications reviewed for information on litter, organic matter, and
 505 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.	Microclimate, stand structure, canopy cover	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 (1.6 to 1.9°C), and within buffers adjacent to patch openings (1.3.5°C), than in untreated stands.
Rilby & 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 NO ₃ + NO ₂ concentrations was negligible at all treatment sites following the road construction activities. However, NO ₃ + NO ₂ 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 NO ₃ + NO ₂ concentration during the PH II treatment period. Overall, the cumulative mean NO ₃ + NO ₂ 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

Commented [JK36]: YELLOW: This table is helpful. I find that I still want to see a table that puts the data (results) from each of these papers together in one story - what does it all mean when taken together. How does the empirical data compare to modeled and hypothesized results? This comment applies to all the summaries..

Commented [JK37R36]: Address: Suggest a tabulation of data from reviewed studies.

Commented [bs38R36]: Tables tabulating treatment, response and type of study has been added to the questions section. These tables have been moved to an appendix.

Field Code Changed

Field Code Changed

Field Code Changed

Cravello 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 ₃ and NO ₂ 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 ₃ and NO ₂ 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 ⁻¹ .
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 ² (28.6–101.6) and 46 g/m (1.2–04.5), 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 woody debris 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=2), 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 56% higher during the Fall into control and buffered treatments than into the clearcut. Needle inputs remained 6% higher in the buffer and control sites through year 7, and 3-6% higher in year 8 than in the clearcut sites. Twig inputs into the control and buffered sites were ~25% 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).

Field Code Changed

Field Code Changed

Field Code Changed

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 = 0.0024) 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	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);	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 32% 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	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

Field Code Changed

Field Code Changed

Field Code Changed

			detected with turbidity meters.		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-e/D) 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 24.4% 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 m2 ha-1.	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 63% 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

Field Code Changed

Field Code Changed

Field Code Changed

					mean annual values of stream water chemistry among different watersheds
Young 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.

Field Code Changed

507 Large Wood (LW)/wood load/wood recruitment

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

532 A Review of the Available Literature Related to Wood Loading Dynamics in and around Streams
533 in Eastern Washington Forests, was developed for CMER in October of 2004 (CMER 03-308,
534 2004). In this review, the researchers sourced 14 references with quantitative and descriptive
535 information relating to the correlation between wood volume and pieces of wood in streams and
536 the adjacent riparian community. The authors conclude that while the literature was incomplete,
537 several significant correlations existed between LW in streams and riparian zone stand
538 characteristics. For unmanaged (defined as unlogged and un-roaded) sites in Washington,
539 researchers reported positive correlations between the volume of LW in streams with adjacent
540 riparian zone mean tree height ($P < 0.001$), mean tree diameter ($P < 0.001$), and mean basal area
541 ($P < 0.001$). For numbers of LW pieces, positive correlations were found with the basal area
542 ($P < 0.007$) but no other vegetation characteristic of the adjacent riparian area. However,
543 regression analysis showed a significant positive correlation of LW piece quantity with core zone
544 trees/acre ($P < 0.001$, $R^2 = 0.45$) and core zone basal area/acre ($p = 0.004$, $R^2 = 0.29$). Relative to
545 managed riparian areas, streams adjacent to unmanaged riparian areas had significantly higher
546 LW volume. The most relevant sources of these results listed in this review were from Fox
547 (2001), Chesney (2000), Camp et al. (1997), and Knight (1990). Two other studies named in this
548 review (McDade et al., 1990; Fox, 2003) show evidence that as much as half of the wood found

Commented [AJK39]: Which of these factors was more important?

Commented [AK40R39]: ADDRESS

Commented [bs41R39]: Responses included

Commented [AJK42]: The evidence for salmonid population responses to LWD is equivocal...please see <https://cdnsiencepub.com/doi/10.1139/cifas-2014-0344> for a flavor of the overall debate.

Without question, LWD shapes the physical structure of streams and creates salmonid habitat. The challenge is to determine, in a watershed, whether physical structure is the factor limiting fish population growth by influencing recruitment and/or survival.

Commented [AK43R42]: ADDRESS

Commented [bs44R42]: Addressed

Commented [AJK45]: Was this effort based on empirical data?

Commented [AK46R45]: ADDRESS

Commented [bs47R45]: Addressed

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

551 In the western United States, several notable studies since 2000 have continued to investigate
552 and refine the factors important for LW recruitment. For example, Wing & Skaugset (2002)
553 investigated the relationships between land use, land ownership, and channel and habitat
554 characteristics with LW quantity and volume in stream reaches in western Oregon. The relevant
555 results (those derived for forested streams only) showed that stream gradient was the most
556 important explanatory variable for in-stream LW volume with the split in the regression analysis
557 occurring at 4.7%. Stream reaches with gradients less than 4.7% had on average less than half
558 the in-stream LW volume (11.3 m³ vs. 25.2 m³ per reach) than reaches with gradients >4.7%.
559 Results for LW pieces (logs at least 0.15 m diameter, and 3 m long) per 100 m length showed
560 bankfull width (BFW) as the most important explanatory variable with a split in the regression
561 analysis occurring at 12.2 m BFW. Reaches with a BFW <12.2 m averaged 11.1 LW pieces per
562 100 m compared to wider streams which averaged 4.9 pieces per 100 m. When the analysis was
563 constrained to “key” LW pieces (logs at least 0.6 m diameter and 10 m long), stream gradient
564 again emerged as the most important explanatory variable with the split in the regression
565 occurring at 4.9% stream gradient (mean key pieces per 100 m were 0.5 and 0.9 for gradients <,
566 and >4.9%, respectively). Following stream gradient and BFW, lithology was also an important
567 explanatory variable showing splits for Mesozoic and sedimentary lithologies (in 3 out of 4
568 analyses) grouped as containing half the LW quantity (pieces, key pieces, volume) on average
569 than all other geologies (basalt, cascade, and marine sedimentary geologies). Wing & Skaugset
570 (2002) suggests that geomorphic characteristics, in particular stream gradient and bankfull width,
571 but also underlying lithology in forested areas correlate best with LW presence in headwater
572 streams of western Oregon.

573 Another study from the Oregon Coast Range, May & Gresswell (2003), compared LW
574 recruitment processes between small colluvial channels and larger alluvial channels. Results
575 from this study showed that LW derived from local hillslopes and riparian areas accounted for
576 the majority of pieces (63%) in small colluvial channels. In contrast, the larger alluvial channel
577 received wood from a greater variety of sources, including recruitment from local hillslopes and
578 riparian areas (36%), fluvial redistribution (9%), and debris flow transported wood (33%).
579 Further, distributions of the source distance of wood pieces were significantly different between
580 colluvial and alluvial channels. In colluvial streams, 80% of total wood and 80% of total wood
581 volume recruited to colluvial streams originated from trees rooted within 50 m of the channel. In
582 the alluvial channel, 80% of the pieces of wood and 50% of the total volume originated from
583 trees which came from within 30 m of the channel. Considering the mechanisms responsible for
584 recruitment, for both colluvial and alluvial stream channels, slope instability exhibited the
585 longest source distance (median source distance = 40 m), followed by windthrow (median source
586 distance = 20 m), then natural mortality (median source distance = 18 m), and for obvious
587 reasons, bank erosion had the shortest median source distance (2 m). Compared between channel
588 types (colluvial vs. alluvial), the median source distance of wood recruited by windthrow was
589 significantly greater in colluvial channels than in the alluvial channel ($p < 0.05$). Source
590 distances for all other processes did not differ significantly between channel types. May &

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

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

606
607 Fox & Bolton (2007) modeled LW values from 150 stream segments located in unmanaged
608 watersheds, across Washington, with landscape, reach, and stand characteristics to understand the
609 central tendency of instream LW values in “natural” fish-bearing streams. Outputs from their
610 models show evidence that in-stream wood volume (m³ per 100 m stream length) and LW piece
611 count for streams up to 20 m in bankfull width (BFW) increased with drainage area and as
612 streams became less confined with BFW being a significantly better predictor of wood
613 parameters than basin size. Also, in-stream wood volume increased with adjacent riparian timber
614 age as determined by the last stand replacing fire. In this study (Fox & Bolton, 2007), the authors
615 noted that other predictor variables (e.g., gradient, bedform) also showed some evidence of an
616 effect but the variability of these variables were too great to evaluate with confidence.

617
618 Hough-Snee et al. (2016) reported similar issues with their results using Random Forest (RF)
619 models developed from field data to identify relationships between hydrogeomorphic and
620 ecological attributes that influence instream wood accumulation. Final RF models explained
621 43.5% of the variance in volume and 42.0% of the variance in frequency of in stream wood
622 loads. Mean annual precipitation, riparian large tree cover, and watershed area were estimated as
623 the most important predictors of in stream wood loads. However, so did individual watershed
624 which showed there was an interaction with site (i.e., site conditions unaccounted for may be
625 affecting the response). Given the heterogeneous results across all sub-basins studied, the authors
626 conclude by emphasizing the importance of incorporating local data and context when building
627 wood models to inform future management decisions.

628
629 Multiple studies have also investigated the effects of timber harvest under varying riparian
630 management zone prescriptions on LW recruitment. Specific to Washington, Schuett-Hames and
631 Stewart (2019a) compared in stand structure, tree fall rates, and LW recruitment between riparian
632 management zones harvested under the current standard Shade Rules (SR), the All-Available
633 Shade Rule (AAS), and unharvested references for fish-bearing streams in the mixed conifer

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

653 Four similar studies conducted for non-fish bearing streams in western Washington compared
654 changes in LW recruitment and stand mortality following harvest (Ehinger et al., 2021; McIntyre
655 et al., 2021; Schuett-Hames et al., 2011; Schuett-Hames et al., 2019b. Schuett-Hames et al.,
656 (2011) and Schuett-Hames & Stewart(2019b) investigated changes in riparian stand mortality
657 and LW recruitment into the bankfull channel 5- and 10-years post-harvest, respectively.
658 Treatments for riparian forests adjacent to non-fish-bearing streams evaluated in these studies
659 include clearcut to stream edge, upland clearcut with a 50-foot no cut buffer, and these were
660 compared to unharvested reference streams. Results showed that tree fall rates (annual fall rates
661 of live and dead standing stems combined) was over 8 times and 5 times higher in the 50-foot
662 buffers than in the reference buffers 3 years after treatment when compared as a percentage of
663 standing trees and as trees/acre/yr, respectively. These differences were significant for both
664 metrics ($p \leq 0.001$). Total tree-fall rates in the period 4-5 years after treatment, while still higher
665 in the 50-foot buffers was not significant.

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

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

678 For LW recruitment into the bankfull channel, results showed during the first three years after
679 treatment recruitment rates were 8 times and 14 times higher in the 50-foot buffers than in the
680 reference buffers respectively. The differences in pieces/acre/year and volume/acre/year between
681 reference and 50-foot buffers were significant. In years 4-5 after harvest LW recruitment
682 decreased in the 50-ft buffers and increased in the reference patches, and the number of recruited
683 LW pieces/acre/yr was greater in the reference patches, although the volume of LW recruited was
684 greater in the 50-ft buffers. Differences in recruitment rates between the 50-foot buffer and the
685 reference buffers for the 4–5-year period were not significant. For the entire first 5 years after
686 harvest, the 50-ft buffers recruited about twice the number of LW pieces recruited in the
687 reference patches, and over 3 times the volume; differences were marginally significant.

688 The results of the 10-year follow-up study for these sites (Schuett-Hames & Stewart, 2019b)
689 showed that stand mortality in the 50-foot buffer sites had stabilized and showed a cumulative
690 14.1% reduction in live basal area, while the reference stands showed a 2.7% increase in live
691 basal area. The differences in these values were not significant. Cumulative LW recruited into the
692 stream channel over the 10-period was double in the 50-ft treatment streams compared to the
693 reference streams. However, the majority of the LW recruited in the 50-ft treatment streams came
694 to rest above the streams, providing shade but not affecting streamflow, pool formation, or
695 sediment storage. Further, while the 50-ft buffer treatment provided more LW recruitment in the
696 short-term (10-years), the authors speculate there is a reduction in future LW recruitment
697 potential given the removal of trees outside the 50-ft buffer.

698 Two other studies which evaluated changes in LW following riparian forest harvest along non-
699 fish-bearing streams in western Washington were complimentary studies. Treatment sites in these
700 studies were underlain by either competent (McIntyre et al., 2021; also referred to as Phase 2 of
701 the “Hard Rock” study), or incompetent (easily eroded) marine sedimentary lithologies (Ehinger
702 et al., 2021; also referred to as the “Soft Rock” study). The buffer treatments evaluated for these
703 studies were compared against unharvested reference sites ("REF") and included a two-sided 50-
704 ft wide riparian buffer along the entire reach (“100%”), and the standard Forest Practices
705 treatment (FP), a two-sided 50-ft wide riparian buffer along at least 50% of the RMZ (buffered
706 and unbuffered portions were analyzed separately; hereafter referred to as FPB for the buffered
707 portion, and 0% for the unbuffered portion). However, because of unstable slopes in some of the
708 sites in the Soft Rock study (Ehinger et al., 2021), many of the buffers were required to be wider
709 than 50-feet (ranging from 18–160% wider than 50-feet). Conversely, some of the sites treated
710 ended up with buffers narrower than 50 feet. Further, there was limited availability of sites that
711 fit the criteria (marine sediment lithology, timing of treatment). Because of these limitations,
712 statistical analysis and comparison of LW response between treatments and references could not
713 be performed. Thus, the results are only descriptive, but they provide useful information for
714 comparison to the Hard Rock study.

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

717 contrast, mean post-harvest mortality in the full buffer sites and the <50 ft buffer sites were 31
718 and 25% of density, respectively. However, there was considerable variation in mortality among
719 sites, exceeding 65% in two full buffer treatment sites. Windthrow and physical damage from
720 falling trees accounted for ~75% of mortality in the full and <50 ft buffers. In contrast to the
721 treated sites, <10% of trees died due to wind or physical damage in the reference sites. For LW
722 recruitment, there was an increase in pieces of LW per 100 m length of stream in the full buffers
723 (8%) and the unbuffered treatments (13%) and a decrease in the streams adjacent to buffers < 50
724 feet wide (-15%) 3 years after harvest. The Hard Rock study did not require changes to the
725 grouping of treatments (i.e., all treatment buffers were harvested as described above; e.g.,
726 Reference, 100%, FPB, 0%). Also, the Hard Rock study collected up to 9 years of post-harvest
727 data that allowed for the comparison of LW changes over time pre- to post-harvest, and between
728 treatments.

729 Results for the Hard Rock study showed that by year 8 post-harvest mortality as a percentage of
730 pre-harvest basal area was lower in the reference (16.1%) than in the 100% (24.3%) and FPB
731 (50.8%) treatments. The FPB–Reference contrast in mortality was not significant 2 years post-
732 harvest, but it was at 5- and 8-years post-harvest as mortality in FPB increased relative to the
733 Reference over time. The contrast in mortality between the 100% and Reference were not
734 significant for any time interval 8 years post-harvest. Wind/physical damage was the primary
735 cause of mortality for all treatments, including the Reference. In the 100% treatment it accounted
736 for 78% and 90% of the loss of basal area and density (trees/ha), respectively; in FPB it
737 accounted for 78% and 65% of the loss. Wind accounted for a smaller proportion of mortality in
738 the Reference RMZ (52% and 43%, respectively). LW recruitment to the channel was greater in
739 the 100% and FPB RMZs than in the reference for each pre- to post-harvest time interval. Eight
740 years post-harvest mean recruitment of large wood volume was two to nearly three times greater
741 in 100% and FPB RMZs than in the references. Annual LW recruitment rates were greatest
742 during the first two years, then decreased. However, there was a great deal of variability in
743 recruitment rates within treatment sites and the differences between treatments were not
744 significant. Mean LW loading into the channel (pieces/m of channel length) differed significantly
745 between treatments in the magnitude of change over time. There was a 66%, 44% and 47%
746 increase in mean large wood density in the 100%, FP and 0% treatments, respectively, in the first
747 2 years post-harvest compared with the pre-harvest period and after controlling for temporal
748 changes in the references. By year 8, only the FP treatment showed a significantly higher
749 proportional increase (41%) in wood loading when compared to the reference. In the time
750 interval 2-8 years post-harvest wood loading in the 100% treatment stabilized and began to
751 decrease in the 0% treatment.

752 The Hard Rock and Soft Rock studies showed similar results. Both studies showed an increase in
753 stand mortality that also led to an increase in LW recruitment into the channels adjacent to 50-
754 foot (and greater in the Soft Rock) buffer treatments relative to unharvested reference sites.
755 However, the longer time period of study in the Hard Rock study showed mortality and thus LW
756 recruitment began to stabilize after year five. The results presented by Schuett-Hames (2012,
757 2019b) showed a similar pattern of an initial increase in mortality rates and LW recruitment rates
758 in treated stands relative to untreated stands within three years of treatment, but stabilization

759 within 5-10 years. Unfortunately, because of the limitations in sample size and buffer width
760 consistency in the Soft Rock study, confident conclusions on the effects of lithological
761 competency on LW recruitment post-harvest cannot be drawn.

762 All studies reviewed above which investigate the effect of timber harvest with riparian buffers
763 show that the initial increase in mortality within treatment buffers relative to reference buffers is
764 primarily a result of increased windthrow mortality. Liquori (2006) found similar results in an
765 investigation of treefall characteristics within riparian buffer sites ranging in width from 25-100
766 feet along non-fish bearing and fish bearing streams. Within no-cut buffers, windthrow caused
767 mortality was up to 3 times greater than competition induced mortality for 3 years following
768 treatment with tree fall probability highest in the outer areas (closest to upland clearcuts) of the
769 buffers. Their results showed that treefall was generally highest at the outside edges of buffers
770 (50+ feet), representing about 60% of the total observed treefall, while the 0–25-foot zone
771 represented ~18%, and the 25–50-foot zone represented ~22%. This suggests an increase in
772 windthrow susceptibility within riparian buffers with increasing distance from the stream.
773 Liquori (2006), however, did not differentiate thinning treatments applied to the outer zones of
774 the buffer in their analysis mentioning “very modest” thinning was applied to some buffers. They
775 suggest in their interpretation of the results that buffer thinning may influence the depth to which
776 wind forces can penetrate into the buffer. The results from Schuett-Hames & Stewart (2019a),
777 discussed above, show evidence that thinning in the outer area (30-75 feet from bankfull width)
778 changed the source distance curve of wood recruitment from fallen trees with thinned buffers
779 (SR treatments). The results exhibited statistically higher overall treefall rates with a larger
780 percentage coming from the outer area in the SR treatments than in the reference and more
781 lightly thinned (AAS) treatment buffers.

782 Outside of Washington, but in areas with similar habitats (Oregon, British Columbia) several
783 experimental studies that have investigated the effects of timber harvest on treefall, mortality,
784 LW recruitment, and LW source distance have found comparable results to those conducted in
785 Washington. For example, Martin & Grotefendt (2007) compared riparian stand mortality and in-
786 stream LW recruitment characteristics between riparian buffer strips with upland timber harvest
787 and riparian stands of unharvested watersheds using aerial photography in the northern and
788 southern portions of Southeast Alaska. All buffer strips in this study were a minimum of 20 m
789 wide and included selective harvest within the 20 m zone (thinning intensity not specified or
790 included in the analyses as an effect). The results from this study showed significantly higher
791 mortality (based on cumulative stand mortality: downed tree counts divided by standing tree
792 counts + downed tree counts), significantly lower stand density (269 trees/ha in buffer units and
793 328 trees/ha in reference units), and a significantly higher proportion of LW recruitment from the
794 buffer zones of the treatment sites than in the reference sites. Also, results showed that mortality
795 varied with distance to the stream. Differences in mortality for the treatment sites were similar to
796 the reference sites for the first 0-10 m from the stream (only a 22% increase in the treated sites).
797 However, mortality in the outer half of the buffers (10-20 m) from the stream in the treatment
798 sites was more than double (120% increase) what was observed in the reference sites. The
799 authors attribute the difference in cumulative stand mortality to the increase in windthrow

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

802

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

814 Burton et al. (2016) examined the relationship between annual in-stream wood loading and
815 riparian buffer widths adjacent to upland thinning operations. No-cut buffer widths were 6, 15, or
816 70 meters, and upland thinning was to 200 trees per ha (tph), with a second thinning (~10 years
817 later) to ~85 tph, alongside an unthinned reference stand ~400 tph. Their results showed that
818 slightly higher volumes of wood were found in sites with a narrow 6-m buffer, as compared with
819 the 15-m and 70-m buffer sites in the first 5 years after the first harvest and maintained through
820 year 1 of the second harvest (end of study). The authors attributed this difference to a higher
821 likelihood of logging debris and/or windthrow, but these factors were not analyzed. Considering
822 source distance, the authors used a mixed modeling approach to assess the relationship between
823 wood volume and source distance for in-stream wood with an identifiable source. This model
824 was only applied to the 70-meter buffer. The results showed that 82-85% of the wood with
825 discernable sources (90% for wood in early stages of decay; 45% of wood in late stages of
826 decay) came from within 15 m of the stream, and the relative contribution of wood to streams
827 declined rapidly with increasing distance. Still, these results are similar to those presented by
828 Schuett-Hames & Stewart (2019a) which showed the majority of the LW recruited (72-76% for
829 treated stands) into the channel were from within the first 30 feet (~9.1 m) of the stream even
830 though upland harvest prescriptions in this study differed from those evaluated by Burton et al.
831 (2016) (e.g., clearcut vs thinning).

832

833 *Summary of Factors Impacting LW Loads and Recruitment*

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

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

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

865 ~~Considering the influence of landscape and site factors on LW dynamics factors such as stand~~
866 ~~density (stems per unit area), basal area, stand age, stream bankfull width, stream gradient, valley~~
867 ~~constraint, lateral slope steepness, lithology, and mean annual precipitation have all been shown~~
868 ~~to influence LW recruitment and instream wood loads. Repeatedly, one or more of these factors~~
869 ~~have emerged as important predictor variables of LW dynamics in watersheds with and without~~
870 ~~management.]~~

Commented [JK48]: Yellow: There is a difference between modeled or simulated results and empirical results and this should be taken into account in this summary of findings. How do they compare, with the observed data presented? Again, a table that contains information with treatment and impact would be helpful for the reader.

Commented [JK49R48]: Address (as above)

Commented [bs50R48]: Tables tabulating treatment, response and type of study has been added to the questions section.

Commented [WB51]: This doesn't really say anything

Commented [WB52R51]: Address

Commented [bs53R51]: Removed

Commented [JK54]: Red: I agree with Welles, this paragraph adds no further information that isn't provided above.

Commented [JK55R54]: Address

Commented [bs56R54]: Removed

Table 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 & Measeon, 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, understorey 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 9 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 9 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 (RCWM) developed for the Alsea watershed in central coastal Oregon. Data was sourced from FIA.	Single entry thinning reduced in stream wood by 32 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	LW recruitment, in-stream wood volume, biomass, and	LW volume, LW characteristics and source evidence, reach	Wood surveys were carried out at four times during the study: (1) prior to the	In stream wood volume increased significantly with drainage basin area, for every 1 ha increase in drainage basin area, wood volume increased by 0.62%. 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

Commented [AJK57]: This table should be placed in an appendix.

Also, I would reconsider how much information is placed in the table...as it stands, it is less a summary table than massive blocks of text with lines around them.

Commented [AK58R57]: ADDRESS

Commented [bs59R57]: Moved to Appendix. Smaller tables outlining treatment and impact have been added to the Question sections.

	of buffer, all treatment stands were thinned first to 200 trees per hectare (tph), then again to 85 tph ≈ 40 years later. Uncut reference was ≈ 400 tph.		and stream characteristics:	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 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	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 (~90 years); (2) the stream-side forests were not disturbed by human activities, such as harvesting, road	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 2.2 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 ³), significantly higher than stream size I (0.06 m ³). LW density (pieces per 100 m ² of stream area), however, decreased as stream size increased. For example, LW density (defined as piece numbers per 100 m ²) numbers were 19, 17, 12, and 4 for stream size I, II, III, and IV respectively. Increases in channel bank full width (R ² = 0.52) and stream area (R ² = 0.50) was found to be strongly inversely correlated with LW density.

	variability of LW characteristics			building; (2) the streams were not salvaged.	
Ehinger et al., 2011	1) Buffers encompassing the full width (50 feet), 2) <50ft buffers, 3) Unbuffered, harvested to the edge of the channel, and 4) Reference sites in unharvested forests.			Soft Rock study. Only descriptive statistics were applied for changes in stand structure and wood loading. Small sample sizes.	There was little post harvest large wood input in reference sites: an average of 4.3 pieces and 0.34 m ³ of combined in- and over-channel volume per 100 m of channel. In contrast, the full buffer sites and <50 ft buffer sites received an average of 22 and 10 pieces/100 m and 2.2 and 0.7 m ³ /100 m of large wood, respectively. Piece counts remained stable in the reference sites through year 3 post harvest, increased in the full buffer and unbuffered sites (8 and 13%, respectively), and decreased in the <50 ft buffers (-15%).
Fox & Bolton, 2007	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 91-1,996 m, drainage areas 0.4-225 km ² , 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 ³)" (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 0.7 m ³ for the 20- to 30 m BFW class, 10.5 m ³ for the 30- to 50 m BFW class, and 10.7 m ³ for channels greater than 50 m BFW per 100 m length of stream.

Gomi et al., 2004	Five management or disturbance regimes: old growth (OG), recent clear cut (CC; 3 years), young conifer forest (YC; 27 years after clear cut), young alder (YA; 20 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.
Hough-Snee et al., 2016	In-stream wood volume and frequency were quantified across multiple sub-basins.	LW frequency and volume, hydrologic and geomorphic attributes	Models were calibrated with site characteristics from multiple riparian stands in the Columbia River Basin.	Results show a high level of variability between sub-basins studied. The overall model shows site (watershed) was an important predictor.	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.

Hyatt & Naiman, 2001	LW data was collected from multiple sites in the Queets River Watershed.	LW in stream and in riparian forests.	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.	The depletion constant was developed for a large, mostly alluvial river and should probably not be applied to smaller streams.	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 2 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.
Jackson & Wohl, 2015	in-stream wood volume and frequency were quantified along 23 pool-riffle or plane-bed stream reaches in the Arapaho and Roosevelt National Forests in Colorado.	Sediment storage, channel geometry, in-stream wood load, and forest stand characteristics	Wood loads, wood jam volumes, log jam frequencies, residual pool volume, and fine sediment storage around wood, stand age, and disturbance history.	Old growth defined as forests ≥ 200 years. Age range of young forests not reported. Sample sizes include 10 old growth and 23 younger forests.	Results indicated that channel wood load (OG = 304.4 ± 161.1 ; Y = 197.9 ± 245.5 m ³ /ha), floodplain wood load (OG = 109.4 ± 80 ; Y = 47.1 ± 52.8 m ³ /ha), and total wood load (OG = 154.7 ± 64.1 ; Y = 97.8 ± 100.6 m ³ /ha) 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 = 7.10 ± 6.9 m ³ ; Y = 1.71 ± 2.91 m ³). Although wood load in streams draining from pine beetle-infected 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.
Jackson et al., 2001	2 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	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.

	buffers were as thin as 2.3 m.				
Liquori, 2006	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 2 times greater than competition induced mortality for 2 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.
Martin & Grotfeldt, 2007	Buffer widths a minimum of 20 m. Multiple buffer widths and harvest intensities.	Instream wood load, stand mortality	Counts of downed wood, tree stumps, stand characteristics, instream wood from aerial photographs taken post-logging	Stand and stream characteristic, and LW data was surveyed from aerial photographs.	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.
May & Gresswell, 2003	Survey of LW in three second-order streams and the mainstem of the North Fork of Cherry creek.	LW, delivery mechanism	LW > 20 cm diameter, and > 2 m length was categorized by 4 delivery mechanisms; Delivery process, disturbance type, and channel characteristics.	Although mean edge of Douglas-fir trees was identified to be excess of 200 years old, further information on differences in stand structure or development stage between sites are not included.	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 20 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 (44%).
McIntyre et al., 2021	(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, (3) 0% treatment, clearcut to stream edge (no buffer).			Hard Rock Study Physical constraints such as a lack of suitable low gradient reaches and/or issues with accessibility related to weather limited downstream measurements of exports to just eight sites.	Large wood recruitment to the channel was greater in the 100% and FP 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 FP RMZs than in the references. Annual LW recruitment rates were greatest during the first two years, then decreased. However, these differences were 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 43% ($P = 0.08$), 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.
Melesson 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 is 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 > 20 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.
Newakowski & 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 ³ /100 m) when compared to unmanaged reference watersheds (3.3 m ³ /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 ($r^2 = 0.5749$). Shear stress was the best predictor of wood load in the managed watersheds ($r^2 = 0.2402$). 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 ³ /100 m) had, on average, 2-3 times lower in stream wood loads than unmanaged (3.3 m ³ /100 m) watersheds.

Reid & Hassan, 2020	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.
Schuett Hames & Stewart, 2019a	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.
Schuett Hames et al, 2011; Schuett Hames & Stewart, 2010b	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.

Sebota 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 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 98 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 20, 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 98 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 ³ , which was less than half of the average found at higher gradient reaches (25.2 m ³); in this model the stream gradient split explained 11% of the variation observed of in-stream 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 4.0%. The streams with a gradient < 4.0% averaged 0.5 key LW pieces per

					reach while streams with higher gradients averaged 0.0 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 DFW.
--	--	--	--	--	--

872

873

874

875 [Bank Stability and Sediment](#)

876 *Bank Stability*

877 Few studies could be found that quantify how riparian area harvest directly affects bank stability
878 or bank erosion based on our search criteria. Many studies published since 1999 that investigate
879 bank stability and bank erosion compare relative rates of erosion based on the presence/absence
880 of vegetation, type of vegetation (e.g., grassland vs. forest cover), and soil types or lithology
881 (Konsoer et al., 2015; Micheli et al., 2004; Simon & Collision, 2001; Wynn & Mostaghimi,
882 2006). Also, many studies have investigated the relative effects of different types of land use
883 (e.g., agricultural, urban, forested) as well as cattle grazing intensity (McInnis & McIver, 2009;
884 Zaines & Schultz, 2014). The only studies that could be found that provide some experimental
885 evidence as to how timber harvest within the riparian area affects bank stability or erosion come
886 from 3 CMER reports (Ehinger et al. 2021; McIntyre et al. 2018, Schuett-Hames et al., 2011;
887 Schuett-Hames & Stewart, 2019).

888 Schuett-Hames et al. (2011) investigated how soils and streambanks were disturbed following
889 harvest within the riparian area along perennial non-fish bearing streams (Type Np) in western
890 Washington. To evaluate post-harvest soil and stream bank disturbance, Schuett-Hames et al.
891 (2011) first described a soil erosion feature as areas of exposed soil that (1) had a surface area of
892 greater than 10 square feet, and (2) was caused by harvest practice (e.g., felling, bucking, or
893 yarding). If both criteria were met, the length, width, and distance to stream were recorded, and
894 evidence of sediment delivery to the stream was noted. The number of harvest related soil
895 disturbances were grouped by 100 ft lengths of stream, as were the number of features delivering
896 sediment to the stream. Disturbances along stream bank were quantified using the same methods.
897 The surface area (mean width x length) of disturbance features were used to estimate the percent
898 coverage of soil disturbance within 50-feet of bankfull width and in the equipment exclusion
899 zone (ELZ; within 30 feet of the bankfull width). Finally, the percent of harvested patches with a
900 greater than 10% coverage of soil disturbance features in the ELZ were also quantified
901 (performance target for bank stability). These methods were used to collect data for all 3 harvest
902 treatments. These harvest treatments included 1) a 50-foot wide no cut buffer, 2) clearcut, no
903 buffer, and 3) a 56-foot radius no-cut buffer surrounding the perennial initiation point (PIP). A
904 non-parametric, two-sample Mann-Whitney U test was used to test differences in mean soil and
905 stream bank disturbance metrics between the 50-foot buffer patches and the clearcut (no buffer)
906 patches. A Fisher's exact test was used to test for differences in the relative frequency of patches
907 exceeding the performance target (more than 10% of ELZ area disturbed by management related
908 activities) between 50-foot and the clearcut buffer prescriptions.

909 Results showed that the differences between the mean values of harvest related soil and
910 streambank disturbances for clear-cut patches and the 50-ft buffers were significant for all
911 metrics (e.g., # of bank disturbance features per 100 ft, # of soil disturbance features per 100 feet,
912 # of soil disturbance features, # of soil disturbance features delivering sediment to stream, % of
913 ELZ with soil disturbance; $P \leq 0.082$). Results for soil disturbance performance targets showed
914 that all of the 50-foot buffer and PIP prescriptions met the performance targets (i.e., maintained
915 <10% harvest-related soil disturbance in the ELZ). One clearcut patch exceeded the 10%

916 coverage performance target. The difference between clearcut patches and 50-foot buffer patches
917 was significant ($p = 0.007$). The average size of harvest related soil disturbances that delivered
918 sediment to streams was 752 ft² (range: 31-9060 ft²). The average size of soil disturbance
919 features that did not deliver sediment to streams was 65 ft² (range: 13 – 214 ft²). Delivery of
920 sediment to streams was best predicted by the horizontal distance between the soil disturbance
921 and the stream channel ($P < 0.0001$). The average distance to the stream for soil disturbance
922 features that delivered sediment was 1 ft (max. = 7.7), while the average distance for non-
923 delivering soil disturbance features was 14 ft (min 3.3). Using distance-to-stream alone, 96% of
924 the observations were correctly predicted based on whether the horizontal distance to the stream
925 was greater or less than 5.4 ft ($R^2 U4 = 0.80$). The authors concluded there were more harvest-
926 related soil disturbances following harvest in the clear-cut patches than the 50-ft buffers. Further,
927 that the management practices for the 50-foot and PIP buffers were sufficient at maintaining
928 bank stability performance targets. The clearcut patches were mostly sufficient at maintaining
929 performance targets with the exception of one site.

930 Schuett-Hames et al. (2011) also collected data on soil disturbance associated with post-harvest
931 root pits created from trees being uprooted by wind or other disturbances. Four metrics were
932 used to evaluate soil disturbance associated with uprooted trees: *Root-pits per acre*. Root-
933 pits/acre was calculated by tallying the number of root-pits in each patch and dividing by the
934 patch acreage. *Root-pits per 100 ft of stream length*. Root-pits/100 ft of stream length was
935 calculated by tallying the number of root-pits in each patch (both sides of the stream), dividing
936 by the stream length, and multiplying by 100. *Root-pits with sediment delivery per acre*. Root-
937 pits/acre with evidence of sediment delivery to the channel was calculated by tallying the number
938 of root-pits where evidence of sediment delivery to the stream channel is observed in each patch
939 and dividing by the patch acreage. *Root-pits with sediment delivery per 100 ft of stream length*.
940 Root-pits with sediment delivery/100 ft of stream length were calculated by tallying the number
941 of root-pits with evidence of sediment delivery in each patch (both sides of the stream), dividing
942 by the stream length, and multiplying by 100. These metrics were measured 3 years and 5 years
943 following harvest to give an annual rate of change for each metric at 3 years, from 3-5 years, and
944 for the entire 5 years. These standardized annual rates were compared between each treatment
945 patch type and a unharvested reference patch of the same size.

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

958 pattern was similar for root-pits with sediment delivery, however the difference between the
959 reference and buffer patches was less pronounced due to the higher percentage of root-pits
960 delivering sediment in the reference patches. The percentage of root-pits with evidence of
961 sediment delivery was much higher in the clear-cut patches than in the 50-ft buffers (20.1%) and
962 the reference (26.0%) patches but was not significantly different. Results for the PIP buffers
963 showed a similar trend as the 50-foot buffers with an increase in root pits delivering sediment to
964 the stream in the first three years, but a sharp decline after the third year. Over the course of the
965 full five years Over the entire 5 year period, the percentage of root-pits with evidence of
966 sediment delivery in the PIP buffers (17.6%) was similar to the percentage for the 50-ft buffers
967 (19.8%). These values did not differ significantly from the references.

968 The authors also investigated the factors affecting whether the post-harvest root pits delivered
969 sediment to streams for 2006 and 2008 (3 and 5 years post-harvest). In both years, sediment
970 delivery to streams was best predicted by the distance of the root-pit from the stream ($P <$
971 0.0001). Mean horizontal distance to the stream for root-pits that delivered sediment was 8.2 ft
972 compared to 28.0 ft for those that did not deliver. Using horizontal distance to stream, the
973 proportion of the total uncertainty that was attributed to the model fit was 0.39, and 80% of the
974 observations were correctly predicted based on whether the horizontal distance to stream was
975 greater or less than 12.5 ft. Width of root pits delivering soil to the stream were also larger on
976 average but its inclusion to the model did not increase fitness. The authors speculate from their
977 observations that the higher tree-fall rates in the 50-foot buffer during the first 3 years after
978 harvest was due to an increase in wind-throw. However, in the second time period the reference
979 patches showed an increase in windthrow following stronger storms during the 2006-2008
980 period. One of the two reference streams did show string evidence of mass wasting.

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

993 McIntyre et al. (2018; Hard Rock Study) also investigated post-harvest surface erosion following
994 harvest along Type Np streams (same prescriptions as Schuett-Hames, 2011) on competent
995 lithologies in western Washington. They conducted visual surveys to identify recently eroded
996 areas (source of erosion not discerned) in the treated riparian areas that were 10 m² or larger.
997 Post-harvest stream-delivering surface erosion was documented at 11 of 17 sites observed. The
998 total erosion area exceeded 110 m² at 5 of the 17 sites: 2 reference sites, 2 50-foot buffer sites,

999 and 1 clearcut sites. At these five sites, post-harvest surface erosion was evident adjacent to only
1000 1.5 to 4.6% (average = 2.2%) of the total stream channel length (including both mainstem and
1001 tributaries). At the remaining study sites where stream-delivering erosion events occurred, the
1002 total eroded area was 60 m² or less and occurred adjacent to 0.3% to 0.8% (average = 0.6%) of
1003 the stream channel length. There were no statistically significant differences in stream-delivering
1004 surface erosion among treatments ($\alpha = 0.05$), and on average, reference and buffer treatments
1005 visually exhibited a similar amount of exposed bank.

1006 The researchers also investigated the frequency of uprooted trees that developed root pits during
1007 the post-harvest period. The average rate of root pits developed in the 50-foot buffers was
1008 approximately 3 times higher (3.6 pits/ha/yr) than in the reference sites (1.2 pits/ha/yr) for 3
1009 years following harvest. However, year to year values were highly variable with reference sites
1010 showing higher numbers of root pits per acre than either buffer treatment in the first year
1011 following treatment (27.4 vs. 18.5 vs. 6.4 for reference, 50-foot, and clearcuts respectively).

1012 The results of the above studies on bank and riparian surface erosion after harvest show some
1013 evidence that bank erosion and soil disturbance is generally higher in treated areas than in
1014 untreated areas. Further, that bank erosion is likely higher in clearcut treatments without buffers
1015 than in treatments with no-cut buffers. However, development of root-pits (with and without
1016 sediment delivery pathways to streams) are more likely in treatments with no-cut buffers which
1017 is likely because no trees were left in the clearcuts to be toppled. When compared to a reference,
1018 the trends of surface erosion and soil disturbance shows there is generally an increase in the
1019 treated buffers within the first few years. However, these differences appear to stabilize within
1020 five years. Finally, soil disturbance and bank erosion (especially when caused by windthrow) are
1021 highly variable and in many instances (e.g., Ehinger et al. 2021; McIntyre et al. 2018) do not
1022 exceed the natural range of variability found in reference streams.

1023 Sediment~~Nutrients~~

1024 The function of riparian areas to regulate and filter the flow of sediments into streams is essential
1025 not only for water clarity and pool formation but also because of the ability of sediments to carry
1026 nutrients and pollutants (Cooper et al., 1987; Hoffman et al., 2009; Polyakov et al., 2005). .

1027 Sediment flux into streams can be affected by landscape factors, streamflow, vegetation
1028 composition, and disturbance including riparian and adjacent upland forest management
1029 (Crandall et al., 2021; Devotta et al., 2021; Vanderbilt et al., 2003). The movement of sediment
1030 into the active channel can, in turn, impact aquatic habitat and geomorphic processes, especially
1031 in small, forested streams (Benda et al. 2005; Gomi et al., 2005; Hassan et al., 2005).

1032 The effects of riparian area timber harvest on sediment flux into streams has been documented,
1033 investigated, and incorporated into riparian forest management plans in western North America
1034 since the 1970s with the development of the Clean Water Act of 1972 (Bilby et al., 1989;
1035 Gregory 1990; Gresswell et al., 1989; Naiman et al., 1998; Salo & Cundy, 1986; Swanson et al.,
1036 1982; Swanson & Dyrness, 1975). Prior to the Forests and Fish Report (FFR 1999), several
1037 studies from western North America investigated the effects of riparian zone timber harvest
1038 practices on sediment flux into streams.

Commented [WB60]: Should be sediment

Commented [WB61R60]: Address

Commented [bs62R60]: Corrected

1039 Specific to Washington, Rashin et al. (2006) evaluated the effectiveness of Washington State best
1040 management practices (BMPs) for controlling sediment related water quality impacts. Although
1041 this study was published in 2006, the data analyzed in this study were collected between 1992
1042 and 1995. In their evaluation, Rashin et al. (2006) assessed site erosion, sediment delivery,
1043 channel disturbance, and aquatic habitat condition within the first two years of harvest along
1044 fish- and non-fish bearing streams across Washington state. From their results, the authors
1045 concluded that the site-specific factors influencing the effectiveness of BMPs in preventing
1046 chronic sediment delivery into streams were 1) the proximity of ground disturbance to the
1047 stream, 2) presence of a stream buffer, 3) falling and yarding practices that minimized
1048 disturbance to stream channel, and 4) timing of harvest activities for certain climate zones where
1049 frozen ground or snow cover may be exploited. The landscape factors that influenced BMP
1050 effectiveness were 1) the density (specific metric not reported) of unbuffered small streams at
1051 harvest sites, and 2) steepness of stream valley slopes. The authors conclude with a
1052 recommendation of excluding timber falling and yarding activities at least 10 m from streams
1053 and outside of steep inner gorges.

1054 Similar results were reported by Lewis (1998) in their evaluation of logging activities' effect on
1055 erosion and suspended sediment transport in the Caspar Creek Watersheds of northwestern
1056 California. From their results the authors concluded that the dominant factors influencing the
1057 difference in suspended sediment loads between watersheds was the difference in road
1058 alignment, yarding methods, and presence of stream protection zones (i.e., buffers). Because of
1059 studies like these reviewed, contemporary riparian forest management practices in the western
1060 United States include rules that limit harvesting, use of equipment, and procedures that disturb
1061 soil in areas closest to the stream or on steep and unstable slopes ([WAC 222-30-022](#); WAC 22-
1062 30-021; 2022 [ODF](#); IDAPA 20.02.01)

1063 Since 2000, many of the studies published that evaluate changes in sediment delivery or water
1064 turbidity following riparian timber harvest show similar results in that contemporary BMPs are
1065 effective in mitigating increases in sediment delivery to streams (Hatten et al., 2018; Reiter et al.,
1066 2009). For example, the studies reviewed that report a significant change in sediment delivery
1067 following harvest show evidence that these changes only persist for a short period of time (1-3
1068 years) and that the magnitude of these changes are related to the intensity of the harvest
1069 prescriptions (Karwan et al., 2007; Macdonald et al., 2003a).

1070
1071 For example, Macdonald et al. (2003a) compared changes in stream discharge rates and in-
1072 stream suspended sediment concentrations during spring snowmelt between two harvest
1073 intensities and one unharvested control, for pre- and post-harvest in first order streams of interior
1074 British Columbia. Both treated riparian areas received a harvest of 55% of the watershed; one
1075 (low-retention) removed all merchantable timber >15 cm DBH for pine and > 20 cm DBH for
1076 spruce within 20 m of the stream; the other (high-retention) removed all merchantable timber >
1077 30 cm within 20 m of the stream. The results showed an increase in spring snowmelt discharge
1078 for both treatments above predicted values for the study (5 years). However, increased in-stream
1079 total suspended sediments (TSS) only persisted for two-years post-harvest in the high-retention
1080 treatment, and for 3-years in the low-retention.

1081
1082 Karwan et al. (2007) investigated the effects of riparian timber harvest and road construction on
1083 TSS concentrations in the Mica Creek Experimental Watershed in northern Idaho. Treatments in
1084 the paired-watershed experiment consisted of 1) commercial clearcut of the watershed area by
1085 50%, and was broadcast burned and replanted, 2) partial cut in which half the canopy was
1086 removed in 50% of the watershed area 3) a no-harvest control. All harvests were done according
1087 to best management practices and the Idaho Forest Practices Act. This included equipment
1088 exclusion zones of 50- and 30-feet for fish- and non-fish-bearing streams, respectively. On all
1089 skid trails, drainage features, such as water bars, were installed for erosion control at the end of
1090 the harvest period. Results showed that road construction in both watersheds did not result in
1091 significant impacts on monthly sediment loads in either treated watershed during the immediate
1092 (1-year post-harvest) or recovery (2-4 years post-harvest) time intervals. A significant and
1093 immediate impact of harvest on monthly sediment loads in the clear-cut watershed ($p = 0.00011$),
1094 and a marginally significant impact of harvest on monthly sediment loads in the partial cut ($p =$
1095 0.081) were observed. However, after one year, the TSS loads in both treatments became
1096 statistically indistinguishable from the control.

1097
1098 Specific to Washington, McIntyre et al. (2021) evaluated the effectiveness of riparian buffers on
1099 non-fish-bearing streams underlain by competent lithologies (“Hard Rock”) in western
1100 Washington. Buffers were treated with one of three prescriptions 1) unharvested reference, 2) a
1101 two-sided 50-ft riparian buffer along the entire riparian management zone (RMZ), 3) a two-sided
1102 50-ft riparian buffer along at least 50% of the RMZ, and 4) clearcut to stream edge (no-buffer).
1103 Results for suspended sediment export (SSE) following treatment showed episodic increases
1104 with storm events that rapidly declined. However, changes in SSE were poorly correlated with
1105 discharge and exhibited high variation between treatment sites. The authors suggest that these
1106 results show evidence that changes in SSE magnitudes were not related to harvest. Further, they
1107 conclude that the sites were likely sediment-limited considering the underlying lithology.

1108
1109 Site factors such as underlying lithology and physiography can interact with the effect of timber
1110 harvest operations on sediment delivery into streams. Bywater-Reyes et al. (2017) assessed the
1111 influence of natural controls (basin lithology and physiography) and forest management on
1112 suspended sediment yields in temperate headwater catchments in northeastern Oregon. Results
1113 from this study indicate that site lithology was the first order control over suspended sediment
1114 yield (SSY) with SSY varying by an order of magnitude across lithologies observed.
1115 Specifically, SSY was greater in catchments underlain by Siletz Volcanics ($r = 0.6$), the Trask
1116 River Formation ($r = 0.4$), and landslide deposits ($r = 0.9$) and displayed an exponential
1117 relationship when plotted against the percentage of watershed area underlain by these lithologies.
1118 In contrast, lithology had a strong negative correlation with percent area underlain by diabase (r
1119 $= 0.7$), with the lowest SSY associated with 100% diabase. Following timber harvest, increases
1120 in SSY occurred in all harvested catchments but returned to pre-harvest levels within 1 year
1121 except for sites that were underlain by sedimentary formations and were clearcut without
1122 protective buffers. The authors conclude that sites underlain with a friable lithology (e.g.,

1123 sedimentary formations) had, on average, SSYs an order of magnitude higher following harvest
1124 than those on more resistant lithologies (intrusive rocks).

1125 Mueller & Pitlick, (2013) found similar results in their assessment of the relative effect of
1126 lithology, basin relief, mean basin slope, and drainage density on in stream sediment supply for
1127 83 drainage basins in Idaho and Wyoming. The strongest correlation of in stream sediment
1128 supply was with lithology relative softness (based on grouping of rock types – granitic,
1129 metasedimentary, volcanic, and sedimentary). Sediment concentrations at bankfull width
1130 increased by as much as 100-fold as basin lithology became dominated by softer sedimentary
1131 and volcanic rock compared to lithologies dominated by harder granitic and metasedimentary
1132 rock. Finally, Wissmar et al. (2004), developed and field-tested erosion risk indices for
1133 watersheds in western Washington based on land cover. These erosion risk indices used the
1134 presence of unstable soils (determined by geological formation and underlying lithology), rain-
1135 on-snow events, immature forest cover (stands <35 years old where open canopies and
1136 undeveloped root systems could contribute to hillslope instability), presence and coverage of
1137 roads, and critical slope (hillslope gradients >36%, for terrain with surficial deposits of coarse-
1138 textured colluvial materials). Results of this study showed these variables could explain ~65% of
1139 the variation associated with sediment input into channels. The lowest risk areas contained the
1140 fewest of these variables (most commonly critical slope with either rain-on snow events or
1141 immature forests), while higher risk areas contained a combination of 4 or more of these factors
1142 indicating a compounding effect.

1143 Changes in sediment yield may also interact with increases in discharge rates caused by timber
1144 harvest as well as physiographic site factors. For example, Bywater-Reyes et al. (2018)
1145 quantified how sediment yields vary with catchment lithology and physiography, discharge, and
1146 disturbance history over 60 years in the H.J. Andrews experimental watershed in the western
1147 Cascade Range of Oregon. Methods for determining suspended sediment concentration involved
1148 using either vertically integrated storm-based grab samples, or discharge-proportional composite
1149 samples where composite samples were collected every three weeks at the outlet of each
1150 catchment. Data sets were taken from 10 watersheds, 7 with a history of management (mixture of
1151 selective canopy removal, patch-cut, 25-100% clearcut, broadcast burning, road building, and
1152 thinning), and 3 with no history of management that were used as a reference. A linear mixed
1153 effects model (log transformed to meet the normality assumption) was used to predict annual
1154 sediment yield. In this model, site was treated as a random effect while discharge and
1155 physiographic variables were treated as fixed variables. This allowed for the evaluation of the
1156 relationships between sediment yield and physiographic features (slope, elevation, roughness,
1157 and index of sediment connectivity) while accounting for site. To account for the effect of
1158 disturbance history a variable was added to the model when the watershed had a history of
1159 management or natural disturbances. If the models for the disturbed watersheds significantly
1160 underpredicted the sediment discharge, the timing of the sudden increases were further examined
1161 to assess whether it correlated with a disturbance event (e.g., harvesting, road building, and
1162 slash-burning.) The results of this study show that watershed physiography combined with
1163 cumulative annual discharge explains 67% of the variation in annual sediment yield across the
1164 60-year data set regardless of lithology. Relative to other physiographic variables, watershed

1165 slope was the greatest predictor of annual suspended sediment yield. However, the results
1166 showed that annual sediment yields also moderately correlated with many other physiographic
1167 variables and caution that the strong relationship with watershed slope is likely a proxy for many
1168 processes, encompassing multiple catchment characteristics.

1169 In contrast, Safeeq et al. (2020) compared instream and bedload sediment supply under multiple
1170 harvesting treatments in watersheds of western Oregon that were paired with control watersheds
1171 by size, aspect, and topography. The treatment watershed was 100% clearcut during the period
1172 from 1962-1966, broadcast burned in 1966, and re-seeded in 1968. For this study 15-minute
1173 streamflow data was recorded for both watersheds, and after large storm events. Sediment data
1174 was collected from 1952 (pre-harvest) through 1988 for suspended sediment data, and 2016 for
1175 sediment bedload. The control watershed was forested, and had no treatments (e.g., harvest)
1176 during the study period. Their results estimate that following streamside harvest, increased
1177 streamflow alone is estimated to be responsible for <10% of sediment transport into streams
1178 while the increased sediment supply caused by harvest operations is responsible for >90% of the
1179 sediment transported into streams.

1180 Puntenney-Desmond et al. (2020) found similar results in their assessment of differences in
1181 instream sediment contributions from the buffer area, harvest area, and buffer-harvest interface.
1182 Sediment concentration in the runoff was 15.8 times higher for the harvested area than in the
1183 riparian buffer, and 4.2 times greater than in the harvest-buffer interface. Total sediment yields
1184 ($\text{mg m}^{-2} \text{min}^{-1}$) from the harvested area (sediment concentration x flow rate) were approximately
1185 2 times greater than in the buffer areas, and 1.2 times greater in the harvest-buffer interface than
1186 in the buffer area.

1187 *Summary of Factors Impacting Sediment Delivery into Streams*

1188 From the studies reviewed there is evidence that sediment delivery into streams following timber
1189 harvest is influenced by not only the intensity of the harvest operation (e.g., presence of retention
1190 buffers, yarding and equipment use immediately adjacent to the stream, upland clearcut vs.
1191 thinning), but also by physiography (e.g., hillslope gradient), lithology relative softness, and
1192 climate (e.g., precipitation, frequency of large storm events). Thus, the change in magnitude of
1193 sediment delivery following harvest is context dependent and these landscape factors can interact
1194 with one another to compound these changes. However, from the studies reviewed above there is
1195 evidence that the implementation of BMPs since the 1970s in the northwestern United States
1196 lessen the impact and duration of these changes.

1197 Table 4. List of treatments, variables, metrics, and results from publications reviewed for information on sediment inputs and source.

Reference	Treatment	Variables	Metrics	Notes	Results
Bywater-Reyes et al., 2017	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 Northeast 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).
Bywater-Reyes et al., 2018	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 ($\sim 0.2\text{--}0.52\text{ t/(km}^2\text{)}$) 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.
Hutton et al., 2018	Data from pre restriction and post-Oregon BMPs prescriptions for non fish bearing streams.	suspended sediment concentrations (SSC)	suspended sediment, stream discharge, and daily precipitation	Phase I harvest: 2009 harvest of upper half of watershed. Phase II harvest: 2015 harvest of lower half of watershed.	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 92 mg L^{-1} ($\sim 63\%$) lower after the Phase I harvest and

Field Code Changed

Field Code Changed

Field Code Changed

	BMPs: no buffer in non-fish-bearing streams with equipment exclusion zones, and a 15-m no-cut buffer in fish-bearing streams				28.2 mg L ⁻¹ (-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 NDLG 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.
Karwan et al., 2007	clearcut of the watershed area of by 50%, partial cut of 50% canopy removal, timber road construction Riparian zone harvest followed Idaho EPA rules.	Total suspended solid (TSS) yields	Monthly total suspended solid readings from multiple flume locations for pre- and post-harvest, and pre- and post-road construction.		A significant and immediate impact of harvest on monthly sediment loads in the clear-cut watershed ($p = 0.00011$), and a marginally significant impact of harvest on monthly sediment loads in the partial-cut ($p = 0.081$) 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: $p = 0.2336$, partial cut: $p = 0.1730$) 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.
Litschert & MacDonald, 2009	Data collected from 4 NF of Nort CA, ~200 harvest sites near riparian zones with 00 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.

Field Code Changed

Field Code Changed

Macdonald et al., 2002a	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 generate 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 edge (no buffer).	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.
Mueller & Pitlick, 2013	The study used sediment concentration data from 93 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 side slope, and drainage density showed little correlation strength with bankfull sediment supply.

Field Code Changed

Field Code Changed

Field Code Changed

Puntunney-Desmond et al., 2020	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 x 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.
Rachels et al., 2020	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 ± 2.1% from hillslope) and unharvest (92.1 ± 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 ± 2.3 g).
Safecq et al., 2020	Long term (51 years) effects of clearcut to stream followed by broadcast burn.	streamflow, sediment transport	Historical streamflow data, precipitation data, sediment grab samples for bedload and suspended sediment.	Data compared one treatment watershed and one control watershed across 51 years.	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 ² (10%) of the suspended load and 113 Mg/km ² (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, 92% 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 < 10% of the resulting suspended sediment transported into streams, while the increase in sediment supply due to harvest disturbance was responsible for >90%.

Field Code Changed

Field Code Changed

Field Code Changed

Wise, 2010	Streamflow patterns derived from instrumental data and from reconstructed tree-ring chronologies were compared with other previously reconstructed rivers in similar climates.	Streamflow	Dendrochronology, historical data records, seasonal patterns	The reconstruction model developed for the analysis explained 62% of the variance in the instrumental record after adjustment for degrees of freedom.	Results showed evidence that droughts of the recent past are not yet as severe, in terms of overall magnitude, as a 20-year extended period of drought discovered in the mid-1600s. However, in terms of number of individual years of < 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.
Wissmar et al., 2004	Data sourced from management records and geospatial data to identify high erosion risk areas.	Sediment, weather, stand characteristics, landscape factors	unstable soils, immature forests, roads, critical slopes for land failure, and rain-on-snow events		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.

Field Code Changed

Field Code Changed

1199 [Shade and stream temperature](#)

1200 Canopy cover provides shade for streams that decreases the amount of incoming solar radiation
1201 and thus influences stream temperatures, although that influence can be highly variable
1202 depending on shade structure and density surrounding stream courses. Temperature regulation is
1203 vital for sensitive salmonid fish species that require cooler waters, and shade is often the primary
1204 function assessed when developing state regulations (Groom et al., 2011; Groom et al., 2018;
1205 Teply et al., 2014). The importance of shade and cooler in-stream temperatures for fish habitat
1206 has been thoroughly investigated (Bjornn & Reiser, 1991; Chapman & Bjornn, 1969; Ebersole et
1207 al., 2001; Sullivan et al., 2000). The streamside shade will likely become even more critical with
1208 the predicted increases in air temperature over the next century (Manuta et al., 2009). While
1209 stream temperature is initially reflective of moisture source (e.g., snowmelt, liquid precipitation,
1210 groundwater inputs) and watershed subsurface soil characteristics. As water flows downstream
1211 and into higher-order streams, the net rate of temperature gain or loss is the sum of incident
1212 radiation, evaporation, conduction, and advection (Brown, 1983; Bescheta et al., 1987).

1213 Bescheta et al. (1987) presented evidence that direct beam solar radiation inputs are of the
1214 highest importance to the stream's net heat exchange rate per unit area compared to other factors.
1215 Within the net heat exchange calculation, the heat released from evaporation generally cancels
1216 out the heat gained from warm air temperatures (convective and advective heat transfer). Thus,
1217 temperature fluctuations are expected to be more severe in less-shaded/more-exposed streams.
1218 This has been supported by many experimental field and simulation studies showing evidence
1219 that the reduction of effective shade can lead to considerable increases in peak summer stream
1220 temperatures primarily due to the increase of incoming solar radiation. However, while increases
1221 in solar radiation are accepted as the most important factor in stream temperature changes and
1222 fluctuations following harvest, other factors are also important and may compound these effects.

1223 [For example,](#)

1224 Guenther et al. (2014) investigated the relationship between changes in stream temperature and
1225 changes in wind speed, vapor pressure, and evaporation following riparian thinning treatments
1226 along headwater streams in southwestern British Columbia. Treatment involved reduction of
1227 basal area by 50% (resulting in 14% reduction in canopy closure) in the upland and riparian
1228 forests. Results showed a post-harvest increase in wind speed, vapor pressure deficit, air
1229 temperature and evaporation above the stream, which coincided with increased stream
1230 temperatures and lower stability. The authors report that prior to harvest, vapor pressure
1231 gradients often favored condensation over evaporation. Further, they concluded that the
1232 relationships between the riparian and microclimate variables after harvesting became more
1233 strongly coupled to ambient climatic conditions due to increased ventilation. Contemporary
1234 riparian management practices in western North America vary by state. However, all require
1235 retention of protective buffers that preserve some percentage of shade or canopy cover to
1236 maintain or mitigate changes in stream temperatures, especially along fish-bearing streams.
1237 Many studies published in the last two decades report evidence that these practices have been
1238 effective in mitigating stream temperature changes after harvest.

1239 For example, Bladon et al. (2016), assessed the effectiveness of riparian management
1240 prescriptions developed for the Oregon Forest Practices Act (FPA). Oregon State requires a 15 m
1241 buffer on either side of small fish-bearing streams with a 6 m no-cut buffer, and a minimum
1242 retention for conifer basal area of $\sim 3.7 \text{ m}^2$ for every 300 m (~ 1000 ft) length of stream. This
1243 resulted in a reduction of mean canopy closure from $\sim 96\%$ in the pre-harvest period to $\sim 89\%$ in
1244 the post-harvest period in the treatment reaches. In contrast, mean canopy closure in the
1245 reference reaches changed from $\sim 92\%$ to $\sim 91\%$ from pre- to post-treatment periods. Results
1246 showed there was a significant increase in the 7-day moving maximum temperature from pre- to
1247 post-harvest values when data was constrained to the period of July 15 – August 15 by $0.6 \pm$
1248 $0.2 \text{ }^\circ\text{C}$. However, when analyzed by individually paired sites, and when interannual and site
1249 variability was accounted for, no significant changes in stream temperature were observed for 3
1250 years post-harvest (length of study).

1251 However, Groom et al., (2011a, b) showed evidence that the more stringent rules of the
1252 Northwest Oregon State Forest Management Plan (FMP; applied to riparian management zones
1253 on state owned land) was even more effective at maintaining stream temperatures post-harvest.
1254 The FMP requires a 52 m wide buffer for all fish-bearing streams, with an 8 m no cut buffer
1255 immediately adjacent to the stream. The results from Groom et al. (2011b) showed that FPA
1256 (Oregon Forest Practices) post-harvest shade values differed from pre-harvest values (mean
1257 change in Shade from 85% to 78%), while no difference was found for FMP shade values pre-
1258 harvest to post-harvest (mean change in Shade from 90% to 89%). Following harvest, maximum
1259 temperatures at FPA increased relative to FMP on average by $0.71 \text{ }^\circ\text{C}$. Similarly, mean
1260 temperatures increased by $0.37 \text{ }^\circ\text{C}$ (range: $0.24 - 0.50$), minimum temperatures by $0.13 \text{ }^\circ\text{C}$
1261 (range: $0.03 - 0.23$), and diel fluctuation increased by $0.58 \text{ }^\circ\text{C}$ (range: $0.41 - 0.75$) relative to
1262 FMP sites.

1263 Groom et al (2011a) developed prediction models from this data to estimate the probability of
1264 riparian harvest under each regulation causing an increase in stream temperatures $>0.3 \text{ }^\circ\text{C}$ (the
1265 Protecting Cold Water criterion developed by the Department of Environmental Quality). Results
1266 indicate that sites harvested according to FPA standards exhibited a 40.1% probability of a
1267 temperature change of $> 0.3^\circ\text{C}$ from pre- to post harvest. Conversely, harvest to FMP standards
1268 resulted in an 8.6% probability of exceedance that did not significantly differ from all other
1269 comparisons.

1270 In Montana, Sugden et al. (2019) investigated the effectiveness of state regulation which requires
1271 timber be retained within a minimum of 15.2 m (50 feet) of the stream. Within the riparian
1272 management zone, no more than half the trees greater than 204 mm (8 in) diameter at breast
1273 height (DBH) can be removed. In no case, however, can stocking levels of leave trees be reduced
1274 to less than 217 trees per hectare. Data for canopy cover, stream temperature, and fish population
1275 were collected for 30 harvest reaches in western Montana (northern Rocky Mountain Region),
1276 for a minimum of one-year pre- and one-year post-harvest. Shade over the stream surface was
1277 not directly measured in this study. Instead, canopy cover was used as proxy, using two
1278 independent estimates of canopy cover (1) used cruise data to populate a canopy cover model
1279 within Forest Vegetation Simulator, and (2) measured canopy cover in the harvested reach every

1280 30 m, before and after harvest. Within harvest units, mean basal area was reduced by 13%
1281 (range: 0 – 36%), and again further by a mean of 2% due to windthrow. Mean canopy cover
1282 within the riparian management area reduced from 77% (pre-treatment) to 74% (post-treatment),
1283 and mean canopy cover over the stream changed from 66% (pre-treatment) to 67% (post-
1284 treatment) based on densiometer measurements. Neither of these changes were significant.
1285 Results for stream temperature also showed no significant changes in stream temperatures or fish
1286 populations in one-year post treatment compared to pre-treatment values.

1287 Specific to Washington, Cupp & Lofgren (2014) conducted a study to test the effectiveness of
1288 riparian timber harvest rules for eastern Washington in preserving shade and stream
1289 temperatures. Regulations for fish-bearing streams in eastern Washington (in the mixed
1290 conifer/mid elevation zone) includes an “All Available Shade Rule” (ASR) for streams in the bull
1291 trout habitat zones, and a “Standard Shade Rule” (SR). Under the ASR it is required to retain all
1292 available shade within 75 feet of the stream. Under SR some harvest of shade providing trees is
1293 allowed within the 75-foot buffer depending on elevation and pre-harvest canopy cover.
1294 Unharvested reference reaches were located upstream from treatment reaches. Prior to harvest
1295 treatments, canopy closure measurements ranged from 89% to 97%, with a mean of 93%.
1296 Results showed post-harvest shade values decreased in SR sites (mean effect of -2.8%, $p =$
1297 0.002), as did the canopy closure values (mean effect of -4.5%, $p < 0.001$). Shade and canopy
1298 closure values did not significantly change after treatment in the ASR sites. Post-harvest mean
1299 daily maximum stream temperature increased 0.16 °C in the SR harvest reaches, whereas stream
1300 temperatures in both the ASR sites and in the no-harvest reference reaches increased on average
1301 by 0.02 °C. Sample period means of daily maximum temperature responses varied from -1.1 °C
1302 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
1303 sites, and -0.5 to 0.9 °C in the reference sites. While these values show a slight increase in mean
1304 temperatures and temperature ranges with treatment, the authors interpret these results as
1305 evidence that temperature effects of the SR and ASR were similar to reference conditions along
1306 sampled reaches.

1307 Riparian harvest rules along non-fish bearing streams tend to allow for narrower buffer widths
1308 (sometimes with no retention buffers) or more intense thinning within the buffer than for fish-
1309 bearing streams. For example, in western Washington the Forest Practices (FP) buffer
1310 prescription requires a two-sided 15 m (50 ft) wide buffer along a minimum of 50% of the length
1311 of a non-fish-bearing perennial stream (i.e., up to 50% of the stream may have no buffer) with a
1312 9.1 m (30 ft) equipment exclusion zone. Two recent studies (Ehinger et al., 2021; McIntyre et al.,
1313 2021) have compared these FP buffers to two experimental buffer treatments, a 50 ft buffer along
1314 100% of the stream length (100%), and no buffer (0%) treatment, and an unharvest reference
1315 (REF) on sites underlain by competent lithologies (McIntyre et al., 2021; “Hard Rock”) or
1316 incompetent (friable) lithologies (Ehinger et al. 2021; “Soft Rock”).

1317 Results from the Hard Rock study showed that riparian canopy cover declined after harvest in all
1318 buffer treatments reaching a minimum around 4 years post-harvest (after mortality stabilized).
1319 The treatments, ranked from least to most change, were REF, 100%, FP, and 0% for all metrics
1320 and across all years. Effective shade results showed decreases of 11, 36, and 74 percent in the

1321 100%, FP, and 0% treatments, respectively. These changes in shade were significant for all
 1322 treatments. [This led to changes in mean stream temperature from pre- to post-harvest in the
 1323 100% treatment by 1.1°C 2.4°C in the first two years following treatment, but returned to pre-
 1324 harvest levels by post-harvest year 3 never exceeded 1.0°C in any year after (for up to 8 years). In
 1325 contrast, the mean difference in pre- to post-harvest stream temperatures in the FP ranged from
 1326 0.5°C to 1.1°C exceeded 1.0°C in the first year, and changed little over the entire post-harvest
 1327 period, declined in years 2-5 post-harvest, and then exceeded 1.0°C again in years 6-9. Results for
 1328 the 0% treatment showed a mean increased difference of 3.85.3°C immediately following harvest
 1329 and declined over time to but never below 0.89°C by year 9. [These results suggest that the 100%
 1330 treatment was most effective at preventing increases to stream temperature followed by the FP
 1331 and 0% treatments. Comparatively, mean pre- to post-harvest differences in stream temperature
 1332 never exceeded 1.0°C in the reference sites. Changes in mean difference from pre- to post-
 1333 harvest stream temperatures were significant for all treatments at some point during the study.
 1334 However, by year 11 mean stream temperatures had recovered to within 0.2°C of pre-harvest
 1335 values for all treatments.] A weak and nearly significant (P-value range: 0.008 - 0.108) negative
 1336 relationship between canopy cover and stream temperature for the first 4 years after treatment
 1337 was detected. These results provide evidence that the effectiveness of buffers in maintaining
 1338 stream temperatures post-harvest is relative to the intensity of the treatment (e.g., presence of
 1339 buffer, reduction in canopy cover). Further, post-treatment mortality within the buffer from
 1340 events such as windthrow can cause fluctuations in stream temperature response during the first
 1341 decade. Results from the Soft Rock Study showed similar trends in canopy cover reduction and
 1342 stream temperature increases. [Authors of the Soft Rock study note that stream temperature
 1343 changes varied as a function of the proportion of the stream buffered and tree mortality, but
 1344 limited and unbalanced sample sizes did not allow for statistical analysis.]

1345 Outside of Washington, several studies conducted in western North America since 2000 have
 1346 shown results similar to the Hard Rock and Soft Rock studies. For example, [Roon et al. (2021b)]
 1347 compared stream temperature changes following variable riparian thinning intensities in the
 1348 redwood forests of northern California. Treatments to riparian stands included reduction of
 1349 canopy cover that resulted in reduction of effective shade by either (19-30%) or by (4-5%). Their
 1350 results showed that local changes in stream temperature were dependent on thinning intensity,
 1351 with higher levels of canopy cover reduction leading to higher increases in local stream
 1352 temperatures. In the reaches with higher reductions in shade (19-30%) there was accumulation of
 1353 45° to 115°C additional degree days from pre- to post treatment years, while the reaches with
 1354 lower reductions in shade (4-5%) only accumulated 10° to 15°C additional degree days. Further,
 1355 travel distance of increased stream temperatures also appeared to be dependent on thinning
 1356 intensity. The lower shade reduction reaches had an increased temperature effect downstream
 1357 with travel distance of 75-150 m, while the high shade reduction sites had a downstream travel
 1358 distance of 300- ~1000 m. Roon et al. (2021a) reported changes in -in average daily maximum,
 1359 maximum weekly average of the maximum (MWMT), average daily mean, or maximum weekly
 1360 average of the mean (MWAT) at these same sites under the same timeline. The lower thinning
 1361 intensity (4-5% effective shade reduction) showed no significant changes in any temperature
 1362 metrics. However, The more intensely thinned sites (19-30% reduction in effective shade)
 1363 showed an increase MWMT during spring by a mean of 1.7°C (0.9 - 2.5 °C), summer by a mean

Commented [BW(63)]: Numbers are incorrect. Please see Buffer Treatment Table 4-18, and 4-6.3 Summary in McIntyre et al 2021.

Commented [WB64R63]: Address

Commented [bs65R63]: Yes, this was accidentally taken from the mean within treatment differences, thank you. Corrected

Commented [BW(66)]: There was no post harvest year 11. Is this meant to be year 11 of the study? Also, 0.2 is incorrect, see above comment for locations of stream temperature effects.

Commented [WB67R66]: Address

Commented [bs68R66]: Addressed

Commented [BW(69)]: A statistical analysis was performed, see Figure 4A-3, Table 4A-8, Figure 4A-4, and section 4A-2.3 Stream Temperature of Ehinger et al 2021

Commented [WB70R69]: Address

Commented [bs71R69]: Removed

Commented [BW(72)]: I would also include Roon et al 2021a, which is more directly about temperature and shade response at the same study sites.

Commented [WB73R72]: Address

Commented [bs74R72]: Included

1364 of 2.8°C (1.8, 3.8 °C), and fall by a mean of 1.0°C (0.5, 1.5 °C) and increased in downstream
1365 reaches during spring 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
1366 °C). The authors interpret their results as evidence that that changes in shade of 5% or less
1367 caused minimal changes in temperature while reductions in shade of 20–30% resulted in much
1368 larger increases in temperature.

1369 Reiter et al. (2020) compared the changes in stream temperatures following different harvest
1370 treatments along headwater streams in the Trask River Watershed in the northwestern coast range
1371 of Oregon. Treatments included a clearcut to stream (no buffer but half of sites contained some
1372 leave trees along stream bank), upland clearcut with a 10 m no-cut buffer, upland thinning (basal
1373 area reduction to 30-50% of original stand) with a 10 m no-cut buffer, and an unharvested
1374 reference. Results showed that post-harvest stream temperature increases were only significant in
1375 the clear-cut treatments without buffers with a mean increase of 3.6°C (SE = 0.4°C) for four
1376 years after the study. They note that temperature changes were more severe in the unbuffered
1377 streams with no leave trees (4.2 and 4.4°C), however, this difference was not analyzed. No
1378 significant changes in stream temperature were detected in either treatment with a 10 m no-cut
1379 buffer. The authors speculate that 10 m wide buffers were sufficient in maintaining stream
1380 temperatures post-harvest in small, forested headwater streams.

1381 In the sub-boreal forest ecosystems of British Columbia, Canada, Macdonald et al. (2003b)
1382 compared pre- to post-harvest stream temperature changes in first-order headwater streams under
1383 3 different riparian forest treatments. These treatments included 1) low-retention – removal of all
1384 merchantable timber >15 or >20 cm DBH for pine or spruce respectively, within 20 m of the
1385 stream 2) high-retention – removal of merchantable timber >30 cm DBH within 20-30 m of the
1386 stream, and 3) patch-cut – high retention for the lower 60% of watershed approaching streams
1387 and removal of all vegetation in the upper 40% of the watershed. Results showed significant
1388 increase in stream temperatures ranging from 4 – 6 °C in the low-retention and patch cut in the
1389 first three years following harvest. However, by year five, mortality in the high-retention buffer
1390 (due to windthrow) resulted in canopy cover reduction and increases in stream temperatures that
1391 became equivalent to the other treatments. The authors conclude that while the variation in
1392 harvest intensity initially appeared to dictate stream temperature responses, site effects (e.g.,
1393 windthrow susceptibility) can impact the effectiveness of the buffer. While the studies above all
1394 show evidence that the impact of riparian forest harvest on stream temperatures are related to the
1395 severity of the harvest prescription (e.g., buffer width, thinning intensity, canopy reduction) the
1396 results are variable within treatments indicating other site factors are also important when
1397 evaluating buffer effectiveness. For example, in their review of experimental studies conducted
1398 in the Pacific Northwest of Canada and the United States, Martin et al. (2021) reported high
1399 variability in temperature response to streamside buffers. They report a substantial variability and
1400 overlap in the effect size of the mean 7-day maximum temperature metric with no-cut buffers,
1401 no-cut plus variable retention buffers, and no-cut patch buffers ≤ 20 m wide. The largest
1402 temperature response (> 3.4 °C) occurred in the clearcut buffers while treatments with buffers
1403 (i.e., no cut buffers without variable retention) had the smallest response (< 0 °C). The variable
1404 retention buffers < 20 m showed variable response (0.6 – 1.4 °C). They conclude that the

1405 variation in temperature response following riparian harvest may be associated with multiple
1406 factors such as geology, hydrology, topography, latitude, and stream azimuth.

1407 Bladon et al. (2018) investigated the changes in stream temperatures following treatments that
1408 varied from clearcuts to stream to buffers > 20 m in western Oregon. They performed a
1409 regression analysis to assess the relative relationship between catchment lithology and the
1410 percentage catchment harvested with stream temperature at all sites. Their results showed that at
1411 the upstream harvested sites there was a strong relationship between stream temperature
1412 increases and catchment lithologies, but no statistically significant relationship between stream
1413 temperature changes and percent of catchment harvested. Sites downstream from harvested areas
1414 showed a significant relationship with the interaction of percentage of catchment harvested and
1415 the underlying lithologies ($p = 0.01$). The greatest temperature increases at downstream sites
1416 were in areas with a higher percentage of catchment harvested and were underlain by more
1417 resistant lithologies. There was no evidence for increases in stream temperatures in catchments
1418 with a high percentage of harvest that were underlain by permeable geology. The authors suggest
1419 that this relationship may be due to the buffering effect of increases in summer low flows and
1420 greater groundwater or hyporheic exchange. They conclude that the variability of rock
1421 permeability and the relative contribution of groundwater during summer months, and their
1422 effect on stream temperatures following harvest should be investigated further.

1423 There is evidence that geomorphology alone can impact stream temperature fluctuations
1424 throughout the year. Hunter & Quinn, (2009) compared seasonal fluctuations in stream
1425 temperatures between two watersheds in the Olympic Peninsula, Washington. Both watersheds
1426 were similar in all characteristics except for bed substrate. One was underlain by alluvial bed
1427 substrate while the other was underlain by bedrock. Results from this study show consistent
1428 differences in stream temperature response in alluvial versus bedrock channels. Seasonal
1429 maximum and minimum average daily temperatures varied less at the alluvial site compared to
1430 the bedrock site. This, the authors suggest, may be due to hyporheic exchange in alluvial
1431 channels helping to buffer surface water temperatures from gaining or losing heat. In addition,
1432 groundwater may also contribute to the increased stability at the alluvial site. Aside from shade
1433 reduction from timber harvest, there is evidence that light availability and canopy cover naturally
1434 changes over time as riparian stands develop. For example, Warren et al. (2013) compared
1435 canopy cover and stream light availability between old-growth-forests (>500 years old) and
1436 young harvest-aged stands (~40-60 years old) in the H.J. Andrews Experimental Forest in the
1437 Cascade mountains of Oregon. Streams were paired based on reach length and bankfull width,
1438 and north ($n=2$), and south ($n=2$) facing watersheds. Canopy cover was estimated using a
1439 convex spherical densiometer, and light reaching the stream bed was estimated using a
1440 fluorescent dye that degrades overtime from light exposure. Overall, three of the four paired old-
1441 growth reaches (2 south-facing, 1 north-facing) had significantly lower mean percent canopy
1442 cover ($p < 0.10$), and significantly higher mean decline in fluorescent dye concentrations ($p <$
1443 0.01). The authors interpret these results as evidence that old-growth forest canopies were more
1444 complex and had more frequent gaps allowing for more light availability and lower mean canopy
1445 cover, on average, than in adjacent young, second growth forests.

1446 Kaylor et al. (2017) presented similar results when they compared canopy cover and light
1447 availability between small mountain streams adjacent to late-successional forests (dominant
1448 canopy trees >300 years old) and second-growth forests that had been harvested to the stream
1449 50-60 years prior to data collection. Like Warren et al. (2013), canopy cover was estimated with
1450 a convex spherical densiometer; and light availability to streams was estimated with a
1451 photodegrading fluorescent dye. However, for this study, fluorescent dye degradation was
1452 converted to photosynthetically active radiation (PAR) by building a linear relationship between
1453 the dye degradation and PAR sensors. Results showed that mean PAR reaching streams was 1.7
1454 times greater, and canopy openness was 6.1% greater in >300-year-old forests than in 30–100-
1455 year-old forests. Of the 14 paired sites, differences in canopy openness and PAR were significant
1456 for 6 sites. The authors compared and combined their data with published data from 10 other
1457 similar studies. The combined datapoints for canopy openness (%) were plotted against stand age
1458 and fit it with a negative exponential curve. From the slope of the curve, the authors estimate that
1459 canopy openness reaches its minimum value in regenerating forests at ~30 years and maintains
1460 with little variability until ~100 years.

1461 *Summary of Factors Affecting Shade and Stream Temperature*

1462 From the studies reviewed above, the results show evidence that changes in canopy cover and
1463 effective shade are, not surprisingly, directly related to the intensity of harvest operation. Initial
1464 reduction in canopy cover and shade from pre- to post-harvest are influenced by the basal area
1465 removed and the width of the retention buffer. However, there is evidence that multiple site
1466 factors can interact with harvest operations (e.g., target basal areas).

1467

Commented [WB75]: Please expand on this summary. This does not include clear-cut vs thinning, complexities in riparian stands (e.g. conifer vs broadleaf), hyporheic exchange, topographic shading, etc.. This is a complex topic that deserves more attention.

Commented [WB76R75]: Do not Address

Table 5. List of treatments, variables, metrics, and results from publications reviewed for information on shade and stream temperature.

Reference	Treatment	Variables	Metrics	Notes	Results
Bladon et al., 2016	15 m buffer with a minimum of $\approx 3.7 \text{ m}^2$ 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 $0.6 \pm 0.2^\circ\text{C}$. 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).
Bladon et al., 2018	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.

Cole & Newton, 2013	clearcut to stream, partial buffer (12-m width on predominant sun side)), Oregon state BMP (15-30-m no-cut buffer both sides)	Stream temperature	Controlled for yearly fluctuations in temperatures by analyzing the difference in stream temperature entering and exiting the reach with digital temperature data loggers	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.	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.
Cupp & Lofgren, 2014	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.	Canopy closure, shade measurements, stream temperature	Hand-held densiometer (canopy closure), self-leveling fisheye lens digital camera (shade), temperature data loggers	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.	Results showed post-harvest shade values decreased in SR sites (mean effect of -2.8%, p = 0.002), as did the canopy closure values (mean effect of -4.5%, p < 0.001). 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 AS 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.
Ehinger et al., 2021	1) Buffers encompassing the full width (50 feet), 2) <50ft buffers, 3) Unbuffered, harvested to the edge of the channel, and 4) Reference sites in unharvested forests.			Soft Rock study. Only descriptive statistics. Small sample sizes.	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 0.2 °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 only in 2 treatment sites by up to 1.8 °C at one site (for 5 years post-harvest) and by 0.1 °C at another (at year 5 post-

Commented [WB77]: Also included many metrics mentioned elsewhere in this table.

Commented [WB78R77]: Address

Commented [bs79R77]: included

Commented [WB80]: Multiple statistical analyses were run on the temperature response (e.g. GLS, GLIMMIX), see 4-3.4 of Ehinger et al 2021. “Small sample size” is not an informative metric, please provide actual sample sizes if mentioned in this table to provide reader with information to determine how the sample sizes of the studies compare to each other. If possible find a way to normalize the data for comparison. E.g. Soft Rock - 7 treatment basins (~7000 m of streams treated with current forest practice buffers), 3 reference basins (~3000m of streams), and 57 temperature stations. This study had an unbalanced design (reference sites were well matched and in close proximity with treatments).

Commented [WB81R80]: Address

Commented [bs82R80]: Addressed

					harvest). None of the three REF sites exceeded 16.5°C during the study.
Gravelle & Link, 2007	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.
Groom et al., 2011a	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
Groom et al., 2011b	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 ² /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.
Guenther et al., 2014	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.

Hunter & Quinn, 2009	an alluvial study site and a bedrock study site whose overall characteristics were otherwise comparable apart from geomorphology.	Stream temperature, Alluvial depth	Water temperature was recorded at 75-m intervals along each channel during the summers of 2003 and 2004	Small sample sizes, results only from two sites for two summers. Actual numeric values not reported but shown in graphs.	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.
Janisch et al., 2012	clearcut logging with two riparian buffer designs: a continuous buffer and a patched buffered stream. Buffers were 10-15 m wide.	Stream temperature	Channel and catchment attributes (e.g., BFW, Confinement, slope, FPA, etc.), Stream temperatures were recorded with a Tidbit datalogger in areas persistently submerged.	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.	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 ($p=0.0000$), and the day of the year ($p=0.0007$) were all significant effects explaining observed changes in temperature. Wetland area (0.96 , $p<0.01$) and length of surface flow (0.67 , $p=0.05$) were strongly correlated with post-logging temperature changes.
Johnson & Jones, 2000	clearcut-to-stream, patch cutting followed by debris flows (resulted in the removal of all streamside vegetation), 450+ yo-Doug fir forest reference.	Stream temperature	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.	The experimental design used historic stream temperature data to examine changes in stream temperatures. This required conflating data from 2 different devices.	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. Differences in treatment streams and reference stream

					temperatures were less than 1.1°C pre-treatment and 30-years post-treatment.
Kaylor et al., 2017	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.
Kibler et al., 2013	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.
Macdonald et al., 2003b	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 20–30 m 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.

McIntyre et al., 2021	(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).			Hard Rock Study.	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 5.2°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.
Pollock et al., 2009	A range of harvest from 0-100%, <20 years old regrowth, ~40 years old regrowth. Unharvested sites were estimated as being >150 years old	Stream temperature, time since harvest, percent of watershed and stream network harvested.	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.	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.	Results of general temperature patterns showed that average daily maximum (ADM) were strongly correlated with average diurnal fluctuations ($r^2 = 0.87, p < 0.001, n = 40$), 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 ($r^2 = 0.39, p < 0.001, n = 40$) and 32% of variation in the average daily range (ADR) ($r^2 = 0.32, p < 0.001, n = 40$). 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 ($p < 0.001$). 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 ($p < 0.001$). Results for the correlations between the riparian network scale forest harvest and stream temperature showed that the total

				<p>percentage of the riparian forest network upstream of temperature loggers harvested explained 33% of the variation in the ADM among subbasins ($r^2 = 0.33$, $p < 0.001$, $n = 40$) and 20% of variation in the ADR ($r^2 = 0.20$, $p = 0.003$, $n = 40$). 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 ($r^2 = 0.03$, $p = 0.79$, $n = 40$), the ADR of stream temperatures ($r^2 = 0.02$, $p = 0.61$, $n = 40$) or any other stream temperature parameters. The proportion of total harvested near upstream riparian forest (avg = 0.66, SD \pm 0.34, range = 0.0-1.0) was weakly correlated with ADM ($r^2 = 0.12$, $p = 0.02$, $n = 40$) and not significantly correlated with ADR ($r^2 = 0.07$, $p = 0.06$, $n = 40$). 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 <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 >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>
--	--	--	--	---

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); and unharvested reference (REF) streams.	Stream temperature	Temperature data was separated into 5 th , 25 th , 50 th , 75 th , and 95 th percentiles. The researchers also quantified the percentage of summer where temperatures were above 16 and 15 °C.	Sample sizes are relatively low for some treatments. (CC_NB; n = 4); (CC_B; n = 3); (TH_B; n = 1); (REF; n = 7).	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.
Reiter et al., 2015	Various buffer prescriptions as regulations changed over time. (mid 1970s—1980s = “nominal”; mid 1980s—mid 1990s = 23-m; 2001—2009 = 30-m buffers)	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)	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.	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.	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

					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.
Roon et al., 2021a	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 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.
Roon et al., 2021b	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.
Sugden et al., 2019	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 ² /ha pre-harvest to 26.4 m ² /ha post harvest (mean = -13%, range from -32% to 0%). Windthrow further reduced the mean BA to 25.9 m ² /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.
Swartz et al., 2020	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 ² to 1,374 m ² with a mean of 962 m ² .	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 ⁻² day ⁻¹ , overall resulting in a mean change in light of 2.93 (SD ± 1.50) moles of photons m ⁻² day ⁻¹ . Through the entirety of the treatment reach mean shading declined by only 4% (SD ± 0.02%). Overall, the gap treatments did not change summer T7DayMax or T7DayMean 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.

1469
1470

1471 **Results/discussion by focal question**

1472 **Focal Question 1**

1473 1. *What are the effects of timber harvest intensities and extent on the riparian functions, with an*
 1474 *emphasis on the five key functions listed above, in comparison to conditions before harvest?*

1475 From the perspective of an experimental design, this question inquires how the values of the
 1476 metrics used to describe the five key functions (large woody debris recruitment, sediment
 1477 filtration, stream bank stability, shade, litterfall and nutrients) differ from pre- to post-harvest
 1478 within particular riparian areas of interest. An attempt to answer this question would require data
 1479 collection before and after treatment with or without a control site. Thus, only studies that used a
 1480 BACI or BAI approach are appropriate for discussing this question. From our review, 22 papers
 1481 report pre- to post-harvest changes in the magnitude of one or more of the key functions with the
 1482 majority of these papers focusing on changes in shade. No studies published since 2000 that
 1483 apply an experimental design in western North America to quantify changes in bank stability
 1484 could be found in the literature.

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

1485

1486 *Shade*

1487 **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	1-2 years post-harvest *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	1- and 2-years post-harvest 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.

Commented [WB83]: Answers to focal questions appear to just be additional summaries of specific studies. This reads more like an annotated bibliography broken up by topic. Very little synthesis of these papers in a way that could address the focal questions appears to have been done. One benefit of a literature synthesis is to provide the reader with a comparison and integration of the full breadth of literature around a specific topic. This can provide information on how all of the literature together can and cannot answer these specific questions. The way this is written puts the onus on the reader to make the comparisons to the studies reviewed. There should be more of an effort to provide a narrative structure that tries to answer these questions by integrating findings of multiple studies that either support or potentially don't support (and try to provide a possible reason why) an answer to these questions.

Commented [WB84R83]: Do not Address

Commented [JK85]: Red: Again, I find the answers to the focal questions appear to be a reiteration of the summaries provided above. What can we infer or learn from this collection of studies that may help answer or reframe the focal questions?

As above, I suggest tabulating the findings from the studies by treatment or maybe treatment range when there isn't consistent buffer width for example. What are the key factors that affect the five functions in question?

Commented [JK86R85]: Do not address given above suggestions already made to tabulate the data from reviewed studies.

Commented [AJK87]: I am confused here, too. It seems that many of the studies are summarized more than once.

Commented [AK88R87]: ADDRESS

Commented [bs89R87]: Yes, in the previous reviews it was requested to include background review and summary when discussed in each question. While it is redundant, it provides context for the reader within each question. This way, the reader does not need to go back to the original summary for these details.

Commented [JK90]: Green: I recognize this question was presented to the contractor and was even perhaps vetted by CMER or Policy. I wonder, however, if better question is about "desired future conditions" as conditions before harvest may not be optimal to meet the goals of the FFR.

Commented [JK91R90]: Do not address

Formatted: Font: Not Italic

Groom et al. (2011b)	Buffer width of 21 meters (~69 feet; Private); 52 meters (~170 feet; State)	1 year post-harvest 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)	9 years post-harvest Riparian cover declined after harvest in all buffer treatments reaching a minimum at 4 years. 100% buffers (n=4) showed a change in mean shade ranging from +1 to -10 % over nine years. FP buffers (n = 4) 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).	1 year post-harvest 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)	1 year post-harvest 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)	1 year post-harvest 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
Swartz et al. (2020)	Buffer width of ~20-meter (65 feet) diameter gaps along streambank	1-2 years post-harvest Treatment reach (n = 4) mean shading declined by only 4% (SD ± 0.02%) post-harvest.

1488

1489 Specific to fish-bearing streams of eastern Washington, Cupp & Lofgren (2014) reported changes
1490 in canopy closure (quantified with handheld densiometer) and shade (quantified with fisheye lens
1491 digital camera) within reaches adjacent to riparian forests harvested under the All Available
1492 Shade Rule (ASR) and the Standard Shade Rule (SR). Both shade rules have a 30-ft no-cut
1493 buffer (core zone) immediately adjacent to the stream. The SR prescription allows thinning in the
1494 buffer zone 30-75 feet (inner zone) from the stream while the AAS prescription requires
1495 retention of all shade providing trees in this area. Results showed post-harvest shade values
1496 decreased in SR sites (mean effect of -2.8%, p = 0.002), as did the canopy closure values (mean
1497 effect of -4.5%, p < 0.001). Shade and canopy closure values did not significantly change in the
1498 treatment reaches of the ASR sites. Mean shade reduction in the SR treatment sites exceeded the

1499 mean shade reduction in the ASR sites by 3%. Canopy closure reduction was also greater in the
1500 SR sites than in the ASR sites by a mean of 4%.

1501 For non-fish bearing streams of western Washington, McIntyre et al. (2021) report changes in
1502 canopy closure following 3 different harvest prescriptions. Prescriptions included a two-sided
1503 50-ft wide riparian buffer along the entire stream (100%), a two-sided 50-ft riparian buffer along
1504 at least 50% of the stream consistent with the current Forest Practices buffer prescription (FP),
1505 and a clearcut to stream edge without a buffer (0%). The canopy cover was estimated at mid-
1506 stream with a handheld densiometer and was converted to effective shade values (for 5 years
1507 post-harvest). Hemispherical canopy photos were also taken for 4 years pre-harvest and 3 years
1508 post-harvest and converted to Canopy and Topographic Density (percentage of the photograph
1509 obscured by vegetation or topography). Results for canopy cover showed that riparian cover
1510 declined after harvest in all buffer treatments reaching a minimum around 4 years post-harvest.
1511 The treatments, ranked from least to most change, were 100%, FP, and 0% for all metrics and
1512 across all years. Effective shade results showed decreases of 11, 36, and 74 percent in the 100%,
1513 FP, and 0% treatments, respectively, by 3 years post-harvest. However, by post-harvest year 9,
1514 canopy closure returned to pre-harvest levels in the 100% treatment but remained 15% and 27%
1515 below pre-harvest values at the FP and 0% treatments, respectively. Significant post-harvest
1516 decreases were noted for all treatments and all years (9 years post-harvest). Another study,
1517 Janisch et al. (2012) also compared the effects of similar treatments (clearcut to stream, a full
1518 continuous buffer (10-15 m wide), and a patched buffer (~50-110 m long were retained in
1519 distinct patches along some portion of the channel) to canopy cover. Canopy cover in all streams
1520 averaged 95% (SE = 0.4) prior to harvest. Following treatment, canopy cover in the clearcut
1521 catchments averaged 53%, (SE = 7.4) canopy cover in the patch buffer treatment averaged 76%,
1522 (SE = 5.1) and canopy cover in the continuous buffer treatment averaged 86% (SE = 1.7). The
1523 changes were significant in the clearcut and patch buffers.
1524

1525 Outside of Washington, Bladon et al. (2016) assessed the effects of harvest treatments under the
1526 Oregon Forest Practices Act (FPA) on shade reduction and stream temperature. This study took
1527 place in the Siuslaw National Forest in the Oregon Coast Range in the Alsea Watershed.
1528 Treatment under the FPA includes a 15 m riparian management area with a minimum of ~3.7 m²
1529 conifer basal area retained for every 300 m length of stream and an additional 4-5 wildlife leave
1530 trees per hectare. This resulted in a mean canopy closure reduction from ~96% (pre-harvest) to
1531 ~89% (post-harvest) based on measurements from a densiometer along the stream channel for 3
1532 years pre- and 3 years post-harvest. Unfortunately, the authors did not compare these changes
1533 with statistical analysis. Groom et al. (2011b) compared changes in shade from pre- to post-
1534 harvest under the FPA and under the Northwest Oregon State Forest Management Plan (FMP).
1535 The FMP requires a 52 m wide buffer for all fish-bearing streams, with an 8 m no cut buffer
1536 immediately adjacent to the stream.

1537 Results from Groom et al. (2011b) showed that FPA site post-harvest shade values differed from
1538 pre-harvest values (mean change in Shade from 85% to 78%); While no difference was found for
1539 FMP site shade values pre-harvest to post-harvest (mean change in Shade from 90% to 89%). In
1540 the Trask Watershed of the northwestern Oregon Coast range, Reiter et al. (2020) compared three
1541 riparian zone treatments: 1) clearcut, no buffer (CC_NB; n = 4), 2) clearcut with 10-m no cut

Formatted: Default

Commented [WB92]: Canopy photos were also taken

Commented [WB93R92]: Address

Commented [bs94R92]: included

Commented [WB95]: Only through post 5

Commented [WB96R95]: Address

Commented [bs97R95]: addressed

Commented [WB98]: This statement is incorrect, there were some non-significant decreases in the 100% buffer treatments and later FP at stream level.

Commented [WB99R98]: Address

Commented [bs100R98]: Addressed

Formatted: Font:

1542 buffer (CC_B; n = 3), 3) thinning with 10 m no-cut buffer (TH_B; n =1) in small non-fish
 1543 bearing streams. Pre- to post-harvest values in shade were quantified with hemispherical analysis
 1544 over the stream one-year prior and one-year post-treatment. However, post-harvest overstory
 1545 buffer width varied within each treatment depending on landscape factors. For this reason, we
 1546 will present the change in percent shade with residual buffer width (Table 6). Again, changes in
 1547 shade were not statistically analyzed.

1548 [In fish-bearing streams within the McKenzie River basin in the western Cascade Mountains of
 1549 Oregon Swartz et al. (2020) assessed the effects of experimental canopy gap treatments on shade
 1550 and light availability to the stream. In each treatment reach (n = 6), 20 m gaps were prescribed to
 1551 mimic gap openings that naturally occur after individual large tree mortality or small-scale
 1552 disturbance events in late successional forests. Shade was recorded in the year before and the
 1553 year after treatment with hemispherical photos. Changes in effective shade (1 year post-harvest)
 1554 were estimated in HemiView 2.1 software. Mean stream shading could not be evaluated in the
 1555 full BACI analysis because post-treatment hemispherical photographs could not be taken at all
 1556 sites due to fire impeding access in 2018. For the remaining sites, the areas beneath each gap had
 1557 notable localized declines in shade, through the entirety of the treatment reach mean shading
 1558 declined by only 4% (SD ± 0.02%).]

1559 Table 56. Results for changes in shade following treatment for the Trask River Watershed Study
 1560 headwaters. Reproduced from Reiter et al (2020).

Treatment	Mean residual buffer width (2-sided)	Pre-harvest shade (%)	Post-harvest shade (%)
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

1561 CC_B = clearcut with 10 m buffer, CC_NB = clearcut no buffer, TH_B upland thinning with
 1562 buffer. *Unable to determine exact buffer width because adjacent to thinning

1563 Gravelle & Link (2007) compared changes in shade following treatment for non-fish bearing
 1564 streams in northern Idaho. For non-fish-bearing streams there is a 30 ft (9.1 m) equipment
 1565 exclusion zone on each side of the ordinary high-water mark (definable bank). There are no
 1566 shade requirements and no leave tree requirements, but skidding logs in or through streams is
 1567 prohibited. Harvesting treatments included (1) clearcut and (2) thinning to a 50% shade removal.
 1568 Canopy cover measurements were made using a concave spherical densiometer. Preharvest
 1569 canopy measurements ranged from 56% to 88%, with an average of 63% in the clearcut reaches,
 1570 and 74% in the partial cut reaches. In the clearcut reaches, canopy was reduced to 52% in 2002
 1571 and 41% in 2003, immediately following broadcast burning and replanting. In 2004 and 2005,

Commented [AJK101]: This study had a very modest sample size, if I recall correctly...

Commented [AK102R101]: ADDRESS

Commented [bs103R101]: included

1572 overall canopy was measured at 56% and 54%, respectively. Streamside shade recovery can be
1573 attributed entirely to low-lying understory species, as evidenced by the increase in
1574 understory/deciduous cover of 26% in 2003 to 39% and 37% in 2004 and 2005, respectively. In
1575 the partial cut reaches, canopy shade remained near 75%.

1576 In fish-bearing streams of Montana, Sugden et al. (2019) assessed the effectiveness of state
1577 riparian management harvest prescriptions in maintaining canopy cover. Montana state law
1578 requires timber be retained within a minimum of 15.2 m of fish-bearing streams, with equipment
1579 exclusion zones extended on steep slopes for up to 30.5 m. Within the riparian management
1580 zone, no more than half the trees greater than 204 mm (8 in) diameter at breast height (DBH) can
1581 be removed. In no case, however, can stocking levels of leave trees be reduced to less than 217
1582 trees per hectare. Shade over the stream surface was not directly measured in this study. Rather,
1583 canopy cover was used as a general proxy, with two independent estimates of canopy cover
1584 employed. One method used the riparian cruise data to populate a canopy cover model within the
1585 Forest Vegetation Simulator (FVS), which estimated canopy cover for each study site, pre- and
1586 post-harvest. The second method measured canopy cover in the harvest reach every 30 m, both
1587 before and after timber harvest, using a concave spherical forest densiometer. Mean canopy
1588 cover in the SMZ, as modelled in FVS, decreased from 77% to 74% following timber harvest
1589 and 73% when subtracting windthrow to differentiate between direct and indirect impacts of
1590 management (Table 3). The mean canopy cover over the stream channel based on densiometer
1591 measurements was 66% pre-harvest and 67% post-harvest. Neither of these changes was
1592 statistically significant.

1593 Roon et al. (2021a) compared the effects of two experimental thinning treatments on shade in
1594 second growth redwood stands (40-60 years old) of northern California. This study took place
1595 between 2016 and 2018 with thinning treatments applied during 2017 giving 1-year pre-
1596 treatment and 1-year of post-treatment data. Two study sites prescribed treatment on one side of
1597 the stream of a 45 m buffer width with a 22.5 m inner zone with 85% canopy retention and a
1598 22.5 m outer zone that retained 70% canopy cover (Tectah watershed). At the third treatment site,
1599 thinning prescriptions included removal of up to 40% of the basal area within the riparian zone
1600 on slopes less than 20% on both sides of the channel along a ~100–150 m reach (Lost Man
1601 watershed, Redwood national park). Shade over streams was measured with hemispherical
1602 photos and effective shade was calculated in HemiView Canopy Analysis Software version 2.1.
1603 Results for the Tectah watershed showed a significant reduction in canopy closure by a mean of
1604 18.7%, (95% CI: -21.0, -16.3) and a significant reduction of effective shade by a mean of 23.0%
1605 (-25.8, -20.1) one-year post treatment. In the Lost man watershed, a non-significant reduction of
1606 mean shade by 4.1% (-8.0, -0.5), and mean canopy closure by 1.9% was observed in 2018.
1607 Results for below canopy light availability showed significant increases by a mean of 33% (27.3,
1608 38.5) in the Tectah watershed, and non-significant increases in Lost man watershed of 2.5% (-
1609 1.6, 5.6) by 2018.

1610 In general, the results from the studies reviewed above suggest changes in shade or canopy cover
1611 from pre- to post-harvest are directly impacted by the intensity of the treatment prescription.
1612 Buffer treatments vary between states and within states by stream type (e.g., fish-bearing or non-

Commented [WB104]: Why was this done?

Commented [WB105R104]: Address

Commented [bs106R104]: addressed

1613 fish-bearing), For the studies that quantified pre- to post-changes in shade along fish-bearing
 1614 streams (Cupp & Lofgren, 2014; Sugden et al. 2019), results show evidence that the application
 1615 of best management practices (BMPs) cause minimal or non-significant changes in shade
 1616 following harvest. For non-fish-bearing streams harvest prescriptions are much more variable.
 1617 Further, there are many more examples of application and comparison of different experimental
 1618 buffer treatments which vary by width or thinning targets.

Commented [WB107]: Please expand. There has been decades of research on this topic in WA, OR and CA and the differences in approaches, results, site specific responses could all be discussed here.

1619 Litter

Commented [WB108R107]: Do not Address

1620

Commented [JK109]: Red: This is another summary of the studies presented above. A table or graph with the combined data would be more helpful in answering the question. What are the buffers in place? 10, 20, 30m? What is the % change in shade observed following each treatment? While not inaccurate, the conclusion isn't a synthesis of the data.

1621 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)	2-years post-harvest 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	8 years post-harvest 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.

Commented [JK110R109]: See previous comments: Do not address

Commented [WB111]: Another aspect of litter is quality and decomposition rate and how that affects macroinvertebrate communities. This seems to be a missing piece of this review.

Commented [WB112R111]: Do not Address

Formatted: Font: Not Italic

1622

1623 Specific to western Washington, McIntyre et al. (2018) compared the change in litterfall inputs
 1624 from pre- to post-harvest under three different riparian harvest treatments. Treatments included a
 1625 two-sided 50-ft riparian buffer along at least 50% of the stream (FP; with clearcut to stream's
 1626 edge outside of the buffer), a two sided 50-ft buffer along the entire stream (100%), and a
 1627 clearcut to stream without a buffer (0%). Litterfall was collected with litter traps placed along the
 1628 mainstem channel of each site. Litter was dried and sorted by type (e.g., deciduous, conifer,
 1629 small wood) and ashed to compare weight. Results for litterfall input showed a decrease in total
 1630 litterfall input in the FP (P = 0.0034) and 0% (P = 0.0001) treatments between pre- and post-
 1631 treatment periods. Leaf litterfall (deciduous and conifer leaves combined) input decreased in the
 1632 FP (P = 0.0114) and 0% (P < 0.0001) treatments in the post-treatment period. In addition, conifer
 1633 (conifer needles and scales) litterfall input decreased in the FP (P = 0.0437) and 0% (P < 0.0001)
 1634 treatments, deciduous leaves in the 0% (P < 0.0001) treatment, wood (twigs and cones) in the FP

Commented [WB113]: This has been repeated multiple times now. Maybe include this in a table once and then refer to it throughout the document

Commented [WB114R113]: Do not Address

1635 (P = 0.0044) and 0% (P = 0.0153) treatments, and misc. (e.g., moss and flowers) in the 0% (P =
 1636 0.0422) treatment.

1637 In the Malcom Knapp Research Forests of British Columbia, Canada, Kiffney & Richardson
 1638 (2010) compared changes in litter input between riparian harvest prescriptions that included
 1639 clear-cut to stream edge, 10 m wide buffer reserve, and 30 m buffer reserves over the course of 8
 1640 years. No thinning was applied within the reserves. Upland treatment at all sites used clearcutting
 1641 methods. Vertical litter inputs were collected monthly and at approximately 6–8-week intervals
 1642 during each season for years 1,2,6,7, and 8 years after harvest. Litter was separated into
 1643 broadleaf deciduous, twig, needles, and other (seeds, cones, and moss) categories following
 1644 collection and subsequently dried and weighed using a microbalance. Results for post-harvest
 1645 changes in litterfall input by treatment per year are summarized in **Table 7**. Actual values of pre-
 1646 post-harvest changes in litterfall input by type, treatment, and year were not directly reported,
 1647 however, the authors report that post-harvest inputs of needles, twigs, and total particulate matter
 1648 were significantly lower for clearcuts compared to all other treatments.

1649 Table 7. Percent change in total litterfall percentage post-harvest by treatment per year from
 1650 Kiffney & Richardson (2010). Table reproduced and modified from Yeung et al. (2019)
 1651 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

Commented [JK115]: Red: this is a good example of the type of presentation of the data that is most useful, adding the data from McIntyre et al. for comparison will help tie the studies together for a broader picture of the affects of buffers on the measured variables.

Commented [JK116R115]: Address

Commented [bs117R115]: Unfortunately for the Litterfall study, McIntyre et al. only present 2 years of post harvest data, and only the total change (not presented yearly). Also, the reduction in litterfall in the McIntyre study is presented as change in ash-free dry mass, not as a percentage. A new table has been added that presents both results.

1652
1653 *Large Wood (LW) recruitment*

1654 Table 8. Treatment and responses for selected publications investigating Large Wood relevant to
1655 Q1.

<u>Reference</u>	<u>Treatment</u>	<u>Response</u>
<u>McIntyre et al. (2021)</u>	<u>Buffer width of 50 feet (100%); 50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)</u>	<u>8 years post-harvest</u> <u>Large wood recruitment rates were greatest during the first two years, 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, in the first 2 years. LW density continued to increase in the FP treatment by 42 and 41%, respectively, in years 5 and 8 post-harvest.</u>
<u>Ehinger et al. (2021)</u>	<u>1) 50 feet, 2) <50ft buffers (variable), and 3) unbuffered, harvested to the edge of the channel</u>	<u>3 years post-harvest</u> <u>Mean LW piece counts increased in the 50 feet (n = 8) and unbuffered (n = 7) treatments by 8 and 13%, respectively, and decreased in the <50 feet (n = 6) treatments by 15%.</u>

Formatted: Font: Not Italic

Formatted: Font: Not Italic

1656
1657 Specific to western Washington, McIntyre et al. (2021) compared the change in mean in-stream
1658 large wood from pre- to post-harvest under three different riparian harvest treatments in non-fish-
1659 bearing streams. Treatments included a two-sided 50-ft riparian buffer along at least 50% of the
1660 stream (FP; with clearcut to stream's edge outside of the buffer), a two sided 50-ft buffer along
1661 the entire stream (100%), and a clearcut to stream without a buffer (0%). Results showed a 66%
1662 (P < 0.001), 44% (P = 0.05) and 47% (P = 0.01) increase in mean large wood density in the 100%,
1663 FP and 0% treatments, respectively, in the first 2 years post-harvest compared with the pre-
1664 harvest period and after controlling for temporal changes in the references. Five years post-
1665 treatment the mean LW density in the FP continued to increase 42% (P = 0.08), and again 8 years
1666 post-treatment (41%; P = 0.09).

1667 Ehinger et al. (2021) also quantified changes in in-stream LW following similar riparian harvest
1668 prescription. Because of unstable slopes, total buffer area was 18 to 163% greater than the
1669 prescribed 50-foot-buffer. This resulted in 2 different buffer types 1) buffers encompassing the
1670 full width (50 feet), 2) <50ft buffers, and 3) unbuffered, harvested to the edge of the channel.
1671 Because of the separation into multiple treatments, sample sizes became small and unbalanced.
1672 Thus, no statistical analyses were conducted, and only descriptive statistics were applied for

1673 changes in stand structure and wood loading. However, given the lack of studies presenting
 1674 changes in LW recruitment from pre- to post-harvest, it is presented here for comparison. Results
 1675 showed the full buffer sites and <50 ft buffer sites received an average of 23 and 10 pieces/100 m
 1676 and 2.3 and 0.7 m³/100 m of large wood, respectively, post-harvest. The majority of recruited
 1677 large wood pieces had stems with roots attached (SWRW); 70, and 100% in the full buffer, and
 1678 <50 ft buffer types, respectively. Pre-harvest channel large wood loading ranged from 55.8 to 111
 1679 pieces/100 m and from 9.8 to 25.2 m³/100 m among buffer types. Piece counts increased in the
 1680 full buffer and unbuffered sites (8 and 13%, respectively), and decreased in the <50 ft buffers
 1681 (15%).

1682 *Sediment*

1683 [Table 9. Treatment and responses for selected publications investigating Sediment relevant to](#)
 1684 [Q1.](#)

<u>Reference</u>	<u>Treatment</u>	<u>Response</u>
Hatten et al. (2018)	Buffer width of 15 m (~ 50 feet), Oregon Forest practices	1 year post-harvest after 2 harvest events (n = 3) Mean suspended sediment concentrations (SSC) was 32 mg L⁻¹ (~63%) lower after the first harvest and 28.3 mg L⁻¹ (~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	3 years post-harvest 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	4 years post-harvest Total suspended sediment (TSS) load from the clearcut exceeded the predicted load by 152% (6,791 kg km⁻²) 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.

1685
 1686 No studies from Washington published since 2000 provide changes in sediment concentration or
 1687 transport from pre- to post-harvest. The Hard Rock study (McIntyre et al., 2021) reported their
 1688 results for water turbidity and suspended sediment export (SSE) were stochastic in nature and the
 1689 relationships between SSE export and treatment effects were not strong enough to confidently
 1690 draw conclusions. The lack of SSE in some high discharge events suggests that the basins are
 1691 likely to be supply limited. The Soft Rock study (Ehinger et al., 2021) similarly reported that
 1692 their results for changes in sediment post-harvest were highly variable. [The SSE data in the Soft](#)
 1693 [Rock study indicated that the marine sedimentary lithologies were more erodible than then](#)

1694 lithologies sampled in the Hard Rock Study. However, prediction equations could not be
 1695 calculated to predict the response of the treatment sites after harvest. Thus, strong conclusions
 1696 about the effectiveness of the Forest Practices harvest prescription rules on discharge and SSE
 1697 could not be drawn. [Harvest treatment effects on suspended sediment export could not be
 1698 calculated.]

Commented [WB118]: SSE was calculated, the authors state that it was difficult to draw any solid conclusions on the effectiveness of rule.

Commented [WB119R118]: Address

Commented [bs120R118]: Addressed

1699 Hatten et al. (2018) compared pre- to post-harvest suspended sediment concentrations (SSC) in a
 1700 western Oregon Alsea watershed. Treatments followed contemporary harvesting practices (no
 1701 buffer in non-fish-bearing streams with equipment exclusion zones, and a 15 m no-cut-buffer in
 1702 fish-bearing streams) resulted in non-significant changes in SSC at all treatment sites.
 1703 Surprisingly, in the fish-bearing streams there was a decrease in SSC (~63% and ~55%, after first
 1704 and second harvest, respectively) compared to pre-harvest values. Bywater-Reyes et al. (2017)
 1705 compared pre- to post-harvest changes in suspended sediment yield (SSY) following harvest in
 1706 the Trask River Watershed of western Oregon. Harvest treatments of study sub-watersheds
 1707 consisted of clearcuts (UM2 and GC3) and a clearcut with buffers (50 ft; ~15 m; PH4).
 1708 Following timber harvest, (water year 2013), increases in SSY occurred in all harvested
 1709 catchments. The SSY in both PH4 (clearcut with buffers) and GC3 (clearcut without buffers)
 1710 declined to pre-harvest levels by water year 2014. Interestingly, the SSY in UM2 (clearcut
 1711 without buffers) increased annually throughout the post-harvest period, ultimately resulting in
 1712 the highest SSY of all catchments during the final two years (2015-2016) of the study after
 1713 producing the lowest SSY in the pre-harvest period. Actual values for SSY and significance were
 1714 not reported.

1715 Karwan et al. (2007) compared changes in total suspended solids (TSS) in streams from pre- to
 1716 post-harvest in northern Idaho. Treatments in the paired-watershed experiment consisted of 1)
 1717 commercial clearcut of the watershed area of 50%, and was broadcast burned and replanted by
 1718 the end of May 2003, and 2) partial cut in which a target of 50% the canopy was removed in 50%
 1719 of the watershed in 2001, with final 10% of log processing and hauling in early summer of 2002.
 1720 All harvests were carried out according to best management practices and in accordance with the
 1721 Idaho Forest Practices Act. Results showed a significant and immediate impact of harvest on
 1722 monthly sediment loads in the clear-cut watershed ($p = 0.00011$), and a marginally significant
 1723 impact of harvest on monthly sediment loads in the partial cut ($p = 0.081$). Total sediment load
 1724 from the clearcut over the immediate harvest interval exceeded predicted load by 152% (6,791
 1725 kg km⁻²); however, individual monthly loads varied around this amount. The largest increases in
 1726 percentage and magnitude occurred during snowmelt months, namely April 2002 (560%, 2,958
 1727 kg km⁻²) and May 2002 (171%, 3,394 kg km⁻²). Neither treatment showed a statistical
 1728 difference in TSS during the recovery time 2-4 years after harvest (clearcut: $p = 0.2336$; partial
 1729 cut: $p = 0.1739$) compared to the calibration loads (pre-harvest).

1730 *Nutrients*

1731 Table 9. Treatment and responses for selected publications investigating Sediment relevant to
 1732 Q1.

<u>Reference</u>	<u>Treatment</u>	<u>Response</u>
------------------	------------------	-----------------

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 2 years post-harvest; and of 6.20, 5.34, and 8.49 kg/ha/yr in the extended period (7-8 years post-harvest). 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	4 years post-harvest Significant increases in nitrogen were observed in the clearcut (n = 1) and partial cut treatments (n = 1). Increases at the clearcut site were greatest from 0.06 mg-N L ⁻¹ (pre-harvest) to 0.35 mg-N L ⁻¹ (post-harvest period, 4 years). There was also an observable seasonal effect on NO ₃ + NO ₂ concentrations with the peak concentration of 0.89 mg-N L ⁻¹ , with mean monthly concentrations of 0.43 mg-N L ⁻¹ and 0.59 mg-N L ⁻¹ 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)	6 years post-harvest (Phase I), 8 years post-harvest (Phase II); (n = 7) Mean annual NO ₃ + NO ₂ 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 ⁻¹ yr ⁻¹ – 3.95 kg ha ⁻¹ yr ⁻¹). NO ₃ + NO ₂ concentrations followed an increasing trend throughout the post-harvest period with evidence of recovery in year 8 indicated by the flattening of the cumulative load curve.

1733

1734 The “Hard Rock” study (McIntyre et al., 2021) results showed an increase in total-N export of
 1735 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,
 1736 and 0% treatments, respectively, in the first 2 years; and of 6.20 (P = 0.095), 5.34 (P = 0.147),
 1737 and 8.49 (P = 0.026) kg/ha/yr in the extended period (7-8 years post-harvest). Results for nitrate-
 1738 N export showed changes similar to but slightly less than those seen in the total-N analysis with
 1739 a relative increase in nitrate-N export of 4.79 (P = 0.123), 9.63 (P = 0.004), and 14.41 (P <0.001)
 1740 kg/ha/yr post-harvest in the 100%, FP, and 0% treatments, respectively in the first 2 years. None
 1741 of the changes in the extended period were significant. However, the authors note that there was
 1742 high variability in the data for the extended period and nitrate-N export only returned to pre-
 1743 harvest levels in one watershed. Total phosphorus export increased post-harvest by a similar
 1744 magnitude in all treatments: 0.10 (P = 0.006), 0.13 (P = 0.001), and 0.09 (P = 0.010) kg/ha/yr in
 1745 the 100%, FP, and 0% treatments, respectively in the first 2 years post-harvest. Changes in
 1746 phosphorus were not reported in the extended period.

1747 Gravelle et al. (2009) compared pre- to post changes in NO³ and NO² concentrations in
 1748 headwater streams following a clearcut and a partial cut (50% removal of canopy cover) in
 1749 northern Idaho. Riparian buffers and leave trees are not required for non-fish bearing headwater

1750 streams in Idaho. Results showed statistically significant increases in NO^3 and NO^2
1751 concentrations following clearcut and partial harvest cuts in headwater streams ($p < 0.001$).
1752 Increases at the clearcut treatment site were greatest, where mean monthly concentrations
1753 increased from 0.06 mg-N L⁻¹ during the calibration period to 0.35 mg-N L⁻¹ in the post-
1754 harvest period. Mean monthly concentrations in the partial cut increased from 0.04 mg-N L⁻¹ in
1755 the pre-harvest period to 0.05 mg-N L⁻¹ in the post-harvest period. No significant changes of
1756 in-stream concentration of any other nutrient recorded (total Kjeldahl nitrogen (TKN), TP, total
1757 ammonia nitrogen (TAN) consisting of unionized (NH_3) and ionized (NH_4^+) ammonia, and
1758 unfiltered orthophosphate (OP)) were found between time periods and treatments.

1759 Deval et al. (2021) compared changes in the same nutrient concentrations in the same area of
1760 northern Idaho but with an additional harvest prescription several years later. For this analysis,
1761 time periods were broken into four distinct phases: 1) pre-disturbance (1992–1997), 2) post-road
1762 (1997–2001), 3) experimental-harvest Phase I (PH-I) (2001–2007), and 4) operational sequential
1763 harvest Phase II (PH-II) when the extent and frequency of harvests increased (2007–2016). PH-I
1764 represents an experimental treatment phase during which harvest activities were experimentally
1765 controlled (only upstream headwater watersheds were harvested and mature vegetation (size or
1766 age threshold for “mature” not reported) removal ranged between 24% and 47%) followed by
1767 site management operations including broadcast burning and replanting. PH-II represents the
1768 post-experimental phase where the study area transitioned to operational treatments that
1769 consisted of additional road construction and timber harvest, with site management operations
1770 including pile burning and competition release herbicide application. During this operational
1771 phase, the mature vegetation (size or age threshold for “mature” not reported) removal in the
1772 upstream watersheds ranged between 36% and 50%. The response in $\text{NO}^3 + \text{NO}^2$ concentrations
1773 was negligible at all treatment sites following the road construction activities. However, $\text{NO}^3 +$
1774 NO^2 concentrations during the PH-I period increased significantly ($p < 0.001$) at all treatment
1775 sites. Similar to the PH-I period, all watersheds experienced significant increases in $\text{NO}^3 + \text{NO}^2$
1776 concentration during the PH-II treatment period ($p < 0.001$). Similar to Gravelle et al. (2009),
1777 significant increases in all other nutrients recorded were not detected.

1778

1779 Focal Question 1a

1780 *1a. What are the effects of thinning (intensity, extent) on the riparian functions, over the short*
1781 *and long-term compared to untreated stands?*

1782 Based on the wording of this question, papers deemed appropriate are those that compare
1783 changes in measurable data indicative of the riparian functions between harvested and
1784 unharvested stands. Further, studies chosen for this question should compare the response of
1785 these functions based on different thinning intensities. Thus, the design of the studies reviewed
1786 for this review should be a BACI or ACI design with results reported for differences between
1787 treatment and reference reaches. Also included are a few simulation modeling experiments that
1788 follow these designs.

1789

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

1790

1791 *Shade*

1792 Table 10. Treatment and responses for selected publications investigating Shade relevant to Q1a.

Formatted: Font:

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)	2-5 years post-harvest 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)	1-year post-harvest 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%.

1793

1794 Anderson et. al. (2007) compared changes in canopy cover at stream centers between sites
 1795 adjacent to different riparian zone treatments and an untreated control. This study was conducted
 1796 in young headwater forests of western Oregon. Treatments included three buffer widths: 1) one
 1797 site-potential tree averaging 69 m (B1), 2) variable width buffer averaging 22 m (VB), or 3)
 1798 streamside retention buffer averaging 9 m (SR-T). Adjacent upland to each buffer treatment was
 1799 thinned to ~198 trees per hectare. Results showed that visible sky at stream center only differed
 1800 significantly between SR-T (9.6%) and the untreated (4.2%) sites post-harvest. These results
 1801 were reported for the period 2-5 years post-harvest.

1802 Roon et. al. (2021a) used a BACI analysis to evaluate significant changes in canopy cover
 1803 relative to untreated reaches following 2 different thinning intensities in second growth redwood
 1804 forests of northern California. One study site prescribed treatment on one side of the stream of a
 1805 45 m buffer width with a 22.5 m inner zone with a target 85% canopy retention and a 22.5 m
 1806 outer zone that retained 70% canopy cover (Green Diamond Resource Company, Tectah
 1807 watershed). The treatment site, thinning prescriptions included removal of up to 40% of the basal
 1808 area within the riparian zone on slopes less than 20% on both sides of the channel along a ~100–
 1809 150 m reach (Lost Man watershed, Redwood national park). Control reaches were located

1810 upstream from treatment reaches. Data analysis was conducted separately for each experimental
 1811 watershed (i.e., 1 Lost man site, 2 Tectah sites). Results for the Tectah watershed showed a
 1812 significant reduction in canopy closure by a mean of 18.7%, (95% CI: -21.0, -16.3) and a
 1813 significant reduction of effective shade by a mean of 23.0% (-25.8, -20.1) one-year post
 1814 treatment. In the Lost Man watershed, a non-significant reduction of mean shade by 4.1% (-8.0, -
 1815 0.5), and mean canopy closure by 1.9% was observed. Results for below canopy light availability
 1816 showed significant increases by a mean of 33% (27.3, 38.5) in the Tectah watershed, and non-
 1817 significant increases in Lost Man watershed of 2.5% (-1.6, 5.6). Data for canopy closure and
 1818 effective shade were recorded for 1-year pre- and 1-year post-harvest.

1819 LW

1820 Table 10. Treatment and responses for selected publications investigating Large Wood relevant to
 1821 Q1a.

Formatted: Line spacing: Multiple 1.08 li
 Formatted: Font: Not Italic

Reference	Treatment	Response
Benda et al. (2016)	simulation modeling 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	Simulated 100-year post harvest results 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)	5-years post-harvest The SR and AAS LW input rates by volume were nearly 300% and 50% higher than the reference stream rates, respectively. Wood recruitment in the SR sites was significantly greater than in the AAS and reference sites. Conversely, differences in wood recruitment did not differ significantly between the AAS and reference sites.

Formatted: Font: Not Italic

1822
 1823 Benda et al. (2016) used simulation modeling to estimate the changes in in-stream LW volume
 1824 over time between sites with thinning treatments and unharvested reference sites. They used
 1825 ORGANON growth models to simulate forest growth and LW recruitment over a 100-year
 1826 period. The model simulated treatments of single entry thinning from below (thinning from
 1827 below removes the smallest trees to simulate suppression mortality) with and without a 10 m
 1828 width no-cut buffers; and a double entry thinning from below with the second thinning occurring
 1829 25 years after the first with and without 10 m no-cut buffers (results with 10 m buffer presented
 1830 in question 1b). Each thinning treatment was also combined with some mechanical introduction
 1831 of thinned trees into the stream encompassing a range between 5 and 20 % of the thinned trees.
 1832 The single-entry thin reduces stand density to 225 tph in 2015 (-67 %) and declines further to
 1833 160 tph by 2110 (-77 %). The double entry thinning resulted in 123 tph after the second thinning
 1834 in 2040 (-82%) and maintained that density until 2110. Both thinning treatments resulted in a

1835 substantial reduction of dead trees that could contribute to in-stream wood loads. The model
1836 output for single entry thinning treatments predicts a 33% or 66% reduction of in-stream wood
1837 over a century relative to the unharvested reference for harvest on one side or both sides of the
1838 stream, respectively. Including mechanical tipping of 5,10,15, and 20% of cut stems without a
1839 buffer in the single-entry thinning treatment changes the relative in-stream percentages of wood
1840 relative to the reference stream to -15, -6, +1, and +6%, respectively. Double entry thinning
1841 treatments without a buffer predicted further reduction in wood recruitment over a century of
1842 simulation with 42 and 84% reduction of in stream wood relative to the reference stream when
1843 one side and both sides of the channel were harvested. To offset the predicted changes of in
1844 stream wood volume following double entry harvest would require tipping of 10% of cut stems.
1845 The authors conclude that thinning without some mitigation efforts resulted in large losses of in
1846 stream wood over a century.

1847 Schuett Hames and Stewart (2019a) compared recruitment rates of LW and volume of in-stream
1848 LW between different riparian buffer thinning treatments and unharvested reference sites.
1849 Treatments evaluated included prescriptions for standard shade rule (a 30-ft no-cut buffer width,
1850 and thinning 30-75 ft from the stream), and all available shade rule (requires retention of all
1851 shade providing trees in this area) for eastern Washington. Results showed cumulative wood
1852 recruitment from tree fall over the five-year post-harvest interval was highest in the standard
1853 shade rule (SR) group, lower in the all-available-shade rule (AAS) group and lowest in the
1854 reference (REF) group. The SR and AAS rates by volume were nearly 300% and 50% higher
1855 than the REF rates, respectively. Wood recruitment in the SR sites was significantly greater than
1856 in the AAS and reference sites ($P < 0.05$). Conversely, differences in wood recruitment did not
1857 differ significantly between the AAS and reference sites.

1858 *Sediment*

1859 Karwan et al. (2007) used BACI analysis to compare changes in total suspended solid (TSS)
1860 yields between thinned sites and unharvested reference sites. This study was conducted in the
1861 Mica Creek Experimental watershed of northern Idaho and focused on non-fish bearing
1862 headwater streams. The thinning treatment included a target 50% canopy removal without no-cut
1863 buffers. Results showed a marginally significant ($P = 0.081$) increase in TSS relative to the
1864 reference streams in the first year following treatment. However, differences in TSS between the
1865 treatment streams and reference streams were not significant ($p = 0.174$) in the period 2-4 years
1866 post-harvest.

1867 *Nutrients*

1868 Yang et al. (2021) compared changes in stream chemistry between streams along thinned stands
1869 and unharvested reference stands in young mixed conifer headwater basins of the Sierra National
1870 Forest. Thinning treatment included mastication of shrub cover to $< 10\%$ and harvesting of trees
1871 to a target basal area of $27\text{--}55 \text{ m}^2 \text{ ha}^{-1}$. Data for dissolved organic carbon (DOC) and dissolved
1872 organic nitrogen (DON) were recorded for 2 years prior to and 3 years after treatment. For
1873 stream water, volume-weighted concentrations of DOC were 66- 94% higher in thinned
1874 watersheds than in control watersheds for all three consecutive drought years following thinning
1875 ($p = 0.06, 0.01, \text{ and } 0.05$ for years 1,2, and 3 post-harvest, respectively). No differences in DOC

1876 concentrations were found between thinned and control watersheds before thinning ($p = 0.50$,
 1877 and 0.74 for pre-harvest years 1 and 2, respectively). Volume-weighted concentrations of DIN
 1878 were 24% higher in thinned than in control watersheds only in the third year following thinning
 1879 ($p = 0.04$). No differences in DIN were detected between treatment and reference streams in the
 1880 2 pre-harvest years ($P \geq 0.44$). Note: Drought occurred at both sites during the three post-harvest
 1881 years which may have compounded these effects. This is discussed in more detail in question 3.

1882

1883 **Focal Question 1b**

1884 *1b. How do buffer widths and adjacent upland timber harvest prescriptions influence impacts of*
 1885 *riparian thinning treatments?*

1886 An experimental design that could provide information useful in answering this question would
 1887 involve a comparison of sites with different buffer widths, all with upland harvest, and data
 1888 would need to be recorded before and after thinning, with or without a control site (BAI, BACI),
 1889 or differences after thinning between treatment and control sites (ACI). Three papers include an
 1890 experimental design that investigate different buffer widths or different upland treatments along
 1891 with riparian thinning treatments.

1892 *Shade*

1893 Anderson et al. (2007) compared changes in canopy cover at stream centers between sites
 1894 adjacent to different riparian zone treatments and an untreated control. This study was conducted
 1895 in young headwater forests of western Oregon. Treatments included three buffer widths (1) one
 1896 site-potential tree averaging 69 m (B1), (2) variable width buffer averaging 22 m (VB), or (3)
 1897 streamside retention buffer averaging 9 m (SR-T); the adjacent upland to each buffer was thinned
 1898 to ~198 trees per hectare. Results showed that visible sky at stream center only differed
 1899 significantly between SR-T (9.6%) and the untreated (4.2%) sites post-harvest. These results
 1900 were reported for the period 2-5 years post-harvest.

1901 *LW*

1902 [Table 11. Treatment and responses for selected publications investigating Large Wood relevant to](#)
 1903 [Q1b.](#)

<u>Reference</u>	<u>Treatment</u>	<u>Response</u>
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.	5 years post-harvest 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;	Simulated 100-year post harvest results 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.
-------------------------------------	--	--

1904

1905 Burton et al. (2016) examined the relationship between annual in-stream wood loading and
 1906 riparian buffer widths adjacent to upland thinning operations. Buffer widths were 6, 15, or 70
 1907 meters and upland thinning was to 200 trees per ha (tph), with a second thinning (~10 years later)
 1908 to ~85 tph, alongside an unthinned reference stand of ~400 tph. Their results showed that slightly
 1909 higher volumes of wood were found in sites with a narrow 6-m buffer (not significant), as
 1910 compared with the 15-m and 70-m buffer sites in the first 5 years after the first harvest and
 1911 maintained through year 1 of the second harvest (end of study). The authors attributed this
 1912 difference to a higher likelihood of logging debris and/or windthrow, but these factors were not
 1913 analyzed.

1914 Benda et al. (2016) used simulation modeling to estimate the changes in in-stream LW volume
 1915 over time between sites with thinning treatments and unharvested reference sites. They used
 1916 ORGANON growth models to simulate forest growth and LW recruitment over a 100-year
 1917 period. The model simulated treatments of single entry thinning from below (thinning from
 1918 below removes the smallest trees to simulate suppression mortality) with and without a 10 m
 1919 width no-cut buffers; and a double entry thinning from below with the second thinning occurring
 1920 25 years after the first with and without 10 m no-cut buffers. Each thinning treatment was also
 1921 combined with some mechanical introduction of thinned trees into the stream encompassing a
 1922 range between 5 and 20 % of the thinned trees. The single-entry thin reduces stand density to 225
 1923 tph in 2015 (-67 %) and declines further to 160 tph by 2110 (-77 %). The double entry thinning
 1924 resulted in 123 tph after the second thinning in 2040 (-82%) and maintained that density until
 1925 2110. Both thinning treatments resulted in a substantial reduction of dead trees that could
 1926 contribute to in-stream. The model output for single entry thinning treatments predicts a 33% or
 1927 66% reduction of in-stream wood over a century relative to the unharvested reference for harvest
 1928 on one side or both sides of the stream, respectively. Adding the 10-m no cut buffer reduced total
 1929 loss to 7 and 14%. Including mechanical tipping of 5,10,15, and 20% of cut stems without a
 1930 buffer in the single-entry, thinning treatment changed the relative in-stream percentages of wood
 1931 relative to the reference stream to -15, -6, +1, and +6%, respectively. To completely offset the
 1932 loss of in stream wood due to single entry thinning, mechanical tipping of 14 and 12% were
 1933 required without and with buffers. Double entry thinning treatments without a buffer predicted
 1934 further reduction in wood recruitment over a century of simulation with 42 and 84% reduction of
 1935 in stream wood relative to the reference stream when one side and both sides of the channel were
 1936 harvested. Adding a 10 m buffer reduced total reduction of in stream wood to 11 and 22% for
 1937 thinning on one and both sides of the channel. To offset the predicted changes of in stream wood
 1938 volume following double entry harvest would require tipping of 10 and 7% of cut stems without

1939 and with the 10-m buffer. The authors conclude that thinning without some mitigation efforts
1940 resulted in large losses of in stream wood over a century.

1941

1942 [Focal Question 1c](#)

1943 *1c. What are the effects of clearcut gaps in riparian stands (intensity, extent) on the riparian*
1944 *functions, over the short and long-term, compared to untreated stands?*

1945 This question uses the general term “clearcut gaps” as a treatment within the riparian area but
1946 does not define a minimum or maximum threshold for gap size. Thus, studies reviewed that used
1947 a “patch” treatment were included as having information useful in answering this question. The
1948 question also identifies a comparison with untreated stands. Therefore, any design with a control
1949 site (BACI, ACI) is appropriate.

1950 There appears to be a paucity of studies in the literature that investigate the effects of gaps or
1951 patch harvesting treatments on riparian function within riparian stands. Only 4 papers discussed
1952 the effects of prescribed gaps or patches in the riparian area on riparian function.

1953 The “Hard Rock” study from McIntyre et al. (2021) and the “Soft Rock” study from Ehinger et
1954 al. (2021) present the most relevant results useful for answering this question. Riparian buffer
1955 prescriptions for non-fish bearing streams in western Washington use a gap design. In this
1956 design, a 50-foot buffer is required along at least 50% of the treated stream length. The
1957 remaining 50% or less of the treated riparian management zone can be clear cut to the stream
1958 edge. The Hard Rock study compared differences in shade, in-stream sediment and nutrient
1959 concentrations, and large wood recruitment between treated and unharvested reaches for 8-9
1960 years post-harvest. The first iteration of the Hard Rock study (McIntyre et al. 2021) also
1961 compared differences in litter inputs following treatment for 2 years post-harvest between
1962 treatment and reference reaches.

1963 The Soft Rock study compared differences in the same functions between treated and
1964 unharvested reaches, with 3-6 years of post-harvest sampling depending on the function under
1965 investigation, but only for 3 years post-harvest. However, because of unstable slopes in some of
1966 the sites in the Soft Rock study, many of the buffers were required to be wider than 50-feet
1967 (ranging from 18 –160% wider than 50-feet). Conversely, some of the sites treated ended up with
1968 buffers narrower than 50 feet. Further, there was limited availability of sites that fit the criteria
1969 (marine sediment lithology, timing of treatment). Because of these limitations, statistical
1970 analysis, and comparison of response between treatments and references for many stream
1971 temperature and shade functions, could not be performed. However, descriptive statistics were
1972 provided that contain useful information. Results from formal statistical analyses are provided
1973 for all other functions. Thus, the results are only descriptive, but they provide useful information
1974 for comparison to the Hard Rock study.

1975 *Shade*

1976 [Table 12. Treatment and responses for selected publications investigating Shade relevant to Q1c.](#)

Commented [WB121]: Extended monitoring was conducted (through Post 6) and included as addendum chapters.

Commented [WB122R121]: Address

Commented [bs123R121]: Corrected.

Commented [WB124]: This could provide some relevant information on patches of narrow buffers.

Commented [WB125R124]: Do not Address

Commented [WB126]: Statistical analyses were performed. Some descriptive statistics were used in Chapter 3, the remaining 4 chapters had formal statistical analysis done. Please go through document and accurately reflect the statistical analyses performed when discussing portions of the Soft Rock report.

Commented [WB127R126]: Address

Commented [bs128R126]: Addressed

1977

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	1-year post-harvest 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.	1-year post-harvest (n = 6) Post-harvest significant increase in mean reach light to a mean of 3.91 (SD ± 1.63) moles of photons m ⁻² day ⁻¹ , overall resulting in a mean change in light of 2.93 (SD ± 1.50) moles of photons m ⁻² day ⁻¹ . 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%).

1978

1979 [The Hard Rock study reported significant that decreases in canopy cover (measured at 1 meter
1980 above the stream surface with a spherical densiometer) for all were significant across all years
1981 for the treated sites immediately following harvest compared to the reference sites (p < 0.05).

1982 [The mean canopy cover decreased from 96% (pre-harvest) to 72% in the first-year post-harvest
1983 and continued to decline for four years reaching a minimum of 54%. After year four, mean
1984 canopy cover began to recover increasing annually until year 9 to 74%. In contrast, mean canopy
1985 cover in the reference sites was 95% before harvest and never fell below 85% for 9 years. In the
1986 Soft Rock study, mean canopy closure decreased in the treatment sites from 97% in the pre-
1987 harvest period to 75%, 68%, and 69% in the first, second, and third post-harvest years,
1988 respectively; and was further related to the proportion of stream buffered and to post-harvest
1989 windthrow within the buffer. Canopy closure remained stable in the reference sites throughout
1990 the course of the study, ranging from 95 to 99%.

1991 Janisch et al. (2012) compared canopy cover before and after application of a “patched buffer”
1992 treatment with unharvested control reaches in headwater streams of western Washington. The
1993 “patched buffer” treatment included retention of portions of the riparian forests ~50-110 m long
1994 in distinct patches along the channel with the remaining riparian area clearcut. There was no
1995 standard width for patched buffers, with buffers spanning the full width of the floodplain area
1996 and/or extending some undefined distance away from the stream. Canopy density was measured
1997 once in the summer prior to logging and once in the summer following logging. The percentage
1998 of visible sky was determined from digital photos taken with a fish-eye lens using Hemiview
1999 Canopy Analysis software. Canopy cover in all streams averaged 95% prior to harvest and did

Commented [WB129]: Was not significant across all years for all treatments, see previous comment on this subject.

Commented [WB130R129]: Address

Commented [bs131R129]: corrected

2000 not differ between treatment and reference streams. Following treatment, canopy cover in the
2001 patch buffer treatment averaged 76% and differed significantly from reference reaches.

2002 Swartz et al. (2020) tested the effects of adding canopy gaps within young (40 – 60 years old),
2003 regenerating forests of western Oregon on stream light availability and stream temperatures.
2004 While light availability and stream temperature are not functions described in the FPHCP, they
2005 are directly related to shade (an FPHCP function). Also, they directly affect water quality and
2006 aquatic habitat productivity which are functional objectives within the FPHCP. Further,
2007 considering the paucity of studies available that investigate the effects of clearcut gaps, the
2008 results are presented here. The addition of gaps in the young regenerating forests were used to
2009 theoretically mimic the natural disturbance regimes and the higher canopy complexity of late-
2010 successional forests. The researchers used a BACI design on six replicated streams within the
2011 McKenzie River Basin. In each treatment reach, gaps were designed to create openings in the
2012 canopy that were approximately 20 m in diameter. Gaps were centered on a tree next to the
2013 stream and spaced approximately 30 meters apart along each reach. The BACI analysis showed
2014 strong evidence for significant increase in mean reach light ($p < 0.01$) up to 3.91 ($SD \pm 1.63$)
2015 moles of photons $m^{-2} day^{-1}$ and an overall mean change in light of 2.93 ($SD \pm 1.50$) moles of
2016 photons $m^{-2} day^{-1}$. Mean stream shading could not be evaluated in the full BACI analysis
2017 because post-treatment hemispherical photographs could not be taken at all sites due to fire
2018 impeding access. For the remaining sites, the areas beneath each gap had notable localized
2019 declines in shade, though the entirety of the treatment reach mean shading declined by only 4%
2020 ($SD \pm 0.02\%$).

2021 Litter

2022 The Hard Rock study only quantified changes in litter input for 2 years after treatment (McIntyre
2023 et al., 2018). While significant decreases in litter input were observed from pre- to post-harvest
2024 in the treatment sites (described in focal question 1) these values were not significant when
2025 compared to the changes in the reference sites. Litter input was not quantified in the Soft Rock
2026 study.

2027 LW

2028 For the Hard Rock study, large wood recruitment and loading were only compared between the
2029 reference reaches and the buffered portion of the treatment reaches. The authors report large
2030 wood recruitment into the channel was 3 times greater on average in the treatment buffer than in
2031 the reference over the 8-year post-treatment period. However, while considerable, these
2032 differences were not significant for any analyzed post-harvest interval (e.g., 1-2 years post, 1-5
2033 years post, or 1-8 years post). The lack of significance was attributed to the large variability in
2034 recruitment values among treatment sites. The greatest increase in LW recruitment in the
2035 treatment sites relative to the reference sites occurred in the first 2 years post-harvest. Large
2036 wood loading (pieces/m of channel length) increased significantly ($\alpha = 0.10$) in the treatment
2037 reaches, relative to the reference sites in the first 2 years (47%; $p = 0.05$), 5 years (42%; $p =$
2038 0.08), and 8 years (41%; $p = 0.09$) post-harvest. For the Soft Rock study there was little post-
2039 harvest large wood input in reference sites: an average of 4.3 pieces and 0.34 m³ of combined in-

Commented [WB132]: How young, please provide age of forest if available

Commented [WB133R132]: Address

Commented [bs134R132]: included

Commented [WB135]: Stream temperature is included in the FPHCP under Performance Goals 3. "meet or exceed water quality standards" - This includes stream temperature.

It's also a functional objective and a performance target in Appendix N (Schedule L-1).

If it's necessary to point out that stream temperature is not described as a function, please provide the connections to water quality and shade that are within the FPHCP.

Commented [WB136R135]: Address

Commented [bs137R135]: Addressed.

2040 and over-channel volume per 100 m of channel. In contrast, the full buffer sites and <50 ft buffer
2041 sites received an average of 23 and 10 pieces/100 m and 2.3 and 0.7 m³/100 m of large wood,
2042 respectively.

2043 *Sediment*

2044 For the Hard Rock study, results for water turbidity and suspended sediment export (SSE) were
2045 stochastic in nature and the relationships between SSE and treatment effects were not strong
2046 enough to confidently draw conclusions. Water turbidity and SSE increased with stream
2047 discharge during large storm events but rapidly declined. The Soft Rock study reported similar
2048 issues with the data for SSE in that it appeared to be driven by site and event specific factors and
2049 strong conclusions could not be drawn. The authors report that the softer lithologies sampled as
2050 part of this study were more erodible than the competent lithologies sampled in the companion
2051 Hard Rock Study.

2052 *Nutrients*

2053 The Hard Rock study analyzed changes in total nitrogen and nitrate export in the gap buffers
2054 relative to untreated reference streams. Results showed an increase in total nitrogen export in the
2055 treatment sites of 10.85 kg/ha/yr ($p = 0.006$) in the first two years post-harvest relative to the
2056 reference sites. In the extended periods, total nitrogen export increased by 5.34 ($p = 0.147$)
2057 kg/ha/yr relative to the reference streams. Results for NO³ export showed similar but slightly
2058 lower increases than total nitrogen with a relative increase in NO³ export of 9.63 ($p = 0.004$)
2059 kg/ha/yr for the first two years post-harvest relative to the reference. None of the changes in
2060 nitrate exports in the extended period were significant. The Soft Rock study reported significant
2061 increases in concentrations of total nitrogen ($p < 0.05$) and NO³ ($p < 0.05$) post-harvest in the
2062 treatment sites relative to the reference sites. The change in export appeared related to the
2063 proportion of stream buffered.

2064

2065 [Focal Question 1d](#)

2066 *1d. How do buffer widths and upland timber harvest influence impacts of clearcut gaps*
2067 *treatments?*

2068 The wording of this question implies that the effects of clearcut gaps (discussed in focal question
2069 1c) on riparian function could be impacted when paired with different buffer widths and upland
2070 harvest prescriptions. Similar to the results of the search in literature for focal question 1c, there
2071 was a paucity of riparian function studies that implemented a clearcut gap or patch cutting
2072 method within the riparian area. The added layer of complexity in this question specifying
2073 differences in buffer widths and upland harvests only further refined the selection of appropriate
2074 papers. Of the studies reviewed above, none included the evaluation of different buffer widths or
2075 different upland harvests in their experimental design. The Hard Rock study compared the
2076 clearcut gap buffers to full retention buffer and unbuffered sites (discussed in the literature
2077 review section), but different widths were not compared in the gap buffer treatments.

2078

2079 Focal Question 1e

2080 *I.e. What are the effects of any combinations of the above treatments?*

2081 No studies found in our search compared the effects of combined treatments on one or more of
2082 the five functions, likely because combining multiple treatments into one design has the potential
2083 to confound results and are difficult to implement with sufficient sample sizes. The majority of
2084 the studies listed in our review investigate the effects of buffer width, thinning treatments, and
2085 upland treatments separately.

2086 The only papers with some extractable evidence of the compounding/ameliorating effects of
2087 combined treatments were focused on shade. One study, Reiter et al. (2020), compared the
2088 effects of thinned and unthinned buffers, and clearcut on changes in percent shade over adjacent
2089 streams (discussed in focal question 1). However, changes in shade were not statistically
2090 analyzed and the implementation of the upland thinning treatment only occurred at one site
2091 (Table 6).

2092

2093 Focal Question 2

2094 *2. How and to what degree do specific site conditions (e.g., topography, channel width and*
2095 *orientation, riparian stand age and composition) influence the response of the riparian*
2096 *functions?*

2097 Multiple studies have investigated the influences of site conditions on riparian function. Few
2098 studies reviewed (4) investigated the interaction between specific site conditions (e.g., slope,
2099 lithology, elevation) and harvest on the response of riparian function. However, if these specific
2100 site conditions influence the magnitude of riparian function in the absence of harvest, it is
2101 possible they can compound the effects of harvest on their response. Thus, studies that assess the
2102 relationship between site factors and riparian function may provide some useful insight for
2103 management and are presented below. Further, we also included studies that investigated the
2104 relationships between road development and sediment transport because road development is
2105 directly related to changes in local topography.

2106 *Litter*

2107 **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	1-2 years post-treatment (n = 5). Deciduous-site vertical litter input (504 g m ⁻¹ y ⁻¹) exceeded that from coniferous sites (394 g m ⁻¹ y ⁻¹ , 336.4–451.7) by 110 g/m ² (28.6–191.6) over the full year. Annual lateral inputs at deciduous sites (109 g m ⁻¹ y ⁻¹) were 46 g/m more than at coniferous sites (63 g m ⁻¹ y ⁻¹). 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 increased with slope at deciduous sites (R² = 0.4073) , but showed no strong relationship at coniferous sites (R² = 0.1863) .

Commented [AJK138]: Again, what conclusions can be drawn, collectively, from the studies?

Commented [TA(139R138)]: Per 9/5 email from AJ: DO NOT ADDRESS

Formatted: Font: Italic

	<u>and parallel to the stream</u>	
<u>Bilby & Heffner (2016)</u>	<u>Simulation modeling and field sampling</u>	<u>1-year of litterfall data 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.</u>

2108

Formatted: Font: Not Italic

2109 Hart et al. (2013) compared litter delivery into streams between riparian zones dominated by
 2110 deciduous (red alder) and coniferous (Douglas-fir) tree species in western Oregon. Results from
 2111 this study show that deciduous forests dominated by red alder delivered significantly greater
 2112 vertical and lateral inputs ($\text{g m}^{-2} \text{y}^{-1}$) to adjacent streams than did coniferous forests dominated by
 2113 Douglas-fir. Deciduous-site vertical litter input (mean = $504 \text{ g m}^{-2} \text{y}^{-1}$) exceeded that from
 2114 coniferous sites ($394 \text{ g m}^{-2} \text{y}^{-1}$) by 110 g/m^2 over the full year. Annual lateral inputs at
 2115 deciduous sites ($109 \text{ g m}^{-2} \text{y}^{-1}$) were $46 \text{ g m}^{-2} \text{y}^{-1}$ more than at coniferous sites ($63 \text{ g m}^{-2} \text{y}^{-1}$).
 2116 The timing of the inputs also differed, with the greatest differences occurring in November
 2117 during autumn peak inputs for the deciduous forests. Further, annual lateral litter input increased
 2118 with slope at deciduous sites ($R^2 = 0.4073$, $p = 0.0771$), but showed no strong relationship at
 2119 coniferous sites ($R^2 = 0.1863$, $p = 0.2855$). These results were partially consistent with Bilby &
 2120 Heffner (2016) in that they suggest litter type, and topography (slope) can affect the litter input
 2121 rates.

2122 Bilby & Heffner (2016) used a combination of field experiments, literature review, and modeling
 2123 to estimate the relative importance of factors affecting litter delivery from riparian areas into
 2124 streams of western Washington in the Cascade mountains at high and low elevations. Their
 2125 results for conifer needles released at mature sites had a higher proportion of cumulative input
 2126 from greater distances than needles or leaves released at younger sites. The authors suggest from
 2127 their interpretation of the model that the width of the litter contributing area was $\sim 35\%$ greater at
 2128 mature sites than at young sites. The mean age of “mature” and “young” sites was not specified
 2129 but the mean tree heights were 47.0 m and 32.4 m for the mature and young sites, respectively.
 2130 Thus, tree height is related to the width of the litter contributing area for conifer needles. Litter
 2131 travel distance was also linearly related to wind speed ($p < 0.0001$). Doubling wind speed at one
 2132 site led to a 67-87% expansion of the riparian litter contribution zone in the study area.
 2133 Interpretation of the regression curves revealed a trend that suggests hillslope gradient affects the
 2134 width of the litter contributing area as well. However, the authors did not apply statistical

2135 analysis to these values and only speculated that increasing the slope from 0-45% would increase
 2136 the width of the litter contributing area by up to 70%.

2137 *LW*

2138

2139

2140 Table 14. Treatment and responses for selected publications investigating Large Wood relevant to
 2141 Q2.

Formatted: Font: Not Italic

<u>Reference</u>	<u>Treatment</u>	<u>Response</u>
<u>Wing & Skaugset (2002)</u>	<u>Relationships between channel and habitat characteristics with LW piece count and volume</u>	<u>Observation data from in 3793 stream reaches in western Oregon State. LW volume: reaches with < 2.3% gradient averaged 5.8 m³ while higher gradient streams averaged 17.9 m³ per reach for all land types (ownership, forested and non-forested). Reaches with gradients less than 4.7% averaged LW volume of 11.5 m³, while mean volume at higher gradient reaches was 25.2 m³. LW pieces: Streams <12.2 m bank full width 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), stream gradient 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.</u>
<u>Sobota et al. (2006)</u>	<u>patterns of riparian tree fall directions</u>	<u>Data was collected from 21 field sites. 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%).</u>

2142

2143 Wing & Skaugset (2002) investigated the relationships between channel and habitat
 2144 characteristics with LW piece count and volume in stream reaches in western Oregon. This study
 2145 analyzed an extensive spatial database of aquatic habitat conditions created for western Oregon
 2146 using stream habitat classification techniques and a geographic information system (GIS).
 2147 Regression tree analysis (an exploratory regression analysis that allows for the inclusion of
 2148 multiple explanatory variables) was used to compare the relative strength of each variable in
 2149 predicting LW volume. Explanatory variables used in this analysis included morphology of
 2150 active channel (hillslope, terrace, terrace hillslope, unconstrained), and lithology (e.g., alluvium,
 2151 basalt, etc.). Results for channel characteristics showed that stream gradient was the most
 2152 important explanatory variable for LW volume. The split for stream gradient occurred for reaches
 2153 with < 2.3% gradient (mean LW volume: 5.8 m³ per reach) while higher gradient streams showed
 2154 a mean LW volume of 17.9 m³ per reach.

2155
2156 For LW pieces in forested stream reaches bankfull channel width was the most important
2157 explanatory variable with the split occurring for streams channels less than 12.2 m wide. LW
2158 pieces for streams <12.2 m wide averaged 11.1 LW pieces per reach while larger channels
2159 averaged 4.9 pieces per reach; in this model the BFW split explained 7% of the variation in LW
2160 pieces found in forested streams. For key LW pieces (logs at least 0.60 m in diameter and 10 m
2161 long) in forested reaches, stream gradient was again the most important explanatory variable
2162 with the split occurring at a slope of 4.9%. The streams with a gradient < 4.9% averaged 0.5 key
2163 LW pieces per reach while streams with higher gradients averaged 0.9 key LW pieces per reach;
2164 in this model stream gradient explained 8% of the variation in key LW pieces found in streams.

2165
2166 Lithology caused second, third or fourth level splits after stream gradient or BFW. Specifically,
2167 Mesozoic sedimentary and metamorphic geologies, located in southern Oregon stream reaches,
2168 were grouped and split from basalt, Cascade, and marine sedimentary geologies. In stream
2169 reaches with Mesozoic sedimentary and metamorphic geologies, the quantity of LWD was
2170 roughly half the amount found in other geologies. The only exception to this grouping was for
2171 LW volume in larger stream reaches, where basalt and marine sedimentary geologies contained
2172 more LW volume when grouped separately from all other geologies in a fourth-level split. The
2173 authors conclude that the geomorphic characteristic of stream reaches, in particular stream
2174 gradient and bankfull width, correlated best with LW presence.

2175
2176 Sobota et al. (2006), evaluated patterns of riparian tree fall directions in diverse environmental
2177 conditions and evaluate correlations with tree characteristics, forest structural variables, and
2178 topographic features. Specifically, the authors were interested in correlations between fall
2179 directionality and tree species type, tree size, riparian forest structure, and valley topography
2180 (side slope). Data was collected from 21 field sites located west of the Cascade Mountains crest
2181 (11 sites: Coast Range and west slopes of the Cascades), and in the interior Columbia Basin (10
2182 sites: east slopes of the Cascades, Blue Mountains, and Northern Rockies) of Oregon,
2183 Washington, Idaho, and Montana, USA. Streams were second- to fourth-order channels and had
2184 riparian forests that were approximately 40 to >200 years old. Model projections of LW
2185 recruitment estimated that sites with uniform steep side slopes (>40%) produced between 1.5 to
2186 2.4 times more in stream LW by number of tree boles than sites with uniform moderate side
2187 slopes (< 40%). The authors warn that while side slope categories (>40%, <40%) was the
2188 strongest predictor of tree fall direction in this study, they believe the differences in tree fall
2189 direction between these categories mainly characterized differences between fluvial (88% of
2190 moderate slope sites) and hillslope landforms (71% of steep slope sites). They suggest that the
2191 implications from this study are most applicable to small- to medium-size streams (second to
2192 fourth order) in mountainous regions where sustained large wood recruitment from riparian
2193 forest mortality is the significant management concern.

2194 *Sediment*

2195 [Table 15. Treatment and responses for selected publications investigating Sediment relevant to](#)
2196 [Q2.](#)

Formatted: Space After: 8 pt, Line spacing: Multiple
1.08 li

Formatted: Font: Not Italic

<u>Reference</u>	<u>Treatment</u>	<u>Response</u>
<u>Bywater-Reyes et al. (2017)</u>	<u>basin lithology and physiography effects on sediment delivery</u>	<u>6 years of data from the Trask River Watershed</u> <u>Site lithology 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</u>
<u>Bywater-Reyes et al. (2018)</u>	<u>catchment lithography, physiography, discharge, and disturbance history effects on sediment delivery</u>	<u>60 years of data in the H.J. Andrews experimental watershed (n = 10) Watershed slope variability combined with cumulative annual discharge explained 67% of the variation in annual sediment yield. When considering disturbance, the largest magnitude changes in sediment movement, were after floods with a > 30-year return interval.</u>
<u>Mueller & Pitlick (2013)</u>	<u>correlation analysis to assess the relative impact of lithology, basin relief, mean basin slope, and drainage density on in stream sediment supply.</u>	<u>Data sets ranging 1-96 years for 83 basins</u> <u>the strongest correlation of bankfull sediment concentration was with basin lithology, 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.</u>
<u>Litschert & MacDonald (2009)</u>	<u>Post-harvest stream sediment delivery pathway development frequency and characteristics.</u>	<u>1-year post-harvest data (n = 200 harvest units)</u> <u>19 harvest units developed sediment delivery pathways. Pathway length and probability of connecting to stream was significantly correlated with mean annual precipitation, cosine of the aspect, elevation, and hillslope gradient.</u>

2197
2198

2199 Bywater-Reyes et al. (2017) assessed the influence of natural controls (basin lithology and
2200 physiography) and forest management on suspended sediment yields in temperate headwater
2201 catchments. This study analyzed 6 years of data from the Trask River Watershed in northeastern
2202 Oregon and included data from harvested and unharvested sub-catchments underlain by
2203 heterogenous lithologies. Results from this study indicate that site lithology was the first order
2204 control over suspended sediment yield (SSY) with SSY varying by an order of magnitude across
2205 lithologies observed. Specifically, SSY was greater in catchments underlain by Siletz Volcanics
2206 (r = 0.6), the Trask River Formation (r = 0.4), and landslide deposits (r = 0.9) and displayed an
2207 exponential relationship when plotted against the percentage of watershed area underlain by
2208 these lithologies. In contrast, site lithology had a strong negative correlation with percent area
2209 underlain by diabase (r = 0.7), with the lowest SSY associated with 100% diabase. Following
2210 timber harvest, increases in SSY occurred in all harvested catchments but returned to pre-harvest
2211 levels within 1 year except for sites that were underlain by sedimentary formations and were
2212 clearcut without protective buffers. The authors conclude that sites underlain with a friable
2213 lithology (e.g., sedimentary formations) had, on average, SSYs an order of magnitude higher
2214 following harvest than those on more resistant lithologies (intrusive rocks).

Formatted: Font: Not Italic

2215 Bywater-Reyes et al. (2018) quantified how sediment yields vary with catchment lithography and
2216 physiography, discharge, and disturbance history (management or natural disturbances) over 60
2217 years in the H.J. Andrews experimental watershed in the western Cascade Range of Oregon. A
2218 linear mixed effects model (log transformed to meet the normality assumption) was used to
2219 predict annual sediment yield. In this model, site was treated as a random effect while discharge
2220 and physiographic variables were treated as fixed variables. This allowed for the evaluation of
2221 the relationships between sediment yield and physiographic features (slope, elevation, roughness,
2222 and index of sediment connectivity) while accounting for site. To account for the effect of
2223 disturbance history a variable was added to the model when the watershed had a history of
2224 management or natural disturbances. If the models for the disturbed watersheds significantly
2225 underpredicted the sediment discharge, the timing of the sudden increases were further examined
2226 to assess whether it correlated with a disturbance event. The results showed that watershed
2227 physiography combined with cumulative annual discharge explained 67% of the variation in
2228 annual sediment yield across the 60-year data set. Relative to other physiographic variables,
2229 watershed slope was the greatest predictor of annual suspended sediment yield. However, the
2230 results showed that annual sediment yields also moderately correlated with many other
2231 physiographic variables and caution that the strong relationship with watershed slope is likely a
2232 proxy for many processes, encompassing multiple catchment characteristics.

2233 Mueller & Pitlick (2013) used correlation analysis to assess the relative impact of lithology,
2234 basin relief, mean basin slope, and drainage density on in stream sediment supply defined by the
2235 bankfull sediment concentration (bedload and suspended load). The study used sediment
2236 concentration data from 83 drainage basins in Idaho and Wyoming. Lithologies of the study area
2237 were divided into four categories ranging from hardest to softest- granitic, metasedimentary,
2238 volcanic, and sedimentary. The results showed the strongest correlation of bankfull sediment
2239 concentration was with basin lithology, and showed little correlation strength with slope, relief
2240 and drainage density. As lithologies become dominated by softer parent materials (volcanic and
2241 sedimentary rocks), bankfull sediment concentrations increased by as much as 100-fold. The
2242 authors interpret these results as evidence that lithology can be more important in estimating
2243 sediment supply than topography.

2244 Rachels et al. (2020) used sediment source fingerprinting techniques to quantify the proportional
2245 relationship of sediment sources (hillslope, roads, streambanks) in harvested and un-harvested
2246 watersheds of the Oregon Coast Range. The study included one catchment (Enos Creek) that was
2247 partially clearcut harvested in the summer of 2016 and an unharvested reference catchment
2248 (Scheele Creek) located ~3.5 km northwest of Enos Creek. The paired watersheds had similar
2249 road networks, drainage areas, lithologies and topographies. The treatment watershed was
2250 harvested with a skyline buffer technique in the summer of 2016 under the Oregon Forest
2251 practices Act policy that requires a minimum 15 m no-cut buffer. The proportion of suspended
2252 sediment sources were similar in the harvested ($90.3 \pm 3.4\%$ from stream bank; $7.1 \pm 3.1\%$ from
2253 hillslope) and unharvest ($93.1 \pm 1.8\%$ from streambank; $6.9 \pm 1.8\%$ from hillslope) watersheds.
2254 However, the harvested watershed contained a small portion of sediment from roads ($3.6 \pm$
2255 3.6%), while the unharvested reference watershed suspended sediment contained no sediment
2256 sourced from roads. In the harvested watersheds the sediment mass eroded from the general

2257 harvest areas (96.5 ± 57.0 g) was approximately 10 times greater than the amount trapped in the
2258 riparian buffer (9.1 ± 1.9 g), and 4.6 times greater than the amount of sediment collected from
2259 the unharvested hillslope (21.0 ± 3.3 g). These results suggest that the riparian buffer was
2260 efficient in reducing sediment erosion relative to the harvested area. The caveat of this study was
2261 the limited sample size (1 treatment, 1 paired reference watershed) and does not incorporate the
2262 effects of different watershed physiography on sediment erosion. However, it is presented here as
2263 evidence that the formation of roads within a riparian area may interact with timber harvest to
2264 increase the potential flow of sediments from roads.

2265 Litschert & MacDonald, (2009) investigated the frequency of sediment delivery pathways in
2266 riparian management areas and their physical characteristics and connectivity following harvest.
2267 In this study the authors describe sediment delivery pathways (“features”) as rills, gullies, and
2268 sediment plumes that form when excess sediment relative to overland flows transports sediment
2269 from the hillslope to the stream. The authors surveyed 200 riparian management areas (RMA) in
2270 four different National Forests of the Sierra Nevada and Cascade Mountains of California. USFS
2271 policy requires 90-m wide RMA along each side of perennial streams and 45-m wide RMA along
2272 each side of all ephemeral and intermittent streams. When features were found within an RMA,
2273 data for years since harvest, soil depth, soil erodibility (K), feature length, feature gradient,
2274 aspect, elevation, hillslope gradient, hillslope curvature, surface roughness, and connectivity
2275 were recorded for analysis. Association between these variables were analyzed with a
2276 Spearman’s rank correlation. The variables most strongly associated with feature length were
2277 used to develop a multiple linear regression model to predict feature length. Only 19 of the 200
2278 harvest units had sediment development pathways. Feature pathways ranged in age (time since
2279 harvest) from 2 to 18 years, and in length from 10 m to 220 m. Of the 19 feature pathways, only
2280 six were connected to streams, and five of those originated from skid trails. Feature pathway
2281 length was significantly related to mean annual precipitation, cosine of the aspect, elevation, and
2282 hillslope gradient ($R^2 = 64\%$, $p = 0.004$). These results suggest that within treated riparian areas
2283 topographic characteristics such as aspect, elevation and hillslope gradient can affect delivery of
2284 sediment into streams.

2285 Rashin et al. (2006) evaluated the effectiveness of Washington State best management practices
2286 (BMPs) for controlling sediment related water quality impacts. Although this study was
2287 published in 2006, the data analyzed in this study were collected between 1992 and 1995. In their
2288 evaluation, Rashin et al. (2006) assessed site erosion, sediment delivery, channel disturbance,
2289 and aquatic habitat condition within the first two years of harvest along fish- and non-fish
2290 bearing streams across Washington state. From their results, the authors concluded that the site-
2291 specific factors influencing the effectiveness of BMPs in preventing chronic sediment delivery
2292 into streams were 1) the proximity of ground disturbance to the stream, 2) presence of a stream
2293 buffer, 3) falling and yarding practices that minimized disturbance to stream channel, and 4)
2294 timing of harvest activities for certain climate zones where frozen ground or snow cover may be
2295 exploited. The landscape factors that influenced BMP effectiveness were 1) the density (specific
2296 metric not reported) of unbuffered small streams at harvest sites, and 2) steepness of stream
2297 valley slopes. The authors conclude with a recommendation of excluding timber falling and
2298 yarding activities at least 10 m from streams and outside of steep inner gorges.

2299 From the studies reviewed there is evidence that sediment delivery into streams following timber
2300 harvest is influenced by not only the intensity of the harvest operation (e.g., presence of retention
2301 buffers, yarding and equipment use immediately adjacent to the stream, upland clearcut vs.
2302 thinning), but also by physiography (especially hillslope gradient), lithology relative softness,
2303 and the presence of roads. Thus, the change in magnitude of sediment delivery following harvest
2304 is context dependent and these landscape factors can interact with one another to compound
2305 these changes. However, from the studies reviewed in the sediment section of the literature
2306 review, there is evidence that the implementation of BMPs since the 1970s in the northwestern
2307 United States has lessened the impact and duration of these changes.

2308 *Nutrient*

2309 None of the studies published since 2000 and conducted in western North America provide
2310 experimental evidence of the effects of site factors on nutrient flux into streams. However, Zhang
2311 et al. (2010) conducted a global review and meta-analysis of the effectiveness of buffers in
2312 reducing nonpoint source pollution. They reported slope (hillslope gradient) as having a linear
2313 relationship with buffer pollutant removal efficacy that switched from positive to negative when
2314 slope increased beyond 10% (i.e., hillslope gradients of ~10% were optimal for buffer efficacy in
2315 removing pollutants).

2316

2317 *Focal Question 3*

2318 *3. What is the frequency of weather-related effects (e.g., windthrow, ice storms, excessive heat,*
2319 *flood and drought events) on riparian areas? What are the weather-related effects (positive and*
2320 *negative) on the riparian functions, and how are they distinguished from harvest effects? How do*
2321 *these effects differ between treated and untreated riparian forests?*

2322 The first part of this question “What is the frequency of weather-related effects (e.g., windthrow,
2323 ice storms, excessive heat, flood and drought events) on riparian areas?” is a generally worded
2324 question asking how often weather events in riparian areas occur. The second part of this
2325 question “What are the weather-related effects (positive and negative) on the riparian functions,
2326 and how are they distinguished from harvest effects?” contains within it 2 parts 1) what the
2327 effects on the riparian functions are, and 2) how they are distinguished from timber harvest
2328 effect. Any study reviewed that answers one or more parts of this question have been included.

2329 *Shade*

2330 McIntyre et al. (2021), the “Hard Rock” study, compared changes in shade from pre- to post-
2331 harvest between three riparian harvest treatments and a reference. Treatments included a two-
2332 sided 50-ft riparian buffer along at least 50% of the stream (FP; with clearcut to stream’s edge
2333 outside of the buffer), a two sided 50-ft buffer along the entire stream (100%), and a clearcut to
2334 stream without a buffer (0%). The canopy cover was measured 1 meter above the stream surface
2335 with a spherical densiometer. The changes in canopy cover were distinguished from harvest
2336 effects and compared to unharvested reference sites by using a BACI design. For the FP
2337 treatment, mean canopy cover declined from 96% to 72% in the first-year post-harvest but

2338 continued to decline for 4 years to a minimum of 54%. In the 100% treatment mean canopy
 2339 cover was more stable, decreasing from 94% to 88% in the first year and reaching a minimum of
 2340 82% also by year 4. Canopy cover began to increase after year 4 through year 9 in both
 2341 treatments. In contrast, the reference sites experienced much smaller reductions in canopy cover
 2342 from 95% to 89% in the first four years. The cause of mortality in the treatment sites was
 2343 primarily attributed to windthrow. However, while post-harvest mortality in the treatment sites
 2344 were higher on average than in the reference sites there was a high amount of variability between
 2345 sites in both the treated and reference sites. For example, in the first 2 years following harvest
 2346 mortality ranged from 1.8 to 34.6% (loss of basal area) between sites in the FP treatment. In
 2347 contrast, mortality in the reference sites ranged from 1.1 to 20.4% (loss of basal area) during the
 2348 same period.

2349 *Litter*

2350 Bilby & Heffner (2016) showed evidence that wind speed has a strong effect on the width of
 2351 litter delivery areas within riparian areas. They used a combination of field experiments and
 2352 simulation modeling to estimate the influence of different site factors (physiography, stand age,
 2353 species composition, wind speed) on litter delivery into streams. Their results showed that litter
 2354 travel distance was also linearly related to wind speed ($p < 0.0001$). Doubling wind speed at one
 2355 site led to a 67-87% expansion of the riparian litter contribution zone in the study area. However,
 2356 this study does not compare the differences in the influence of wind speed on the width of the
 2357 litter contributing area between harvested and unharvested sites.

2358 *LW*

2359 [Table 16. Treatment and responses for selected publications investigating Large Wood relevant to](#)
 2360 [Q3.](#)

Reference	Treatment	Response
McIntyre et al. (2021)	50 feet (100%); 50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)	8-years post-harvest data 100% (n = 4), FP (n =3), and unbuffered treatments (n = 4) 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. Wind/physical damage was the primary cause of mortality 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	3 years post-harvest (n = 20) within no-cut buffers, windthrow caused mortality was up to 3 times greater than competition induced mortality for 3 years following treatment 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, 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

Formatted Table

		<u>in the reference sites. The authors estimate that windthrow mortality was twofold and fivefold greater in the inner and outer halves of the treatment buffers than in the reference buffers, respectively.</u>
<u>Bahuguna et al. (2010)</u>	<u>Buffer widths 10 m, and 30 m</u>	<u>7-years post-harvest (n = 3) In the first 2 years, 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 8 years post-harvest, a significant amount of annual windthrow caused mortality occurred in the unharvested control at 30%, compared to 15% in both 30 m and 10 m buffers.</u>
<u>Schuett-Hames & Stewart (2011, 2019b)</u>	<u>Buffer widths 50 feet</u>	<u>10-years post-harvest 3 years after treatment annual tree fall rates (live and dead) were over 8 times (by % of standing trees) and 5 times (by trees/acre/yr) higher in the 50-foot buffers than in the reference. 4-5 years after treatment mortality was still higher in the treated sites (27.3%) than in the reference (13.6%), but the difference was not significant. 10 years after treatment stand mortality in the 50-ft buffer treatment stabilized.</u>

2361

Formatted: Font: Not Italic

2362 Chapter 3 of the Hard Rock study compared changes in stand mortality and LW input from pre-
 2363 post-harvest and between treated and untreated reference sites. Results showed that by year 8,
 2364 post-harvest mortality as a percentage of pre-harvest basal area was lower in the reference
 2365 (16.1%) than in the 100% (24.3%) and FP (50.8%) treatments. The FP–Reference contrast in
 2366 mortality was not significant 2 years post-harvest, but it was at 5- and 8-years post-harvest as
 2367 mortality in FP increased relative to the Reference over time. The contrast in mortality between
 2368 the 100% and Reference were not significant for any time interval 8 years post-harvest.
 2369 Wind/physical damage was the primary cause of mortality for all treatments, including the
 2370 Reference. In the 100% treatment it accounted for 78% and 90% of the loss of basal area and
 2371 density (stem/ha), respectively; in FP it accounted for 78% and 65% of the loss. Wind accounted
 2372 for a smaller proportion of mortality in the reference (52% and 43%, respectively).

2373
 2374 LW recruitment to the channel was greater in the 100% and FP treatment than in the reference for
 2375 each pre- to post-harvest time interval. Eight years post-harvest mean recruitment of large wood
 2376 volume was two to nearly three times greater in 100% and FPB RMZs than in the references.
 2377 Annual LW recruitment rates were greatest during the first two years, then decreased. However,
 2378 there was a great deal of variability in recruitment rates within treatment sites and the differences
 2379 between treatments were not significant. Mean LW loading into the channel (pieces/m of channel
 2380 length) differed significantly between treatments in the magnitude of change over time. There
 2381 was a 66%, 44% and 47% increase in mean large wood density in the 100%, FP and 0%
 2382 treatments, respectively, in the first 2 years post-harvest compared with the pre-harvest period
 2383 and after controlling for temporal changes in the references. By year 8, only the FP treatment
 2384 showed a significantly higher proportional increase (41%) in wood loading when compared to
 2385 the reference. In the time interval 2-8 years post-harvest wood loading in the 100% treatment
 2386 stabilized.

2387
 2388 Liquori (2006) investigated treefall characteristics within riparian buffer sites in a managed tree

2389 farm in the Cascade Mountains of western Washington. Buffer widths ranged between 25-100
2390 feet along non-fish bearing and fish bearing streams. Results showed that within no-cut buffers,
2391 windthrow caused mortality was up to 3 times greater than competition induced mortality for 3
2392 years following treatment with tree fall probability highest in the outer areas (closest to upland
2393 clearcuts) of the buffers. Their results showed that treefall was generally highest at the outside
2394 edges of buffers (50+ feet), representing about 60% of the total observed treefall, while the 0–25-
2395 foot zone represented ~18%, and the 25–50-foot zone represented ~22%. The researchers
2396 interpret these results as evidence that windthrow susceptibility within riparian buffers increases
2397 with increasing distance from the stream.

2398
2399 Martin & Grotenfendt (2007) compared riparian stand mortality and in-stream LW recruitment
2400 characteristics between riparian buffer strips with upland timber harvest and riparian stands of
2401 unharvested watersheds using aerial photography in the northern and southern portions of
2402 Southeast Alaska. All buffer strips in this study were a minimum of 20 m wide and included
2403 selective harvest within the 20 m zone (thinning intensity not specified or included in the
2404 analyses as an effect). The results from this study showed significantly higher mortality (based
2405 on cumulative stand mortality: downed tree counts divided by standing tree counts + downed tree
2406 counts by number/ha), significantly lower stand density (269 trees/ha in buffer units and 328
2407 trees/ha in reference units), and a significantly higher proportion of LW recruitment from the
2408 buffer zones of the treatment sites than in the reference sites. Also, results showed that mortality
2409 varied with distance to the stream. Differences in mortality for the treatment sites were similar to
2410 the reference sites for the first 0-10 m from the stream (only a 22% increase in the treated sites).
2411 However, mortality in the outer half of the stream buffers (10-20 m) across treatment sites was
2412 more than double (120% increase) that observed within the reference sites. The authors estimate
2413 that windthrow mortality was twofold and fivefold greater in the inner and outer halves of the
2414 treatment buffers than in the reference buffers, respectively.

2415
2416 Bahuguna et al. (2010) evaluated the difference in windthrow caused mortality between 10 m, 30
2417 m buffer widths (neither had thinning within the buffer and both had upland clear-cuts) and
2418 unharvested controls in the Coast Mountains, British Columbia. Following harvest, 11% of
2419 initially standing timber was blown down in the first and second years in the 10 m buffer,
2420 compared to 4% in the 30 m buffer, and 1% in the unharvested controls. However, after 8 years
2421 post-harvest, a significant amount of annual mortality occurred when winter storms brought
2422 down multiple trees in the unharvested control at 30%, compared to 15% in both 30 m and 10 m
2423 buffers. These results show evidence that timber harvest can increase windthrow caused
2424 mortality within protective buffers in the short term but can stabilize within a decade. Further,
2425 this study shows evidence that windthrow caused mortality is stochastic and large storm events
2426 can cause significant mortality within untreated riparian forests.

2427
2428 Schuett-Hames and Stewart (2019a) compared changes in stand mortality and LW recruitment
2429 between treated and untreated riparian areas along fish-bearing streams in eastern Washington.
2430 Treatments were prescribed under the Standard Shade Rule (SR), under the All-Available Shade
2431 rule (AAS), and unharvested reference sites. Both shade rules have a 30-ft no-cut buffer (core

2432 zone) immediately adjacent to the stream. The SR prescription allows thinning in the buffer zone
2433 30-75 feet (inner zone) from the stream while the AAS prescription requires retention of all
2434 shade providing trees in this area. Thinning non-shade providing trees within the inner zone is
2435 allowed under the AAS rule. Results from a mixed model comparison showed that the frequency
2436 of wood input from fallen trees was significantly greater in SR group compared to both the
2437 reference and AAS groups ($p < 0.001$), while the difference between reference and AAS groups
2438 was not significant. Over 60% of pieces recruited from AAS and SR fallen trees consisted of
2439 stems with attached rootwads (SWAR), double the proportion in the reference sites. The
2440 reference-AAS and reference-SR differences in recruitment of SWAR pieces were significant (p
2441 < 0.001). The authors comment that the higher mortality and recruitment of LW in the SR sites
2442 was primarily due to windthrow.

2443 Schuett-Hames et al, (2011) compared tree mortality and LW recruitment between treated and
2444 untreated riparian stands along non-fish bearing streams in western Washington. Treated sites
2445 were prescribed a 50-foot-wide no-cut buffer. Annual fall rates of live and dead standing stems
2446 combined were over 8 times (by % of standing trees) and 5 times (by trees/acre/yr) higher in the
2447 50-foot buffers than in the reference buffers 3 years after treatment. These differences were
2448 significant for both metrics ($p < 0.001$). Over the entire five-year period, the percentages of
2449 standing trees that were uprooted and broken (as well as the combined total) were significantly
2450 greater in the 50-foot buffer. Wind was the dominant tree fall process, accounting for nearly 75%
2451 of combined fallen trees, 11% fell from other trees falling against them and 1.8% of fallen trees
2452 fell from bank erosion. Differences in mortality followed a similar pattern to tree fall rates. In the
2453 50-foot buffer sites mortality rates were significantly higher (3.5 times higher) than in the
2454 reference sites for the first three years following harvest. However, in years 4-5 mortality rates
2455 increased in the reference buffers after high-intensity storms resulting in non-significant
2456 differences in mortality during this period. The cumulative percentage of live trees that died over
2457 the entire five-year period was 27.3% in the 50-ft buffers compared to 13.6% in the reference
2458 reaches, but the difference was not statistically significant. The authors suggest that the lack of
2459 significance was likely due to the high variability in mortality between sites in the 50-foot
2460 buffers.

2461 In the follow-up study, Schuett-Hames & Stewart (2019b) reported that over a 10-year period,
2462 stand mortality in the 50-ft buffer treatment stabilized and showed a cumulative 14.1% reduction
2463 in live basal, while the reference stands showed a 2.7% increase in live basal area. The
2464 differences in these values were not significant. Cumulative LW recruited into stream channel
2465 over the 10-period was double in the 50-ft buffer treatment streams than in the reference streams.

2466 In general, the studies reviewed above show evidence that upland timber harvest with riparian
2467 retention buffers initially increases stand mortality within the buffers and increases LW
2468 recruitment relative to unharvested reference stands in the short-term. Hence, treated riparian
2469 forests appear to have a higher susceptibility to windthrow caused mortality, at least in the short
2470 term, compared to untreated stands. Depending on the streams in question, an increase in LW
2471 could be considered a positive or negative impact This increase in mortality and LW recruitment
2472 is attributed to an increase in the susceptibility to windthrow within the riparian buffers relative
2473 to the unharvested controls. Further, multiple studies (Liquori, 2006; Martin & Grotenfendt, 2007,

2474 Schuett-Hames & Stewart 2019a) showed evidence that the increase in windthrow caused
 2475 mortality is highest in the outer area of the riparian buffers (area closest to upland treatments).
 2476 There is some evidence that thinning within the buffer can also affect mortality rates, but these
 2477 studies are few. In the three studies that collected post-harvest data for 8 or more years
 2478 (Bahuguna et al., 2010; McIntyre et al., 2021; Schuett-Hames & Stewart 2019b), there is
 2479 indication that mortality in the riparian buffers and annual LW recruitment into adjacent streams
 2480 stabilizes within 5-10 years. However, in the subsequent decades following treatments with
 2481 upland clearcuts there is evidence that LW recruitment rates can continue to decrease and in
 2482 stream wood loads may become depleted before recruitment rates can recover (Nowakowski &
 2483 Wohl, 2008; Reid & Hassan, 2020) depending on applied management practices (e.g., buffer
 2484 widths, road construction, etc.). For example, Teply et al. (2007) used simulation modeling to
 2485 estimate the effectiveness of Idaho Forest Practices for riparian buffers and found no significant
 2486 difference between predicted LW loads for harvested and unharvested sites 30-, 60-, or 100-years
 2487 post-harvest.

2488 *Nutrient*

2489 Table 17. Treatment and responses for selected publications investigating Nutrients relevant to
 2490 Q3.

Reference	Treatment	Response
Vanderbilt et al. (2003)	long-term datasets from six watersheds in the H.J. Andrews Experimental Watershed	20-30 years of historical data Total annual discharge was a positive predictor of annual dissolved organic nitrogen (DON) export in all watersheds with R2 values ranging between 0.42 to 0.79. No other nutrients nitrate (NO3-N), ammonium (NH4-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 the earliest rain events , and peak DON concentrations occurred in October through December before the peak in the hydrograph. DON concentrations then declined during the winter months.
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 Drought alone 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. Drought combined with thinning showed 66- 94% higher DOC than in unthinned watersheds.

Formatted: Font: Not Italic

Formatted Table

2491 Formatted: Font: Not Italic

2492 Vanderbilt et al. (2003) analyzed long-term datasets (ranging 20-30 years for each watershed)
 2493 from six watersheds in the H.J. Andrews Experimental Watershed in the west-central Cascade
 2494 Mountains of Oregon to investigate patterns in dissolved organic nitrogen (DON) and dissolved
 2495 inorganic nitrogen (DIN) export with watershed hydrology. The researchers used regression
 2496 analysis of annual N inputs and outputs with annual precipitation and stream discharge to
 2497 analyze patterns. Their results showed that total annual discharge was a positive predictor of

2498 annual DON export in all watersheds with R^2 values ranging between 0.42 to 0.79. In contrast,
2499 relationships between total annual discharge and annual export of nitrate ($\text{NO}_3\text{-N}$), ammonium
2500 ($\text{NH}_4\text{-N}$), and particulate organic nitrogen (PON) were variable and inconsistent across
2501 watersheds. The authors speculate that different factors may control organic vs. inorganic N
2502 export. The authors emphasize the importance of analyzing data from multiple watersheds in a
2503 single climactic zone to make inferences about stream chemistry.

2504 Yang et al. (2021) investigated the effects of drought and forest thinning operations
2505 (independently and combined) on stream water chemistry in the Mediterranean climate
2506 headwater basins of the Sierra National Forest. The effects of drought alone were examined by
2507 comparing water samples collected from control watersheds for 2 years before and 3 years after
2508 drought. The effects of drought and thinning combined were examined by comparing water
2509 samples collected from treated sites to reference sites for three years post-harvest (all drought
2510 years). Drought alone altered the concentration of dissolved organic carbon (DOC) in stream
2511 water. Volume-weighted concentration of DOC was 62% lower ($p < 0.01$) and the ratio of
2512 dissolved organic carbon to dissolved inorganic nitrogen (DOC:DON) was 82% lower ($p =$
2513 0.004) in stream water in years during drought (WY 2013–2015) than in years prior to drought
2514 (WY 2009 and 2010). Drought combined with thinning altered DOC and DIN concentrations in
2515 stream. For stream water, volume-weighted concentrations of DOC were 66–94% higher in
2516 thinned watersheds than in control watersheds for all three consecutive drought years following
2517 thinning. No differences in DOC concentrations were found between thinned and control
2518 watersheds before thinning. The authors conclude that their results showed evidence that the
2519 influences of drought and thinning are more pronounced for DOC than for DIN in streams.

2520 *Drought Frequency*

2521 Wise (2010) used reconstructed newly collected tree-ring data augmented with existing
2522 chronologies from sites at three headwater streams in the Snake River Basin to estimate
2523 streamflow patterns for the 1600–2005 time-period. Streamflow patterns derived from
2524 instrumental data and from reconstructed chronologies were compared with other streamflow
2525 previously reconstructions of three other western rivers (the upper Colorado, the Sacramento,
2526 and the Verde Rivers) in similar climates to examine synchronicity among the rivers and gain
2527 insight into possible climatic controls on drought episodes. The reconstruction model developed
2528 for the analysis explained 62% of the variance in the instrumental record after adjustment for
2529 degrees of freedom. Results showed evidence that droughts of the recent past are not yet as
2530 severe, in terms of overall magnitude, as a 30-year extended period of drought discovered in the
2531 mid-1600s. However, in terms of number of individual years of $< 60\%$ mean-flow (i.e., low-flow
2532 years), the period from 1977–2001 were the most severe. Considering the frequency of
2533 consecutive drought years, the longest (7-year-droughts), occurred in the early 17th and 18th
2534 centuries. However, the 5-year drought period from 2000–2004 was the second driest period over
2535 the 415-year period examined. The correlative analysis of the chronologies developed for the
2536 upper Snake River with other rivers of the West showed mixed results with periods of positive
2537 and negative correlations. The author interprets these results as evidence that drought frequency,
2538 in general, in this area appears to be increasing in severity and that mean annual flow appears to

2539 be reducing in the latter half of the 20th and the beginning of the 21st century. The exceptions
2540 being the 1930's dustbowl, and an unusually long dry period in the early 1600s.

2541 *Fire Frequency*

2542 Dwire & Kauffman (2003) in their reviewed and summarized the available conducted on fire
2543 regimes in forested riparian areas relative to uplands in the western United States. They
2544 summarized the distinctive features of riparian areas that can influence the properties of fire as
2545 (1) higher fuel loads because of higher net primary productivity, (2) higher fuel moisture content
2546 due to proximity to water, shallow water tables, and dense shade, (3) active channels gravel bars
2547 and wet meadows may act as fuel breaks, (4) topographic position (canyon bottoms, low point on
2548 landscape) leads to higher relative humidity, fewer lightning strikes, but more human-caused
2549 ignitions, (5) microclimate may lead to cooler temperatures and higher humidity that can lessen
2550 fire intensity and spread. They highlight a need for more extensive research on the history and
2551 ecological role of fire in the riparian areas of the western United States.

2552 There is a logical assumption that fire in riparian zones would be less frequent than in adjacent
2553 uplands because of its proximity to water. However, several studies have been conducted which
2554 reconstruct historical fire regimes in riparian areas relative to adjacent uplands and have
2555 provided varying results. Everett et al. (2003) used fire-scar and stand-cohort records to estimate
2556 the frequency and seasonality of fire in Douglas-fir dominated riparian areas and adjacent
2557 uplands. They sampled sites along 49 stream segments on 24 different streams in the Wenatchee
2558 (33 segments) and Okanogan (16 segments) National Forests. The data collected allowed for
2559 reconstruction of fire occurrence back to 1896. Their results showed that the mean count of fire
2560 scars was significantly fewer in riparian areas than in adjacent uplands regardless of valley type,
2561 aspect, or plant association group. However, the difference between riparian and upland fire scars
2562 was greatest for western aspects and least for northern aspects. Also, the differences were
2563 greatest for the 'warm mesic shrub/herb' plant association group (e.g., common snowberry), and
2564 least in the cool dry grass plant association group (e.g., pinegrass, or elk sedge).

2565 Prichard et al. (2020) evaluated drivers of fire severity and fuel treatment effectiveness at the
2566 2014 Carlton Complex in north-central Washington State. While this study's objective does not
2567 specifically evaluate differences in fire severity between riparian and upland forests, it did
2568 evaluate differences in fire severity based on variations in topographic and vegetation type
2569 variables. One vegetation variable was classified broadly as "riparian vegetation" from the
2570 publicly available data set LANDFIRE. The authors used a combination of simultaneous
2571 autoregression and random forests approaches to model drivers of fire severity. In the study
2572 area's southern section (1 of 2 designated study areas), the results showed cover type was a
2573 significant predictor with negative correlations with fire severity in non-forest types and riparian
2574 forests.

2575 Conversely, Olson & Agee (2005) provide evidence that fire return intervals in the riparian areas
2576 of the Umpqua National Forests, Oregon, may not have differed significantly from adjacent
2577 upland forests. They reconstructed historical fire return intervals from fire scar cross sections
2578 taken from 15 stream reaches and 13 paired upland forests. Sites were primarily dominated by

2579 Douglas-fir, western red cedar, and western hemlock. The number of fires per plot, maximum
2580 and minimum fire return intervals, and the Weibull median fire return interval (WMPs)
2581 were compared between riparian and upland stands using the Wilcoxon signed rank test, the Mann-
2582 Whitney U-test for unmatched samples, and the Kruskal-Wallis one-way analysis of variance.
2583 The results showed that between 1650 and 1900, 43 fire years occurred on 80 occasions. Of these
2584 80 occasions, 33 were recorded in the riparian and adjacent upslope forest, 23 were recorded in
2585 only the riparian area, and 24 were recorded only in the upland forests. The riparian WMPs
2586 were somewhat longer (ranging from 35-39 years, with fire return intervals ranging from 4-167
2587 years) than upslope WMPs (ranging from 27-36 years, with fire return intervals ranging from 2-
2588 110 years), but these differences were not significant. The authors, Olson & Agee (2005),
2589 interpret these results as evidence that fires in this area were likely patchy and smaller in scale
2590 with a high incidence of fires occurring only in the riparian area or only in the upland forests,
2591 and less commonly in both. The authors also suggest that fire is a natural occurrence in the
2592 riparian areas of this area and should be restored to protect riparian forest health.

2593 Another study from the Klamath Mountains in northern California showed evidence that fires in
2594 riparian forests may have been more frequent than in adjacent upland forests (Skinner, 2003).
2595 Skinner (2003) used dendrochronological methods to construct fire return intervals for 5 riparian
2596 and adjacent upland forests sites, each between 1-2 hectares. Because of the small sample size,
2597 statistical analysis was not conducted, and their results are only descriptive. The ranges of fire
2598 return intervals (FRIs) were similar between riparian and upland forests. However, the median
2599 FRI for the riparian forests was nearly double that in adjacent uplands. The authors conclude that
2600 these limited data suggest fire in the riparian areas may be more variable than in the uplands in
2601 frequency and intensity.

2602 Yet, another study from Harley et al. (2020) showed evidence that the differential fire occurrence
2603 riparian and adjacent uplands may have been dependent on weather (i.e. drought). Harley et al.
2604 (2020) reconstructed low-severity fire histories from tree rings in 38 1-ha plots. This data was
2605 supplemented with existing fire histories from 104 adjacent upland plots. 2633 fire scars were
2606 sampled from 454 (127 riparian; 329 upland) trees from two sites in the Blue Mountains in
2607 north-eastern Oregon: One in the Wallowa-Whitman (WWNF) and one in the Malheur (MNF)
2608 National Forests. Fire-scar dates were used to construct plot composite fire chronologies,
2609 excluding fire dates recorded from only one tree. These were used to compute median fire
2610 intervals for riparian and upland forests for each site and for both sites combined. A mixed linear
2611 model with fire interval as a response and plot type (riparian vs. Upland) as a predictor was used
2612 to check for statistical difference in fire frequency. The influence of climate on fire occurrence
2613 was inferred by assessing whether the summer Palmer Drought Severity Index (PDSI) differed
2614 significantly during the fire year or preceding or following years (-3 to +1 years) using
2615 superimposed epoch analysis. Results showed that Fires burned synchronously in riparian and
2616 upland plots during more than half of the fire years at both WWNF and MNF (55% and 57%,
2617 respectively). At WWNF, fires burned during 65 years of the analysis period (1650–1900); 36
2618 burned in both riparian and upland plots, 7 burned only in riparian plots and 22 burned only in
2619 upland plots. At MNF, fires burned during 74 years of the analysis period; 42 burned in both
2620 riparian and upland plots, 3 burned only in riparian plots and 29 burned only in upland plots. At

2621 both sites, average PDSI was significantly warm–dry during synchronous fire years. However,
2622 climate was not significantly cool–wet during non-synchronous fire years at either site. The
2623 authors interpret these results as evidence that historical synchronized fire occurrence was more
2624 likely during excessively dry or drought years.

2625 There is also evidence that riparian forest fire regimes have been altered in many areas from pre-
2626 Euro-American settlement due to fire suppression. Messier et al. (2012), used dendro-ecological
2627 methods to reconstruct pre-Euro-American settlement riparian forest structure and fire frequency
2628 for comparison of changes post-settlement in the Rouge River of southwestern Oregon. Fire
2629 events were dated from increment cores and fire-scar cross-sections back to the year 1600,
2630 approximately. Changes in annual radial growth rates were used to infer changes in stand density
2631 over time. Results showed the age distribution prior to 1850 followed a pulse pattern of
2632 recruitment with recruitment peaks occurring around 1850, 1800, and between 1740-1770
2633 (though this pulse was difficult to discern because the sample size of trees established prior to
2634 1740 were relatively few). After 1900, many mixed conifer sites showed a dramatic increase in
2635 the recruitment of more more-shade tolerant white fir (*Abies concolor*) compared to Douglas-fir
2636 (*Pseudotsuga menziesii*). White fir comprised 51% of the live trees recruited after 1900, but only
2637 18% of the live trees before 1900. Results from the 26 cross-dated fire scars spanned from 1748
2638 – 1919 with the highest number of detected fires occurring in the early-settlement period (1850-
2639 1900). The authors interpret these results as evidence that fire suppression over the last century
2640 has changed the successional pathway and stand structure of riparian forests in this area.

2641 Van de Water & North (2011) found similar results from their study in the northern Sierra
2642 Nevada. They compared current field data with reconstructed data to estimate changes in stand
2643 structure, fuel loads, and potential fire behavior over time. Additionally, they estimated how
2644 these conditions for riparian forests compared to adjacent upland forests during the reconstructed
2645 and current periods. Data for current forest structure, species composition, and fuel loads were
2646 collected from 36 adjacent riparian and upland sites (72 sites total). The reconstruction period
2647 was set at the year of the last fire (ranging from 1848 – 1990), determined from fire-scar records.
2648 Potential fire behavior, effects, and canopy bulk density were estimated for current and
2649 reconstructed stand conditions for riparian and upland sites using Forest vegetation Simulator
2650 (FVS). Stand structure (BA, stand density, snag volume, QMD, average canopy base height),
2651 species composition, fuel load, potential fire behavior, canopy bulk density, and mortality were
2652 compared between current and reconstructed periods for riparian and upland sites, and between
2653 sampling areas (riparian vs. Upland) with an analysis of variance (ANOVA). Results showed that
2654 under current conditions, riparian forests were significantly more fire prone than upland forests,
2655 with greater stand density (635 vs. 401 stems/ha), probability of torching (0.45 vs. 0.22),
2656 predicted mortality (31% vs. 16% BA), and lower quadratic mean diameter (46 vs. 55 cm),
2657 canopy base height (6.7 vs. 9.4 m), and frequency of fire tolerant species (13% vs. 36% BA).
2658 However, the reconstructed periods showed no significant difference between riparian and
2659 upland forests for fuels and structure. The authors suggest that these results provide evidence that
2660 the historic fire return intervals may not have differed significantly between riparian and upland
2661 forests in this area.

2662 *Fire Effects on Function*

2663 *Litter and Nutrients*

2664 Table 17. Treatment and responses for selected publications investigating the relationship
2665 between wildfire Litter and Nutrients relevant to Q3.

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

Formatted Table

2666

Formatted: Font: Not Italic

2667 Musetta-Lambert et al. (2017) compared changes in leaf-litter inputs into streams following
2668 adjacent riparian forest harvesting or wildfire to reference sites. This study took place in the
2669 boreal forest of the White River Forest management Area in Ontario, Canada, ~75 km inland
2670 from the northern shore of Lake Superior. This study is outside of western North America (the
2671 focal area for this review), but it is the only study found that provides experimental evidence of
2672 wildfire's effects on litter inputs. The study sites consisted of ~50 m reaches in 25 catchments, 10
2673 that were harvested, 7 that experienced wildfire, and 8 references. Of these reaches a subset was
2674 used to riparian forest structure, leaf litter inputs, and water chemistry (5 harvest, 7 fire, 6
2675 reference). The harvested catchments were harvested 7-17 years prior to the study (minimum 30
2676 m riparian buffers; specific harvest rules/methods not described). The wildfire catchments had
2677 burned 12 years prior to the study and had no dead material removed. The reference catchments
2678 had no fire or harvesting for a minimum of 40 years. Water grab samples were collected in
2679 September, October and November 2010, and May, June and September of 2011 from the study
2680 reaches.

2681 Water samples were analyzed to obtain measurements for pH, conductivity, dissolved organic
2682 carbon (DOC) and dissolved inorganic carbon (DIC) concentrations, soluble reactive
2683 phosphorous (SRP), along with a suite of other major elements and nutrient measurements (total

2684 N, NH₄, total P, Ca, K, Mg, etc.). Vertical leaf litter traps consisting of plastic bins were placed at
2685 10 locations along the bankfull width of each site. Lateral leaf fall was not collected or analyzed.
2686 Leaf litter inputs were focused on leaves from deciduous trees and shrubs. Leaves were separated
2687 to the lowest possible taxonomic level, dried and weighed for analysis.

2688 Univariate one-way ANOVA models were used to determine differences in water chemistry,
2689 riparian forest characteristics of juvenile tree and shrub communities (richness, Shannon's
2690 diversity index, relative occurrence of individual taxa), mature tree communities (total basal
2691 area, stem density), and litter subsidies (richness, mass input). Results for water chemistry
2692 showed that Conductivity, pH, and dissolved inorganic carbon were significantly higher at fire
2693 sites than at reference sites ($p = 0.02$, $p = 0.04$, $p = 0.03$, respectively) but did not differ between
2694 harvested and fire sites or harvested and reference sites.

2695 Results for stand structure showed there was significantly higher taxa richness in fire sites than
2696 in reference sites or harvested sites ($p = 0.04$). Taxa richness did not differ significantly between
2697 reference and harvested sites. Reference sites had significantly higher total mean densities (# ha
2698 ⁻¹) of mature riparian trees (>10 cm DBH) than fire ($p < 0.001$) and harvested sites ($p = 0.036$).
2699 Total mature tree densities in reference sites were 1.7x and 4x higher than in harvested and fire
2700 sites, respectively. **3.3. Leaf litter subsidies** Taxa richness in leaf litter subsidies did not
2701 significantly differ among disturbances ($p = 0.477$). Total leaf litter input (g m^{-1}) significantly
2702 higher at fire sites than at harvest ($p = 0.02$) or reference sites ($p = 0.02$). Fire sites had
2703 significantly greater leaf litter inputs of willow spp. ($p = 0.0002$, 0.006 , respectively), Atlantic
2704 ninebark ($p = 0.002$, 0.003 , respectively) and speckled alder ($p = 0.02$, 0.04 , respectively) than in
2705 both reference and harvested sites. The authors interpret these results as evidence that natural fire
2706 disturbance in low-order boreal forest streams had higher leaf litter inputs, and different stand
2707 structures and composition than harvested or untreated riparian stands. They suggest that while
2708 harvested stands were more structurally similar to fire affected stands than reference stands, the
2709 future implementation of these treatments should intend to emulate the patchy nature of wildfire
2710 disturbance. This would enhance the diversity of riparian forest structure and increase litter
2711 subsidies into streams.

2712 *Nutrients*

2713 Rhoades et al. (2011) monitored stream chemistry and sediment 1-year before and for 5-years
2714 after the 2002 Hayman Fire in Colorado. Monthly water samples were collected from streams in
2715 three burned and three unburned watersheds. Pre-fire and post-fire water nitrate, cation
2716 concentration (Ca^{2+} , Mg^{2+} , K^{+}), acid neutralizing capacity (ANC) and turbidity were compared
2717 graphically and statistically between the three burned and unburned basins. Results for cation
2718 concentrations and ANC showed an immediate and significant increase that peaked during the 4-
2719 month period following the fire. The Ca^{2+} concentrations, ANC, and conductivity remained
2720 elevated in the burned streams for 2 years compared to pre-fire conditions, and unburned
2721 streams. Stream water nitrate and turbidity increased linearly with the proportion of a basin
2722 burned or burned at high severity. No other chemical analyte showed a significant response to
2723 fire severity or extent. Streams draining basins affected by extensive stand-replacement fires
2724 showed a 3.3-fold higher ($p = 0.000$) nitrate concentration than basins that burned less. Also,

2725 turbidity was 2.4-fold ($p = 0.000$) higher average turbidity compared to streams in basins burned
2726 less severely or extensively. In the extensively burned basins, stream water nitrate concentrations
2727 did not decline over the five years of the study and the mean concentrations of nitrate in the fifth
2728 year did not differ from the fourth year. The authors conclude that wildfire can have immediate
2729 and mid-term (up to 5 years) impacts on water chemistry and turbidity. Further, the magnitude
2730 and temporal increases of nitrate and turbidity, specifically, have a positive relationship with burn
2731 severity and extent.

2732 Son et al. (2015) compared stream water samples before and after an intense wildfire in the
2733 Cache la Poudre River basin in Colorado. Stream water samples for total phosphorus (TP) and
2734 total nitrogen (TN) were collected over 2 years (2010 – May 2012) before the fire in June 2012.
2735 Two post-fire water samples were taken: 1) immediately following containment of the fire (July
2736 4, 2012) and 2) twelve days after the fire was contained (July 16, 2012). For each pre- and post-
2737 fire sampling date water samples were collected at three randomly selected points at two sites.
2738 Riverbed sediments were also collected at each site and sieved through a 2 mm sieve to capture
2739 the geochemically reactive portion of the riverbed. The pre- and post-fire sediment and stream
2740 water quality were compared with t-test. Correlations of sediment and stream water quality with
2741 other factors (e.g., stream temperature, precipitation, streamflow) were evaluated with a
2742 Pearson's correlation at 0.05 and 0.1 significance levels. Results for turbidity showed no
2743 significant differences between pre- and post-fire ranges immediately following fire. However,
2744 after the first post-fire rainfall (2.5 mm) nephelometric turbidity ranged from 113.6 - 2099.4
2745 NTU (mean = 641.62 NTU), a considerable increase from pre-fire data (mean 11.3 NTU), and
2746 post-fire data before rainfall (47.3 NTU). Post-fire aqueous TP and TN loads ranged from 30.5 -
2747 56,086 and 45.4 - 1203 kg/day, respectively, and were significantly higher than pre-fire values
2748 (390 and 6 times higher than pre-fire values for TP and TN, respectively). The authors note that
2749 this is likely due to the transport and input of ash into the stream. After the first rainfall, all forms
2750 of P were significantly higher than pre-fire concentrations, such as soluble reactive phosphorus
2751 (SRP; $p = 0.000$), dissolved organic phosphorus (DOP; $p = 0.009$), and particulate phosphorus
2752 (PP; $p = 0.02$). Riverbed sediment equilibrium P concentrations increased significantly ($p =$
2753 0.007) from pre- to post-fire in all sites. The authors conclude that this study shows evidence that
2754 stream TP and TN, and riverbed sediment TP all increased significantly after the first rainfall,
2755 post-fire. They further suggest that the effects of wildfire on riverbed sorption mechanisms are
2756 very complex but further research would be valuable because fire impacted sediments highly
2757 concentrated P can become a long-term source of P.

2758 *LW*

2759 Bendix & Cowell (2010) investigated the effects of fire and flooding on LW input in two
2760 tributaries of Sespe Creek (Potrero John Creek and Piedra Blanca Creek) in the Los Padres
2761 national Forest in southern California. Both sites were located within the perimeter of the Wolf
2762 Fire that burned in June of 2002. Extensive flooding in the area occurred during January and
2763 February of 2005. The study area is characterized by chaparral dominated communities and a
2764 Mediterranean-type climate. While there is a scarcity of trees in the uplands, the riparian areas
2765 contained substantial growth of *Alnus rhombifolia* (white alder), *Populus fremontii* (Fremont

2766 cottonwood), *Quercus agrifolia* (coast live oak), *Quercus dumosa* (scrub oak) and *Salix* sp.
2767 (willows) on the valley floors. Thus, any change in in-stream or riparian area LW was sourced
2768 exclusively from the riparian area. Data for LW and standing live and dead stems in the riparian
2769 area were collected in July, of 2003 (1-year pre-fire) and again in July of 2005 (3-years post-fire,
2770 5-6 months after flood events). This data was used to answer 4 questions: 1) How many of the
2771 burned snags fell during this time, and what was the species composition?, 2) Did snags differ by
2772 species or size in the rate at which they fell?, 3) How did flooding after the fire affect the rate at
2773 which snags fell?, 4) How did flooding affect the mobilization of fallen snags? Questions 1 was
2774 analyzed by comparing descriptive data (i.e., no statistical analysis). A t-test was used to compare
2775 mean diameter of standing and fallen stems (question 2). T-tests were also used to analyze
2776 differences in mean flow depth for standing vs. fallen snags and for fallen snags still present vs.
2777 snags that had been transported after flooding (questions 4 and 5). Results showed high post-fire
2778 mortality (94%) with 339 of 362 stems killed. By 2005, 57 of the 339 snags had fallen (16.8%).
2779 The majority of fallen stems were either *Alnus* or *Salix* species. Standing snags varied in size
2780 from 3 cm to 69.2 cm, whereas those that had fallen ranged from 3 cm to 33 cm. Among the
2781 fallen snags, those <10 cm were not proportionate to the overall numbers, whereas snags between
2782 10 cm and 30 cm were disproportionately likely to fall. While fewer snags in the larger size
2783 classes the mean diameter of fallen snags was larger than the mean diameter of standing snags
2784 (11.4±10.9 cm vs. 11.0±8.0 cm) and did not differ significantly. The mean flood depth for fallen
2785 snags (1.05±0.68 m) was significantly greater than those still standing (0.40±0.56 m; $p < 0.0001$,
2786 $n=339$). The three species experiencing no snagfall at all (*Abies glauca*, *Rhamnus californica* and
2787 *Quercus agrifolia*) occurred only in higher quadrats, which had experienced virtually no
2788 flooding. Of the 57 snags that had fallen by July 2005, 43 (75%) were gone from the quadrats in
2789 which they had been recorded in 2003. The snags that had been mobilized were from quadrats
2790 that had experienced deeper flood depths (1.14±0.69 m) than those that had remained. (0.80±0.62
2791 m), but the difference is insignificant. The authors interpret these findings as an indication that
2792 short-term rates of snagfall following wildfire are influenced by the species composition of
2793 burned stems and by post-fire flood depth. Thus, although wildfire resulted in many burned snags
2794 across the valley floor, the rate at which these stems are recruited into the fluvial system as
2795 woody debris varies by the ecological characteristics and the geomorphic setting.

2796

2797 Focal Question 4

2798 4. How do various treatments within riparian buffers relate to forest health and resilience to fire,
2799 disease, and other forest disturbances?

2800 While there are several studies that discuss the frequency, dynamics, or potential for
2801 disturbances, especially fire, in riparian areas of the western United States (Dwire & Kauffman,
2802 2003; Everett et al., 2003; Merschel et al., 2014) there is a dearth of studies that investigate how
2803 treatments within the riparian area or in riparian buffers relate to the riparian area's resilience to
2804 disturbance. No studies found in our literature search and review were suitable for providing
2805 direct experimental evidence of the effects of riparian buffer treatments on riparian health and
2806 resilience to disturbance except for several studies that provide evidence that riparian harvest

2807 treatments have the potential to increase susceptibility to windthrow caused mortality. Post-
 2808 harvest changes in windthrow susceptibility are discussed in focal question 3. One study used
 2809 simulation modeling to estimate changes in health and susceptibility to disturbance with and
 2810 without treatment.

2811 Ceder et al. (2018) used Forest Vegetation Simulator (FVS) to predict how treatment along fish-
 2812 bearing streams of eastern Washington affects riparian stand health and susceptibility to insects,
 2813 disease, and crown fire. The projected changes in susceptibility were produced for the low- and
 2814 mid-elevation regulatory zones for timber harvest. Models were run for 50 years with and
 2815 without application of prescribed treatments. Prescriptions for these zones include a buffer width
 2816 of 75-130 ft depending on stream width category. For all treatments, no harvest is allowed within
 2817 the first 30 feet from the bankfull channel. Timber harvest is allowed in the remaining width of
 2818 the buffer but must meet a minimum basal area based on the regulatory zone. The authors report
 2819 high variability in the data and the outputs of each modeling scenario. However, they report that
 2820 overall, as riparian zone growth was simulated with and without management, tree size and stand
 2821 density increased, along with some increases in insect and disease susceptibility and potential
 2822 fire severity without management and decreases with management.

2823 Focal Question 5

2824 5. How do the functions provided by riparian stands change over time (e.g., large woody debris
 2825 recruitment from farther away from the stream)?

2826 This question addresses the effect of time on riparian function. This question addresses a
 2827 temporal and spatial component to changes in function. The question specifies “change over
 2828 time” but provides an example with a spatial component. While harvest is not specified as a
 2829 factor, studies that quantify changes to riparian function in harvested reaches have been included.
 2830 Studies that compare differences in one or more functions between comparable sites in different
 2831 successional stages (i.e., different mean age) are also included. Papers that investigate the
 2832 changes in LW source distance following harvest have been included because of the given
 2833 example (large woody debris recruitment from farther away from the stream).

Commented [AJK140]: I would be very careful about conflating spatial and temporal variation in this response.

Commented [AK141R140]: ADDRESS

Commented [bs142R140]: Corrected

2834 Shade

2835 Table 18. Treatment and responses for selected publications investigating Shade relevant to Q5.

Formatted: Font: Not Italic

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 canopy openness reaches its minimum value in regenerating forests at ~30 years and maintains with little variability until ~100 years. 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)

Formatted Table

Warren et al. (2013)	old-growth-forests (>500 years old) and young second-growth stands (~40-60 years old)	Three of the four paired old-growth reaches had significantly lower mean percent canopy cover
----------------------	---	--

2836

Formatted: Font: Not Italic

2837 Kaylor et al. (2017) compared canopy cover throughout stream networks adjacent to old-growth
 2838 (> 300 years old) and mid-successional (50-60 years old) Douglas-fir dominated forests in the
 2839 H.J. Andrews Experimental Forest in the Cascade Mountains of Oregon. Canopy openness was
 2840 quantified with a handheld spherical densiometer. Data was supplemented with a review of
 2841 literature studies conducted in the Pacific Northwest that reported stand age and canopy cover
 2842 over the stream. The combined datapoints for canopy openness (%) were plotted against stand
 2843 age and fit with a negative exponential curve. From the slope of the curve, the authors estimate
 2844 that canopy openness reaches its minimum value in regenerating forests at ~30 years and
 2845 maintains with little variability until ~100 years. Mean canopy openness in stands 30-100 years
 2846 old was 8.7% with a range from 1.2 to 32.0% (standard deviation = 5.7). Canopy openness over
 2847 streams in old-growth forests averaged 18.0% but was highly variable and ranged from 3.4 to
 2848 34.0% (standard deviation = 5 7.9).

2849 Warren et al. (2013) compared canopy cover between old-growth-forests (>500 years old) and
 2850 young second-growth stands (~40-60 years old) in the H.J. Andrews Experimental Forest in the
 2851 Cascade Mountains of Oregon. Canopy cover was estimated using a convex spherical
 2852 densiometer. Streams were paired based on reach length, bankfull width, and north (n =2), vs.
 2853 south (n=2) facing watersheds. Results showed significant differences in percent forest cover
 2854 between old-growth and second-growth reaches in both south-facing watersheds in mid-summer
 2855 (p < 0.10). For the north-facing watersheds, differences in canopy cover and light availability (p
 2856 < 0.10) were only significant at 1 of the two reaches. Overall, three of the four paired old-growth
 2857 reaches had significantly lower mean percent canopy cover. The authors interpret these results as
 2858 evidence that old-growth forest canopies were more complex and had more frequent gaps.

2859 *Litter*

2860 Table 19. Treatment and responses for selected publications investigating Litter relevant to Q5.

Formatted: Line spacing: Multiple 1.08 li

Reference	Treatment	Response
Kiffney & Richardson (2010)	Upland clearcuts with (1) no buffer, (2) 10 m buffer (~30 feet), (3) 30 m buffers	8 years post-harvest data Differences in litter flux relative to riparian treatment persisted through year 7, while a positive trend between buffer width and litter flux remained through year 8 . 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.

Formatted Table

<u>Bilby & Heffner (2016)</u>	<u>Litter samples released from canopy height at one old-growth site and one young forest site.</u>	<u>1-year of data actual age of stands not quantified, estimated by mean height (47.0 and 32.4 m)</u> <u>Needles released at mature sites had a higher proportion of cumulative input from greater distances than needles or alder leaves released at younger sites. The model estimated that the width of the contributing area for needles was ~35% greater at older sites than at younger sites.</u>
-----------------------------------	---	---

2861

2862 Kiffney & Richardson (2010) compared changes in litter input between riparian harvest
 2863 prescriptions that included clear-cut to stream edge, 10 m wide buffer reserve, 30 m buffer
 2864 reserves, and an uncut control over the course of 8 years. No thinning was applied within the
 2865 reserves. Upland treatment at all sites applied clearcut. Results showed differences in litter flux
 2866 relative to riparian treatment persisted through year 7, while a positive trend between reserve
 2867 width and litter flux remained through year 8. Needle inputs remained 6x higher in the buffer and
 2868 control sites through year 7, and 3-6x higher in year 8 than in the clearcut sites. Twig inputs into
 2869 the control and buffered sites were ~25x higher than in the clearcut sites in the first year after
 2870 treatment. The linear relationship between reserve width and litter inputs was strongest in the
 2871 first year after treatment, explaining ~57% of the variation, but the relationship could only
 2872 explain ~17% of the variation in litter input by buffer width by year 8 (i.e., the relationship
 2873 degraded over time). The authors interpret these results as evidence that litter flux from riparian
 2874 plants to streams, was affected by riparian reserve width and time since logging.

2875 Bilby & Heffner (2016) used linear mixed effects models developed for young and old-growth
 2876 forests of western Washington to estimate controls on litter delivery. Litter samples were released
 2877 from canopy height at one old-growth forest site and one young forest site. The mean age of
 2878 “mature” and “young” sites was not specified but the mean tree heights were 47.0 m and 32.4 m
 2879 for the mature and young sites, respectively. Results showed that needles released at mature sites
 2880 had a higher proportion of cumulative input from greater distances than needles or alder leaves
 2881 released at younger sites. The model estimated that the width of the contributing area for needles
 2882 was ~35% greater at older sites than at younger sites.

2883 *Source distance curves for LW*

2884 Table 20. Treatment and responses for selected publications investigating LW source distance
 2885 curves relevant to Q5.

<u>Reference</u>	<u>Treatment</u>	<u>Response</u>
------------------	------------------	-----------------

Formatted: Line spacing: Multiple 1.08 li

Formatted: Font: Not Italic

Formatted Table

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)	5-years post-harvest 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 (30–75 feet from the stream) was ~10% greater for the SR group compared to the AAS and REF groups.
Burton et al. (2016)	Buffer widths of 6, 15, or 70 meters	6 years post-harvest 82-85% of the wood with discernable sources came from within 15 m of the stream, and the relative contribution of wood to streams declined rapidly with increasing distance.
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 outer 10 – 20 m was more than double in the buffered units than in the reference units. Estimate that future supply of LW is diminished by ~10% in the treated sites compared to the reference sites.

2886

Formatted: Font: Not Italic

2887 Schuett-Hames & Stewart (2019a) compared differences in LW recruitment between riparian
 2888 management zones harvested under the current standard Shade Rules (SR), the All-Available
 2889 Shade Rule (AAS), and unharvested references for fish-bearing streams in the mixed conifer
 2890 habitat type (2500 - 5000 feet elevation) for eastern Washington. Both shade rules have a 30-ft
 2891 no-cut buffer (core zone) immediately adjacent to the stream. The SR prescription allows
 2892 thinning in the buffer zone 30-75 feet (inner zone) from the stream while the AAS prescription
 2893 requires retention of all shade providing trees in this area. Results showed that cumulative wood
 2894 recruitment from tree fall after the five-year post-harvest interval was highest in the SR group,
 2895 lower in the AAS group and lowest in the REF group. The SR and AAS LW recruitment rates by
 2896 volume were nearly 300% and 50% higher than the REF rates, respectively. Wood recruitment in
 2897 the SR sites was significantly greater than in the AAS and reference sites. Conversely,
 2898 differences in wood recruitment did not differ significantly between the AAS and reference sites.
 2899 Considering the source distance of post-harvest recruited LW, most recruited fallen trees
 2900 originated in the core zone (76%, 72%, and 64% for the REF, AAS and SR groups, respectively),
 2901 while the proportion from the inner zone (30–75 feet from the stream) was ~10% greater for the
 2902 SR group compared to the AAS and REF groups. These results provide evidence that the
 2903 thinning treatments applied in the inner zone of the SR treatment changed the spatial pattern
 2904 (source distance) of wood recruitment from fallen trees within 5 years post-harvest.

2905 Burton et al. (2016) examined the relationship between annual in-stream wood loading and
 2906 riparian buffer widths adjacent to upland thinning operations. Buffer widths were 6, 15, or 70
 2907 meters and upland thinning was to 200 trees per ha (tph), with a second thinning (~10 years later)
 2908 to ~85 tph, alongside an unthinned reference stand ~400 tph. Data for LW in streams were
 2909 collected for 6 years (5 years after the first harvest and 1 additional year after the second
 2910 harvest). The results showed that between 82-85% of the wood with discernable sources (90%

2911 for wood in early stages of decay; 45% of wood in late stages of decay) came from within 15 m
2912 of the stream, and the relative contribution of wood to streams declined rapidly with increasing
2913 distance.

2914 Martin & Grotenfendt (2007) compared riparian stand mortality and in-stream LW recruitment
2915 characteristics between riparian buffer strips with upland timber harvest and riparian stands of
2916 unharvested watersheds using aerial photography. All buffer strips in this study were a minimum
2917 of 20 m wide and included selective harvest within the 20 m zone (thinning intensity not
2918 specified or included in the analyses as an effect). The results showed significantly higher
2919 mortality (based on cumulative stand mortality: downed tree counts divided by standing tree
2920 counts + downed tree counts), significantly lower stand density (269 trees/ha in buffer units and
2921 328 trees/ha in reference units), and a significantly higher proportion of LW recruitment from the
2922 buffer zones of the treatment sites than in the reference sites. LW recruitment based on the
2923 proportion of stand recruited (PSR) was significantly higher in the buffered units compared to
2924 the reference units. However, PSR from the inner 0-20 m was only 17% greater in the buffer
2925 units than in the reference units; while PSR of the outer unit (10 – 20 m) was more than double
2926 in the buffered units than in the reference units. From their analysis they also estimate that future
2927 potential supply of LW is diminished by ~10% in the buffered sites compared to the reference
2928 sites.

2929 *LW and stand age*

2930 Jackson and Wohl (2015) compared in-stream wood loads between old-growth (> 200 years) and
2931 young forests (age not reported). This study took place within the Arapaho and Roosevelt
2932 National Forests in Colorado. In-stream wood loads (m³/ha) were recorded for reaches in 10 old-
2933 growth forests and 23 young forests. Paired t- test or Kruskal-Wallis tests were used to check for
2934 significant differences in wood load. Results indicated that channel wood load (OG = 304.4 +
2935 161.1; Y = 197.8 + 245.5 m³ /ha), floodplain wood load (OG = 109.4 + 80; Y = 47.1 + 52.8 m³
2936 /ha), and total wood load (OG = 154.7 + 64.1; Y = 87.8 + 100.6 m³ /ha) per 100 m length of
2937 stream and were significantly higher in streams of old-growth forests than in young forests.
2938 Streams in old-growth forests also had significantly more wood in jams, and more total wood
2939 jams per unit length of channel than in younger forests (jam wood volume: OG = 7.10 +/- 6.9
2940 m³; Y = 1.71 +/- 2.81 m³)

2941 *Nutrient dynamics over time*

2942 Vanderbilt et al. (2003) investigated long-term datasets (ranging from 20-30 years) from six
2943 watersheds in the H.J. Andrews Experimental Watershed (HJA) in the west-central Cascade
2944 Mountains of Oregon. Their objective was to characterize long-term patterns of N dynamics in
2945 precipitation and stream water at the HJA. Patterns between nitrogen with precipitation and
2946 discharge were analyzed with logistic regression. Results showed that dissolved organic nitrogen
2947 (DON) concentrations increased in the fall in every watershed. The increase in concentration
2948 began in July or August with the earliest rain events, and peak DON concentrations occurred in
2949 October through December before the peak in the hydrograph. DON concentrations then
2950 declined during the winter months. However, other forms of N showed inconsistent patterns

2951 across all other watersheds. The authors conclude that total annual stream discharge was a
2952 positive predictor of DON output suggesting a relationship to precipitation. Also, DON had a
2953 consistent seasonal concentration pattern. All other forms of N observed showed variability and
2954 inconsistencies with annual and seasonal stream discharge. The authors speculate that different
2955 factors may control organic vs. inorganic N export. Specifically, DIN may be strongly influenced
2956 by terrestrial or in-stream biotic controls, while DON is more strongly influenced by climate.
2957 Last, the authors suggest that DON in streams may be recalcitrant, and largely unavailable to
2958 stream organisms.

2959

2960 Focal Question 6

2961 6. Are there feedback mechanisms (e.g., microclimate changes within the riparian buffer) related
2962 to forest management that affect the recovery rates of riparian functions?

2963 The studies considered appropriate for answering this question are those that quantify how forest
2964 management practices impact one or more factors that can in-turn impact the rate of recovery of
2965 riparian function. The regeneration, growth and development of vegetation within the riparian
2966 area following treatment can impact the rate of recovery of litter inputs, shade, sediment and
2967 nutrient filtration. Reduction in shade may affect the amount of light reaching the forest
2968 understory that then could impact productivity in the riparian area. Also, disturbance of soil and
2969 removal of vegetation during riparian management operations can impact streamflow and
2970 sediment supply, which in turn impacts sediment flux into streams. [The studies summarized
2971 below provide experimental evidence in how these factors (e.g., vegetation productivity,
2972 streamflow discharge, sediment disturbance) are impacted by management.]

2973 However, considering the second part of this question on how these feedback mechanisms affect
2974 the recovery rates of riparian function can only be inferred. To properly answer the full question
2975 a study would require an experimental design which 1) tracks the changes in site conditions (e.g.,
2976 microclimate, light availability to groundcover, exposed soil...) after treatment relative to
2977 untreated stands, 2) evaluates how these changes in site conditions lead to changes in stand
2978 development that can then impact function (e.g., vegetation), and finally 3) how these changes in
2979 development affect the recovery rates of function. This third step would require separating out
2980 the effect of these “feedback mechanism” so that the differences in recovery rates in treated
2981 stands with and without these effects (e.g., blocking newly available light to the understory) can
2982 be compared quantitatively. No studies that specifically, and entirely address these 3 objectives
2983 collectively could be found in the literature. Thus, the following reviewed studies provide
2984 evidence of how feedback mechanisms can affect function (e.g., increased light = increased
2985 primary productivity), but how these mechanisms affect the recovery rates of any particular
2986 function (e.g., timing of recovery with and without the feedback mechanism) can only be
2987 assumed.

2988 Litter

2989 Yeung et al. (2019) simulated post-harvest responses to leaf-litter derived coarse particulate
2990 organic matter (CPOM) quantity in a coastal rainforest stream in British Columbia. This study

Commented [JK143]: Yellow: Answering this question may best be achieved through extensive monitoring and landscape assessment in areas that have experienced a time gradient of management. Like a chronosequence conducted where conditions are similar or the same.

Commented [JK144R143]: Do NOT address

Commented [AJK145]: What do these studies tell us, collectively?

Commented [AK146R145]: DO NOT ADDRESS

2991 used a CPOM model that was calibrated using data from multiple published studies from,
 2992 primarily the Pacific Northwest region, and several other North American regions. Calibration
 2993 data included stream flow and temperature, and CPOM following different timber harvest
 2994 intensities within 4 years of harvest. The model used estimated litterfall decreases of (-10%, -
 2995 30%, -50%, -90%) for low, moderate, high, and very high basal area removal ; peak streamflow
 2996 increases of +20%, +40%, +100%, +300%); and stream temperature increases of +1°C, +2°C,
 2997 +4°C, and +6 °C. Treatment intensities in litterfall, peak flow, and stream temperature were
 2998 modeled and analyzed individually and cumulatively to estimate their relative and combined
 2999 effects on in-stream CPOM standing stocks. Results of the model showed that, in general, the
 3000 standing stocks of CPOM decreased under the independent effects of reduced litterfall and
 3001 elevated peak flows and increased with higher stream temperatures.

3002 Along the gradient of increasing timber removal, litterfall reductions on depleting CPOM
 3003 standing stocks were at least an order of magnitude greater than those of elevated peak flows.
 3004 The magnitude of CPOM changes induced by litterfall reductions was consistently greater than
 3005 stream temperature increases, but their differences in magnitude became smaller at higher levels
 3006 of disturbance severity. Only the effects of litterfall-temperature interactions on CPOM standing
 3007 stocks were significant (p < 0.001). The authors interpret these results as evidence that litterfall
 3008 reduction from timber harvest was the strongest control on in-stream CPOM quantity for 4 years
 3009 post-harvest. However, the authors propose that the decreased activity of CPOM consumers
 3010 caused by increasing stream temperatures may be enough to offset the loss of litterfall inputs on
 3011 standing CPOM stocks. The caveat of this study is that it did not include LW dynamics in
 3012 preserving CPOM post-harvest. There is evidence that in-stream LW can act as a catchment for
 3013 CPOM (May & Gresswell, 2003; Richardson et al. 2007).

3014 *Sediment*

3015 [Table 21. Treatment and responses for selected publications investigating Sediment relevant to](#)
 3016 [Q5.](#)

Reference	Treatment	Response
Safaeq et al. (2020)	Long-term dataset with mixture of management, storm events, and	estimate that following harvest, changes on streamflow alone was estimated in being responsible for < 10% of the resulting suspended sediment transported into streams, while the increase in sediment supply due to harvest disturbance was responsible for >90%.
Litschert & MacDonald (2009)	Post-harvest stream sediment delivery pathway development frequency.	1-year post-harvest data (n = 200 harvest units) 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 evidence that skid trails should be directed away from streams, maintain surface roughness, and promptly decommissioned.

Formatted Table

3017
 3018 Safaeq et al. (2020) analyzed a long-term data set to changes in streamflow, and suspended
 3019 sediment load and sediment bedload in streams between two watersheds; one with a history of

3020 timber management and one with no history of timber management. The two watersheds were
3021 located in the H.J. Andrews Experimental Forest and were paired by size, aspect, and
3022 topography. The treatment watershed was 100% clearcut during the period from 1962-1966,
3023 broadcast burned in 1966, and re-seeded in 1968. Streamflow and sediment data were taken
3024 intermittently; suspended sediment data after large storm events between 1952 (pre-harvest) and
3025 1988; and sediment bedload in 2016. The researchers used a reverse regression technique to
3026 evaluate the relative and absolute importance of changes in streamflow versus changes in
3027 sediment supply from timber harvest on sediment transport. There were no significant changes in
3028 precipitation patterns before or after harvest. The results for post-treatment sediment yields
3029 showed suspended load declined to pre-treatment levels in the first two decades following
3030 treatment and bedload remained elevated, causing the bedload proportion of the total load to
3031 increase through time. Changes in streamflow alone account for 477 Mg/km² (10%) of the
3032 suspended load and 113 Mg/km² (5%) of the bedload over the post-treatment period. Increase in
3033 suspended sediment yield due to increase in sediment supply from timber harvest activities was
3034 84% of the measured post-treatment total suspended sediment yield. The authors estimate that
3035 following harvest, changes on streamflow alone was estimated in being responsible for < 10% of
3036 the resulting suspended sediment transported into streams, while the increase in sediment supply
3037 due to harvest disturbance was responsible for >90%. Thus, while timber harvest-induced
3038 increases in streamflow does increase sediment transport, it is negligible compared to the
3039 increase in sediment source created from management practices.

3040 Litschert & MacDonald (2009) investigated the frequency of sediment delivery pathways in
3041 riparian management areas and their physical characteristics and connectivity following harvest.
3042 In this study the authors describe sediment delivery pathways (“features”) as rills, gullies, and
3043 sediment plumes that form when excess sediment relative to overland flows transports sediment
3044 from the hillslope to the stream. The authors surveyed 200 riparian management areas (RMA) in
3045 four different National Forests of the Sierra Nevada and Cascade Mountains of California. USFS
3046 policy requires 90-m wide RMA along each side of perennial streams and 45-m wide RMA along
3047 each side of all ephemeral and intermittent streams. When features were found within an RMA,
3048 data for years since harvest, soil depth, soil erodibility (K), feature length, feature gradient,
3049 aspect, elevation, hillslope gradient, hillslope curvature, surface roughness, and connectivity
3050 were recorded for analysis. Association between these variables were analyzed with a
3051 Spearman’s rank correlation. The variables most strongly associated with feature length were
3052 used to develop a multiple linear regression model to predict feature length. Only 19 of the 200
3053 harvest units had sediment development pathways. Feature pathways ranged in age (time since
3054 harvest) from 2 to 18 years, and in length from 10 m to 220 m. Of the 19 feature pathways, only
3055 six were connected to streams, and five of those originated from skid trails. Feature pathway
3056 length was significantly related to mean annual precipitation, cosine of the aspect, elevation, and
3057 hillslope gradient ($R^2 = 64\%$, $p = 0.004$). The authors conclude that in general, USFS riparian
3058 forest harvest practices are effective in reducing the development of sediment delivery pathways.
3059 They also interpret these results as evidence that skid trails should be directed away from
3060 streams, maintain surface roughness, and promptly decommissioned.
3061

3062 *Impacts on Microclimate*

3063 Anderson et al. (2007) compared changes in understory microclimate above the stream, within
3064 the channel, and within the riparian area between thinned and unthinned riparian stands. The
3065 focus of this study was on second-growth (30- to 80-year-old) riparian Douglas-fir forests along
3066 headwater streams in the western Oregon Coast and Cascade Range. Stands were either thinned
3067 to approximately 198 trees per acre (TPA) or were left unthinned and ranged from 500-865 TPA.
3068 Streams within treated stands were surrounded by buffers of either 1) one site-potential tree
3069 averaging 69 m (B1, B1-T thinned and unthinned respectively), 2) variable width buffer
3070 averaging 22 m (VB, and VB-T), or 3) streamside retention buffer averaging 9 m (SR, and SR-
3071 T). Further, directly adjacent randomly selected B1-T and VB-T buffers patch openings (0.4 ha)
3072 were created (B1-P, VB-P). Microsite and microclimate responses were repeat sampled for each
3073 treatment and compared with untreated stands (UT). Within the riparian buffer zones, daily
3074 maximum temperatures were higher in all treated stands when compared to UT stands. The
3075 differences in daily maximum temperatures between treated and untreated stands ranged from
3076 1.1°C (B1) to 4.0°C (SR-T), but the difference was only significant in one SR-T stand. Daily
3077 maximum air temperature within buffer zones adjacent to patch openings were 3.5°C higher than
3078 in UT stands. Within patch openings daily maximum temperatures were on average 6 to 9°C
3079 higher than in UT stands. Soil temperature changes were only evident within patch openings
3080 ranging from 3.6 - 8.8°C higher than in UT stands. VB-T buffers that were 15 m wide or wider
3081 exhibited changes in daily maximum air temperature above stream centers <1°C and daily
3082 minimum relative humidity <5% lower than in untreated stands. The authors conclude that in
3083 general, thinned stands are warmer and drier than unthinned stands. However, the results for
3084 differences in microclimate were only significant in narrow (9 m) thinned buffers and patch
3085 openings.

3086 Anderson & Meleason (2009) conducted a companion study to Anderson et al. (2007) and
3087 compared changes in small (5-29 cm diameter) and large (≥ 30 cm diameter) downed wood
3088 abundance and understory vegetation between treated and untreated stands 5 years after harvest.
3089 Treatments compared were the same as those described in Anderson et al. (2007) discussed
3090 above. The results for small and large downed wood were highly variable between pre- and post-
3091 harvest periods and between treatments but the authors speculate from trends in the data that
3092 both wood and vegetation responses within buffers ≥ 15 m wide were insensitive to treatments.
3093 The strongest contrast in rate of change in herb cover was between the SR-T and VB-T buffers
3094 with higher herbaceous cover in the SR-T buffers and highest in SR-T buffers adjacent to patch
3095 openings. The authors conclude that in general these thinning treatments only led to subtle
3096 changes in understory vegetation cover and composition. Because of the high variability in
3097 responses among and between treatments significance could not be confirmed. The authors
3098 further conclude that a better functional understanding of the changes in ecological processes
3099 associated with changes in habitat characteristics following changes in understory wood and
3100 vegetation cover is needed to help discern ecological significance.

3101

3102 Focal Question 7

3103 7. What major data gaps and uncertainties exist relative to effects of timber harvest (both
3104 riparian and adjacent upland) on the riparian functions?

3105 No studies that provide experimental evidence that quantifies how specific treatments within the
3106 riparian area affect bank stability were found based on our search criteria (published after 2000,
3107 conducted in western North America). [However], this may be because bank erosion relates
3108 directly to sediment transport and thus bank stability is inferred by the magnitude of change in
3109 sediment export. Furthermore, the importance of vegetation retention and equipment exclusion in
3110 areas closest to the stream for maintaining bank stability appears to be well understood
3111 considering its prevalence in riparian forest management plans (WAC 222-30-022; WAC 22-30-
3112 021; 2022 ODF; IDAPA 20.02.01).

3113 Our search of the literature focused on how treatments within or adjacent to forested riparian
3114 areas impact one or more of the riparian functions. Most of the studies found in our search focus
3115 on the impacts of riparian treatment on LW and shade (commonly coupled with stream
3116 temperature). There is also a significant body of research that considers the impact of harvest on
3117 nutrient and sediment flux into streams. Fewer studies could be found that quantify changes in
3118 litter input following riparian management. ~~No studies that provide experimental evidence that~~
3119 ~~quantifies how specific treatments within the riparian area affect bank stability were found based~~
3120 ~~on our search criteria (published after 2000, conducted in western North America). [However], this~~
3121 ~~may be because bank erosion relates directly to sediment transport and thus bank stability is~~
3122 ~~inferred by the magnitude of change in sediment export. Furthermore, the importance of~~
3123 ~~vegetation retention and equipment exclusion in areas closest to the stream for maintaining bank~~
3124 ~~stability appears to be well understood considering its prevalence in riparian forest management~~
3125 ~~plans (WAC 222-30-022; WAC 22-30-021; 2022 ODF; IDAPA 20.02.01).~~

3126 While few studies could be found that provide direct experimental evidence of how bank
3127 stability is affected by timber harvest, two studies were found that compared the relative
3128 influence of different factors on bank stability. Both of which showed evidence that bank
3129 stability is influenced by the type of vegetation dominating the riparian area. Rood et al. (2015)
3130 compared the relative erosion resistance of riverbanks occupied by forests versus grassland along
3131 the Elk River in British Columbia, Canada. This study used a combination of field sampling and
3132 aerial photo analysis from 1995 to 2013 to estimate the differences in channel migration between
3133 forest and grass dominated riparian areas. Relative tree cover was binned into 5 categories
3134 ranging from (1) no trees to (5) completely treed. Relative channel change was binned into 2
3135 categories as ‘moderate change’ for channels that migrated between 45 and 75 m, and as ‘major
3136 change’ for channels that migrated more than 75 m. Chi square analysis was used to assess the
3137 distributions of vegetation of channels with moderate and major changes. Results of the chi
3138 square analysis showed that the distribution of the observed vegetation types differed
3139 significantly ($p < 0.05$) by channel change categories. Of the 15 sites assessed with moderate or
3140 major erosion (changes), 7 were along banks dominated by grasslands without trees (‘1’), four
3141 were assessed as a ‘2’, with some trees, and three were in a ‘3’ with a mixed zone of similar
3142 proportions of trees and clearing. Only one site with a ‘4’ showed a moderate amount of change.

Commented [AJK147]: I had to read this paragraph several times before I understood that you were identifying bank stability as an information gap (or uncertainty).

Each one of the responses (or narratives) for each focal question should be written in a manner so that the reader is introduced, in the first paragraph, to the general aspects of your response.

Commented [AK148R147]: ADDRESS

Commented [bs149R147]: Moved to front

Field Code Changed

Field Code Changed

Field Code Changed

Field Code Changed

Commented [AJK150]: I had to read this paragraph several times before I understood that you were identifying bank stability as an information gap (or uncertainty).

Each one of the responses (or narratives) for each focal question should be written in a manner so that the reader is introduced, in the first paragraph, to the general aspects of your response.

Commented [AK151R150]: ADDRESS

Commented [bs152R150]: Moved to front

Field Code Changed

Field Code Changed

Field Code Changed

Field Code Changed

3143 The authors interpret these results as evidence that trees are better than grass at stabilizing banks,
3144 and that stability increases with tree cover.

3145 Outside of the U.S., Krzeminska et al. (2019), investigated the effect of different types of
3146 riparian vegetation on stream bank stability in a small agricultural catchment in South-Eastern
3147 Norway. The dominating soil type within the catchment is coarse moraine in the forested areas
3148 and marine deposits with silt loam and silty clay loam texture in agriculture areas. The
3149 researchers used a combination of field collected data with stream bank stability modeling using
3150 Bank-Stability and Toe-Erosion Modeling (BSTEM). Three experimental plots were established,
3151 one for each dominant vegetation type, grass dominated, shrub dominated, and tree dominated.
3152 Investigations of in-situ undrained shear strength of the root-reinforced soil were done with a
3153 Field Inspection Vane Tester. Additionally, potential changes in the bank profile were monitored
3154 with a series of erosion pins, 6 pins per each plot. Changes in root cohesion and % cover over
3155 time for each vegetation type were estimated using the RipRoots sub-model in BSTEM. Their
3156 results showed a difference in bank stability based on vegetation type, that varied seasonally with
3157 groundwater level and stream water level. The grass dominated and tree dominated plots,
3158 specifically, showed the lowest estimated stability during spring (March to April) and early
3159 autumn (September to November), and the highest estimated stability during the summer months
3160 (May-June). This seasonal trend was also observed for the shrub plots but not as strongly.
3161 Steeper slopes in the grass and shrub dominated plots showed a trend of reduced stability for
3162 plots 54° slopes showing potential for failure. The tree dominated plots showed a trend of lower
3163 stability for steeper slopes, however, it wasn't as strong of a trend and the model did not predict
3164 potential for failure or 'instability'. Regardless of season, groundwater levels, or slope steepness
3165 the tree plots showed the highest estimated bank stability overall.

3166 These two studies that investigate bank stability use methods which could be applied to an
3167 experimental design that also considers differences in stability between treated (harvested) and
3168 untreated stands. The combination of field observation and simulation modeling used by
3169 Krzeminska et al. (2019), especially, could be used to estimate how timber harvest affects bank
3170 stability (or erosion) while also accounting for geomorphic and hydrological differences.

3171 Considering the topics included in the focal questions, studies that investigate the effects of
3172 clearcut gaps, and studies that quantify how treatment within the riparian zone relates to
3173 resilience to fire had the fewest studies providing experimental evidence. Other than the Hard
3174 Rock and Soft Rock studies, only 2 other studies (Janisch et al., 2012, Swartz et al., 2020) were
3175 found that investigate the effects of similar buffer treatment designs (patched buffers and riparian
3176 canopy gaps). For how treatments within the riparian zone relate to resilience to fire, there were
3177 no studies that provide experimental evidence on this topic based on the search criteria. Some
3178 studies were found to quantify the probability of fire or fire severity within riparian zones in
3179 general (Reeves et al. 2006; Van de Water & North, 2011). However, none compares the
3180 resilience of riparian stands between treated and untreated stands after fire. One study, Ceder et
3181 al. (2018) used simulation modeling to compare fire susceptibility between managed and
3182 unmanaged stands and has been included in focal question 4.

3183 Indeed, Stone et al. (2010) surveyed fire management officers from 55 national forests across 11
3184 western states and found that fewer than half (43%) of them indicated that they were conducting
3185 fuel reduction treatments in riparian areas. The primary objective for most of these treatments
3186 involved some form of fuel reduction (83%), while others focused on multiple objectives such as
3187 ecological restoration and habitat improvement. Most of these treatments (93%) were of small
3188 extent (< 300 acres) and occurred in the wildland urban interface (73%). The authors conclude
3189 that these results are promising, but that well-designed monitoring programs are needed to
3190 estimate the consequences of these treatments on fire risk and other ecological effects.

3191 The study from Prichard et al. (2020), discussed in question 3, used a combination of
3192 simultaneous autoregression (SAR) and random forest (RF) modeling approaches to model the
3193 drivers of fire severity and the effectiveness of fuel treatments in mitigating fire severity in the
3194 2014 Carlton Complex. Results from this study provided evidence on how vegetation (based on
3195 broad LANDFIRE classifications), topography, and different fuel treatments (e.g., thinning only,
3196 thin and pile burn, thin and broadcast burn, etc.) related to fire severity and fire spread. This
3197 approach has potential to be used in riparian areas burned by wildfires. In terms of the topic of
3198 how various treatments relate to riparian forest resistance and resilience to fire would require
3199 using a dataset of riparian forest stand characteristics that includes information on fuel
3200 treatments, time since last fire, and basin characteristics. This information could be used along
3201 with spatial information of burn severity immediately following a fire.

3202 References

- 3203 Anderson, P. D., & Meleason, M. A. (2009). Discerning responses of down wood and understory
3204 vegetation abundance to riparian buffer width and thinning treatments: an equivalence–inequivalence
3205 approach. *Canadian Journal of Forest Research*, 39(12), 2470-2485.
- 3206 Anderson, P. D., Larson, D. J., & Chan, S. S. (2007). Riparian buffer and density management influences on
3207 microclimate of young headwater forests of western Oregon. *Forest Science*, 53(2), 254-269.
- 3208 Arkle, R. S., & Pilliod, D. S. (2010). Prescribed fires as ecological surrogates for wildfires: a stream and
3209 riparian perspective. *Forest Ecology and Management*, 259(5), 893-903.
- 3210 Bahuguna, D., Mitchell, S. J., & Miquelajauregui, Y. (2010). Windthrow and recruitment of large woody
3211 debris in riparian stands. *Forest Ecology and Management*, 259(10), 2048-2055.
- 3212 Benda, L. E., Litschert, S. E., Reeves, G., & Pabst, R. (2016). Thinning and in-stream wood recruitment in
3213 riparian second growth forests in coastal Oregon and the use of buffers and tree tipping as
3214 mitigation. *Journal of forestry research*, 27(4), 821-836.
- 3215 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., &
3216 Dunne, T. (2003, January). Wood recruitment processes and wood budgeting. In *American Fisheries
3217 Society Symposium* (pp. 49-74). American Fisheries Society.
- 3218 Beschta, R. L., Bilby, R.E.Brown, G.W., Holtby, L.B., and Hofstra., T.D., 1987. Stream Temperature and
3219 Aquatic Habitat: Fisheries and Forestry Interactions. In: *Streamside Management: Forestry and Fisheries
3220 Interactions*. E. O. Salo and T. W. Cundy (Editors). Contribution No. 57, University of Washington, Institute
3221 of Forest Resources, 471 pp.

Commented [WB153]: This would be good to include in this review:

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

Commented [WB154R153]: Address, but no need to be detailed. Maybe just a brief summary using section 1.1 in the annotated biblio, then maybe introduce it somewhere in background.

Commented [bs155R153]: Included

- 3222 Bilby, R. E., & Heffner, J. T. (2016). Factors influencing litter delivery to streams. *Forest Ecology and*
3223 *Management*, 369, 29–37. <https://doi.org/10.1016/j.foreco.2016.03.031>
- 3224 Bilby, R. E., Sullivan, K., & Duncan, S. H. (1989). The generation and fate of road-surface sediment in
3225 forested watersheds in southwestern Washington. *Forest Science*, 35(2), 453-468.
- 3226 Bjornn, T. C., & Reiser, D. W. (1991). Habitat requirements of salmonids in streams. *American Fisheries*
3227 *Society Special Publication*, 19(837), 138.
- 3228 Bladon, K. D., Cook, N. A., Light, J. T., & Segura, C. (2016). A catchment-scale assessment of stream
3229 temperature response to contemporary forest harvesting in the Oregon Coast Range. *Forest Ecology and*
3230 *Management*, 379, 153-164.
- 3231 Bladon, K. D., Segura, C., Cook, N. A., Bywater-Reyes, S., & Reiter, M. (2018). A multicatchment analysis of
3232 headwater and downstream temperature effects from contemporary forest harvesting. *Hydrological*
3233 *Processes*, 32(2), 293-304.
- 3234 Brown, G. W. (1983). *Forestry and water quality. Forestry and water quality., (Ed. 2).*
- 3235 Brown, G. W., Gahler, A. R., & Marston, R. B. (1973). Nutrient losses after clear-cut logging and slash
3236 burning in the Oregon Coast Range. *Water Resources Research*, 9(5), 1450-1453.
- 3237 Burton, J. I., Olson, D. H., & Puettmann, K. J. (2016). Effects of riparian buffer width on wood loading in
3238 headwater streams after repeated forest thinning. *Forest Ecology and Management*, 372, 247–257.
3239 <https://doi.org/10.1016/j.foreco.2016.03.053>
- 3240 Bywater-Reyes, S., Bladon, K. D., & Segura, C. (2018). Relative influence of landscape variables and
3241 discharge on suspended sediment yields in temperate mountain catchments. *Water Resources*
3242 *Research*, 54(7), 5126-5142.
- 3243 Bywater-Reyes, S., Segura, C., & Bladon, K. D. (2017). Geology and geomorphology control suspended
3244 sediment yield and modulate increases following timber harvest in temperate headwater
3245 streams. *Journal of Hydrology*, 548, 754-769.
- 3246 Camp, A., C. Oliver, P. Hessburg, and R. Everett. (1997). Predicting late-successional fire refugia pre-dating
3247 European settlement in the Wenatchee Mountains. *Forest Ecology and Management* 95 63-77
- 3248 Chan, S., P. Anderson, J. Cissel, L. Lateen. and C. Thompson. 2004. Variable density management in
3249 Riparian Reserves: lessons learned from an operational study in managed forests of western Oregon,
3250 USA. *For. Snow Landsc. Res.* 78,1/2:151-172.
- 3251 Chapman, D. W., & Bjornn, T. C. (1969). Distribution of salmonids in streams. In *Symp. Salmon Trout*
3252 *Streams*. Institute of Fisheries, University of British Columbia, Vancouver (pp. 153-176).
- 3253 Chen, X., Wei, X., & Scherer, R. (2005). Influence of wildfire and harvest on biomass, carbon pool, and
3254 decomposition of large woody debris in forested streams of southern interior British Columbia. *Forest*
3255 *Ecology and Management*, 208(1-3), 101-114.
- 3256 Chen, X., Wei, X., Scherer, R., Luider, C., & Darlington, W. (2006). A watershed scale assessment of in-
3257 stream large woody debris patterns in the southern interior of British Columbia. *Forest Ecology and*
3258 *Management*, 229(1-3), 50-62.

3259 Chesney, C. (2000). Functions of wood in small, steep streams in eastern Washington: Summary of
3260 results for project activity in the Ahtanum, Cowiche, and Tieton basins. *Timber, Fish, Wildlife*.

3261 CMER 03-308 (2004) Review of the Available Literature Related to Wood Loading Dynamics in and
3262 around Stream in eastern Washington Forests

3263 Cole, E., & Newton, M. (2013). Influence of streamside buffers on stream temperature response
3264 following clear-cut harvesting in western Oregon. *Canadian journal of forest research*, 43(11), 993-1005.

3265 Cooper, J. R., Gilliam, J. W., Daniels, R. B., & Robarge, W. P. (1987). Riparian areas as filters for agricultural
3266 sediment. *Soil science society of America journal*, 51(2), 416-420.

3267 Crandall, T., Jones, E., Greenhalgh, M., Frei, R. J., Griffin, N., Severe, E., ... & Abbott, B. W. (2021).
3268 Megafire affects stream sediment flux and dissolved organic matter reactivity, but land use dominates
3269 nutrient dynamics in semiarid watersheds. *PLoS one*, 16(9), e0257733.

3270 Cupp, C.E. and T.J. Lofgren. 2014. Effectiveness of riparian management zone prescriptions in protecting
3271 and maintaining shade and water temperature in forested streams of Eastern Washington. Cooperative
3272 Monitoring Evaluation and Research Report CMER 02-212. Washington State Forest Practices Adaptive
3273 Management Program. Washington Department of Natural Resources, Olympia, WA.

3274 Deval, C., Brooks, E. S., Gravelle, J. A., Link, T. E., Dobre, M., & Elliot, W. J. (2021). Long-term response in
3275 nutrient load from commercial forest management operations in a mountainous watershed. *Forest
3276 Ecology and Management*, 494, 119312.

3277 Devotta, D. A., Fraterrigo, J. M., Walsh, P. B., Lowe, S., Sewell, D. K., Schindler, D. E., & Hu, F. S. (2021).
3278 Watershed Alnus cover alters N: P stoichiometry and intensifies P limitation in subarctic
3279 streams. *Biogeochemistry*, 153(2), 155-176.

3280 Dwire, K. A., & Kauffman, J. B. (2003). Fire and riparian ecosystems in landscapes of the western USA.
3281 *Forest Ecology and Management*, 178(1-2), 61-74.

3282 Ebersole, J. L., Liss, W. J., & Frissell, C. A. (2001). Relationship between stream temperature, thermal
3283 refugia and rainbow trout *Oncorhynchus mykiss* abundance in arid-land streams in the northwestern
3284 United States. *Ecology of freshwater fish*, 10(1), 1-10.

3285 Ehinger, W.J., W.D. Bretherton, S.M. Estrella, G. Stewart, D.E. Schuett-Hames, and S.A. Nelson. 2021.
3286 Effectiveness of Forest Practices Buffer Prescriptions on Perennial Non-fish-bearing Streams on Marine
3287 Sedimentary Lithologies in Western Washington. Cooperative Monitoring, Evaluation, and Research
3288 Committee Report CMER 2021.08.24, Washington State Forest Practices Adaptive Management
3289 Program, Washington Department of Natural Resources, Olympia, WA.

3290 Everett, R., Schellhaas, R., Ohlson, P., Spurbeck, D., & Keenum, D. (2003). Continuity in fire disturbance
3291 between riparian and adjacent sideslope Douglas-fir forests. *Forest Ecology and Management*, 175(1-3),
3292 31-47.

3293 Fox, M. J. (2001). A new look at the quantities and volumes of instream wood in forested basins within
3294 Washington State (Doctoral dissertation, University of Washington).

3295 Fox, M., & Bolton, S. (2007). A regional and geomorphic reference for quantities and volumes of instream
3296 wood in unmanaged forested basins of Washington State. *North American Journal of Fisheries*
3297 *Management*, 27(1), 342-359.

3298 Fox, M., Bolton, S., & Conquest, L. (2003). 14Reference conditions for instream wood in western
3299 Washington. University of Washington Press: Seattle, WA, 361-393.

3300 Fratkin, M. M., Segura, C., & Bywater-Reyes, S. (2020). The influence of lithology on channel geometry
3301 and bed sediment organization in mountainous hillslope-coupled streams. *Earth Surface Processes and*
3302 *Landforms*, 45(10), 2365-2379.

3303 Fredriksen, R. L. (1975). Nitrogen, phosphorus and particulate matter budgets of five coniferous forest
3304 ecosystems in the western Cascades Range, Oregon.

3305 Gomi, T., Dan Moore, R., & Hassan, M. A. (2005). Suspended sediment dynamics in small forest streams
3306 of the Pacific Northwest 1. *JAWRA Journal of the American Water Resources Association*, 41(4), 877-898.

3307 Gomi, T., Sidle, R. C., Bryant, M. D., & Woodsmith, R. D. (2001). The characteristics of woody debris and
3308 sediment distribution in headwater streams, southeastern Alaska. *Canadian Journal of Forest*
3309 *Research*, 31(8), 1386-1399.

3310 Gravelle, J. A., & Link, T. E. (2007). Influence of timber harvesting on headwater peak stream
3311 temperatures in a northern Idaho watershed. *Forest Science*, 53(2), 189-205.

3312 Gravelle, J. A., Ice, G., Link, T. E., & Cook, D. L. (2009). Nutrient concentration dynamics in an inland
3313 Pacific Northwest watershed before and after timber harvest. *Forest Ecology and Management*, 257(8),
3314 1663-1675.

3315 Gregory, S. (1990). *Riparian management guide: Willamette National Forest*. US Department of
3316 Agriculture, Forest Service, Pacific Northwest Region.

3317 Gresswell, R. E., Barton, B. A., & Kershner, J. L. (Eds.). (1989). *Practical approaches to riparian resource*
3318 *management: an educational workshop: May 8-11, 1989, Billings, Montana*. US Bureau of Land
3319 Management.

3320 Groom, J. D., Dent, L., Madsen, L. J., & Fleuret, J. (2011b). Response of western Oregon (USA) stream
3321 temperatures to contemporary forest management. *Forest Ecology and Management*, 262(8), 1618-
3322 1629.

3323 Groom, J. D., L. Dent, and L. J. Madsen. (2011a). Stream temperature change detection for state and
3324 private forests in the Oregon Coast Range, *Water Resour. Res.* 47, W01501

3325 Guenther, S. M., Gomi, T., & Moore, R. D. (2014). Stream and bed temperature variability in a coastal
3326 headwater catchment: influences of surface-subsurface interactions and partial-retention forest
3327 harvesting. *Hydrological Processes*, 28(3), 1238-1249.

3328 Harmon, M. E., Franklin, J. F., Swanson, F. J., Sollins, P., Gregory, S. V., Lattin, J. D., ... & Cummins, K. W.
3329 (1986). Ecology of coarse woody debris in temperate ecosystems. *Advances in ecological research*, 15,
3330 133-302.

- 3331 Hart, Stephanie K., David E. Hibbs, and Steven S. Perakis. (2013) "Riparian litter inputs to streams in the
3332 central Oregon Coast Range." *Freshwater Science* 32.1 (2013): 343-358.
- 3333 Hartman, G. F., & Scrivener, J. C. (1990). Impacts of forestry practices on a coastal stream ecosystem,
3334 Carnation Creek, British Columbia.
- 3335 Hassan, M. A., Church, M., Lisle, T. E., Brardinoni, F., Benda, L., & Grant, G. E. (2005). Sediment transport
3336 and channel morphology of small, forested streams 1. *Jawra journal of the american water resources*
3337 *association*, 41(4), 853-876.
- 3338 Hatten, J. A., Segura, C., Bladon, K. D., Hale, V. C., Ice, G. G., & Stednick, J. D. (2018). Effects of
3339 contemporary forest harvesting on suspended sediment in the Oregon Coast Range: Alsea Watershed
3340 Study Revisited. *Forest Ecology and Management*, 408, 238-248.
- 3341 Hoffmann, C. C., Kjaergaard, C., Uusi-Kämppe, J., Hansen, H. C. B., & Kronvang, B. (2009). Phosphorus
3342 Retention in Riparian Buffers: Review of Their Efficiency. *Journal of Environmental Quality*, 38(5), 1942–
3343 1955. <https://doi.org/10.2134/jeq2008.0087>
- 3344 Hough-Snee, N., Kasprak, A., Rossi, R. K., Bouwes, N., Roper, B. B., & Wheaton, J. M. (2016).
3345 Hydrogeomorphic and Biotic Drivers of Instream Wood Differ Across Sub-basins of the Columbia River
3346 Basin, USA. *River Research and Applications*, 32(6), 1302–1315. <https://doi.org/10.1002/rra.2968>
- 3347 Hunter, M. A., & Quinn, T. (2009). Summer water temperatures in alluvial and bedrock channels of the
3348 Olympic Peninsula. *Western Journal of Applied Forestry*, 24(2), 103-108.
- 3349 Hyatt, T. L., & Naiman, R. J. (2001). The residence time of large woody debris in the Queets River,
3350 Washington, USA. *Ecological Applications*, 11(1), 191-202.
- 3351 Jackson, C. R., C. A. Sturm, and J. M. Ward. 2001. Timber harvest impacts on small headwater stream
3352 channels in the coast ranges of Washington. *JAWRA Journal of the American Water Resources*
3353 *Association* 37(6):1533-1549.
- 3354 Jackson, K. J., & Wohl, E. (2015). Instream wood loads in montane forest streams of the Colorado Front
3355 Range, USA. *Geomorphology*, 234, 161-170.
- 3356 Janisch, J. E., Wondzell, S. M., & Ehinger, W. J. (2012). Headwater stream temperature: Interpreting
3357 response after logging, with and without riparian buffers, Washington, USA. *Forest Ecology and*
3358 *Management*, 270, 302-313.
- 3359 Johnson, S. L., & Jones, J. A. (2000). Stream temperature responses to forest harvest and debris flows in
3360 western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences*, 57(S2), 30-39.
- 3361 Karwan, D. L., Gravelle, J. A., & Hubbard, J. A. (2007). Effects of timber harvest on suspended sediment
3362 loads in Mica Creek, Idaho. *Forest Science*, 53(2), 181-188.
- 3363 Kaylor, M. J., Warren, D. R., & Kiffney, P. M. (2017). Long-term effects of riparian forest harvest on light in
3364 Pacific Northwest (USA) streams. *Freshwater Science*, 36(1), 1-13.
- 3365 Kibler, K. M., Skaugset, A., Ganio, L. M., & Huso, M. M. (2013). Effect of contemporary forest harvesting
3366 practices on headwater stream temperatures: Initial response of the Hinkle Creek catchment, Pacific
3367 Northwest, USA. *Forest ecology and management*, 310, 680-691.

3368 Kiffney, P., and J. Richardson. 2010. Organic matter inputs into headwater streams of southwestern
3369 British Columbia as a function of riparian reserves and time since harvesting. *Forest Ecology and*
3370 *Management*. 260:1931-1942.

3371 Knight, S.M. (1990). Forest harvesting impacts on coarse woody debris and channel form in central
3372 Oregon streams. M.S. Thesis. Oregon State University, Corvallis, Oregon.

3373 Lewis, D. D. (1998, April). Naive (Bayes) at forty: The independence assumption in information retrieval.
3374 In *European conference on machine learning* (pp. 4-15). Berlin, Heidelberg: Springer Berlin Heidelberg.

3375 Liquori, M. K. (2006). POST-HARVEST RIPARIAN BUFFER RESPONSE: IMPLICATIONS FOR WOOD
3376 RECRUITMENT MODELING AND BUFFER DESIGN 1. *JAWRA Journal of the American Water Resources*
3377 *Association*, 42(1), 177-189.

3378 Litschert, S. E., & MacDonald, L. H. (2009). Frequency and characteristics of sediment delivery pathways
3379 from forest harvest units to streams. *Forest Ecology and Management*, 259(2), 143-150.

3380 Macdonald, J. S., Beaudry, P. G., MacIsaac, E. A., & Herunter, H. E. (2003). The effects of forest harvesting
3381 and best management practices on streamflow and suspended sediment concentrations during
3382 snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada. *Canadian Journal of*
3383 *Forest Research*, 33(8), 1397-1407.

3384 Macdonald, J. S., MacIsaac, E. A., & Herunter, H. E. (2003b). The effect of variable-retention riparian
3385 buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of
3386 British Columbia. *Canadian journal of forest research*, 33(8), 1371-1382.

3387 Martin, D. J., & Grotefendt, R. A. (2007). Stand mortality in buffer strips and the supply of woody debris
3388 to streams in Southeast Alaska. *Canadian Journal of Forest Research*, 37(1), 36-49.

3389 Martin, D. J., Kroll, A. J., & Knoth, J. L. (2021). An evidence-based review of the effectiveness of riparian
3390 buffers to maintain stream temperature and stream-associated amphibian populations in the Pacific
3391 Northwest of Canada and the United States. *Forest Ecology and Management*, 491, 119190.

3392 May, C. L., & Gresswell, R. E. (2003). Large wood recruitment and redistribution in headwater streams in
3393 the southern Oregon Coast Range, USA. *Canadian Journal of Forest Research*, 33(8), 1352-1362.

3394 McDade, M, F. Swanson, W. McKee, J. Franklin and J. Van Sickle. (1990). Source distances for coarse
3395 woody debris entering small streams in western Oregon and Washington. *Canadian Journal of Forest*
3396 *Resources* 20, 326-330

3397 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn (technical
3398 coordinators). 2018. Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing
3399 Streams on Competent Lithologies in Western Washington. Cooperative Monitoring, Evaluation and
3400 Research Report CMER 18-100, Washington State Forest Practices Adaptive Management Program,
3401 Washington Department of Natural Resources, Olympia, WA.

3402 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D.E. Schuett-Hames, R. Ojala-Barbour, G. Stewart
3403 and T. Quinn (technical coordinators). 2021. Effectiveness of experimental riparian buffers on perennial
3404 non-fish-bearing streams on competent lithologies in western Washington – Phase 2 (9 years after
3405 harvest). Cooperative Monitoring, Evaluation and Research Report CMER 2021.07.27, Washington State

3406 Forest Practices Adaptive Management Program, Washington Department of Natural Resources,
3407 Olympia, WA.

3408 Meleason, M. A., Gregory, S. V., & Bolte, J. P. (2003). Implications of riparian management strategies on
3409 wood in streams of the Pacific Northwest. *Ecological Applications*, 13(5), 1212-1221.

3410 Merschel, A. G., Spies, T. A., & Heyerdahl, E. K. (2014). Mixed-conifer forests of central Oregon: effects of
3411 logging and fire exclusion vary with environment. *Ecological Applications*, 24(7), 1670-1688.

3412 Mueller, E. R., & Pitlick, J. (2013). Sediment supply and channel morphology in mountain river systems: 1.
3413 Relative importance of lithology, topography, and climate. *Journal of Geophysical Research: Earth*
3414 *Surface*, 118(4), 2325-2342.

3415 Murray, G. L. D., Edmonds, R. L., & Marra, J. L. (2000). Influence of partial harvesting on stream
3416 temperatures, chemistry, and turbidity in forests on the western Olympic Peninsula,
3417 Washington. *Northwest science.*, 74(2), 151-164.

3418 Naiman, R. J., Fetherston, K. L., McKay, S. J., & Chen, J. (1998). Riparian forests. *River ecology and*
3419 *management: lessons from the Pacific Coastal Ecoregion*, 289-323.

3420 Nowakowski, A. L., & Wohl, E. (2008). Influences on wood load in mountain streams of the Bighorn
3421 National Forest, Wyoming, USA. *Environmental Management*, 42(4), 557-571.

3422 Polyakov, V., Fares, A., & Ryder, M. H. (2005). Precision riparian buffers for the control of nonpoint source
3423 pollutant loading into surface water: A review. *Environmental Reviews*, 13(3), 129-144.

3424 Puntenney-Desmond, K. C., Bladon, K. D., & Silins, U. (2020). Runoff and sediment production from
3425 harvested hillslopes and the riparian area during high intensity rainfall events. *Journal of Hydrology*, 582,
3426 124452.

3427 [Quinn, T., G.F. Wilhere, and K.L. Krueger, technical editors. \(2020\). Riparian Ecosystems, Volume 1:
3428 Science Synthesis and Management Implications. Habitat Program, Washington Department of Fish and
3429 Wildlife, Olympia.](#)

3430 Rachels, A. A., Bladon, K. D., Bywater-Reyes, S., & Hatten, J. A. (2020). Quantifying effects of forest
3431 harvesting on sources of suspended sediment to an Oregon Coast Range headwater stream. *Forest*
3432 *Ecology and Management*, 466, 118123.

3433 Reid, D. A., & Hassan, M. A. (2020). Response of in-stream wood to riparian timber harvesting: Field
3434 observations and long-term projections. *Water Resources Research*, 56(8), e2020WR027077.

3435 Reiter, M., Bilby, R. E., Beech, S., & Heffner, J. (2015). Stream temperature patterns over 35 years in a
3436 managed forest of western Washington. *JAWRA Journal of the American Water Resources*
3437 *Association*, 51(5), 1418-1435.

3438 Reiter, M., Heffner, J. T., Beech, S., Turner, T., & Bilby, R. E. (2009). Temporal and Spatial Turbidity Patterns
3439 Over 30 Years in a Managed Forest of Western Washington 1. *JAWRA Journal of the American Water*
3440 *Resources Association*, 45(3), 793-808.

3441 Reiter, M., Johnson, S. L., Homyack, J., Jones, J. E., & James, P. L. (2020). Summer stream temperature
3442 changes following forest harvest in the headwaters of the Trask River watershed, Oregon Coast
3443 Range. *Ecohydrology*, 13(3), e2178.

3444 [Roni, P., Beechie, T., Pess, G., & Hanson, K. \(2015\). Wood placement in river restoration: fact, fiction, and
3445 future direction. *Canadian Journal of Fisheries and Aquatic Sciences*, 72\(3\), 466–478.
3446 <https://doi.org/10.1139/cifas-2014-0344>](#)

3447 Roon, D. A., Dunham, J. B., & Groom, J. D. (2021a). Shade, light, and stream temperature responses to
3448 riparian thinning in second-growth redwood forests of northern California. *PLoS One*, 16(2), e0246822–
3449 e0246822. <https://doi.org/10.1371/journal.pone.0246822>

3450 Roon, D. A., Dunham, J. B., & Torgersen, C. E. (2021b). A riverscape approach reveals downstream
3451 propagation of stream thermal responses to riparian thinning at multiple scales. *Ecosphere*, 12(10),
3452 e03775.

3453 Safeeq, M., Grant, G. E., Lewis, S. L., & Hayes, S. K. (2020). Disentangling effects of forest harvest on long-
3454 term hydrologic and sediment dynamics, western Cascades, Oregon. *Journal of Hydrology*, 580, 124259.

3455 Salo, E. O., & Cundy, T. W. (1986, February). Streamside management: forestry and fishery interactions.
3456 In *Proceedings of a conference sponsored by the College of Forest Resources, University of Washington*
3457 *and others, and held at the University of Washington*.

3458 Schuett-Hames, D., & Stewart, G. (2019a). Post-Harvest Change in Stand Structure, Tree Mortality and
3459 Tree Fall in Eastern Washington Riparian Buffers. Cooperative Monitoring Evaluation and Research
3460 Report. Washington State Forest Practices Adaptive Management Program. Washington Department of
3461 Natural Resources, Olympia, WA.

3462 Schuett-Hames, D. & Stewart, G. (2019b). Changes in stand structure, buffer tree mortality and riparian-
3463 associated functions 10 years after timber harvest adjacent to non-fish-bearing perennial streams in
3464 western Washington. Cooperative Monitoring Evaluation and Research Report. Washington State Forest
3465 Practices Adaptive Management Program. Washington Department of Natural Resources, Olympia, WA.

3466 Schuett-Hames, D., Martin, D., Mendoza, C., Flitcroft, R., & Haemmerle, H., (2015). Westside Type F
3467 Riparian Prescription Monitoring Project Technical Writing and Implementation Group (TWIG).
3468 Cooperative Monitoring Evaluation and Research Report. Washington State Forest Practices Adaptive
3469 Management Program. Washington Department of Natural Resources, Olympia, WA.

3470 Schuett-Hames, D., Roorbach, A., & Conrad, R. (2011). Results of the Westside Type N Buffer
3471 Characteristics, Integrity and Function Study Final Report. Cooperative Monitoring Evaluation and
3472 Research Report, CMER 12-1201, Washington Department of Natural Resources, Olympia, WA.

3473 Shah, N. W., & Nisbet, T. R. (2019a). The effects of forest clearance for peatland restoration on water
3474 quality. *Science of the Total Environment*, 693, 133617.

3475 Sievers, M., Hale, R., & Morrongiello, J. R. (2017). Do trout respond to riparian change? A meta-analysis
3476 with implications for restoration and management. *Freshwater Biology*, 62(3), 445–457.
3477 <https://doi.org/10.1111/fwb.12888>

3478 Six, L. J., Bilby, R. E., Reiter, M., James, P., & Villarin, L. (2022). Effects of current forest practices on
3479 organic matter dynamics in headwater streams at the Trask river watershed, Oregon. *Trees, Forests and*
3480 *People*, 8, 100233.

3481 Sobota, D. J., Gregory, S. V., & Sickler, J. V. (2006). Riparian tree fall directionality and modeling large
3482 wood recruitment to streams. *Canadian Journal of Forest Research*, 36(5), 1243–1254.
3483 <https://doi.org/10.1139/x06-022>

3484 Sugden, B. D., Steiner, R., & Jones, J. E. (2019). Streamside management zone effectiveness for water
3485 temperature control in Western Montana. *International Journal of Forest Engineering*, 30(2), 87-98.

3486 Sullivan, P. F., Neale, M. C., & Kendler, K. S. (2000). Genetic epidemiology of major depression: review
3487 and meta-analysis. *American journal of psychiatry*, 157(10), 1552-1562.

3488 Swanson, F. J., & Dyrness, C. T. (1975). Impact of clear-cutting and road construction on soil erosion by
3489 landslides in the western Cascade Range, Oregon. *Geology*, 3(7), 393-396.

3490 Swanson, F. J., Gregory, S. V., Sedell, J. R., & Campbell, A. G. (1982). Land-water interactions: the riparian
3491 zone.

3492 Swartz, A., Roon, D., Reiter, M., & Warren, D. (2020). Stream temperature responses to experimental
3493 riparian canopy gaps along forested headwaters in western Oregon. *Forest Ecology and*
3494 *Management*, 474, 118354.

3495 Sweeney, B. W., & Newbold, J. D. (2014). Streamside forest buffer width needed to protect stream water
3496 quality, habitat, and organisms: a literature review. *JAWRA Journal of the American Water Resources*
3497 *Association*, 50(3), 560-584.

3498 Teply, M., McGreer, D., & Ceder, K. (2014). Using simulation models to develop riparian buffer strip
3499 prescriptions. *Journal of Forestry*, 112(3), 302-311.

3500 Teply, M., McGreer, D., Schult, D., & Seymour, P. (2007). Simulating the effects of forest management on
3501 large woody debris in streams in northern Idaho. *Western Journal of Applied Forestry*, 22(2), 81–87.
3502 <https://doi.org/10.1093/wjaf/22.2.81>

3503 Vanderbilt, K. L., Lajtha, K., & Swanson, F. J. (2003). Biogeochemistry of unpolluted forested watersheds
3504 in the Oregon Cascades: temporal patterns of precipitation and stream nitrogen
3505 fluxes. *Biogeochemistry*, 62(1), 87-117.

3506 Warren, D. R., Keeton, W. S., Bechtold, H. A., & Rosi-Marshall, E. J. (2013). Comparing streambed light
3507 availability and canopy cover in streams with old-growth versus early-mature riparian forests in western
3508 Oregon. *Aquatic sciences*, 75(4), 547-558.

3509 Wing, M. G., & Skaugset, A. (2002). Relationships of channel characteristics, land ownership, and land
3510 use patterns to large woody debris in western Oregon streams. *Canadian Journal of Fisheries and*
3511 *Aquatic Sciences*, 59(5), 796-807.

3512 Wise, E. K. (2010). Tree ring record of streamflow and drought in the upper Snake River. *Water Resources*
3513 *Research*, 46(11).

3514 Wissmar, R.C., Beer, W.N. & Timm, R.K. Spatially explicit estimates of erosion-risk indices and variable
3515 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)
3516 004-0714-9

3517 Yang, Y., Hart, S. C., McCorkle, E. P., Stacy, E. M., Barnes, M. E., Hunsaker, C. T., ... & Berhe, A. A. (2021).
3518 Stream water chemistry in mixed-conifer headwater basins: role of water sources, seasonality,
3519 watershed characteristics, and disturbances. *Ecosystems*, 24(8), 1853-1874.

3520 Yeung, A. C., Stenroth, K., & Richardson, J. S. (2019). Modelling biophysical controls on stream organic
3521 matter standing stocks under a range of forest harvesting impacts. *Limnologica*, 78, 125714.

3522 Zhang, X., & Shu, C. W. (2010). On positivity-preserving high order discontinuous Galerkin schemes for
3523 compressible Euler equations on rectangular meshes. *Journal of Computational Physics*, 229(23), 8918-
3524 8934.

3525

3526

3527

3528

3529

3530

3531

3532

3533

3534

3535

3536

3537

3538

3539

3540

3541

3542

3543

3544

3545

3546
3547

Appendix I

3548 Table A-1.2. List of treatments, variables, metrics, and results from publications reviewed for information on litter, organic matter, and
3549 nutrient inputs.

Reference	Treatment	Variables	Metrics	Notes	Results
<u>Anderson et al., 2007</u>	<u>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.</u>	<u>Microsite, microclimate, stand structure, canopy cover</u>	<u>Microsite and microclimate data (humidity, temperature sensors). Stand basal area. Canopy cover was estimated through photographic techniques.</u>	<u>Many of the reported differences in temperature and humidity were considerable but not significant. Results for changes in upland areas not reported here.</u>	<u>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.</u>
<u>Bilby & Heffner, 2016</u>	<u>Various wind speeds for young and old-growth conifer and deciduous forests. Distance of litter delivery.</u>	<u>Litter input</u>	<u>Models were developed with site characteristics and litter release experiments from sites along Humphrey Creek in the cascade mountains of western Washington.</u>	<u>Wind speeds, direction, and litter release data were collected for only one year in one area of western Washington.</u>	<u>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.</u>
<u>Deval et al., 2021</u>	<u>clearcut to stream, 50% shade retention, with site management operations including pile burning and competition release herbicide application.</u>	<u>Changes in nitrogen and phosphorus compounds.</u>	<u>monthly grab samples from multiple flume sites pre- and post-harvest, laboratory chemical analysis</u>	<u>Data was compared from pre-harvest to post experimental harvest (PH-I), and post operational harvest (PH-II)</u>	<u>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</u>

Formatted: Width: 11", Height: 8.5"

Commented [JK156]: YELLOW: This table is helpful. I find that I still want to see a table that puts the data (results) from each of these papers together in one story - what does it all mean when taken together. How does the empirical data compare to modeled and hypothesized results? This comment applies to all the summaries..

Commented [JK157R156]: Address: Suggest a tabulation of data from reviewed studies.

Commented [bs158R156]: Tables tabulating treatment, response and type of study has been added to the questions section. These tables have been moved to an appendix.

					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 NO3 and NO2 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 NO3 and NO2 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⁻¹.
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/m² (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 woody debris 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	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).	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 m² ha⁻¹.	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.

3550

3551 [Table A-23. List of treatments, variables, metrics, and results from publications reviewed for information on large wood \(LW\), wood loads, and wood](#)
 3552 [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.

Commented [AJK159]: This table should be placed in an appendix.

Also, I would reconsider how much information is placed in the table...as it stands, it is less a summary table than massive blocks of text with lines around them.

Commented [AK160R159]: ADDRESS

Commented [bs161R159]: Moved to Appendix. Smaller tables outlining treatment and impact have been added to the Question sections.

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 ³), significantly higher than stream size I (0.06 m ³). LW density (pieces per 100 m ² of stream area), however, decreased as stream size increased. For example, LW density (defined as piece numbers per 100 m ²) numbers were 19, 17, 12, and 4 for stream size I, II, III, and IV respectively. Increases in channel bank full width (R ² = 0.52) and stream area (R ² = 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 ³ 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 ³ /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%).
Fox & Bolton, 2007	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 km², 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 (m³)" (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 m³ for the 20- to 30-m BFW class, 10.5 m³ for the 30- to 50-m BFW class, and 10.7 m³ for channels greater than 50 m BFW per 100 m length of stream.
Gomi et al., 2001	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.

Hough-Snee et al., 2016	In-stream wood volume and frequency were quantified across multiple sub basins.	LW frequency and volume, hydrologic and geomorphic attributes	Models were calibrated with site characteristics from multiple riparian stands in the Columbia River Basin.	Results show a high level of variability between sub basins studied. The overall model (watershed) was an important predictor.	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.
Hyatt & Naiman, 2001	LW data was collected from multiple sites in the Queets River Watershed.	LW in stream and in riparian forests.	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.	The depletion constant was developed for a large, mostly alluvial river and should probably not be applied to smaller streams	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.
Jackson & Wohl, 2015	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.	Sediment storage, channel geometry, in-stream wood load, and forest stand characteristics	Wood loads, wood jam volumes, log jam frequencies, residual pool volume, and fine sediment storage around wood, stand age, and	Old growth defined as forests > 200 years. Age range of young forests not reported. Sample sizes include 10 old-growth and	Results indicated that channel wood load (OG = 304.4 + 161.1; Y = 197.8 + 245.5 m³ /ha), floodplain wood load (OG = 109.4 + 80; Y = 47.1 + 52.8 m³ /ha), and total wood load (OG = 154.7 + 64.1; Y = 87.8 + 100.6 m³ /ha) 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 = 7.10 + 6.9 m³; Y =

			<u>disturbance history.</u>	<u>23 younger forests.</u>	<u>1.71 + 2.81 m3). 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.</u>
<u>Jackson et al., 2001</u>	<u>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.</u>	<u>Instream LW, particle size, surface roughness</u>	<u>LW as functional and nonfunctional (not altering flow hydraulics). Particle size distributions.</u>	<u>Data collected for only 1-year pre- and 1-month post-harvest. These results only describe immediate effects of harvest on stream conditions.</u>	<u>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.</u>
<u>Liquori, 2006</u>	<u>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.</u>	<u>Tree and tree fall characteristics, Site characteristics</u>	<u>Tree characteristic data estimated cause of mortality, and distance to the stream. Tree recruitment probability curves were developed as a function of tree height.</u>		<u>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.</u>

Martin & Grotefendt, 2007	Buffer widths a minimum of 20 m. Multiple buffer widths and harvest intensities.	Instream wood load, stand mortality	Counts of downed wood, tree stumps, stand characteristics, instream wood from aerial photographs taken post-logging	Stand and stream characteristic, and LW data was surveyed from aerial photographs.	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.
May & Gresswell, 2003	Survey of LW in three second-order streams and the mainstem of the North Fork of Cherry creek.	LW, delivery mechanism	LW > 20 cm diameter, and >2 m length was categorized by 4 delivery mechanisms. Delivery process, disturbance type, and channel characteristics.	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.	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%).
Mcintyre et al., 2021	(1) unharvested reference, (2) 100% treatment, a two-sided 50-ft riparian buffer along the entire Riparian Management Zone			Hard Rock Study Physical constraints such as a lack of suitable low gradient reaches and/or issues with	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

	<u>(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).</u>			<u>accessibility related to weather limited downstream measurements of exports to just eight sites.</u>	<u>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.</u>
<u>Meleason et al., 2003</u>	<u>Multiple buffer widths and upland harvest intensities</u>	<u>Change in instream wood load over time</u>	<u>Simulation metrics for forest growth, tree breakage, and in-channel process</u>	<u>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.</u>	<u>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.</u>
<u>Nowakowski & Wohl, 2008</u>	<u>History of regulated and unregulated timber harvest practices.</u>	<u>Instream wood volume</u>	<u>LW volume, LW characteristics source evidence, buffer widths, reach and stream characteristics.</u>		<u>In-stream LW was 2-3 times lower in a watershed with a history (>100 years) of timber harvest (1.1 m³/100 m) when compared to unmanaged reference watersheds (3.3 m³/100 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 (r² = 0.8048). For the unmanaged watershed the model produced stream valley sideslope gradient as the single best predictor of wood load (r² = 0.5748). Shear stress was the best predictor of wood load in the managed watersheds (r² = 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</u>

					<u>(1.1 m³/100 m) had, on average, 2-3 times lower in-stream wood loads than unmanaged (3.3 m³/100 m) watersheds.</u>
<u>Reid & Hassan, 2020</u>	<u>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).</u>	<u>Instream LW</u>	<u>Models were calibrated with long-term data for site and LW characteristics in treatment reaches dating back to 1973.</u>	<u>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.</u>	<u>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.</u>
<u>Schuett-Hames & Stewart, 2019a</u>	<u>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.</u>	<u>LW recruitment, instream wood volume, stand structure</u>	<u>LW volume, LW characteristics, LW source evidence, reach and stream characteristics, basin metrics, stand metrics</u>	<u>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.</u>	<u>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.</u>
<u>Schuett-Hames et al., 2011; Schuett-Hames & Stewart, 2019b</u>	<u>Clearcut to stream with 30-foot equipment exclusion zone, and 50-foot no-cut buffers</u>	<u>LW, mortality, stand structure, canopy cover</u>	<u>QMD, basal area, tree fall rates, instream LW counts and volume, canopy percentage from densiometer.</u>	<u>1) Substantial variability among sites. 2) Due to scale of study, results only applicable to immediate vicinity of buffer treatment.</u>	<u>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.</u>

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 m³, which was less than half of the average found at higher gradient reaches (25.2 m³); 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

					<p>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.</p>
--	--	--	--	--	--

3553

3554 Table A-34. List of treatments, variables, metrics, and results from publications reviewed for information on sediment inputs and
 3555 source.

Reference	Treatment	Variables	Metrics	Notes	Results
<u>Bywater-Reyes et al., 2017</u>	<u>Harvest had a mixture of intensities including clearcut to stream and clearcut with 15 m buffers.</u>	<u>Sediment concentration, basin lithology, geomorphology</u>	<u>Channel, stream, and riparian area characteristics sourced from a mixture of LiDAR and management data.</u>	<u>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.</u>	<u>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).</u>
<u>Bywater-Reyes et al., 2018</u>	<u>long-term data (60 years) of sediment, discharge, weather, and disturbance.</u>	<u>Sediment yield, discharge history, physiography.</u>	<u>suspended sediment concentration involved using either vertically integrated storm-based grab samples, or discharge-proportional composite samples.</u>	<u>The authors caution that the high variability of sediment yield over space and time (~0.2 - ~953 t/km²) indicates that the factors tested in this study should be tested more broadly to investigate their</u>	<u>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</u>

				<u>utility to forest managers.</u>	<u>watersheds with high slope variability and within a decade of forest management and a large flood event.</u>
<u>Hatten et al., 2018</u>	<u>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</u>	<u>suspended sediment concentrations (SSC)</u>	<u>suspended sediment, stream discharge, and daily precipitation</u>	<u>Phase I harvest; 2009 harvest of upper half of watershed. Phase II harvest: 2015 harvest of lower half of watershed.</u>	<u>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⁻¹ (~63%) lower after the Phase I harvest and 28.3 mg L⁻¹ (~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.</u>
<u>Karwan et al., 2007</u>	<u>clearcut of the watershed area of by 50%, partial cut of 50% canopy removal, timber road construction Riparian zone harvest followed Idaho FPA rules.</u>	<u>Total suspended solid (TSS) yields</u>	<u>Monthly total suspended solid readings from multiple flume locations for pre-, and post-harvest, and pre- and post-road construction.</u>		<u>A significant and immediate impact of harvest on monthly sediment loads in the clear-cut watershed (p = 0.00011), and a marginally significant impact of harvest on monthly sediment loads in the partial-cut (p = 0.081) 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: p = 0.2336; partial-cut: p = 0.1739) compared to the control watersheds. Road construction in both watersheds did not result in statistically significant</u>

					<u>impacts on monthly sediment loads in either treated watershed during the immediate or recovery time intervals.</u>
<u>Litschert & MacDonald, 2009</u>	<u>Data collected from 4 NF of Nort CA. ~200 harvest sites near riparian zones with 90 m and 45 m buffer widths.</u>	<u>Sediment delivery pathway frequency and characteristics.</u>	<u>Pathway length, width, origins, and connectivity of sediment delivery pathways to streams.</u>	<u>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.</u>	<u>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.</u>
<u>Macdonald et al., 2003a</u>	<u>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.</u>	<u>suspended sediment yields, stream discharge</u>	<u>Discharge rate and total suspended sediments (TSS) collected using Parshall flumes</u>	<u>Only 1-year pre-harvest data was collected to generated predicted TSS and discharge values post-harvest.</u>	<u>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.</u>
<u>McIntyre et al., 2021</u>	<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</u>	<u>stream discharge, turbidity, and suspended sediment export.</u>		<u>Type N (non-fish-bearing streams). Hard-Rock study.</u>	<u>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.</u>

	<u>least 50% of the RMZ, (4) 0% treatment, clearcut to stream edge (no-buffer).</u>				
<u>Mueller & Pitlick, 2013</u>	<u>The study used sediment concentration data from 83 drainage basins in Idaho and Wyoming.</u>	<u>Sediment concentration, basin lithology, geomorphology</u>	<u>Sediment concentration distribution, geomorphology, and weather data from multiple sources.</u>		<u>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.</u>
<u>Puntenney-Desmond et al., 2020</u>	<u>Variable retention buffers with clearcut.</u>	<u>surface and subsurface runoff rates, sediment.</u>	<u>Simulation metrics calibrated with runoff and sediment samples from sample area. Precipitation calibrated for 100-year-rain events.</u>	<u>Differences in sediment yield not statistically significant.</u>	<u>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.</u>
<u>Rachels et al., 2020</u>	<u>harvested following the current Oregon Forest Practices Act policies and BMPs</u>	<u>proportion of sediment from sources</u>	<u>Sediment collected in traps; sourced using chemical analysis</u>	<u>limited sample size (1 treatment, 1 paired reference watershed) and does not incorporate the effects of different watershed physiography on sediment erosion.</u>	<u>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).</u>

Safeeq et al., 2020	Long term (51 years) effects of clearcut to stream followed by broadcast burn.	streamflow, sediment transport	Historical streamflow data, precipitation data, sediment grab samples for bedload and suspended sediment.	Data compared one treatment watershed and one control watershed across 51+ years.	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/km2 (10%) of the suspended load and 113 Mg/km2 (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 < 10% of the resulting suspended sediment transported into streams, while the increase in sediment supply due to harvest disturbance was responsible for >90%.
Wise, 2010	Streamflow patterns derived from instrumental data and from reconstructed tree-ring chronologies were compared with other previously reconstructed rivers in similar climates.	Streamflow	Dendrochronology, historical data records, seasonal patterns	The reconstruction model developed for the analysis explained 62% of the variance in the instrumental record after adjustment for degrees of freedom.	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 < 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.
Wissmar et al., 2004	Data sourced from management records and geospatial data to identify high erosion-risk areas.	Sediment, weather, stand characteristics, landscape factors	unstable soils, immature forests, roads, critical slopes for land failure, and rain-on-snow events		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.

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
Bladon et al., 2016	15 m buffer with a minimum of ~3.7 m² 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 + 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).
Bladon et al., 2018	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

					<u>catchments with a high percentage of harvest that were underlain by permeable geology</u>
<u>Cole & Newton, 2013</u>	<u>clearcut to stream, partial buffer (12 m width on predominant sun-side), Oregon state BMP (15-30 m no-cut buffer both sides)</u>	<u>Stream temperature</u>	<u>Controlled for yearly fluctuations in temperatures by analyzing the difference in stream temperature entering and exiting the reach with digital temperature data loggers</u>	<u>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.</u>	<u>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.</u>
<u>Cupp & Lofgren, 2014</u>	<u>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.</u>	<u>Canopy closure, shade measurements, stream temperature</u>	<u>Hand-held densiometer (canopy closure), self-leveling fisheye lens digital camera (shade), temperature data loggers</u>	<u>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.</u>	<u>Results showed post-harvest shade values decreased in SR sites (mean effect of -2.8%, p = 0.002), as did the canopy closure values (mean effect of -4.5%, p < 0.001). 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.</u>

<p>Ehinger et al., 2021</p>	<p>1) Buffers encompassing the full width (50 feet), 2) <50ft buffers, 3) Unbuffered, harvested to the edge of the channel, and 4) Reference sites in unharvested forests.</p>	<p>Canopy closure estimated from densitometer, 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) and by 0.1°C at another (at year 5 post-harvest). None of the three REF sites exceeded 16°C during the study.</p>
<p>Gravelle & Link, 2007</p>	<p>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.</p>	<p>stream temperatures at the headwater streams immediately adjacent to treatments, and downstream in larger fish-bearing streams.</p>	<p>Stream temperature data collected from digital sensors.</p>	<p>for the non-fish-bearing, headwater sites pre-treatment data was only collected one season prior to treatment.</p>	<p>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 site. 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.</p>

Commented [WB162]: Also included many metrics mentioned elsewhere in this table.

Commented [WB163R162]: Address

Commented [bs164R162]: included

Commented [WB165]: Multiple statistical analyses were run on the temperature response (e.g. GLS, GLIMMIX), see 4-3.4 of Ehinger et al 2021. "Small sample size" is not an informative metric, please provide actual sample sizes if mentioned in this table to provide reader with information to determine how the sample sizes of the studies compare to each other. If possible find a way to normalize the data for comparison. E.g. Soft Rock - 7 treatment basins (~7000 m of streams treated with current forest practice buffers), 3 reference basins (~3000m of streams), and 57 temperature stations. This study had an unbalanced design (reference sites were well matched and in close proximity with treatments).

Commented [WB166R165]: Address

Commented [bs167R165]: Addressed

<p>Groom et al., 2011a</p>	<p>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.</p>	<p>Stream temperature</p>	<p>Stream temperature collected with digital temperature sensors within harvested areas before and after treatment.</p>	<p>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 harvested along one stream bank, of which 13 were state forest sites. The remaining 16 sites were harvested along both banks.</p>	<p>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 stream temperature by more than 0.3 °C above its ambient temperature</p>
<p>Groom et al., 2011b</p>	<p>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) m²/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.</p>	<p>Stream temperature, Shade, canopy cover</p>	<p>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.</p>	<p>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.</p>	<p>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.</p>

Guenther et al., 2014	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.
Hunter & Quinn, 2009	an alluvial study site and a bedrock study site whose overall characteristics were otherwise comparable apart from geomorphology.	Stream temperature, Alluvial depth	Water temperature was recorded at 75-m intervals along each channel during the summers of 2003 and 2004	Small sample sizes, results only from two sites for two summers. Actual numeric values not reported but shown in graphs.	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.
Janisch et al., 2012	clearcut logging with two riparian buffer designs: a continuous buffer and a patched buffered stream. Buffers were 10-15 m wide.	Stream temperature	Channel and catchment attributes (e.g., BFW, Confinement, slope, FPA, etc.), Stream temperatures were recorded with a Tidbit datalogger in areas persistently submerged.	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.	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<0.01) and length of surface flow (0.67, p=0.05) were strongly correlated with post-logging temperature changes.

Johnson & Jones, 2000	clearcut to stream, patch cutting followed by debris flows (resulted in the removal of all streamside vegetation), 450+ yo Doug-fir forest reference.	Stream temperature	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.	The experimental design used historic stream temperature data to examine changes in stream temperatures. This required conflating data from 2 different devices.	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. Differences in treatment streams and reference stream temperatures were less than 1.1°C pre-treatment and 30-years post-treatment.
Kaylor et al., 2017	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.
Kibler et al., 2013	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.

<p><u>Macdonald et al., 2003b</u></p>	<p>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.</p>	<p>Stream temperature</p>	<p>Temperature data were recorded with Vemco dataloggers. Canopy cover was estimated with densiometers.</p>		<p>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>
<p><u>McIntyre et al., 2021</u></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><u>Pollock et al., 2009</u></p>	<p><u>A range of harvest from 0 – 100%, < 20 years old regrowth, ~ 40 years old regrowth . Unharvested sites were estimated as being >150-years old</u></p>	<p><u>Stream temperature, time since harvest, percent of watershed and stream network harvested.</u></p>	<p><u>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.</u></p>	<p><u>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.</u></p>	<p><u>Results of general temperature patterns showed that average daily maximum (ADM) were strongly correlated with average diurnal fluctuations ($r^2 = 0.87$, $p < 0.001$, $n = 40$), 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 ($r^2 = 0.39$, $p < 0.001$, $n = 40$) and 32% of variation in the average daily range (ADR) ($r^2 = 0.32$, $p < 0.001$, $n = 40$). 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 ($p < 0.001$). 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 ($p < 0.001$). Results for the correlations between the riparian network scale forest harvest and stream temperature showed that the total percentage of the riparian forest network upstream of temperature loggers harvested explained 33% of the variation in the ADM among subbasins ($r^2 = 0.33$, $p < 0.001$, $n = 40$) and 20% of variation in the ADR ($r^2 = 0.20$, $p = 0.003$, $n = 40$). 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 ($r^2 = 0.03$, $p = 0.79$, $n = 40$), the ADR of stream temperatures ($r^2 = 0.02$, $p = 0.61$, $n = 40$) or any other stream temperature parameters. The proportion of total harvested near upstream riparian forest (avg = 0.66, SD ± 0.34, range = 0.0-1.0) was weakly correlated with ADM ($r^2 = 0.12$, $p = 0.02$, $n = 40$) and not significantly correlated with ADR ($r^2 = 0.07$, $p = 0.06$, $n = 40$). Even when the upstream riparian corridor length was shortened to 400 m and</u></p>
------------------------------------	---	--	--	--	--

					then to 200 m, and the definition of recently harvested was narrowed to <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 >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.
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), and unharvested reference (REF) streams.	Stream temperature	Temperature data was separated into 5 th , 25 th , 50 th , 75 th , and 95 th percentiles. the researchers also quantified the percentage of summer where temperatures were above 16 and 15 °C.	Sample sizes are relatively low for some treatments. (CC_NB; n = 4); (CC_B; n = 3); (TH_B; n =1); (REF; n = 7).	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.
Reiter et al., 2015	. Various buffer prescriptions as regulations changed over time. (mid1970s – 1980s = “nominal”; mid 1980s – mid	Stream temperature data from four permanent sampling stations in the Deschutes River Watershed	Long term stream and air temperature collected from sampling stations. To detect correlations of stream and air temperature change with land	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.	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

	<p><u>1990s = 23 m; 2001 – 2009 = 30 m buffers)</u></p>	<p><u>from 1975- 2009. Results for this analysis are for 3 watersheds (1- large, 1-medium, 1- small)</u></p>	<p><u>management activity separately from climate changes the data was fit to a model that included the effects of climate.</u></p>	<p><u>0.09°C for 1984 – 1999, and 0.02°C. for 2000 – 2009.</u></p>	<p><u>(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 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.</u></p>
--	---	--	---	--	--

<p>Roon et al., 2021a</p>	<p>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)</p>	<p>Stream temperature, solar radiation, Shade</p>	<p>Stream temperature was collected using digital sensors; solar radiation was measured using silicon pyranometers; riparian shade was measured using hemispherical photography.</p>	<p>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.</p>	<p>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.</p>
<p>Roon et al., 2021b</p>	<p>Effective shade reductions ranging between 19-30% along 200 m reach, or 4-5% along 100 m reach.</p>	<p>local and downstream temperature</p>	<p>Stream temperature collected with digital temperature sensors within harvest area and every 200 m downstream of stream network.</p>	<p>Stream temperature data was only collected for one-year pre- and one-year post-harvest.</p>	<p>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.</p>

<p><u>Sugden et al., 2019</u></p>	<p>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.</p>	<p>Stream temperature, fish population, Canopy cover</p>	<p>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.</p>	<p>Data only collected for one year pre-harvest and one year post-harvest.</p>	<p>The mean basal area (BA) declined from 30.2 m²/ha pre-harvest to 26.4 m²/ha post-harvest (mean = -13%, range from -32% to 0%). Windthrow further reduced the mean BA to 25.9 m²/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 (MWM) 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.</p>
<p><u>Swartz et al., 2020</u></p>	<p>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 m² to 1,374 m² with a mean of 962 m².</p>	<p>Stream temperature, Light reaching stream, canopy cover</p>	<p>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.</p>	<p>Data was collected for one year pre-harvest, during harvest year (harvest took place in late fall 2017), and one-year post-harvest.</p>	<p>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⁻² day⁻¹, overall resulting in a mean change in light of 2.93 (SD ± 1.50) moles of photons m⁻² day⁻¹. 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</p>

					<p><u>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.</u></p>
<p><u>Warren et al., 2013</u></p>	<p><u>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.</u></p>	<p><u>Light reaching bottom of stream, canopy cover</u></p>	<p><u>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</u></p>	<p><u>Relatively small sample sizes (n = 4). Significant differences were only found in 3 of the four paired reaches.</u></p>	<p><u>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.</u></p>

3559

3560 [Appendix II](#)

3561

3562 **Shade and LW**

3563

3564 Anderson & Meleason, 2009

3565

3566 Anderson, P.D., Meleason, M.A., 2009. Discerning responses of down wood and understory
3567 vegetation abundance to riparian buffer width and thinning treatments: an equivalence-
3568 inequivalence approach. *Can. J. For. Res.* 39, 2470–2485 <https://doi.org/10.1139/X09-151>

3569

3570 The purpose of this study was to determine the effect of buffer width on understory vegetation
3571 and down woody responses both within the unthinned buffer and in the adjacent thinned stand. A
3572 secondary objective of this study was to explore the ability of equivalence-nonequivalence
3573 statistical tests at assessing the degree of similarity between stands. The focus of this study was
3574 on second-growth stands dominated by Douglas-fir at multiple sites along the coast and Cascade
3575 Range in western Oregon. Six combinations of buffer width and upslope density management
3576 prescription were evaluated: one site potential tree height buffer averaging 69 m adjacent to
3577 thinning and a 0.4 patch opening; variable width buffer averaging 22 m adjacent to thinning and
3578 a 0.4 patch opening; streamside retention width averaging 9 m adjacent to thinning; and an
3579 unthinned stand serving as a reference. Pearson correlation and multivariate analysis of variation
3580 were used to examine data on percent cover of small and large down wood, and percent cover of
3581 shrubs, herbs, and moss. Inferences on buffer performance were generated using linear mixed
3582 model analysis, equivalence-inequivalence tests, and two post-hoc comparisons. The results from
3583 this study show upland thinning led only to subtle changes in understory vegetation cover and
3584 composition with vegetation responses most prevalent with narrow buffer widths and particularly
3585 when adjacent to patch openings. There was a lack of significant change in down wood response
3586 to treatments.

3587

3588 **Shade**

3589

3590 Anderson et al., 2007

3591

3592 Anderson, P.D., Larson, D.J., Chan, S.S., 2007. Riparian buffer and density management
3593 influences on microclimate of young headwater forests of western Oregon. *For. Sci.* 53, 254–
3594 269. <https://doi.org/10.1093/forestscience/53.2.254>

3595

3596 The purpose of this study was to characterize variation in overstory density, canopy closure, and
3597 microclimate as a function of distance from headwater streams, and (2) determine differences in the
3598 ability of thinned stands and unthinned stands to maintain understory microclimate above the stream
3599 channel and in the riparian zone. The focus of this study was on second-growth (30- to 80-year-old)
3600 Douglas-fir forests characteristic of western Oregon. The study was located at four sites along the
3601 Oregon coast and at one site on the western Oregon Cascade Range. Stands were either thinned to
3602 approximately 198 trees per acre (TPA) or were left unthinned and ranged from 500-865 TPA. Within
3603 thinned stands, 10% of the area was harvested to create patch openings and 10% was left as clusters of
3604 “leave islands”. Streams within treated stands were surrounded by buffers of either (1) one site-potential
3605 tree averaging 69 m (B1), (2) variable width buffer averaging 22 m (VB), or (3) streamside retention
3606 buffer averaging 9 m (SR-T). These six combinations of buffer width and adjacent density management
3607 were evaluated using univariate linear modeling and compared with untreated (UT) stands. Microsite
3608 and microclimate data were obtained through repeated transect measurements extending laterally from
3609 stream center and into the riparian zone and upland treated stand 2-5 years after treatment. The stand
3610 basal area was determined through variable radius plot sampling. Canopy cover was estimated through
3611 photographic techniques during the summer leaf-on period. The results from this study show that the
3612 ability of narrow streamside buffers (SR-T) at moderating stream microclimate in treated stands was
3613 questionable. Visible sky at stream center only differed significantly between SR-T (9.6%) and UT
3614 (4.2%) stands. The SR-T stands showed a +4.5°C difference in daily maximum temperatures just above
3615 stream center when compared to the UT stands. However, this difference was not statistically significant.
3616 The researchers report that SR-T had a weak temperature gradient (tested at 0-10 m and 10-30 m
3617 increments from stream center) indicating the stream center and buffer microclimates were nearly the
3618 same as upslope in the thinned stand. Within the riparian buffer zones daily maximum temperatures
3619 were higher in all treated stands when compared to UT stands. The differences in daily maximum
3620 temperatures ranged from 1.1°C (B1) to 4.0°C (SR-T), but the difference was only significant in one
3621 SR-T stand. The maximum air temperature within buffer zones adjacent to patch openings was 3.5°C
3622 higher than in UT stands. Soil temperature changes were only evident within patch openings ranging
3623 from 3.6 - 8.8°C higher than in UT stands. The researchers of this study conclude by saying that buffers
3624 with widths defined by the transition of riparian to upslope vegetation or significant topographic slope
3625 breaks appear sufficient at mitigating effects from upslope harvests on the above-stream microclimate.
3626 Their suggestions for further study center around cross-disciplinary research into the relationships
3627 between forest structure, microclimate, and habitat suitability on headwater riparian organisms.

3628

3629 **Stream Temperatures**

3630

3631 Cole & Newton, 2013

3632

3633 Cole, E., & Newton, M. (2013). Influence of streamside buffers on stream temperature response
3634 following clear-cut harvesting in western Oregon. Canadian journal of forest research, 43(11), 993-1005.
3635 <https://doi.org/10.1139/cjfr-2013-0138>

3636

3637 This study compares the changes in stream temperatures following a clearcut with three different buffer
3638 treatments – no tree buffer, predominantly sun-sided 12 m wide partial buffer, and a two-sided 15-30 m
3639 buffer (BMP for this area). The study was conducted on four small fish bearing streams in the area
3640 surrounding Corvallis, Oregon. Streams were dominated by both hardwood and conifers and were
3641 located at low- and mid-elevations. Each treatment alternated with unharvested reference sections along
3642 study reaches spanning 1800-2600 meters. Stream temperature data adjacent to treatment and
3643 downstream of treatment were collected for 2 –years prior and 4 to 5 years following harvest. Time-
3644 series regression analysis was used to evaluate the change in temperatures between pre- and post-
3645 harvest. The researchers controlled for yearly fluctuations in temperatures by analyzing the difference in
3646 stream temperature entering and exiting the experimental reaches. Results ~~showed significant~~ showed
3647 significant increases in daily maximum, mean, and diel fluctuations in temperatures post-harvest for all
3648 no tree buffers (up to 3.8 °C). The no tree buffers also showed small but significant changes below
3649 predicted summer minima by as much as 1.2°C. The partial buffer units varied in their response to
3650 treatment exhibiting increases, decreases, and no change from preharvest trends. For example, at one
3651 site, there were no detectable changes in means, minima, or diel fluctuations but significantly lower
3652 maximum temperatures post-harvest ($p = 0.0021$; actual temperatures not reported). Partial buffers at
3653 another site reported lower trends in mean, maxima, and diel fluctuations in temperature post-harvest,
3654 and no difference in minima. Only one partial buffer site showed increases in all recorded trends (mean,
3655 minima, maxima, diel fluctuations). The BMP buffered treatment sites also showed variation in results.
3656 One site showed no detectable changes, one site showed small but significant ($p < 0.0350$; actual
3657 temperatures not reported) decreases in downstream temperatures. Only two BMP buffered sites showed
3658 significant ($p < 0.0499$) increases in mean, maxima, and diel fluctuations in temperatures. The highest
3659 increase in maxima for any BMP buffered site was 5.3°C. Changes in temperature trends in uncut
3660 reference post-treatment were minimal and attributed to downstream effects from the treatment reaches.
3661 However, when post-harvest trends in upstream treated sites were higher than pre-harvest temperatures
3662 tended to fall below pre-harvest values when passing through the unharvested downstream units. For
3663 within-unit trends, unharvested units downstream from no tree and partial buffers showed trends of
3664 significantly decreasing daily maximum temperatures. When the data was analyzed by 7-day moving
3665 mean maximum temperatures, the no tree buffers showed significant increases after harvest. The authors
3666 report that most partial and BMP buffers resulted in minimal increases or negligible changes to the 7-day
3667 moving mean maximum temperatures (actual values not reported). Significant changes in one or more
3668 temperature trends (mean, minima, maxima, diel fluctuations) were detected in all treatment stream
3669 post-harvest with only one exception at a BMP buffered site This was a well planned and executed
3670 experimental design that shows how changes in stream temperatures post-harvest are directly related to
3671 residual buffer treatment while also showing evidence that many other factors such as stream features

3672 (orientation, topography, ground water source) can compound or ameliorate these effects (I.e., changes
3673 in temperature were highly affected by site factors).

3674

3675 **Stream Temperature**

3676

3677 Johnson & Jones, 2000

3678

3679 Johnson, S. L., & Jones, J. A. (2000). Stream temperature responses to forest harvest and debris flows in
3680 western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences*, 57(S2), 30-39.
3681 <https://doi.org/10.1139/f00-109>

3682

3683 This paper is a study of the changes in mean stream temperature minimum, maximum, diurnal
3684 fluctuation, and interannual and seasonal variability following harvest in three small basins of the
3685 H.J. Andrews experimental watershed between 1962 and 1966. The experimental design used
3686 historic stream temperature data to examine changes in stream temperature following clear-cut
3687 (no buffer) and burning in one watershed; patch cutting and debris flows (resulted in the removal
3688 of all streamside vegetation 3 years after cut) treatments in another watershed; and one old-
3689 growth uncut reference watershed. All watersheds were dominated by 450-year-old Doug-fir
3690 forests prior to harvest. Data was analyzed for the period 1959-1997. Mean weekly temperature
3691 maximum, minimum, and annual fluctuations were compared between all three watersheds using
3692 a complete factor analysis of variance (ANOVA). The experiment also involved long-term
3693 monitoring to evaluate time until recovery of pre-treatment temperature fluctuations. Results
3694 showed a significant increase in stream temperatures in both treatment watersheds after treatment
3695 compared to the unharvested site. The unharvested watershed showed higher interannual
3696 variability in maximum stream temperatures ranging from 15 to 19°C. The two treatment
3697 watersheds, despite differences in disturbances, (clear-cut and burn vs. Patch cut and debris-
3698 flow) followed similar trajectories from 1966-1982. Stream temperature summer maximums
3699 reached 23.9°C and 21.7°C 1-2 years post-harvest (clear-cut/burn and patch-cut/debris flow
3700 respectively) and returned to pre-harvest summer temperatures by 1980 (~15 years post-harvest).
3701 Both treatment watersheds exhibited significant increases in mean weekly minimum and
3702 maximum stream temperatures in the summer months immediately following harvest and for at
3703 least 3 years compared to the unharvested reference. The clear-cut and burn watershed's
3704 weekly maximum summer temperatures ranged between 5.4 and 6.4°C higher, and mean weekly
3705 minimum ranged 1.6-2.0°C higher than the reference streams for 4 years post-harvest. The patch-
3706 cut and debris-flow watershed exhibited mean weekly maximum stream temperatures 3.5-5.2°C
3707 higher than in the reference stream for 3 years following harvest/disturbance. Prior to harvest and
3708 30 years post-harvest the mean weekly maximum and minimum stream temperatures for both
3709 treatment streams differed less than 1.1°C from the reference stream. These differences in stream
3710 temperatures from treated and untreated sites were amplified during periods of high solar inputs

3711 and reduced during periods of cloud cover. Differences in stream temperatures were greatest
3712 during the end of July and beginning of June. Diurnal fluctuations in stream temperatures were
3713 also significantly higher in both treatment watersheds (6-8 °C in the clearcut, and 5-6 °C in the
3714 patch-cut) relative to the reference stream (1-2 °C). Stream temperatures returned to pre-harvest
3715 levels after 15 years of growth.

3716

3717 **Large Wood (LW)**

3718

3719 Bahuguna et al., 2010

3720

3721 Bahuguna, D., Mitchell, S.J., Miquelajauregui, Y., 2010. Windthrow and recruitment of large woody
3722 debris in riparian stands. *Forest Ecology and Management* 259, 2048–2055.
3723 <https://doi.org/10.1016/j.foreco.2010.02.015>

3724

3725 The purpose of this paper was to evaluate the effect of riparian buffer width on windthrow and LW
3726 recruitment and to contrast data with unharvested controls. This paper also seeks to document the
3727 geometry of post-harvest windthrow from buffers of varying widths and to develop a model framework
3728 for incorporating supply of LW originating from windthrow to streams from riparian buffers. The focus
3729 of this paper is on dense young conifer-dominated forests originating from harvest followed by wildfire.
3730 This study is located in the Coast Mountains, approximately 60 km east of Vancouver, BC. Two buffer
3731 widths on each side of the stream (10 m and 30 m) along with an unharvested control were each
3732 replicated three times in the experiment. The researchers used a strip plot sampling method running
3733 parallel to the stream to collect data on species, diameter, height, and status (standing live/dead)
3734 beginning in the year prior to harvest and annually thereafter for seven years. A General Linear Model
3735 Procedure was used to determine the significance of variables. The Pearson correlation coefficient was
3736 used to assess correlations and potential predictor variables. Multiple linear regression was then used to
3737 determine the utility of the variables at determining LW height above the stream. Following harvest,
3738 11% of initially standing timber was blown down in the first and second years in the 10 m buffer,
3739 compared to 4% in the 30 m buffer, and 1% in the unharvested controls. Following 8 years post-harvest,
3740 a significant amount of annual mortality occurred in the unharvested control at 30%, compared to 15%
3741 in both 30 m and 10 m buffers. 77% of LW was in the 10 cm - 20 cm diameter class while the mean
3742 diameter of standing trees in riparian buffers was 30 cm indicating small diameter trees were
3743 significantly more represented in streams. Only 3% of windthrown logs fell perpendicular to the stream
3744 with the majority falling diagonal-perpendicular relative to the stream. The researchers of this study
3745 conclude that recruitment of logs into streams lags behind the post-harvest pulse of windthrow by
3746 several years. The lag depends on the size, species, and condition of logs, and their direction of fall
3747 relative to stream valley geometry.

3748

3749 **Species Richness**

3750

3751 Baldwin et al., 2012 (Removed from focal list)

3752

3753 Baldwin, L.K., Petersen, C.L., Bradfield, G.E., Jones, W.M., Black, S.T., Karakatsoulis, J., 2012.

3754 Bryophyte response to forest canopy treatments within the riparian zone of high-elevation small streams.

3755 Can. J. For. Res. 42, 141–156. <https://doi.org/10.1139/x11-165>

3756

3757 The purpose of this study was to examine the influence of forest harvesting practices and distance from
3758 the stream on riparian-bryophyte communities. The experiment was limited to the montane spruce forest
3759 type which is considered moderately open and dominated by lodgepole pine in the uplands and by
3760 hybrid spruce in well-developed riparian areas. The study took place at five different watersheds located
3761 approximately 70 km from Kamloops, BC. Three primary treatments: clear-cut (n=7), two-sided buffer
3762 averaging approximately 15 m on both sides (n=10), and a continuous forest (n=6) were used to sample
3763 numerous environmental variables including elevation, aspect, slope, buffer width, and CWD decay
3764 class. Bryophytes (classified into life history strategies), stand structure, and microhabitat were also
3765 measured 1, 5, and 10 m from the streams edge. Additionally, the DBH of all conifer stems as well as
3766 percent vegetation cover were measured along transects. All data were collected in July-August of 2007
3767 and 2008. Minimum time since disturbance for clearcut sites was 13 years versus a minimum of 5 years
3768 in buffered sites. An analysis of variance was used to compare environmental, stream, and stand
3769 structure characteristics among canopy treatments. Mean values were calculated for stand structure and
3770 substrate variables recording in transects. Bryophytes were analyzed within functional groups based on
3771 growth form, substrate affiliations, and life history. Linear models were used to evaluate the effects of
3772 distance to stream, forest canopy treatment, and their interaction on response variables. Overall CWD
3773 did not differ significantly among treatments, although buffer treatment sites had significantly higher
3774 volume of CWD in early decay classes compared to clearcut and continuous forests. The researchers
3775 suggest the early decay class CWD in buffer treated sites was likely the result of increased stem
3776 breakage. After accounting for distance from the stream, the richness and frequency of bryophyte
3777 functional communities was intermediate to continuous and clearcut sites. Compared to continuous sites,
3778 buffered sites featured significantly lower richness and frequency of many forest-associated groups.
3779 Furthermore, buffered sites also did not support increased richness or frequency of disturbance-
3780 associated species. Clearcut treatments featured higher levels of disturbance associated species including
3781 colonists, canopy species, and species typically found on mineral soil. Data from this study also showed
3782 bryophyte species richness and frequency decline with increasing distance from the stream. The authors
3783 conclude by noting that while bryophyte communities in buffered sites are significantly more diverse
3784 than communities in clearcut sites, reductions in forest-associated species as well as in the bryophyte
3785 mat as a result of large-scale forestry indicate that the ecological function of buffer-dwelling bryophyte
3786 communities may be hindered and could benefit alongside large uncut forest reserves.

3787

3788 **Sediment**

3789

3790 Mueller & Pitlick, 2013

3791

3792 Mueller, E. R., & Pitlick, J. (2013). Sediment supply and channel morphology in mountain river
3793 systems: 1. Relative importance of lithology, topography, and climate. *Journal of Geophysical*
3794 *Research: Earth Surface*, 118(4), 2325-2342. <https://doi.org/10.1002/2013JF002843>

3795

3796 This study used correlation analysis to assess the relative impact of lithology, basin relief, mean basin
3797 slope, and drainage density on in stream sediment supply defined by the bankfull sediment concentration
3798 (bedload and suspended load). The study used sediment concentration data from 83 drainage basins in
3799 Idaho and Wyoming. Lithologies of the study area were divided into four categories ranging from
3800 hardest to softest- granitic, metasedimentary, volcanic, and sedimentary. The results showed the
3801 strongest correlation of bankfull sediment concentration was with basin lithology, and showed little
3802 correlation strength with slope, relief and drainage density. As lithologies become dominated by softer
3803 parent materials (volcanic and sedimentary rocks), bankfull sediment concentrations increased by as
3804 much as 100-fold. These results suggest that lithology can be more important in estimating sediment
3805 supply than topography. The authors discuss using a correlative analysis but give little description of
3806 what that analysis was or how they compare the values of each correlation strength to see if the
3807 differences were significant.

3808

3809 **CWD Modeling**

3810

3811 Benda et al., 2016

3812

3813 Benda, L.E., Litschert, S.E., Reeves, G., Pabst, R., 2016. Thinning and in-stream wood
3814 recruitment in riparian second growth forests in coastal Oregon and the use of buffers and tree
3815 tipping as mitigation. *J. For. Res.* 27, 821–836. <https://doi.org/10.1007/s11676-015-0173-2>

3816

3817 The purpose of this study was to develop a model which examines the effects of riparian thinning
3818 on in-stream wood recruitment in second growth stands. A secondary objective of this study was
3819 to model how manual felling of trees in no-harvest buffer zones impacts the effects of thinning.
3820 The study site was located within the Alcea watershed in central coastal Oregon. Silvicultural
3821 simulation treatments used the reach scale wood model (RSWM) and included: (1) no harvest
3822 control; (2) single entry thinning from below (thinning from below removes the smallest trees to
3823 simulate suppression mortality) with and without a 10 m width no-cut buffers; (3) double entry

3824 thinning from below with the second thinning occurring 25 years after the first with and without
3825 10 m no-cut buffers (4) Each thinning treatment was also combined with some mechanical
3826 introduction of thinned trees into the stream encompassing a range between 5 and 20 % of the
3827 thinned trees. . The simulation model RSWM was run for 100 years in 5-year time steps. In the
3828 no-harvest control, the model output shows the density of live trees declines from 687 trees-per-
3829 hectare (tph) in 2015 to 266 tph in 2110 due to natural suppression mortality (-61 % from initial
3830 conditions). The single-entry thin reduces stand density to 225 tph in 2015 (-67 %) and declines
3831 further to 160 tph by 2110 (-77 %). The double entry thinning resulted in 123 tph after the
3832 second thinning in 2040 (-82%) and maintained that density until 2110. Both thinning treatments
3833 resulted in a substantial reduction of dead trees that could contribute to in-stream wood over
3834 time. The model output for single entry thinning treatments predicts a 33% or 66% reduction of
3835 in-stream wood over a century relative to the unharvested reference for harvest on one side or
3836 both sides of the stream, respectively. Adding the 10-m no cut buffer reduced total loss to 7 and
3837 14%. Including mechanical tipping of 5,10,15, and 20% of cut stems without a buffer in the
3838 single entry thinning treatment changes the relative in-stream percentages of wood relative to the
3839 reference stream to -15, -6, +1, and +6%, respectively. To completely offset the loss of in stream
3840 wood due to single entry thinning mechanical tipping of 14 and 12% were required without and
3841 with buffers. Double entry thinning treatments without a buffer predicted further reduction in
3842 wood recruitment over a century of simulation with 42 and 84% reduction of in stream wood
3843 relative to the reference stream when one side and both sides of the channel were harvested.
3844 Adding a 10 m buffer reduced total reduction of in stream wood to 11 and 22% for thinning on
3845 one and both sides of the channel. To offset the predicted changes of in stream wood volume
3846 following double entry harvest would require tipping of 10 and 7% of cut stems without and with
3847 the 10-m buffer. The authors conclude that thinning without some mitigation efforts resulted in
3848 large losses of in stream wood over a century. However, by including a 10-m no cut buffer or a
3849 practice of mechanical tipping can offset these losses Although predictions from this study
3850 contribute to the in-stream wood recruitment conversation moving forward, the model contained
3851 limitations such as utilizing data from FIA plots which only approximate riparian forest
3852 conditions.

3853

3854 **Modeling Stream Litter Delivery**

3855

3856 Bilby & Heffner, 2016

3857

3858 Bilby, R.E., Heffner, J.T., 2016. Factors influencing litter delivery to streams. *Forest Ecology and*
3859 *Management* 369, 29–37. <https://doi.org/10.1016/j.foreco.2016.03.031>

3860 The purpose of this study was to understand the relative influence of wind speed and direction,
3861 topography, litter type, species, and stand conditions on the distance from which litter is
3862 delivered to streams. This study utilized a combination of field experiments, literature, and

3863 simple models to estimate the width of a delivery areas. The effects of wind speed on litter
3864 delivery distance were measured on litter samples from two common species of the Pacific
3865 Northwest, Douglas-fir and red alder by releasing litter from a riparian tree canopy at various
3866 wind speeds and recording the distances traveled for each litter type at each wind speed. The
3867 relationship between distance of litter recruitment area and variables of interest (e.g., wind speed,
3868 topography, litter type...) were determined with a linear mixed effects model Data for wind speed
3869 and direction was recorded for one year in 30 min intervals along Humphrey Creek in the
3870 Cascade Mountains of western Washington. Results showed that under the wind conditions
3871 recorded at Humphrey Creek the majority of the litter recruited into the stream originated from
3872 within 10 m of the stream regardless of litter or stand type. No difference was found in delivery
3873 distance and litter type (needles or broadleaf) at young sites. However, needles released at mature
3874 sites had a higher proportion of cumulative input from greater distances than needles or alder
3875 leaves released at younger sites. This is likely due to the higher canopy and thus higher release
3876 position. Litter travel distance was linearly related to wind speed ($p < 0.0001$) Doubling wind
3877 speed at one site led to a 67-87% expansion of the riparian contribution zone in the study area.
3878 The results reveal a trend that suggests slope also contributes to the width of the litter
3879 contributing area. However, the authors did not apply statistical analysis to these values and only
3880 speculate that increasing the slope from 0-45% would increase the width of the litter contributing
3881 area by 70%. Overall, the results of this study show evidence that wind speed has a strong effect
3882 on the width of litter delivery areas within riparian areas, but that relationship is also affected
3883 stand age and litter type. Trends in the data also suggest that topography is an important factor,
3884 but it was not quantified.

3885

3886 **Stream Temperature**

3887

3888 Bladon et al., 2016

3889

3890 Bladon, K.D., Cook, N.A., Light, J.T., Segura, C., 2016. A catchment-scale assessment of stream
3891 temperature response to contemporary forest harvesting in the Oregon Coast Range. *Forest
3892 Ecology and Management* 379, 153–164. <http://dx.doi.org/10.1016/j.foreco.2016.08.021>

3893

3894 The purpose of this study was to compare the effects of contemporary riparian forest harvest
3895 treatments under the Oregon Forest Practices Act (15 m riparian management area with a
3896 minimum of ~ 3.7 m² conifer basal area retained for every 300 m length of stream) with historical
3897 riparian forest harvest practices (no maintenance of streamside vegetation) on stream
3898 temperatures. This study took place in the Siuslaw National Forest in the Oregon Coast Range
3899 as part of the Alsea Watershed Study Revisited. Historical records of stream temperatures were
3900 sourced from the original Alsea Watershed Study that monitored stream temperature changes
3901 from 1958-1973, before and after streamside timber harvesting in 1966. Stream temperature data

3902 was collected for contemporary forest practices over a 6-year period (3 years pre- and 3 years
3903 post-harvest; 2006-2012). Data for the contemporary harvest was also compared with stream
3904 temperature changes in unharvested reference streams to support a Before-After-Control Impact
3905 (BACI) design. Stream temperature thermistors were installed, and data was taken at 30-minute
3906 intervals at three sections of both the harvested (2 within harvest boundary and 1 downstream)
3907 and reference sites. Mean canopy closure, as measured with a densiometer, along the stream
3908 channel in the harvested portion of Needle Branch was reduced from ~96% in the pre-harvest
3909 period to ~89% in the post-harvest period. Comparatively, mean canopy closure along the stream
3910 channel in the reference sites were ~92% in the pre-harvest period and 91% in the post-harvest
3911 period. Data was analyzed to assess whether there were changes in the 7-day moving mean of
3912 daily maximum stream temperature, mean daily stream temperature, and diel stream temperature
3913 following harvest. The results showed no significant changes in any of the three parameters
3914 measured following contemporary forest harvesting practices when analyzed across all
3915 catchments for all summer months (July to September). When the mean 7-day moving maximum
3916 temperature was constrained to the summer period between July 15 – August 15 across all sites
3917 there was a significant increase in stream temperatures in the harvested sites by $0.6 + 0.2^{\circ}\text{C}$
3918 following harvest. However, when the data was arranged for individual pair-wise comparisons
3919 with the unharvested sites, and intrinsic annual and site variability was accounted for, the
3920 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
3921 any site. The only comparison made in the study to the original Alsea Watershed study was with
3922 the single day maximum stream temperatures for pre- and post-harvest. The contemporary
3923 practices showed a change of single day maximum stream temperatures from 15.7°C to 14.7°C
3924 (a reduction) from pre- to post-harvest. In contrast, the historical stream temperature data showed
3925 an increase in single day maximum stream temperatures from 13.9°C (pre-harvest) to as much
3926 as 29.4°C (2-years post-harvest). The authors caution that while these results support the
3927 conclusion that contemporary forest practices in Oregon are sufficient in maintaining stream
3928 temperatures after riparian forest harvest, and much more efficient than historical practices; these
3929 results should not be generalized to areas outside of coastal Oregon. The authors caution that the
3930 streams in this study have potential for a muted stream temperature response following harvest
3931 relative to other regions because of the (1) north-south stream orientation, which would
3932 maximize RMA effectiveness (2) steep catchment and channel slopes that can increase stream
3933 velocity and hyporheic exchange, (3) potential increases in groundwater contributions after
3934 harvest.

3935

3936 **Stream temperature**

3937

3938 Bladon et al., 2018

3939

3940 Bladon, K.D., Segura, C., Cook, N.A., Bywater-Reyes, S., Reiter, M., 2018. A multicatchment
3941 analysis of headwater and downstream temperature effects from contemporary forest harvesting.
3942 *Hydrological Processes* 32, 293–304. <https://doi.org/10.1002/hyp.11415>

3943

3944 The purpose of this study was to (1) examine the effects of contemporary forest harvesting
3945 practices on headwater stream temperature, (2) determine if increased temperatures from
3946 harvesting was detectable in downstream fish-bearing streams, and (3) examine the relative role
3947 of geology and forest management on influencing the differential stream temperature responses
3948 in both headwater and downstream reaches. This study took place at three paired watershed
3949 studies, of which two (Aalsea, Trask) were located in the Oregon coast range, and one (Hinkle)
3950 was located in the western Cascades of Oregon. This study featured pre- and post-harvest
3951 measurements, as well as measurements within and downstream from harvested and reference
3952 sites. Buffer widths at harvested sites varied but averaged 20 m on either side of streams.
3953 Statistical models were generated which analyzed whether (a) the 7-day moving average of daily
3954 maximum stream temperature (*7daymax*) changed between pre- and post-harvest sites, and (b)
3955 whether post-harvest changes in *7daymax* were detectable downstream. A regression analysis
3956 was also performed to assess the relative relationship between catchment lithology and percent
3957 catchment harvested on temperature at all sites. Statistical models were generated for each
3958 harvest site and reference pair. The pre-harvest relationship in stream temperatures for paired
3959 sites were used to create predicted changes in stream temperatures post-harvest. The post-harvest
3960 stream temperatures were then compared to the predicted values and the 95% prediction
3961 intervals. If post-harvest values of the *7daymax* were outside the prediction interval the authors
3962 referred to these observations as statistical “exceedances”. Results showed that the *7daymax*
3963 exceeded the predictive interval at 7 of the 8 harvested headwater sites (within the harvested
3964 boundary) when analyzed across all harvest years. The exceedances were largest in the first year
3965 after harvest but diminished in the second and third year at two treatment sites. However, at one
3966 site, the elevated *7daymax* continued for three years post-harvest. In 4 of the 7 harvested sites
3967 with exceedances, the exceedances were recorded between 22 and 100% of the time. Smaller
3968 increases in stream temperatures were detected in the other 3 streams with exceedances, the
3969 exceedances occurred < 15% of the time. There was no evidence of elevated stream temperatures
3970 beyond the predicted intervals in any of the downstream sites following harvesting. The
3971 magnitude of change in stream temperature and transmission of warmer water downstream were
3972 a function of percentage of catchment harvested and the underlying geology. Although, these
3973 relationships were scale dependent. At the upstream, harvested sites there was a strong
3974 relationship between stream temperature increases and catchment lithologies, but no statistically
3975 significant relationship between stream temperature changes and percent of catchment harvested.
3976 Sites downstream from harvested areas showed a strong relationship with the interaction of
3977 percentage of catchment harvested and the underlying lithologies. The greatest temperature
3978 increases at downstream sites were in areas with a higher percentage of catchment harvested and
3979 were underlain by more resistant lithologies. There was no evidence for increases in stream
3980 temperatures in catchments with a high percentage of harvest that were underlain by permeable
3981 geology. The authors suggest that this relationship may be due to the buffering effect of increases

3982 in summer low flows and greater groundwater or hyporheic exchange. They conclude that the
3983 variability of rock permeability and the relative contribution of groundwater during summer
3984 months, and their effect on stream temperatures following harvest should be investigated further.

3985

3986 **Wood Loading**

3987

3988 Burton et al., 2016

3989

3990 Burton, J.I., Olson, D.H., Puettmann, K.J., 2016. Effects of riparian buffer width on wood
3991 loading in headwater streams after repeated forest thinning. *Forest Ecology and Management*
3992 372, 247–257. <https://doi.org/10.1016/j.foreco.2016.03.053>

3993

3994 The purpose of this study was to examine the relationship between in-stream wood loading and
3995 riparian buffer width in thinned stands in conjunction with several stand, site, and stream
3996 variables. This study is a part of a larger density management study which covered 6 sites along
3997 the coastal and western Cascade Range of Oregon. The sites used for this study were dominated
3998 by Douglas-fir and ranged in age from 30-70 years old. Two consecutive thinning treatments
3999 took place on a portion of each site, while the other portions were designated as an unthinned
4000 control. Treated sites featured one of four buffer width prescriptions: (1) ~ 70-m buffer
4001 representative of one site potential tree, (2) ~15-m buffer, (3) a 6-m buffer representative of trees
4002 immediately adjacent to the stream. Wood surveys were carried out at four times during the
4003 study: (1) prior to the first thinning, (2) five years after the first thinning, (3) 9-13 years after the
4004 first thinning and just prior to the second thinning, and (4) one year after the second thinning. At
4005 each site, the first thinning was to 200 trees per ha (tph), the second thinning (~10 years later)
4006 was to ~85 tph, alongside an unthinned reference stand ~400 tph. Spatial and geomorphic
4007 characterization were measured using a combination of field and geospatial data. Hierarchical
4008 linear mixed models were developed with repeated measures using a multi-step process to
4009 examine relationships between large wood volume in headwater streams over time and in-stream
4010 wood characteristics (decay stage, zone), buffer width, time since thinning, and reach and
4011 geomorphology (drainage basin area, width:depth ratio, gradient). Wood volume was found to
4012 increase exponentially with drainage basin area; for every 1-ha increase in drainage basin area,
4013 wood volume increased by 0.63%. Slightly higher volumes of wood were found in sites with a
4014 narrow 6-m buffer, as compared with the 15-m and 70-m buffer sites in the beginning 5 years
4015 after the first harvest and maintained through year 1 of the second harvest (end of study). The
4016 authors attributed this difference to a higher likelihood of logging debris and/or windthrow but
4017 was not analyzed. Low volumes of wood from stands in the stem-exclusion phase were found to
4018 contribute to overall in-stream wood. The results showed that between 82-85% of the wood with
4019 discernable sources (90% for wood in early stages of decay; 45% of wood in late stages of
4020 decay) came from within 15 m of the stream, and the relative contribution of wood to streams

4021 declined rapidly with increasing distance. The authors hypothesize that this finding in
4022 conjunction with their results, which show a positive relationship between basin area and wood
4023 volume suggests a greater role for other large wood recruitment processes such as creep,
4024 landslides, and debris flow.

4025

4026 **Sediment**

4027

4028 Bywater-Reyes et al., 2018

4029

4030 Bywater-Reyes, S., Bladon, K.D., Segura, C., 2018. Relative Influence of Landscape Variables
4031 and Discharge on Suspended Sediment Yields in Temperate Mountain Catchments. *Water*
4032 *Resources Research* 54, 5126–5142. 10.1029/2017WR021728

4033

4034 The purpose of this paper was to improve our ability to predict suspended sediment yields by
4035 quantifying how sediment yields vary with catchment lithography and physiography, discharge,
4036 and disturbance history. This study took place at the HJ. Andrews Experimental Site in the
4037 Western Cascade Range of Oregon. The questions this paper sought to answer were (1) What is
4038 the relative association between discharge and catchment setting (i.e., lithology and
4039 physiography) and suspended sediment yields over an ~60-year period? (2) Is there an
4040 association between historical forest management activities (i.e., forest harvesting and road
4041 building) or extreme hydrologic events and the spatial and temporal trends in suspended
4042 sediment yield? Data was collected from 10 catchments, 8 within the Lookout Creek Watershed,
4043 1 just below the Lookout Creek Watershed, and 1 that drains to the adjacent Blue River. The data
4044 set spanned a 60-year period from 1955-2015. Methods for determining suspended sediment
4045 concentration involved using either vertically integrated storm-based grab samples, or discharge-
4046 proportional composite samples where composite samples were collected every three weeks at
4047 the outlet of each catchment. A linear mixed effects model (log transformed to meet the
4048 normality assumption) was used to predict annual sediment yield. In this model, site was treated
4049 as a random effect while discharge and physiographic variables were treated as fixed variables.
4050 This allowed for the evaluation of the relationships between sediment yield and physiographic
4051 features (slope, elevation, roughness, and index of sediment connectivity) while accounting for
4052 site. To account for the effect of disturbance history a variable was added to the model when the
4053 watershed had a history of management or natural disturbances. If the models for the disturbed
4054 watersheds significantly underpredicted the sediment discharge, the timing of the sudden
4055 increases were further examined to assess whether it correlated with a disturbance event. Last,
4056 the authors considered changes in stage derived from comparing measured historic stage values
4057 to those predicted from current rating curves. Changes in stage were interpreted as a relative bed-
4058 elevation change resulting from changes in scour and deposition of material likely moved as
4059 bedload. The results of this study show that sediment yield varied greatly across space and time

4060 with the lowest annual yield occurring in 2001 (~0.2 t/km²) at one catchment, and the highest
4061 annual yield (~953 t/km²) occurring in 1969 at another catchment. Annual suspended sediment
4062 yield was most strongly correlated with the standard deviation of watershed slope (r=0.72), Only
4063 moderately correlated with slope (r = 0.32), and with drainage area (r = 0.38). Standard deviation
4064 of slope was also strongly correlated with TPI (a surface roughness index), and standard
4065 deviation of index of connectivity. When considering disturbance, the largest magnitude changes
4066 in bed-elevation (I.e., sediment movement), were after floods with a ≥ 30-year return interval.
4067 The authors conclude that variability in watershed slope was the best predictor of annual
4068 suspended sediment yield relative to other physiographic variables. The authors report that the
4069 variability in watershed slope combined with cumulative annual discharge explained 67% of the
4070 variation in annual sediment yield across the 60-year data set. The results, however, show that
4071 annual sediment yields also moderately correlated with many other physiographic variables and
4072 caution that the strong relationship with watershed slope variability is likely a proxy for many
4073 processes, encompassing multiple catchment characteristics. For example, the strong relationship
4074 between watershed slope standard deviation and surface roughness. For the relationships
4075 between disturbance and sediment yield the authors conclude that the few anomalous years of
4076 high sediment yield occurred in watersheds with high slope variability and within a decade of
4077 forest management and a large flood event. The authors further caution that the high variability
4078 of sediment yield over space and time indicate that the factors tested in this study should be
4079 tested more broadly to investigate their utility to forest managers.

4080

4081

4082 **LW, Wildfire**

4083

4084 Chen et al., 2005

4085

4086 Chen, X., Wei, X., Scherer, R., 2005. Influence of wildfire and harvest on biomass, carbon pool,
4087 and decomposition of large woody debris in forested streams of southern interior British
4088 Columbia. *Forest Ecology and Management* 208, 101–114. doi:10.1016/j.foreco.2004.11.018

4089

4090 The purpose of this study was to compare the components of in-stream LW features between
4091 wildfire and forest harvesting disturbances. This study focuses particularly on the change in
4092 biomass and carbon pool among LW under different disturbances. This study was located in the
4093 central Okanagan Valley, Kelowna, British Columbia. A total of 19 forest streams, first and
4094 second order, within the study area were divided into four categories based on disturbance
4095 history of the adjacent upland forest and included: (1) riparian forest harvested 10 years ago; (2)
4096 riparian forest harvested 30 years ago; (3) riparian forest burnt ~ 40 years ago; and (4)
4097 undisturbed old-growth riparian forests that had a mean forest age of 163 years.. All harvested
4098 streams were clear-cut to the stream edge. New trees had established on these sites within 1-3

4099 years of harvest (planted or natural growth) and resulted in lodgepole pine being the dominant
4100 species. The wildfire streams included those that had been burnt ~40 years ago with no post-fire
4101 harvest or salvage logging. In stream LW was recorded for analysis if it had a minimum diameter
4102 of 10 cm and length of 1.0 m and were situated within the bankfull width. LW biomass was
4103 determined through the conversion of wood density and wood volume. LW was also categorized
4104 by decay class (3 classes), species, orientation submergence, and distance from the beginning of
4105 the study reach. Sampling took place during the period between July and October 2003 along a
4106 150 m study reach for each stream. An analysis of variance was used to determine the
4107 relationships between the chosen variables. When significant differences were found, the data
4108 was further analyzed with the data was fitted with a linear regression model to obtain
4109 correlations between the three variables (volume, biomass, and carbon). Results from this study
4110 show that on average the riparian sites disturbed by wildfire had the highest biomass, volume,
4111 and carbon content for individual LW pieces, followed by the 10-year harvest, then the old-
4112 growth forest; the 30-year harvest had the lowest of all streams for all parameters. Mean LW
4113 biomass of each individual piece of wood was significantly higher in sites which had been
4114 burned than in harvested sites. Biomass values were, on average, 31 kg in the wildfire sites,
4115 compared to 21 kg and 19 kg for sites harvested 10 years ago and 30 years ago, respectively. The
4116 volume of individual pieces in wildfire sites was significantly higher than in old-growth sites,
4117 and nearly significantly higher than in sites harvested 30 years ago. No statistical significance
4118 was found comparing piece volume in wildfire sites to sites harvested 10 years ago. The average
4119 carbon content of individual pieces of wood was also highest in the wildfire sites but the
4120 differences were not significant. The authors present data that the LW found in the wildfire and
4121 30-year harvest sites was mostly in the third decay class (most decayed), with less than 1% of
4122 LW in the class 1 decay class. Statistical significance was not discussed in the results for
4123 differences in decay class. The authors conclude that streams adjacent to wildfire disturbed and
4124 recently harvested (10-years post-harvest) forests contained significantly higher LW individual
4125 pieces and total volume than old-growth and 30-year post-harvest sites. Further because biomass,
4126 volume, and carbon were significantly higher in the 10-year post harvest sites, but there was no
4127 difference in the 30-year post-harvest sites and the old-growth sites; the authors speculate that
4128 harvest can increase the abundance of LW in the short-term from leaving harvest residues but
4129 reduces the abundance of LW over the long-term (~30 years post) due to a lack of recruitment
4130 from the young forests, and loss of in-stream LW from decomposition. The three main takeaways
4131 presented by the authors for this paper were (1) LWD input in old growth forested streams was
4132 relatively stable, (2) timber harvesting activities would cause a short-term increase of LWD
4133 stocks and might greatly reduce LWD loadings over a long-term, and (3) wildfire disturbance
4134 would delay LWD recruitment because not all burnt trees would fall in the stream immediately
4135 after the wildfire.

4136

4137

4138 **LW**

4139

4140 Chen et al., 2006

4141

4142 Chen, X., Wei, X., Scherer, R., Luider, C., Darlington, W., 2006. A watershed scale assessment of
4143 in-stream large woody debris patterns in the southern interior of British Columbia. *Forest*
4144 *Ecology and Management* 229, 50–62. <https://doi.org/10.1016/j.foreco.2006.03.010>

4145

4146 The purpose of this study was to (1) determine the spatial distribution and variation of LW
4147 characteristics (size, amount, volume, mass, orientation, position) within different order streams
4148 of forested watersheds; (2) to examine the relationship between LW characteristics and stream
4149 features through channel networks; and (3) to estimate the total density, volume and mass of LW
4150 at the watershed scale using a combination of field surveys and GIS data. This study took place
4151 at three different watersheds located in the south-central interior of British Columbia near
4152 Kelowna. A total of 35 study reaches with stream orders ranging from first- through fifth-order
4153 were selected to measure spatial distribution and variability of LW characteristics. Data collected
4154 for each reach was binned into 4 stream size categories (I = first order; II = second to third order;
4155 III = third to fourth order; IV = fourth to fifth order). Study sites were selected based on the
4156 following criteria. (1) the streams were in areas of intact mature riparian forests (>80 years); (2)
4157 the stream side forests were not disturbed by human activities, such as harvesting, road building;
4158 (3) the streams were not salvaged. Therefore, the results from this study provide a baseline of
4159 LWD characteristics in intact mature riparian forests in the southern interior of British Columbia.
4160 LW in this study is defined as having a diameter of > 0.1 m and a length > 1.0 m. LW
4161 characteristics (decay class, orientation, position within channel, distance from downstream end
4162 of channel) were recorded for any piece of LW that was within or above the bankfull width of the
4163 channel. Watershed features and the distribution of stream orders were derived from remotely
4164 sensed data. Mean values of LW density, volume, and biomass were compared between stream
4165 size classes with an analysis of variance (ANOVA). Results from this study show that LW size,
4166 volume, and biomass generally increased with increasing stream size. For example, the mean
4167 LWD diameter in stream size I (16.4 cm) was lower than that in stream size III (20.6 cm) and
4168 size IV (20.5 cm), respectively. Mean LW length also increases with stream size from 2.3 m in
4169 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
4170 volume (0.18 m³), significantly higher than stream size I (0.06 m³). LW volume was also
4171 significantly lower than in stream sizes II, and III. LW density (pieces per 100 m² of stream
4172 area), however, decreased as stream size increased. For example, LW density (defined as piece
4173 numbers per 100 m²) numbers were 19, 17, 12, and 4 for stream size I, II, III, and IV
4174 respectively. Increases in channel bankfull width ($R^2 = 0.52$) and stream area ($R^2 = 0.58$) was
4175 found to be strongly inversely correlated with LW density. Taken together, this study shows that
4176 spatial variation and distribution of LW characteristics vary as a function of stream size. From
4177 their results the authors conclude that in small sized streams, LW exhibit high density (number of
4178 pieces per 100 m²), low volume and biomass per unit area of stream. While in large sized
4179 streams, LW number, volume and biomass per unit of stream area are low but mean individual
4180 LW size was high.

4181

4182 **Stream Temperature Response to Harvesting**

4183

4184 Gravelle & Link, 2007

4185

4186 Gravelle, J.A., Link, T., 2007. Influence of Timber Harvesting on Headwater Peak Stream
4187 Temperatures in a Northern Idaho Watershed. *Forest Science* 53, 189–205.

4188

4189 The purpose of this study was to examine the effects of clearcutting and partial cutting on
4190 summer peak water temperatures in downstream fish-bearing streams, and to measure direct
4191 harvesting impacts on peak water temperature within headwater catchments. This study took
4192 place at the Mica Creek Experimental Watershed in Northern Idaho. Three headwater drainages
4193 were used to assess harvesting impacts on stream temperatures: (1) Watershed 1 which had 50%
4194 of the drainage area clearcut in 2001; (2) Watershed 2 which was thinned to a 50% target shade
4195 removal in Fall 2001; (3) and an unimpacted control. Riparian buffers were applied adjacent to
4196 the streams under the Idaho Forest Practices Act. This means, for fish-bearing streams the
4197 riparian management area must be at least 75 ft (22.9 m) wide on each side of the ordinary high-
4198 water mark (definable bank). Harvesting is still permitted, but there is a restriction where 75% of
4199 existing shade must be left. There are also leave tree requirements, which is a target number of
4200 trees per 1,000 linear feet (305 m), depending on stream width. For non-fish-bearing streams
4201 there is a 30 ft (9.1 m) equipment exclusion zone on each side of the ordinary high-water mark
4202 (definable bank). There are no shade requirements and no leave tree requirements, but skidding
4203 logs in or through streams is prohibited. Stream temperature data and canopy cover percentage
4204 data were collected at multiple sites within and downstream of treatment areas between 1992-
4205 2005. However, for the non-fish-bearing, headwater sites pre-treatment data was only collected
4206 one season prior to treatment. Temperature data was summarized as maximum daily temperature
4207 and was analyzed using simple linear regression to estimate changes in stream temperature
4208 following harvest during the summer months (July 1 – September 1). Results from this study
4209 show that there is no strong evidence of a posttreatment increase in stream temperature at long-
4210 term downstream sampling points for each harvest treatment. In general, the downstream sites
4211 showed a cooling effect between -0.2 and -0.3°C . The estimated cooling effect could not be
4212 attributed to any cause (e.g., increase in water yield), but the authors conclude that there was no
4213 post-harvest increase in peak summer temperatures at the downstream sites. For streams
4214 immediately adjacent to the clearcut treatment (headwater streams) a significant increase in
4215 temperature was detected at 2 sites ranging between 0.4 and 1.9°C , while a marginally
4216 significant decrease in temperature was detected at the third site (-0.1°C , $p = 0.06$). At the sites
4217 located immediately adjacent to partial cuts, results showed mixed results with decreases in
4218 temperature (-0.1°C ; non-significant) at one site and significant but minimal changes at another
4219 site (0.0 - 3.0°C) across the individual post-harvest years. Overall, there were minimal to no
4220 changes in stream peak temperatures following treatment in the partial-cut riparian areas. The

4221 authors go on to point out that headwater stream temperatures were highly variable, and that the
4222 shade value of understory vegetation may be an important factor contributing to results.

4223

4224 **SED**

4225

4226 Bywater-Reyes et al., 2017

4227

4228 Bywater-Reyes, S., Segura, C., Bladon, K.D., 2017. Geology and geomorphology control
4229 suspended sediment yield and modulate increases following timber harvest in temperate
4230 headwater streams. *Journal of Hydrology* 548, 754–769.
4231 <https://doi.org/10.1016/j.jhydrol.2017.03.048>

4232

4233 The purpose of this study was to assess the influence of natural controls (basin lithology and
4234 physiography) and forest management on suspended sediment yields in temperate headwater
4235 catchments. The study sought to achieve three objectives: (1) Quantify how suspended sediment
4236 yield varies by catchment setting in forested headwater catchments, (2) Determine whether
4237 contemporary forest management practices impact annual suspended sediment yield (SSY) in
4238 forested headwater catchments (3) Determine whether there are natural catchment settings that
4239 result in different levels of vulnerability or resilience to increases in suspended sediment yield
4240 associated with disturbances (e.g., harvest activities). This study analyzed 6 years of data from
4241 the Trask River Watershed in Northeastern Oregon and included data from harvested and
4242 unharvested sub-catchments underlain by heterogenous lithologies. Baseline SSY data collection
4243 began in water year 2010 and continued through water year 2015, with road upgrades (July–
4244 August 2011) and harvest (May–November 2012) occurring in the middle of the study period.
4245 Generalized least square candidate models quantifying the parameters from each site were used
4246 to test differences in the relationship between suspended sediment yield and catchment setting.
4247 Results from this study indicate that site lithology was a first order control over SSY with SSY
4248 varying by an order of magnitude across lithologies observed. Specifically, SSY was greater in
4249 catchments underlain by Siletz Volcanics ($r = 0.6$), the Trask River Formation ($r = 0.4$), and
4250 landslide deposits ($r = 0.9$) and displayed an exponential relationship when plotted against
4251 percent watershed area underlain by these lithologies, combined. In contrast, the site effect had a
4252 strong negative correlation with percent area underlain by diabase ($r = 0.7$), with the lowest SSY
4253 associated with 100% diabase independent of whether or not earthflow terrain was present.
4254 Following timber harvest (water year 2013), increases in SSY occurred in all harvested
4255 catchments. The SSY in both PH4 (clearcut with buffers) and GC3 (clearcut without buffers)
4256 declined to pre-harvest levels by water year 2014. Interestingly, the SSY in UM2 (clearcut
4257 without buffers) increased annually throughout the post-harvest period, ultimately resulting in
4258 the highest SSY of all catchments during the final two years of the study after producing the
4259 lowest SSY in the pre-harvest period. Catchment physiographic variables (hypsoetry, slope,

4260 standardized topographic position index (SD TPI), and sediment connectivity (IC)) appeared to
4261 be good indicators of the underlying lithology of each site. Principle component analysis
4262 constructed from physiographic variables separated sites underlain by resistant diabase from
4263 those underlain by mixed lithologies along the PC1 axis. While sites along the second axis (PC2)
4264 were separated by relative values of earthflow terrain (high proportion vs. Little to none). Sites
4265 with low SSY and underlain by more resistant lithologies were also resistant to harvest-related
4266 increases in SSY. The authors conclude that sites underlain with a friable lithology (e.g.,
4267 sedimentary formations) had SSYs an order of magnitude higher, on average, following harvest
4268 than those on more resistant lithologies (intrusive rocks). In general, sites with higher SSY also
4269 had 1) lower mean elevation and slope, 2) greater landscape roughness, and 3) lower sediment
4270 connectivity (potential for sediment transport based on physiography). The authors suggest that
4271 their research be undertaken in different regions with different disturbance types to broadly apply
4272 their findings.

4273

4274 **Plant Communities**

4275

4276 D'Souza et al., 2012

4277

4278 D'Souza, L.E., Six, L.J., Bakker, J.D., Bilby, R.E., 2012. Spatial and temporal patterns of plant
4279 communities near small mountain streams in managed forests. *Can. J. For. Res.* 42, 260–271.
4280 <https://doi.org/10.1139/x11-17>

4281

4282 The purpose of this study was to examine spatial and temporal patterns in plant communities
4283 along fish-bearing streams in western Washington. The focus of this study is on areas which were
4284 harvested to the streambank within the last 100 years. The study took place in the western
4285 Cascade Mountains of Washington. Sites were randomly selected using a geographic information
4286 system. Stands that had been impacted by road development were excluded. Stands were
4287 stratified into a chronosequence of age classes: young (31-51 years), mature (52-70 years), old
4288 (>100 years). Due to availability, the sample sizes included 11 young stands, 10 mature stands,
4289 but only 4 old stands. Vegetation characteristics were captured in each stand using 0.16 ha plots
4290 located 30 m from stand edges to limit the influence of adjacent stands. Transects perpendicular
4291 to the stream were used 10 m apart and extended 80 m upslope. Vegetation and physical features
4292 along each transect were sampled using a series of subplots at 10 m intervals from the channel.
4293 The authors found little variation in riparian landform type and or canopy cover and were not
4294 included in the analysis for their effect on vegetation. Plant communities were examined
4295 spatially as a function of distance to stream and temporally by using the chronosequence of stand
4296 ages. Three distinct plant communities were observed in the shrub and herb layer (riparian: 0-9
4297 m; transitional: 10-29 m; and upslope: 30-80 m) and their composition differed significantly
4298 between communities. A total of 12 species were identified as indicators of these communities.

4299 For the shrub layer, community composition differed between old stands and young and mature
4300 stands. In the herb layer, community composition differed between all age classes. The results
4301 from this study suggest that plant communities along small fish-bearing streams have distinct
4302 changes in community with distance to stream, but also reflect successional status in nearby
4303 forests. The authors conclude by suggesting increased research in understanding the effects of
4304 forest management on streamside vegetation.

4305

4306 **LW Residence Time**

4307

4308 Hyatt & Naiman, 2001

4309

4310 Hyatt, T.L., Naiman, R.J., 2001. The Residence Time of Large Woody Debris in the Queets
4311 River, Washington, Usa. *Ecological Applications* 11, 191–202. [https://doi.org/10.1890/1051-
4312 0761\(2001\)011\[0191:TRTOLW\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2001)011[0191:TRTOLW]2.0.CO;2)

4313

4314 The purpose of this study was to determine the depletion rate of LW by examining differences in
4315 size and species composition in the Queets River compared to the adjacent forest. This study
4316 took place in the Queets River Watershed located on the west slope of the Olympic Mountains in
4317 Washington. Field sampling was carried out at 25 transects and four different sites. Increment
4318 cores from in-stream LW were cross-dated against cores from riparian conifers to estimate the
4319 time which LW was recruited into the channel. LW pieces which were in a heightened state of
4320 decay were dated using carbon-dating techniques. the most common tree species (> 30 cm
4321 diameter) in the riparian zone is red alder, followed by Sitka spruce and western hemlock,
4322 whereas the most common species of LWD (> 30 cm diameter) is Sitka spruce, followed by red
4323 alder and western hemlock. Each of the hardwood species is better represented among standing
4324 trees than among LWD, and each of the conifers are better represented as LW than among trees
4325 in the riparian zone. The depletion curve developed in the results was based only on conifer LW
4326 because hardwood LW was either too small or too young to provide accurate estimates of
4327 residence time in the stream. Based on the depletion curve developed for all available LW
4328 showed that wood typically disappears from the active channel within the first 50 years, while
4329 some pieces may remain for several hundred years. By cross-referencing the LW depletion
4330 curves with field notes the authors suggest that the longer residence time, beyond 50 years, was
4331 dependent on more than one process such as burial. Decay class was not an accurate predictor of
4332 LW age. Also, Dependent vegetation on or around LWD was a poor and often misleading
4333 indicator of residence time. Many LWD pieces that had 1–5 year old vegetation growing on

4334 or around them were discovered to have died and presumably recruited to the channel 20 years
4335 previous. The authors conclude that LW originating from hardwoods is depleted faster than
4336 conifers. Considering the depletion rate curve, the authors speculate that the majority of LW is
4337 transported out of the system within 50 years, while pieces of LW that are buried or jammed in

4338 the river floodplain may remain for hundreds of years. Overall, ~80% of LW residing in the
4339 active channel were living within 50 years of the study. The authors explain there are several
4340 caveats to the depletion curve created for this study (1) the depletion constant was developed for
4341 a large, mostly alluvial river and should probably not be applied to smaller streams (mean
4342 bankfull width at study transects on the Queets is 165 m and the range is 51–398 m; mean key
4343 LWD length is 23.4 m, and the range is 5.3–69.0 m). Also, from the data the authors infer that
4344 alluvial channel trap wood from upstream, and constrained channels export LWD downstream,
4345 so it is not to be expected that the LWD resident in a channel was recruited from the riparian
4346 zone in that reach. In general, the authors conclude that for this study the depletion curve shows
4347 that the half-life of LW is ~20 years and thus all resident LW will be exported, buried, or broken
4348 down within 3-5 decades. Also, hardwood LW will be depleted from the channel more rapidly
4349 than conifers.

4350

4351 **Litter Input**

4352

4353 Hart et al., 2013

4354

4355 Hart, S.K., Hibbs, D.E., Perakis, S.S., 2013. Riparian litter inputs to streams in the central
4356 Oregon Coast Range. *Freshwater Science* 32, 343–358. <https://doi.org/10.1899/12-074.1>

4357

4358 The purpose of this study was to understand how riparian vegetation composition, understory
4359 density, and topography affect the quantity and quality of litter input to streams throughout the
4360 annual cycle. This study took place within 5 contiguous watersheds located in the central Coast
4361 Range of Oregon. At each of the study sites uniform areas along a ≤ 300 m stream reach, 3 plots
4362 were delineated on 1 side of the stream, each 8x 25 m along the stream. Three treatments were
4363 applied: (1) a no cut or fence control; (2) cut and remove a 5 x 8 m section adjacent to stream
4364 plants < 10 cm DBH and >12 cm height every 2 months; and (3) 5 m fence extending
4365 underground and parallel to the stream to block litter moving downslope from reaching stream.
4366 Vertical and lateral litter traps were installed at each site and collected monthly between August
4367 2003-August 2004. Variation of riparian vegetation and woody debris characteristics were
4368 analyzed with a 3-way ANOVA using overstory, treatments, and sections and their interactions.
4369 Two-way ANOVA with repeated measures was used to compare seasonal and monthly control
4370 and treatment inputs for different overstory and litter types. 1-way ANOVA was used to test for
4371 differences in nutrient concentration flux between overstory type. Results from this study show
4372 that deciduous forests dominated by red alder delivered significantly greater vertical and lateral
4373 inputs to stream than did coniferous forests dominated by Douglas-fir. Deciduous-site vertical
4374 litter input (mean, 95% CI; 504 g m⁻¹ y⁻¹, 446.6–561.9) exceeded that from coniferous sites
4375 (394 g m⁻¹ y⁻¹, 336.4–451.7) by 110 g/m² (28.6–191.6) over the full year. Annual lateral inputs
4376 at deciduous sites (109 g m⁻¹ y⁻¹, 75.6–143.3) were 46 g/m (1.2– 94.5) more than at coniferous

4377 sites (63 g m⁻¹ y⁻¹, 28.9–96.6). Lateral inputs calculated for a 3-m-wide stream accounted for
4378 9.6% (5.4–12.5) of total annual inputs at coniferous sites and 12.7% (10.2–14.5) of total inputs at
4379 deciduous sites. Composition of litter also differed significantly by overstory type. Annual lateral
4380 inputs at coniferous sites were dominated by deciduous leaves (.33%), twigs (.23%), and leftover
4381 (.18%) litter types, whereas annual lateral inputs at deciduous sites were deciduous leaves (.61%)
4382 and leftover (.15%) litter types. Leftover litter types were defined as those that were too small or
4383 decayed to identify, bark, moss, or lichens. Vertical litter inputs at deciduous sites were
4384 dominated by deciduous leaves (.65%) and deciduous-other (.15%) litter types. While deciduous
4385 leaves (.33%), coniferous needles (.24%), and twigs (.21%) composed the annual vertical litter
4386 inputs at coniferous sites. The strongest deciduous inputs to streams occurred in November.
4387 Annual lateral litter input increased with slope at deciduous sites ($R^2 = 0.4073$, $p = 0.0771$), but
4388 showed no strong relationship at coniferous sites ($R^2 = 0.1863$, $p = 0.2855$). Total nitrogen flux
4389 to streams at deciduous sites was twice as much as recorded at coniferous sites. However, there
4390 was seasonal effect where the N fluxes in deciduous sites was only higher in autumn. The
4391 authors of this study conclude by suggesting management in riparian areas consider utilizing
4392 deciduous species such as red alder for greater total N input to aquatic and terrestrial ecosystems
4393 along with the increased shade and large woody debris provided by coniferous species.

4394

4395 **Effect of Contemporary Management on Nutrient Concentration and Cycling**

4396

4397 Gravelle et al., 2009

4398

4399 Gravelle, J.A., Ice, G., Link, T.E., Cook, D.L., 2009. Nutrient concentration dynamics in an
4400 inland Pacific Northwest watershed before and after timber harvest. *Forest Ecology and*
4401 *Management* 257, 1663–1675. <https://doi.org/10.1016/j.foreco.2009.01.017>

4402

4403 The purpose of this study was to assess the effects of contemporary forest harvesting practices on
4404 nutrient cycling and concentrations. This study took place at the Mica Creek Experimental
4405 Watershed in Northern Idaho. Seven steel Parshall flumes were installed at select locations
4406 within the watershed to assess the effects of clearcut to stream and partial cut (50% shade
4407 retention) harvesting practices. All harvesting was conducted in compliance with the Idaho
4408 Forest Practices Act. Within fish-bearing streams (Class I) Harvesting is permitted, but 75% of
4409 existing shade must be retained. There are also leave tree requirements for a target number of
4410 trees per 1000 linear feet (305 m), depending on stream width. In Mica Creek, this was roughly
4411 200 trees in the 3–12 in. (8–30 cm) diameter class per 305 m of the riparian management zone
4412 (RMZ). Along non-fish-bearing streams (Class II) the RMZ is 30 feet (9.1 m) of equipment
4413 exclusion zone on each side of the ordinary high-water mark (definable bank); skidding logs in
4414 or through streams is prohibited. There are no shade requirements and no requirements to leave
4415 merchantable trees. Two-sided riparian buffers were left on all Class I streams during harvest

4416 operations. Timber was removed from both sides of the Class II streams. In the post-harvest and
4417 post-burn conditions, Class II streams in clearcut treatments had only a small amount of green
4418 tree retention within the riparian zone, while in partial cut treatments equal amounts of canopy
4419 cover (approximately 50%) were removed from both sides of the stream. This study followed the
4420 BACI design and featured a pre-treatment measurement phase (1992-1997), a post-road
4421 construction phase (1997-2001), and a post-harvest phase (2001-2006). A student's t-test was
4422 used to analyze the data between the observed and predicted values of post-treatment sites for
4423 several nitrogen and phosphorus compound concentrations (Kjeldahl nitrogen (TKN), nitrate +
4424 nitrite (NO₃ + NO₂), TP, total ammonia nitrogen (TAN) consisting of unionized (NH₃) and
4425 ionized (NH₄⁺) ammonia, and unfiltered orthophosphate (OP) samples). Results from the post-
4426 road construction period showed no significant changes in concentrations of any nutrients
4427 analyzed. Results from this study show statistically significant increases in NO₃ and NO₂
4428 concentrations following clearcut and partial harvest cuts in headwater streams. Increases at the
4429 clearcut treatment site were greatest, where mean monthly concentrations increased from 0.06
4430 mg-N L⁻¹ during the calibration and post-road periods to 0.35 mg-N L⁻¹. There was also an
4431 observable seasonal effect on NO₃ + NO₂ concentrations with the peak concentration of 0.89
4432 mg-N L⁻¹ occurred at F1 in April 2004, with mean monthly concentrations of 0.43 mg-N L⁻¹
4433 and 0.59 mg-N L⁻¹ in water years (October–September) 2004 and 2005, respectively. Similar
4434 results were also observed at sites further downstream although changes were smaller which, the
4435 authors point out this may be due to in-stream uptake and/or dilution. No significant changes of
4436 in-stream concentration of any other nutrient recorded were found between time periods and
4437 treatments except for one downstream site that showed a small increase in orthophosphate by
4438 0.01 mg P L⁻¹. In general, the results of this study show that forest management influences in-
4439 stream NO₃ + NO₂ immediately adjacent to treatment and downstream of treatment. The authors
4440 conclude by suggesting future research in understanding variability in nutrient concentrations
4441 and cycling as affected by seasons and storm runoff events.

4442

4443 **Organic Matter Inputs**

4444

4445 Kiffney & Richardson, 2010

4446

4447 Kiffney, P.M., Richardson, J.S., 2010. Organic matter inputs into headwater streams of
4448 southwestern British Columbia as a function of riparian reserves and time since harvesting.
4449 *Forest Ecology and Management* 260, 1931–1942. <https://doi.org/10.1016/j.foreco.2010.08.016>

4450

4451 The purpose of this paper was to assess how differences in riparian buffer width and timing since
4452 harvest affect terrestrial particulate organic matter flux into streams. The focus of this paper was
4453 on 1st and 2nd order headwater streams located approximately 45 km east of Vancouver in
4454 British Columbia, Canada. Sites were measured over an 8-year period and included clear-cut

4455 (n=3), 10-m buffered reserve (n=3), 30-m buffered reserve (n=2), and uncut control (n=2)
4456 treatments. For streams receiving a 10 or 30-m reserve, there was no logging on either side of the
4457 stream within these reserves. Study reaches were approximately 200m long. Vertical litter inputs
4458 were collected monthly and at approximately 6–8-week intervals during each season for years
4459 1,2,6,7, and 8 years after harvest. Litter was separated into broadleaf deciduous, twig, needles,
4460 and other (seeds, cones, and moss) categories following collection and subsequently dried and
4461 weighed using a microbalance. A mixed-model analysis of covariance was used for Fall data
4462 with riparian treatment as a fixed effect and year as a covariate. Secondly, ordinary least
4463 squares regression was used to quantify the functional relationship between reserve width and
4464 litter flux within each year. Results show riparian treatments having significant effects on the
4465 quantity and composition of litter input into streams. Inputs consisting of needles and twigs were
4466 significantly lower while deciduous inputs were higher in clearcuts compared to other
4467 treatments. Differences in litter flux relative to riparian treatment persisted through year 7, while
4468 a positive trend between reserve width and litter flux remained through year 8. For example, one-
4469 year post-treatment, needle inputs were 56x higher during the Fall into control and buffered
4470 treatments than into the clearcut. Needle inputs remained 6x higher in the buffer and control sites
4471 through year 7, and 3-6x higher in year 8 than in the clearcut sites. Twig inputs into the control
4472 and buffered sites were ~25x higher than in the clearcut sites in the first year after treatment.
4473 There was no significant difference in treatment for deciduous litter but a trend of increasing
4474 deciduous litter input in the clear cut was observed in the data. For example, one-year post-
4475 treatment deciduous litter was lowest in the clearcut, but by year 8 deciduous litter was highest in
4476 the clearcut sites relative to control and buffered sites. The linear relationship between reserve
4477 width and litter inputs was strongest in the first year after treatment, explaining ~57% of the
4478 variation, but the relationship could only explain ~17% of the variation in litter input by buffer
4479 width by year 8 (i.e., the relationship degraded over time). The authors interpret these results as
4480 evidence that riparian reserves showed a similar litter flux to streams when compared to uncut
4481 controls. They also conclude that litter flux from riparian plants to streams, was affected by
4482 riparian reserve width, time since logging, and potentially channel geomorphology.

4483

4484 **In-stream Wood Loads**

4485

4486 Jackson & Wohl, 2015

4487

4488 Jackson, K.J., Wohl, E., 2015. Instream wood loads in montane forest streams of the Colorado
4489 Front Range, USA. *Geomorphology* 234, 161–170.
4490 <http://dx.doi.org/10.1016/j.geomorph.2015.01.022>

4491

4492 The purpose of this study was to examine in-stream wood loads and geomorphic effects between
4493 stands of different ages and stands with different disturbance histories The first objective of this

4494 study was to determine whether instream wood and geomorphic effects differ significantly
4495 among old-growth, younger, healthy, and beetle-infested forest stands. The second objective of
4496 this study was to determine whether instream wood loads correlate with valley and channel
4497 characteristics. The authors hypothesized that streams in old-growth montane forests have (1)
4498 significantly larger in stream and floodplain wood loads than those in younger stands, (2) greater
4499 frequency of volume of jams than those in younger forests, and (3) more wood created
4500 geomorphic effects. They also hypothesized that instream wood loads in healthy montane forests
4501 are significantly smaller than in beetle-infested forests. Last, they hypothesized that instream
4502 wood load correlates with lateral valley confinement, with unconfined valleys having the greatest
4503 in-stream and total wood loads. This study took place within the Arapaho and Roosevelt National
4504 Forests in Colorado. Sediment storage, channel geometry, in-stream wood load, and forest stand
4505 characteristics were measured along 33 pool-riffle or plane-bed stream reaches (10 located in
4506 old-growth (> 200 years); 23 located in younger forests (age range not reported)). LW
4507 characteristics were recorded for all in-stream wood ≥ 10 cm diameter and ≥ 1 m in length. Pair-
4508 wise t-test or Kruskal-Wallis tests were used to check for significant differences in wood load,
4509 logjam volume, and logjam frequencies. To test for significant differences in wood created
4510 geomorphic effects a principal component analysis was used. Results indicated that channel
4511 wood load (OG = 304.4 ± 161.1 ; Y = 197.8 ± 245.5 m³/ha), floodplain wood load (OG = 109.4
4512 ± 80 ; Y = 47.1 ± 52.8 m³/ha), and total wood load (OG = 154.7 ± 64.1 ; Y = 87.8 ± 100.6 m³
4513 /ha) per 100 m length of stream and per unit surface area were significantly larger in streams of
4514 old-growth forests than in young forests. Streams in old-growth forests also had significantly
4515 more wood in jams, and more total wood jams per unit length of channel than in younger forests
4516 (jam wood volume: OG = 7.10 ± 6.9 m³; Y = 1.71 ± 2.81 m³). When standardized to stream
4517 gradient, old-growth streams had significantly greater pool volume and significantly greater
4518 sediment volume than younger stands. No significant difference was detected in in-stream wood
4519 loads between healthy and beetle-infested stands. Although wood load in streams draining from
4520 pine beetle infested forests did not differ significantly from healthy forests, best subset regression
4521 (following principal component analysis) indicated that elevation, stand age, and pine beetle
4522 infestation were the best predictors of wood load in channels and on floodplains. The authors
4523 speculate that beetle infestation is affecting in-stream wood, but perhaps not enough time has
4524 passed since the infestation for the affected trees to fall into the stream. Time since beetle-
4525 infestation was not reported.

4526

4527 **LW Recruitment**

4528

4529 May & Gresswell, 2003

4530

4531 May, C.L., Gresswell, R.E., 2003. Large wood recruitment and redistribution in headwater
4532 streams in the southern Oregon Coast Range, U.S.A. *Can. J. For. Res.* 33, 1352–1362.

4533 <https://doi.org/10.1139/x03-023>

4534

4535 The purpose of this study was to understand the relative influence of processes that recruit and
4536 redistribute wood into channels and to understand how these processes vary spatially. Specific
4537 research questions included the following: (i) Do processes that deliver and redistribute wood
4538 differ in small colluvial channels compared with larger alluvial channels? (ii) Do proximal and
4539 distal controls on wood delivery differ for colluvial and alluvial channels? (iii) How do input and
4540 redistribution processes influence the functional role of wood in the channel? The focus of this
4541 research is specifically on differences between small colluvial channels and large alluvial
4542 channels in the southern Oregon Coast Range. All downed wood exceeding 20 cm mean
4543 diameter and 2 m in length, and in contact with the bank-full channel were measured in three
4544 second order and one third-order stream. Large wood was categorized based on the various
4545 mechanisms delivering it to the stream channel. Categories included (i) direct delivery from local
4546 hillslopes and riparian areas, (ii) fluvial redistribution, (iii) debris flow transported, or (iv) an
4547 unidentified source. Results from this study show that stream size and topographic position
4548 strongly influence processes that recruit and redistribute wood in channels. Processes of slope
4549 instability were shown to be important conveyors of wood from upland forests to small colluvial
4550 channels. In the larger alluvial channels, windthrow was found to be the dominant recruitment
4551 process from adjacent riparian area. Results showed that Wood derived from local hillslopes and
4552 riparian areas accounted for the majority of pieces (63%) in small colluvial channels. The larger
4553 alluvial channel received wood from a greater variety of sources, including recruitment from
4554 local hillslopes and riparian areas (36%), fluvial redistribution (9%), and debris flow transported
4555 wood (33%). However, because pieces recruited from local sources (hillslope and riparian area)
4556 were larger, these sources of wood had a disproportionately large contribution to volume of wood
4557 in the stream. For example, wood recruited from the local hillslopes and riparian areas accounted
4558 for 36% of wood pieces in the alluvial stream, which accounted for 74% of the total volume of
4559 wood. Slope instability and windthrow were the dominant mechanisms for wood recruitment into
4560 small colluvial channels. Windthrow was the dominant recruitment mechanism for wood
4561 recruitment into larger alluvial channels. Distributions of the source distance of wood pieces
4562 were significantly different between colluvial and alluvial channels. In colluvial streams, 80% of
4563 total wood and 80% of total wood volume recruited originated from trees rooted within 50 m of
4564 the channel. In the alluvial channel, 80% of the pieces of wood and 50% of the total volume
4565 originated from trees which came from 30 m of the channel. The primary function of wood in
4566 smaller colluvial channels was sediment storage (40%) and small wood storage (20%). The
4567 primary function of wood in larger alluvial channels is bank scour (26%), stream bed scour
4568 (26%), and sediment storage (14%). Recruitment and redistribution processes were shown to
4569 affect the location of the piece relative to the channel/flow direction, thus influencing its
4570 functional role. The authors conclude that wood recruited from local sources is variable by
4571 position in the stream network because of differences in recruitment processes, degree of
4572 hillslope constriction, and slope steepness.

4573

4574 **Sediment**

4575

4576 Macdonald et al., 2003^a

4577

4578 Macdonald, J. S., Beaudry, P. G., MacIsaac, E. A., & Herunter, H. E. (2003). The effects of forest
4579 harvesting and best management practices on streamflow and suspended sediment concentrations
4580 during snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada.
4581 Canadian Journal of Forest Research, 33(8), 1397-1407. <https://doi.org/10.1139/x03-110>

4582

4583 (BACI, only single year pre-harvest)

4584

4585 This study investigates the changes in suspended sediment concentration and stream discharge
4586 during freshet (spring snowmelt) at two harvest intensities relative to each other and an
4587 unharvested control watershed, pre- and post-harvest. The design included three small sub-
4588 boreal, first order, forest streams (<1.5 m width) in the central interior of British Columbia
4589 (Baptiste watershed). Both treatment streams received a 55% harvest treatment; one (low-
4590 retention) removed all merchantable timber >15 cm DBH for pine and > 20 cm DBH for spruce
4591 within 20 m of the stream; the other treatment (high-retention) removed all merchantable timber
4592 > 30 cm within 20 m of the stream; and an un-harvested control. Data for stream flow and total
4593 suspended sediments (TSS) was collected using Parshall flumes downstream from the treatment
4594 and control sites for one-year pre- and four-years post-harvest during snowmelt periods.
4595 Regression analysis was used to analyze relationships between treatment and control reaches pre-
4596 and post-treatment to estimate and compare predicted changes in TSS. The results showed an
4597 increase in freshet discharge for both treatments above predicted values for the entirety of the
4598 study. During the year prior to treatment, TSS relationships of both treatment watersheds during
4599 freshet closely matched those of the control. Immediately following harvest TSS concentrations
4600 increased above predicted values for both treatment streams. Increased TSS persisted for two-
4601 years post-harvest in the high-retention treatment, and for 3-years in the low-retention. The
4602 authors speculate that the treatment areas may have accumulated more snow (e.g., more exposed
4603 area below canopy) than in the control reaches leading to the increase in discharge. This study
4604 shows evidence that harvest intensity (low vs. high retention) is proportional to the increase in
4605 stream discharge, TSS, and recovery time to pre-harvest levels.

4606

4607 **LW**

4608

4609 Fox & Bolton, 2007

4610

4611 Fox, M., & Bolton, S. (2007). A regional and geomorphic reference for quantities and volumes of
4612 instream wood in unmanaged forested basins of Washington State. *North American Journal of*
4613 *Fisheries Management*, 27(1), 342-359. <https://doi.org/10.1577/M05-024.1>

4614

4615 This study uses in-stream LW values from 150 stream segments located in unmanaged
4616 watersheds, across all of Washington State, to investigate the relationships between
4617 geomorphology, forest zone, and disturbance regimes with LW recruitment. The purpose of this
4618 study was to create a base-line value of central tendency for in-stream LW values in “natural”
4619 streams for which salmonids are theoretically adapted. The authors define natural and
4620 unmanaged as streams that (1) had no part of the basin upstream of the survey site ever logged
4621 using forest practices common after European settlement and (2) the basin upstream of the
4622 survey site contains no roads or human modifications to the landscape that could affect the
4623 hydrology, slope stability, or other natural processes of wood recruitment and transport in
4624 streams. Sites were stratified to capture the variations in forest types, channel morphologies, and
4625 hydrological origins. The authors used descriptive statistics to establish and evaluate correlations
4626 between wood loading and watershed characteristics to reveal the highest valued variables
4627 influencing wood loading. Following this analysis, the variables with the highest mechanistic
4628 values in determining wood loading were evaluated and compared using simulation modeling.
4629 Results showed that in-stream wood volume increased with drainage area and as streams became
4630 less confined. However, bank full width (BFW) was a significantly better predictor of wood
4631 parameters than basin size. There was observational evidence that alluvial channels contained
4632 more wood volume on average than bedrock channels. However, due to limits in sample size
4633 following stratification, statistical analysis could not be completed. Sample sizes for isolating
4634 gradient and confinement were also too small to apply statistical analyses. Fire was found to
4635 influence in-stream wood quantities and volumes west of the Cascade crest; In-stream wood
4636 volume increased with adjacent riparian timber age as determined by the last stand replacing fire.
4637 Other disturbances such as debris flow, snow avalanche, and flooding were too few in frequency
4638 in the study area to be analyzed statistically. From these results the authors developed thresholds
4639 for expected “key piece volume (m^3)” (pieces with independent stability) of wood for three BFW
4640 classes (20-30 m, >30 – 50 m, > 50 m width) per 100 m stream length for streams with BFW
4641 greater than 20 m. From percentile distributions the authors recommend minimum volumes,
4642 defined by the 25th percentiles, of approximately 9.7 m³ for the 20- to 30-m BFW class, 10.5 m³
4643 for the 30- to 50-m BFW class, and 10.7 m³ for channels greater than 50 m BFW per 100 m
4644 length of stream. The results of this study suggest that BFW is the single greatest predictor of in-
4645 stream wood quantity and volume relative to other predictor variables. However, this result
4646 comes with the caveat that other processes and geomorphologies (e.g., channel bed form,
4647 gradient, confinement) are also important in the mechanisms for wood recruitment, modeling in
4648 this study showed too much inconsistency with these predictor variables to draw strong
4649 conclusions. Further the authors warn that these values for reference conditions are only
4650 applicable to streams with bank-full widths between 1 and 100 m, gradients between 0.1% and
4651 47%, elevations between 91 and 1,906 m, drainage areas between 0.4 and 325 km², glacial and
4652 rain- or snow-dominated origins, forest types common to the Pacific Northwest.

4653

4654 **LW and sediment**

4655

4656 Gomi et al., 2001

4657

4658 Gomi, T., Sidle, R. C., Bryant, M. D., & Woodsmith, R. D. (2001). The characteristics of woody
4659 debris and sediment distribution in headwater streams, southeastern Alaska. *Canadian Journal of*
4660 *Forest Research*, 31(8), 1386-1399. <https://doi.org/10.1139/x01-070>

4661

4662 This study investigated different riparian conditions related to harvest and disturbance
4663 (landslides), their influence on woody debris and sediment distributions, and their related
4664 functions in headwater streams. This study examined the effects of recent and past timber
4665 harvests on woody debris abundance and distribution, landslides and debris flow on woody
4666 debris abundance and sediment accumulations, and the function of in-stream woody debris on
4667 sediment storage. The researchers examined 15 steep headwater streams in the Maybeso
4668 Experimental Forest and Harris River basin in the Tongass National Forest, Prince of Wales
4669 Island, southeastern Alaska. Treatments of headwater streams included five management or
4670 disturbance regimes: old growth (OG), recent clear-cut (CC; 3 years), young growth conifer
4671 forest (YC; 37 years after clear-cut), young growth alder (YA; 30 years after clear-cut), and
4672 recent landslide and debris flow channels (LS). Three headwater streams were sampled for each
4673 of the 5 treatments, 15 streams total. Analysis of covariance (ANCOVA) was used to compare
4674 LW quantity and distribution, and sediment quantity and distribution, across plots nested within
4675 each treatment site. Results showed in-channel numbers of LW pieces were significantly higher
4676 in YC and CC sites when compared to OG, YA, and LS sites. The number of LW pieces was
4677 highest in YC streams even though logging concluded 3 decades prior to sampling. No
4678 significant differences in LW volume were found among OG, CC, and YC streams. However,
4679 LW volume per 100 m of stream length in YC was twice that in OG. The total volume of LW per
4680 100 m associated with CC channels was half that in OG channels. However, the majority of the
4681 LW volume in OG systems was outside of the bank-full area. When the data was stratified by
4682 channels that experienced landslides (LS and YA), the number of LW pieces among OG, YA, and
4683 LS was not statistically significant. However, the in-channel volumes of LW in LS and YA
4684 channels were significantly lower than in OG sites because individual LW pieces in the OG sites
4685 were relatively larger than in the LS and YA sites. There was high variability among sites in the
4686 amount of sediment stored within streams. The authors conclude that timber harvesting and
4687 related landslides and debris flows affect the distribution and accumulation of LW and related
4688 sediment accumulation in headwater streams. These effects are summarized as (i) inputs of
4689 logging slash and unmerchantable logs significantly increase the abundance of in-channel woody
4690 debris; (ii) in the absence of landslides or debris flows, these woody materials remain in the
4691 channel 50–100 years after logging; (iii) relatively smaller woody debris initially stores
4692 sediment; (iv) when landslides and debris flows occur 3–15 years after logging because of

4693 intensive rain and weakening of root strength (Sidle et al. 1985), woody debris is evacuated from
4694 headwater streams and deposited in downstream reaches; (v) although less woody debris remains
4695 in the scour zone, woody debris pieces and jams contribute to sediment storage in both the scour
4696 and deposition zones of landslide and debris flow channels; (vi) red alder stands actively
4697 recolonize riparian zones of headwater streams for 20–50 years after mass movement and recruit
4698 woody debris and organic materials, which in turn provide sediment storage sites; and (vii)
4699 subsequent sediment movement after landslides and debris flows are affected by residual woody
4700 debris and newly introduced debris.

4701

4702 **LW and sediment**

4703

4704 Johnson et al., 2000 (removed from focal list)

4705

4706 Johnson, S. L., Swanson, F. J., Grant, G. E., & Wondzell, S. M. (2000). Riparian forest
4707 disturbances by a mountain flood—the influence of floated wood. *Hydrological processes*,
4708 14(16-17), 3031-3050. [https://doi.org/10.1002/1099-1085\(200011/12\)14:16/17<3031::AID-](https://doi.org/10.1002/1099-1085(200011/12)14:16/17<3031::AID-HYP133>3.0.CO;2-6)
4709 [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)

4710

4711 This study examined the differences in riparian forest responses to a 100-year flood event along
4712 eight third- to fifth-order streams in the Cascade Mountain Range of Oregon. Disturbance
4713 intensities were grouped into three categories: purely fluvial (high water flow only), fluvial with
4714 uncongested wood transport, and fluvial with congested wood transport. Riparian forest
4715 responses were heavily influenced by pre-flood forest structure and disturbance/harvest history,
4716 especially the characteristics of LW presence within streams and along channels. The quantity
4717 and severity of toppled trees (fully uprooted vs. partially uprooted) during the flood event was
4718 proportional to the quantity and congestion of LW already present (i.e., higher volumes of LW
4719 already present during the flood event increased the frequency of toppled trees and newly
4720 deposited LW in streams). Further, stands that experienced higher frequencies of toppled trees
4721 also showed higher frequencies and magnitudes of debris flow. The authors concluded that the
4722 land use practices, and disturbance histories influenced the age and structure of the riparian
4723 forests, but also the availability of the agents of disturbance (presence of LW) during the 100-
4724 year flood event. This paper is a good discussion of how pre-disturbance structure affects the
4725 response of riparian forests to disturbances (in this case, flood), however, there is no statistical
4726 analysis discussed in the methods. This is purely descriptive science that involves an intensive
4727 survey of before and after riparian forest structures.

4728

4729 **Sediment**

4730
4731 Yang et al., 2022 (removed from focal list)
4732
4733 Yang, Y., Safeeq, M., Wagenbrenner, J. W., Asefaw Berhe, A., & Hart, S. C. (2022). Impacts of
4734 climate and forest management on suspended sediment source and transport in montane
4735 headwater catchments. *Hydrological Processes*, 36(9), e14684.
4736 <https://doi.org/10.1002/hyp.14684>
4737
4738 This paper investigates the changes in annual hysteresis patterns for in-stream suspended
4739 sediment in 10 headwater streams at 2 sites, Providence Creek (rain-snow-dominated,
4740 transitional), and Kings River Experimental Watershed (snow-dominated). Aside from
4741 precipitation pattern differences in the two catchments, the researchers also compared differences
4742 in hysteresis patterns for forested riparian control, burn-only, thin-only, and thin-and-burn
4743 combined areas. The differences in the proportion of clockwise-loop hysteresis patterns for
4744 suspended sediments in the warmer rain-snow-transition sites compared to the colder snow-
4745 dominated sites suggests that warming temperatures may cause the snow-dominated basins to
4746 receive sediment from extended source areas and for longer periods if they transition to rain
4747 dominated catchments. The results found no discernable difference in hysteresis loops between
4748 the control, burn-only, thin-only, and thin-and-burn combined areas. Further, there seemed to be
4749 little change in the hysteresis loops during drought, average, and excessively wet years. The
4750 authors speculate that local conditions will be more important in understanding the impacts of
4751 climate change than changes in precipitation patterns or average annual temperatures alone.
4752 Mainly, there is evidence that if snow-dominated watersheds become warm enough to transition
4753 to rain-dominated, there is potential for disruption to sediment discharge frequency, rates, and
4754 source distance. The indiscernible difference in hysteresis loops for the different treatments also
4755 suggests that management practices imposed to ameliorate these changes may not be completely
4756 effective.
4757
4758 **Nutrients**
4759
4760 Vanderbilt et al., 2003
4761
4762 Vanderbilt, K. L., Lajtha, K., & Swanson, F. J. (2003). Biogeochemistry of unpolluted forested
4763 watersheds in the Oregon Cascades: temporal patterns of precipitation and stream nitrogen
4764 fluxes. *Biogeochemistry*, 62(1), 87-117. DOI:10.1023/A:1021171016945
4765

4766 This study uses long-term datasets (ranging from 20-30 years) from six watersheds in the H.J.
4767 Andrews Experimental Watershed (HJA) in the west-central Cascade Mountains of Oregon to
4768 investigate patterns in dissolved organic nitrogen (DON) and dissolved inorganic nitrogen (DIN)
4769 export with watershed hydrology. The objectives of this study were to 1) characterize long-term
4770 patterns of N dynamics in precipitation and stream water at the HJA, 2) analyze relationships
4771 between annual output of N solutes and annual stream discharge, 3) analyze relationships
4772 between seasonal stream water N solute concentrations and precipitation and stream discharge,
4773 and 4) compare results with those from other forested watersheds. Precipitation data were
4774 collected at three-week intervals from 10/1/1968 until 5/24/1988 and at one-week intervals
4775 thereafter. Stream chemistry samples were collected weekly for the entirety of the study. Stream
4776 discharge was measured continuously throughout the study. The researchers used regression
4777 analysis of annual N inputs and outputs with annual precipitation and stream discharge to
4778 analyze patterns. The results showed DON was the largest component of N input at the low-
4779 elevation collector, followed by PON (particulate organic N), NO₃-N, and NH₄-N. At the high-
4780 elevation collector, NO₃-N input was higher than at low elevation and was the largest component
4781 of N in bulk and wet-only inputs, followed by NH₄-N, DON, and PON. For annual stream
4782 outputs, DON was the largest fraction of annual N output, followed by PON, NH₄-N and then
4783 NO₃-N. Total annual discharge was a positive predictor of annual DON export in all watersheds
4784 with r² values ranging from 0.42 to 0.79. In contrast, significant relationships between total
4785 annual discharge and annual export of NO₃-N, NH₄-N, and PON were not found in all
4786 watersheds. No systematic long-term average seasonal trends were observed for NO₃-N or PON
4787 concentrations. Elevated concentrations of NH₄-N occurred in spring and early summer in all
4788 three watersheds, although they are not convincingly synchronous. DON concentrations
4789 increased in the fall in every watershed. The increase in concentration began in July or August
4790 with the earliest rain events, and peak DON concentrations occurred in October through
4791 December before the peak in the hydrograph. DON concentrations then declined during the
4792 winter months. The authors conclude that total annual stream discharge was a positive
4793 predictor of DON output suggesting a relationship to precipitation. Also, DON had a consistent
4794 seasonal concentration pattern. All other forms of N observed showed variability and
4795 inconsistencies with annual and seasonal stream discharge. The authors speculate that different
4796 factors may control organic vs. Inorganic N export. Also, DIN may be strongly influenced by
4797 terrestrial or in-stream biotic controls, while DON is more strongly influenced by climate. Last,
4798 the authors suggest that DON in streams may be recalcitrant, and largely unavailable to stream
4799 organisms. The authors emphasize the importance of analyzing data from multiple watersheds in
4800 a single climactic zone to make inferences about stream chemistry.

4801

4802 **Stream temperature**

4803

4804 Roon et al., 2021b

4805

4806 Roon, D. A., Dunham, J. B., & Torgersen, C. E. (2021). A riverscape approach reveals
4807 downstream propagation of stream thermal responses to riparian thinning at multiple scales.
4808 *Ecosphere*, 12(10), e03775. <https://doi.org/10.1002/ecs2.3775>

4809

4810 This study uses a riverscape approach to evaluate the effects of streamside forest thinning on
4811 stream temperatures at multiple spatiotemporal scales. This study addresses the question of how
4812 thinning second-growth riparian forests influences local and downstream temperatures at
4813 watershed extents. This study attempts to answer this question by addressing four objectives: (1)
4814 quantify pretreatment spatial and temporal variability in stream temperature conditions; (2)
4815 evaluate local responses in stream temperature to riparian thinning; (3) assess the spatial extent
4816 and temporal duration of downstream effects to local responses in temperature; and (4)
4817 characterize local and downstream responses to thinning with a conceptual framework based on
4818 waveforms. The researchers compared upstream, local, and downstream, stream temperature
4819 fluctuations following different intensities of streamside forest thinning at 10 treatment reaches
4820 across three watersheds in the redwood forests of northern California. Treatments varied by
4821 landowners. In two watersheds thinning treatments were intended to reduce 50% of canopy
4822 closure within the riparian zone along a 200 m reach on both sides of the active channel. This
4823 treatment resulted in a reduction in effective shade over the stream between 19-30%. In the other
4824 treatment watershed, thinning treatments reduced basal area by as much as 40% on both sides of
4825 the active channel along a 100 m long reach. Reductions in effective shade over the stream in
4826 these sites ranged from 4-5%. The analysis considered each reach both individually and
4827 collectively to understand how site and treatment heterogeneity may affect thermal responses at
4828 local and watershed extents. Temperature data were collected before, during, and after treatment
4829 and in the thinned experimental reaches and in adjacent unthinned control reaches with digital
4830 temperature sensors. Temperature data was collected for only 1-year pre-treatment and 1-year
4831 post-treatment. For data analysis, semivariograms of summer degree days were used to
4832 determine the presence of spatial autocorrelation. To control temporal variations in local and
4833 downstream responses summer cumulative degree-days were plotted for pre- and post- treatment
4834 temperatures and along a longitudinal gradient. A Lagrangian framework was used to track
4835 changes in temperature through space and time. Results showed that increases in thermal
4836 heterogeneity occurred in the treatment reaches, in the year following treatment (20° to 139°C),
4837 compared to the pre-treatment year (66° to 112°C). Local changes in stream temperature were
4838 dependent on thinning intensity, with higher levels of canopy cover reduction leading to higher
4839 increases in local stream temperatures. In the reaches with higher reductions in shade (19-30%)
4840 there was accumulation of 45° to 115°C additional degree days from pre- to post treatment years,
4841 while the reaches with lower reductions in shade (4-5%) only accumulated 10° to 15°C
4842 additional degree days. Travel distance of increased stream temperatures also appeared to be
4843 dependent on thinning intensity. The lower shade reduction reaches had an increased temperature
4844 effect downstream with travel distance of 75-150 m, while the high shade reduction sites had a
4845 downstream travel distance of 300- ~1000 m. In the high shade reduction sites, treatment reaches
4846 that were further apart (> 400 m) showed dissipation in increased stream temperatures
4847 downstream, while in parts of the stream where treatments were <400 m apart, temperature

4848 increases did not always dissipate before entering another the next treatment reach. The analyses
4849 with the conceptual framework based on waveforms showed there was no evidence of
4850 cumulative watershed effects at the downstream extent. The authors conclude that their results
4851 show evidence that riparian forest management impacts may extend beyond local stream
4852 environments. Further, the authors propose that riparian forest management that uses a holistic
4853 approach may be more effective in preserving some functions (e.g., shade).

4854

4855 **Sediment**

4856

4857 Wissmar et al., 2004

4858

4859 Wissmar, R.C., Beer, W.N. & Timm, R.K. (2004) Spatially explicit estimates of erosion-risk
4860 indices and variable riparian buffer widths in watersheds. *Aquat. Sci.* 66, 446–455 . DOI:
4861 10.1007/s00027-004-0714-9

4862

4863 The purpose of this study is to use management records, the spatial distribution, and the
4864 variability of different landcover types that can contribute to unstable conditions to develop
4865 erosion-risk indices and variable riparian buffer widths in watersheds of different drainages in
4866 the State of Washington. The objectives of this study were to 1) define erosion risk indices based
4867 on “different land cover types,” 2) evaluate erosion risk indices with sediment inputs into
4868 streams, 3) use erosion risk categories to define locations of stream reaches that are susceptible
4869 to different levels of erosion 4) use categories to identify distribution of channels requiring
4870 variable width buffers for protection 5) Test procedure by applying ground-truthed data from the
4871 upper Cedar River drainage near Seattle, Washington. The land cover types used to assess risk
4872 included unstable soils, immature forests, roads, critical slopes for land failure, and rain-on-snow
4873 events. Based on available data, the researchers developed a map of these land cover features
4874 with sediment input values to define erosion risk indices. The indices were used to categorize the
4875 landscape into 6 levels of erosion risk. Results of the mapped erosion risk categories explained
4876 65% of the variation associated with sediment inputs. The highest-risk areas contained a
4877 combination of all landscape cover factor combinations (rain-on-snow zone, critical failure
4878 slope, unstable soil, immature forests, and roaded areas). The lowest risk categories contained
4879 only rain-on-snow zones, and critical failure slopes. Roaded areas and unstable soils were only
4880 present in risk categories 3-6. This paper shows the importance of investigating multiple factors
4881 when evaluating the controls on sediment discharge and stream inputs. Further, when factors
4882 influencing erosion combine in an area, their effects are compounded.

4883

4884 **Nutrient and forest structure**

4885
4886 Devotta et al., 2021 (removed)
4887
4888 Devotta, D. A., Fraterrigo, J. M., Walsh, P. B., Lowe, S., Sewell, D. K., Schindler, D. E., & Hu,
4889 F. S. (2021). Watershed Alnus cover alters N: P stoichiometry and intensifies P limitation in
4890 subarctic streams. *Biogeochemistry*, 153(2), 155-176. DOI:10.1007/s10533-021-00776-w

4891
4892 This study investigates how coverage of alder species affects the aquatic N and P availability
4893 across a natural alder coverage gradient in 26 streams of southwestern Alaska. Alder coverage in
4894 the Alaskan streams was inversely related to elevation (i.e., lower coverage at higher elevations).
4895 To identify the presence of alder as the N and p contributing factor, the researchers analyzed
4896 resin lysimeter samples from select watershed soils supporting variable percent coverages of
4897 alder. Soils supporting alders leached, on average, three times more N and two times more P than
4898 soils not containing alders. The relationship between alder coverage and N and P values was not
4899 linear. Still, the authors identified 30% alder coverage as a transitional threshold from low to
4900 markedly higher soil N and p availability. The higher soil N and P resulted in higher dissolved N
4901 in streams, but the higher soil P under alder coverage did not translate to higher stream P
4902 availability. The authors speculate that soil chemistry or local soil biota may be immobilizing the
4903 soil P from transport into the streams. This led to a high N:P ratio in the spring and summer
4904 stream chemistry of reaches supporting >30% alder coverage. As climate change causes
4905 increasing temperatures, alder may begin to expand its range into higher elevations. This, in turn,
4906 may lead to increased N availability, but higher P limitations in high-elevation montane streams.

4907
4908 **Sediment and lithology**

4909
4910 Fratkin et al., 2020 (removed from focal, scope and results not relevant to review)

4911
4912 Fratkin, M. M., Segura, C., & Bywater-Reyes, S. (2020). The influence of lithology on channel
4913 geometry and bed sediment organization in mountainous hillslope-coupled streams. *Earth*
4914 *Surface Processes and Landforms*, 45(10), 2365-2379. <https://doi.org/10.1002/esp.4885>

4915
4916 This study compares the differences in channel form patterns, sediment flow, grain size, and
4917 shear stress thresholds between two gravel-bed streams, one on basalt and one on sandstone
4918 parent material in the Oregon Coast Range. Study sites were in a region where widespread
4919 landslides and debris flows occurred in 1996. The researchers compared channel
4920 geomorphologies (e.g., slope, valley width, channel geometry, etc.) to evaluate thresholds and

4921 channel bed adjustments since the 1996 events. The results showed similar sediment coarsening
4922 patterns in the first several kilometers indicating hillslope influence, but downstream fining was
4923 lithology dependent. The authors hypothesized threshold channel conditions in the basalt basin,
4924 and non-threshold conditions in the sandstone basin with a tendency to expose bedrock, based on
4925 the relative competencies (i.e., basalt = high-competency, sandstone = low-competency).
4926 However, results showed evidence of threshold conditions for over 60% of the streams in both
4927 basins. The authors inferred a cycle adjustment to correct the assumed sediment delivery from
4928 the 1996 flood season. The authors speculate that the basalt basins would act as threshold
4929 channels over longer time periods despite a higher debris flow frequency. This paper provides
4930 some evidence that lithologies impose control on channel adjustments driven by different rock
4931 competencies. This difference in rock competency ultimately controls the grain size fining rates
4932 and bed load transport (sediment availability).

4933

4934 **Nutrient and species composition**

4935

4936 Whigham et al., 2017 (removed from focal)

4937

4938 Whigham, D. F., Walker, C. M., Maurer, J., King, R. S., Hauser, W., Baird, S., ... & Neale, P. J.
4939 (2017). Watershed influences the structure and function of riparian wetlands associated with
4940 headwater streams—Kenai Peninsula, Alaska. *Science of the Total Environment*, 599, 124-134.
4941 <https://doi.org/10.1016/j.scitotenv.2017.03.290>

4942

4943 This field study was designed to test the hypothesis that alder cover in watersheds influences the
4944 structure and function of riparian wetlands adjacent to headwater streams. The researchers
4945 compared biomass production, biomass distribution (aboveground vs. belowground),
4946 decomposition rates, and chemical characteristics of interstitial groundwater, between watersheds
4947 with and without alder coverage. Study sites were located on two headwater streams located in
4948 the Kenai Peninsula in south-central Alaska. The results showed that aboveground biomass was
4949 higher in watersheds with alder cover, but the largest differences were in the litter layer and the
4950 belowground biomass. Watersheds without alder had significantly higher belowground root
4951 biomass. The litter overhanging the stream was higher in N content at the alder sites than in the
4952 no-alder sites. The quantity of litter overhanging the stream was higher in the no-alder sites.
4953 Interstitial groundwater was significantly higher in dissolved N at the alder sites. The results of
4954 this study show that species composition within the riparian area can have a considerable effect
4955 on nutrient concentrations which consequently affect stream chemistry, biomass production,
4956 vegetation structure, and decomposition rates.

4957

4958 **LW**

4959

4960 Wing & Skaugset, 2002

4961

4962 Wing, M. G., & Skaugset, A. (2002). Relationships of channel characteristics, land ownership,
4963 and land use patterns to large woody debris in western Oregon streams. *Canadian Journal of*
4964 *Fisheries and Aquatic Sciences*, 59(5), 796-807. <https://doi.org/10.1139/f02-052>

4965

4966 This study investigated the relationships of land use, land ownership, and channel and habitat
4967 characteristics with LW quantity and volume in 3793 stream reaches in western Oregon State
4968 (west of Cascade crest). This study analyzed an extensive spatial database of aquatic habitat
4969 conditions created for western Oregon using stream habitat classification techniques and a
4970 geographic information system (GIS). The overall objectives of this study were to identify the
4971 database factors most strongly related to LWD abundance and to determine whether ownership
4972 and land use patterns are related to LWD abundance. Regression tree analysis is an exploratory
4973 regression analysis that allows for the inclusion of multiple explanatory variables. LW counts (by
4974 piece, and by key pieces (logs at least 0.60 m in diameter and 10 m long)) and volume were used
4975 as the response variables and explanatory variables included morphology of active channel
4976 (hillslope, terrace, terrace hillslope, unconstrained), lithology (e.g., alluvium, basalt, etc.), Land
4977 use and land cover (e.g., young timber, old timber, rural resident, agriculture, etc.), ownership
4978 (private industrial (PI), private non-industrial (PNI), state, federal (BLM, USFS)), vegetation
4979 type, and other channel characteristics. The analysis was run at the reach scale. Results showed
4980 that the most important predictor for LW volume was land ownership with PNI split from all
4981 other ownership types. Mean LW volumes in stream reaches with PNI ownership were 3.1 m³
4982 while mean volume of LW in reaches in all other ownerships (PI, state, BLM, USFS) were 17.9
4983 m³. However, this was likely because the PNI lands held a disproportionately higher percentage of
4984 unforested lands compared to all other ownership types. When the ownership and land use
4985 variables were removed, stream gradient became the most important explanatory variable for LW
4986 volume. The split for stream gradient occurred for reaches with < 2.3% gradient averaged 5.8 m³
4987 while higher gradient streams averaged 17.9 m³ per reach. When ownership and land use were
4988 included but non-forested lands were removed, stream gradient again was the most important
4989 predictor with the split occurring for stream reaches with gradients less than 4.7% averaging 11.5
4990 m³, which was less than half of the average found at higher gradient reaches (25.2 m³); in this
4991 model the stream gradient split explained 11% of the variation observed of instream LW volume.
4992 For LW pieces in forested stream reaches bankfull channel width was the most important
4993 explanatory variable with the split occurring for streams channels less than 12.2 m wide. LW
4994 pieces for streams <12.2 m wide averaged 11.1 LW pieces per reach while larger channels
4995 averaged 4.9 pieces per reach; in this model the BFW split explained 7% of the variation in LW
4996 pieces found in forested streams. For key LW pieces (logs at least 0.60 m in diameter and 10 m
4997 long) in forested reaches, stream gradient was again the most important explanatory variable
4998 with the split occurring at a slope of 4.9%. The streams with a gradient < 4.9% averaged 0.5 key
4999 LW pieces per reach while streams with higher gradients averaged 0.9 key LW pieces per reach;

5000 in this model stream gradient explained 8% of the variation in key LW pieces found in streams.
5001 For forested streams, lithology caused second, third or fourth level splits after stream gradient or
5002 BFW. In three of these four splits, Mesozoic sedimentary and metamorphic geologies, located in
5003 southern Oregon stream reaches, were grouped and split from basalt, cascade, and marine
5004 sedimentary geologies. In stream reaches in Mesozoic sedimentary and metamorphic geologies,
5005 the quantity of LWD was roughly half the amount found in other geologies. The only exception
5006 to this grouping was for LW volume in larger stream reaches, where basalt and marine
5007 sedimentary geologies were grouped separately from all other geologies in a fourth-level split
5008 and contained more LW volume. The authors conclude that the geomorphic characteristics of
5009 stream reaches, in particular stream gradient and bankfull width, in forested areas correlated best
5010 with LW presence.

5011

5012

5013 **LW and plant communities**

5014

5015 Rot et al., 2000 ([removed from focal list](#))

5016

5017 Rot, B. W., Naiman, R. J., & Bilby, R. E. (2000). Stream channel configuration, landform, and
5018 riparian forest structure in the Cascade Mountains, Washington. *Canadian Journal of Fisheries
5019 and Aquatic Sciences*, 57(4), 699-707. <https://doi.org/10.1139/f00-002>

5020

5021 This study investigates the hierarchical relationships between the “five key elements”, valley
5022 constraint, riparian landform, riparian plant community, channel type, and channel configuration.
5023 for 21 sites in mature old-growth riparian forests of the western Cascade Mountains in
5024 Washington State. The objective of this article is to expand this perspective over several spatial
5025 scales and the temporal life span of a conifer by examining how channel configuration interacts
5026 with valley constraint, streamside landform, channel bedform, and successional processes within
5027 the riparian forest. Stepwise regression was used to examine the relationship between physical
5028 and biological characteristics and the individual elements of channel configuration. Channel
5029 configuration is the channel elements at the habitat unit scale, including channel units (total
5030 number of pool–riffle habitat units per 100 m of channel length), LW pieces (per 100 m of
5031 channel length), LW volume (cubic meters per 100 m of channel length), pool spacing, percent
5032 pools, and percent LW-formed pools. Results showed that significantly more total LW pieces
5033 were found in forced pool–riffle channels than in the bedrock and plane-bed channels (Kruskal–
5034 Wallis, $p < 0.05$). Forced pool–riffle channels averaged 16.4 pieces per 100 m, bedrock 10.8
5035 pieces, and plane-bed 10.1 pieces. The volume of LW (cubic meters per 100 m) followed a
5036 similar trend. The percentage of deep pools (>0.5 m) formed by LW increased with stand age (r^2
5037 = 0.36). LW diameters were significantly smaller for ages 55–220 than for ages 333–727
5038 (Kruskal–Wallis, $p = 0.01$). The authors conclude that scale is an important consideration for

5039 management of aquatic habitat. At the largest spatial scale, results showed valley constraint
5040 significantly influenced off-channel habitat (plant communities associations and landform
5041 categories) and in-stream LW volume within forced pool-riffle channels. At the smallest scale,
5042 channel type (bedrock, plane-bed, and forced pool-riffle) was most closely related to LW
5043 volume, density, and the number of LW-formed pools. The diameter of the in-channel LW
5044 increased with riparian forest stand age. Streams adjacent to old-growth forests in-channel LW
5045 diameter were equivalent to or greater than the average standing riparian tree diameter at all
5046 sites. In younger stands, the relationship of in-stream LW diameter had a mixed relationship with
5047 riparian tree average diameters. The authors speculate this may be due to many in-stream LW
5048 pieces being relics from previous old-growth communities. In this area, four landform classes
5049 differentiated the riparian communities (floodplain, low terrace, high terrace, slope). Most were
5050 dominated by conifers, except the floodplain landforms, which supported a higher density of
5051 deciduous species, but a higher basal area of conifer species. The results of this study provide
5052 more evidence, similar to other studies, that channel geomorphology and valley constraint are
5053 important predictors of LW abundance (quantity and volume) in streams. The novelty in this
5054 study is how the riparian area landforms lead to different riparian plant communities, which
5055 consequently affect the input of LW.

5056

5057 **Nutrients**

5058

5059 Yang et al., 2021

5060

5061 Yang, Y., Hart, S. C., McCorkle, E. P., Stacy, E. M., Barnes, M. E., Hunsaker, C. T., ... & Berhe,
5062 A. A. (2021). Stream water chemistry in mixed-conifer headwater basins: role of water sources,
5063 seasonality, watershed characteristics, and disturbances. *Ecosystems*, 24(8), 1853-1874.
5064 DOI:10.1007/s10021-021-00620-0

5065

5066 This study investigated the effects of drought and forest thinning operations (independently and
5067 combined) on water chemistry from multiple basin water sources (snowmelt, soil solution,
5068 stream water) in the Mediterranean climate headwater basins of the Sierra National Forest. Data
5069 on water chemistry was taken 2 years prior and 3 years following drought and thinning
5070 operations in two watersheds, each with thinned and control stands. This data was analyzed to
5071 answer 3 questions: 1. How does the chemistry of different water sources (that is, snowmelt, soil
5072 solution at two depths, stream water) vary monthly and interannually prior to drought and
5073 thinning? 2. How does drought alone and drought combined with thinning impact water
5074 chemistry? 3. Can watershed characteristics predict stream water chemistry over contrasting
5075 water years? The authors used general linear models to analyze differences in chemistry by water
5076 source, repeated measures analysis of variance for effects of drought and thinning on water
5077 chemistry, and linear regression to predict water chemistry based on watershed characteristics.

5078 Results showed that monthly concentrations of dissolved C and N varied among different water
5079 sources prior to drought and thinning. For dissolved organic carbon (DOC) soil solution at 13 cm
5080 depth (mean \pm SE of $25.97 \pm 2.75 \text{ mg l}^{-1}$, across months for 2 years) had higher monthly
5081 concentrations than soil solution collected at 26 cm depth ($16.93 \pm 1.55 \text{ mg l}^{-1}$). Snowmelt (9.67
5082 $\pm 0.89 \text{ mg l}^{-1}$) and stream water ($5.33 \pm 0.52 \text{ mg l}^{-1}$) had the lowest concentrations. For total
5083 dissolved Nitrogen (TDN) and dissolved organic nitrogen (DON), soil solution at 13 cm depth
5084 (1.72 ± 0.57 and $1.66 \pm 0.57 \text{ mg l}^{-1}$, respectively), soil solution at 26 cm depth (0.94 ± 0.32 and
5085 $0.92 \pm 0.32 \text{ mg l}^{-1}$), and snowmelt (0.94 ± 0.17 and $0.73 \pm 0.18 \text{ mg l}^{-1}$) had higher
5086 concentrations than stream water (0.11 ± 0.02 and $0.08 \pm 0.01 \text{ mg l}^{-1}$). For dissolved inorganic
5087 nitrogen (DIN), snowmelt ($0.25 \pm 0.05 \text{ mg l}^{-1}$) had the highest concentration followed by the soil
5088 solution at 13 cm depth ($0.06 \pm 0.01 \text{ mg l}^{-1}$). Soil solution at 26 cm depth ($0.03 \pm 0.01 \text{ mg l}^{-1}$)
5089 and stream water had the lowest values ($0.04 \pm 0.01 \text{ mg l}^{-1}$). For pH, snowmelt (pH 6.09 ± 0.06)
5090 was more acidic than soil solutions at both depths (7.52 ± 0.23 at 13 cm depth and 7.79 ± 0.11 at
5091 26 cm depth) and stream water (7.37 ± 0.07). Drought alone altered DOC in stream water, and
5092 DOC:DON in soil solution in unthinned (control) watersheds. Volume-weighted concentration of
5093 DOC was 62% lower ($p < 0.01$) and DOC:DON was 82% lower ($p = 0.004$) in stream water in
5094 years during drought (WY 2013–2015) than in years prior to drought (WY 2009 and 2010).
5095 Drought combined with thinning altered DOC and DIN in stream water, and DON and TDN in
5096 soil solution. For stream water, volume-weighted concentrations of DOC were 66–94% higher in
5097 thinned watersheds than in control watersheds for all three consecutive drought years following
5098 thinning. No differences in DOC concentrations were found between thinned and control
5099 watersheds before thinning. Watershed characteristics explained inconsistently the variation in
5100 volume-weighted mean annual values of stream water chemistry among different watersheds.
5101 The authors conclude that their results showed evidence that the influences of drought and
5102 thinning are more pronounced for DOC than for N in streams.

5103

5104 **Geology**

5105

5106 Kusnierz and Sivers, 2018 (removed from focal)

5107

5108 Kusnierz, P.C., Sivers, E., 2018. How important is geology in evaluating stream habitat? *J Soils*
5109 *Sediments* 18, 1176–1184. DOI:10.1007/s11368-017-1885-z

5110

5111 The purpose of this study was to assess the importance of considering geology when evaluating
5112 stream habitat conditions. Stream habitat data were collected from 424 sites on federally
5113 managed lands in western Montana, USA. These sites represented a variety of ecoregions, stream
5114 types, management practices, and geologies. The importance of accounting for geology in data
5115 analysis was evaluated using five sediment-related habitat variables and three analyses that
5116 examined (1) differences across geology for the entire dataset and for sites in reference and

5117 managed watersheds; (2) differences between reference and managed sites within geologies; and
5118 (3) the relative strength of geology as a factor when accounting for the effects of management,
5119 stream type, and ecoregion. This objective was pursued by using five sediment-related habitat
5120 variables (Log instability index, Log roughness-corrected index of relative bed stability, Median
5121 substrate size, Percent pool tail fines < 6 mm, Percent stable banks). Five sediment-related
5122 habitat variables were collected from 424 sites on federally managed lands between 2009-
5123 2012. Factorial ANOVA on ranks was performed to evaluate the relative importance of geology
5124 when other factors were taken into account. Results from this study show that differences in
5125 sediment-related habitat variables did not differ significantly according to geology; however,
5126 observed differences were typically drawn from managed sites. The authors conclude by
5127 advising against using geology as the sole means of stratifying habitat data when attempting to
5128 account for between-site variability.

5129

5130 **Stream Temperatures**

5131

5132 Leach et al., 2017 (removed from focal list)

5133

5134 Leach, J.A., Olson, D.H., Anderson, P.D., Eskelson, B.N.I., 2017. Spatial and seasonal variability
5135 of forested headwater stream temperatures in western Oregon, USA. *Aquat Sci* 79, 291–307.
5136 DOI:10.1007/s00027-016-0497-9

5137

5138 This study is a case study of thermal regimes for headwater streams in the Keel Mountain Study
5139 area. This study examined (1) forested headwater stream temperature variability in space and
5140 time; (2) relationships between stream temperature patterns and weather, above-stream canopy
5141 cover, and geomorphic attributes; and (3) the predictive ability of a regional stream temperature
5142 model to account for headwater stream temperature heterogeneity. Stream temperature data was
5143 collected at 48 sites within a 128-ha watershed in western Oregon between 2012 and 2013.
5144 Spatial statistical modeling was used to relate stream temperature patterns to site characteristics
5145 (elevation, stream width, catchment area, slope, aspect, channel substrate, and terrain shading), a
5146 cluster analysis was used to capture the full variability in annual stream temperatures. Results
5147 from this study show considerable variability in stream temperature over relatively small areas,
5148 and between seasons. The greatest spatial variability existed during summer (up to 10 Celsius)
5149 and during cold and dry winter periods (up to 7.5 Celsius). Geomorphic attributes typically used
5150 in stream temperature models were not good predictors of variability at headwater scales.

5151

5152 **Stream Temperatures**

5153

5154 Groom et al., 2011b

5155

5156 Groom, J.D., Dent, L., Madsen, L.J., Fleuret, J.(2011b). Response of western Oregon (USA)
5157 stream temperatures to contemporary forest management. *Forest Ecology and Management* 262,
5158 1618–1629. <https://doi.org/10.1016/j.foreco.2011.07.012>

5159

5160 The objective of this paper was to assess the riparian characteristics that best predict shade, and
5161 to determine the stream temperature changes that result following harvest. This study took place
5162 in the Oregon Coastal Range at 33 sites (15 state-owned and 18 private-owned). The 33 sites
5163 studied were approximately 50-70 years old and predominately composed of Douglas-fir and red
5164 alder. Private sites (n = 18) followed FPA rules whereby the riparian management area (RMA)s
5165 are 15 and 21 m wide on small and medium fish-bearing streams, with a 6 m no-cut zone
5166 immediately adjacent to the stream. Harvesting is allowed in the remaining RMA to a minimum
5167 basal area of 10.0 (small streams) and 22.9 (medium streams) m²/ha. State sites (N = 15)
5168 followed the state management plan whereby a 52 m wide buffer is required for all fish-bearing
5169 streams, with an 8 m no cut buffer immediately adjacent to the stream. Limited harvest is
5170 allowed within 30 m of the stream only to create mature forest conditions. Harvest operations
5171 within this zone must maintain 124 trees per hectare and a 25% Stand Density Index. Additional
5172 tree retentions of 25–111 conifer trees and snags/hectare are required between 30 and 52 m. A
5173 site's control reach was located immediately upstream of its treatment reach. The control reaches
5174 were continuously forested to a perpendicular slope distance of at least 60 m from the average
5175 annual high-water level. Reach lengths varied from 137 m to 1,829 m with means of 276 m and
5176 684 m for the control and treatment reaches, respectively. Temperature recording stations were
5177 located upstream and downstream of both control and treatment sites. Stream temperature data
5178 was summarized to provide daily minimum, maximum, mean, and fluctuation for analysis. The
5179 temperature data was modeled using mixed-effects linear regression. Shade analysis included
5180 trees per hectare, basal area per hectare, vegetation plot blowdown, and tree height. A linear
5181 regression analysis of shade data (n = 33) was performed and compared small-sample AIC values
5182 to determine relative model performance among 8 a priori models. Results showed that average,
5183 minimum, and diel stream temperatures increased on private sites following harvest, suggesting a
5184 relationship between decreased shade derived from buffer width and an increase in stream
5185 temperature. Outputs from the model predicted an increase of ~2 °C for minimum shade
5186 conditions and a decrease of ~-1 °C for maximum shade conditions. For sites that exhibited an
5187 absolute change of shade > 6% from pre-harvest to post-harvest experienced an increase in
5188 maximum temperatures. Further, the model predicted an increase in stream temperature
5189 proportional to treatment reach length. The authors estimate an increase in maximum and
5190 minimum temperatures of 0.73 and 0.59 °C per km, respectively. Following harvest, maximum
5191 temperatures at private sites increased relative to state sites on average by 0.71 °C. Similarly,
5192 mean temperatures increased by 0.37 °C (0.24 - 0.50), minimum temperatures by 0.13 °C (0.03 -
5193 0.23), and diel fluctuation increased by 0.58 °C (0.41 - 0.75) relative to state sites. A comparison
5194 of within site changes in maximum temperatures pre-harvest to post-harvest showed an overall

5195 increase at private sites, but not all sites behaved the same and some had decreases in maximum
5196 temperatures. The average of maximum state site temperature changes = 0.0 °C (range = -0.89 to
5197 2.27 °C). Observed maximum temperature changes at private sites averaged 0.73 °C (range = -
5198 0.87 to 2.50 °C) and exhibit a greater frequency of post-harvest increases from 0.5 to 2.5 °C
5199 compared to state sites. Private site shade values also appeared to decrease pre-harvest to post-
5200 harvest. Private post-harvest shade values differed from pre-harvest values (mean change in
5201 Shade from 85% to 78%); however, no difference was found for state site shade values pre-
5202 harvest to post-harvest (mean change in Shade from 90% to 89%). They did not find evidence
5203 that shade differed if one or both banks were harvested for private sites although the sample size
5204 for single sided harvests was low. Similarly, private site shade values did not appear to differ
5205 between medium or small streams. Results from this study also show that between 68% and
5206 75% of variability in post-harvest shade may be accounted for by basal area within 30 m of the
5207 stream, tree height, and potentially blowdown. The authors speculate that their results suggest
5208 sites with shorter trees have higher post-harvest shade and this may be due to the negative
5209 correlation between crown ratios and tree heights. Overall, this study shows that buffers managed
5210 by state sites were sufficient at mitigating the effects of upland harvesting on stream temperature.
5211 Increases in stream temperature on private sites were related to decreases in shade, which were
5212 related to decreases in basal area on sites with greater tree heights. The authors suggest that their
5213 results are likely relevant to other high-rainfall low-order Douglas-fir dominated streams in the
5214 Pacific Northwest that are subject to similar harvest practices.

5215

5216 **Litter**

5217

5218 Yeung et al., 2019

5219

5220 Yeung, A. C., Stenroth, K., & Richardson, J. S. (2019). Modelling biophysical controls on stream
5221 organic matter standing stocks under a range of forest harvesting impacts. *Limnologica*, 78,
5222 125714. <https://doi.org/10.1016/j.limno.2019.125714>

5223

5224 This study investigates the relative impact of major biophysical controls (stream temperature,
5225 riparian litterfall, and stream discharge) on in-stream CPOM (coarse particulate organic matter)
5226 quantity across a variety of streamside timber harvest intensities using simulation modeling. The
5227 CPOM model used was developed by Stenroth et al., 2014, for similar stream types and
5228 conditions of coastal rainforest streams of British Columbia. The model was calibrated using
5229 data from multiple published studies from, primarily the Pacific Northwest region, and several
5230 other North American regions, that quantified stream flow, temperature, and CPOM following
5231 different timber harvest intensities within 4 years of harvest. The model used an estimated
5232 response of low, moderate, high, and very high severity timber harvest for litterfall (-10%, -30%,
5233 -50%, -90%), peak flows (+20%, +40%, +100%, +300%), and stream temperature (+1°C, +2°C,

5234 +4°C, +6 °C). These changes in litterfall, peak flow, and stream temperature were modeled and
5235 analyzed individually and cumulatively to estimate their relative and combined effects on in
5236 stream CPOM standing stocks. Results of the model showed that in general the standing stocks
5237 of CPOM decreased under the independent effects of reduced litterfall and elevated peak flows
5238 and increased with higher stream temperatures. Along the gradient of harvest severities, litterfall
5239 reductions on depleting CPOM standing stocks were at least an order of magnitude greater than
5240 those of elevated peak flows. At low severity, litterfall reductions led to a 13.5% reduction of
5241 CPOM stocks while peak flow increases at high severity harvest only led to a 5% reduction in
5242 CPOM stocks. The magnitude of CPOM changes induced by litterfall reductions was
5243 consistently greater than stream temperature increases, but their differences in magnitude became
5244 smaller at higher levels of disturbance severity. For example, at low severity, stream
5245 temperatures only led to an increase on CPOM stocks by 1.1% while litter fall reductions led to a
5246 reduction of CPOM by 13.5%. However, at the high intensity treatment CPOM stocks changed
5247 by -90.24%, and +72.07% for litterfall, and stream temperature respectively. For scenarios
5248 involving perturbations of multiple model drivers (combined effects), the effect size of
5249 disturbance was significantly negative (indicating significantly lower CPOM standing stocks
5250 than in undisturbed conditions) whenever litterfall reductions reached 50% or above (i.e., high
5251 severity). When litterfall reductions were 30% or below, the effect size of disturbance varied with
5252 the relative changes in peak flows and stream temperature. Only the effects of litterfall-
5253 temperature interactions on CPOM standing stocks were significant ($p < 0.001$). The authors
5254 interpret these results as evidence that litterfall reduction from timber harvest was the strongest
5255 control on in-stream CPOM quantity for 4 years post-harvest. Further, the authors propose that
5256 the decreased activity of CPOM consumers caused by increasing stream temperatures may be
5257 enough to offset the loss of litterfall inputs on CPOM stocks. The caveat of this study is that it
5258 did not include LW dynamics in preserving CPOM post-harvest. As other studies have shown,
5259 harvest can increase in-stream LW, and in-stream LW can act as a catchment for CPOM.

5260

5261 **Drought Frequency**

5262

5263 Wise, 2010

5264

5265 Wise, E. K. (2010). Tree ring record of streamflow and drought in the upper Snake River. *Water*
5266 *Resources Research*, 46(11). <https://doi.org/10.1029/2010WR009282>

5267

5268 This study used newly collected tree-ring data augmented with existing chronologies from sites
5269 at three headwater streams in the Snake River Basin to estimate streamflow patterns for the
5270 1600-2005 time-period. The reconstructed chronologies were tested for significant correlations
5271 with streamflow patterns during the 1911-2005 time period prior to extrapolation. Streamflow
5272 patterns derived from instrumental data and from reconstructed chronologies were compared

5273 with other streamflow reconstructions of three other western rivers in similar climates to
5274 examine synchronicity among the rivers and gain insight into possible climatic controls on
5275 drought episodes. The reconstruction model developed for the analysis explained 62% of the
5276 variance in the instrumental record after adjustment for degrees of freedom. Results showed
5277 evidence that droughts of the recent past are not yet as severe, in terms of overall magnitude, as a
5278 30-year extended period of drought discovered in the mid-1600s. However, in terms of number
5279 of individual years of < 60% mean-flow (i.e., low-flow years), the period from 1977-2001 were
5280 the most severe. Considering the frequency of consecutive drought years, the longest (7-year-
5281 droughts), occurred in the early 17th and 18th centuries. However, the 5-year drought period from
5282 2000-2004 was the second driest period over the 415-year period examined. The author explains
5283 that the area has continued to experience a drought period, but its severity could not be
5284 calculated as it hadn't ended by the time of the study (2010). The correlative analysis of the
5285 chronologies developed for the upper Snake River with other rivers of the West (the upper
5286 Colorado, the Sacramento, and the Verde Rivers) showed mixed results with periods of positive
5287 and negative correlations. The author interprets these results as evidence that drought frequency
5288 in general, in this area appears to be increasing in severity and that mean annual flow appears to
5289 be reducing in the latter half of the 20th and the beginning of the 21st century. The exceptions
5290 being the 1930's dustbowl, and an unusually long dry period in the early 1600s.

5291

5292 **Shade and structure**

5293

5294 Warren et al., 2013

5295

5296 Warren, D. R., Keeton, W. S., Bechtold, H. A., & Rosi-Marshall, E. J. (2013). Comparing
5297 streambed light availability and canopy cover in streams with old-growth versus early-mature
5298 riparian forests in western Oregon. *Aquatic sciences*, 75(4), 547-558. DOI:10.1007/s00027-013-
5299 0299-2

5300

5301 This study investigates the differences in canopy cover and streambed light availability between
5302 paired reaches in old-growth (> 500 years old) and secondary-growth (~40-60 years old) riparian
5303 forests on canopy cover and streambed light exposure in four second order fish-bearing streams
5304 in the H.J. Andrews Experimental Forest. Streams were paired based on reach length and
5305 bankfull width and north (n =2), and south (n=2) facing watersheds. The overall mean percentage
5306 of canopy cover was estimated using a convex spherical densiometer every five meters along the
5307 thalweg of each stream reach. At each point densiometer readings were taken from four
5308 directions (upstream, downstream, left bank, right bank) The amount of light reaching the bottom
5309 of the stream was estimated every five meters using fluorescent dye that degrades overtime from
5310 light exposure. Differences in light availability and canopy cover were analyzed separately for
5311 each of the four reaches using a single factor ANOVA. To avoid the inclusion of overlapping

5312 canopy images from adjacent densiometer sampling locations, the canopy cover data from sites
5313 every 15 m (rather than every 5 m) were used in the comparison of canopy cover between the
5314 two age classes along each reach. Linear regression was used to compare values from mean
5315 densiometer readings with mean dye photodegradation site (every 5 meters). To evaluate the
5316 hypothesis that light availability in old-growth forested streams would be more variable than in
5317 second-growth forested streams, the standard deviations of the mean densiometer readings and
5318 mean photodegradation values were compared between old-growth and second-growth forested
5319 streams with an ANOVA. Results showed that the differences in stream light availability and
5320 percent forest cover between old-growth and second-growth reaches were significant in both of
5321 the south-facing watersheds in mid-summer at an alpha of 0.01 for the dye results and 0.10 for
5322 the cover results. For the north-facing watersheds differences in canopy cover and light
5323 availability (alpha = 0.01, and 0.10 respectively) were only significant at 1 of the two reaches.
5324 Overall, three of the four paired old-growth reaches had significantly lower mean percent canopy
5325 cover, and significantly higher mean decline in fluorescent dye concentrations. The authors
5326 interpret these results as evidence that old-growth forest canopies were more complex and had
5327 more frequent gaps allowing for more light availability and lower mean canopy cover, on
5328 average, than in adjacent mature second-growth forests.

5329

5330 **LW**

5331

5332 Teply et al., 2007

5333

5334 Teply, M., McGreer, D., Schult, D., & Seymour, P. (2007). Simulating the effects of forest
5335 management on large woody debris in streams in northern Idaho. *Western Journal of Applied*
5336 *Forestry*, 22(2), 81–87. <https://doi.org/10.1093/wjaf/22.2.81>

5337

5338 This paper uses simulation modeling to estimate the effects of timber harvest, under the Idaho
5339 Forest Plan (IFP), on in-stream LW loading for Class I streams (fish-bearing streams) of the
5340 Priest Lake Watershed in northern Idaho relative to unharvested riparian forest streams. Under
5341 the IFP, class one streams have a 25-foot no-cut-buffer that extends out from the high-watermark,
5342 and an additional 50 feet beyond the edge of the no-cut-buffer where harvest requires retention of
5343 88-trees-per-acre that are greater than 8-in diameter at breast height (DBH). This study used the
5344 Riparian Aquatic Interaction Simulator (RAIS) to estimate the potential wood loading for 58
5345 randomly selected north Idaho stream segments with and without harvest. Stream segments were
5346 measured in the field along the stream centerline from the upstream starting point (0 ft) to a
5347 downstream ending point (200 ft). Riparian stand conditions were measured within 75 ft-long by
5348 10-ft-wide strips oriented perpendicular to the stream at 25, 75, 125, and 175 ft downstream of
5349 the upstream starting point on each side of the stream segment to provide a total of eight strips
5350 for each stream segment. Along each strip, live trees and snags greater than 8 in dbh within the

5351 strip were located and measured. Three circular subplots, each 10 ft in diameter, were located
5352 along each 75-foot strip plot at 12.5, 37.5, and 62.5 ft from the stream edge. Within the subplots,
5353 smaller live trees (less than 8-in. dbh) were tallied by 1-in. dbh classes. Instream LW loads were
5354 surveyed along the same 200-ft stream segments located for measuring riparian stand conditions.
5355 Qualifying LW (greater than 4-in diameter and longer than 6.6 ft) occurring within the high-
5356 water mark along the entire extent of the segment was tallied. Observed instream LW loads
5357 ranged from 10 to 710 pieces per 1,000 ft of stream. Stream size measured by bank full width
5358 covered a wide range (1 ft to 190 ft), averaging 32.5 ft (SD = 28.1). The authors determined that
5359 active streambank erosion was uncommon in the study area and did not include it as a LW
5360 recruitment mechanism in their analysis. Simulation was based on a four-step process applied to
5361 each riparian stand: 1) Harvest the stand according to riparian management prescriptions, 2)
5362 Predict stand characteristics using growth and yield simulators, 3) Estimate the number of trees
5363 that fall due to mortality in each time step, 4) Calculate the probability that a tree would deliver
5364 LWD to the stream. The simulation evaluated both a harvest and a no-harvest scenario to predict
5365 mean in-stream LW loads after 30, 60, and 100 years. The results predicted mean LW loads at 30
5366 years for the 58 segments studied were 151.1 pieces per 1,000 ft for the no-harvest scenario (SD
5367 = 76.2) and 145.1 pieces per 1,000 ft for the harvest scenario (SD = 75.6), which were not
5368 significantly different ($P = 0.67$). However, on a pairwise basis, loads predicted for these
5369 segments using the harvest scenario were significantly lower by an average of about 6.0 pieces
5370 per 1,000 ft than those predicted via the no-harvest scenario ($P < 0.001$). Compared to the initial
5371 surveyed LW loads, LW loads at 30 years predicted in the no-harvest scenario decreased by an
5372 average of 19.5 pieces per 1,000 ft, representing a significant ($P < 0.007$) downward shift in the
5373 distribution. Predicted mean LW loads at 60 years were 136.1 pieces per 1,000 ft in the no-
5374 harvest scenario (SD = 49.2) and 128.3 pieces per 1,000 ft under the harvest scenario (SD =
5375 48.3). At 100 years, predicted mean LW loads were 122.5 (SD = 35.4) and 116.7 (SD = 35.8),
5376 respectively. Based on 20-piece LW classes, the frequency distributions of predicted loads
5377 between the scenarios were not significantly different at either time step. However, on a pairwise
5378 basis, predicted loads for the harvest scenario were significantly lower than the no-harvest
5379 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
5380 100 years, respectively. Compared to LW loads predicted at 30 years and 60 years, LWD loads
5381 decreased significantly on a pairwise basis by an average of 15.1 ($P < 0.001$) and 13.6 ($P <$
5382 0.001) at 60 and 100 years, respectively. The authors note that the collective effect of the
5383 assumptions made for the simulation is likely to underestimate the number and variability of LW
5384 pieces recruited and retained in the streams sampled. The authors interpreted these results as
5385 evidence that the IFP prescriptions for class I Idaho streams were sufficient in maintaining LW
5386 recruitment potential.

5387

5388 **Shade**

5389

5390 Swartz et al., 2020

5391

5392 Swartz, A., Roon, D., Reiter, M., & Warren, D. (2020). Stream temperature responses to
5393 experimental riparian canopy gaps along forested headwaters in western Oregon. *Forest Ecology*
5394 *and Management*, 474, 118354. <https://doi.org/10.1016/j.foreco.2020.118354>

5395

5396 This study tested the effects of adding canopy gaps within young, regenerating forests of western
5397 Oregon on stream light availability and stream temperatures. The addition of gaps in the young
5398 regenerating forests were used to theoretically mimic the natural disturbance regimes and the
5399 higher canopy complexity of late-successional forests. The researchers used a before-after-
5400 control-impact design on six replicated streams within the Mckenzie River Basin. In the
5401 experimental reaches 30 m gaps were created, centered on a tree next to the stream and at least
5402 30 m in from the beginning of the reach. The study reaches were located on second- and third-
5403 order fish-bearing steep step-pool and cascade dominated headwater streams with boulder
5404 substrate that ranged from 2.2 to 6.4 m in bankfull width and were lined by 40- to 60-year-old
5405 riparian forests. Study sites in each stream encompassed two 120 m reaches with no large
5406 tributary inputs within or between the study reaches, and reference and treatment reaches were
5407 separated by a buffer section of 30–150 m. In each treatment reach, gaps were designed to create
5408 openings in the canopy that were approximately 20 m in diameter. Gaps were centered on a tree
5409 next to the stream at approximately meter 30 along each reach. The gaps sizes were intended to
5410 mimic naturally occurring gaps from an individual large tree mortality or small-scale disturbance
5411 events found in these systems which range from 0.05 to 1.0 gap diameter to tree height ratio with
5412 smaller gaps occurring more frequently. Using the Douglas-fir canopy height of 50 m, gaps were
5413 created in the 0.4–1.0 gap diameter to tree height ratio range (approximately 314 m² – 1,963
5414 m²). Actual gap sizes varied across sites from approximately 514 m² to 1,374 m² (0.45 – 0.74
5415 gap ratios) with a mean of 962 m² (mean gap ratio 0.61). Riparian shade was quantified with
5416 hemispherical photos. Light reaching the stream was quantified using photodegradation of
5417 fluorescent dyes placed at 5 m intervals, over a 24 -hour period. Stream temperature was
5418 recorded continuously, at 15-minute intervals, using HOBO sensors to quantify the seven-day
5419 moving average of mean and maximum temperatures. Data was collected for one year pre-
5420 harvest, during harvest year (harvest took place in late fall 2017), and one-year post-harvest. To
5421 determine the effects of experimental canopy gaps on stream light as well as reach responses a
5422 linear mixed-effects model was fit to the data. The results showed that after gaps were cut, the
5423 BACI analysis showed strong evidence for significant increase in mean reach light ($p < 0.01$) to
5424 a mean of 3.91 (SD \pm 1.63) moles of photons m⁻² day⁻¹. overall resulting in a mean change in
5425 light of 2.93 (SD \pm 1.50) moles of photons m⁻² day⁻¹. Mean stream shading could not be
5426 evaluated in the full BACI analysis because post-treatment hemispherical photographs could not
5427 be taken at all sites due to fire impeding access in 2018. For the remaining sites, the areas
5428 beneath each gap had notable localized declines in shade, through the entirety of the treatment
5429 reach mean shading declined by only 4% (SD \pm 0.02%). Overall, the gap treatments did not
5430 change summer T 7DayMax or T 7DayMean significantly across the 6 study sites. The mean
5431 response (change in reach difference before and after the cut) indicated an increase on average
5432 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
5433 °C); however, there was not statistical support of the BACI effect for either metric. The light

5434 response was not correlated with T 7DayMax responses ($r^2 < 0.01$, $p = 0.69$), nor was gap area
5435 ($r^2 = 0.01$, $p = 0.63$), but there was a significant relationship between discharge ($r^2 = 0.73$, $p =$
5436 0.03), and bankfull width ($r^2 = 0.93$, $p < 0.01$) and the T7DayMax response. Wetted width was
5437 also highly correlated with T 7DayMax responses, but the relationship was not as strong with
5438 this stream size metric as with discharge or bankfull width ($r^2 = 0.65$, $p = 0.05$). In contrast to the
5439 summary values, results from the analysis of individual days throughout the full 40-day summer
5440 period identifying differences in the relationships of daily maximums and daily means between
5441 reaches showed a statistically significant effect of the gap for average daily maximums ($p < 0.01$)
5442 and for average daily means ($p = 0.02$). The regression comparison reveals there will be on
5443 average an additional $0.12\text{ }^{\circ}\text{C}/^{\circ}\text{C}$ increase in daily maximum temperature in the reach with a gap.
5444 Likewise, for the daily mean, for every degree increase in the shaded reference reach, an average
5445 additional increase of $0.05\text{ }^{\circ}\text{C}$ in a reach with a small gap is expected. The authors conclude that
5446 adding gaps to young regenerating forests only minimally increases temperatures, dependent on
5447 stream size, and that riparian canopy gaps may be a viable management strategy that can be
5448 implemented with minimal effects on stream temperatures. This paper does not quantify changes
5449 in stream productivity, also expected from the increase in available light.

5450

5451 **Shade**

5452

5453 Sugden et al., 2019

5454

5455 Sugden, B. D., Steiner, R., & Jones, J. E. (2019). Streamside management zone effectiveness for
5456 water temperature control in Western Montana. *International Journal of Forest Engineering*,
5457 30(2), 87-98. <https://doi.org/10.1080/14942119.2019.1571472>

5458

5459 This study investigates the effects of riparian forest timber harvest, under the Montana
5460 Streamside Management Zone (SMZ) laws, on stream temperature in Class 1 streams (fish-
5461 bearing, or flow more than 6 months per year and are connected to downstream waters).
5462 Montana state law requires timber be retained within a minimum of 15.2 m of the class 1
5463 streams, with equipment exclusion zones extended on steep slopes for up to 30.5 m. Within the
5464 SMZ no more than half the trees greater than 204 mm (8 in) diameter at breast height (DBH) can
5465 be removed, and trees retained must be representative of the pre-harvest stand. In no case,
5466 however, can stocking levels of leave trees be reduced to less than 217 trees per hectare. The
5467 objectives of this study were to fill the information gap in this region by: (1) evaluating the
5468 performance of 15.2 m SMZs retained during harvest activities for protecting against adverse
5469 changes in summer maximum stream temperatures, (2) quantifying the level of timber removal
5470 occurring within operational SMZs that may help explain any observed changes, and (3)
5471 Evaluating fish response that may be associated with a stream temperature change. Data for
5472 stream temperature and fish population response was collected for 30 harvest reaches in western

5473 Montana (northern Rocky Mountain Region), for a minimum of one-year pre- and one-year post-
5474 harvest. Data for stream temperatures and fish populations were also collected from unharvested
5475 reference reaches upstream from the harvest sites as a control. Temperature data was collected
5476 with Optic StowAway™ and StowAway TidBit™ digital temperature loggers manufactured by
5477 Onset Computer Corporation. Shade over the stream surface was not directly measured in this
5478 study. Canopy cover was estimated using a combination of simulation modeling and using a
5479 concave spherical densiometer. Fish populations were estimated for 100 m reaches at study sites
5480 using an electro-fishing pass of capture method. Linear mixed effects models were used to
5481 analyze the relationship between year, stream position, harvest, fish populations and stream
5482 temperatures. The results showed that within harvest areas, the mean basal area (BA) declined
5483 from 30.2 m²/ha pre-harvest to 26.4 m²/ha post-harvest (mean = -13%, range from -32% to
5484 0%). Windthrow further reduced the mean BA to 25.9 m²/ha (mean = -2%, range = -32% -0%).
5485 Changes in mean canopy cover were not significant based on the simulation modeling (-3%), or
5486 densiometer readings (+1%). Results of the model for the effect of harvest on stream
5487 temperature showed no detectable increase in treatment streams relative to control streams. The
5488 estimated mean site level response in maximum weekly maximum temperatures (MWMT)
5489 varied from -2.1 °C to +3.3 °C. Overall, 20 of 30 sites had estimated site level response within
5490 ±0.5 °C. There were five sites that had an estimated site-level response greater than 0.5 °C (i.e.
5491 warming) and five sites that had an estimated site level response less than -0.5 °C (i.e. cooling).
5492 Results for the fish population showed approximately 7% increase in trout population from pre-
5493 harvest to post-harvest, but this difference was not significant. The authors conclude that the
5494 results suggest that Montana's 15.2 m SMZs retained during timber harvest activities are highly
5495 protective (change <0.5°C) of stream temperatures.

5496

5497 **LW**

5498

5499 Sobota et al., 2006

5500

5501 Sobota, D. J., Gregory, S. V., & Sickle, J. V. (2006). Riparian tree fall directionality and
5502 modeling large wood recruitment to streams. *Canadian Journal of Forest Research*, 36(5), 1243–
5503 1254. <https://doi.org/10.1139/x06-022>

5504

5505 The objectives of this study were to evaluate patterns of riparian tree fall directions in diverse
5506 environmental conditions and evaluate correlations with tree characteristics, forest structural
5507 variables, and topographic features. Specifically, the authors were interested in correlations
5508 between fall directionality and tree species type, tree size, riparian forest structure, and valley
5509 topography (side slope). Data was collected from 21 field sites located west of the Cascade
5510 Mountains crest (11 sites: Coast Range and west slopes of the Cascades), and in the interior
5511 Columbia Basin (10 sites: east slopes of the Cascades, Blue Mountains, and Northern Rockies)

5512 of Oregon, Washington, Idaho, and Montana, USA. Streams were second- to fourth-order
5513 channels and had riparian forests that were approximately 40 to >200 years old. The location of
5514 specific study reaches (200–300 m stream length) on each stream were selected randomly.
5515 Minimum size criteria for a fallen tree in this study were diameter at breast height (DBH) of 0.1
5516 m and height of 5 m. All fallen trees up to 50 m slope distance from stream or the first 100 trees
5517 were measured at all sites. Tree fall direction was standardized among sites by streamside
5518 location (upstream = 0° and 360°; toward stream = 90°; downstream = 180°; away from stream =
5519 –90° and 270°). Spearman rank correlations were used to compare site level statistics of tree fall
5520 directions with physical and riparian forest characteristics. Then trees were pooled among sites
5521 and classified by species for analysis of species, tree size, and valley side slope effects. To avoid
5522 small sample sizes species were grouped by side slope categories (< 40%, >40%). Average
5523 direction of tree fall by site was significantly correlated with valley constraint (Spearman $r = -$
5524 0.53; $P = 0.02$). Average direction of tree fall by site was weakly correlated with active channel
5525 width, tree stem density, and basal area ($P > 0.05$), with Spearman r coefficients of 0.22, –0.21,
5526 and 0.39, respectively. Trees on valley side slopes >40% for each species had a 95% CI that only
5527 included falls directly towards the stream channel; trees on side slopes <40% had a 95% CI for
5528 mean fall direction that included directly upstream, downstream, away from the stream, towards
5529 the stream, or all four directions simultaneously (consistent with random fall directions),
5530 depending on species. Tree size was only different between side slope categories for coastal
5531 Douglas fir on >40% side slopes which had a median DBH 1.2 to 1.9 times greater than trees on
5532 <40% side slopes. Also, red alder trees on side slopes > 40% had a median DBH 1.1 to 1.6 times
5533 greater than on side slopes < 40%. Model projections of LW recruitment calibrated with the
5534 results of the spearman rank correlations estimated that sites with uniform steep side slopes
5535 (>40%) produced between 1.5 (first resolution) to 2.4 (second resolution) times more in stream
5536 LW by number of tree boles than sites with uniform moderate side slopes (< 40%). The authors
5537 interpret their results as evidence that edaphic, topographic, and hydrologic characteristics are
5538 related to greater variability of tree fall directions on moderate slopes than on steep slopes. The
5539 authors conclude that models that use tree fall directions in predictions of LW recruitment should
5540 consider stream valley topography. The authors warn that while side slope categories (>40%,
5541 <40%) was the strongest predictor of tree fall direction in this study, they believe the differences
5542 in tree fall direction between these categories mainly characterized differences between fluvial
5543 (88% of moderate slope sites) and hillslope landforms (71% of steep slope sites). They suggest
5544 that the Implications from this study are most applicable to small- to medium-size streams
5545 (second- to fourth-order) in mountainous regions where sustained large wood recruitment from
5546 riparian forest mortality is the significant management concern.

5547

5548 **LW**

5549

5550 Schuett-Hames & Stewart, 2019a

5551

5552 Schuett-Hames, D., & Stewart, G. (2019a). Post-Harvest Change in Stand Structure, Tree
5553 Mortality and Tree Fall in Eastern Washington Riparian Buffers: Comparison of the Standard
5554 and All Available Shade Rules for the Fish-Bearing Streams in the Mixed Conifer Timber Habitat
5555 Type Under Washington's Forest Practices Habitat Conservation Plan. Cooperative Monitoring
5556 Evaluation and Research Report CMER. Washington State Forest Practices Adaptive
5557 Management Program. Washington Department of Natural Resources, Olympia, WA.

5558

5559 This report is a comparative analysis of the differences in stand structure, tree fall, and LW
5560 recruitment between riparian sites of eastern Washington harvested under the current Standard
5561 Shade Rule (SR), under the All-Available Shade rule (AAS), and unharvested reference sites
5562 (REF). Both shade rules have a 30-ft no-cut buffer (core zone) immediately adjacent to the
5563 stream. The SR prescription allows thinning in the buffer zone 30-75 feet (inner zone) from the
5564 stream while the AAS prescription requires retention of all shade providing trees in this area.
5565 Post-harvest surveys were completed at each site one–two years and five years post-harvest. A
5566 census was done of all standing trees ≥ 4 inches diameter at breast height (DBH) within 75 feet
5567 (horizontal distance) of the channel on both sides of the stream in each treatment and reference
5568 reach. The condition (live or dead), species, canopy class, and DBH were recorded for each tree.
5569 Dead or fallen trees with a decay class of 1 or 2 were classified as post-harvest mortality and a
5570 mortality agent was recorded (e.g. wind, erosion, suppression, fire, insects, disease, and physical
5571 damage). Metrics were calculated separately for regulatory zones defined by horizontal distance
5572 from the channel, including the core zone (0–30 feet) and inner zone (30–75 feet) and the
5573 combined core and inner zone (the full RMZ). Mixed model analysis was used to evaluate
5574 differences in treatment response. Results showed Cumulative wood recruitment from tree fall
5575 over the five-year post-harvest interval was highest in the SR group, lower in the AAS group and
5576 lowest in the REF group. The SR and AAS rates by volume were nearly 300% and 50% higher
5577 than the REF rates, respectively. The mixed model comparisons indicated that the frequency of
5578 wood input from fallen trees was significantly greater in SR group compared to both the REF
5579 and AAS groups ($p < 0.001$), while the difference between REF and AAS groups was not
5580 significant. Over 60% of pieces recruited from AAS and SR fallen trees consisted of stems with
5581 attached rootwads (SWAR), double the proportion in the REF sites. The REF-AAS and REF-SR
5582 differences in recruitment of SWAR pieces were significant ($p < 0.001$). Most recruiting fallen
5583 trees originated in the core zone (76%, 72%, and 64% for the REF, AAS and SR groups,
5584 respectively), while the proportion from the inner zone (30–75 feet from the stream) was ~10%
5585 greater for the SR group compared to the AAS and REF groups. The authors interpret the results
5586 and conclude that harvest of the adjacent stand outside the RMZ appeared to alter the spatial
5587 pattern of wood recruitment from fallen trees, increasing recruitment from trees located farther
5588 from the stream. Recruitment of fallen trees from the inner zone of the AAS and SR sites were
5589 two and four times the rate for the inner zones of the unharvested reference sites due to increased
5590 tree fall from wind disturbance in the buffers after harvest of the adjacent stand, as reported in
5591 other studies. It is important to note that this was a short-term study (5 years). The authors note
5592 that LW recruitment is a process that can change over decadal time scales. Adding that thinning

5593 and post-harvest mortality also reduced the standing stock of trees available for wood
5594 recruitment in the SR and AAS RMZs compared to unharvested REF RMZs.

5595

5596 **Litter and LW**

5597

5598 Six et al., 2022

5599

5600 Six, L. J., Bilby, R. E., Reiter, M., James, P., & Villarin, L. (2022). Effects of current forest
5601 practices on organic matter dynamics in headwater streams at the Trask River watershed,
5602 Oregon. *Trees, Forests and People*, 8, 100233. <https://doi.org/10.1016/j.tfp.2022.100233>

5603

5604 This study investigates the effects of different riparian timber harvest intensities on changes in
5605 canopy cover, and litter input into streams and litter transport downstream. The objective of this
5606 study was to investigate whether differing levels of tree retention adjacent to the channel altered
5607 coarse particulate organic matter (CPOM) delivery, retention, and transport. The authors
5608 hypothesized an inverse relationship between tree removal and litter delivery (i.e., increase in
5609 tree removal adjacent to the channel would result in a reduction of litter delivery). Data was
5610 collected for leaf litter in streamside litter traps, canopy cover percentage using hemispherical
5611 photos in-stream LW, and litter retention in stream flume litter traps pre- and post-treatment at
5612 five watersheds of the Trask River in the northern Oregon Coast range. The experimental design
5613 included three treatment watersheds: clearcut with no leave trees or retention buffer (CC),
5614 clearcut with leave trees (CC w/LT; retention of 5 trees per hectare/2 trees per acre), and clearcut
5615 with 15 m wide retention buffer (CC c/B) and two uncut references (REF 1, and 2) along
5616 headwater streams. Because there were no replication sites for treatments, data was analyzed
5617 using descriptive and graphical summaries of the data (i.e., no quantitative statistical analysis).
5618 Results showed a reduction of canopy cover from 91.4% to 34.4% in the clearcut treatment with
5619 no leave trees, from 89.8% to 76.1% in the clearcut treatment with leave trees, and from 89.5%
5620 to 86.9% in the clearcut treatment with the 15 m retention buffer. Change in canopy cover in the
5621 reference streams was < 1% for both reaches. Post harvest litter delivery decreased for the
5622 clearcut with no leave trees but increased for both the clearcut with leave tree and clear cut with
5623 retention buffer. The number of logjams, the total weight of logjams, and the volume of LW in
5624 streams increased for all treatment sites. The results of this study were consistent with similar
5625 studies and provide supporting evidence that riparian timber harvest can affect litter and LW
5626 delivery into and retention in streams.

5627

5628 **Shade and LW**

5629

5630 Schuett-Hames et al., 2011

5631

5632 Dave Schuett-Hames, Ashley Roorbach, Robert Conrad. 2011. Results of the Westside Type N
5633 Buffer Characteristics, Integrity and Function Study Final Report. Cooperative Monitoring
5634 Evaluation and Research Report, CMER 12-1201. Washington Department of Natural Resources,
5635 Olympia, WA.

5636

5637 This report presents the results from the Washington State Westside Type N Buffer
5638 Characteristics, Integrity and Function (BCIF) study. The purpose of the study was to evaluate
5639 the effects of westside riparian timber harvest prescriptions for Type Np (perennial non-fish-
5640 bearing) streams on resource objectives (riparian stand tree mortality, wood recruitment, channel
5641 debris, shade, and soil disturbance) described in the Forest and Fish Report of 1999. Three
5642 treatment prescriptions were evaluated, 1) clearcut harvest to the edge of the stream (CC) at eight
5643 sites, 50-foot-wide no-cut-buffers (50-ft) at 13 sites, and 56-foot radius circular no-cut-buffer at
5644 the perennial initiation point (PIP) at three sites (not used in statistical analysis due to small
5645 sample sizes). Each treatment site was paired with an uncut reference site as a control. The CC
5646 and 50-ft treatments were compared with treatment sites at three time periods (the first 1-3 years,
5647 years 4-5, and the whole 5-year period). Differences in variable mean values were checked for
5648 statistical significance between treatment and reference streams using non-parametric Mann-
5649 Whitney U tests. Tree fall rates (annual fall rates of live and dead standing stems combined) was
5650 over 8 times and 5 times higher in the 50-foot buffers than in the reference buffers 3 years after
5651 treatment when compared as a percentage of standing trees and as trees/acre/yr, respectively.
5652 These differences were significant for both metrics ($p \leq 0.001$). In the period 4-5 years post
5653 treatment rate of tree uprooting decreased but rate of stem breakage increased in the 50-foot
5654 buffer. For this period only the percentage of broken trees were significantly different (higher)
5655 than what was observed in the reference buffers. Over the entire five-year period, the percentages
5656 of standing trees that were uprooted and broken (as well as the combined total) were
5657 significantly greater in the 50-foot buffer. Wind was the dominant tree fall process, accounting
5658 for nearly 75% of combined fallen trees, 11% fell from other trees falling against them and 1.8%
5659 of fallen trees fell from bank erosion. Differences in mortality followed a similar pattern to tree
5660 fall rates. In the 50-foot buffer sites mortality rates were significantly higher (3.5 times higher)
5661 than in the reference sites for the first three years following harvest. However, in years 4-5
5662 mortality rates increased in the reference buffers after high-intensity storms resulting in non-
5663 significant differences in mortality during this period. The cumulative percentage of live trees
5664 that died over the entire five-year period was 27.3% in the 50-ft buffers compared to 13.6% in
5665 the reference reaches, but the difference was not statistically significant. This was likely because
5666 of the high variability in mortality between sites in the 50-foot buffers. LW recruitment into the
5667 channel after treatment was higher in the 50-ft buffers than in the reference patches during the
5668 first three years after harvest, over 8 times higher in pieces/acre/yr and over 14 times higher in
5669 volume/acre/yr. In years 4-5 after harvest LW recruitment decreased in the 50-ft buffers and
5670 increased in the reference patches, and the number of recruited LW pieces/acre/yr was greater in

5671 the reference patches, although the volume of LW recruited was greater in the 50-ft buffers. For
5672 the entire first 5 years after harvest, the 50-ft buffers recruited about twice the number of LW
5673 pieces recruited in the reference patches, and over 3 times the volume. The CC treatment,
5674 unsurprisingly, had significantly lower LW recruitment following harvest relative to the reference
5675 streams. Mean overhead shade (from trees and tall shrubs) was 13% lower in the 50-ft treatment,
5676 and 77% lower in the CC treatment relative to reference streams. The CC treatment, however,
5677 increased by 25% five years after harvest relative to values recorded 1-year following harvest.
5678 The implications of these results suggest that immediate and direct changes in stand structure,
5679 canopy cover, and LW are most severe for clear-cut treatments, but that the 50-foot buffer
5680 treatment showed an increase in LW and stand mortality, and a decrease in shade over the five-
5681 year period. Limitations of this study were the lack of pre-harvest data and the relatively short
5682 time-period (5-years) in evaluating impacts that may last for several decades.

5683

5684 Schuett-Hames & Stewart, 2019b (BCIF)

5685

5686 Schuett-Hames, D & Stewart, G. (BCIF), (2019). Changes in stand structure, buffer tree
5687 mortality and riparian-associated functions 10 years after timber harvest adjacent to non-fish-
5688 bearing perennial streams in western Washington. Cooperative Monitoring Evaluation and
5689 Research Report. Washington State Forest Practices Adaptive Management Program. Washington
5690 Department of Natural Resources, Olympia, WA.

5691

5692 This paper presents a 10 -year follow-up to the results of the BCIF report (Schuett-Hames et al.,
5693 2012) that originally presented 5-year post-treatment results. Over the 10-year period stand
5694 mortality in the 50-ft buffer treatment stabilized and showed a cumulative 14.1% reduction in
5695 live basal, while the reference stands showed a 2.7% increase in live basal area. The differences
5696 in these values were not significant. Cumulative LW recruited into stream channel over the 10-
5697 period was double in the 50-ft treatment streams than in the reference streams. However, the
5698 majority of the LW recruited in the 50-ft treatment streams came to rest above the streams,
5699 providing shade but not affecting streamflow, pool formation, or sediment storage. Further, while
5700 the 50-ft buffer treatment provided more LW recruitment in the short-term (10-years), the authors
5701 speculate there is a reduction in future LW recruitment potential given the removal of trees
5702 outside the 50-ft buffer. Canopy cover in the 50-ft treatment streams recovered to similar
5703 percentages as the reference's streams by the end of the 10-year period. The authors speculate
5704 that the 50-ft buffer was better at maintaining resource objectives than the clearcut but propose
5705 that the narrow buffers presented variable increases in mortality (specifically increased
5706 susceptibility to windthrow) and recommend further research before drawing definitive
5707 conclusions.

5708

5709 **Riparian thinning effects on shade, light, and temperature**

5710

5711 Roon et al., 2021a

5712

5713 Roon, D.A., Dunham, J.B., Groom, J.D., 2021. Shade, light, and stream temperature responses to
5714 riparian thinning in second-growth redwood forests of northern California. PLOS ONE 16,
5715 e0246822. <https://doi.org/10.1371/journal.pone.0246822>

5716

5717 The purpose of this study was to evaluate the effects of riparian thinning on shade, light, and
5718 temperature in three watersheds located in second-growth redwood stands in northern California.
5719 The objectives of this study were to evaluate: 1) the effects of experimental riparian thinning
5720 treatments on shade and light conditions; 2) how changes in shade and light associated with
5721 thinning affected stream temperatures at a reach-scale both locally and downstream; 3) how
5722 thermal responses varied seasonally; and 4) how these thermal responses were expressed across
5723 the broader thermal regime to gain a more complete understanding of thinning on stream
5724 temperatures in these watersheds. This study took place between 2016 and 2018 with thinning
5725 treatments applied during 2017 giving 1-year pre-treatment and 1-year of post-treatment data.
5726 Two study sites prescribed treatment on one side of the stream of a 45 m buffer width with a 22.5
5727 m inner zone with 85% canopy retention and a 22.5 m outer zone that retained 70% canopy
5728 cover (Green Diamond Resource Company, Tectah watershed). At the third treatment site
5729 thinning prescriptions included removal of up to 40% of the basal area within the riparian zone
5730 on slopes less than 20% on both sides of the channel along a ~100–150 m reach (Lost Man
5731 watershed, Redwood national park). Control reaches were located upstream from treatment
5732 reaches. Data analysis was conducted separately for each experimental watershed (i.e., 1 Lost
5733 man site, 2 Tectah sites). Stream temperature was collected using digital sensors; solar radiation
5734 was measured using silicon pyranometers; riparian shade was measured using hemispherical
5735 photography. A classical BACI analysis was performed to test the effects of riparian thinning on
5736 shade, light, and stream temperature using linear-effects models. Results for the Tectah
5737 watershed showed a significant reduction in canopy closure by a mean of 18.7%, (95% CI: -21.0,
5738 -16.3) and a significant reduction of effective shade by a mean of 23.0% (-25.8, -20.1) one-year
5739 post treatment. In the Lost man watershed, a non-significant reduction of mean shade by 4.1% (-
5740 8.0, -0.5), and mean canopy closure by 1.9% was observed in 2018. Results for below canopy
5741 light availability showed significant increases by a mean of 33% (27.3, 38.5) in the Tectah
5742 watershed, and non-significant increases in Lost man watershed of 2.5% (-1.6, 5.6) by 2018.
5743 Results for stream temperature changes showed variation seasonally and between watersheds.
5744 The Lost Man watershed showed no significant changes in average daily maximum, maximum
5745 weekly average of the maximum (MWMT), average daily mean, or maximum weekly average of
5746 the mean (MWAT). In the Tectah watershed, MWMT increased during spring by a mean of 1.7°C
5747 (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)
5748 and increased in downstream reaches during spring by a mean of 1.0°C (0.0, 2.0) and summer by
5749 a mean of 1.4°C (0.3, 2.6). Thermal variability of streams in the Tectah watershed were most
5750 pronounced during summer increasing the daily range by a mean of 2.5°C (95% CI:

5751 1.6, 3.4) and variance by a mean of 1.6°C (0.7, 2.5), but also increased during spring (daily
5752 range: 0.5°C; variance: 0.3°C) and fall (daily range: 0.4°C; variance: 0.1°C). Increases in thermal
5753 variability in downstream reaches were limited to summer (daily range: 0.7°C; variance: 0.5°C).
5754 Again, no significant changes in stream and downstream temperature variability were detected in
5755 the Lost Man watershed. In the Techtah watersheds the frequency of days with temperatures
5756 greater than 16°C increased in summer by a mean of 42.9 more days (95% CI: 31.5, 53.8) in
5757 thinned reaches and a mean of 16.3 more days (6.1, 27.4) in downstream reaches. Temperatures
5758 greater than 16°C persisted for a mean duration of 31.1 more consecutive days (21.0, 41.1) in
5759 thinned reaches and 11.6 more consecutive days (3.9, 20.0) in downstream reaches under the
5760 BACI analysis. The authors conclude that responses to the experimental riparian thinning
5761 treatments we evaluated differed greatly depending on treatment intensity. For example, they
5762 interpret their results as evidence that that changes in shade of 5% or less caused minimal
5763 changes in temperature while reductions in shade of 20–30% resulted in much larger increases in
5764 temperature. However, the authors warn that their data only evaluated immediate (1-year-post-
5765 treatment) changes in stream shade and temperatures. Also, the study was conducted in relatively
5766 small (< 10 km²) coastal watersheds and may not apply to larger watersheds of different regions.

5767

5768 **Sediment**

5769

5770 Safeeq et al., 2020

5771

5772 Safeeq, M., Grant, G.E., Lewis, S.L., Hayes, S.K., 2020. Disentangling effects of forest harvest
5773 on long-term hydrologic and sediment dynamics, western Cascades, Oregon. *Journal of*
5774 *Hydrology* 580, 124259. <https://doi.org/10.1016/j.jhydrol.2019.124259>

5775

5776 The purpose of this study was to separate and investigate the effects of changes in streamflow
5777 and sediment supply due to disturbances (specifically timber harvest), on sediment transport into
5778 streams. Timber harvest affects both streamflow and sediment supply simultaneously. The
5779 researchers used a reverse regression technique to evaluate the relative and absolute importance
5780 of changes in streamflow versus changes in sediment supply on sediment transport. The
5781 technique was applied to long-term data collected from two paired experimental watersheds in
5782 the H.J. Andrews Experimental Forest, Oregon. The two watersheds were paired by size, aspect,
5783 and topography. The treatment watershed was 100% clearcut during the period from 1962-1966,
5784 broadcast burned in 1966, and re-seeded in 1968. Streamflow, and sediment data were taken
5785 intermittently, and after large storm events from 1952 (pre-harvest) through 1988 for suspended
5786 sediment data, and 2016 for sediment bedload. The control watershed was forested, and had no
5787 treatments (e.g., harvest) during the study period. The results that considered the effects of
5788 harvest on streamflow alone showed an increase in annual water yield in the treatment watershed

5789 by 10% (136 mm/year) over the 51-year post-treatment period. There were no significant
5790 changes in precipitation patterns before or after harvest. Further, the patterns of streamflow in the
5791 control watershed showed diverging patterns in streamflow after the harvest period. The authors
5792 state that these patterns strongly suggest that the increase in streamflow in the treatment
5793 watershed was caused by timber harvest. The results for post-treatment sediment yields showed
5794 suspended load declined to pre-treatment levels in the first two decades following treatment,
5795 bedload remained elevated, causing the bedload proportion of the total load to increase through
5796 time. Changes in streamflow alone account for 477 Mg/km² (10%) of the suspended load and
5797 113 Mg/km² (5%) of the bedload over the post-treatment period. Increase in suspended sediment
5798 yield due to increase in sediment supply is 84% of the measured post-treatment total suspended
5799 sediment yield. In terms of bedload, 93% of the total measured bedload yield during the
5800 posttreatment period can be attributed to an increase in sediment supply. The authors interpret
5801 these results as evidence that while streamflow alone can cause a modest increase in sediment
5802 transport, it is negligible compared to the increases in sediment transport following harvest.
5803 Following harvest, changes on streamflow alone was estimated in being responsible for < 10% of
5804 the resulting suspended sediment transported into streams, while the increase in sediment supply
5805 due to harvest disturbance was responsible for >90%. The authors suggest these results provide
5806 evidence for a need to investigate thresholds for specific watershed management regimes to
5807 ameliorate these impacts following harvest, or thinning treatments. Also, the sharp increases in
5808 sediment transport following logging can be confidently attributed to the increase in sediment
5809 supply and delivery to streams due to the ground disturbances associated with logging rather than
5810 increased streamflow.

5811

5812 **Stream Temperature**

5813

5814 Reiter et al., 2020

5815

5816 Reiter, M., Johnson, S. L., Homyack, J., Jones, J. E., & James, P. L. (2020). Summer stream
5817 temperature changes following forest harvest in the headwaters of the Trask River watershed,
5818 Oregon Coast Range. *Ecohydrology*, 13(3), e2178. <https://doi.org/10.1002/eco.2178>

5819

5820 This paper investigates the effects of different riparian forest harvest treatments on stream
5821 temperature. Stream temperature data was collected from 2006 to 2016 for multiple small (<50
5822 ha), non-fish-bearing headwater stream watersheds in the Trask River Watershed of the
5823 northwestern Oregon Coast range. The experiment followed a BACI design with four treatments,
5824 1) clearcut, no buffer (CC_NB; n = 4), 2) clearcut with 10-m no cut buffer (CC_B; n = 3), 3)
5825 Thinning with 10 m no-cut buffer (TH_B; n = 1), and 4) unharvested, reference streams (REF; n
5826 = 7). Temperature data was collected at 30-minute increments for all streams using continuously
5827 recording thermistors. Harvest operations occurred in the Summer of 2012 giving 6 summers of

5828 pre-treatment and 4 summers of post-treatment data collection. Temperature data was separated
5829 into 5th, 25th, 50th, 75th, and 95th percentiles, with each percentile being treated as independent
5830 response variables in a linear mixed model. Treatments were compared to reference watersheds
5831 to check for significant differences in temperature percentiles. For ecological context, the
5832 researchers also quantified the percentage of summer where temperatures were above 16 and 15
5833 °C, the preferred thermal regime limits for two local amphibian larvae (coastal tailed frog,
5834 coastal giant salamander). Results showed that even the small (10 m buffer; CC_B, TH_B) buffer
5835 was efficient in maintaining similar temperature changes throughout the summers compared to
5836 reference streams. There were no significant changes in the buffered watersheds with
5837 temperature responses in these watersheds ranging from negative values to negative values close
5838 to zero. The treatments with no buffer (CC_NB), however, showed significant increases in
5839 temperature for all percentiles with the greatest increases occurring in the 95th percentile,
5840 showing a mean increase of 3.6 °C (SE = 0.4). For the 5th percentile, the CC_NB also showed a
5841 mean temperature response 1.7°C (SE = 0.3; range from 1.5 - 2.8°C). Temperature changes were
5842 more severe in the CC_NB watersheds with no leave trees (4.2 and 4.4°C), however, this
5843 difference was not analyzed. The percentage of time the post-harvest, no-buffer treatments spent
5844 above the 16 and 15 °C thresholds were 1.3% and 4.7%, respectively. This was an increase from
5845 pre-harvest values that showed no instances of temperatures above 16°C, and only 0.2% of the
5846 recorded time above 15°C. The authors conclude that their evaluation of temperature responses
5847 as potential biologically significant changes adds context to the changes and fluctuations
5848 observed in each harvest design. While significant changes in mean and percentile changes in
5849 temperature were observed, the amount of time spent above critical temperature thresholds for
5850 important amphibian species was minimal.

5851

5852 **SHD, Stream temperature**

5853

5854 Chan et al., 2004 (Removed from focal list, significant results only apply to fauna)

5855

5856 Chan, S.S., Anderson, P.D., Cissel, J., Larsen, L., Thompson, C., 2004. Variable density
5857 management in Riparian Reserves: lessons learned from an operational study in managed forests
5858 of western Oregon, USA. USDA Forest Service. <https://doi.org/10.1016/j.foreco.2013.06.055>

5859

5860 The purpose of this study was to assess the ability of variable retention thinning and riparian
5861 buffers at accelerating late-seral habitat, facilitating rare species management, and maintaining
5862 ecological functions within riparian zones of 40–70-year-old headwater forests in western
5863 Oregon. This study evaluated 13 separate sites each averaging ~ 100 ha whereby 4 buffer width
5864 treatments adjacent to variable retention thinning prescriptions were assessed. Buffer treatments
5865 include: (1) one site potential tree; (2) two-site potential trees; (3) variable buffer width based on
5866 vegetation and/or topographic site factors; (4) streamside buffer of only the first tree whereby

5867 thinning treatments applied up to 6 m of stream. Thinning treatments included: (1) Unthinned
5868 control - 500-750 trees per hectare; (2) High density retention - 70-75% of area thinned to 300
5869 TPH, 25-30% unthinned riparian reserves or leave islands; (3) Moderate density retention - 60-
5870 65% area thinned to 200 TPH, 25-30% unthinned riparian reserves or leave islands with 10%
5871 circular patch openings; (4) Variable density retention - 10% area thinned to 100 TPH, 25-30%
5872 thinned to 200 TPH, 25-30% thinned to 300 TPH, 20-30% unthinned riparian reserves or leave
5873 islands with 10% circular patch openings. Variables measured include stand development
5874 metrics, understory vegetation, microclimate, aquatic ecology, invertebrates, lichens, and
5875 bryophytes. Early findings from this study show that relatively small changes in the riparian
5876 environment are attributed to different residual thinning densities and different buffer widths.
5877 According to the results, the most suitable habitat for many species of fauna is consistently found
5878 within 5 m of the stream. The largest changes in relative humidity in warm and dry summer
5879 conditions occur within 15 m of the stream channel and begin to stabilize at 25 m. In summary,
5880 the early findings of this study indicate the near-stream riparian environment provides critical
5881 functions and habitat for a wide variety of organisms.

5882

5883 **Sediment**

5884

5885 Reiter et al., 2009

5886

5887 Reiter, M., Heffner, J. T., Beech, S., Turner, T., & Bilby, R. E. (2009). Temporal and Spatial
5888 Turbidity Patterns Over 30 Years in a Managed Forest of Western Washington 1. *JAWRA Journal*
5889 *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)
5890 [1688.2009.00323.x](https://doi.org/10.1111/j.1752-1688.2009.00323.x)

5891

5892 This study evaluates the efficacy of the changes in a forest practices plan developed in 1974 to
5893 reduce sediment inputs into streams in the Deschutes River watershed of western Washington. To
5894 test this, the researchers analyzed 30 years of data (1975-2005) on water levels, discharge,
5895 suspended sediment, turbidity, and water and air temperature from four permanent sampling sites
5896 representing a range of basin sizes from small tributary headwaters to the mainstem of the
5897 Deschutes River. In the 1970s roughly 30% of the watershed had been harvested and
5898 approximately 63% of the existing road network had been constructed. Timber harvest continued
5899 until the early 1990s and the road network was completed in the late 1970s but updated to
5900 include culverts and sediment traps in the early 2000s. The researchers used turbidity as a proxy
5901 for suspended sediment correlation and corrected for typical seasonal increases in streamflow.
5902 The results showed a declining trend in turbidity at all permanent sampling sites during the study
5903 period even with active forest management. Following the road construction and harvest
5904 activities of the 1980s turbidity levels continued to decline until the year 2000 when they

5905 returned to pre-logging levels. The authors interpret these results as evidence that management's
5906 increased attention to reducing sediment is responsible for the reduction in sediment transport.

5907

5908 **Effect of debris torrents on shade, vegetation, and stream temperature**

5909

5910 D'Souza et al., 2011

5911

5912 D'Souza, L.E., Reiter, M., Six, L.J., Bilby, R.E., 2011. Response of vegetation, shade and stream
5913 temperature to debris torrents in two western Oregon watersheds. *Forest Ecology and*
5914 *Management* 261, 2157–2167. <https://doi.org/10.1016/j.foreco.2011.03.015>

5915

5916 The purpose of this study was to examine the effects of debris torrents on vegetation, shade, and
5917 stream temperature eight years after an extreme storm-related disturbance. This study examined
5918 two separate managed watersheds which were affected by storm-related debris torrents in 1996.
5919 This study addressed several questions regarding the patterns and rate of vegetation, shade and
5920 water temperature change post-disturbance: (1) What is the relationship between vegetation and
5921 local landform and substrate types along the study streams? (2) Does vegetation composition and
5922 structure, stream shade and water temperature in debris torrented streams differ between the two
5923 watersheds? and (3) How does recovery of stream temperature relate to vegetation and shade
5924 recovery and does this differ through time between watersheds? Data was gathered from
5925 multiple headwater streams following the disturbance in 1996 at 2 managed watersheds: the
5926 Williams River watershed (WRW), and the Calapooia River watershed (CRW). Data for stream
5927 temperature, to analyze stream temperature recovery, was collected immediately following the
5928 disturbance event in 5 streams, 3 at the CRW (2 disturbed; 1 reference), and 3 at the WRW (1
5929 disturbed, 1 reference) and for 8 years through the summer of 2004. Eight years post-disturbance
5930 12 disturbed streams (n = 6 for each watershed) were selected for data collection to examine the
5931 relationships between riparian vegetation, shade, and stream temperatures. Data on landform,
5932 substrate, and vegetation (density, species, and seedlings) were collected at each stream. Stream
5933 shade was estimated using hemispherical photographs taken 1 m above the stream center during
5934 summer and winter months and compared using t-tests. Stream temperature data was collected
5935 using continuously recording thermistors. Data were averaged and analyzed using t-tests, chi-
5936 square tests, simple linear regression, Pearson's correlation coefficient, and analysis of
5937 covariance. Results from this study show early successional species red alder and willow species
5938 dominated areas affected by debris torrents. All red alder variables (density, basal area, and
5939 height) showed a significant relationship with vegetation-related shade. Red alder showed a
5940 significantly higher density ($p = 0.0277$) and basal area ($p = 0.0367$) in the WRW sites. While
5941 stem density of red alder was similar in both watersheds, the size of the trees differed suggesting
5942 that colonization and/or growth of red alder in the WRW occurred more rapidly than in the CRW.
5943 However, there was no statistical difference in landforms or site factors between watersheds that

5944 explained these differences. The only correlations found were a negative relationship between
5945 alder density and rock; and a positive relationship between alder basal area and moss suggesting
5946 a relationship between moisture availability and red alder establishment and growth. The authors
5947 note that the WRW sites experienced greater precipitation in the years following disturbance and
5948 may have contributed to the greater growth rates of red alder, but no analysis was conducted.
5949 Total shade was also significantly higher in the WRW ($p = 0.0049$). Mean maximum daily
5950 temperature fluctuations ($p = 0.0483$), and 7-day maximum temperatures ($p=0,0483$) were also
5951 significantly lower in the WRW streams. Mean max daily stream temperatures were lower in the
5952 WRW streams but the difference was not significant ($p = 0.0779$). The authors conclude that
5953 even though the debris torrents resulted in poor soil conditions, the ability of red alder to thrive
5954 in these conditions resulted in rapid recovery of shade and thermal control.

5955

5956 **Stream temperature, shade and climate**

5957

5958 Reiter et al., 2015

5959

5960 Reiter, M., Bilby, R. E., Beech, S., & Heffner, J. (2015). Stream temperature patterns over 35
5961 years in a managed forest of western Washington. *JAWRA Journal of the American Water*
5962 *Resources Association*, 51(5), 1418-1435. <https://doi.org/10.1111/1752-1688.12324>

5963

5964 This study was an analysis of long-term stream temperature data in a western Washington
5965 watershed to evaluate the effects of forest management, before and after implementation of
5966 riparian forest best management practices, and climate change on stream temperatures. Stream
5967 temperature data from four permanent sampling stations in the Deschutes River Watershed.
5968 Stream and air temperature data was analyzed on a monthly basis from 1975-2009. This long-
5969 term dataset allowed for the examination of changes in stream temperature in four basins of
5970 varying size across a period from before stream buffers were implemented, during their
5971 implementation, and several instances of buffer expansion. Because the study period covered
5972 such a long time the changes in stream temperature based on climate change needed to be
5973 accounted for as well. The recovery of shade was estimated using the shade recovery function
5974 developed by R. Summers of Oregon State University (1983), whereby stream shade is estimated
5975 by angular canopy density (ACD) as a function of the age of stream-adjacent harvest units. To
5976 detect correlations of stream and air temperature change with land management activity
5977 separately from climate changes the data was fit to a model that included the effects of climate.
5978 The researchers accomplished this with a technique for deriving the residuals between stream
5979 temperature and climate called locally weighted scatterplot smoothing (LOWESS). The four
5980 watersheds varied in size from small (2 sites: Hard Creek, 2.4 km²; Ware Creek, 2.9 km²),
5981 medium (1 site: Thurston Creek, 9.3 km²), and large (1 site: The Deschutes River Station, 150
5982 km²). In the 1970s nominal buffer widths were required along fish-bearing streams, which

5983 expanded in the 1980s (requirements not listed), again in the mid-1990s to 23 m, and again to 30
5984 m in 2001. Methods for stream temperature data collection varied at different periods resulting in
5985 a margin of error for monthly temperatures of 0.14°C for 1975 - 1983, 0.09°C for 1984 – 1999,
5986 and 0.02°C. for 2000 – 2009. Because these margins of error were smaller than what the authors
5987 expected from climate and management, they were not accounted for in confidence intervals and
5988 p-values. The results for air temperature changes showed a statistically significant ($p \leq 0.05$)
5989 increasing trend in regional air temperatures for July TMAX_AIR and June and July
5990 TMIN_AIR. The trend for TMAX_AIR for July resulted in a trend magnitude of +0.07°C per
5991 year, for a total increase of 2.45°C over the 35-year record. For minimum air temperatures the
5992 magnitude of the June trend was +0.03°C per year while July TMIN_AIR had a trend magnitude
5993 of +0.04°C per year. The resulting increases in minimum temperatures for the period of record
5994 are 1.05°C and 1.40°C for June and July TMIN_AIR, respectively. Results for trends in stream
5995 temperature over the 35-year study period without adjustment for climate change showed no
5996 statistically significant trend in water temperature changes for the large watershed, while the
5997 medium watershed (Thurston Creek) showed decreasing trends in TMAX_WAT for June, July,
5998 and August, ranging in magnitude from 0.05°C (August) to 0.08°C (July) per year. For the
5999 smaller watershed, Hard Creek (Ware Creek was not included in this analysis), had significant
6000 decreasing trends in TMAX_WAT for July, August, and September. The magnitude of these
6001 trends was yearly decreases of TMAX_WAT by 0.05, 0.08, and 0.05°C, for July, August, and
6002 September, respectively. Significant changes in trends for TMIN_WAT were only found for the
6003 large basin site with yearly increases of 0.04, 0.03, and 0.04°C for July, August, and September,
6004 respectively. Results for stream temperature trends after adjusting for changes in air temperature
6005 (climate) showed significant decreasing trends in TMAX_WAT for the large basin by 0.04, 0.03,
6006 and 0.04°C yearly, for July, August, and September, respectively. For the medium basin, trends
6007 showed yearly decreases in TMAX_WAT of 0.07, 0.08, 0.06, and 0.03 for June, July, August,
6008 and September, respectively. For the small basin, climate adjusted trends in TMAX_WAT
6009 showed significant decreases in yearly trends by 0.05, 0.08, and 0.05 for July, August, and
6010 September, respectively. When stream temperature was examined with its correlation with
6011 estimated annual shade recovery from initial harvest (indexed by ACD). Significant correlations
6012 were found for monthly temperature metrics that were adjusted for climate, for all basins. The
6013 strongest correlations were for the smallest basin (Ware Creek) with correlation coefficients for
6014 climate adjusted maximum water temperatures (CTMAX_WAT) with ACD valuing -0.66, -0.78,
6015 -0.65, and -0.69 for June, July, August, and September, respectively. Correlation coefficients for
6016 Ware Creek CTMIN_WAT with ACD were -0.46, -0.64, -0.71, and -0.52 for June July, August,
6017 and September respectively. The largest basin (The Deschutes River) only showed significant
6018 correlations of CTMAX_WAT with ACD with July (-0.39) and August (-0.25); and only showed
6019 significant correlations of CTMIN_WAT with ACD for the months of August (+0.27), and
6020 September (+0.37). The authors interpret their results as evidence that following canopy
6021 recovery after implementation of riparian harvest rules the larger mainstem of the Deschutes
6022 River decreased in average maximum temperatures by approximately 1.3 °C when accounting for
6023 climate driven changes. The effects of canopy closure cooling were even more dramatic in the
6024 smaller headwater streams by 2.67 and 1.6 °C during the study period when accounting for
6025 climate driven changes (this includes a 0.5 °C correction based on climate warming). However,

6026 following re-initiation of timber harvest in 2001 for the area, when riparian protection buffers of
6027 30 m minimum were required, there was no detectable change in stream temperatures. The
6028 authors conclude that the results of this study show evidence that implementation of protection
6029 buffers in this area were sufficient in maintaining stream temperatures. Conversely, this study
6030 also shows evidence that despite these protections from land management induced stream
6031 temperature changes, these protections have been somewhat offset by the warming climate
6032 conditions.

6033

6034 **Overstory structure effects on understory light and vegetation**

6035

6036 Giesbrecht et al., 2017 (removed from focal, not relevant to questions, essentially a case study)

6037

6038 Giesbrecht, I.J.W., Saunders, S.C., MacKinnon, A., Lertzman, K.P., 2017. Overstory structure
6039 drives fine-scale coupling of understory light and vegetation in two temperate rainforest
6040 floodplains. *Can. J. For. Res.* 47, 1244–1256. [dx.doi.org/10.1139/cjfr-2016-0466](https://doi.org/10.1139/cjfr-2016-0466)

6041

6042 The purpose of this paper was to characterize the overstory structure and understory light
6043 regimes of temperate rainforest floodplains, and to assess the role of light and other site variables
6044 in driving stand vegetation patterns and processes. This study took place along two 1-ha coastal
6045 BC, Canada floodplain sites. These sites were selected as representative examples of floodplain
6046 forests in the Coastal Temperate Rainforest (CTR) as part of a larger network of long-term, old-
6047 growth monitoring plots. These sites were in the submontane variant of the very wet maritime
6048 subzone of the Coastal Western Hemlock zone (CWHvm1) of the B.C. coast. In each stand, the
6049 largest overstory trees are *Picea sitchensis* (Bong.) Carr., with several individuals taller than 60 m
6050 in height (maximum of 62 to 93 m). Based on coring a sample of main canopy trees, stand age at
6051 Kitlope is at least 95 years. Stand age at Carmanah is at least 350 years, based on a core from a
6052 50 m tall *P. sitchensis*. All trees ≥ 5 cm were measured along with all understory vegetation
6053 within 25 2m x 2m subplots. Stand characteristics were recorded as well as information on gap
6054 origins. Hemispheric canopy photographs were taken to estimate understory light penetration.
6055 Visual estimations of organic material, mineral layer, CWD, and other substrates were taken in
6056 each vegetation subplot. Relationships among measures of light transmission, vegetation
6057 structure, and diversity were analyzed with linear correlation analysis. Nonmetric
6058 multidimensional scaling was used to describe variation in species composition on multivariate
6059 axes. Results from this study show both sites as having a relatively high degree of canopy
6060 openness (11-11.6%) and light transmission (median 18% full sun) compared to many other
6061 tropical and temperate forests. Light transmission at both sites is however significantly lower
6062 than a number of old-growth sites in Quebec and northern BC. The origins of canopy openness
6063 and stand shade differ between both sites indicating distinct stand processes and different stages
6064 of stand development. Further, light levels vary substantially within short distances at each site

6065 reflecting a complex overstory structure. Although results from this study are reflective
6066 specifically of the coastal temperate rainforests of BC, the descriptive assessment of these two
6067 separate floodplain forests reveal a natural disturbance history which fostered a high degree of
6068 canopy openness and structural heterogeneity which may ultimately aid in informing future
6069 temperate rainforest floodplain restoration efforts.

6070

6071 **LW**

6072

6073 Reid & Hassan, 2020

6074

6075 Reid, D. A., & Hassan, M. A. (2020). Response of in-stream wood to riparian timber harvesting:
6076 Field observations and long-term projections. *Water Resources Research*, 56(8),
6077 e2020WR027077. <https://doi.org/10.1029/2020WR027077>

6078

6079 This paper proposes a conceptual model of wood storage response to different harvesting
6080 intensities. The model predicts how LW in streams is expected to change spatially and
6081 temporally following three different harvest patterns. The model was developed with 45 years of
6082 LW data retrieved from the Pacific coastal region of Vancouver Island, British Columbia. The
6083 Carnation Creek watershed, which supports gravel bed forested streams, contains riparian forests
6084 that have received a wide range of harvest plans implemented. During logging in the 1970s and
6085 '80s riparian forests of one region were harvested with buffer widths ranging from 1 – 70 meters
6086 in upstream reaches, and another region with near complete or complete removal of vegetation to
6087 the streams edge in downstream reaches. In-stream wood volume and characteristics data has
6088 been collected in eight of these study reaches since 1973 (pre-harvest). The researchers used this
6089 data with simulation modelling to develop a reach-scale wood budget model that predicts wood
6090 loss and recover patterns for 300 years (1900-2200). This paper has two objectives: (i) to use this
6091 field data and modeling approach to examine LW storage changes, the time to minimum wood
6092 load, and wood load recovery times as a result of riparian timber harvesting and forest
6093 regeneration, and (ii) to describe the characteristics of in stream wood, with particular focus to
6094 spatial and temporal patterns in wood storage over the multidecade scale following harvesting in
6095 riparian areas. The model was based upon the proposed response outlined by Murphy and Koski
6096 (1989). Wood budget responses were estimated using three management scenarios. Scenario 1 is
6097 a no harvest scenario, in this configuration, the loss of wood supply from the landscape has little
6098 to no impact on input from wood mortality or bank erosion, and therefore in-stream storage,
6099 decay, and transport of wood is not affected. Scenario 2 represents partial loss of forested area in
6100 the riparian zone, which will lead to a near-immediate reduction in wood recruitment to the
6101 channel from mortality and bank erosion along harvested areas. Wood decay and other
6102 components of wood loss will exceed rates of input, leading to a reduction in storage until time
6103 T_{min} , the point where wood recruitment equals losses as the forest regrows in riparian areas and

6104 the greatest overall reduction in storage has occurred (ΔS_{max}). Wood storage increases
6105 thereafter, eventually recovering to preharvest levels after time T_{rec} . Scenario 3 represents an
6106 intensive harvest scenario where most of the riparian area has undergone harvesting over a short
6107 period of time, a major reduction of input from bank erosion and mortality occurs. This greater
6108 reduction leads to a much larger ΔS_{max} than in Figure 1b as wood losses exceed recruitment.
6109 However, as the dominant wood sources recover at the same rate, the time to T_{min} and T_{rec} is
6110 similar under both the moderate and intensive harvest scenarios. Results of the model show
6111 evidence that wood storage in streams of harvested reaches, hits its minimum value in 50 years
6112 or more following loss of LW input, decay, and export of current stock. Recovery of LW volume
6113 in-streams following harvest is estimated to take approximately 150-200 years. The pattern and
6114 intensity of the harvesting operation had little effect on LW loss and recovery times but did affect
6115 the estimated magnitude of LW volume loss in the first 50 – 80 years. These results show
6116 evidence that timber harvest has a long-term effect on LW storage and loading dynamics even
6117 with protective buffers. However, buffers can ameliorate the magnitude of LW loss during the
6118 recovery period. The one caveat of this model is it doesn't account for as much variability on
6119 stream configuration or valley morphologies that are likely to affect LW storage.

6120

6121 **Buffers and LW Recruitment**

6122

6123 Grizzel et al., 2000 (Removed)

6124

6125 Grizzel, J., McGowan, M., Smith, D., Beechie, T., 2000. STREAMSIDE BUFFERS AND
6126 LARGE WOODY DEBRIS RECRUITMENT: EVALUATING THE EFFECTIVENESS OF
6127 WATERSHED ANALYSIS PRESCRIPTIONS IN THE NORTH CASCADES REGION
6128 (Timber/Fish/Wildlife Monitoring Advisory Group and the Northwest Indian Fisheries
6129 Commission). fp_tfw_mag1_00_003

6130

6131 This study analyzed the effectiveness of the Washington Watershed Analysis (WWA)
6132 prescriptions at recruiting large woody debris. This study took place at 10 riparian sites
6133 distributed across 5 watershed administrative units in the Northern Cascades of Washington. Ten
6134 sites were randomly chosen with gradients and buffer width classes in compliance with WWA
6135 indices. To analyze WWA effectiveness, debris frequency and size at each site were compared to
6136 targets derived from WWA. In addition, debris recruitment was compared between three buffer
6137 width classes. Geometric mean diameter and geometric mean length of debris was calculated
6138 based on measurements of midpoint diameter and total lengths. This data was then compared to
6139 targets derived from a channel width-dependent regression. Results show post-harvest mortality
6140 substantially decreasing stand density at several sites. In stream frequency targets were met at
6141 most sites; however, debris categorized as "good" for habitat was only achieved at four out of ten
6142 sites. At the time of data collection, a large portion of debris recruited from buffers was either

6143 above or outside the bankfull flow zone. The authors point out that the degree to which the debris
6144 will influence fluvial processes in the future will depend on whether or not they are recruited into
6145 the stream and will also depend on the size and state of decay. The size of debris recruited from
6146 buffers was significantly smaller than recruited from unmanaged old-growth stands.
6147 Interestingly, data shows recruitment occurring from the outermost margins of the widest buffers
6148 (20-30 m, >30 m), suggesting narrow buffers may limit recruitment. The authors point out that
6149 the large degree of variability in recruitment from site to site suggests windthrow as an important
6150 causal factor. In channels oriented perpendicular to damaging winds (east-west), there was a
6151 higher likelihood of potential recruitment as compared to channels oriented parallel to damaging
6152 winds. The authors conclude with multiple recommendations for future study. First, they suggest
6153 integrating habitat inventory with recruitment to achieve a better understanding of relationships.
6154 Second, they suggest future study into the fate of debris suspended above channels given much
6155 of our current understanding is based on assumptions of decay and breakage. Finally, they
6156 recommend study into factors influencing windthrow in riparian buffers.

6157

6158 **Sediment**

6159

6160 Rachels et al., 2020

6161

6162 Rachels, A. A., Bladon, K. D., Bywater-Reyes, S., & Hatten, J. A. (2020). Quantifying effects of
6163 forest harvesting on sources of suspended sediment to an Oregon Coast Range headwater stream.
6164 *Forest Ecology and Management*, 466, 118123. <https://doi.org/10.1016/j.foreco.2020.118123>

6165

6166 This study uses sediment source fingerprinting techniques to quantify the proportional
6167 relationship of sediment sources (hillslope, roads, streambanks) in harvested and un-harvested
6168 watersheds of the Oregon Coast Range. The researchers used sediment traps, and chemical
6169 analysis to estimate the origin of suspended sediment in the stream and to quantify magnitude of
6170 sediment stored in protection buffers. The study included one catchment (Enos Creek) that was
6171 partially clearcut harvested in the summer of 2016 and an unharvested reference catchment
6172 (Scheele Creek) located ~3.5 km northwest of Enos Creek. The paired watersheds had similar
6173 road networks, drainage areas, lithologies and topographies. The treatment watershed was
6174 harvested with a skyline buffer technique in the summer of 2016 under the Oregon Forest
6175 practices Act policy that requires a minimum 15 m no-cut buffer. The proportion of suspended
6176 sediment sources were similar in the harvested ($90.3 \pm 3.4\%$ from stream bank; $7.1 \pm 3.1\%$ from
6177 hillslope) and unharvest ($93.1 \pm 1.8\%$ from streambank; $6.9 \pm 1.8\%$ from hillslope) watersheds.
6178 However, the harvested watershed contained a small portion of sediment from roads ($3.6 \pm$
6179 3.6%), while the unharvested reference watershed suspended sediment contained no sediment
6180 sourced from roads. In the harvested watersheds the sediment mass eroded from the general
6181 harvest areas (96.5 ± 57.0 g) was approximately 10 times greater than the amount trapped in the

6182 riparian buffer (9.1 ± 1.9 g), and 4.6 times greater than the amount of sediment collected from
6183 the unharvested hillslope (21.0 ± 3.3 g). These results suggest that the riparian buffer was
6184 efficient in reducing sediment erosion relative to the harvested area. The caveat of this study was
6185 the limited sample size (1 treatment, 1 paired reference watershed) and does not incorporate the
6186 effects of different watershed physiography on sediment erosion.

6187

6188 **SED**

6189

6190 Puntenney-Desmond et al., 2020

6191

6192 Puntenney-Desmond, K. C., Bladon, K. D., & Silins, U. (2020). Runoff and sediment production
6193 from harvested hillslopes and the riparian area during high intensity rainfall events. *Journal of*
6194 *Hydrology*, 582, 124452. <https://doi.org/10.1016/j.jhydrol.2019.124452>

6195

6196 This study uses simulation modeling to evaluate the differences in run-off rates, sediment
6197 concentrations, and sediment yields between watershed harvested areas, along the interface of
6198 harvested areas and riparian buffers, and within riparian buffers during periods of high-intensity
6199 rainfall events. The model simulations were calibrated with soil and watershed characteristic data
6200 collected from the Star Creek catchment located in southeastern Alberta. Fifteen plots were
6201 selected for rainfall simulations along three transects on a north facing hillslope (aspect: $\sim 358^\circ$)
6202 and along two transects on a southeast facing hillslope (aspect: $\sim 129^\circ$). Each transect consisted
6203 of three plots that were spaced ~ 20 m apart along the planar hillslopes. Each plot was one
6204 square-meter, which was bounded by a three-sided steel frame that was inserted into the soil with
6205 the open side facing down the slope. The plots were located either (a) within the general harvest
6206 area, (b) along the edge of the riparian buffer at the interface with the harvested area, or (c)
6207 within the riparian buffer. The high-intensity rainfall events were calibrated to mimic 100-year,
6208 or greater, storm events of the Northern Rocky Mountains (1-hour high intensity rainfall). The
6209 results showed runoff rates and surface and shallow subsurface were greatest in the buffer areas
6210 than in the harvested areas or in the harvest-buffer interfaces especially during dry conditions.
6211 During the dry condition rainfall simulations, the general pattern of runoff rates (surface/shallow
6212 subsurface flow) was riparian buffer (175.6 ± 17.3 [SE] ml min^{-1}) > harvest-riparian edge
6213 (125.8 ± 18.2 ml min^{-1}) > general harvest area (37.2 ± 8.5 ml min^{-1}). Mean runoff rates within
6214 the riparian buffer plots were greater than within the general harvest area plots ($t = 2.90$, $p = .03$).
6215 Runoff ratios were only statistically greater in the riparian buffer plots ($13.9 \pm 3.1\%$) relative to
6216 the general harvest area ($2.9 \pm 1.5\%$) during the dry conditions. All runoff ratios declined during
6217 the wet condition rainfall simulations relative to the dry condition simulations with no evidence
6218 for differences between any of the plot positions ($p > .27$ for all pairwise comparisons). During
6219 the dry condition rainfall simulations, the general patterns of sediment concentrations and
6220 sediment yields were opposite of the runoff rates, with the general harvest area > harvest-riparian

6221 edge > riparian buffer. The sediment concentration was (a) 424.8 mg l⁻¹ (151.0–1195.3 mg l⁻¹)
6222 in the general harvest area, (b) 100.9 mg l⁻¹ (45.8–222.1 mg l⁻¹) along the harvest riparian
6223 edge, and (c) 26.9 mg l⁻¹ (12.2–59.1 mg l⁻¹) in the riparian buffer. Statistically, there was
6224 strong evidence for differences in sediment concentrations between the general harvest area and
6225 along the harvest-riparian edge ($t = 3.21$, $p = .01$) and between the harvest area and the riparian
6226 buffer ($t = 6.17$, $p < .001$). Statistically, there was no evidence for differences in sediment yields
6227 between any of the plot positions. Sediment concentration among plot positions remained the
6228 same during the wet rainfall simulations as the dry rainfall simulations—general harvest area >
6229 harvest-riparian edge > riparian buffer. The geometric mean and 95% confidence intervals (back-
6230 transformed) for the sediment concentration was (a) 285.7 mg l⁻¹ (67.9–1201.5 mg l⁻¹) in the
6231 general harvest area, (b) 79.6 mg l⁻¹ (36.5–173.5 mg l⁻¹) along the harvest-riparian edge, and
6232 (c) 22.3 mg l⁻¹ (3.5–141.7 mg l⁻¹) in the riparian buffer. However, while sediment
6233 concentrations differed most strongly between the general harvest area and the riparian buffer (t
6234 $= 3.51$, $p = .01$), other pairwise comparisons were not significant ($p > .20$). Statistically, there
6235 was no evidence for differences in sediment yields between any of the plot positions for rainfall
6236 simulations during wet conditions. The authors speculate this was likely due to the greater soil
6237 porosity in the disturbed, harvested areas. Sediment concentration in the runoff, however, was
6238 approximately 15.8 times higher for the harvested area than in the riparian buffer, and 4.2 times
6239 greater than in the harvest-buffer interface. Total sediment yields from the harvested area (runoff
6240 + sediment concentration) were approximately 2 times greater than in the buffer areas, and 1.2
6241 times greater in the harvest-buffer interface (however, these proportions were not statistically
6242 different). Replication of the model showed high levels of variability in total run off rate,
6243 sediment concentrations, and sediment yields but the relationships between timing and relative
6244 magnitudes between the three experimental areas were consistent. The authors speculate that
6245 these results will become more relevant as climate change is expected to increase the frequency
6246 of high-intensity rainfall events following dry periods in this area. They suggest expanding
6247 similar methods to understand these effects in areas of different hydro-climatic settings.

6248

6249 **Stream Temperature**

6250

6251 Pollock et al., 2009

6252

6253 Pollock, M. M., Beechie, T. J., Liermann, M., & Bigley, R. E. (2009). Stream temperature
6254 relationships to forest harvest in western Washington 1. JAWRA Journal of the American Water
6255 Resources Association, 45(1), 141-156. <https://doi.org/10.1111/j.1752-1688.2008.00266.x>

6256

6257 This study investigates the effect of watershed harvest percentage, and time since harvest on
6258 summer stream temperatures at different scales in the Olympic Peninsula, Washington. The
6259 researchers examined recorded stream temperature data in 40 small watersheds that experienced

6260 a range of harvest from 0 – 100% (7 unharvested, 33 harvested between 25-100%), with
6261 regrowth age groups binned for analysis as recently clear cut (< 20 years old) and less recently
6262 clearcut (mostly < 40 years old). Unharvested sites were estimated as being >150-years old.
6263 Clearcut is defined in this paper as removing any protective canopy cover for streams. This study
6264 tested 3 hypotheses: (1) the condition of the riparian forest immediately upstream of a site
6265 primarily controls stream temperature, (2) the condition of the entire riparian forest network
6266 affects stream temperature, and (3) the forest condition of the entire basin affects stream
6267 temperature. These hypotheses were test by examining correlations of stream temperature with
6268 the condition of the immediate upstream riparian forest, or more correlated with forest conditions
6269 more spatially distant and on a coarser scale, such as the entire upstream riparian forest network
6270 or the forest condition of the entire basin. To avoid site effects in their analysis sites were chosen
6271 from a narrow range of subbasin sizes (approximately 1-10 km²) and elevation (75-400 m).
6272 Further, all sites were underlain by sedimentary rock and had perennial flow. Each hypothesis
6273 was tested with linear regression to evaluate the correlations of each age group at each scale with
6274 stream temperature data. The researchers also used AIC value comparisons for model selection to
6275 assess the correlation of other physiographic features (elevation, basin area, aspect, slope, or
6276 geologic composition) with stream temperatures. Results of general temperature patterns showed
6277 that average daily maximum (ADM) were strongly correlated with average diurnal fluctuations
6278 ($r^2 = 0.87$, $p < 0.001$, $n = 40$), indicating that cool streams also had more stable temperatures. For
6279 basin-level harvest effects on stream temperatures. The percentage of the basin harvested
6280 explained 39% of the variation in the ADM among subbasins ($r^2 = 0.39$, $p < 0.001$, $n = 40$) and
6281 32% of variation in the average daily range (ADR) ($r^2 = 0.32$, $p < 0.001$, $n = 40$). The median
6282 ADM for the unharvested subbasins was 12.8 °C (mean = 12.1 °C), which was significantly
6283 lower than 14.5 °C, the median (and average) ADM for the harvested subbasins ($p < 0.001$).
6284 Likewise, the median (and average) ADR for the unharvested subbasins was 0.9 °C, which was
6285 significantly lower than 1.6 °C, the median ADR (average = 1.7 °C) for the harvested subbasins
6286 ($p < 0.001$). Results for the correlations between the riparian network scale forest harvest and
6287 stream temperature showed that the total percentage of the riparian forest network upstream of
6288 temperature loggers harvested explained 33% of the variation in the ADM among subbasins ($r^2 =$
6289 0.33 , $p < 0.001$, $n = 40$) and 20% of variation in the ADR ($r^2 = 0.20$, $p = 0.003$, $n = 40$).
6290 However, the total percentage of upstream riparian forest harvested within the last 20 years was
6291 not significantly correlated to ADM or ADR. Results for near upstream riparian harvest and
6292 stream temperature showed either non-significant, or very weakly significant correlations. For
6293 example, there were no significant correlations between the percentage of near upstream riparian
6294 forest recently clear-cut and ADM temperature ($r^2 = 0.03$, $p = 0.79$, $n = 40$), the ADR of stream
6295 temperatures ($r^2 = 0.02$, $p = 0.61$, $n = 40$) or any other stream temperature parameters. The
6296 proportion of total harvested near upstream riparian forest (avg = 0.66, SD \pm 0.34, range = 0.0-
6297 1.0) was weakly correlated with ADM ($r^2 = 0.12$, $p = 0.02$, $n = 40$) and not significantly
6298 correlated with ADR ($r^2 = 0.07$, $p = 0.06$, $n = 40$). Even when the upstream riparian corridor
6299 length was shortened to 400 m and then to 200 m, and the definition of recently harvested was
6300 narrowed to <10 year, no significant relationships between temperature and the condition of the
6301 near upstream riparian forest was found. Results for the effect of physical landscape variables on
6302 stream temperature found that the variables of elevation, slope, aspect, percent of the basin with

6303 a glacial surficial geology, upstream distance of the site to sedimentary (bedrock) geology, and
6304 the percent of sedimentary surficial geology in the basin individually explain between 5% and
6305 14% more of the variability relative to basin harvest. Adding any one of these variables to the
6306 model increases the r^2 from 0.40 up to between 0.48 and 0.51. However, the coefficient for
6307 percent of basin harvested and its standard error stay essentially the same, thus the authors
6308 concluded that adding additional variables to the model did not change the basic finding that
6309 there is a strong relationship between ADM and total amount of harvest in a basin. Thus, for
6310 these models, the percentage of basin area harvested was the best predictor of variation in mean
6311 maximum stream temperatures. The probability of stream temperatures increasing beyond DOE
6312 standards (16 °C for seven-day average of maximum temperatures) increased with percent
6313 harvest. Nine of the 18 sites with 50-75% harvest and seven of the nine sites with >75% harvest
6314 failed to meet these standards. The authors interpret these results as evidence that the total
6315 amount of forest harvested within a basin, and within a riparian stream network are the most
6316 important predictors of changes in summer stream temperatures. They conclude that watersheds
6317 with 25-100% of their total area harvested had higher stream temperatures than watersheds with
6318 little or no harvest. Furthermore, they speculate that past basin-wide timber management can
6319 impact stream temperatures over long periods of time in a way that riparian buffer treatments
6320 cannot entirely ameliorate.

6321

6322 **Stream Temperature**

6323

6324 Groom et al., 2011a

6325

6326 Groom, J.D., Dent, L., Madsen, L.J., 2011. Stream temperature change detection for state and
6327 private forests in the Oregon Coast Range. *Water Resources Research* 47.
6328 <https://doi.org/10.1029/2009WR009061>

6329

6330 The purpose of this study was to evaluate the effectiveness of private and state forest buffer rules
6331 on state water quality stream temperature antidegradation standards in the Oregon Coast Range.
6332 According to the Department of Environmental Quality (DEQ), under the Protecting Cold Water
6333 (PCW) criterion, anthropogenic activities are not permitted to increase stream temperature by
6334 more than 0.3 °C above its ambient temperature. In addition, the cumulative amount of
6335 anthropogenic temperature increase allowed in streams with temperature total maximum daily
6336 loads (TMDLs) is 0.3 °C for all sources combined. Stream temperature and riparian stand
6337 conditions were measured pre- and post-harvest between 2002 and 2008 at 33 sites (18 private-
6338 owned, 15 state-managed). Treatment stands included 26 clear-cuts and 7 partial cuts (leave tree
6339 requirements not specified), all of which were harvested in adherence to FPA (private) and FMP
6340 (state) standards. Private sites followed FPA rules whereby the riparian management area
6341 (RMA)s are 15 and 21 m wide on small and medium fish-bearing streams, respectively, with a 6

6342 m no-cut zone immediately adjacent to the stream. State sites followed the state management
6343 plan whereby a 52 m wide buffer is required for all fish-bearing streams, with an 8 m no cut
6344 buffer immediately adjacent to the stream. Stream temperature data was collected for at least 2
6345 years prior to harvest. Reference reaches were located immediately upstream from the harvested
6346 reaches. Generalized least square regression was used to model ambient conditions while
6347 accounting for temporal autocorrelation. The authors examined prediction intervals to assess the
6348 rule exceedance (>0.3 °C increase in temperature). Results indicate that sites harvested according
6349 to FPA standards exhibited a 40.1% probability that a pre harvest to post harvest comparison of
6350 2 years of data will detect a temperature change of > 0.3 °C. Conversely, harvest to state FMP
6351 standards resulted in an 8.6% probability of exceedance that did not significantly differ from all
6352 other comparisons. The a priori and secondary post hoc multimodel comparisons did not indicate
6353 that timber harvest increased the probability of PCW exceedance at state sites. The authors point
6354 out that the 0.3 °C change threshold still lies 1 or 2 orders of magnitude lower than previous
6355 findings from studies which took place prior to the enactment of the riparian protection
6356 standards. The authors recommend further research looking into the potential persistence of
6357 stream temperature change downstream after harvest. In addition, they recommend looking into
6358 the biological significance of increases in stream temperature change particularly to aquatic life.

6359

6360 **Stream and subsurface water temperature**

6361

6362 Guenther et al., 2014

6363

6364 Guenther, S.M., Gomi, T., Moore, R.D., 2014. Stream and bed temperature variability in a
6365 coastal headwater catchment: influences of surface-subsurface interactions and partial-retention
6366 forest harvesting. *Hydrological Processes* 28, 1238–1249. <https://doi.org/10.1002/hyp.9673>

6367

6368 This study documented changes in stream and subsurface water temperature in response to forest
6369 harvesting in two paired headwater catchments. Specifically, the researchers hypothesized that
6370 post-logging changes in bed temperatures should be greatest in locations experiencing hyporheic
6371 downwelling (DW) and least in areas with lateral inflow/groundwater discharge. This study took
6372 place in the University of British Columbia Malcolm Knapp Research Forest near Vancouver,
6373 Canada. As a part of an ongoing study into the effects of riparian buffers on stream ecology,
6374 the catchments of 3 southerly-aspect first order streams were harvested using partial retention
6375 (50% removal of basal area including riparian zone) methods resulting in approximately 14%
6376 reduction in canopy cover on average; 3 other southerly-aspect streams served as unharvested
6377 controls. Before thinning treatments, the harvested riparian forests were dominated by western
6378 hemlock, (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), and Douglas-fir (*Pseudotsuga*
6379 *menziesii*). The forests were mature second growth forests with trees approximately 30–40 m tall,
6380 and canopy closure than 90%. Harvest operations began in September 2004 and completed in

6381 November of 2004. Temperature data was summarized from 10-minute intervals to daily
6382 minimum, maximum, and mean temperatures for stream and bed temperatures for one-year prior
6383 to, and one year following harvest. An analysis of the post-harvesting effects was conducted
6384 using a paired-catchment analysis. Results from this study show treatment sites resulted in higher
6385 daily maximum stream and bed temperatures after harvest but smaller changes in daily minima.
6386 Daily maximum post-harvest stream temperatures averaged over July and August ranged from
6387 1.6°C to 3°C at different locations. Post harvest changes in bed temperature at the lower reaches
6388 were smaller than changes in stream temperature, but was greater at sites with downwelling (DF)
6389 flow, and decreased with depth at upwelling (UW) and DF sites dropping to approximately 1°C
6390 at a depth of 30 cm. Changes did not vary significantly with depth at the middle reach, and
6391 averaged approximately 1°C change in daily maximum bed temperature over July and August. In
6392 summary, stream temperature responses differed at different locations within the cutblock. Bed
6393 temperatures also differed between UW and DW zones as well as between reaches with different
6394 contributions of lateral inflow. Given evidence that stream/bed temperature is shown to change
6395 spatially and with differences in hyporheic exchange and lateral inflow, the authors conclude by
6396 suggesting further research into the how these results might impact biological and ecological
6397 processes.

6398

6399 **Stream Temperature and evaporation/wind speed**

6400

6401 Guenther et al., 2012 (not in focal, does not separate the effects of shade reduction from wind
6402 speed/)

6403

6404 Guenther, S. M., Moore, R. D., & Gomi, T. (2012). Riparian microclimate and evaporation from
6405 a coastal headwater stream, and their response to partial-retention forest harvesting. *Agricultural
6406 and Forest Meteorology*, 164, 1-9.

6407

6408 The purpose of this study was to (1) develop and test an evaporimeter designed specifically to
6409 measure stream surface evaporation from headwater streams; (2) fit a wind function for
6410 computing evaporation from meteorological observations, and to compare it to previously
6411 published wind functions for evaporation from streams; and (3) quantify the influence of partial-
6412 retention forest harvesting on riparian microclimate and evaporation. This study was conducted
6413 in the University of British Columbia Malcom Knapp Research Forest (MKRF), approximately
6414 60 miles east of Vancouver, Canada and focused on the headwater stream of Griffith Creek. The
6415 harvesting treatment involved removal of 50% of the basal area from within the cut block,
6416 including the riparian zone. Smaller stems were removed, leaving the larger stems for harvest at
6417 a later date. creek. Analysis of paired pre- and post-logging hemispherical photographs indicated
6418 that canopy closure decreased by about 14% due to the logging treatment. Air temperature and
6419 relative humidity were measured by a Campbell Scientific CS500 sensor with stated accuracies

6420 of ± 0.5 °C for temperature and ± 3 – 6% for relative humidity. Wind speed was measured with a
6421 Met One anemometer with a stall speed of 0.447 m s⁻¹. Instruments were scanned every 10 s by
6422 a Campbell Scientific CR10x data logger; observations were averaged and stored every 10
6423 minutes. Evaporation was measured using four specially designed evaporimeters comprising an
6424 evaporation pan connected to a Mariotte cylinder. Results showed that Daily mean wind speeds
6425 increased following harvest, but were still consistently lower than wind speeds at the control site,
6426 with a maximum of 1.09 m s⁻¹. Vapor pressure was generally lower after harvesting. Vapor
6427 pressure deficit (vpd) increased following harvesting, but tended to remain lower than vpd
6428 measured at the control site. After harvesting, the relatively high wind speeds in the afternoon
6429 generally coincided with higher water temperatures, which in turn are associated with higher vpd
6430 at the water surface and a stronger vapor pressure gradient to drive evaporation. After harvest,
6431 wind speeds and vapor pressure gradients were higher and stability was weaker, consistent with
6432 the observed increase in evaporation. The authors conclude that the generally stronger relations
6433 between riparian and open microclimate variables after harvesting suggest that the riparian zone
6434 became more strongly coupled to ambient climatic conditions after harvesting as a result of
6435 increased ventilation. Further, that stream evaporation increased markedly as a result of partial
6436 retention harvest, consistent with the decrease in atmospheric vapor pressure, the increase in
6437 stream vapor pressure, the increase in wind speed and the decreased stability. In fact, prior to
6438 harvest, vapor pressure gradients often favored condensation rather than evaporation.

6439

6440 **LW**

6441

6442 Opperman, 2005 (Not in focal)

6443

6444 Opperman, J. J. (2005). Large woody debris and land management in California's hardwood-
6445 dominated watersheds. *Environmental Management*, 35(3), 266-277. DOI:10.1007/s00267-004-
6446 0068-z

6447

6448 The purpose of this paper was to evaluate the effects of stream and riparian area characteristics
6449 (bankfull width, gradient, basal area), and land ownership (public vs. private) on LW loading,
6450 and frequency, and debris jam frequency (response variables) in 21 hardwood-dominated forests
6451 of a Mediterranean climate region of northern California. The relationship between the stream
6452 and riparian area characteristics (explanatory variables: basal area of riparian trees, bankfull
6453 width, and gradient), and the response variables (woody debris loading and frequency, and
6454 debris-jam frequency) were evaluated with linear regression. The characteristics were then
6455 combined with ownership categories and their relative weight in explaining LW loading,
6456 frequency and pool frequency were assessed with a multi-variate analysis. Debris jam frequency
6457 was also analyzed by channel position with a chi-square. Results showed that debris jam
6458 frequency in the 21 reaches analyzed were strongly influenced by living standing trees rooted at

6459 the margins of the bank, especially in channel positions near the stream bank, but also spanning
6460 the channel partially, or completely. In general, LW loading was significantly higher in reaches
6461 adjacent to public lands (104 ± 13 m³/ha) than in those adjacent to private lands (46 ± 8 m³/ha;
6462 $P = 0.0015$). The strongest relationship for LW loading was with bankfull width ($r^2 = 0.32$; $p =$
6463 0.0006), and riparian basal area ($r^2 = 0.22$; $p = 0.006$) riparian basal area. This is likely the cause
6464 of the difference in public vs. private, as the public lands had significantly higher basal area in
6465 the riparian areas at distances >5 m from the stream, than the private lands. Debris jam frequency
6466 was also significantly influenced by riparian area gradient ($r^2 = 0.14$; $p = 0.03$) and basal area (r^2
6467 $= 0.11$; $p = 0.05$). The author concludes that landownership, and thus, land-management
6468 practices are driving factors in LW dynamics in this region.

6469

6470 **LW**

6471

6472 Nowakowski & Wohl, 2008

6473

6474 Nowakowski, A. L., & Wohl, E. (2008). Influences on wood load in mountain streams of the
6475 Bighorn National Forest, Wyoming, USA. *Environmental Management*, 42(4), 557-571.
6476 DOI:10.1007/s00267-008-9140-4

6477

6478 The purpose of this paper is to evaluate the relationship between riparian area characteristics, and
6479 land management practices with in-stream wood-loads in the Bighorn National Forest of
6480 northern Wyoming. The authors hypothesized that 1) valley geometry correlates with wood load,
6481 2) stream gradient correlates with wood load, 3) wood loads are significantly lower in managed
6482 watersheds than in similar unmanaged watersheds. The study analyzed data from 19 conifer
6483 dominated, forested headwater reaches in the bighorn mountains. Study reaches were separated
6484 by two watersheds, managed and unmanaged, with similar drainages, elevation, and lithology.
6485 Unmanaged watersheds were defined as having a history of minimal anthropogenic influences.
6486 The managed watershed had a history of different harvest prescriptions from unregulated in the
6487 late 1800s, clearcutting in the mid-1900s with tie floating practices. The relationship between in-
6488 stream wood loads (m³/ha) was analyzed with 11 valley-scale (elevation, forest type, forest stand
6489 density, etc.) and 13 channel-scale (reach gradient, channel width, etc.) variables with linear
6490 regression. Results support the first and third hypotheses. Across all streams, the highest
6491 explanatory power of all models tested produced land use (managed vs unmanaged), and basal
6492 area as a significant predictor of wood loads ($r^2 = 0.8048$). For the unmanaged watershed the
6493 model produced stream valley sideslope gradient as the single best predictor of wood load ($r^2 =$
6494 0.5748) supporting the first hypothesis. Shear stress was the best predictor of wood load in the
6495 managed watersheds ($r^2 = 0.2403$), These results did not directly support the second hypothesis.
6496 The authors suggest that while shear stress is a function of stream gradient (shear stress and
6497 stream gradient were significantly correlated, $r^2 = 0.9392$), gradient itself did not have the

6498 highest explanatory power of wood load in any of the models tested. Valley characteristics
6499 consistently explained more of the variability in wood load (42-80%) than channel characteristics
6500 (21-33%). When land use (managed vs. Unmanaged) effect on wood loads was analyzed the
6501 number of wood pieces per 100 m of stream was marginally significant ($p = 0.0565$), and the
6502 difference in wood volume per channel was significant ($p = 0.0200$) supporting the third
6503 hypothesis. When the significant valley and channel characteristics of the managed and
6504 unmanaged watersheds were controlled for, the significant difference in wood loads between
6505 managed and unmanaged watersheds were enhanced ($p = 0.0006$). Managed watersheds (1.1
6506 $m^3/100\ m$) had, on average, 2-3 times lower in-stream wood loads than unmanaged (3.3 $m^3/100$
6507 m) watersheds. These results suggest watersheds with a history of timber harvest have a decrease
6508 in stream wood loads than unmanaged watersheds, and that wood load dynamics can be driven
6509 by valley morphology, specifically, slope.

6510

6511 **Harvesting Practices on Suspended Sediment Yields**

6512

6513 Hatten et al., 2018

6514

6515 Hatten, J.A., Segura, C., Bladon, K.D., Hale, V.C., Ice, G.G., Stednick, J.D., 2018. Effects of
6516 contemporary forest harvesting on suspended sediment in the Oregon Coast Range: Alsea
6517 Watershed Study Revisited. *Forest Ecology and Management* 408, 238–248.
6518 <https://doi.org/10.1016/j.foreco.2017.10.049>

6519

6520 The objectives of this study were to (1) determine the effects of contemporary harvesting
6521 practices on suspended sediment yields and concentration, and (2) determine if contemporary
6522 harvesting practices produce lower sediment yields than historic practices. This study took place
6523 in the central Oregon Coast Range and consisted of a paired watershed study whereby Flynn
6524 Creek (FC) served as a reference watershed and Needle Branch (NB) served as a treatment
6525 watershed. A third watershed, Deer Creek (DC) served as a secondary control to compare
6526 historical vs contemporary harvest practices. The upper section of the treatment watershed was
6527 clearcut harvested using contemporary harvest practices (no buffer in non-fish-bearing streams
6528 with equipment exclusion zones, and a 15 m no-cut-buffer in fish-bearing streams) adhering to
6529 BMP's. Daily precipitation, discharge, and suspended sediment were collected at all three
6530 watersheds from October 2005 to June 2016. The upper half of the treatment watershed, (35 ha;
6531 measured at the Needle Branch Upper Gage or NBUG) was harvested in 2009 (Phase I) and the
6532 lower half (NBLG) was harvested in the fall of 2014 and mid-summer 2015 (Phase II). A model
6533 was developed using step wise linear regression to compare suspended sediment concentration
6534 (SSC). Differences in SSC among downstream sites and across harvest entries were compared
6535 utilizing an analysis of covariance. Results of the stepwise multiple linear regression showed
6536 strong evidence ($p < .001$) that all covariates (hydrograph limb, cumulative area discharge within

6537 water year, day of water year, daily precipitation, previous day's precipitation) were related to
6538 SSC across all watersheds. Both the mean and maximum SSC were greater in the reference
6539 catchments (FCG and DCG) compared to the harvested catchment (NBLG) across all water
6540 years. In NBLG the mean SSC was 32 mg L⁻¹ (~63%) lower after the Phase I harvest and 28.3
6541 mg L⁻¹ (~55%) lower after the Phase II harvest when compared to the pre-harvest
6542 concentrations. Compared to the reference watersheds, the mean SSC was 1.5-times greater in
6543 FCG (reference) compared to NBLG during the pre-harvest period. After the Phase I harvest the
6544 mean SSC in FCG (reference) was 3.1-times greater and after the Phase II harvest was 2.9-times
6545 greater when compared to the SSC in NBLG, the harvested watershed. Data from historical and
6546 contemporary harvests indicate contemporary practices are more effective at mitigating
6547 sedimentation. Historical data from the original study show harvesting without buffers, road
6548 building, and slash burning resulted in ~2.8 times increase in annual sediment yields and aquatic
6549 ecosystem degradation. The authors conclude that contemporary harvesting practices (i.e., stream
6550 buffers, smaller harvest units, no broadcast burning, leaving material in channels) using buffers
6551 were shown to sufficiently mitigate sediment delivery to streams, especially when compared to
6552 historic practices.

6553

6554 **Riparian Vegetation Removal Effects on Inputs and Production.**

6555

6556 Hetrick et al., 1998 (Removed, outside of timeline)

6557

6558 Hetrick, N.J., Brusven, M.A., Meehan, W.R., Bjornn, T.C., 1998. Changes in Solar Input, Water
6559 Temperature, Periphyton Accumulation, and Allochthonous Input and Storage after Canopy
6560 Removal along Two Small Salmon Streams in Southeast Alaska. *Transactions of the American*
6561 *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)
6562 [8659\(1998\)127<0859:CISIWT>2.0.CO;2](https://doi.org/10.1577/1548-8659(1998)127<0859:CISIWT>2.0.CO;2)

6563

6564 The purpose of this study was to assess whether or not the removal of second growth riparian
6565 vegetation would affect the production of juvenile coho salmon. In addition, this study aims to
6566 understand whether perceived effects are due to changes in habitat or food availability. This
6567 study took place in the Tongas National Forest on Prince of Wales Island, Alaska. Experimental
6568 reaches were divided into untreated and treated sections whereby treated sections had all
6569 vegetation on both sides of the streambank 6-15 m back removed. Stream discharge, water
6570 temperature, periphyton accumulation, allochthonous inputs, and storage of benthic organic
6571 matter were assessed during the summer and fall of 1988-1989. Differences in measured
6572 variables were assessed with a split-block analysis of variance. Results from this study show
6573 average light intensities reaching the water surface was significantly greater ($P < 0.01$) in the
6574 open canopy block than in the closed canopy block and was influenced significantly by weather
6575 conditions. Removal of riparian vegetation in both sections of the study significantly increased

6576 the accumulation of periphyton biomass and chlorophyll a ($P < 0.01$), and significantly decreased
6577 the amount of allochthonous organic inputs to streams ($P < 0.01$). Average daily allochthonous
6578 input rates for closed and open canopy conditions at Eleven creek were 789 and 6 mg AFDM/m²
6579 respectively, while input rates for closed and open canopy conditions at Woodsy creek were 805
6580 and 6 mg AFDM/m². Average daily water temperatures in open and closed canopy blocks at
6581 Eleven Creek were similar in 1988 but were significantly higher in the open blocks than in the
6582 closed blocks in 1989 ($P < 0.01$). The authors conclude by suggesting a thorough investigation
6583 into the interactions and responses of higher trophic levels to increases in periphyton biomass
6584 production and decreases in allochthonous inputs resulting from removal of riparian vegetation.
6585 Furthermore, the authors point out that the ability of stream segments to retain organic inputs
6586 through in-stream large woody debris may be a more important factor for allochthonous input
6587 processing by stream biota than the amount of allochthonous inputs entering a stream.

6588

6589 **Wood Recruitment and Retention**

6590

6591 Hough-Snee et al., 2016

6592

6593 Hough-Snee, N., Kasprak, A., Rossi, R.K., Bouwes, N., Roper, B.B., Wheaton, J.M., 2016.
6594 Hydrogeomorphic and Biotic Drivers of Instream Wood Differ Across Sub-basins of the
6595 Columbia River Basin, USA. *River Research and Applications* 32, 1302–1315.
6596 <https://doi.org/10.1002/rra.2968>

6597

6598 The purpose of this study was to understand the hydrogeomorphic and ecological processes
6599 which lead to wood recruitment and retention in seven sub-basins of the interior Columbia River
6600 Basin (CRB), USA. To achieve this, in-stream wood volume and frequency are quantified across
6601 sub basins. Following this, the riparian, geomorphic, and hydrologic attributes which are most
6602 strongly correlated to in-stream wood loads were determined. Random forest models were used
6603 to identify relationships between ecological and hydrogeomorphic attributes that influence in-
6604 stream wood within each sub-basin. Non-metric multidimensional scaling was performed on a
6605 matrix of hydrogeomorphic and forest cover variables, excluding instream wood frequency and
6606 volume to visualize reaches and sub-basins' relative similarity. To determine how wood
6607 predictors differed between sub-basins, ordinary least squares regression models of wood volume
6608 and frequency were built within each sub-basin. Results from this study show that in stream
6609 wood volume and frequency were distinctly different across all seven sub-basins. Across the
6610 CRB, wood frequency ranged from 0 to 2117.0 pieces km⁻¹, while volume ranged from 0 to 539
6611 m³ km⁻¹. Large wood volume (PERMANOVA $F = 5.1$; $p = 0.001$) and frequency
6612 (PERMANOVA $F = 5.4$; $p = 0.001$) differed significantly between sub-basins. According to
6613 random forest (RF) models, mean annual precipitation, riparian large tree cover, and individual
6614 watershed were the three most important predictors of wood volume and frequency. Watershed

6615 area was the fourth strongest predictor of wood frequency, while catchment-scale and reach-scale
6616 forest cover were the fourth and fifth strongest predictor of wood volume. In contrast, sinuosity
6617 and measures of streamflow and stream power were relatively weak predictors of wood volume
6618 and frequency. Taken together, wood volume and frequency increased with precipitation and
6619 large riparian tree cover and decreased with watershed area. Final RF models explained 43.5% of
6620 the variance in volume and 42.0% of the variance in frequency of in stream wood loads. Results
6621 for drivers of wood frequency and volume between sub-basins were highly variable either
6622 showing no relationship between candidate models and predictive power (e.g., $r^2 \leq 0.12$; Entiat
6623 sub-basin). The highest predictive models for wood volume ($r^2 > 0.55$) and wood frequency (r^2
6624 ≤ 0.45) were for the John Day sub basin. Depending on the sub basin wood volume and
6625 frequency was positively correlated with forest cover, watershed area, large tree cover, 25-year
6626 flood event stream power, riparian conifer cover, and precipitation. Negative correlations,
6627 depending on sub basin, of wood volume and frequency with baseflow discharge, riparian woody
6628 cover, watershed area, and large tree cover. Given the heterogeneous results across all sub-basins
6629 studied, the authors conclude by emphasizing the importance of incorporating local data and
6630 context when building wood models to inform future management decisions.

6631

6632 **Stream Temperature**

6633

6634 Hunter, 2010 (not in focal, treatments and results not relevant to questions)

6635

6636 Hunter, M.A., 2010. Water Temperature Evaluation of Hardwood Conversion Treatment Sites
6637 Data Collection Report (Data Collection Report). Cooperative Monitoring, Evaluation, and
6638 Research (CMER). Fp_cmer_05_513

6639

6640 The purpose of this study is to evaluate the response of stream temperature to changes in canopy
6641 cover using a before-after-control-impact design. This study took place along nine hardwood-
6642 dominated riparian stands in Western Washington. Variables measured among locations and years
6643 include riparian conditions, canopy cover, channel dimensions, substrate, flow and stream
6644 temperature. Results from this study show that hardwood conversion buffers (HCB -
6645 approximately 15 m width) intended to convert hardwood-dominated riparian areas to conifer-
6646 dominated riparian areas usually resulted in decreased canopy cover of streams. Mean Global
6647 Site Factor (GSF - the proportion of global radiation under a plant canopy relative to the amount
6648 in an open area) increased in most study sites with HCB's. However, mean GSF did not change
6649 substantially at sites with buffers closer to standard (~ 18 – 45 m) non-hardwood conversion
6650 buffers. Temperature was highly variable over time and among locations suggesting stream
6651 temperature is affected by many factors that might differ among locations and throughout time.
6652 Longitudinal patterns of warming and cooling were consistent at all sites indicating the potential

6653 importance of careful site selection to account for changes in the longitudinal distribution of
6654 temperatures.

6655

6656 **Influence of Stream Geomorphology on Water Temperature**

6657

6658 Hunter & Quinn, 2009

6659

6660 Hunter, M.A., Quinn, T., 2009. Summer Water Temperatures in Alluvial and Bedrock Channels
6661 of the Olympic Peninsula. *Western Journal of Applied Forestry* 24, 103–108.
6662 <https://doi.org/10.1093/wjaf/24.2.103>

6663

6664 The purpose of this study was to understand how stream geomorphology influences water
6665 temperature in managed stands on the Olympic Peninsula, Washington. Sites chosen for this
6666 included an alluvial study site and a bedrock study site whose overall characteristics were
6667 otherwise comparable apart from geomorphology. The alluvial study site was a 1.6-km reach of
6668 Thorndyke Creek. The bedrock study site was a 1.4-km reach of the South Fork Pysht River.
6669 Both channels were located in 35–50-year-old managed forests dominated by Douglas-fir
6670 (*Pseudotsuga menziesii*) in the uplands and red alder (*Alnus rubra*) in the riparian zone. Surface
6671 substrate at the alluvial channel was composed mostly of gravel, whereas the bedrock channel
6672 was composed of mostly bedrock, boulder, and cobble. The mean solar input (GSF: global site
6673 factor) did not differ between streams. Water temperature was recorded at 75-m intervals along
6674 each channel during the summers of 2003 and 2004. Results from this study show consistent
6675 differences in stream temperature response in alluvial versus bedrock channels. Seasonal
6676 maximum and minimum average daily temperatures varied less at the alluvial site compared to
6677 the bedrock site. This, the authors suggest may be due to hyporheic exchange in alluvial channels
6678 helping to buffer surface water temperatures from gaining or losing heat. In addition,
6679 groundwater may also contribute to the increased stability at the alluvial site. Two same-day
6680 measurements at each site showed the alluvial site gaining 8% of its flow, as compared to the
6681 bedrock site whose flow decreased by approximately 15%. The bedrock site was also shown to
6682 have the highest variation in reach-scale water temperatures during low flow. The authors
6683 conclude that stream geomorphology may have profound impacts on spatial and temporal
6684 patterns of channel water temperature. The authors suggest temperature reading from a single
6685 location may not accurately represent the entire channel. Additional research involving collection
6686 of temporal and longitudinal data will be needed to tailor riparian buffers to channel type.

6687

6688 **Stream temperature, sediment, nutrient**

6689

6690 Murray et al., 2000

6691

6692 Murray, G. L. D., Edmonds, R. L., & Marra, J. L. (2000). Influence of partial harvesting on
6693 stream temperatures, chemistry, and turbidity in forests on the western Olympic Peninsula,
6694 Washington. *Northwest science.*, 74(2), 151-164. Handle: <https://hdl.handle.net/2376/1065>

6695

6696 This study investigates the effects of partial watershed harvest (7-33%) on stream temperature,
6697 chemistry, and turbidity relative to an unharvested old-growth watershed in the western Olympic
6698 Peninsula, Washington. Both harvested watersheds (Rock and Tower creeks) originally contained
6699 old-growth forests. Rock Creek had 7% of its watershed harvested in 1981, and Tower Creek had
6700 33% of its watershed harvested between 1985 and 1987. Logging extended to the stream edge
6701 near the in-stream monitoring sites. Data for stream daily maximum, minimum, and mean
6702 temperatures, chemistry, and turbidity was recorded and monitored from June 1996 to June 1998
6703 (10-15 years post-harvest). Differences in variables between treatment and reference watersheds
6704 were compared with a one-way ANOVA with a posthoc Tukey HSD test. Results showed higher
6705 maximum summer stream temperatures (15.4 °C), and lower winter maximum stream
6706 temperatures (3.7 °C) in the two treatment watersheds compared to the unharvested reference
6707 watershed (12.1 °C and 6.0 °C for summer max, and winter max, respectively). Winter minimum
6708 temperatures for one of the harvested watersheds reached 1.2 °C (Rock Creek) compared to a
6709 winter minimum of 6 °C. Thus, seasonal variation of stream maximum temperatures and winter
6710 minimum temperatures were more extreme in the treatment watershed than in the control. There
6711 were no seasonal patterns or significant differences between watersheds in stream chemistry
6712 except for calcium and magnesium concentrations being consistently higher in the unharvested
6713 watersheds. Turbidity was low and not significantly different between watersheds. The authors
6714 interpret these results as evidence of partial harvest having minimal impact on stream
6715 temperatures, chemistry, and turbidity long-term (after 10-15 years). The stream temperature
6716 changes were significant but did not exceed the 16 °C threshold used as a standard for salmonid
6717 habitat. However, there was no data collection during the first decade following harvest.

6718

6719 **Channel Habitat, Particle Size, Stream Temperature, and Woody Debris Response to**
6720 **Harvest**

6721

6722 Jackson et al., 2001

6723

6724 Jackson, C.Rhett., Sturm, C.A., Ward, J.M., 2001. Timber Harvest Impacts on Small Headwater
6725 Stream Channels in the Coast Ranges of Washington1. *JAWRA Journal of the American Water*
6726 *Resources Association* 37, 1533–1549.

6727 <https://doi.org/10.1111/j.1752-1688.2001.tb03658.x>

6728

6729 The purpose of this study was to evaluate changes in stream temperature, particle size
6730 distributions of bed material, and channel habitat distributions in 15 first- or second order
6731 streams located on the Coast Range of Western Washington. Four of the fifteen stream basins
6732 were not harvested and served as references; three streams were cut with unthinned riparian
6733 buffers; one with a partial buffer; one with a buffer of non-merchantable trees; and six were
6734 clearcut to the stream edge. Buffer widths varied by operation; the average buffer width varied
6735 from 15 – 21 meters. The narrowest buffer measured on one side of the stream was 2.3 meters.
6736 Data for woody debris, sediment concentrations, turbidity, and stream temperatures were
6737 recorded for one-year prior to harvest (1998). Harvest was conducted in the spring and early
6738 summer of 1999, and post-harvest data was collected for about a month after operations were
6739 complete. Thus, the results presented in this study represent changes in stream attributes and
6740 characteristics immediately following harvest. Results from this study show that logging without
6741 buffers had immediate and dramatic effects on channel morphology. Without buffers, and the
6742 relatively steep topography of the study sites logging debris tended to accumulate at the bottom
6743 of slopes thereby burying or covering many headwater streams. Covered channels were defined
6744 in this study as having flow completely obscured by organic debris, but a recognizable channel
6745 still exists below the debris. Buried channel was defined as having so much organic detritus in
6746 the flow cross-section that the channel was no longer definable. Needles, twigs, whole branches,
6747 and logs buried headwater streams with a mean depth of 0.94 meters of organic debris (range:
6748 0.5 - 2.0 meters). Of the clearcut streams the percent of stream buried with organic matter ranged
6749 from 6 to 90%, and the percent covered by organic matter ranged from 8 to 85%. The sum of
6750 buried and covered for each stream ranged from 72 to 100%. On the other hand, most buffered
6751 streams had 0% covered or buried by organic matter post-harvest with the only exception being
6752 one stream that experienced blowdown post-harvest that covered 29% of the stream. While
6753 debris accumulation tended to protect streams from the effects of solar radiation, organic logging
6754 debris was also shown to trap fine sediment in the channels which, in the near term, greatly
6755 reduced downstream sediment movement. As a result of increased roughness and additional bank
6756 failures within the clearcut sites, sediment size shifted towards finer particles growing from 12 to
6757 44 percent. In contrast, particle size distributions continued nearly unchanged in buffered and
6758 reference sites. In the first summer after logging, significant increases were detected in overall
6759 macroinvertebrate densities, collector densities, shredder abundance and biomass, and organic
6760 and inorganic matter accretion. However, these responses were not detected one year following
6761 logging. For stream temperature changes, because the data collection was for such a short period
6762 of time (1-year pre- and 1-month post-harvest), and because the summer of 1999 was much
6763 cooler than 1998, the assessment of harvest effects on stream temperature changes was difficult.
6764 Thus, to interpret significant changes in stream temperatures from pre- to post- harvest, daily
6765 maximum temperatures were plotted against the appropriate reference stream, and a regression
6766 equation was calculated. The slopes of the regression lines were compared with a student's t-test
6767 to determine significant differences. Of the seven clearcut streams, three showed no significant
6768 changes in temperature, one became cooler (-1.1 °C), one became slightly warmer (+0.8 °C), and

6769 the other 2 became warmer or colder depending on location with decreases in temperature
6770 upstream (-2.2 and -1.7 °C) and increases in temperature downstream (+5.2 and +15.1 °C). The
6771 buffered streams had significant but less dramatic changes in temperature with one decreasing in
6772 temperature (-0.3 °C), and 2 increasing in temperature (+1.6 and +2.4 °C). The one site with the
6773 non-merchantable buffer had much higher temperature increases (+3.7 and +6.6 °C). The authors
6774 posit that sites which retained riparian buffers succeeded in keeping debris out of streams as well
6775 as served to protect streambanks from failure or erosion. Some mature trees left within buffers
6776 experienced blow down and spanned the channel. While the clearcut streams had nearly all
6777 canopy cover removed, the buildup of slash and LW in the stream also provided shade and
6778 insulation that caused reductions in stream temperatures, or slight increases with one exception
6779 (+15.1 °C) The authors point out that this study only served to point out immediate effects of
6780 logging on physical channel conditions. Although important, there are still many questions about
6781 how channel conditions will evolve over time.

6782

6783 **LW**

6784

6785 Meleason et al., 2003

6786

6787 Meleason, M. A., Gregory, S. V., & Bolte, J. P. (2003). Implications of riparian management
6788 strategies on wood in streams of the Pacific Northwest. *Ecological Applications*, 13(5), 1212-
6789 1221. <https://doi.org/10.1890/02-5004>

6790

6791 This study used simulation modeling to evaluate the potential effects of three different riparian
6792 and watershed harvest scenarios on the standing stock of large wood in a hypothetical stream in
6793 the Pacific Northwest. The three scenarios involved harvest 1) clearcut to the streambank, 2)
6794 riparian management buffer widths ranging from 6-75 m, and 3) riparian buffers of various
6795 widths with upland forest plantation. The effects of each scenario on wood load dynamics were
6796 simulated with OSU STREAMWOOD for four harvest rotation periods (no harvest, 60, 90, and
6797 120 years) over the course of 720 years. Results for scenario one (clear-cut to stream) showed
6798 minimal accumulation of wood into the stream with little change over time due to the lack of a
6799 forested riparian management zone. Results for scenario two showed the maximum standing
6800 stock of in-stream wood loads required ≥ 30 m no-cut buffer zones for 500-year-old forests.
6801 Wood loads in streams with 6 m wide buffers showed 32% of standing wood load stocks after
6802 240 years. Results from scenario three showed minimal amounts of wood contributed into
6803 streams from forest plantations when > 10 m wide buffers were used. The authors interpret these
6804 results as evidence that riparian buffer widths and forest age are more important for estimating
6805 changes in wood loads over time than the harvest rotation age of plantation forests.

6806

6807 **LW**

6808

6809 Martin & Grotefendt, 2007

6810

6811 Martin, D. J., & Grotefendt, R. A. (2007). Stand mortality in buffer strips and the supply of
6812 woody debris to streams in Southeast Alaska. *Canadian Journal of Forest Research*, 37(1), 36-49.
6813 <https://doi.org/10.1139/x06-209>

6814

6815 This study compared riparian stand mortality and in-stream LW recruitment characteristics
6816 between riparian buffer strips with upland timber harvest and riparian stands of unharvested
6817 watersheds using aerial photography. This study was conducted in the northern and southern
6818 portions of Southeast Alaska at multiple sites in nine timber harvest areas. All study sites were
6819 along moderate- and low-gradient streams with channel widths ranging from 5 m to 30 m wide.
6820 All buffer strips were conifer dominated and a minimum of 20 m wide that included selective
6821 harvest within the 20 m zone. Reference sites were along unharvested reaches in the same area.
6822 Stand mortality was estimated by the proportion of downed trees within a buffer strip.
6823 Differences in downed tree proportions relative to reference streams were assumed to be caused
6824 by timber harvest, accounting for selective in-buffer harvests. A one-tailed paired t-test or a
6825 Wilcoxon signed rank test was used to check for statistical differences between treatment and
6826 reference sites. Results showed significantly higher mortality (based on cumulative stand
6827 mortality: downed tree counts divided by standing tree counts + downed tree counts),
6828 significantly lower stand density (269 trees/ha in buffer units and 328 trees/ha in reference units),
6829 and a significantly higher proportion of LW recruitment from the buffer zones of the treatment
6830 sites than in the reference sites. Densities within all units ranged from 0 – 1334 trees/ha
6831 depending on location. Overall, mean stand density in the buffer units was 18% lower than in the
6832 reference units. Results also showed that mortality varied with distance to the stream.
6833 Differences in mortality for the treatment sites were similar to the reference sites for the first 0-
6834 10 m from the stream (only a 22% increase in the treated sites). However, mortality in the outer
6835 half of the buffers (10-20 m) from the stream in the treatment sites was more than double (120%
6836 increase) what was observed in the reference sites. This caused a change in the LW recruitment
6837 source distance curves, with a larger proportion of LW recruitment coming from greater
6838 distances in logged watersheds. LW recruitment based on the proportion of stand recruited (PSR)
6839 was significantly higher in the buffered units compared to the reference units. However, PSR
6840 from the inner 0-20 m was only 17% greater in the buffer units than in the reference units, while
6841 PSR of the outer unit (10 – 20 m) was more than double in the buffered units than in the
6842 reference units. The researchers conclude that the increase in mortality was caused by an
6843 increased susceptibility to windthrow. They estimate that future recruitment potential from the
6844 logged sites diminished by 10% relative to the unlogged reference sites.

6845

6846 **Stream temperatures**

6847

6848 Macdonald et al., 2003b

6849

6850 Macdonald, J. S., MacIsaac, E. A., & Herunter, H. E. (2003). The effect of variable-retention
6851 riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest
6852 ecosystems of British Columbia. *Canadian journal of forest research*, 33(8), 1371-1382.
6853 <https://doi.org/10.1139/x03-015>

6854

6855 This study investigates the impacts of forest harvest on stream temperatures under three variable
6856 retention buffer treatments in headwater streams of the interior sub-boreal forests of British
6857 Columbia. Temperature data were recorded for two years pre- and five years post-harvest from
6858 five harvested streams and two unharvested reference streams. Differences between pre- and
6859 post-harvested stream temperatures were compared with the paired reference streams using
6860 repeated measures ANOVA. Treatment riparian areas were harvested with the following
6861 prescriptions: 1) low-retention – removal of all merchantable timber >15 or >20 cm DBH for
6862 pine or spruce respectively, within 20 m of the stream 2) high-retention – removal of
6863 merchantable timber >30 cm DBH within 20-30 m of the stream, and 3) Patch-cut – high
6864 retention for the lower 60% of watershed approaching streams and removal of all vegetation in
6865 the upper 60% of the watershed. Eight first-order streams were included in this study: two

6866 in the Gluskie Creek watershed (G5, G7) and six in the Baptiste Creek watershed (B1–B6). Five
6867 of these streams were within the harvested boundaries (2 high-retention, 2 low-retention, and 1
6868 patch cut), and 3 reaches outside of the harvest boundary served as controls. Results showed a
6869 significant increase in stream temperatures ranging from 4 – 6 °C at five years post-harvest, and
6870 increased ranges of diurnal temperature fluctuations for all treatment streams relative to the
6871 reference streams. Streams that had summer maximum mean weekly temperatures of 8°C before
6872 harvesting had maximum temperatures near 12°C or more following harvesting. Daily ranges of
6873 1.0–1.3°C before harvesting became 2.0–3.0°C following harvesting. Greater temperature ranges
6874 occurred in low-retention and patch treatments than the high-retention or control treatment
6875 locations. The high-retention buffer treatment mitigated temperature increases for the first three
6876 years. Still, increased mortality (windthrow) caused a reduction in the canopy that increased
6877 stream temperatures equivalent to other treatment streams by year five. The results of this study
6878 show evidence that high-retention buffers are no more effective in preserving stream temperature
6879 changes than small retention buffers when treatment areas have a high susceptibility to
6880 windthrow.

6881

6882 **Sediment delivery pathways**

6883

6884 Litschert & MacDonald, 2009

6885

6886 Litschert, S. E., & MacDonald, L. H. (2009). Frequency and characteristics of sediment delivery
6887 pathways from forest harvest units to streams. *Forest Ecology and Management*, 259(2), 143-
6888 150. <https://doi.org/10.1016/j.foreco.2009.09.038>

6889

6890 This study investigates the frequency of sediment delivery pathways (“features”) in riparian
6891 management areas and measures the physical characteristics and connectivity of these pathways
6892 following timber harvest. The results of this study were then used to develop models for
6893 predicting the length and connectivity of pathways formed from harvest units. Data was collected
6894 from over 200 harvest units with riparian management areas in the Eldorado, Lassen, Plumas,
6895 and Tahoe National Forests in the Sierra and Cascade mountains of northern California. Riparian
6896 buffer widths for this area are 90 m and 45 m for perennial and annual streams respectively. No
6897 machinery is allowed in the riparian management areas. Data collected and analyzed for the
6898 pathways included years since harvest, mean annual precipitation, soil depth, soil erodibility,
6899 hillslope gradient, aspect, and elevation. Characteristics of pathway length, gradient, and
6900 roughness were also collected. Relationships between site variables and pathway variables were
6901 assessed using linear regression. The site variables with the most significant relationships with
6902 the pathway variables were used in a multivariate regression model to predict pathway length.
6903 Only 19 of the 200 harvest units had sediment development pathways. Pathways ranged in age
6904 (time since harvest) from 2 to 18 years, and in length from 10 m to 220 m. Of the 19 pathways,
6905 only six were connected to streams, and five of those originated from skid trails. Pathway length
6906 was significantly related to mean annual precipitation, cosine of the aspect, elevation, and
6907 hillslope gradient. The authors conclude that timber prescription practices for these National
6908 Forests are effective in reducing sediment delivery pathways. The authors interpret these results
6909 as evidence that skid trails should be directed away from streams, maintaining surface roughness,
6910 and promptly decommissioning skid trails.

6911

6912 **LW**

6913

6914 Liquori, 2006

6915

6916 Liquori, M. K. (2006). POST-HARVEST RIPARIAN BUFFER RESPONSE: IMPLICATIONS
6917 FOR WOOD RECRUITMENT MODELING AND BUFFER DESIGN 1. *JAWRA Journal of the*
6918 *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)
6919 [1688.2006.tb03832.x](https://doi.org/10.1111/j.1752-1688.2006.tb03832.x)

6920

6921 This study investigates the differences in treefall characteristics in riparian management areas
6922 based on ecological and physiographic variables to give insight on the variables important for
6923 wood recruitment modeling. Data were collected from 20 riparian buffer sites that had all been
6924 clearcut within three years of sampling with standard no-cut buffers 25 ft. An additional 50-100
6925 ft buffer was applied to fish-bearing streams depending on stream type, in a managed tree farm in
6926 the Cascade Mountains of western Washington. These riparian buffers generally consisted of
6927 naturally regenerated, second-growth conifer stands about 45 to 70 years old. “Very modest”
6928 thinning was applied to some stands to meet wildlife objectives and any downed wood not
6929 affecting the channel was removed. Tree characteristic data collected included tree size (DBH
6930 and height), species, fall direction, tree fall angles, estimated cause of mortality, and distance to
6931 the stream. Site characteristics included stream gradient, valley morphology, and time since
6932 harvest. Tree recruitment probability curves were developed as a function of tree height using
6933 methods described by Beschta, (1990). Results showed that wind-caused mortality and tree fall
6934 rates were significantly higher, up to three times higher, than competition-induced mortality
6935 within buffers for three years following treatment. The median observed treefall per site was
6936 15% of all trees in each buffer, ranging from 1 to 57%. total treefall at each site for one, two, and
6937 three years since harvest was $16 \pm 10\%$, $28 \pm 21\%$, and $10 \pm 10\%$, respectively. Total treefall
6938 percentage for each site was not correlated to years since harvest (Spearman $R = 0.11$; $p = 0.34$).
6939 The mean and standard deviation of the total normalized treefall for one-year old sites was $405 \pm$
6940 394 trees/km ($n = 9$), for two-year old sites was 264 ± 280 trees/km ($n = 7$), and for three-year
6941 old sites was 556 ± 316 trees/km ($n = 4$). Treefall varied significantly by species. Downed red
6942 alder (*Alnus rubra*), western red cedar (*Thuja plicata*), and Douglas-fir (*Pseudotsuga menziesii*)
6943 comprised 3 percent to 8 percent of all downed trees; these species had treefall rates ranging
6944 from 5 percent to 9 percent of the total number of trees of the same species. By contrast, treefall
6945 rates for western hemlock (*Tsuga heterophylla*) and Pacific silver fir (*Abies amabilis*) ranged
6946 from 23 percent to 26 percent. Treefall rates also varied somewhat by size, with the 31 to 41 cm
6947 (12 to 16 in) diameter class having the greatest treefall rates (All trees were grouped into size
6948 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
6949 than 20 in). Treefall following harvest greatly exceeded the expected competition induced
6950 mortality rates (posited by Franklin, 1970) of 0.5%, and the model of average competition
6951 mortality used in Rainville et al. (1985), which ranged from 0.7 - 1.6%, and 2% per year for bank
6952 undercutting. Treefall direction was heavily biased towards the channel regardless of channel or
6953 buffer orientation and tree fall probability was highest in the outer areas of the buffers (adjacent
6954 to the harvest area). Fall direction bias increased significantly in the inner portions of the buffer.
6955 Within the 0 to 7 m zone and 7 to 15 m zone, 68% and 67% of the trees, respectively, fell toward
6956 the channel ($n = 125$ and 153 , respectively). Only 44% of the outer zone (> 15 m) downed trees
6957 fell toward the channel ($n = 403$). Generally, recruitment was negatively correlated to buffer
6958 width ($r^2 = 0.40$). Treefall was generally highest at the outside edges of buffers (50+ feet),
6959 representing about 60% of the total observed treefall, while the 0–25-foot zone represented
6960 ~18%, and the 25–50-foot zone represented ~22%. The authors interpret their results as evidence
6961 that tree fall models that use a random fall direction may underrepresent the probability of LW
6962 recruitment into streams. Further, they suggest that the increase in windthrow mortality and the
6963 probability of tree fall with increasing distance from the stream should be considered.

6964

6965 **LW**

6966

6967 Lininger et al., 2021 (removed from focal list, this is a case study)

6968

6969 Lininger, K. B., Scamardo, J. E., & Guiney, M. R. (2021). Floodplain large wood and organic
6970 matter jam formation after a large flood: Investigating the influence of floodplain forest stand
6971 characteristics and river corridor morphology. *Journal of Geophysical Research: Earth Surface*,
6972 126(6), e2020JF006011. <https://doi.org/10.1029/2020JF006011>

6973

6974 This study examines how river corridor morphology and forest stand density influence LW and
6975 coarse particulate matter (CPOM) deposition patterns in the flood plain resulting from a 400-year
6976 flood event in West Creek in the Colorado Front Range in 2013. The researchers tested the
6977 hypothesis that if river corridor geomorphology affects LW and CPOM deposition then there
6978 should be an inverse relationship between elevation above and distance from the stream's edge.
6979 Further, that deposition frequency would be higher in unconfined portions of the corridor.
6980 Considering forest stand structure, the researchers hypothesized that LW/CPOM jams would be
6981 pinned by trees, higher in intermediate forest densities, and decrease in size with increasing
6982 forest stand density. Field data of LW/CPOM jams were analyzed with non-parametric Spearman
6983 correlation tests to determine the strength of their relationship with channel and stand
6984 characteristics. Results showed support for most of the hypotheses. LW accumulations did
6985 decrease in size with distance from the stream, but CPOM did not. Confined channels (steeper
6986 reaches) contained fewer LW/CPOM loads per unit area. The authors speculate that these reaches
6987 had higher flow rates and thus lower deposition during the flood. CPOM jams increased in
6988 number per area with increasing stand density with most jams pinned against live trees. The
6989 authors conclude that the effect of riparian forest stand density is evidence that riparian forests in
6990 the floodplains should be preserved to increase LW and CPOM trapping probability.

6991

6992 **Stream Temperature**

6993

6994 Janisch et al., 2012

6995

6996 Janisch, J.E., Wondzell, S.M., Ehinger, W.J., 2012. Headwater stream temperature: Interpreting
6997 response after logging, with and without riparian buffers, Washington, USA. *Forest Ecology and*
6998 *Management* 270, 302–313. <https://doi.org/10.1016/j.foreco.2011.12.035>

6999

7000 The purpose of this study was to assess the stream temperature response to three different
7001 harvesting treatments in small, forested headwater catchments in western Washington. The pre-
7002 logging calibration period lasted 1–2 summers and stream temperatures were monitored for two
7003 or more summers after logging. Harvest treatments occurred between September 2003 and July
7004 2005; catchments were clustered by harvest year for analysis. A before-after-control-impact
7005 study design was used to contrast stream temperature responses for three forest harvest
7006 treatments: clearcut logging to the stream (n=5), a continuous buffer (n=6) with widths 10-15 m
7007 on each side of the channel, and a patched buffered (n=5) where portions of the riparian forests
7008 ~50-110 m long were retained in distinct patches along some portion of the channel with the
7009 remaining riparian area clearcut. For the patch buffers there was no standard width, the buffer
7010 spanned the full width of the floodplain area and extended well away from the stream. Upland
7011 areas adjacent to buffers were clearcut. Regression relationships were developed between
7012 temperatures measured in the treatments and corresponding reference catchments. A simple
7013 ANOVA model was used that only included fixed effects for treatment, years since treatment,
7014 and day of year. Because of the unbalanced experimental design and variation in time of harvest,
7015 clustering of treatments caused the sample sizes to become too small to apply a more complex
7016 nested, repeated measures ANOVA could not be used. Correlation analysis was conducted
7017 between post-harvest stream temperatures and descriptive variables on a subset of catchments to
7018 examine possible factors that might control post-harvest thermal responses. Results from this
7019 study show significant increases in stream temperature in all treatments. Although temperature
7020 responses were highly variable within treatments, July and August daily maximum temperatures
7021 increased in clearcut catchments during the first year after logging by an average of 1.5°C (range
7022 0.2 to 3.6°C), in patch-buffered catchments by 0.6°C (range – 0.1 to 1.2°C), and in continuously
7023 buffered catchments by 1.1°C (range 0.0 to 2.8°C). Canopy cover in all streams averaged 95%
7024 prior to harvest and did not differ between treatment and reference streams. Following treatment,
7025 canopy cover in the clearcut catchments averaged 53%, canopy cover in the patch buffer
7026 treatment averaged 76%, and canopy cover in the continuous buffer treatment averaged 86%.
7027 Following treatment, the canopy cover of the clearcut and patch buffer treatments were
7028 significantly lower than in the reference streams. The continuous buffer treatments did not differ
7029 significantly from the reference streams for canopy cover. Further analyses which attempted to
7030 identify variables responsible for controlling the extent of stream temperature responses showed
7031 the amount of cover retained in the riparian buffer was not a strong explanatory variable. Post-
7032 treatment temperature changes suggested that treatments ($p = 0.0019$), the number of years after
7033 treatment ($p = 0.0090$), and the day of the year ($p = 0.0007$) were all significant effects
7034 explaining observed changes in temperature. Wetland area ($r^2 = 0.96$, $p < 0.01$) and length of
7035 surface flow ($r^2 = 0.67$, $p = 0.05$) were strongly correlated with post-logging temperature
7036 changes. Regression analysis of these variables showed streams with fine-textured substrates
7037 responded differently than coarse textured substrates. The authors speculate this is possibly due
7038 to groundwater interactions which can buffer thermal responses of small streams. In summary,
7039 the authors conclude that their results suggest small headwater streams may be fundamentally
7040 different than larger streams partly because factors other than canopy shade can greatly influence
7041 stream energy budgets to moderate stream temperatures despite changes and/or removal of the
7042 overstory canopy.

7043

7044 **Large woody debris**

7045

7046 Jones et al., 2011 (Removed from focal list, study not relevant to focal questions)

7047

7048 Jones, T.A., Daniels, L.D., Powell, S.R., 2011. Abundance and function of large woody debris in
7049 small, headwater streams in the Rocky Mountain foothills of Alberta, Canada. *River Research*
7050 *and Applications* 27, 297–311. <https://doi.org/10.1002/rra.1353>

7051

7052 The purpose of this study was to assess LW abundance in the upper foothills of the Rocky
7053 Mountains in Alberta, Canada. This study also sought to understand key processes that underlie
7054 changes in LW function. Finally, this study used results to develop a LW recruitment, decay and
7055 interaction model. This research was conducted in 21 headwater streams spanning two
7056 watersheds. At each site, all LW was sampled and was classified according to decay, orientation,
7057 position and function. LW frequency, total volume, and total in-stream volume were calculated
7058 and analyzed for differences using a one-way ANOVA followed by a Tukey post hoc test to
7059 differentiate among significant classes. Results show LW frequency was greater in the Alberta
7060 foothills (64.0 ±3.3 LW 100 m¹) than in many small, headwater streams in mountain (46.2 ±3.6),
7061 coastal (47.6 ±3.8), mixed broad-leaf (47.0 ±4.2) and boreal (31.0 ±3.0) streams. This, the
7062 authors suggest, is likely due to the narrow bankfull width channels characteristic of the Alberta
7063 foothills which are less able to transport LW downstream. LW with ≥20 cm was more frequent in
7064 coastal streams, and overall LW volume was also greatest in coastal streams (721.0 ±99.9 m³ ha⁻¹).
7065 The authors note that large LW volumes in coastal streams are likely due to geomorphic
7066 disturbances alongside large, long-lived, decay resistant tree species. According to Harmon et al.
7067 1986, much of the variation in LW recruitment is due to differences in species life history and
7068 forest type which together govern log size and decay rates.

7069

7070 **Suspended Sediment**

7071

7072 Karwan et al., 2007

7073

7074 Karwan, D., Gravelle, J., Hubbart, J., 2007. Effects of timber harvest on suspended sediment
7075 loads in Mica Creek, Idaho. *Forest Science* 53, 181–188.
7076 <https://doi.org/10.1093/forestscience/53.2.181>

7077

7078 The purpose of this study was to examine the effects of forest road construction and timber
7079 harvest on total suspended solids (TSS) in a forested watershed. This study took place at the
7080 Mica Creek Experimental Watershed in northern Idaho. The study area consisted of dense,
7081 naturally regenerated, even-aged stands ~65 years old and ~300 trees per acre. Timber harvesting
7082 and heavy road use began in 2001. Treatments in the paired-watershed experiment consisted of
7083 (1) commercial clearcut of the watershed area of 50%, and was broadcast burned and replanted
7084 by the end of May 2003, (2) partial cut in which half the canopy was removed in 50% of the
7085 watershed in 2001, with final 10% of log processing and hauling in early summer of 2002, and
7086 (3) a no-harvest control. All harvests were carried out according to best management practices
7087 and in accordance with the Idaho Forest Practices Act. At the time of the study this involved a
7088 22.86 m (75 ft) stream protection zones (SPZs) on each side of fish-bearing (Class I) streams.
7089 The inner 50 ft is an equipment exclusion zone where no ground-based skidding machinery is
7090 allowed. Timber harvesting is allowed in Class I SPZs, but 75% percent of existing shade must
7091 be retained. Along non-fish-bearing (Class II) streams, harvesting equipment was excluded from
7092 entering within 9.14 m (30 ft) of definable stream channels and any cut trees were felled away
7093 from the stream; however, there were no tree retention requirements. In the clearcut and partial
7094 cut units, line skidding was used on slopes in the watershed exceeding approximately 20%, while
7095 tractor skidding was used on the lower gradient slopes. On all skid trails, drainage features, such
7096 as water bars, were installed for erosion control at the end of the harvest period. Time series data
7097 were compiled for all measured TSS values from 1991 through 2004. Data was collected via
7098 seven stream monitoring flumes located within the Mica Creek Watershed. Monthly TSS loads
7099 were compared across watersheds for five time intervals: (1) pretreatment: ~6 years, (2)
7100 immediate post-road construction: ~1 year, (3) recovery post-road construction: ~3 years, (4)
7101 immediate post-harvest: ~1 year, and (5) recovery post-harvest: ~3 years. Trends in the
7102 relationship between treatment and control watersheds were statistically examined for each of the
7103 time intervals. Treatments in the paired-watershed experiment consisted of (1) commercial
7104 clearcut of the watershed area of 50%, and was broadcast burned and replanted, (2) partial cut in
7105 which half the canopy was removed in 50% of the watershed (3) a no-harvest control. All
7106 harvests were done according to best management practices and the Idaho Forest Practices Act.
7107 This included equipment exclusion zones of 50- and 30-feet for fish- and non-fish-bearing
7108 streams, respectively. On all skid trails, drainage features, such as water bars, were installed for
7109 erosion control at the end of the harvest period. Analysis of covariance was used for each
7110 treatment-control watershed pair. Results show monthly TSS loads from watersheds 1 (clearcut),
7111 2 (partial cut), and 3 (no-harvest) ranged from 0.4 kg km⁻² to above 10,000 kg km⁻², with a
7112 maximum in the spring months and minimum in the winter and late summer months similar to
7113 intra-annual trends in water yield. Road construction in both watersheds did not result in
7114 statistically significant impacts on monthly sediment loads in either treated watershed during the
7115 immediate or recovery time intervals. A significant and immediate impact of harvest on monthly
7116 sediment loads in the clear-cut watershed ($p = 0.00011$), and a marginally significant impact of
7117 harvest on monthly sediment loads in the partial-cut ($p = 0.081$) were observed. Total sediment
7118 load from the clearcut over the immediate harvest interval exceeded predicted load by 152%
7119 (6,791 kg km⁻²); however, individual monthly loads varied around this amount. The largest
7120 increases in percentage and magnitude occurred during snowmelt months, namely April 2002

7121 (560%, 2,958 kg km⁻²) and May 2002 (171%, 3,394 kg km⁻²). Neither treatment showed a
7122 statistical difference in TSS during the recovery time (clearcut: p = 0.2336; partial-cut: p =
7123 0,1739) compared to calibration loads (pre-treatments). The authors conclude that best
7124 management practices for road construction, including improvement of existing roads, did not
7125 produce significant changes in TSS. Significant changes in TSS only occurred immediately after
7126 harvest. However, after one year, the TS load became statistically indistinguishable from the
7127 control.

7128

7129 **Harvest effects on Instream light**

7130

7131 Kaylor et al., 2017

7132

7133 Kaylor, M.J., Warren, D.R., Kiffney, P.M., 2017. Long-term effects of riparian forest harvest on
7134 light in Pacific Northwest (USA) streams. *Freshwater Science* 36, 1–13.
7135 <https://doi.org/10.1086/690624>

7136

7137 The purpose of this study was to evaluate relationships between riparian forest stand age and
7138 stream light availability. The specific goals dealt with evaluating characteristics of late-
7139 successional forest light regimes, and whether canopy openness and light differed between
7140 streams flowing through harvested units and late-successional forest units. This study took place
7141 at the HJ Andrews Experimental Forest in the Cascade Mountain, Oregon. Approximately 11.5
7142 km of stream length were sampled in the McCrae Basin which consists mostly of old-growth
7143 forests Douglas-fir forests with small patch clear cuts. All treatment sites were harvested within
7144 50 to 60 years before the study. Clearing up to both stream banks occurred at two of seven
7145 treated sites and clearing up to one bank occurred on all other treated sites. Stream bank-full
7146 width, wetted width, canopy openness, % red alder, and estimated photosynthetically active
7147 radiation (PAR) were quantified at 25-m intervals to evaluate relationships between channel and
7148 riparian characteristics and stream light. Results from this study show mean estimated PAR
7149 reaching the streams was lower in the recovering harvested units (50-year post-treatment) than
7150 in up and downstream reaches bordered by old growth for all comparisons (n=14), while only 6
7151 were significant (p<0.05). All in all, old growth reaches averaged 1.7 times greater PAR values
7152 than in nearby harvested units with the greatest differences occurring when harvest was
7153 implemented on both banks. Mean canopy openness was higher in late-successional forests (>
7154 300 years old) than in young second growth forests (30–100-year-old forests), 18% and 8.7%
7155 respectively. Results also indicate the relationship between canopy openness and PAR was
7156 stronger at the reach scale than at individual locations with mean canopy openness explaining
7157 78% of the variance in mean PAR estimates. The researchers also conducted a review of
7158 available literature of studies that contained information on the effects of Northwest Douglas-fir
7159 forest growth dynamics on canopy cover and light availability. The researchers concluded from

7160 this review that canopy closure, and thus lower light availability, occurs approximately 30 years
7161 after growth and maintained until after 100 years of growth when the canopy structure begins to
7162 open and produce gaps. Altogether, this study suggests stream light regimes are affected by
7163 initial canopy removal and subsequent recovery. Depending on forest type, dominant species and
7164 the age of the stand, different stages of stand development may reflect complex overstory
7165 structures allowing variable levels of light to the stream.

7166

7167 **Stream Temperatures**

7168

7169 Kibler et al., 2013

7170

7171 Kibler, K.M., Skaugset, A., Ganio, L.M., Huso, M.M., 2013. Effect of contemporary forest
7172 harvesting practices on headwater stream temperatures: Initial response of the Hinkle Creek
7173 catchment, Pacific Northwest, USA. *Forest Ecology and Management* 310, 680–691.
7174 <https://doi.org/10.1016/j.foreco.2013.09.009>

7175

7176 The purpose of this study was to investigate the effects of contemporary forest harvesting
7177 practices on headwater stream temperatures using a BACI design. This study was conducted as
7178 part of the Hinkle Creek paired Watershed Study (HCPWS). This study consisted of a nested,
7179 paired watershed study in which harvesting treatments in accordance with the Oregon Forest
7180 Practices Act (FPA) were applied to four headwater catchments in southern Oregon. Oregon FPA
7181 does not require retention of fixed-width buffer strips adjacent to non-fish-bearing streams. Thus,
7182 as a part of the harvest activities, fixed-width buffer strips containing merchantable overstory
7183 conifers were not left adjacent to the non-fish-bearing streams. Clearcut harvest took place
7184 between August 2005 and May 2006. Streamflow and temperature were measured at 8 locations
7185 within the basin from autumn 2002 until autumn of 2006 giving 3 years of pre-harvest data and
7186 <1 year of post-harvest data. Treatment and reference catchments were paired based on similarity
7187 in catchment area, aspect, stream orientation, stream length, and discharge. Significant
7188 differences between pre- and post-harvest daily max temperature measurements were detected
7189 across all sites, however, magnitude and direction of changes were inconsistent. Results for daily
7190 mean maximum stream temperatures show a variable response across all four harvested streams
7191 ranging from 1.5°C cooler to 1.1°C warmer relative to pre-harvest years. No statistically
7192 significant changes in max, mean, or minimum daily stream temperatures to timber harvest were
7193 observed. The authors suggest possible explanations for lack of consistent temperature increases
7194 to shading provided by logging slash. Interestingly, statistically significant changes to
7195 relationship between treatment and reference site pairs with respect to minimum and mean
7196 stream temperatures resulted in decreased minimum daily stream temperatures on days where
7197 high temperatures were observed in reference streams. At one treatment site, mean minimum
7198 temperatures across the warm season decreased 1.9°C relative to pre-harvest years, and the

7199 minimum temperature on the warmest day decreased by 2.8°C relative to pre-harvest years.
7200 Except for one treatment-reference pair, highly significant changes to slope and intercept
7201 parameters of minimum daily stream temperatures were detected for each stream pair ($p < 0.001$).
7202 The authors suggest decreases in daily minimum stream temperature is a likely consequence of
7203 timber harvest.

7204

7205 **Shade and Stream temperature**

7206

7207 Cupp & Lofgren, 2014

7208

7209 Cupp, C.E. & Lofgren, T.J. (2014). Effectiveness of riparian management zone prescriptions in
7210 protecting and maintaining shade and water temperature in forested streams of Eastern
7211 Washington. Cooperative Monitoring Evaluation and Research Report CMER 02-212.
7212 Washington State Forest Practices Adaptive Management Program. Washington Department of
7213 Natural Resources, Olympia, WA.

7214

7215 The purpose of this study was to assess the percent reduction in canopy cover, and the response
7216 in stream temperatures following riparian timber harvest under the “all available shade” rule
7217 (ASR), and the standard rule (SR) in eastern Washington. The ASR is applied to areas in the Bull
7218 Trout Habitat Overlay (BTO; map of bull trout habitat) that requires retention of all available
7219 shade within 75 feet of the stream. Under the standard shade rule (SR) some harvest is allowed
7220 within the 75-foot buffer depending on elevation and pre-harvest canopy cover. The primary
7221 objectives of this study were to (1) Quantify and compare differences in post-harvest canopy
7222 closure between the SR and the ASR riparian prescriptions of eastern Washington; and (2)
7223 Quantify and compare differences in stream temperature effects of the two riparian prescriptions:
7224 the SR and the ASR. This study was conducted at 30 sites in eastern Washington. Sites were
7225 between 65-100 years old and were situated along second to fourth order streams with harvest-
7226 regenerated or fire-regenerated forests. Reference reaches were located upstream from treatment
7227 reaches where harvest was applied. Eighteen sites were located on state owned and managed
7228 forests and 12 sites were located on private industrial forests. Prior to harvest treatments, canopy
7229 closure measurements ranged from 89% to 97%, with a mean of 93%. The riparian management
7230 zone (RMZ) consists of three zones: The core zone is nearest to the edge of the stream and
7231 extends out 30 feet horizontally from the bankfull edge or outer edge of the channel migration
7232 zone (CMZ), whichever is greater. The inner zone is situated immediately outside of the core
7233 zone. For streams with a bankfull width of less than or equal to 15 feet wide, the inner zone
7234 width is 45 feet wide. All streams assessed in this study were less than or equal to 15 feet wide.
7235 The outer zone of the RMZ is the zone furthest from the water and its width varies according to
7236 stream width and site class for the land. The specific site class (a measure of site productivity) at
7237 each treatment site would vary the outer zone width from 0 to 55 feet wide. Seven sites had up to

7238 four years pre-harvest temperature data with only two years post-harvest data. Nine sites had
7239 three years pre-harvest data and one site had only one year pre-harvest data. The remaining 13
7240 sites had two years pre-harvest data. Following harvest treatments, all 30 sites had at least two
7241 years post-harvest temperature data collection, although 21 of the 30 sites had at least three years
7242 post-harvest monitoring. Data collection included twice hourly stream and air temperature data
7243 during each sample period. Canopy, shade, riparian, and channel data were collected during the
7244 first-year pre-harvest and the first year post-harvest. Stream temperature data were collected at
7245 30-minute intervals between 1 July and 15 September for a total of 77 days each year a site was
7246 investigated. Stream canopy closure and shade were quantified at 75-ft intervals within each
7247 reach using a hand-held densiometer (for canopy closure measurements) and a self-leveling
7248 fisheye lens digital camera (for shade measurements). A t-test was used to evaluate differences in
7249 pre-harvest canopy cover between reference and treatment reaches, and between ASR and SR
7250 sites. A correlation analysis between post-harvest change in shade and the descriptive riparian
7251 and channel values (e.g., trees per acre, basal area, channel gradient, etc.) was also used to
7252 examine possible factors that may control post-harvest changes in shade. A linear mixed effects
7253 model was used to quantify and compare differences in daily max stream temperatures (DMAX)
7254 between no harvest, ASR and SR prescriptions. Results showed post-harvest shade values
7255 decreased in SR sites (mean effect of -2.8%, $p = 0.002$), as did the canopy closure values (mean
7256 effect of -4.5%, $p < 0.001$). Shade and canopy closure values did not significantly change in the
7257 treatment reaches of the ASR sites. Mean shade reduction in the SR treatment sites exceeded the
7258 mean shade reduction in the ASR sites by 3%. Canopy closure reduction was also greater in the
7259 SR sites than in the ASR sites by a mean of 4%. Specifically, the mean shade reduction in ASR
7260 sites was 1% with a maximum reduction of 4%. The mean reduction of shade in the SR sites was
7261 4% with a maximum reduction of 10%. Mean shade contribution of upland trees (trees outside of
7262 the RMZ) per study site was calculated as $< 1\%$. Shade reduction levels did not differ between
7263 the sites receiving RMZ-harvest only and the sites receiving standard operational upland harvest.
7264 Site seasonal means of daily maximum stream temperature treatment responses in the first two
7265 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
7266 the SR reaches. Site seasonal mean post-harvest background responses in reference reaches
7267 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
7268 stream temperature increased $0.16\text{ }^{\circ}\text{C}$ in the SR harvest reaches, whereas stream temperatures in
7269 both the ASR sites and in the no-harvest reference reaches increased on average by $0.02\text{ }^{\circ}\text{C}$.
7270 Seasonal mean stream temperature responses of up to $0.5\text{ }^{\circ}\text{C}$ in the no-harvest references were
7271 common during the post-harvest test period. Sample period means of daily maximum
7272 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
7273 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
7274 authors interpret these results as evidence that temperature effects of the SR, and ASR were
7275 similar to reference conditions along sampled reaches for small streams in the mixed fir zone
7276 mid-successional forests of eastern Washington. Further, that processes not directly related to
7277 canopy cover alteration over streams may be primarily responsible for the small variations
7278 observed in stream temperatures following harvest.

7279

7280

7281 Ehinger et al., 2021 (~~results are only descriptive~~)

7282

7283 Ehinger, W.J., W.D. Bretherton, S.M. Estrella, G. Stewart, D.E. Schuett-Hames, and S.A. Nelson.
7284 2021. Effectiveness of Forest Practices Buffer Prescriptions on Perennial Non-fish-bearing
7285 Streams on Marine Sedimentary Lithologies in Western Washington. Cooperative Monitoring,
7286 Evaluation, and Research Committee Report CMER 2021.08.24, Washington State Forest
7287 Practices Adaptive Management Program, Washington Department of Natural Resources,
7288 Olympia, WA.

7289

7290 The purpose of this study was to assess the effectiveness of riparian management zone
7291 prescriptions in maintaining functions and processes in headwater perennial, non-fish-bearing
7292 streams in incompetent (easily eroded) marine sedimentary lithologies in western Washington.
7293 Specifically, this study used a multiple before after control impact (MBACI) design to compare
7294 unharvested reference sites to sites harvested under the western Washington Forest Practices for
7295 non-fish-bearing streams to assess the effects of these rules on riparian vegetation and wood
7296 recruitment, canopy closure and stream temperature, stream discharge and downstream transport
7297 of suspended sediment and nitrogen, and benthic macroinvertebrates. The Forest Practices rules
7298 for non-fish-bearing streams in the study area includes clearcut harvest with a two-sided 50-foot-
7299 wide riparian buffer along at least 50% of the riparian management zone, including buffers
7300 prescribed for sensitive sites and unstable slopes. Ten study sites were chosen with first-, second-
7301 , and third-order non-fish-bearing streams. Data was collected for 1-2 years of pre-harvest,
7302 during the harvest period (2012 – 2014), and at least 2 years post-harvest at all sites. Because of
7303 unstable slopes, total buffer area was 18 to 163% greater than the 50-foot-buffer. This resulted in
7304 4 different buffer types 1) Buffers encompassing the full width (50 feet), 2) <50ft buffers, 3)
7305 Unbuffered, harvested to the edge of the channel, and 4) Reference sites in unharvested forests.
7306 Because of the separation into multiple treatments, sample sizes became small and unbalanced.
7307 Thus, no statistical analyses were conducted, and only descriptive statistics were applied for
7308 changes in stand structure and wood loading. Density decreased by 33 and 51% and basal area
7309 by 26 and 49% in the full and <50ft buffers, respectively, with high variability among sites.
7310 Nearly all trees were removed from Unbuffered sites during harvest (>99% of basal area). In the
7311 reference plots, cumulative post-harvest mortality during the 3-year post-harvest interval was
7312 only 6.5% of live density. In contrast, mean post-harvest mortality in the full buffer sites and the
7313 <50 ft buffer sites were 31 and 25% of density, respectively. However, there was considerable
7314 variation in mortality among sites exceeding 65% in two full buffer treatment sites. Windthrow
7315 and physical damage from falling trees accounted for ~75% of mortality in the full and <50 ft
7316 buffers. In contrast to the treated sites, <10% of trees died due to wind or physical damage in the
7317 reference sites. There was little post-harvest large wood input in reference sites: an average of
7318 4.3 pieces and 0.34 m³ of combined in- and over-channel volume per 100 m of channel. In
7319 contrast, the full buffer sites and <50 ft buffer sites received an average of 23 and 10 pieces/100

7320 m and 2.3 and 0.7 m³/100 m of large wood, respectively. The majority of recruited large wood
7321 pieces had stems with roots attached (SWRW); 60, 70, and 100% in the reference, full buffer,
7322 and <50 ft buffer types, respectively. Pre-harvest channel large wood loading ranged from 55.8 to
7323 111 pieces/100 m and from 9.8 to 25.2 m³/100 m among buffer types. Piece counts remained
7324 stable in the reference sites through year 3 post-harvest, increased in the full buffer and
7325 unbuffered sites (8 and 13%, respectively), and decreased in the <50 ft buffers (15%). For effects
7326 of treatment on shade, data was analyzed with generalized linear mixed-effects models. For
7327 effects of treatment on stream temperature, data was analyzed for the seven-day average in a
7328 linear-mixed-effects model analysis of variance. Mean canopy closure decreased in the treatment
7329 sites from 97% in the pre-harvest period to 75%, 68%, and 69% in the first, second, and third
7330 post-harvest years, respectively, and was related to the proportion of stream buffered and to post-
7331 harvest windthrow within the buffer. The seven-day average temperature response increased by
7332 0.6°C, 0.6°C, and 0.3°C in the first, second, and third post-harvest years, respectively. During
7333 and after harvest, mean monthly water temperatures were higher, but equaled or exceeded
7334 15.0°C only in 2 treatment sites by up to 1.8°C at one site and by 0.1°C at another. None of the
7335 three REF sites exceeded 15°C during the study. Predictive models could not be fitted to the
7336 temperature data for statistical analysis. Results for changes in nutrient concentrations post-
7337 harvest were highly variable. Harvest treatment effects on nutrient concentrations, discharge, and
7338 suspended sediment export could not be calculated because prediction equations could not be
7339 developed.

7340

7341

7342 McIntyre et al., 2018

7343

7344 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn
7345 (technical coordinators). 2018. Effectiveness of Experimental Riparian Buffers on Perennial
7346 Non-fish-bearing Streams on Competent Lithologies in Western Washington. Cooperative
7347 Monitoring, Evaluation and Research Report CMER 18-100, Washington State Forest Practices
7348 Adaptive Management Program, Washington Department of Natural Resources, Olympia, WA.

7349

7350 The purpose of the study was to evaluate the effectiveness of forest management prescriptions in
7351 maintaining aquatic conditions and processes for small non-fish-bearing (Type N) headwater
7352 stream basins underlain by competent “hard rock” lithologies (i.e., volcanic or igneous rock) in
7353 western Washington. Specifically, this study quantified and compared the effects of timber
7354 harvest adjacent to Type N streams on riparian stand structure and tree mortality, in stream wood
7355 loading and recruitment, stream temperature and canopy cover, stream discharge, turbidity, and
7356 suspended sediment export, nitrogen export, and response of stream associated amphibians. This
7357 study used a before-after control-impact (BACI) study design. This involved evaluation of four
7358 experimental treatments: (1) unharvested reference (n = 6), (2) 100% treatment (n = 4), a two-

7359 sided 50-ft riparian buffer along the entire Riparian Management Zone (RMZ), (2) FP treatment
7360 (n = 3), a two-sided 50-ft riparian buffer along at least 50% of the RMZ, consistent with the
7361 current Forest Practices buffer prescription for Type N streams, This treatment also included a
7362 circular buffer protecting the uppermost points of perennial flow (PIP), (3) 0% treatment (n = 4),
7363 clearcut to stream edge (no-buffer). The upland forests of all treatments were clearcut harvested.
7364 The study design included data collection for at least two years pre-harvest (2006 –2008), and
7365 three years of post-harvest data (2009 – 2011). Results for stand structure and tree mortality
7366 showed that in the RMZs, the proportional changes in stem count (dstems) and basal area (dBA)
7367 were similar for the reference (mean dstems: -11.8, SE 5.3; dBA: -6.9, SE 5.4) and 100% (mean
7368 dstems: -3.8, SE 5.9; dBA -6.7, SE 6.0) treatment. In contrast, the magnitude of decrease was
7369 significantly greater in the FPB (portion of FP containing trees; mean dstems: -29.6, SE 6.5; dBA
7370 124.4, SE 6.7) treatment than in either the reference or 100% treatment. The pattern was similar
7371 in the PIPs. 2 years post-harvest tree mortality was mostly (70%) attributed to wind/mechanical
7372 agents (pre-harvest wind/mechanical agent caused mortality was 70%). In the reference sites,
7373 trees that died post-harvest had smaller diameters (mean 10.3 in) and fewer came from the
7374 overstory crown class (59.0%) than the other treatments. In contrast, in the 100% and FPB
7375 treatments, ~70% of trees that died were from the overstory crown class and their mean
7376 diameters were 1 (11.2 in) and 2 (12.2 in) in greater than those in the reference sites,
7377 respectively. Results for wood recruitment and loading showed that tree fall rates were highly
7378 variable during the pre-harvest period between sites ranging from 0 to 239.9 trees/ha/yr. Large
7379 wood (LW) recruitment rates in the pre-harvest period were also highly variable ranging from 0
7380 to 121.6 pieces/ha/yr, along with recruitment volume (0-16.2 m³/ha/yr). 2 years post-harvest
7381 recruitment rates in the reference riparian management zones (RMZs) were lower and less
7382 variable (5.9 to 37.3 trees/ha/yr) than in buffer treatments. Tree fall rates for the 100% treatment
7383 ranged from 7.7 to 76.4 trees/ha/yr, and for the FPB treatments tree fall rates ranged from 4.2 to
7384 152.2 trees/ha/yr. Post-harvest LW recruitment volumes in reference RMZs were relatively low,
7385 ranging from 0.7 to 2.2 m³/ha/yr. Post-harvest LW recruitment volumes were generally higher
7386 and more variable in the 100% and FPB RMZs, ranging from 0.3 to 14.0 m³/ha/yr in the 100%
7387 treatment and 0 to 7.6 m³/ha/yr in the FPB. Because of the high variability between sites in all
7388 treatments the p values for comparisons between treatments were generally high (p ≥ 0.35),
7389 except for the FPB vs. reference comparison for piece count which was nearly significant (p =
7390 0.13). The only significant differences were for the 0% treatments which had significantly lower
7391 LW recruitment by volume than the Reference RMZ (P = 0.02). For PIPs, LW recruitment in the
7392 100% treatment was over 12 times the reference rate by piece count (P = 0.03) and 30 times the
7393 reference rate by volume (P = 0.04). Recruitment in the FPB PIPs was also high, over nine times
7394 the reference rate by piece count (P = 0.08) and 18 times the reference rate by volume (P = 0.11).
7395 The amount of change in the number of LW pieces per meter from pre-harvest to post-harvest
7396 depended on treatment (P <0.01). Analysis estimated the changes in 100%, FP and 0% treatments
7397 to be different from the change in the reference (P <0.001, 0.03 and <0.01, respectively). The
7398 percentage of the stream channel length covered by newly recruited wood in the second post-
7399 harvest year ranged from 0 to 11% in the reference, 1 to 15% in the 100% treatment and 0 to
7400 10% in the FP treatment and was 0% in all four of the 0% treatments. The percent of stream
7401 channel covered by new wood differed between the 0% treatment and the reference (P = 0.03),

7402 100% ($P < 0.01$), and FP treatments ($P = 0.03$). Overall, the authors estimated a mean between-
7403 treatment increase of 60% (95% CI: 0–150%), 70% (95% CI: 0–190%) and 170% (95% CI:
7404 80–330%) in the number of SW pieces per stream meter in the 100%, FP and 0% treatments
7405 compared with the reference, respectively. Also, a between-treatment increase of 60% (95% CI:
7406 30–110%), 40% (95% CI: 0–100%) and 50% (95% CI: 10–90%) in the number of LW pieces
7407 per stream meter in the 100%, FP and 0% treatments compared with the reference, respectively.
7408 The authors conclude that windthrow was responsible for much of the increase in LW. However,
7409 they also posit that the timing and magnitude of wood inputs was inconsistent, resulting in
7410 considerable variability between and within sites, especially in the FP treatment. Results for
7411 shade response to treatments post-harvest was greatest in the 0% treatment than in either the
7412 100% or the FP treatment. Effective shade decreased to 77, 52, and 14% 2 years post-treatment,
7413 in the 100%, FP, and 0% buffer treatments, respectively. Canopy and Topographic Density
7414 (CTD), defined as the percentage of the photograph obscured by vegetation or topography
7415 decreased from an average of 95% pre-harvest to 86, 71, and 43% 2 years post-harvest in the
7416 100%, FP, and 0% buffer treatments, respectively. All were significantly lower than the reference
7417 (92% 2 years post-treatment). Results for stream temperature showed maximum daily water
7418 temperatures increased post-harvest in all but one of the harvested sites and was elevated over
7419 much of the year at most of the sites. Daily temperature response (TR) increased in late winter or
7420 early spring, reached a maximum in July–August and was still elevated well into the fall. This
7421 pattern was observed at most of the sites. For the Buffer Treatment locations, 94 of the 131
7422 calculated mean monthly temperature responses (MMTRs) were significant and 91 of these
7423 significant responses were positive. In comparison, only 52 of 156 MMTR values calculated for
7424 the reference sites were significant and these were nearly evenly split with 25 positive and 27
7425 negative responses. This strongly suggests that the pattern of post-harvest increases in daily
7426 maximum water temperature is real even though the magnitude of some of the individual
7427 MMTRs is relatively small ($< 0.5^{\circ}\text{C}$). Warming tended to be greatest in July or August with
7428 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
7429 in the 0% treatments. Post-harvest, Max7D (seven-day-average maximum stream temperature)
7430 was higher at 36 of the 40 locations within the harvest units across all 11 buffer treatment sites
7431 regardless of presence or absence of a buffer, buffer width, and longitudinal location along the
7432 stream. Relative to the unharvested sites, there were summertime temperature increases
7433 throughout the stream length and across all buffer treatment sites. The authors conclude that none
7434 of the buffer treatments were successful in preventing significant increases in maximum stream
7435 temperature. The generalizable conclusions made by the authors from this portion of the study
7436 are that 1) Buffer widths greater than 50 ft (15.2 m) are needed to prevent shade loss and (2)
7437 Maximum water temperature decreased below the harvest unit after flowing through
7438 approximately 100 m of intact forest but was still elevated compared to pre-harvest conditions.
7439 Results for nitrogen and phosphorus concentrations showed that post-harvest changes for total-N
7440 or total-P were not significant for any of the treatments relative to the Reference. The only
7441 significant difference detected within 2 years post-harvest was for nitrate-N concentration
7442 between the 0% buffer treatment and all other treatments. However, for annual export, total-N
7443 and nitrate-N export increased post-harvest at all sites, with the smallest increase in the 100%
7444 treatment and the largest in the 0% treatment. Compared to the reference sites, the GLMM

7445 analysis showed a relative increase in total-N export post-harvest of 5.52 (P = 0.051), 11.52 (P =
7446 0.0007), and 17.16 (P <0.0001) kg ha⁻¹ yr⁻¹ in the 100%, FP, and 0% treatments. The GLMM
7447 analysis showed a relative increase in nitrate-N export post-harvest of 4.83 (P = 0.048), 10.24 (P
7448 = 0.001), and 15.35 (P <0.0001) kg ha⁻¹ yr⁻¹ in the 100%, FP, and 0% treatments, respectively,
7449 only slightly less than the changes in total-N. Total-P export increased post-harvest by a similar
7450 magnitude in all treatments: 0.10 (P = 0.006), 0.13 (P = 0.001), and 0.09 (P = 0.010) kg ha⁻¹ yr⁻¹
7451 in the 100%, FP, and 0% treatments, respectively. The increase in N, total-N and nitrate-N, from
7452 the treatment watersheds post-harvest was strongly correlated with the increase in annual runoff
7453 (R² = 0.970 and 0.971; P = 0.001 and 0.001, respectively) and with the proportion of the basin
7454 harvested (R² = 0.854 and 0.852; P = 0.031 and 0.031, respectively). The correlation with the
7455 proportion of stream length buffered was weaker (R² = 0.761 and 0.772; P <0.079 and 0.072,
7456 respectively). In contrast, total-P export was uncorrelated with all three variables. Overall, the
7457 authors concluded that mean flow-weighted concentration of total-N and nitrate-N increased at
7458 all buffer treatment sites post-harvest, however the magnitude was variable and significant only
7459 for the 0% treatment. However, the export of total-N increased in the FP and 0% treatments and
7460 nitrate-N increased in all buffer treatments. Increases in N export was correlated with increased
7461 stream discharge and the proportion of the site that was harvested. Pre-harvest total-P
7462 concentration was low and remained so post-harvest, although P export increased slightly post-
7463 harvest in all treatments due to the increase in discharge. Results for changes in water turbidity
7464 and suspended sediment concentrations (SSC) showed both turbidity and SSC increased with
7465 increasing discharge during storm events but then rapidly fell off. Analysis of treatment effects
7466 revealed no significant effects of harvest and no clear pattern regarding the relative effectiveness
7467 of buffer treatments at mitigating the effects of clearcut harvests on suspended sediment export
7468 (SSE). The general conclusions made by the authors were that all sites appeared to be supply
7469 limited both pre- and post-harvest. Results for litterfall input showed a decrease in TOTAL
7470 litterfall input in the FP (P = 0.0034) and 0% (P = 0.0001) treatments between pre- and post-
7471 treatment periods. LEAF litterfall (deciduous and conifer leaves combined) input decreased in
7472 the FP (P = 0.0114) and 0% (P <0.0001) treatments in the post-treatment period. In addition,
7473 CONIF (conifer needles and scales) litterfall input decreased in the FP (P = 0.0437) and 0% (P
7474 <0.0001) treatments, DECID (deciduous leaves) in the 0% (P <0.0001) treatment, WOOD (twigs
7475 and cones) in the FP (P = 0.0044) and 0% (P = 0.0153) treatments, and MISC (e.g., moss and
7476 flowers) in the 0% (P = 0.0422) treatment. Results for comparison of the post-harvest effects
7477 between treatments showed LEAF litterfall input decreased in the 0% treatment relative to the
7478 reference (P = 0.0040), 100% (P = 0.0008), and FP (P = 0.0267) treatments. Likewise, there was
7479 a decrease in DECID litterfall input in the 0% treatment relative to the Reference (P = 0.0001),
7480 100% (P <0.0001), and FP (P = 0.0015) treatments. Results for detritus with comparisons
7481 between the pre- and post-treatment periods showed an increase in TOTAL detritus export in the
7482 100% treatment (P = 0.0051) and a decrease in the 0% treatment (P = 0.0046; Table 12-9).
7483 Likewise, there was an increase in CPOM, WOOD, MISC, and FPOM detritus export in the
7484 100% treatment (P <0.05), but a decrease in the 0% treatment (P <0.05) The authors for this
7485 portion of the study conclude that overall, total litterfall input was slightly higher after harvest in
7486 the 100% treatment, lower in the FP treatment and lowest in the 0% treatment; however,
7487 statistical differences were only detected for deciduous inputs between the 0% treatment and the

7488 other treatments. Total detritus export decreased in the 0% treatment relative to the reference,
7489 and in the FP and 0% treatments relative to the 100% treatment.

7490

7491

7492 McIntyre et al., 2021

7493

7494 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D.E. Schuett-Hames, R. Ojala-Barbour,
7495 G. Stewart and T. Quinn (technical coordinators). 2021. Effectiveness of experimental riparian
7496 buffers on perennial non-fish-bearing streams on competent lithologies in western Washington –
7497 Phase 2 (9 years after harvest). Cooperative Monitoring, Evaluation and Research Report CMER
7498 2021.07.27, Washington State Forest Practices Adaptive Management Program, Washington
7499 Department of Natural Resources, Olympia, WA.

7500

7501 This study was a follow-up study to the hard-rock Phase 1 study (McIntyre et al., 2018) to assess
7502 changes over longer time periods (up to 9 years post-harvest). The purpose of the study was to
7503 evaluate the effectiveness of forest management prescriptions in maintaining aquatic conditions
7504 and processes for small non-fish-bearing (Type N) headwater stream basins underlain by
7505 competent “hard rock” lithologies (i.e., volcanic or igneous rock) in western Washington.
7506 Specifically, this study quantified and compared the effects of timber harvest adjacent to Type N
7507 streams on riparian stand structure and tree mortality, in stream wood loading and recruitment,
7508 stream temperature and canopy cover, stream discharge, turbidity, and suspended sediment
7509 export, nitrogen export, and response of stream associated amphibians. This study used a before-
7510 after control-impact (BACI) study design. This involved evaluation of four experimental
7511 treatments: (1) unharvested reference (n = 6), (2) 100% treatment (n = 4), a two-sided 50-ft
7512 riparian buffer along the entire Riparian Management Zone (RMZ), (2) FP treatment (n = 3), a
7513 two-sided 50-ft riparian buffer along at least 50% of the RMZ, consistent with the current Forest
7514 Practices buffer prescription for Type N streams, (3) 0% treatment (n = 4), clearcut to stream
7515 edge (no-buffer). The upland forests of all treatments were clearcut harvested. The study design
7516 included data collection for at least two years pre-harvest (2006 –2008), and up to nine years
7517 post-harvest from 2009 (harvest began in 2008) until 2016 or 2017 depending on the variable
7518 (e.g., wood loading, shade, etc.). Results for stand structure showed that in the buffered portions
7519 of the FP treatments (FPB) density, basal area and relative density (RD) decreased by 59%, 55%
7520 and 54%, respectively, 8 years after harvest. For the same variables, reductions in the 100%
7521 RMZs were 30%, 14%, and 17%, respectively. In contrast, stand structure in the reference RMZs
7522 was more stable, with a 17% decrease in density and little change in basal area or RD. Change in
7523 live basal area did not differ statistically between 100% and REF RMZs for any time interval
7524 although the differences increased over time. The FPB–REF contrast was not significant in the
7525 first interval (years 1 and 2 post-harvest), but it was in subsequent intervals (5- and 8-years post-
7526 harvest) as the magnitude of change in FPB RMZs increased over time. The FPB–100% contrast

7527 was not significant until the last interval when basal area stabilized in the 100% treatment but
7528 continued to decline in FPB. Between treatment comparison of cumulative change in live basal
7529 area (m²/ha) between the 100% treatment and the Reference was -2.9 (CI: -16.9, 11.0), -6.0 (CI:
7530 -20.0, 8.0), and -6.8 (CI -20.8, 7.1) for the first-, second-, and third-time intervals respectively
7531 (none were significant). Comparison between the FPB and Reference were -10.2 (CI: -25.5, 5.2),
7532 -16.1 (CI: -31.4, -0.8), and -21.1 (CI: -36.4, -5.8) for the first-, second-, and third-time intervals
7533 respectively (differences for intervals 2 and 3 were significant). For tree mortality, results
7534 showed that by year 8 post-harvest mortality as a percentage of pre-harvest basal area was lower
7535 in the reference (16.1%) than in the 100% (24.3%) and FPB (50.8%). The FPB–Reference
7536 contrast was not significant 2 years post-harvest, but it was at 5- and 8-years post-harvest as
7537 mortality in FPB increased relative to the reference. The contrast between the 100% and Ref
7538 were not significant for any time interval 8 years post-harvest. The contrasts 100% vs. REF and
7539 FPB vs. 100%—were not significant for any time interval. This may have been because of the
7540 high variability in the data. There was a temporal pattern to mortality in 100% and FPB RMZs.
7541 Annual rates of mortality as percentage of live basal area and density were highest in the first
7542 two years after harvest, then decreased. Wind/physical damage was the primary cause of
7543 mortality. In the 100% treatment it accounted for 78% and 90% of the loss of basal area and
7544 density, respectively; in FPB it accounted for 78% and 65% of the loss. Wind accounted for a
7545 smaller proportion of mortality in reference RMZ (52%). Large wood recruitment to the channel
7546 was greater in the 100% and FPB RMZs than in the reference for each pre- to post-harvest time
7547 interval. Eight years post-harvest mean recruitment of large wood volume was two to nearly
7548 three times greater in 100% and FPB RMZs than in the references. Large wood recruitment rates
7549 were greatest during the first two years, then decreased. However, these differences were not
7550 significant between any treatment comparisons, again, likely due to the high variability in the
7551 data. Mean large wood loading differed significantly between treatments in the magnitude of
7552 change overtime. Results showed a 66% (P < 0.001), 44% (P = 0.05) and 47% (P = 0.01) increase
7553 in mean large wood density in the 100%, FP and 0% treatments, respectively, in the first 2 years
7554 post-harvest compared with the pre-harvest period and after controlling for temporal changes in
7555 the references. Five years post-treatment the mean LW density in the FP continued to increase
7556 42% (P = 0.08), and again 8 years post-treatment (41%; P = 0.09). Results for canopy cover
7557 showed that riparian cover declined after harvest in all buffer treatments reaching a minimum
7558 around 4 years post-harvest. The treatments, ranked from least to most change, were REF, 100%,
7559 FP, and 0% for all metrics and across all years. Effective shade results showed decreases of 11,
7560 36, and 74 percent in the 100%, FP, and 0% treatments, respectively. Significant post-harvest
7561 decreases were noted for all treatments and all years. Results for stream temperature showed that
7562 within treatment mean post–pre-harvest difference in the REF treatment never exceeded 1.0°C.
7563 In contrast, the mean within treatment difference in the 100% treatment was 2.4°C in 2009 (Post-
7564 harvest year 1) but never exceeded 1.0°C in later years. The mean difference in the FP treatment
7565 exceeded 1.0°C immediately after harvest then again in 2014–2016 (post-harvest years 6–9)
7566 while in the 0% treatment the mean difference was 5.3°C initially, then decreased over time to
7567 near, but never below, 0.9°C. Stream temperature increased post-harvest at most locations within
7568 all 12 harvested sites and remained elevated in the FP and 0% treatments over much of the nine
7569 years post-harvest. Temperature responses varied by treatment, by season, and over the years. In

7570 three out of the first four post-harvest years there was, at least, a weak ($r < -0.48$) negative
7571 correlation between July monthly mean temperature response (MMTR) and the change in
7572 riparian cover based on each of the four shade metrics. The correlation was generally weaker
7573 ($-0.4 < r$ and $P > 0.10$) after post-harvest year 4, except for post-harvest year 9 ($-0.6 < r < -0.4$).
7574 However, there were only eight data pairs available for Post 9, compared to ten to twelve for the
7575 other years, which affected the correlation coefficient and p-value. However, there was a great
7576 deal of variability in the correlation coefficient of July MMTR with shade across post-harvest
7577 years among sites and treatments with some sites showing negative correlations and others
7578 positive for some treatments in some years. Considering site characteristics, aspect showed an
7579 influence on stream temperature response. In the first five post-harvest years and in Post 7 the
7580 highest MMTR in each treatment was nearly always the site with a southern (SE or SW) aspect.
7581 No significant correlation between July MMTR and either mean July discharge or the post-
7582 harvest difference in discharge was observed. For the effects of harvest on stream discharge,
7583 cumulative results of regression analysis (forward and reverse regression approaches) indicated
7584 that discharge did increase following harvest. In relative terms, discharge increased by 5-7% on
7585 average in the 100% treatments while increasing between 26-66% in the FP and 0% treatments.
7586 The change in discharge following harvest was also affected by climate, weather, and physical
7587 hydrology of the watershed. In all basins, discharge varied with precipitation, but this was a
7588 complex relationship showing lag time between precipitation events and discharge rate response
7589 in some watersheds. This indicated a potential relationship with physical hydrology at some
7590 watersheds. Results for water turbidity and suspended sediment export (SSE) were stochastic in
7591 nature and the relationships between SSE export and treatment effects were not strong enough to
7592 confidently draw conclusions. Results for harvest effects on total nitrogen export following a
7593 generalized linear mixed effects model, however, showed significant ($P < 0.05$) treatment effects
7594 were present in the FP treatment post-harvest and in the 0% treatment in the post-harvest (2-
7595 years immediately following harvest) and extended periods (2015 – 2017; 7 and 8 years post-
7596 harvest) relative to the reference sites, but there were no significant differences in total-N export
7597 between the treatments. Analysis showed an increase in total-N export of 5.73 ($P = 0.121$), 10.85
7598 ($P = 0.006$), and 15.94 ($P = 0.000$) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments,
7599 respectively, and of 6.20 ($P = 0.095$), 5.34 ($P = 0.147$), and 8.49 ($P = 0.026$) kg/ha/yr in the
7600 extended period. Results for nitrate-N export showed changes similar to but slightly less than
7601 those seen in the total-N analysis with a relative increase in nitrate-N export of 4.79 ($P = 0.123$),
7602 9.63 ($P = 0.004$), and 14.41 ($P < 0.001$) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments,
7603 respectively. None of the changes in the extended period were significant. However, the authors
7604 note that there was high variability in the data for the extended period and nitrate-N export only
7605 returned to pre-harvest levels in one watershed. The increase in total-N and nitrate-N export
7606 tended to be highest during the high flow months in the fall and early winter. The authors
7607 conclude that the 100% treatment was generally the most effective in minimizing changes from
7608 pre-harvest conditions, the FP was intermediate, and the 0% treatment was least effective. The
7609 collective effects of timber harvest were most apparent in the 0% treatment in the two years
7610 immediately post-harvest.

7611

7612 **LW**

7613

7614 Johnston et al., 2011

7615

7616 Johnston, N. T., Bird, S. A., Hogan, D. L., & MacIsaac, E. A. (2011). Mechanisms and source
7617 distances for the input of large woody debris to forested streams in British Columbia, Canada.
7618 *Canadian journal of forest research*, 41(11), 2231-2246. <https://doi.org/10.1139/x11-110>

7619

7620 The purpose of this study was to determine whether the processes and source distances from
7621 which LW entered streams differed among channel types and sizes, to describe LW source
7622 distance curves for a wide range of undisturbed stream and forest types, and to characterize the
7623 relationships between LW input mechanism, source distance, and piece size. Input processes,
7624 source distances, and physical characteristics of approximately 2100 pieces of LW at 51
7625 anthropogenically undisturbed stream reaches throughout south and central British Columbia
7626 were determined. Large wood (LW) was defined in this study as pieces within or suspended
7627 above the active channel, with a minimum length of 1 m. and capable of inducing sediment scour
7628 or deposition. A delivery mechanism was assigned to each LW piece, when it could be
7629 determined, as bank erosion, landslide, windthrow of live trees, stem snap, or standing dead tree
7630 fall. Differences in the frequencies of count data among LW delivery mechanisms, LW positions,
7631 or LWD functions were assessed using chi-square tests. The effects of channel (type, width) and
7632 forest (maximum tree height) characteristics on the proportions of LWD pieces entering the
7633 channel by a given input mechanism were examined using ANCOVA. Channel type for this
7634 study was grouped into 3 categories; riffle-pool (RP), cascade-pool (CP), and step-pool (SP).
7635 Results showed that tree mortality was the most common entry mechanism at all channel types
7636 and width categories and accounted for 65% of all LW pieces sampled. Both channel and
7637 riparian forest characteristics influenced the proportion of LW pieces that entered streams by tree
7638 mortality ($P < 0.05$) but did not vary significantly among channel types ($P = 0.13$). The
7639 proportion of LW pieces recruited by tree mortality decreased with increasing channel width and
7640 with increasing maximum tree height. Bank erosion inputs accounted for 20%–25% of all LW
7641 pieces at the lower-gradient RP and CP sites but were much less important at the SP channels.
7642 Erosion inputs increased with increasing stream size within all channel types ($P = 0.0004$). Wind-
7643 induced inputs (windthrow and stem snap) accounted for 13%–20% of inputs over the channel
7644 types and generally increased in importance in the smaller channels. The proportion of LW
7645 recruited to the stream by stem breakage increased with increasing tree height ($P < 0.0001$) and
7646 varied among channel types ($P = 0.040$), being about twice as prevalent at SP channels as
7647 elsewhere. Landslide inputs of LWD were a minor delivery mechanism. There was considerable
7648 variability in distances from which LW entered the stream. However, based on the cumulative
7649 distributions over sites, 90% of the LW pieces or volume entering the channels originated within
7650 18 m of the stream in 90% of all cases (between 2 and 23 m in all cases). The distances from
7651 which LW entered the streams differed significantly among the various input mechanisms ($P <$

7652 0.001), the rank ordering of the mean source distances being bank erosion < tree mortality <
7653 stem breakage < windthrow < landslides. Bank erosion and landslides delivered the largest LW
7654 pieces and tree mortality and stem breakage the smallest. In general, source distances increased
7655 with increasing tree height, with the effect being stronger in the steeper channel types and
7656 weaker in the wider channels for LW pieces and volume. However, all two-way interactions
7657 among variables were significant implying that the mechanisms through which vegetation and
7658 stream geomorphology influenced LW source distance were complex. Maximum tree height in
7659 the adjacent forest accounted for the greatest variance in in-stream LW source distance for all
7660 models.

7661

7662 **Nutrient**

7663

7664 Deval et al., 2021

7665

7666 Deval, C., Brooks, E. S., Gravelle, J. A., Link, T. E., Dobre, M., & Elliot, W. J. (2021). Long-
7667 term response in nutrient load from commercial forest management operations in a mountainous
7668 watershed. *Forest Ecology and Management*, 494, 119312.
7669 <https://doi.org/10.1016/j.geomorph.2013.11.028>

7670

7671 The purpose of this study was to quantify and compare the differences in nitrogen and
7672 phosphorus concentrations and loads between pre-disturbance, post road construction (post-
7673 road), post experimental harvest (PH-I), and post operational harvest (PH-II) from both a
7674 hydrological yield and nutrient concentration perspective. This study was carried out in the Mica
7675 Creek Experimental Watershed in Northern Idaho. For this analysis time periods have been
7676 broken into four distinct phases: Pre-disturbance (1992–1997), Post-road (1997–2001),
7677 experimental-harvest Phase I (PH-I) (2001–2007), and operational sequential harvest Phase II
7678 (PH-II) when the extent and frequency of harvests increased (2007–2016). PH-I represents an
7679 experimental treatment phase during which harvest activities were experimentally controlled
7680 (only upstream headwater watersheds were harvested and mature vegetation removal ranged
7681 between 24% and 47%) followed by site management operations including broadcast burning
7682 and replanting. PH-II represents the post-experimental phase where the study area transitioned to
7683 operational treatments that consisted of additional road construction and timber harvest, with site
7684 management operations including pile burning and competition release herbicide application.
7685 During this operational phase, the mature vegetation removal in the upstream and cumulative
7686 downstream watersheds ranged between 36% and 50% and 17–28%, respectively. Monthly
7687 annual grab samples of stream water were collected from seven flumes over the course of 25
7688 years (from pre- to post-treatments). The samples were analyzed for six parameters, specifically
7689 nitrate + nitrite (NO₃ + NO₂), total Kjeldhal nitrogen (TKN), total ammonia nitrogen (TAN)
7690 containing un-ionized (NH₃) and ionized (NH₄⁺) ammonia, total nitrogen (TN), total

7691 phosphorus (TP), and orthophosphate (OP). This study used a before-after, control-impact paired
7692 series design (BACIPS) to evaluate direct and cumulative effects of forest management practices
7693 on stream nutrient concentrations in paired and nested watersheds. Results for long-term trends
7694 in stream flow showed a statistically significant increasing trend in all the watersheds during the
7695 fall and winter seasons. Significant increases in summer streamflow only occurred in the control
7696 watersheds. There were minimal changes in TKN concentration with a slight observed reduction
7697 in long-term TKN loads. Overall, the cumulative mean TAN loads from all watersheds did not
7698 show large variations with sequential varying treatments over time. In contrast to TAN, there was
7699 a significant response in NO₃ + NO₂ following timber harvest. The response in NO₃ + NO₂
7700 concentrations was negligible at all treatment sites following the road construction activities.
7701 However, NO₃ + NO₂ concentrations during the PH-I period increased significantly ($p < 0.001$)
7702 at all treatment sites. Similar to the PH-I period, all watersheds experienced significant increases
7703 in NO₃ + NO₂ concentration during the PH-II treatment period. Overall, the cumulative mean
7704 NO₃ + NO₂ load from all watersheds followed an increasing trend with initial signs of recovery
7705 in one treatment watershed after 2014. Mean monthly TP concentrations showed no significant
7706 changes in the concentrations during the post-road and PH-I treatment periods. However, a
7707 statistically significant increase in TP concentrations ($p < 0.001$) occurred at all sites, including
7708 the downstream cumulative sites, during PH-II. Generally, OP concentrations throughout the
7709 study remained near the minimum detectable concentrations. A statistically significant increase
7710 in mean monthly OP concentrations occurred only at the cumulative downstream treatment site
7711 during both Post-road (p -value = 0.021) and PH-I (p -value < 0.001) treatment periods,
7712 respectively. The largest cumulative increase in mean annual loads was largely attributed to
7713 increased flow. The authors conclude that only relatively small increases in nutrient loads were
7714 detected suggesting that Idaho Forest Practices Act regulations and BMPs are effective in
7715 minimizing the delivery of particulate-bound pollutants. Forest management activities increased
7716 stream NO₃ + NO₂ concentrations and loads following timber harvest activities, but these effects
7717 were also attenuated in downstream reaches and reduced through time as vegetation regrowth
7718 occurred.

7719 Quinn, T., G.F. Wilhere, and K.L. Krueger, technical editors. 2020. Riparian Ecosystems, Volume
7720 1: Science Synthesis and Management Implications. Habitat Program, Washington Department
7721 of Fish and Wildlife, Olympia.
7722 This publication is a synthesis of scientific literature concerning riparian areas (function, process,
7723 characteristics, etc.) for the purpose of informing management and the development of policies
7724 related to management of riparian areas and watersheds of Washington State. The most relevant
7725 information in the publication to this review are in chapters 3 (large wood), 4 (stream
7726 temperature), and 6 (Nutrient dynamics in Riparian ecosystems).
7727 The main conclusions from chapter 3 (large wood) state that the successful conservation, or
7728 restoration, of fish habitats in forested areas requires management practices that deliver adequate
7729 wood into aquatic systems. They purpose the main scientific uncertainties, from a management
7730 perspective, is (1) the shape of the wood recruitment curves under different watershed and site-
7731

Formatted: Font: (Default) Times New Roman, 12 pt

7732 level conditions. These curves describe the function of wood input into streams from greater
7733 distances from the stream (source distance curves) based on stand structure (e.g., young-old, tree
7734 height variability, density metrics, etc.), species compositions (especially conifer vs. hardwood),
7735 and site conditions (e.g., slope, moisture availability, soils, site index). The second uncertainty is
7736 the effects of the potential wood delivery mechanisms that occur outside of the riparian area
7737 (e.g., landslides, debris flows). They posit that management objectives for large wood
7738 recruitment potential should aim to restore site composition and structure that is similar to
7739 unmanaged riparian forests. The authors suggest that much is known about large wood
7740 recruitment potential from within the riparian forests based on site potential and stand structure.
7741 Previous work and the development of source distance curve equations show the range of
7742 “effective” tree heights (trees with the bulk of stem > 10 cm, functionally classified as large
7743 wood) is between 85 and 230 feet. This means 100% of a sites wood recruitment potential is
7744 within 85 -230 feet of the stream. However, these equations do not account for the presence of
7745 smaller trees or the potential of tree recruitment from outside of the riparian area, or from
7746 extreme channel migration events.

7747 The main conclusions from chapter 4 identify that the science surrounding stream thermal
7748 regimes is uneven. Scientists are certain that stream temperatures and thermal regimes are
7749 important to aquatic species, and thus it is important for management practices to restore and
7750 conserve these conditions. However, while the general conclusions of most studies show that
7751 land use changes (urbanization, and agriculture) and forest management within riparian areas
7752 leads to warmer stream temperatures, the spatial and temporal effects of any specific riparian
7753 management action remain uncertain. Recovery rates for stream temperature post-treatment vary
7754 greatly based on site location because of differences in stream size (width and depth), and
7755 physiography (climate, physical geography). Shade from the adjacent riparian area is widely
7756 accepted as the most important, and most directly manageable, factor affecting stream
7757 temperature. However, because of the variability in other factors affecting stream temperature,
7758 predicting changes in stream temperature from shade removal will likely always suffer from
7759 imprecision.

7760 The main conclusions from chapter 6 (Nutrient dynamics in Riparian ecosystems) list land use,
7761 forest age and composition, Climate and seasonality, elevation and topography, hydrology,
7762 nutrient concentrations, forms and inputs, soil properties and geology, and biota as the major
7763 factors influencing nutrient dynamics in riparian ecosystems. Riparian areas that are structurally
7764 diverse in physiography and soil are most likely to support diverse biota (vegetation, animals,
7765 and microbial communities. More diverse riparian communities are considered best in processing
7766 and assimilating nutrient loads. The authors identify headwater streams as important zones for
7767 active nutrient processing because they affect downstream nutrient loads. Also, maintaining the
7768 connection between the aquatic and terrestrial environments via floodplain conservation and
7769 restoration is important. While there is still a lot of uncertainty involved in the mechanisms
7770 responsible for nutrient transport through the system, it is clear that riparian areas are vital for the
7771 not only providing nutrients to stream, but also in filtering, processing, and storing nutrients in
7772 the short and long term. The results of most studies indicate that the storage and filtering of
7773 nitrogen is most effective in areas with wide vegetated buffers compared to narrower buffers or

7774 unvegetated riparian areas regardless of the type of vegetation present. However, the type of
7775 vegetation directly impacts the quality and quantity of nutrients available. Deciduous trees
7776 generally provide more litter with higher nutrient content. Coniferous trees, on the other hand,
7777 life longer and provide more shade and large wood input potential. Thus, the authors conclude
7778 that riparian management should consider both the structural and food web roles of each species
7779 present in a forested riparian area.

7780 ▲

Formatted: Font: (Default) Times New Roman, 12 pt