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Riparian Function Literature Synthesis

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

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57 Background

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

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

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

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

91

92 Focal Questions

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

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

121 Methods

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

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

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

149 Table 1. List of terms used in search of keywords and titles of literature sourced from Web of
 150 Science. Terms in **bold** were used in all searches. Terms were grouped by topic (e.g.,
 151 management, physiography, disturbance, etc.). Results show the number of publications returned
 152 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
(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
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154 From the initial title and topic review of the 16,516 papers sourced in our search, we refined the
 155 list to 528 papers for consideration based on the 3 criteria listed above (e.g., utilize experimental
 156 design with results focusing on at least one of the five key functions; published after 1999; were
 157 conducted in western North America). From these 528 papers we further refined our list to 105
 158 articles based on information gleaned from the abstract, introduction and methods sections
 159 regarding study design and relevant geography. Of these 105 articles 91 provided information on
 160 at least one of the five key functions and were thoroughly read and used to develop an annotated
 161 bibliography (Appendix). The other 14 articles provided information and experimental results
 162 about fire frequency and fire behavior in riparian areas, or effects of fire on one of the five key
 163 functions. These 14 papers about fire were not included in the literature review but were
 164 reviewed and discussed in focal questions 3 and 7. Frequency of the top 8 keywords were
 165 represented in a histogram to express the popularity of topics in the literature since the year 2000
 166 (Figure 1). We organized our review of the relevant literature by (1) FPHCP objective and (2)
 167 focal question. A table was submitted along with this report that gives a more thorough
 168 description of details used to categorize publications in supplemental materials (supplemental
 169 table of references; S1).

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

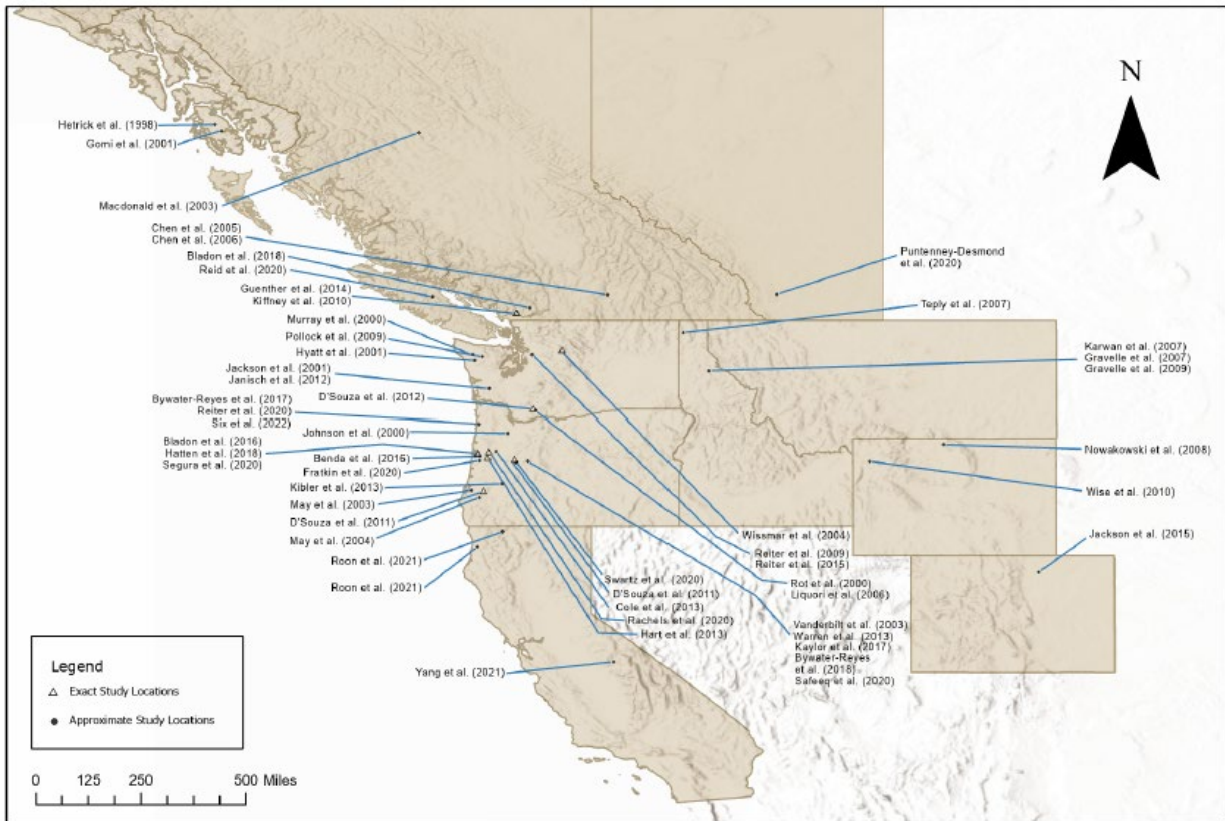
Keywords	Count
Water quality	1165
Streams	1004
Watershed	1000
Climate change	848
Watershed management	729
Riparian	652
Stream	604
Nitrogen	489

171

172 Results/Summary of Review

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

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



196
 197 Figure 2. Locations where studies were conducted. References not listed include studies that
 198 sourced data from multiple locations.

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

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

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

Guenther et al. (2014)	Differences in surface/sub-surface variability as well as influences of partial retention harvesting on stream temperature.	2 years	3	SHD	BACI	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, correlative 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

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

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

Martin & Grotfendt (2007)	Compared site conditions between riparian buffer strips and unlogged riparian stands using aerial photography to determine mortality and LWD recruitment	1 year data collection	9	LWD	ACI	Regional, northern and southern portions of southeast Alaska	Alaska
May & Gresswell (2003)	Investigates the mechanisms responsible for LWD recruitment into streams.	2-year data collection	4	LWD, SED	modeling, Regression analysis	Local, North Fork of Cherry Creek Research Natural Area	Oregon
Meleason et al. (2003)	Evaluate of the potential effects of different riparian mgmt. strategies on the standing stock of wood.	Simulation modeling of 720 years	1	LWD	Modeling	simulation of stream types common in PNW	PNW, hypothetical stream
Mueller & Pitlick (2013)	Examines the relative importance of lithology as a driver of sediment delivery into streams.	multiple datasets ranging 5-90 years	83	SED	spatial modeling, correlative analysis of historical data	Regional, Northern Rocky Mountains	ID, WY, MT
Murray et al. (2000)	Examined the influence of partial harvesting on stream temperature, chemistry, and turbidity.	2 years data collection, 10-15 years after treatment	1	NUT, SED, SHD	ACI	Local, Rock and Tower Creek watersheds	Washington

Nowakowski & Wohl (2008)	Examined differences in wood load and valley/channel characteristics between managed and unmanaged riparian areas.	1 year 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

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

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

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

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

203 Discussion of findings relative to FPHCP objectives

204 Litter/Organic matter inputs/Nutrients

205 Prior to the Forest and Fish Report (1999), studies that directly quantify the effects of timber
206 harvest within riparian areas on litter and organic matter (OM) input into streams in managed
207 watersheds of western north America are sparse. Two seminal studies, one from the H.J.
208 Andrews experimental watershed studies (Gregory et al., 1987) and one from the Carnation
209 Creek experimental watershed (Hartman & Scrivener, 1990) present results that estimate loss of
210 litter input following harvest. Gregory et al., (1987) which was part of the Streamside
211 Management: Forestry and Fishery Management collection produced by Salo & Cundy (1987)
212 noted that removal of the forest canopy from timber harvesting resulted in decreases in annual
213 litter fall from 300-400 g/m² in the mature forests to less than 100 g/m². Further, they posit that
214 decreased litter inputs after logging can persist for 10 – 20 years before recovering. Results from
215 Hartman & Scrivener, (1990) showed that litter inputs post-logging were 25-50% of pre-logging
216 levels with about 50% of the loss recovering within a decade (note: buffer widths varied from 1-
217 70 m, litter input loss was not analyzed by buffer width).

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

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

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

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

264 In terms of the effects of timber harvest on litter and OM quantity in streams, 4 studies in our
265 review were found that provide experimental results that have been conducted since 2000 and
266 focus on western North America. Of these 4 studies, 1 used simulation modeling (Yeung et al.,
267 2019), and the other 3 (Kiffney & Richardson, 2010; McIntyre et al., 2018; Six et al., 2022) used
268 field-based experiments to estimate the effects of timber harvest within riparian forests on OM
269 inputs and dynamics in streams. Yeung et al. (2019) simulated post-harvest responses to leaf-
270 litter derived coarse particulate organic matter (CPOM) quantity in a coastal rainforest stream in
271 British Columbia, Canada. For this study, Yeung et al. (2019) used published empirical data from
272 representative small, forested streams in coastal British Columbia to calibrate and set parameters
273 for their CPOM model. The model compared the effects litterfall reduction, increase in peak
274 flows, and increase in stream temperature (estimated for 4 harvesting intensities based on
275 available data) on in stream CPOM standing stocks. Results showed evidence that litterfall
276 reductions from timber harvest was the strongest control on in-stream CPOM quantity for 4 years
277 post-harvest. However, when litterfall reductions were below 30%, the effect size varied with
278 relative changes to peak flows and stream temperature. Stream temperature increases specifically
279 showed a significant interaction with litterfall reductions. The authors propose that the decreased
280 activity of CPOM consumers caused by increasing stream temperatures by $4 \text{ }^\circ\text{C}$ or more, may be
281 enough to offset the loss of litterfall inputs of CPOM stocks. This speculation was made based on
282 the temperature dependent function of leaf litter consumption by common shredder species and
283 temperature ranges modeled by Stenroth et al. (2014). This model predicts shredder activity is
284 optimized at $\sim 15 \text{ }^\circ\text{C}$ (ranging between $13.7 - 16.7^\circ\text{C}$) but begins to quickly decline at

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

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

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

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

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

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

341 *Nutrients*

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

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

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

377 In our search of the literature, four studies were found that provide experimental evidence of the
378 effects of riparian timber harvest on nutrient flux in western north America and were published
379 since 2000. Gravelle et al., 2009 compared the effects of contemporary forest harvesting
380 practices in Idaho on nutrient cycling and in stream concentrations. This study followed the
381 BACI design and featured a pre-treatment measurement phase (5 years), a post-road construction
382 phase (5 years), and a post-harvest phase (5 years). Treatments imposed included a clearcut to
383 stream with 30-foot equipment exclusion zone (non-fish-bearing), a target reduction of 50% of
384 the canopy removal over 50% of the area, equating to 25% removal of existing shade (fish-
385 bearing streams), and was compared to an uncut reference. Results for the post-road construction
386 period showed no significant changes in any analyzed nutrient concentrations. Results for the
387 post-harvest period showed significant increases in monthly mean nitrate and nitrite (NO^3 and
388 NO^2) at sites immediately downstream from the clearcut, the partial harvest, and at sites
389 downstream from both treatments in the stream network (cumulative). The changes in monthly
390 mean NO_3 and NO^2 during the five years post-harvest were greatest for the clearcut treatment
391 ($+0.29 \text{ mg L}^{-1}$), followed by the cumulative ($+0.07$ and $+0.05 \text{ mg L}^{-1}$) and partial harvest ($+0.03$
392 mg L^{-1}). NO^3 showed progressively increasing monthly concentrations for 3 years after harvest
393 before declining. None of the other nutrients analyzed in this study (Kjeldahl nitrogen (TKN),
394 total phosphorus (TP), total ammonia nitrogen (TAN) consisting of un-ionized (NH^3) and ionized
395 (NH^{4+}) ammonia, and unfiltered orthophosphate (OP) samples) showed significant changes
396 during the post-harvest period.

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

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

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

416 Considering the interaction between climate and forest harvest on nutrient transport, Yang et al.
417 (2021) investigated the effects of drought and forest thinning operations (independently and
418 combined) on stream and soil water chemistry in the Mediterranean climate headwater basins of
419 the Sierra National Forest. Data on water chemistry were taken 2 years prior and 3 years
420 following drought and thinning operations in two watersheds, each with thinned and control
421 stands. Young stands with high shrub cover (> 50%) were masticated to < 10% shrub cover. The
422 thinning prescription in mature stands removed trees across all diameter classes to a target basal
423 area range of 27–55 m² ha⁻¹ with target basal areas varying based on tree density. Thinning
424 extended into the riparian management zone. Trees within 15 m of the stream could be chainsaw-
425 felled and skidded, but mechanical equipment was excluded within 30 m of the stream. Results
426 showed that drought alone altered dissolved organic carbon (DOC) in stream water, as well as
427 altered the proportion of dissolved organic carbon to nitrogen (DOC: DON) in soil solution in
428 unthinned (control) watersheds. Volume-weighted concentration of DOC was 62% lower ($p <$
429 0.01) and DOC:DON was 82% lower ($p = 0.004$) in stream water and soil solution, respectively,
430 during years of drought than in years prior to drought. Drought combined with thinning altered
431 DOC and dissolved inorganic nitrogen (DIN) in stream water, and DON and total dissolved
432 nitrogen (TDN) in soil solution. For stream water, volume-weighted concentrations of DOC were
433 66- 94% higher in thinned watersheds than in control watersheds for all three consecutive
434 drought years following thinning. No differences in DOC concentrations were found between
435 thinned and control watersheds before thinning. The authors conclude that their results provide
436 evidence that the influences of drought and thinning are more pronounced for DOC than for
437 nitrogen in streams. They also speculate that the periodic changes in climate (e.g., seasonal,
438 drought) contribute to the high variability in carbon and nitrogen concentration in streams in
439 Mediterranean climates following harvest.

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

449 yr-1), total-N and nitrate-N export increased post-harvest at all sites, with the smallest increase in
450 the 100% treatment and the largest in the 0% treatment. Compared to the reference sites, analysis
451 showed an increase in total-N export of 5.52 (P = 0.051), 11.52 (P = 0.0007), and 17.16 (P
452 <0.0001) kg ha-1 yr-1 in the 100%, FP, and 0% treatments, respectively, in the first 2 years post-
453 harvest. In the extended period (7-8 years post-harvest) export for total-N remained higher in all
454 treatments compared to the reference by 6.20 (P = 0.095), 5.34 (P = 0.147), and 8.49 (P = 0.026)
455 kg ha-1 yr-1 for the 100%, FP, and 0% treatments, respectively. Nitrate-N showed the same
456 pattern with slightly lower values than total-N. The increase in total-N and nitrate-N export from
457 the treatment watersheds post-harvest was strongly correlated with the increase in annual runoff
458 (R² = 0.970 and 0.971; P = 0.001 and 0.001) and with the proportion of the basin harvested (R²
459 = 0.854 and 0.852; P = 0.031 and 0.031). The authors note that there was high variability in the
460 data for the extended period and nitrate-N export only returned to pre-harvest levels in one
461 watershed. Total-P export increased post-harvest by a similar magnitude in all treatments: 0.10 (P
462 = 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
463 (only analyzed during the 2-year post-harvest period). The authors conclude that the 100%
464 treatment was generally the most effective in minimizing changes from pre-harvest conditions,
465 the FP was intermediate, and the 0% treatment was least effective. Thus, similar to the results of
466 other studies reviewed, these results provide evidence that the effects of timber harvest on
467 nutrient export is proportional to the intensity of the treatment (e.g. percent of basin harvested,
468 presence of protective buffer).

469 *Summary of Factors Impacting Nutrient Concentrations and Export*

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

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

489

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

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

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

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

535 In the western United States, several notable studies since 2000 have continued to investigate
536 and refine the factors important for LW recruitment. For example, Wing & Skaugset (2002)
537 investigated the relationships between land use, land ownership, and channel and habitat
538 characteristics with LW quantity and volume in stream reaches in western Oregon. The relevant
539 results (those derived for forested streams only) showed that stream gradient was the most
540 important explanatory variable for in-stream LW volume with the split in the regression analysis
541 occurring at 4.7%. Stream reaches with gradients less than 4.7% had on average less than half
542 the in-stream LW volume (11.3 m³ vs. 25.2 m³ per reach) than reaches with gradients >4.7%.
543 Results for LW pieces (logs at least 0.15 m diameter, and 3 m long) per 100 m length showed
544 bankfull width (BFW) as the most important explanatory variable with a split in the regression
545 analysis occurring at 12.2 m BFW. Reaches with a BFW <12.2 m averaged 11.1 LW pieces per
546 100 m compared to wider streams which averaged 4.9 pieces per 100 m. When the analysis was
547 constrained to “key” LW pieces (logs at least 0.6 m diameter and 10 m long), stream gradient
548 again emerged as the most important explanatory variable with the split in the regression
549 occurring at 4.9% stream gradient (mean key pieces per 100 m were 0.5 and 0.9 for gradients <,
550 and >4.9%, respectively). Following stream gradient and BFW, lithology was also an important
551 explanatory variable showing splits for Mesozoic and sedimentary lithologies (in 3 out of 4
552 analyses) grouped as containing half the LW quantity (pieces, key pieces, volume) on average
553 than all other geologies (basalt, cascade, and marine sedimentary geologies). Wing & Skaugset
554 (2002) suggests that geomorphic characteristics, in particular stream gradient and bankfull width,
555 but also underlying lithology in forested areas correlate best with LW presence in headwater
556 streams of western Oregon.

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

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

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

590
591 Fox & Bolton (2007) modeled LW values from 150 stream segments located in unmanaged
592 watersheds, across Washington, with landscape, reach, and stand characteristics to understand the
593 central tendency of instream LW values in “natural” fish-bearing streams. Outputs from their
594 models show evidence that in-stream wood volume (m³ per 100 m stream length) and LW piece
595 count for streams up to 20 m in bankfull width (BFW) increased with drainage area and as
596 streams became less confined with BFW being a significantly better predictor of wood
597 parameters than basin size. Also, in-stream wood volume increased with adjacent riparian timber
598 age as determined by the last stand replacing fire. In this study (Fox & Bolton, 2007), the authors
599 noted that other predictor variables (e.g., gradient, bedform) also showed some evidence of an
600 effect but the variability of these variables were too great to evaluate with confidence.

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

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

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

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

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

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

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

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

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

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

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

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

736 The Hard Rock and Soft Rock studies showed similar results. Both studies showed an increase in
737 stand mortality that also led to an increase in LW recruitment into the channels adjacent to 50-
738 foot (and greater in the Soft Rock) buffer treatments relative to unharvested reference sites.
739 However, the longer time period of study in the Hard Rock study showed mortality and thus LW
740 recruitment began to stabilize after year five. The results presented by Schuett-Hames (2012,
741 2019b) showed a similar pattern of an initial increase in mortality rates and LW recruitment rates
742 in treated stands relative to untreated stands within three years of treatment, but stabilization

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

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

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

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

786

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

798 Burton et al. (2016) examined the relationship between annual in-stream wood loading and
799 riparian buffer widths adjacent to upland thinning operations. No-cut buffer widths were 6, 15, or
800 70 meters, and upland thinning was to 200 trees per ha (tph), with a second thinning (~10 years
801 later) to ~85 tph, alongside an unthinned reference stand ~400 tph. Their results showed that
802 slightly higher volumes of wood were found in sites with a narrow 6-m buffer, as compared with
803 the 15-m and 70-m buffer sites in the first 5 years after the first harvest and maintained through
804 year 1 of the second harvest (end of study). The authors attributed this difference to a higher
805 likelihood of logging debris and/or windthrow, but these factors were not analyzed. Considering
806 source distance, the authors used a mixed modeling approach to assess the relationship between
807 wood volume and source distance for in-stream wood with an identifiable source. This model
808 was only applied to the 70-meter buffer. The results showed that 82-85% of the wood with
809 discernable sources (90% for wood in early stages of decay; 45% of wood in late stages of
810 decay) came from within 15 m of the stream, and the relative contribution of wood to streams
811 declined rapidly with increasing distance. Still, these results are similar to those presented by
812 Schuett-Hames & Stewart (2019a) which showed the majority of the LW recruited (72-76% for
813 treated stands) into the channel were from within the first 30 feet (~9.1 m) of the stream even
814 though upland harvest prescriptions in this study differed from those evaluated by Burton et al.
815 (2016) (e.g., clearcut vs thinning).

816

817 *Summary of Factors Impacting LW Loads and Recruitment*

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

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

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

849

850

851

852 Bank Stability and Sediment

853 *Bank Stability*

854 Few studies could be found that quantify how riparian area harvest directly affects bank stability
855 or bank erosion based on our search criteria. Many studies published since 1999 that investigate
856 bank stability and bank erosion compare relative rates of erosion based on the presence/absence
857 of vegetation, type of vegetation (e.g., grassland vs. forest cover), and soil types or lithology
858 (Konsoer et al., 2015; Micheli et al., 2004; Simon & Collision, 2001; Wynn & Mostaghimi,
859 2006). Also, many studies have investigated the relative effects of different types of land use
860 (e.g., agricultural, urban, forested) as well as cattle grazing intensity (McInnis & McIver, 2009;
861 Zaimes & Schultz, 2014). The only studies that could be found that provide some experimental
862 evidence as to how timber harvest within the riparian area affects bank stability or erosion come
863 from 3 CMER reports (Ehinger et al. 2021; McIntyre et al. 2018, Schuett-Hames et al., 2011;
864 Schuett-Hames & Stewart, 2019).

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

886 Results showed that the differences between the mean values of harvest related soil and
887 streambank disturbances for clear-cut patches and the 50-ft buffers were significant for all
888 metrics (e.g., # of bank disturbance features per 100 ft, # of soil disturbance features per 100 feet,
889 # of soil disturbance features, # of soil disturbance features delivering sediment to stream, % of
890 ELZ with soil disturbance; $P \leq 0.082$). Results for soil disturbance performance targets showed
891 that all of the 50-foot buffer and PIP prescriptions met the performance targets (i.e., maintained
892 <10% harvest-related soil disturbance in the ELZ). One clearcut patch exceeded the 10%

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

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

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

935 pattern was similar for root-pits with sediment delivery, however the difference between the
936 reference and buffer patches was less pronounced due to the higher percentage of root-pits
937 delivering sediment in the reference patches. The percentage of root-pits with evidence of
938 sediment delivery was much higher in the clear-cut patches than in the 50-ft buffers (20.1%) and
939 the reference (26.0%) patches but was not significantly different. Results for the PIP buffers
940 showed a similar trend as the 50-foot buffers with an increase in root pits delivering sediment to
941 the stream in the first three years, but a sharp decline after the third year. Over the course of the
942 full five years Over the entire 5 year period, the percentage of root-pits with evidence of
943 sediment delivery in the PIP buffers (17.6%) was similar to the percentage for the 50-ft buffers
944 (19.8%). These values did not differ significantly from the references.

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

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

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

976 and 1 clearcut sites. At these five sites, post-harvest surface erosion was evident adjacent to only
977 1.5 to 4.6% (average = 2.2%) of the total stream channel length (including both mainstem and
978 tributaries). At the remaining study sites where stream-delivering erosion events occurred, the
979 total eroded area was 60 m² or less and occurred adjacent to 0.3% to 0.8% (average = 0.6%) of
980 the stream channel length. There were no statistically significant differences in stream-delivering
981 surface erosion among treatments ($\alpha = 0.05$), and on average, reference and buffer treatments
982 visually exhibited a similar amount of exposed bank.

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

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

1000 Sediment

1001 The function of riparian areas to regulate and filter the flow of sediments into streams is essential
1002 not only for water clarity and pool formation but also because of the ability of sediments to carry
1003 nutrients and pollutants (Cooper et al., 1987; Hoffman et al., 2009; Polyakov et al., 2005). .
1004 Sediment flux into streams can be affected by landscape factors, streamflow, vegetation
1005 composition, and disturbance including riparian and adjacent upland forest management
1006 (Crandall et al., 2021; Devotta et al., 2021; Vanderbilt et al., 2003). The movement of sediment
1007 into the active channel can, in turn, impact aquatic habitat and geomorphic processes, especially
1008 in small, forested streams (Benda et al. 2005; Gomi et al., 2005; Hassan et al., 2005).

1009 The effects of riparian area timber harvest on sediment flux into streams has been documented,
1010 investigated, and incorporated into riparian forest management plans in western North America
1011 since the 1970s with the development of the Clean Water Act of 1972 (Bilby et al., 1989;
1012 Gregory 1990; Gresswell et al., 1989; Naiman et al., 1998; Salo & Cundy, 1986; Swanson et al.,
1013 1982; Swanson & Dyrness, 1975). Prior to the Forests and Fish Report (FFR 1999), several
1014 studies from western North America investigated the effects of riparian zone timber harvest
1015 practices on sediment flux into streams.

1016 Specific to Washington, Rashin et al. (2006) evaluated the effectiveness of Washington State best
1017 management practices (BMPs) for controlling sediment related water quality impacts. Although
1018 this study was published in 2006, the data analyzed in this study were collected between 1992
1019 and 1995. In their evaluation, Rashin et al. (2006) assessed site erosion, sediment delivery,
1020 channel disturbance, and aquatic habitat condition within the first two years of harvest along
1021 fish- and non-fish bearing streams across Washington state. From their results, the authors
1022 concluded that the site-specific factors influencing the effectiveness of BMPs in preventing
1023 chronic sediment delivery into streams were 1) the proximity of ground disturbance to the
1024 stream, 2) presence of a stream buffer, 3) falling and yarding practices that minimized
1025 disturbance to stream channel, and 4) timing of harvest activities for certain climate zones where
1026 frozen ground or snow cover may be exploited. The landscape factors that influenced BMP
1027 effectiveness were 1) the density (specific metric not reported) of unbuffered small streams at
1028 harvest sites, and 2) steepness of stream valley slopes. The authors conclude with a
1029 recommendation of excluding timber falling and yarding activities at least 10 m from streams
1030 and outside of steep inner gorges.

1031 Similar results were reported by Lewis (1998) in their evaluation of logging activities' effect on
1032 erosion and suspended sediment transport in the Caspar Creek Watersheds of northwestern
1033 California. From their results the authors concluded that the dominant factors influencing the
1034 difference in suspended sediment loads between watersheds was the difference in road
1035 alignment, yarding methods, and presence of stream protection zones (i.e., buffers). Because of
1036 studies like these reviewed, contemporary riparian forest management practices in the western
1037 United States include rules that limit harvesting, use of equipment, and procedures that disturb
1038 soil in areas closest to the stream or on steep and unstable slopes ([WAC 222-30-022](#); WAC 22-
1039 30-021; 2022 [ODF](#); IDAPA 20.[02.01](#))

1040 Since 2000, many of the studies published that evaluate changes in sediment delivery or water
1041 turbidity following riparian timber harvest show similar results in that contemporary BMPs are
1042 effective in mitigating increases in sediment delivery to streams (Hatten et al., 2018; Reiter et al.,
1043 2009). For example, the studies reviewed that report a significant change in sediment delivery
1044 following harvest show evidence that these changes only persist for a short period of time (1-3
1045 years) and that the magnitude of these changes are related to the intensity of the harvest
1046 prescriptions (Karwan et al., 2007; Macdonald et al., 2003a).

1047
1048 For example, Macdonald et al. (2003a) compared changes in stream discharge rates and in-
1049 stream suspended sediment concentrations during spring snowmelt between two harvest
1050 intensities and one unharvested control, for pre- and post-harvest in first order streams of interior
1051 British Columbia. Both treated riparian areas received a harvest of 55% of the watershed; one
1052 (low-retention) removed all merchantable timber >15 cm DBH for pine and > 20 cm DBH for
1053 spruce within 20 m of the stream; the other (high-retention) removed all merchantable timber >
1054 30 cm within 20 m of the stream. The results showed an increase in spring snowmelt discharge
1055 for both treatments above predicted values for the study (5 years). However, increased in-stream
1056 total suspended sediments (TSS) only persisted for two-years post-harvest in the high-retention
1057 treatment, and for 3-years in the low-retention.

1058
1059 Karwan et al. (2007) investigated the effects of riparian timber harvest and road construction on
1060 TSS concentrations in the Mica Creek Experimental Watershed in northern Idaho. Treatments in
1061 the paired-watershed experiment consisted of 1) commercial clearcut of the watershed area by
1062 50%, and was broadcast burned and replanted, 2) partial cut in which half the canopy was
1063 removed in 50% of the watershed area 3) a no-harvest control. All harvests were done according
1064 to best management practices and the Idaho Forest Practices Act. This included equipment
1065 exclusion zones of 50- and 30-feet for fish- and non-fish-bearing streams, respectively. On all
1066 skid trails, drainage features, such as water bars, were installed for erosion control at the end of
1067 the harvest period. Results showed that road construction in both watersheds did not result in
1068 significant impacts on monthly sediment loads in either treated watershed during the immediate
1069 (1-year post-harvest) or recovery (2-4 years post-harvest) time intervals. A significant and
1070 immediate impact of harvest on monthly sediment loads in the clear-cut watershed ($p = 0.00011$),
1071 and a marginally significant impact of harvest on monthly sediment loads in the partial cut ($p =$
1072 0.081) were observed. However, after one year, the TSS loads in both treatments became
1073 statistically indistinguishable from the control.

1074
1075 Specific to Washington, McIntyre et al. (2021) evaluated the effectiveness of riparian buffers on
1076 non-fish-bearing streams underlain by competent lithologies (“Hard Rock”) in western
1077 Washington. Buffers were treated with one of three prescriptions 1) unharvested reference, 2) a
1078 two-sided 50-ft riparian buffer along the entire riparian management zone (RMZ), 3) a two-sided
1079 50-ft riparian buffer along at least 50% of the RMZ, and 4) clearcut to stream edge (no-buffer).
1080 Results for suspended sediment export (SSE) following treatment showed episodic increases
1081 with storm events that rapidly declined. However, changes in SSE were poorly correlated with
1082 discharge and exhibited high variation between treatment sites. The authors suggest that these
1083 results show evidence that changes in SSE magnitudes were not related to harvest. Further, they
1084 conclude that the sites were likely sediment-limited considering the underlying lithology.

1085
1086 Site factors such as underlying lithology and physiography can interact with the effect of timber
1087 harvest operations on sediment delivery into streams. Bywater-Reyes et al. (2017) assessed the
1088 influence of natural controls (basin lithology and physiography) and forest management on
1089 suspended sediment yields in temperate headwater catchments in northeastern Oregon. Results
1090 from this study indicate that site lithology was the first order control over suspended sediment
1091 yield (SSY) with SSY varying by an order of magnitude across lithologies observed.
1092 Specifically, SSY was greater in catchments underlain by Siletz Volcanics ($r = 0.6$), the Trask
1093 River Formation ($r = 0.4$), and landslide deposits ($r = 0.9$) and displayed an exponential
1094 relationship when plotted against the percentage of watershed area underlain by these lithologies.
1095 In contrast, lithology had a strong negative correlation with percent area underlain by diabase (r
1096 $= 0.7$), with the lowest SSY associated with 100% diabase. Following timber harvest, increases
1097 in SSY occurred in all harvested catchments but returned to pre-harvest levels within 1 year
1098 except for sites that were underlain by sedimentary formations and were clearcut without
1099 protective buffers. The authors conclude that sites underlain with a friable lithology (e.g.,

1100 sedimentary formations) had, on average, SSYs an order of magnitude higher following harvest
1101 than those on more resistant lithologies (intrusive rocks).

1102 Mueller & Pitlick, (2013) found similar results in their assessment of the relative effect of
1103 lithology, basin relief, mean basin slope, and drainage density on in stream sediment supply for
1104 83 drainage basins in Idaho and Wyoming. The strongest correlation of in stream sediment
1105 supply was with lithology relative softness (based on grouping of rock types – granitic,
1106 metasedimentary, volcanic, and sedimentary). Sediment concentrations at bankfull width
1107 increased by as much as 100-fold as basin lithology became dominated by softer sedimentary
1108 and volcanic rock compared to lithologies dominated by harder granitic and metasedimentary
1109 rock. Finally, Wissmar et al. (2004), developed and field-tested erosion risk indices for
1110 watersheds in western Washington based on land cover. These erosion risk indices used the
1111 presence of unstable soils (determined by geological formation and underlying lithology), rain-
1112 on-snow events, immature forest cover (stands <35 years old where open canopies and
1113 undeveloped root systems could contribute to hillslope instability), presence and coverage of
1114 roads, and critical slope (hillslope gradients >36%, for terrain with surficial deposits of coarse-
1115 textured colluvial materials). Results of this study showed these variables could explain ~65% of
1116 the variation associated with sediment input into channels. The lowest risk areas contained the
1117 fewest of these variables (most commonly critical slope with either rain-on snow events or
1118 immature forests), while higher risk areas contained a combination of 4 or more of these factors
1119 indicating a compounding effect.

1120 Changes in sediment yield may also interact with increases in discharge rates caused by timber
1121 harvest as well as physiographic site factors. For example, Bywater-Reyes et al. (2018)
1122 quantified how sediment yields vary with catchment lithology and physiography, discharge, and
1123 disturbance history over 60 years in the H.J. Andrews experimental watershed in the western
1124 Cascade Range of Oregon. Methods for determining suspended sediment concentration involved
1125 using either vertically integrated storm-based grab samples, or discharge-proportional composite
1126 samples where composite samples were collected every three weeks at the outlet of each
1127 catchment. Data sets were taken from 10 watersheds, 7 with a history of management (mixture of
1128 selective canopy removal, patch-cut, 25-100% clearcut, broadcast burning, road building, and
1129 thinning), and 3 with no history of management that were used as a reference. A linear mixed
1130 effects model (log transformed to meet the normality assumption) was used to predict annual
1131 sediment yield. In this model, site was treated as a random effect while discharge and
1132 physiographic variables were treated as fixed variables. This allowed for the evaluation of the
1133 relationships between sediment yield and physiographic features (slope, elevation, roughness,
1134 and index of sediment connectivity) while accounting for site. To account for the effect of
1135 disturbance history a variable was added to the model when the watershed had a history of
1136 management or natural disturbances. If the models for the disturbed watersheds significantly
1137 underpredicted the sediment discharge, the timing of the sudden increases were further examined
1138 to assess whether it correlated with a disturbance event (e.g., harvesting, road building, and
1139 slash-burning.) The results of this study show that watershed physiography combined with
1140 cumulative annual discharge explains 67% of the variation in annual sediment yield across the
1141 60-year data set regardless of lithology. Relative to other physiographic variables, watershed

1142 slope was the greatest predictor of annual suspended sediment yield. However, the results
1143 showed that annual sediment yields also moderately correlated with many other physiographic
1144 variables and caution that the strong relationship with watershed slope is likely a proxy for many
1145 processes, encompassing multiple catchment characteristics.

1146 In contrast, Safeeq et al. (2020) compared instream and bedload sediment supply under multiple
1147 harvesting treatments in watersheds of western Oregon that were paired with control watersheds
1148 by size, aspect, and topography. The treatment watershed was 100% clearcut during the period
1149 from 1962-1966, broadcast burned in 1966, and re-seeded in 1968. For this study 15-minute
1150 streamflow data was recorded for both watersheds, and after large storm events. Sediment data
1151 was collected from 1952 (pre-harvest) through 1988 for suspended sediment data, and 2016 for
1152 sediment bedload. The control watershed was forested, and had no treatments (e.g., harvest)
1153 during the study period. Their results estimate that following streamside harvest, increased
1154 streamflow alone is estimated to be responsible for <10% of sediment transport into streams
1155 while the increased sediment supply caused by harvest operations is responsible for >90% of the
1156 sediment transported into streams.

1157 Puntteney-Desmond et al. (2020) found similar results in their assessment of differences in
1158 instream sediment contributions from the buffer area, harvest area, and buffer-harvest interface.
1159 Sediment concentration in the runoff was 15.8 times higher for the harvested area than in the
1160 riparian buffer, and 4.2 times greater than in the harvest-buffer interface. Total sediment yields
1161 ($\text{mg m}^{-2} \text{min}^{-1}$) from the harvested area (sediment concentration x flow rate) were approximately
1162 2 times greater than in the buffer areas, and 1.2 times greater in the harvest-buffer interface than
1163 in the buffer area.

1164 *Summary of Factors Impacting Sediment Delivery into Streams*

1165 From the studies reviewed there is evidence that sediment delivery into streams following timber
1166 harvest is influenced by not only the intensity of the harvest operation (e.g., presence of retention
1167 buffers, yarding and equipment use immediately adjacent to the stream, upland clearcut vs.
1168 thinning), but also by physiography (e.g., hillslope gradient), lithology relative softness, and
1169 climate (e.g., precipitation, frequency of large storm events). Thus, the change in magnitude of
1170 sediment delivery following harvest is context dependent and these landscape factors can interact
1171 with one another to compound these changes. However, from the studies reviewed above there is
1172 evidence that the implementation of BMPs since the 1970s in the northwestern United States
1173 lessen the impact and duration of these changes.

1175 [Shade and stream temperature](#)

1176 Canopy cover provides shade for streams that decreases the amount of incoming solar radiation
1177 and thus influences stream temperatures, although that influence can be highly variable
1178 depending on shade structure and density surrounding stream courses. Temperature regulation is
1179 vital for sensitive salmonid fish species that require cooler waters, and shade is often the primary
1180 function assessed when developing state regulations (Groom et al., 2011; Groom et al., 2018;
1181 Teply et al., 2014). The importance of shade and cooler in-stream temperatures for fish habitat
1182 has been thoroughly investigated (Bjornn & Reiser, 1991; Chapman & Bjornn, 1969; Ebersole et
1183 al., 2001; Sullivan et al., 2000). The streamside shade will likely become even more critical with
1184 the predicted increases in air temperature over the next century (Manuta et al., 2009). While
1185 stream temperature is initially reflective of moisture source (e.g., snowmelt, liquid precipitation,
1186 groundwater inputs) and watershed subsurface soil characteristics. As water flows downstream
1187 and into higher-order streams, the net rate of temperature gain or loss is the sum of incident
1188 radiation, evaporation, conduction, and advection (Brown, 1983; Bescheta et al., 1987).

1189 Bescheta et al. (1987) presented evidence that direct beam solar radiation inputs are of the
1190 highest importance to the stream's net heat exchange rate per unit area compared to other factors.
1191 Within the net heat exchange calculation, the heat released from evaporation generally cancels
1192 out the heat gained from warm air temperatures (convective and advective heat transfer). Thus,
1193 temperature fluctuations are expected to be more severe in less-shaded/more-exposed streams.
1194 This has been supported by many experimental field and simulation studies showing evidence
1195 that the reduction of effective shade can lead to considerable increases in peak summer stream
1196 temperatures primarily due to the increase of incoming solar radiation. However, while increases
1197 in solar radiation are accepted as the most important factor in stream temperature changes and
1198 fluctuations following harvest, other factors are also important and may compound these effects.

1199 Guenther et al. (2014) investigated the relationship between changes in stream temperature and
1200 changes in wind speed, vapor pressure, and evaporation following riparian thinning treatments
1201 along headwater streams in southwestern British Columbia. Treatment involved reduction of
1202 basal area by 50% (resulting in 14% reduction in canopy closure) in the upland and riparian
1203 forests. Results showed a post-harvest increase in wind speed, vapor pressure deficit, air
1204 temperature and evaporation above the stream, which coincided with increased stream
1205 temperatures and lower stability. The authors report that prior to harvest, vapor pressure
1206 gradients often favored condensation over evaporation. Further, they concluded that the
1207 relationships between the riparian and microclimate variables after harvesting became more
1208 strongly coupled to ambient climatic conditions due to increased ventilation. Contemporary
1209 riparian management practices in western North America vary by state. However, all require
1210 retention of protective buffers that preserve some percentage of shade or canopy cover to
1211 maintain or mitigate changes in stream temperatures, especially along fish-bearing streams.
1212 Many studies published in the last two decades report evidence that these practices have been
1213 effective in mitigating stream temperature changes after harvest.

1214 For example, Bladon et al. (2016), assessed the effectiveness of riparian management
1215 prescriptions developed for the Oregon Forest Practices Act (FPA). Oregon State requires a 15 m

1216 buffer on either side of small fish-bearing streams with a 6 m no-cut buffer, and a minimum
1217 retention for conifer basal area of $\sim 3.7 \text{ m}^2$ for every 300 m (~ 1000 ft) length of stream. This
1218 resulted in a reduction of mean canopy closure from $\sim 96\%$ in the pre-harvest period to $\sim 89\%$ in
1219 the post-harvest period in the treatment reaches. In contrast, mean canopy closure in the
1220 reference reaches changed from $\sim 92\%$ to $\sim 91\%$ from pre- to post-treatment periods. Results
1221 showed there was a significant increase in the 7-day moving maximum temperature from pre- to
1222 post-harvest values when data was constrained to the period of July 15 – August 15 by $0.6 \pm$
1223 $0.2 \text{ }^\circ\text{C}$. However, when analyzed by individually paired sites, and when interannual and site
1224 variability was accounted for, no significant changes in stream temperature were observed for 3
1225 years post-harvest (length of study).

1226 However, Groom et al., (2011a, b) showed evidence that the more stringent rules of the
1227 Northwest Oregon State Forest Management Plan (FMP; applied to riparian management zones
1228 on state owned land) was even more effective at maintaining stream temperatures post-harvest.
1229 The FMP requires a 52 m wide buffer for all fish-bearing streams, with an 8 m no cut buffer
1230 immediately adjacent to the stream. The results from Groom et al. (2011b) showed that FPA
1231 (Oregon Forest Practices) post-harvest shade values differed from pre-harvest values (mean
1232 change in Shade from 85% to 78%), while no difference was found for FMP shade values pre-
1233 harvest to post-harvest (mean change in Shade from 90% to 89%). Following harvest, maximum
1234 temperatures at FPA increased relative to FMP on average by $0.71 \text{ }^\circ\text{C}$. Similarly, mean
1235 temperatures increased by $0.37 \text{ }^\circ\text{C}$ (range: 0.24 - 0.50), minimum temperatures by $0.13 \text{ }^\circ\text{C}$
1236 (range: 0.03 - 0.23), and diel fluctuation increased by $0.58 \text{ }^\circ\text{C}$ (range: 0.41 - 0.75) relative to
1237 FMP sites.

1238 Groom et al (2011a) developed prediction models from this data to estimate the probability of
1239 riparian harvest under each regulation causing an increase in stream temperatures $>0.3 \text{ }^\circ\text{C}$ (the
1240 Protecting Cold Water criterion developed by the Department of Environmental Quality). Results
1241 indicate that sites harvested according to FPA standards exhibited a 40.1% probability of a
1242 temperature change of $> 0.3^\circ\text{C}$ from pre- to post harvest. Conversely, harvest to FMP standards
1243 resulted in an 8.6% probability of exceedance that did not significantly differ from all other
1244 comparisons.

1245 In Montana, Sugden et al. (2019) investigated the effectiveness of state regulation which requires
1246 timber be retained within a minimum of 15.2 m (50 feet) of the stream. Within the riparian
1247 management zone, no more than half the trees greater than 204 mm (8 in) diameter at breast
1248 height (DBH) can be removed. In no case, however, can stocking levels of leave trees be reduced
1249 to less than 217 trees per hectare. Data for canopy cover, stream temperature, and fish population
1250 were collected for 30 harvest reaches in western Montana (northern Rocky Mountain Region),
1251 for a minimum of one-year pre- and one-year post-harvest. Shade over the stream surface was
1252 not directly measured in this study. Instead, canopy cover was used as proxy, using two
1253 independent estimates of canopy cover (1) used cruise data to populate a canopy cover model
1254 within Forest Vegetation Simulator, and (2) measured canopy cover in the harvested reach every
1255 30 m, before and after harvest. Within harvest units, mean basal area was reduced by 13%
1256 (range: 0 – 36%), and again further by a mean of 2% due to windthrow. Mean canopy cover

1257 within the riparian management area reduced from 77% (pre-treatment) to 74% (post-treatment),
1258 and mean canopy cover over the stream changed from 66% (pre-treatment) to 67% (post-
1259 treatment) based on densiometer measurements. Neither of these changes were significant.
1260 Results for stream temperature also showed no significant changes in stream temperatures or fish
1261 populations in one-year post treatment compared to pre-treatment values.

1262 Specific to Washington, Cupp & Lofgren (2014) conducted a study to test the effectiveness of
1263 riparian timber harvest rules for eastern Washington in preserving shade and stream
1264 temperatures. Regulations for fish-bearing streams in eastern Washington (in the mixed
1265 conifer/mid elevation zone) includes an “All Available Shade Rule” (ASR) for streams in the bull
1266 trout habitat zones, and a “Standard Shade Rule” (SR). Under the ASR it is required to retain all
1267 available shade within 75 feet of the stream. Under SR some harvest of shade providing trees is
1268 allowed within the 75-foot buffer depending on elevation and pre-harvest canopy cover.
1269 Unharvested reference reaches were located upstream from treatment reaches. Prior to harvest
1270 treatments, canopy closure measurements ranged from 89% to 97%, with a mean of 93%.
1271 Results showed post-harvest shade values decreased in SR sites (mean effect of -2.8%, $p =$
1272 0.002), as did the canopy closure values (mean effect of -4.5%, $p < 0.001$). Shade and canopy
1273 closure values did not significantly change after treatment in the ASR sites. Post-harvest mean
1274 daily maximum stream temperature increased 0.16 °C in the SR harvest reaches, whereas stream
1275 temperatures in both the ASR sites and in the no-harvest reference reaches increased on average
1276 by 0.02 °C. Sample period means of daily maximum temperature responses varied from -1.1 °C
1277 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
1278 sites, and -0.5 to 0.9 °C in the reference sites. While these values show a slight increase in mean
1279 temperatures and temperature ranges with treatment, the authors interpret these results as
1280 evidence that temperature effects of the SR and ASR were similar to reference conditions along
1281 sampled reaches.

1282 Riparian harvest rules along non-fish bearing streams tend to allow for narrower buffer widths
1283 (sometimes with no retention buffers) or more intense thinning within the buffer than for fish-
1284 bearing streams. For example, in western Washington the Forest Practices (FP) buffer
1285 prescription requires a two-sided 15 m (50 ft) wide buffer along a minimum of 50% of the length
1286 of a non-fish-bearing perennial stream (i.e., up to 50% of the stream may have no buffer) with a
1287 9.1 m (30 ft) equipment exclusion zone. Two recent studies (Ehinger et al., 2021; McIntyre et al.,
1288 2021) have compared these FP buffers to two experimental buffer treatments, a 50 ft buffer along
1289 100% of the stream length (100%), and no buffer (0%) treatment, and an unharvest reference
1290 (REF) on sites underlain by competent lithologies (McIntyre et al., 2021; “Hard Rock”) or
1291 incompetent (friable) lithologies (Ehinger et al. 2021; “Soft Rock”).

1292 Results from the Hard Rock study showed that riparian canopy cover declined after harvest in all
1293 buffer treatments reaching a minimum around 4 years post-harvest (after mortality stabilized).
1294 The treatments, ranked from least to most change, were REF, 100%, FP, and 0% for all metrics
1295 and across all years. Effective shade results showed decreases of 11, 36, and 74 percent in the
1296 100%, FP, and 0% treatments, respectively. These changes in shade were significant for all
1297 treatments. This led to changes in mean stream temperature from pre- to post-harvest in the

1298 100% treatment by 1.1°C °C in the first two years following treatment, but returned to pre-
1299 harvest levels by post-harvest year 3. In contrast, the mean difference in pre- to post-harvest
1300 stream temperatures in the FP ranged from 0.5°C to 1.1°C in the first year, and changed little over
1301 the entire post-harvest period.. Results for the 0% treatment showed a mean increase of 3.8°C
1302 immediately following harvest and declined over time to 0.8°C by year 9. These results suggest
1303 that the 100% treatment was most effective at preventing increases to stream temperature
1304 followed by the FP and 0% treatments. A weak and nearly significant (P-value range: 0.008 -
1305 0.108) negative relationship between canopy cover and stream temperature for the first 4 years
1306 after treatment was detected. These results provide evidence that the effectiveness of buffers in
1307 maintaining stream temperatures post-harvest is relative to the intensity of the treatment (e.g.,
1308 presence of buffer, reduction in canopy cover). Further, post-treatment mortality within the
1309 buffer from events such as windthrow can cause fluctuations in stream temperature response
1310 during the first decade. Results from the Soft Rock Study showed similar trends in canopy cover
1311 reduction and stream temperature increases. Authors of the Soft Rock study note that stream
1312 temperature changes varied as a function of the proportion of the stream buffered and tree
1313 mortality.

1314 Outside of Washington, several studies conducted in western North America since 2000 have
1315 shown results similar to the Hard Rock and Soft Rock studies. For example, Roon et al. (2021b)
1316 compared stream temperature changes following variable riparian thinning intensities in the
1317 redwood forests of northern California. Treatments to riparian stands included reduction of
1318 canopy cover that resulted in reduction of effective shade by either (19-30%) or by (4-5%). Their
1319 results showed that local changes in stream temperature were dependent on thinning intensity,
1320 with higher levels of canopy cover reduction leading to increases in local stream temperatures. In
1321 the reaches with higher reductions in shade (19-30%) there was accumulation of 45° to 115°C
1322 additional degree days from pre- to post treatment years, while the reaches with lower reductions
1323 in shade (4-5%) only accumulated 10° to 15°C additional degree days. Further, travel distance of
1324 increased stream temperatures also appeared to be dependent on thinning intensity. The lower
1325 shade reduction reaches had an increased temperature effect downstream with travel distance of
1326 75-150 m, while the high shade reduction sites had a downstream travel distance of 300- ~1000
1327 m. Roon et al. (2021a) reported changes in in average daily maximum, maximum weekly
1328 average of the maximum (MWMT), average daily mean, or maximum weekly average of the
1329 mean (MWAT) at these same sites under the same timeline. The lower thinning intensity (4-5%
1330 effective shade reduction) showed no significant changes in any temperature metrics. However,
1331 The more intensely thinned sites (19-30% reduction in effective shade) showed an increase
1332 MWMT during spring by a mean of 1.7°C (0.9 - 2.5 °C), summer by a mean of 2.8°C (1.8, 3.8
1333 °C), and fall by a mean of 1.0°C (0.5, 1.5 °C) and increased in downstream reaches during spring
1334 by a mean of 1.0°C (0.0, 2.0 °C) and summer by a mean of 1.4°C (0.3, 2.6 °C). The authors
1335 interpret their results as evidence that that changes in shade of 5% or less caused minimal
1336 changes in temperature while reductions in shade of 20–30% resulted in much larger increases in
1337 temperature.

1338 Reiter et al. (2020) compared the changes in stream temperatures following different harvest
1339 treatments along headwater streams in the Trask River Watershed in the northwestern coast range

1340 of Oregon. Treatments included a clearcut to stream (no buffer but half of sites contained some
1341 leave trees along stream bank), upland clearcut with a 10 m no-cut buffer, upland thinning (basal
1342 area reduction to 30-50% of original stand) with a 10 m no-cut buffer, and an unharvested
1343 reference. Results showed that post-harvest stream temperature increases were only significant in
1344 the clear-cut treatments without buffers with a mean increase of 3.6°C (SE = 0.4°C) for four
1345 years after the study. They note that temperature changes were more severe in the unbuffered
1346 streams with no leave trees (4.2 and 4.4°C), however, this difference was not analyzed. No
1347 significant changes in stream temperature were detected in either treatment with a 10 m no-cut
1348 buffer. The authors speculate that 10 m wide buffers were sufficient in maintaining stream
1349 temperatures post-harvest in small, forested headwater streams.

1350 In the sub-boreal forest ecosystems of British Columbia, Canada, Macdonald et al. (2003b)
1351 compared pre- to post-harvest stream temperature changes in first-order headwater streams under
1352 3 different riparian forest treatments. These treatments included 1) low-retention – removal of all
1353 merchantable timber >15 or >20 cm DBH for pine or spruce respectively, within 20 m of the
1354 stream 2) high-retention – removal of merchantable timber >30 cm DBH within 20-30 m of the
1355 stream, and 3) patch-cut – high retention for the lower 60% of watershed approaching streams
1356 and removal of all vegetation in the upper 40% of the watershed. Results showed significant
1357 increase in stream temperatures ranging from 4 – 6 °C in the low-retention and patch cut in the
1358 first three years following harvest. However, by year five, mortality in the high-retention buffer
1359 (due to windthrow) resulted in canopy cover reduction and increases in stream temperatures that
1360 became equivalent to the other treatments. The authors conclude that while the variation in
1361 harvest intensity initially appeared to dictate stream temperature responses, site effects (e.g.,
1362 windthrow susceptibility) can impact the effectiveness of the buffer. While the studies above all
1363 show evidence that the impact of riparian forest harvest on stream temperatures are related to the
1364 severity of the harvest prescription (e.g., buffer width, thinning intensity, canopy reduction) the
1365 results are variable within treatments indicating other site factors are also important when
1366 evaluating buffer effectiveness. For example, in their review of experimental studies conducted
1367 in the Pacific Northwest of Canada and the United States, Martin et al. (2021) reported high
1368 variability in temperature response to streamside buffers. They report a substantial variability and
1369 overlap in the effect size of the mean 7-day maximum temperature metric with no-cut buffers,
1370 no-cut plus variable retention buffers, and no-cut patch buffers ≤ 20 m wide. The largest
1371 temperature response (> 3.4 °C) occurred in the clearcut buffers while treatments with buffers
1372 (i.e., no cut buffers without variable retention) had the smallest response (< 0 °C). The variable
1373 retention buffers < 20 m showed variable response (0.6 – 1.4 °C). They conclude that the
1374 variation in temperature response following riparian harvest may be associated with multiple
1375 factors such as geology, hydrology, topography, latitude, and stream azimuth.

1376 Bladon et al. (2018) investigated the changes in stream temperatures following treatments that
1377 varied from clearcuts to stream to buffers > 20 m in western Oregon. They performed a
1378 regression analysis to assess the relative relationship between catchment lithology and the
1379 percentage catchment harvested with stream temperature at all sites. Their results showed that at
1380 the upstream harvested sites there was a strong relationship between stream temperature
1381 increases and catchment lithologies, but no statistically significant relationship between stream

1382 temperature changes and percent of catchment harvested. Sites downstream from harvested areas
1383 showed a significant relationship with the interaction of percentage of catchment harvested and
1384 the underlying lithologies ($p = 0.01$). The greatest temperature increases at downstream sites
1385 were in areas with a higher percentage of catchment harvested and were underlain by more
1386 resistant lithologies. There was no evidence for increases in stream temperatures in catchments
1387 with a high percentage of harvest that were underlain by permeable geology. The authors suggest
1388 that this relationship may be due to the buffering effect of increases in summer low flows and
1389 greater groundwater or hyporheic exchange. They conclude that the variability of rock
1390 permeability and the relative contribution of groundwater during summer months, and their
1391 effect on stream temperatures following harvest should be investigated further.

1392 There is evidence that geomorphology alone can impact stream temperature fluctuations
1393 throughout the year. Hunter & Quinn, (2009) compared seasonal fluctuations in stream
1394 temperatures between two watersheds in the Olympic Peninsula, Washington. Both watersheds
1395 were similar in all characteristics except for bed substrate. One was underlain by alluvial bed
1396 substrate while the other was underlain by bedrock. Results from this study show consistent
1397 differences in stream temperature response in alluvial versus bedrock channels. Seasonal
1398 maximum and minimum average daily temperatures varied less at the alluvial site compared to
1399 the bedrock site. This, the authors suggest, may be due to hyporheic exchange in alluvial
1400 channels helping to buffer surface water temperatures from gaining or losing heat. In addition,
1401 groundwater may also contribute to the increased stability at the alluvial site. Aside from shade
1402 reduction from timber harvest, there is evidence that light availability and canopy cover naturally
1403 changes over time as riparian stands develop. For example, Warren et al. (2013) compared
1404 canopy cover and stream light availability between old-growth-forests (>500 years old) and
1405 young harvest-aged stands (~40-60 years old) in the H.J. Andrews Experimental Forest in the
1406 Cascade mountains of Oregon. Streams were paired based on reach length and bankfull width,
1407 and north ($n=2$), and south ($n=2$) facing watersheds. Canopy cover was estimated using a
1408 convex spherical densiometer, and light reaching the stream bed was estimated using a
1409 fluorescent dye that degrades overtime from light exposure. Overall, three of the four paired old-
1410 growth reaches (2 south-facing, 1 north-facing) had significantly lower mean percent canopy
1411 cover ($p < 0.10$), and significantly higher mean decline in fluorescent dye concentrations ($p <$
1412 0.01). The authors interpret these results as evidence that old-growth forest canopies were more
1413 complex and had more frequent gaps allowing for more light availability and lower mean canopy
1414 cover, on average, than in adjacent young, second growth forests.

1415 Kaylor et al. (2017) presented similar results when they compared canopy cover and light
1416 availability between small mountain streams adjacent to late-successional forests (dominant
1417 canopy trees >300 years old) and second-growth forests that had been harvested to the stream
1418 50-60 years prior to data collection. Like Warren et al. (2013), canopy cover was estimated with
1419 a convex spherical densiometer; and light availability to streams was estimated with a
1420 photodegrading fluorescent dye. However, for this study, fluorescent dye degradation was
1421 converted to photosynthetically active radiation (PAR) by building a linear relationship between
1422 the dye degradation and PAR sensors. Results showed that mean PAR reaching streams was 1.7
1423 times greater, and canopy openness was 6.1% greater in >300-year-old forests than in 30–100-

1424 year-old forests. Of the 14 paired sites, differences in canopy openness and PAR were significant
1425 for 6 sites. The authors compared and combined their data with published data from 10 other
1426 similar studies. The combined datapoints for canopy openness (%) were plotted against stand age
1427 and fit it with a negative exponential curve. From the slope of the curve, the authors estimate that
1428 canopy openness reaches its minimum value in regenerating forests at ~30 years and maintains
1429 with little variability until ~100 years.

1430 *Summary of Factors Affecting Shade and Stream Temperature*

1431 From the studies reviewed above, the results show evidence that changes in canopy cover and
1432 effective shade are, not surprisingly, directly related to the intensity of harvest operation. Initial
1433 reduction in canopy cover and shade from pre- to post-harvest are influenced by the basal area
1434 removed and the width of the retention buffer. However, there is evidence that multiple site
1435 factors can interact with harvest operations (e.g., target basal areas).

1436

1437

1438

1439 Results/discussion by focal question

1440 Focal Question 1

1441 1. *What are the effects of timber harvest intensities and extent on the riparian functions, with an*
 1442 *emphasis on the five key functions listed above, in comparison to conditions before harvest?*

1443 From the perspective of an experimental design, this question inquires how the values of the
 1444 metrics used to describe the five key functions (large woody debris recruitment, sediment
 1445 filtration, stream bank stability, shade, litterfall and nutrients) differ from pre- to post-harvest
 1446 within particular riparian areas of interest. An attempt to answer this question would require data
 1447 collection before and after treatment with or without a control site. Thus, only studies that used a
 1448 BACI or BAI approach are appropriate for discussing this question. From our review, 22 papers
 1449 report pre- to post-harvest changes in the magnitude of one or more of the key functions with the
 1450 majority of these papers focusing on changes in shade. No studies published since 2000 that
 1451 apply an experimental design in western North America to quantify changes in bank stability
 1452 could be found in the literature.

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

1453

1454 *Shade*

1455 Table 4. Treatment and responses for selected publications investigating shade relevant to Q1.

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

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

1456

1457 Specific to fish-bearing streams of eastern Washington, Cupp & Lofgren (2014) reported changes
1458 in canopy closure (quantified with handheld densiometer) and shade (quantified with fisheye lens
1459 digital camera) within reaches adjacent to riparian forests harvested under the All Available
1460 Shade Rule (ASR) and the Standard Shade Rule (SR). Both shade rules have a 30-ft no-cut
1461 buffer (core zone) immediately adjacent to the stream. The SR prescription allows thinning in the
1462 buffer zone 30-75 feet (inner zone) from the stream while the AAS prescription requires
1463 retention of all shade providing trees in this area. Results showed post-harvest shade values
1464 decreased in SR sites (mean effect of -2.8%, p = 0.002), as did the canopy closure values (mean
1465 effect of -4.5%, p < 0.001). Shade and canopy closure values did not significantly change in the
1466 treatment reaches of the ASR sites. Mean shade reduction in the SR treatment sites exceeded the

1467 mean shade reduction in the ASR sites by 3%. Canopy closure reduction was also greater in the
1468 SR sites than in the ASR sites by a mean of 4%.

1469 For non-fish bearing streams of western Washington, McIntyre et al. (2021) report changes in
1470 canopy closure following 3 different harvest prescriptions. Prescriptions included a two-sided
1471 50-ft wide riparian buffer along the entire stream (100%), a two-sided 50-ft riparian buffer along
1472 at least 50% of the stream consistent with the current Forest Practices buffer prescription (FP),
1473 and a clearcut to stream edge without a buffer (0%). The canopy cover was estimated at mid-
1474 stream with a handheld densiometer and was converted to effective shade values (for 5 years
1475 post-harvest). Hemispherical canopy photos were also taken for 4 years pre-harvest and 3 years
1476 post-harvest and converted to Canopy and Topographic Density (percentage of the photograph
1477 obscured by vegetation or topography). Results for canopy cover showed that riparian cover
1478 declined after harvest in all buffer treatments reaching a minimum around 4 years post-harvest.
1479 The treatments, ranked from least to most change, were 100%, FP, and 0% for all metrics and
1480 across all years. Effective shade results showed decreases of 11, 36, and 74 percent in the 100%,
1481 FP, and 0% treatments, respectively, by 3 years post-harvest. However, by post-harvest year 9,
1482 canopy closure returned to pre-harvest levels in the 100% treatment but remained 15% and 27%
1483 below pre-harvest values at the FP and 0% treatments, respectively. . Another study, Janisch et
1484 al. (2012) also compared the effects of similar treatments (clearcut to stream, a full continuous
1485 buffer (10-15 m wide), and a patched buffer (~50-110 m long were retained in distinct patches
1486 along some portion of the channel) to canopy cover. Canopy cover in all streams averaged 95%
1487 (SE = 0.4) prior to harvest. Following treatment, canopy cover in the clearcut catchments
1488 averaged 53%, (SE = 7.4) canopy cover in the patch buffer treatment averaged 76%, (SE = 5.1)
1489 and canopy cover in the continuous buffer treatment averaged 86% (SE = 1.7). The changes were
1490 significant in the clearcut and patch buffers.

1491

1492 Outside of Washington, Bladon et al. (2016) assessed the effects of harvest treatments under the
1493 Oregon Forest Practices Act (FPA) on shade reduction and stream temperature. This study took
1494 place in the Siuslaw National Forest in the Oregon Coast Range in the Alsea Watershed.
1495 Treatment under the FPA includes a 15 m riparian management area with a minimum of ~3.7 m²
1496 conifer basal area retained for every 300 m length of stream and an additional 4-5 wildlife leave
1497 trees per hectare. This resulted in a mean canopy closure reduction from ~96% (pre-harvest) to
1498 ~89% (post-harvest) based on measurements from a densiometer along the stream channel for 3
1499 years pre- and 3 years post-harvest. Unfortunately, the authors did not compare these changes
1500 with statistical analysis. Groom et al. (2011b) compared changes in shade from pre- to post-
1501 harvest under the FPA and under the Northwest Oregon State Forest Management Plan (FMP).
1502 The FMP requires a 52 m wide buffer for all fish-bearing streams, with an 8 m no cut buffer
1503 immediately adjacent to the stream.

1504 Results from Groom et al. (2011b) showed that FPA site post-harvest shade values differed from
1505 pre-harvest values (mean change in Shade from 85% to 78%); While no difference was found for
1506 FMP site shade values pre-harvest to post-harvest (mean change in Shade from 90% to 89%). In
1507 the Trask Watershed of the northwestern Oregon Coast range, Reiter et al. (2020) compared three
1508 riparian zone treatments: 1) clearcut, no buffer (CC_NB; n = 4), 2) clearcut with 10-m no cut
1509 buffer (CC_B; n = 3), 3) thinning with 10 m no-cut buffer (TH_B; n =1) in small non-fish

1510 bearing streams. Pre- to post-harvest values in shade were quantified with hemispherical analysis
 1511 over the stream one-year prior and one-year post-treatment. However, post-harvest overstory
 1512 buffer width varied within each treatment depending on landscape factors. For this reason, we
 1513 will present the change in percent shade with residual buffer width (Table 6). Again, changes in
 1514 shade were not statistically analyzed.

1515 In fish-bearing streams within the McKenzie River basin in the western Cascade Mountains of
 1516 Oregon Swartz et al. (2020) assessed the effects of experimental canopy gap treatments on shade
 1517 and light availability to the stream. In each treatment reach (n = 6), 20 m gaps were prescribed to
 1518 mimic gap openings that naturally occur after individual large tree mortality or small-scale
 1519 disturbance events in late successional forests. Shade was recorded in the year before and the
 1520 year after treatment with hemispherical photos. Changes in effective shade (1 year post-harvest)
 1521 were estimated in HemiView 2.1 software. Mean stream shading could not be evaluated in the
 1522 full BACI analysis because post-treatment hemispherical photographs could not be taken at all
 1523 sites due to fire impeding access in 2018. For the remaining sites, the areas beneath each gap had
 1524 notable localized declines in shade, through the entirety of the treatment reach mean shading
 1525 declined by only 4% (SD ± 0.02%).

1526 Table 5. Results for changes in shade following treatment for the Trask River Watershed Study
 1527 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

1528 CC_B = clearcut with 10 m buffer, CC_NB = clearcut no buffer, TH_B upland thinning with
 1529 buffer. *Unable to determine exact buffer width because adjacent to thinning

1530 Gravelle & Link (2007) compared changes in shade following treatment for non-fish bearing
 1531 streams in northern Idaho. For non-fish-bearing streams there is a 30 ft (9.1 m) equipment
 1532 exclusion zone on each side of the ordinary high-water mark (definable bank). There are no
 1533 shade requirements and no leave tree requirements, but skidding logs in or through streams is
 1534 prohibited. Harvesting treatments included (1) clearcut and (2) thinning to a 50% shade removal.
 1535 Canopy cover measurements were made using a concave spherical densiometer. Preharvest
 1536 canopy measurements ranged from 56% to 88%, with an average of 63% in the clearcut reaches,
 1537 and 74% in the partial cut reaches. In the clearcut reaches, canopy was reduced to 52% in 2002
 1538 and 41% in 2003, immediately following broadcast burning and replanting. In 2004 and 2005,
 1539 overall canopy was measured at 56% and 54%, respectively. Streamside shade recovery can be

1540 attributed entirely to low-lying understory species, as evidenced by the increase in
1541 understory/deciduous cover of 26% in 2003 to 39% and 37% in 2004 and 2005, respectively. In
1542 the partial cut reaches, canopy shade remained near 75%.

1543 In fish-bearing streams of Montana, Sugden et al. (2019) assessed the effectiveness of state
1544 riparian management harvest prescriptions in maintaining canopy cover. Montana state law
1545 requires timber be retained within a minimum of 15.2 m of fish-bearing streams, with equipment
1546 exclusion zones extended on steep slopes for up to 30.5 m. Within the riparian management
1547 zone, no more than half the trees greater than 204 mm (8 in) diameter at breast height (DBH) can
1548 be removed. In no case, however, can stocking levels of leave trees be reduced to less than 217
1549 trees per hectare. Shade over the stream surface was not directly measured in this study. Rather,
1550 canopy cover was used as a general proxy, with two independent estimates of canopy cover
1551 employed. One method used the riparian cruise data to populate a canopy cover model within the
1552 Forest Vegetation Simulator (FVS), which estimated canopy cover for each study site, pre- and
1553 post-harvest. The second method measured canopy cover in the harvest reach every 30 m, both
1554 before and after timber harvest, using a concave spherical forest densiometer. Mean canopy
1555 cover in the SMZ, as modelled in FVS, decreased from 77% to 74% following timber harvest
1556 and 73% when subtracting windthrow to differentiate between direct and indirect impacts of
1557 management (Table 3). The mean canopy cover over the stream channel based on densiometer
1558 measurements was 66% pre-harvest and 67% post-harvest. Neither of these changes was
1559 statistically significant.

1560 Roon et al. (2021a) compared the effects of two experimental thinning treatments on shade in
1561 second growth redwood stands (40–60 years old) of northern California. This study took place
1562 between 2016 and 2018 with thinning treatments applied during 2017 giving 1-year pre-
1563 treatment and 1-year of post-treatment data. Two study sites prescribed treatment on one side of
1564 the stream of a 45 m buffer width with a 22.5 m inner zone with 85% canopy retention and a
1565 22.5 m outer zone that retained 70% canopy cover (Tectah watershed). At the third treatment site,
1566 thinning prescriptions included removal of up to 40% of the basal area within the riparian zone
1567 on slopes less than 20% on both sides of the channel along a ~100–150 m reach (Lost Man
1568 watershed, Redwood national park). Shade over streams was measured with hemispherical
1569 photos and effective shade was calculated in HemiView Canopy Analysis Software version 2.1.
1570 Results for the Tectah watershed showed a significant reduction in canopy closure by a mean of
1571 18.7%, (95% CI: -21.0, -16.3) and a significant reduction of effective shade by a mean of 23.0%
1572 (-25.8, -20.1) one-year post treatment. In the Lost man watershed, a non-significant reduction of
1573 mean shade by 4.1% (-8.0, -0.5), and mean canopy closure by 1.9% was observed in 2018.
1574 Results for below canopy light availability showed significant increases by a mean of 33% (27.3,
1575 38.5) in the Tectah watershed, and non-significant increases in Lost man watershed of 2.5% (-
1576 1.6, 5.6) by 2018.

1577 In general, the results from the studies reviewed above suggest changes in shade or canopy cover
1578 from pre- to post-harvest are directly impacted by the intensity of the treatment prescription.
1579 Buffer treatments vary between states and within states by stream type (e.g., fish-bearing or non-
1580 fish-bearing), For the studies that quantified pre- to post-changes in shade along fish-bearing

1581 streams (Cupp & Lofgren, 2014; Sugden et al. 2019), results show evidence that the application
 1582 of best management practices (BMPs) cause minimal or non-significant changes in shade
 1583 following harvest. For non-fish-bearing streams harvest prescriptions are much more variable.
 1584 Further, there are many more examples of application and comparison of different experimental
 1585 buffer treatments which vary by width or thinning targets.

1586 *Litter*

1587

1588 Table 7. Treatment and responses for selected publications investigating Litter relevant to Q1.

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

1589

1590 Specific to western Washington, McIntyre et al. (2018) compared the change in litterfall inputs
 1591 from pre- to post-harvest under three different riparian harvest treatments. Treatments included a
 1592 two-sided 50-ft riparian buffer along at least 50% of the stream (FP; with clearcut to stream's
 1593 edge outside of the buffer), a two sided 50-ft buffer along the entire stream (100%), and a
 1594 clearcut to stream without a buffer (0%). Litterfall was collected with litter traps placed along the
 1595 mainstem channel of each site. Litter was dried and sorted by type (e.g., deciduous, conifer,
 1596 small wood) and ashed to compare weight. Results for litterfall input showed a decrease in total
 1597 litterfall input in the FP (P = 0.0034) and 0% (P = 0.0001) treatments between pre- and post-
 1598 treatment periods. Leaf litterfall (deciduous and conifer leaves combined) input decreased in the
 1599 FP (P = 0.0114) and 0% (P < 0.0001) treatments in the post-treatment period. In addition, conifer
 1600 (conifer needles and scales) litterfall input decreased in the FP (P = 0.0437) and 0% (P < 0.0001)
 1601 treatments, deciduous leaves in the 0% (P < 0.0001) treatment, wood (twigs and cones) in the FP
 1602 (P = 0.0044) and 0% (P = 0.0153) treatments, and misc. (e.g., moss and flowers) in the 0% (P =
 1603 0.0422) treatment.

1604 In the Malcom Knapp Research Forests of British Columbia, Canada, Kiffney & Richardson
 1605 (2010) compared changes in litter input between riparian harvest prescriptions that included
 1606 clear-cut to stream edge, 10 m wide buffer reserve, and 30 m buffer reserves over the course of 8
 1607 years. No thinning was applied within the reserves. Upland treatment at all sites used clearcutting
 1608 methods. Vertical litter inputs were collected monthly and at approximately 6–8-week intervals
 1609 during each season for years 1,2,6,7, and 8 years after harvest. Litter was separated into
 1610 broadleaf deciduous, twig, needles, and other (seeds, cones, and moss) categories following
 1611 collection and subsequently dried and weighed using a microbalance. Results for post-harvest
 1612 changes in litterfall input by treatment per year are summarized in **Table 7**. Actual values of pre-
 1613 to post-harvest changes in litterfall input by type, treatment, and year were not directly reported,
 1614 however, the authors report that post-harvest inputs of needles, twigs, and total particulate matter
 1615 were significantly lower for clearcuts compared to all other treatments.

1616 Table 7. Percent change in total litterfall percentage post-harvest by treatment per year from
 1617 Kiffney & Richardson (2010). Table reproduced and modified from Yeung et al. (2019)
 1618 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
	~ 74	8

1619

1620 *Large Wood (LW) recruitment*

1621 Table 8. Treatment and responses for selected publications investigating Large Wood relevant to
 1622 Q1.

Reference	Treatment	Response
McIntyre et al. (2021)	Buffer width of 50 feet (100%); 50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)	<u>8 years post-harvest</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.
Ehinger et al. (2021)	1) 50 feet, 2) <50ft buffers (variable), and 3) unbuffered, harvested to the edge of the channel	<u>3 years post-harvest</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%.

1623

1624 Specific to western Washington, McIntyre et al. (2021) compared the change in mean in-stream
 1625 large wood from pre- to post-harvest under three different riparian harvest treatments in non-fish-
 1626 bearing streams. Treatments included a two-sided 50-ft riparian buffer along at least 50% of the
 1627 stream (FP; with clearcut to stream’s edge outside of the buffer), a two sided 50-ft buffer along
 1628 the entire stream (100%), and a clearcut to stream without a buffer (0%). Results showed a 66%
 1629 (P <0.001), 44% (P = 0.05) and 47% (P = 0.01) increase in mean large wood density in the 100%,
 1630 FP and 0% treatments, respectively, in the first 2 years post-harvest compared with the pre-
 1631 harvest period and after controlling for temporal changes in the references. Five years post-
 1632 treatment the mean LW density in the FP continued to increase 42% (P = 0.08), and again 8 years
 1633 post-treatment (41%; P = 0.09).

1634 Ehinger et al. (2021) also quantified changes in in-stream LW following similar riparian harvest
 1635 prescription. Because of unstable slopes, total buffer area was 18 to 163% greater than the
 1636 prescribed 50-foot-buffer. This resulted in 2 different buffer types 1) buffers encompassing the
 1637 full width (50 feet), 2) <50ft buffers, and 3) unbuffered, harvested to the edge of the channel.
 1638 Because of the separation into multiple treatments, sample sizes became small and unbalanced.
 1639 Thus, no statistical analyses were conducted, and only descriptive statistics were applied for
 1640 changes in stand structure and wood loading. However, given the lack of studies presenting
 1641 changes in LW recruitment from pre- to post-harvest, it is presented here for comparison. Results
 1642 showed the full buffer sites and <50 ft buffer sites received an average of 23 and 10 pieces/100 m

1643 and 2.3 and 0.7 m³/100 m of large wood, respectively, post-harvest. The majority of recruited
 1644 large wood pieces had stems with roots attached (SWRW); 70, and 100% in the full buffer, and
 1645 <50 ft buffer types, respectively. Pre-harvest channel large wood loading ranged from 55.8 to 111
 1646 pieces/100 m and from 9.8 to 25.2 m³/100 m among buffer types. Piece counts increased in the
 1647 full buffer and unbuffered sites (8 and 13%, respectively), and decreased in the <50 ft buffers
 1648 (15%).

1649 *Sediment*

1650 Table 9. Treatment and responses for selected publications investigating Sediment relevant to
 1651 Q1.

Reference	Treatment	Response
Hatten et al. (2018)	Buffer width of 15 m (~ 50 feet), Oregon Forest practices	<u>1 year post-harvest after 2 harvest events (n = 3)</u> Mean suspended sediment concentrations (SSC) was 32 mg L ⁻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	<u>3 years post-harvest</u> The first year following harvest suspended sediment yield (SSY) increased in the unbuffered (n = 2) and buffered (n = 1) catchments. By year 2, SSY returned to pre-harvest levels in the buffered, and one of the unbuffered catchments. In one unbuffered catchment, SSY continued to increase annual for all three years.
Karwan et al. (2007)	Buffer width of 75 foot (Idaho Forest Practices Act) with (1) upland clearcut and (2) 50% canopy removal	<u>4 years post-harvest</u> Total suspended sediment (TSS) load from the clearcut exceeded the predicted load by 152% (6,791 kg km ⁻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.

1652

1653 No studies from Washington published since 2000 provide changes in sediment concentration or
 1654 transport from pre- to post-harvest. The Hard Rock study (McIntyre et al., 2021) reported their
 1655 results for water turbidity and suspended sediment export (SSE) were stochastic in nature and the
 1656 relationships between SSE export and treatment effects were not strong enough to confidently
 1657 draw conclusions. The lack of SSE in some high discharge events suggests that the basins are
 1658 likely to be supply limited. The Soft Rock study (Ehinger et al., 2021) similarly reported that
 1659 their results for changes in sediment post-harvest were highly variable. The SSE data in the Soft
 1660 Rock study indicated that the marine sedimentary lithologies were more erodible than then
 1661 lithologies sampled in the Hard Rock Study. However, prediction equations could not be
 1662 calculated to predict the response of the treatment sites after harvest. Thus, strong conclusions

1663 about the effectiveness of the Forest Practices harvest prescription rules on discharge and SSE
1664 could not be drawn.

1665 Hatten et al. (2018) compared pre- to post-harvest suspended sediment concentrations (SSC) in a
1666 western Oregon Alsea watershed. Treatments followed contemporary harvesting practices (no
1667 buffer in non-fish-bearing streams with equipment exclusion zones, and a 15 m no-cut-buffer in
1668 fish-bearing streams) resulted in non-significant changes in SSC at all treatment sites.
1669 Surprisingly, in the fish-bearing streams there was a decrease in SSC (~63% and ~55%, after first
1670 and second harvest, respectively) compared to pre-harvest values. Bywater-Reyes et al. (2017)
1671 compared pre- to post-harvest changes in suspended sediment yield (SSY) following harvest in
1672 the Trask River Watershed of western Oregon. Harvest treatments of study sub-watersheds
1673 consisted of clearcuts (UM2 and GC3) and a clearcut with buffers (50 ft; ~15 m; PH4).
1674 Following timber harvest, (water year 2013), increases in SSY occurred in all harvested
1675 catchments. The SSY in both PH4 (clearcut with buffers) and GC3 (clearcut without buffers)
1676 declined to pre-harvest levels by water year 2014. Interestingly, the SSY in UM2 (clearcut
1677 without buffers) increased annually throughout the post-harvest period, ultimately resulting in
1678 the highest SSY of all catchments during the final two years (2015-2016) of the study after
1679 producing the lowest SSY in the pre-harvest period. Actual values for SSY and significance were
1680 not reported.

1681 Karwan et al. (2007) compared changes in total suspended solids (TSS) in streams from pre- to
1682 post-harvest in northern Idaho. Treatments in the paired-watershed experiment consisted of 1)
1683 commercial clearcut of the watershed area of 50%, and was broadcast burned and replanted by
1684 the end of May 2003, and 2) partial cut in which a target of 50% the canopy was removed in 50%
1685 of the watershed in 2001, with final 10% of log processing and hauling in early summer of 2002.
1686 All harvests were carried out according to best management practices and in accordance with the
1687 Idaho Forest Practices Act. Results showed a significant and immediate impact of harvest on
1688 monthly sediment loads in the clear-cut watershed ($p = 0.00011$), and a marginally significant
1689 impact of harvest on monthly sediment loads in the partial cut ($p = 0.081$). Total sediment load
1690 from the clearcut over the immediate harvest interval exceeded predicted load by 152% (6,791
1691 kg km⁻²); however, individual monthly loads varied around this amount. The largest increases in
1692 percentage and magnitude occurred during snowmelt months, namely April 2002 (560%, 2,958
1693 kg km⁻²) and May 2002 (171%, 3,394 kg km⁻²). Neither treatment showed a statistical
1694 difference in TSS during the recovery time 2-4 years after harvest (clearcut: $p = 0.2336$; partial
1695 cut: $p = 0.1739$) compared to the calibration loads (pre-harvest).

1696 *Nutrients*

1697 Table 9. Treatment and responses for selected publications investigating Sediment relevant to
1698 Q1.

Reference	Treatment	Response
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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	<u>4 years post-harvest</u> Significant increases in nitrogen were observed in the clearcut (n = 1) and partial cut treatments (n = 1). Increases at the clearcut site was greatest from 0.06 mg-N L ⁻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)	<u>6 years post-harvest (Phase I), 8 years post-harvest (Phase II); (n = 7)</u> 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.

1699

1700 The “Hard Rock” study (McIntyre et al., 2021) results showed an increase in total-N export of
1701 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,
1702 and 0% treatments, respectively, in the first 2 years; and of 6.20 (P = 0.095), 5.34 (P = 0.147),
1703 and 8.49 (P = 0.026) kg/ha/yr in the extended period (7-8 years post-harvest). Results for nitrate-
1704 N export showed changes similar to but slightly less than those seen in the total-N analysis with
1705 a relative increase in nitrate-N export of 4.79 (P = 0.123), 9.63 (P = 0.004), and 14.41 (P <0.001)
1706 kg/ha/yr post-harvest in the 100%, FP, and 0% treatments, respectively in the first 2 years. None
1707 of the changes in the extended period were significant. However, the authors note that there was
1708 high variability in the data for the extended period and nitrate-N export only returned to pre-
1709 harvest levels in one watershed. Total phosphorus export increased post-harvest by a similar
1710 magnitude in all treatments: 0.10 (P = 0.006), 0.13 (P = 0.001), and 0.09 (P = 0.010) kg/ha/yr in
1711 the 100%, FP, and 0% treatments, respectively in the first 2 years post-harvest. Changes in
1712 phosphorus were not reported in the extended period.

1713 Gravelle et al. (2009) compared pre- to post changes in NO³ and NO² concentrations in
1714 headwater streams following a clearcut and a partial cut (50% removal of canopy cover) in
1715 northern Idaho. Riparian buffers and leave trees are not required for non-fish bearing headwater

1716 streams in Idaho. Results showed statistically significant increases in NO^3 and NO^2
1717 concentrations following clearcut and partial harvest cuts in headwater streams ($p < 0.001$).
1718 Increases at the clearcut treatment site were greatest, where mean monthly concentrations
1719 increased from 0.06 mg-N L⁻¹ during the calibration period to 0.35 mg-N L⁻¹ in the post-
1720 harvest period. Mean monthly concentrations in the partial cut increased from 0.04 mg-N L⁻¹ in
1721 the pre-harvest period to 0.05 mg-N L⁻¹ in the post-harvest period. No significant changes of
1722 in-stream concentration of any other nutrient recorded (total Kjeldahl nitrogen (TKN), TP, total
1723 ammonia nitrogen (TAN) consisting of unionized (NH_3) and ionized (NH_4^+) ammonia, and
1724 unfiltered orthophosphate (OP)) were found between time periods and treatments.

1725 Deval et al. (2021) compared changes in the same nutrient concentrations in the same area of
1726 northern Idaho but with an additional harvest prescription several years later. For this analysis,
1727 time periods were broken into four distinct phases: 1) pre-disturbance (1992–1997), 2) post-road
1728 (1997–2001), 3) experimental-harvest Phase I (PH-I) (2001–2007), and 4) operational sequential
1729 harvest Phase II (PH-II) when the extent and frequency of harvests increased (2007–2016). PH-I
1730 represents an experimental treatment phase during which harvest activities were experimentally
1731 controlled (only upstream headwater watersheds were harvested and mature vegetation (size or
1732 age threshold for “mature” not reported) removal ranged between 24% and 47%) followed by
1733 site management operations including broadcast burning and replanting. PH-II represents the
1734 post-experimental phase where the study area transitioned to operational treatments that
1735 consisted of additional road construction and timber harvest, with site management operations
1736 including pile burning and competition release herbicide application. During this operational
1737 phase, the mature vegetation (size or age threshold for “mature” not reported) removal in the
1738 upstream watersheds ranged between 36% and 50%. The response in $\text{NO}^3 + \text{NO}^2$ concentrations
1739 was negligible at all treatment sites following the road construction activities. However, $\text{NO}^3 +$
1740 NO^2 concentrations during the PH-I period increased significantly ($p < 0.001$) at all treatment
1741 sites. Similar to the PH-I period, all watersheds experienced significant increases in $\text{NO}^3 + \text{NO}^2$
1742 concentration during the PH-II treatment period ($p < 0.001$). Similar to Gravelle et al. (2009),
1743 significant increases in all other nutrients recorded were not detected.

1744

1745 Focal Question 1a

1746 *1a. What are the effects of thinning (intensity, extent) on the riparian functions, over the short*
1747 *and long-term compared to untreated stands?*

1748 Based on the wording of this question, papers deemed appropriate are those that compare
1749 changes in measurable data indicative of the riparian functions between harvested and
1750 unharvested stands. Further, studies chosen for this question should compare the response of
1751 these functions based on different thinning intensities. Thus, the design of the studies reviewed
1752 for this review should be a BACI or ACI design with results reported for differences between
1753 treatment and reference reaches. Also included are a few simulation modeling experiments that
1754 follow these designs.

1755

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

1756

1757 *Shade*

1758 Table 10. Treatment and responses for selected publications investigating Shade relevant to Q1a.

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

1759

1760 Anderson et. al. (2007) compared changes in canopy cover at stream centers between sites
 1761 adjacent to different riparian zone treatments and an untreated control. This study was conducted
 1762 in young headwater forests of western Oregon. Treatments included three buffer widths: 1) one
 1763 site-potential tree averaging 69 m (B1), 2) variable width buffer averaging 22 m (VB), or 3)
 1764 streamside retention buffer averaging 9 m (SR-T). Adjacent upland to each buffer treatment was
 1765 thinned to ~198 trees per hectare. Results showed that visible sky at stream center only differed
 1766 significantly between SR-T (9.6%) and the untreated (4.2%) sites post-harvest. These results
 1767 were reported for the period 2-5 years post-harvest.

1768 Roon et. al. (2021a) used a BACI analysis to evaluate significant changes in canopy cover
 1769 relative to untreated reaches following 2 different thinning intensities in second growth redwood
 1770 forests of northern California. One study site prescribed treatment on one side of the stream of a
 1771 45 m buffer width with a 22.5 m inner zone with a target 85% canopy retention and a 22.5 m
 1772 outer zone that retained 70% canopy cover (Green Diamond Resource Company, Tectah
 1773 watershed). The treatment site, thinning prescriptions included removal of up to 40% of the basal
 1774 area within the riparian zone on slopes less than 20% on both sides of the channel along a ~100–
 1775 150 m reach (Lost Man watershed, Redwood national park). Control reaches were located

1776 upstream from treatment reaches. Data analysis was conducted separately for each experimental
 1777 watershed (i.e., 1 Lost man site, 2 Tectah sites). Results for the Tectah watershed showed a
 1778 significant reduction in canopy closure by a mean of 18.7%, (95% CI: -21.0, -16.3) and a
 1779 significant reduction of effective shade by a mean of 23.0% (-25.8, -20.1) one-year post
 1780 treatment. In the Lost Man watershed, a non-significant reduction of mean shade by 4.1% (-8.0, -
 1781 0.5), and mean canopy closure by 1.9% was observed. Results for below canopy light availability
 1782 showed significant increases by a mean of 33% (27.3, 38.5) in the Tectah watershed, and non-
 1783 significant increases in Lost Man watershed of 2.5% (-1.6, 5.6). Data for canopy closure and
 1784 effective shade were recorded for 1-year pre- and 1-year post-harvest.

1785 *LW*

1786 Table 10. Treatment and responses for selected publications investigating Large Wood relevant to
 1787 Q1a.

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	<u>Simulated 100-year post harvest results</u> The model output for single entry thinning treatments predicts a 33% or 66% reduction of in-stream wood over a century relative to the unharvested reference for harvest on one side or both sides of the stream, respectively. Double entry thinning treatments without a buffer predicted further reduction in wood recruitment over a century of simulation with 42 and 84% reduction of in stream wood relative to the reference stream when one side and both sides of the channel were harvested.
Schuett Hames and Stewart (2019a)	30-ft no-cut buffer width, and thinning 30-75 ft from the stream (SR); retention of all shade providing trees in this area (AAS)	<u>5-years post-harvest</u> The SR and AAS LW input rates by volume were nearly 300% and 50% higher than the reference stream rates, respectively. 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.

1788

1789 Benda et al. (2016) used simulation modeling to estimate the changes in in-stream LW volume
 1790 over time between sites with thinning treatments and unharvested reference sites. They used
 1791 ORGANON growth models to simulate forest growth and LW recruitment over a 100-year
 1792 period. The model simulated treatments of single entry thinning from below (thinning from
 1793 below removes the smallest trees to simulate suppression mortality) with and without a 10 m
 1794 width no-cut buffers; and a double entry thinning from below with the second thinning occurring
 1795 25 years after the first with and without 10 m no-cut buffers (results with 10 m buffer presented
 1796 in question 1b). Each thinning treatment was also combined with some mechanical introduction
 1797 of thinned trees into the stream encompassing a range between 5 and 20 % of the thinned trees.
 1798 The single-entry thin reduces stand density to 225 tph in 2015 (-67 %) and declines further to
 1799 160 tph by 2110 (-77 %). The double entry thinning resulted in 123 tph after the second thinning
 1800 in 2040 (-82%) and maintained that density until 2110. Both thinning treatments resulted in a

1801 substantial reduction of dead trees that could contribute to in-stream wood loads. The model
1802 output for single entry thinning treatments predicts a 33% or 66% reduction of in-stream wood
1803 over a century relative to the unharvested reference for harvest on one side or both sides of the
1804 stream, respectively. Including mechanical tipping of 5,10,15, and 20% of cut stems without a
1805 buffer in the single-entry thinning treatment changes the relative in-stream percentages of wood
1806 relative to the reference stream to -15, -6, +1, and +6%, respectively. Double entry thinning
1807 treatments without a buffer predicted further reduction in wood recruitment over a century of
1808 simulation with 42 and 84% reduction of in stream wood relative to the reference stream when
1809 one side and both sides of the channel were harvested. To offset the predicted changes of in
1810 stream wood volume following double entry harvest would require tipping of 10% of cut stems.
1811 The authors conclude that thinning without some mitigation efforts resulted in large losses of in
1812 stream wood over a century.

1813 Schuett Hames and Stewart (2019a) compared recruitment rates of LW and volume of in-stream
1814 LW between different riparian buffer thinning treatments and unharvested reference sites.
1815 Treatments evaluated included prescriptions for standard shade rule (a 30-ft no-cut buffer width,
1816 and thinning 30-75 ft from the stream), and all available shade rule (requires retention of all
1817 shade providing trees in this area) for eastern Washington. Results showed cumulative wood
1818 recruitment from tree fall over the five-year post-harvest interval was highest in the standard
1819 shade rule (SR) group, lower in the all-available-shade rule (AAS) group and lowest in the
1820 reference (REF) group. The SR and AAS rates by volume were nearly 300% and 50% higher
1821 than the REF rates, respectively. Wood recruitment in the SR sites was significantly greater than
1822 in the AAS and reference sites ($P < 0.05$). Conversely, differences in wood recruitment did not
1823 differ significantly between the AAS and reference sites.

1824 *Sediment*

1825 Karwan et al. (2007) used BACI analysis to compare changes in total suspended solid (TSS)
1826 yields between thinned sites and unharvested reference sites. This study was conducted in the
1827 Mica Creek Experimental watershed of northern Idaho and focused on non-fish bearing
1828 headwater streams. The thinning treatment included a target 50% canopy removal without no-cut
1829 buffers. Results showed a marginally significant ($P = 0.081$) increase in TSS relative to the
1830 reference streams in the first year following treatment. However, differences in TSS between the
1831 treatment streams and reference streams were not significant ($p = 0.174$) in the period 2-4 years
1832 post-harvest.

1833 *Nutrients*

1834 Yang et al. (2021) compared changes in stream chemistry between streams along thinned stands
1835 and unharvested reference stands in young mixed conifer headwater basins of the Sierra National
1836 Forest. Thinning treatment included mastication of shrub cover to $< 10\%$ and harvesting of trees
1837 to a target basal area of $27\text{--}55 \text{ m}^2 \text{ ha}^{-1}$. Data for dissolved organic carbon (DOC) and dissolved
1838 organic nitrogen (DON) were recorded for 2 years prior to and 3 years after treatment. For
1839 stream water, volume-weighted concentrations of DOC were 66- 94% higher in thinned
1840 watersheds than in control watersheds for all three consecutive drought years following thinning
1841 ($p = 0.06, 0.01, \text{ and } 0.05$ for years 1,2, and 3 post-harvest, respectively). No differences in DOC

1842 concentrations were found between thinned and control watersheds before thinning ($p = 0.50$,
 1843 and 0.74 for pre-harvest years 1 and 2, respectively). Volume-weighted concentrations of DIN
 1844 were 24% higher in thinned than in control watersheds only in the third year following thinning
 1845 ($p = 0.04$). No differences in DIN were detected between treatment and reference streams in the
 1846 2 pre-harvest years ($P \geq 0.44$). Note: Drought occurred at both sites during the three post-harvest
 1847 years which may have compounded these effects. This is discussed in more detail in question 3.

1848

1849 **Focal Question 1b**

1850 *Ib. How do buffer widths and adjacent upland timber harvest prescriptions influence impacts of*
 1851 *riparian thinning treatments?*

1852 An experimental design that could provide information useful in answering this question would
 1853 involve a comparison of sites with different buffer widths, all with upland harvest, and data
 1854 would need to be recorded before and after thinning, with or without a control site (BAI, BACI),
 1855 or differences after thinning between treatment and control sites (ACI). Three papers include an
 1856 experimental design that investigate different buffer widths or different upland treatments along
 1857 with riparian thinning treatments.

1858 *Shade*

1859 Anderson et al. (2007) compared changes in canopy cover at stream centers between sites
 1860 adjacent to different riparian zone treatments and an untreated control. This study was conducted
 1861 in young headwater forests of western Oregon. Treatments included three buffer widths (1) one
 1862 site-potential tree averaging 69 m (B1), (2) variable width buffer averaging 22 m (VB), or (3)
 1863 streamside retention buffer averaging 9 m (SR-T); the adjacent upland to each buffer was thinned
 1864 to ~198 trees per hectare. Results showed that visible sky at stream center only differed
 1865 significantly between SR-T (9.6%) and the untreated (4.2%) sites post-harvest. These results
 1866 were reported for the period 2-5 years post-harvest.

1867 *LW*

1868 Table 11. Treatment and responses for selected publications investigating Large Wood relevant to
 1869 Q1b.

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

Benda et al. (2016)	simulation modeling of thinning from below with and without a 10 m width no-cut buffers;	<u>Simulated 100-year post harvest results</u> Adding a 10 m buffer reduced total reduction of in stream wood to 11 and 22% for thinning on one and both sides of the channel, respectively, from the predicted 42 and 84% reduction without the 10 m buffer.
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1870

1871 Burton et al. (2016) examined the relationship between annual in-stream wood loading and
 1872 riparian buffer widths adjacent to upland thinning operations. Buffer widths were 6, 15, or 70
 1873 meters and upland thinning was to 200 trees per ha (tph), with a second thinning (~10 years later)
 1874 to ~85 tph, alongside an unthinned reference stand of ~400 tph. Their results showed that slightly
 1875 higher volumes of wood were found in sites with a narrow 6-m buffer (not significant), as
 1876 compared with the 15-m and 70-m buffer sites in the first 5 years after the first harvest and
 1877 maintained through year 1 of the second harvest (end of study). The authors attributed this
 1878 difference to a higher likelihood of logging debris and/or windthrow, but these factors were not
 1879 analyzed.

1880 Benda et al. (2016) used simulation modeling to estimate the changes in in-stream LW volume
 1881 over time between sites with thinning treatments and unharvested reference sites. They used
 1882 ORGANON growth models to simulate forest growth and LW recruitment over a 100-year
 1883 period. The model simulated treatments of single entry thinning from below (thinning from
 1884 below removes the smallest trees to simulate suppression mortality) with and without a 10 m
 1885 width no-cut buffers; and a double entry thinning from below with the second thinning occurring
 1886 25 years after the first with and without 10 m no-cut buffers. Each thinning treatment was also
 1887 combined with some mechanical introduction of thinned trees into the stream encompassing a
 1888 range between 5 and 20 % of the thinned trees. The single-entry thin reduces stand density to 225
 1889 tph in 2015 (-67 %) and declines further to 160 tph by 2110 (-77 %). The double entry thinning
 1890 resulted in 123 tph after the second thinning in 2040 (-82%) and maintained that density until
 1891 2110. Both thinning treatments resulted in a substantial reduction of dead trees that could
 1892 contribute to in-stream. The model output for single entry thinning treatments predicts a 33% or
 1893 66% reduction of in-stream wood over a century relative to the unharvested reference for harvest
 1894 on one side or both sides of the stream, respectively. Adding the 10-m no cut buffer reduced total
 1895 loss to 7 and 14%. Including mechanical tipping of 5,10,15, and 20% of cut stems without a
 1896 buffer in the single-entry, thinning treatment changed the relative in-stream percentages of wood
 1897 relative to the reference stream to -15, -6, +1, and +6%, respectively. To completely offset the
 1898 loss of in stream wood due to single entry thinning, mechanical tipping of 14 and 12% were
 1899 required without and with buffers. Double entry thinning treatments without a buffer predicted
 1900 further reduction in wood recruitment over a century of simulation with 42 and 84% reduction of
 1901 in stream wood relative to the reference stream when one side and both sides of the channel were
 1902 harvested. Adding a 10 m buffer reduced total reduction of in stream wood to 11 and 22% for
 1903 thinning on one and both sides of the channel. To offset the predicted changes of in stream wood
 1904 volume following double entry harvest would require tipping of 10 and 7% of cut stems without

1905 and with the 10-m buffer. The authors conclude that thinning without some mitigation efforts
1906 resulted in large losses of in stream wood over a century.

1907

1908 [Focal Question 1c](#)

1909 *1c. What are the effects of clearcut gaps in riparian stands (intensity, extent) on the riparian*
1910 *functions, over the short and long-term, compared to untreated stands?*

1911 This question uses the general term “clearcut gaps” as a treatment within the riparian area but
1912 does not define a minimum or maximum threshold for gap size. Thus, studies reviewed that used
1913 a “patch” treatment were included as having information useful in answering this question. The
1914 question also identifies a comparison with untreated stands. Therefore, any design with a control
1915 site (BACI, ACI) is appropriate.

1916 There appears to be a paucity of studies in the literature that investigate the effects of gaps or
1917 patch harvesting treatments on riparian function within riparian stands. Only 4 papers discussed
1918 the effects of prescribed gaps or patches in the riparian area on riparian function.

1919 The “Hard Rock” study from McIntyre et al. (2021) and the “Soft Rock” study from Ehinger et
1920 al. (2021) present the most relevant results useful for answering this question. Riparian buffer
1921 prescriptions for non-fish bearing streams in western Washington use a gap design. In this
1922 design, a 50-foot buffer is required along at least 50% of the treated stream length. The
1923 remaining 50% or less of the treated riparian management zone can be clear cut to the stream
1924 edge. The Hard Rock study compared differences in shade, in-stream sediment and nutrient
1925 concentrations, and large wood recruitment between treated and unharvested reaches for 8-9
1926 years post-harvest. The first iteration of the Hard Rock study (McIntyre et al. 2021) also
1927 compared differences in litter inputs following treatment for 2 years post-harvest between
1928 treatment and reference reaches.

1929 The Soft Rock study compared differences in the same functions between treated and
1930 unharvested reaches, with 3-6 years of post-harvest sampling depending on the function under
1931 investigation. However, because of unstable slopes in some of the sites in the Soft Rock study,
1932 many of the buffers were required to be wider than 50-feet (ranging from 18 –160% wider than
1933 50-feet). Conversely, some of the sites treated ended up with buffers narrower than 50 feet.
1934 Further, there was limited availability of sites that fit the criteria (marine sediment lithology,
1935 timing of treatment). Because of these limitations, statistical analysis, and comparison of
1936 response between treatments and references for stream temperature and shade could not be
1937 performed. However, descriptive statistics were provided that contain useful information. Results
1938 from formal statistical analyses are provided for all other functions.

1939 *Shade*

1940 Table 12. Treatment and responses for selected publications investigating Shade relevant to Q1c.

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Reference	Treatment	Response
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McIntyre et al. (2021)	Buffer width of 50 feet (100%); 50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)	9-years post-harvest Effective shade results showed decreases of 11, 36, and 74 percent in the 100%, FP, and 0% treatments, respectively. Results for canopy cover showed that riparian cover declined after harvest in all buffer treatments reaching a minimum around 4 years post-harvest (after mortality stabilized).
Janisch et al. (2012)	Patched buffer: clearcut to stream with ~50-110 m patches retained; continuous buffer 10-15 m	<u>1-year post-harvest</u> After treatment, canopy cover in the clearcut catchments averaged 53%, canopy cover in the patch buffer treatment averaged 76%, and canopy cover in the continuous buffer treatment averaged 86%. The canopy cover of the clearcut and patch buffer treatments were significantly lower than in the reference streams.
Swartz et al. (2020)	20 m diameter clearcut gaps over stream at 30 m intervals.	<u>1-year post-harvest (n = 6)</u> Post-harvest significant increase in mean reach light to a mean of 3.91 (SD \pm 1.63) moles of photons $m^{-2} day^{-1}$, overall resulting in a mean change in light of 2.93 (SD \pm 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 \pm 0.02%).

1942

1943 The Hard Rock study reported significant decreases in canopy cover (measured at 1 meter above
 1944 the stream surface with a spherical densiometer) for all treated sites immediately following
 1945 harvest compared to the reference sites ($p < 0.05$). The mean canopy cover decreased from 96%
 1946 (pre-harvest) to 72% in the first-year post-harvest and continued to decline for four years
 1947 reaching a minimum of 54%. After year four, mean canopy cover began to recover increasing
 1948 annually until year 9 to 74%. In contrast, mean canopy cover in the reference sites was 95%
 1949 before harvest and never fell below 85% for 9 years. In the Soft Rock study, mean canopy
 1950 closure decreased in the treatment sites from 97% in the pre-harvest period to 75%, 68%, and
 1951 69% in the first, second, and third post-harvest years, respectively; and was further related to the
 1952 proportion of stream buffered and to post-harvest windthrow within the buffer. Canopy closure
 1953 remained stable in the reference sites throughout the course of the study, ranging from 95 to
 1954 99%.

1955 Janisch et al. (2012) compared canopy cover before and after application of a “patched buffer”
 1956 treatment with unharvested control reaches in headwater streams of western Washington. The
 1957 “patched buffer” treatment included retention of portions of the riparian forests ~50-110 m long
 1958 in distinct patches along the channel with the remaining riparian area clearcut. There was no
 1959 standard width for patched buffers, with buffers spanning the full width of the floodplain area
 1960 and/or extending some undefined distance away from the stream. Canopy density was measured
 1961 once in the summer prior to logging and once in the summer following logging. The percentage
 1962 of visible sky was determined from digital photos taken with a fish-eye lens using Hemiview
 1963 Canopy Analysis software. Canopy cover in all streams averaged 95% prior to harvest and did
 1964 not differ between treatment and reference streams. Following treatment, canopy cover in the
 1965 patch buffer treatment averaged 76% and differed significantly from reference reaches.

1966 Swartz et al. (2020) tested the effects of adding canopy gaps within young (40 – 60 years old),
1967 regenerating forests of western Oregon on stream light availability and stream temperatures.
1968 While light availability and stream temperature are not functions described in the FPHCP, they
1969 are directly related to shade (an FPHCP function) Also, they directly affect water quality and
1970 aquatic habitat productivity which are functional objectives within the FPHCP. Further,
1971 considering the paucity of studies available that investigate the effects of clearcut gaps, the
1972 results are presented here. The addition of gaps in the young regenerating forests were used to
1973 theoretically mimic the natural disturbance regimes and the higher canopy complexity of late-
1974 successional forests. The researchers used a BACI design on six replicated streams within the
1975 McKenzie River Basin. In each treatment reach, gaps were designed to create openings in the
1976 canopy that were approximately 20 m in diameter. Gaps were centered on a tree next to the
1977 stream and spaced approximately 30 meters apart along each reach. The BACI analysis showed
1978 strong evidence for significant increase in mean reach light ($p < 0.01$) up to 3.91 (SD \pm 1.63)
1979 moles of photons $m^{-2} day^{-1}$ and an overall mean change in light of 2.93 (SD \pm 1.50) moles of
1980 photons $m^{-2} day^{-1}$. Mean stream shading could not be evaluated in the full BACI analysis
1981 because post-treatment hemispherical photographs could not be taken at all sites due to fire
1982 impeding access. For the remaining sites, the areas beneath each gap had notable localized
1983 declines in shade, though the entirety of the treatment reach mean shading declined by only 4%
1984 (SD \pm 0.02%).

1985 *Litter*

1986 The Hard Rock study only quantified changes in litter input for 2 years after treatment (McIntyre
1987 et al., 2018). While significant decreases in litter input were observed from pre- to post-harvest
1988 in the treatment sites (described in focal question 1) these values were not significant when
1989 compared to the changes in the reference sites. Litter input was not quantified in the Soft Rock
1990 study.

1991 *LW*

1992 For the Hard Rock study, large wood recruitment and loading were only compared between the
1993 reference reaches and the buffered portion of the treatment reaches. The authors report large
1994 wood recruitment into the channel was 3 times greater on average in the treatment buffer than in
1995 the reference over the 8-year post-treatment period. However, while considerable, these
1996 differences were not significant for any analyzed post-harvest interval (e.g., 1-2 years post, 1-5
1997 years post, or 1-8 years post). The lack of significance was attributed to the large variability in
1998 recruitment values among treatment sites. The greatest increase in LW recruitment in the
1999 treatment sites relative to the reference sites occurred in the first 2 years post-harvest. Large
2000 wood loading (pieces/m of channel length) increased significantly ($\alpha = 0.10$) in the treatment
2001 reaches, relative to the reference sites in the first 2 years (47%; $p = 0.05$), 5 years (42%; $p =$
2002 0.08), and 8 years (41%; $p = 0.09$) post-harvest. For the Soft Rock study there was little post-
2003 harvest large wood input in reference sites: an average of 4.3 pieces and 0.34 m³ of combined in-
2004 and over-channel volume per 100 m of channel. In contrast, the full buffer sites and <50 ft buffer
2005 sites received an average of 23 and 10 pieces/100 m and 2.3 and 0.7 m³/100 m of large wood,
2006 respectively.

2007 *Sediment*

2008 For the Hard Rock study, results for water turbidity and suspended sediment export (SSE) were
2009 stochastic in nature and the relationships between SSE and treatment effects were not strong
2010 enough to confidently draw conclusions. Water turbidity and SSE increased with stream
2011 discharge during large storm events but rapidly declined. The Soft Rock study reported similar
2012 issues with the data for SSE in that it appeared to be driven by site and event specific factors and
2013 strong conclusions could not be drawn. The authors report that the softer lithologies sampled as
2014 part of this study were more erodible than the competent lithologies sampled in the companion
2015 Hard Rock Study.

2016 *Nutrients*

2017 The Hard Rock study analyzed changes in total nitrogen and nitrate export in the gap buffers
2018 relative to untreated reference streams. Results showed an increase in total nitrogen export in the
2019 treatment sites of 10.85 kg/ha/yr ($p = 0.006$) in the first two years post-harvest relative to the
2020 reference sites. In the extended periods, total nitrogen export increased by 5.34 ($p = 0.147$)
2021 kg/ha/yr relative to the reference streams. Results for NO^3 export showed similar but slightly
2022 lower increases than total nitrogen with a relative increase in NO^3 export of 9.63 ($p = 0.004$)
2023 kg/ha/yr for the first two years post-harvest relative to the reference. None of the changes in
2024 nitrate exports in the extended period were significant. The Soft Rock study reported significant
2025 increases in concentrations of total nitrogen ($p < 0.05$) and NO^3 ($p < 0.05$) post-harvest in the
2026 treatment sites relative to the reference sites. The change in export appeared related to the
2027 proportion of stream buffered.

2028

2029 [Focal Question 1d](#)

2030 *1d. How do buffer widths and upland timber harvest influence impacts of clearcut gaps*
2031 *treatments?*

2032 The wording of this question implies that the effects of clearcut gaps (discussed in focal question
2033 1c) on riparian function could be impacted when paired with different buffer widths and upland
2034 harvest prescriptions. Similar to the results of the search in literature for focal question 1c, there
2035 was a paucity of riparian function studies that implemented a clearcut gap or patch cutting
2036 method within the riparian area. The added layer of complexity in this question specifying
2037 differences in buffer widths and upland harvests only further refined the selection of appropriate
2038 papers. Of the studies reviewed above, none included the evaluation of different buffer widths or
2039 different upland harvests in their experimental design. The Hard Rock study compared the
2040 clearcut gap buffers to full retention buffer and unbuffered sites (discussed in the literature
2041 review section), but different widths were not compared in the gap buffer treatments.

2042

2043 [Focal Question 1e](#)

2044 *1e. What are the effects of any combinations of the above treatments?*

2045 No studies found in our search compared the effects of combined treatments on one or more of
 2046 the five functions, likely because combining multiple treatments into one design has the potential
 2047 to confound results and are difficult to implement with sufficient sample sizes. The majority of
 2048 the studies listed in our review investigate the effects of buffer width, thinning treatments, and
 2049 upland treatments separately.

2050 The only papers with some extractable evidence of the compounding/ameliorating effects of
 2051 combined treatments were focused on shade. One study, Reiter et al. (2020), compared the
 2052 effects of thinned and unthinned buffers, and clearcut on changes in percent shade over adjacent
 2053 streams (discussed in focal question 1). However, changes in shade were not statistically
 2054 analyzed and the implementation of the upland thinning treatment only occurred at one site
 2055 (Table 6).

2056

2057 [Focal Question 2](#)

2058 *2. How and to what degree do specific site conditions (e.g., topography, channel width and*
 2059 *orientation, riparian stand age and composition) influence the response of the riparian*
 2060 *functions?*

2061 Multiple studies have investigated the influences of site conditions on riparian function. Few
 2062 studies reviewed (4) investigated the interaction between specific site conditions (e.g., slope,
 2063 lithology, elevation) and harvest on the response of riparian function. However, if these specific
 2064 site conditions influence the magnitude of riparian function in the absence of harvest, it is
 2065 possible they can compound the effects of harvest on their response. Thus, studies that assess the
 2066 relationship between site factors and riparian function may provide some useful insight for
 2067 management and are presented below. Further, we also included studies that investigated the
 2068 relationships between road development and sediment transport because road development is
 2069 directly related to changes in local topography.

2070 *Litter*

2071 Table 13. Treatment and responses for selected publications investigating Litter relevant to Q2.

Reference	Treatment	Response
Hart et al. (2013)	Remove plants in a 5 x 8 m section adjacent to stream < 10 cm DBH and >12 cm height every 2 months. 5 m fence extending underground and parallel to the stream	<u>1-2 years post-treatment (n = 5)</u> . Deciduous-site vertical litter input (504 g m ⁻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 (R ² = 0.4073) but showed no strong relationship at coniferous sites (R ² = 0.1863).

Bilby & Heffner (2016)	Simulation modeling and field sampling	<u>1-year of litterfall data</u> the majority of the litter recruited into the stream originated from within 10 m of the stream regardless of litter or stand type. No difference was found in delivery distance and litter type (needles or broadleaf) at young sites (ages not specified; canopy height mean = 32.4 m). However, needles released at mature (canopy height mean = 47 m) sites had a higher proportion of cumulative input from greater distances than needles or alder leaves released at younger sites. Litter travel distance was linearly related to wind speed ($p < 0.0001$). Doubling wind speed at one site led to a 67-87% expansion of the riparian contribution zone in the study area.
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2072

2073 Hart et al. (2013) compared litter delivery into streams between riparian zones dominated by
 2074 deciduous (red alder) and coniferous (Douglas-fir) tree species in western Oregon. Results from
 2075 this study show that deciduous forests dominated by red alder delivered significantly greater
 2076 vertical and lateral inputs ($\text{g m}^{-2} \text{y}^{-1}$) to adjacent streams than did coniferous forests dominated by
 2077 Douglas-fir. Deciduous-site vertical litter input (mean = $504 \text{ g m}^{-2} \text{y}^{-1}$) exceeded that from
 2078 coniferous sites ($394 \text{ g m}^{-2} \text{y}^{-1}$) by 110 g/m^2 over the full year. Annual lateral inputs at
 2079 deciduous sites ($109 \text{ g m}^{-2} \text{y}^{-1}$) were $46 \text{ g m}^{-2} \text{y}^{-1}$ more than at coniferous sites ($63 \text{ g m}^{-2} \text{y}^{-1}$).
 2080 The timing of the inputs also differed, with the greatest differences occurring in November
 2081 during autumn peak inputs for the deciduous forests. Further, annual lateral litter input increased
 2082 with slope at deciduous sites ($R^2 = 0.4073$, $p = 0.0771$), but showed no strong relationship at
 2083 coniferous sites ($R^2 = 0.1863$, $p = 0.2855$). These results were partially consistent with Bilby &
 2084 Heffner (2016) in that they suggest litter type, and topography (slope) can affect the litter input
 2085 rates.

2086 Bilby & Heffner (2016) used a combination of field experiments, literature review, and modeling
 2087 to estimate the relative importance of factors affecting litter delivery from riparian areas into
 2088 streams of western Washington in the Cascade mountains at high and low elevations. Their
 2089 results for conifer needles released at mature sites had a higher proportion of cumulative input
 2090 from greater distances than needles or leaves released at younger sites. The authors suggest from
 2091 their interpretation of the model that the width of the litter contributing area was ~35% greater at
 2092 mature sites than at young sites. The mean age of “mature” and “young” sites was not specified
 2093 but the mean tree heights were 47.0 m and 32.4 m for the mature and young sites, respectively.
 2094 Thus, tree height is related to the width of the litter contributing area for conifer needles. Litter
 2095 travel distance was also linearly related to wind speed ($p < 0.0001$). Doubling wind speed at one
 2096 site led to a 67-87% expansion of the riparian litter contribution zone in the study area.
 2097 Interpretation of the regression curves revealed a trend that suggests hillslope gradient affects the
 2098 width of the litter contributing area as well. However, the authors did not apply statistical
 2099 analysis to these values and only speculated that increasing the slope from 0-45% would increase
 2100 the width of the litter contributing area by up to 70%.

2101 *LW*

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2103

2104 Table 14. Treatment and responses for selected publications investigating Large Wood relevant to
 2105 Q2.

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

2106

2107 Wing & Skaugset (2002) investigated the relationships between channel and habitat
 2108 characteristics with LW piece count and volume in stream reaches in western Oregon. This study
 2109 analyzed an extensive spatial database of aquatic habitat conditions created for western Oregon
 2110 using stream habitat classification techniques and a geographic information system (GIS).
 2111 Regression tree analysis (an exploratory regression analysis that allows for the inclusion of
 2112 multiple explanatory variables) was used to compare the relative strength of each variable in
 2113 predicting LW volume. Explanatory variables used in this analysis included morphology of
 2114 active channel (hillslope, terrace, terrace hillslope, unconstrained), and lithology (e.g., alluvium,
 2115 basalt, etc.). Results for channel characteristics showed that stream gradient was the most
 2116 important explanatory variable for LW volume. The split for stream gradient occurred for reaches
 2117 with < 2.3% gradient (mean LW volume: 5.8 m³ per reach) while higher gradient streams showed
 2118 a mean LW volume of 17.9 m³ per reach.

2119

2120 For LW pieces in forested stream reaches bankfull channel width was the most important
 2121 explanatory variable with the split occurring for streams channels less than 12.2 m wide. LW
 2122 pieces for streams <12.2 m wide averaged 11.1 LW pieces per reach while larger channels
 2123 averaged 4.9 pieces per reach; in this model the BFW split explained 7% of the variation in LW
 2124 pieces found in forested streams. For key LW pieces (logs at least 0.60 m in diameter and 10 m
 2125 long) in forested reaches, stream gradient was again the most important explanatory variable

2126 with the split occurring at a slope of 4.9%. The streams with a gradient < 4.9% averaged 0.5 key
 2127 LW pieces per reach while streams with higher gradients averaged 0.9 key LW pieces per reach;
 2128 in this model stream gradient explained 8% of the variation in key LW pieces found in streams.
 2129

2130 Lithology caused second, third or fourth level splits after stream gradient or BFW. Specifically,
 2131 Mesozoic sedimentary and metamorphic geologies, located in southern Oregon stream reaches,
 2132 were grouped and split from basalt, Cascade, and marine sedimentary geologies. In stream
 2133 reaches with Mesozoic sedimentary and metamorphic geologies, the quantity of LWD was
 2134 roughly half the amount found in other geologies. The only exception to this grouping was for
 2135 LW volume in larger stream reaches, where basalt and marine sedimentary geologies contained
 2136 more LW volume when grouped separately from all other geologies in a fourth-level split. The
 2137 authors conclude that the geomorphic characteristic of stream reaches, in particular stream
 2138 gradient and bankfull width, correlated best with LW presence.
 2139

2140 Sobota et al. (2006), evaluated patterns of riparian tree fall directions in diverse environmental
 2141 conditions and evaluate correlations with tree characteristics, forest structural variables, and
 2142 topographic features. Specifically, the authors were interested in correlations between fall
 2143 directionality and tree species type, tree size, riparian forest structure, and valley topography
 2144 (side slope). Data was collected from 21 field sites located west of the Cascade Mountains crest
 2145 (11 sites: Coast Range and west slopes of the Cascades), and in the interior Columbia Basin (10
 2146 sites: east slopes of the Cascades, Blue Mountains, and Northern Rockies) of Oregon,
 2147 Washington, Idaho, and Montana, USA. Streams were second- to fourth-order channels and had
 2148 riparian forests that were approximately 40 to >200 years old. Model projections of LW
 2149 recruitment estimated that sites with uniform steep side slopes (>40%) produced between 1.5 to
 2150 2.4 times more in stream LW by number of tree boles than sites with uniform moderate side
 2151 slopes (< 40%). The authors warn that while side slope categories (>40%, <40%) was the
 2152 strongest predictor of tree fall direction in this study, they believe the differences in tree fall
 2153 direction between these categories mainly characterized differences between fluvial (88% of
 2154 moderate slope sites) and hillslope landforms (71% of steep slope sites). They suggest that the
 2155 implications from this study are most applicable to small- to medium-size streams (second to
 2156 fourth order) in mountainous regions where sustained large wood recruitment from riparian
 2157 forest mortality is the significant management concern.

2158 *Sediment*

2159 Table 15. Treatment and responses for selected publications investigating Sediment relevant to
 2160 Q2.

Reference	Treatment	Response
Bywater-Reyes et al. (2017)	basin lithology and physiography effects on sediment delivery	<u>6 years of data from the Trask River Watershed</u> 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

Bywater-Reyes et al. (2018)	catchment lithography, physiography, discharge, and disturbance history effects on sediment delivery	<u>60 years of data in the H.J. Andrews experimental watershed (n = 10)</u> Watershed 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 \geq 30-year return interval .
Mueller & Pitlick (2013)	correlation analysis to assess the relative impact of lithology, basin relief, mean basin slope, and drainage density on in stream sediment supply.	<u>Data sets ranging 1-96 years for 83 basins</u> the strongest correlation of bankfull sediment concentration was with 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.
Litschert & MacDonald (2009)	Post-harvest stream sediment delivery pathway development frequency and characteristics.	<u>1-year post-harvest data (n = 200 harvest units)</u> 19 harvest units developed sediment delivery pathways. Pathway length and probability of connecting to stream was significantly correlated with mean annual precipitation, cosine of the aspect, elevation, and hillslope gradient .

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Bywater-Reyes et al. (2017) assessed the influence of natural controls (basin lithology and physiography) and forest management on suspended sediment yields in temperate headwater catchments. This study analyzed 6 years of data from the Trask River Watershed in northeastern Oregon and included data from harvested and unharvested sub-catchments underlain by heterogeneous lithologies. Results from this study indicate that site lithology was the 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 ($r = 0.9$) and displayed an exponential relationship when plotted against the percentage of watershed area underlain by these lithologies. In contrast, site lithology had a strong negative correlation with percent area underlain by diabase ($r = 0.7$), with the lowest SSY associated with 100% diabase. Following timber harvest, increases in SSY occurred in all harvested catchments but returned to pre-harvest levels within 1 year except for sites that were underlain by sedimentary formations and were clearcut without protective buffers. The authors conclude that sites underlain with a friable lithology (e.g., sedimentary formations) had on average, SSYs an order of magnitude higher following harvest than those on more resistant lithologies (intrusive rocks).

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

2186 and index of sediment connectivity) while accounting for site. To account for the effect of
2187 disturbance history a variable was added to the model when the watershed had a history of
2188 management or natural disturbances. If the models for the disturbed watersheds significantly
2189 underpredicted the sediment discharge, the timing of the sudden increases were further examined
2190 to assess whether it correlated with a disturbance event. The results showed that watershed
2191 physiography combined with cumulative annual discharge explained 67% of the variation in
2192 annual sediment yield across the 60-year data set. Relative to other physiographic variables,
2193 watershed slope was the greatest predictor of annual suspended sediment yield. However, the
2194 results showed that annual sediment yields also moderately correlated with many other
2195 physiographic variables and caution that the strong relationship with watershed slope is likely a
2196 proxy for many processes, encompassing multiple catchment characteristics.

2197 Mueller & Pitlick (2013) used correlation analysis to assess the relative impact of lithology,
2198 basin relief, mean basin slope, and drainage density on in stream sediment supply defined by the
2199 bankfull sediment concentration (bedload and suspended load). The study used sediment
2200 concentration data from 83 drainage basins in Idaho and Wyoming. Lithologies of the study area
2201 were divided into four categories ranging from hardest to softest- granitic, metasedimentary,
2202 volcanic, and sedimentary. The results showed the strongest correlation of bankfull sediment
2203 concentration was with basin lithology, and showed little correlation strength with slope, relief
2204 and drainage density. As lithologies become dominated by softer parent materials (volcanic and
2205 sedimentary rocks), bankfull sediment concentrations increased by as much as 100-fold. The
2206 authors interpret these results as evidence that lithology can be more important in estimating
2207 sediment supply than topography.

2208 Rachels et al. (2020) used sediment source fingerprinting techniques to quantify the proportional
2209 relationship of sediment sources (hillslope, roads, streambanks) in harvested and un-harvested
2210 watersheds of the Oregon Coast Range. The study included one catchment (Enos Creek) that was
2211 partially clearcut harvested in the summer of 2016 and an unharvested reference catchment
2212 (Scheele Creek) located ~3.5 km northwest of Enos Creek. The paired watersheds had similar
2213 road networks, drainage areas, lithologies and topographies. The treatment watershed was
2214 harvested with a skyline buffer technique in the summer of 2016 under the Oregon Forest
2215 practices Act policy that requires a minimum 15 m no-cut buffer. The proportion of suspended
2216 sediment sources were similar in the harvested ($90.3 \pm 3.4\%$ from stream bank; $7.1 \pm 3.1\%$ from
2217 hillslope) and unharvest ($93.1 \pm 1.8\%$ from streambank; $6.9 \pm 1.8\%$ from hillslope) watersheds.
2218 However, the harvested watershed contained a small portion of sediment from roads ($3.6 \pm$
2219 3.6%), while the unharvested reference watershed suspended sediment contained no sediment
2220 sourced from roads. In the harvested watersheds the sediment mass eroded from the general
2221 harvest areas (96.5 ± 57.0 g) was approximately 10 times greater than the amount trapped in the
2222 riparian buffer (9.1 ± 1.9 g), and 4.6 times greater than the amount of sediment collected from
2223 the unharvested hillslope (21.0 ± 3.3 g). These results suggest that the riparian buffer was
2224 efficient in reducing sediment erosion relative to the harvested area. The caveat of this study was
2225 the limited sample size (1 treatment, 1 paired reference watershed) and does not incorporate the
2226 effects of different watershed physiography on sediment erosion. However, it is presented here as

2227 evidence that the formation of roads within a riparian area may interact with timber harvest to
2228 increase the potential flow of sediments from roads.

2229 Litschert & MacDonald, (2009) investigated the frequency of sediment delivery pathways in
2230 riparian management areas and their physical characteristics and connectivity following harvest.
2231 In this study the authors describe sediment delivery pathways (“features”) as rills, gullies, and
2232 sediment plumes that form when excess sediment relative to overland flows transports sediment
2233 from the hillslope to the stream. The authors surveyed 200 riparian management areas (RMA) in
2234 four different National Forests of the Sierra Nevada and Cascade Mountains of California. USFS
2235 policy requires 90-m wide RMA along each side of perennial streams and 45-m wide RMA along
2236 each side of all ephemeral and intermittent streams. When features were found within an RMA,
2237 data for years since harvest, soil depth, soil erodibility (K), feature length, feature gradient,
2238 aspect, elevation, hillslope gradient, hillslope curvature, surface roughness, and connectivity
2239 were recorded for analysis. Association between these variables were analyzed with a
2240 Spearman’s rank correlation. The variables most strongly associated with feature length were
2241 used to develop a multiple linear regression model to predict feature length. Only 19 of the 200
2242 harvest units had sediment development pathways. Feature pathways ranged in age (time since
2243 harvest) from 2 to 18 years, and in length from 10 m to 220 m. Of the 19 feature pathways, only
2244 six were connected to streams, and five of those originated from skid trails. Feature pathway
2245 length was significantly related to mean annual precipitation, cosine of the aspect, elevation, and
2246 hillslope gradient ($R^2 = 64\%$, $p = 0.004$). These results suggest that within treated riparian areas
2247 topographic characteristics such as aspect, elevation and hillslope gradient can affect delivery of
2248 sediment into streams.

2249 Rashin et al. (2006) evaluated the effectiveness of Washington State best management practices
2250 (BMPs) for controlling sediment related water quality impacts. Although this study was
2251 published in 2006, the data analyzed in this study were collected between 1992 and 1995. In their
2252 evaluation, Rashin et al. (2006) assessed site erosion, sediment delivery, channel disturbance,
2253 and aquatic habitat condition within the first two years of harvest along fish- and non-fish
2254 bearing streams across Washington state. From their results, the authors concluded that the site-
2255 specific factors influencing the effectiveness of BMPs in preventing chronic sediment delivery
2256 into streams were 1) the proximity of ground disturbance to the stream, 2) presence of a stream
2257 buffer, 3) falling and yarding practices that minimized disturbance to stream channel, and 4)
2258 timing of harvest activities for certain climate zones where frozen ground or snow cover may be
2259 exploited. The landscape factors that influenced BMP effectiveness were 1) the density (specific
2260 metric not reported) of unbuffered small streams at harvest sites, and 2) steepness of stream
2261 valley slopes. The authors conclude with a recommendation of excluding timber falling and
2262 yarding activities at least 10 m from streams and outside of steep inner gorges.

2263 From the studies reviewed there is evidence that sediment delivery into streams following timber
2264 harvest is influenced by not only the intensity of the harvest operation (e.g., presence of retention
2265 buffers, yarding and equipment use immediately adjacent to the stream, upland clearcut vs.
2266 thinning), but also by physiography (especially hillslope gradient), lithology relative softness,
2267 and the presence of roads. Thus, the change in magnitude of sediment delivery following harvest

2268 is context dependent and these landscape factors can interact with one another to compound
2269 these changes. However, from the studies reviewed in the sediment section of the literature
2270 review, there is evidence that the implementation of BMPs since the 1970s in the northwestern
2271 United States has lessened the impact and duration of these changes.

2272 *Nutrient*

2273 None of the studies published since 2000 and conducted in western North America provide
2274 experimental evidence of the effects of site factors on nutrient flux into streams. However, Zhang
2275 et al. (2010) conducted a global review and meta-analysis of the effectiveness of buffers in
2276 reducing nonpoint source pollution. They reported slope (hillslope gradient) as having a linear
2277 relationship with buffer pollutant removal efficacy that switched from positive to negative when
2278 slope increased beyond 10% (i.e., hillslope gradients of ~10% were optimal for buffer efficacy in
2279 removing pollutants).

2280

2281 *Focal Question 3*

2282 *3. What is the frequency of weather-related effects (e.g., windthrow, ice storms, excessive heat,*
2283 *flood and drought events) on riparian areas? What are the weather-related effects (positive and*
2284 *negative) on the riparian functions, and how are they distinguished from harvest effects? How do*
2285 *these effects differ between treated and untreated riparian forests?*

2286 The first part of this question “What is the frequency of weather-related effects (e.g., windthrow,
2287 ice storms, excessive heat, flood and drought events) on riparian areas?” is a generally worded
2288 question asking how often weather events in riparian areas occur. The second part of this
2289 question “What are the weather-related effects (positive and negative) on the riparian functions,
2290 and how are they distinguished from harvest effects?” contains within it 2 parts 1) what the
2291 effects on the riparian functions are, and 2) how they are distinguished from timber harvest
2292 effect. Any study reviewed that answers one or more parts of this question have been included.

2293 *Shade*

2294 McIntyre et al. (2021), the “Hard Rock” study, compared changes in shade from pre- to post-
2295 harvest between three riparian harvest treatments and a reference. Treatments included a two-
2296 sided 50-ft riparian buffer along at least 50% of the stream (FP; with clearcut to stream’s edge
2297 outside of the buffer), a two sided 50-ft buffer along the entire stream (100%), and a clearcut to
2298 stream without a buffer (0%). The canopy cover was measured 1 meter above the stream surface
2299 with a spherical densiometer. The changes in canopy cover were distinguished from harvest
2300 effects and compared to unharvested reference sites by using a BACI design. For the FP
2301 treatment, mean canopy cover declined from 96% to 72% in the first-year post-harvest but
2302 continued to decline for 4 years to a minimum of 54%. In the 100% treatment mean canopy
2303 cover was more stable, decreasing from 94% to 88% in the first year and reaching a minimum of
2304 82% also by year 4. Canopy cover began to increase after year 4 through year 9 in both
2305 treatments. In contrast, the reference sites experienced much smaller reductions in canopy cover
2306 from 95% to 89% in the first four years. The cause of mortality in the treatment sites was

2307 primarily attributed to windthrow. However, while post-harvest mortality in the treatment sites
 2308 were higher on average than in the reference sites there was a high amount of variability between
 2309 sites in both the treated and reference sites. For example, in the first 2 years following harvest
 2310 mortality ranged from 1.8 to 34.6% (loss of basal area) between sites in the FP treatment. In
 2311 contrast, mortality in the reference sites ranged from 1.1 to 20.4% (loss of basal area) during the
 2312 same period.

2313 *Litter*

2314 Bilby & Heffner (2016) showed evidence that wind speed has a strong effect on the width of
 2315 litter delivery areas within riparian areas. They used a combination of field experiments and
 2316 simulation modeling to estimate the influence of different site factors (physiography, stand age,
 2317 species composition, wind speed) on litter delivery into streams. Their results showed that litter
 2318 travel distance was also linearly related to wind speed ($p < 0.0001$). Doubling wind speed at one
 2319 site led to a 67-87% expansion of the riparian litter contribution zone in the study area. However,
 2320 this study does not compare the differences in the influence of wind speed on the width of the
 2321 litter contributing area between harvested and unharvested sites.

2322 *LW*

2323 Table 16. Treatment and responses for selected publications investigating Large Wood relevant to
 2324 Q3.

Reference	Treatment	Response
McIntyre et al. (2021)	50 feet (100%); 50 feet with clearcut gaps (FP); Clearcut to stream (unbuffered)	<u>8-years post-harvest data 100% (n = 4), FP (n = 3), and unbuffered treatments (n = 4)</u> The FP–Reference contrast in mortality was not significant 2 years post-harvest, but it was at 5- and 8-years post-harvest as mortality in FP increased relative to the Reference over time. 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	<u>3 years post-harvest (n = 20)</u> 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 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.
Bahuguna et al. (2010)	Buffer widths 10 m, and 30 m	<u>7-years post-harvest (n = 3)</u> 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.
Schuett-Hames & Stewart (2011, 2019b)	Buffer widths 50 feet	<u>10-years post-harvest</u> 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.

2325

2326 Chapter 3 of the Hard Rock study compared changes in stand mortality and LW input from pre-
 2327 to post-harvest and between treated and untreated reference sites. Results showed that by year 8,
 2328 post-harvest mortality as a percentage of pre-harvest basal area was lower in the reference
 2329 (16.1%) than in the 100% (24.3%) and FP (50.8%) treatments. The FP–Reference contrast in
 2330 mortality was not significant 2 years post-harvest, but it was at 5- and 8-years post-harvest as
 2331 mortality in FP increased relative to the Reference over time. The contrast in mortality between
 2332 the 100% and Reference were not significant for any time interval 8 years post-harvest.
 2333 Wind/physical damage was the primary cause of mortality for all treatments, including the
 2334 Reference. In the 100% treatment it accounted for 78% and 90% of the loss of basal area and
 2335 density (stem/ha), respectively; in FP it accounted for 78% and 65% of the loss. Wind accounted
 2336 for a smaller proportion of mortality in the reference (52% and 43%, respectively).

2337

2338 LW recruitment to the channel was greater in the 100% and FP treatment than in the reference for
 2339 each pre- to post-harvest time interval. Eight years post-harvest mean recruitment of large wood
 2340 volume was two to nearly three times greater in 100% and FPB RMZs than in the references.
 2341 Annual LW recruitment rates were greatest during the first two years, then decreased. However,
 2342 there was a great deal of variability in recruitment rates within treatment sites and the differences
 2343 between treatments were not significant. Mean LW loading into the channel (pieces/m of channel
 2344 length) differed significantly between treatments in the magnitude of change over time. There
 2345 was a 66%, 44% and 47% increase in mean large wood density in the 100%, FP and 0%
 2346 treatments, respectively, in the first 2 years post-harvest compared with the pre-harvest period
 2347 and after controlling for temporal changes in the references. By year 8, only the FP treatment
 2348 showed a significantly higher proportional increase (41%) in wood loading when compared to
 2349 the reference. In the time interval 2-8 years post-harvest wood loading in the 100% treatment
 2350 stabilized.

2351

2352 Liquori (2006) investigated treefall characteristics within riparian buffer sites in a managed tree
 2353 farm in the Cascade Mountains of western Washington. Buffer widths ranged between 25-100
 2354 feet along non-fish bearing and fish bearing streams. Results showed that within no-cut buffers,
 2355 windthrow caused mortality was up to 3 times greater than competition induced mortality for 3
 2356 years following treatment with tree fall probability highest in the outer areas (closest to upland
 2357 clearcuts) of the buffers. Their results showed that treefall was generally highest at the outside
 2358 edges of buffers (50+ feet), representing about 60% of the total observed treefall, while the 0–25-

2359 foot zone represented ~18%, and the 25–50-foot zone represented ~22%. The researchers
2360 interpret these results as evidence that windthrow susceptibility within riparian buffers increases
2361 with increasing distance from the stream.

2362
2363 Martin & Grotenfendt (2007) compared riparian stand mortality and in-stream LW recruitment
2364 characteristics between riparian buffer strips with upland timber harvest and riparian stands of
2365 unharvested watersheds using aerial photography in the northern and southern portions of
2366 Southeast Alaska. All buffer strips in this study were a minimum of 20 m wide and included
2367 selective harvest within the 20 m zone (thinning intensity not specified or included in the
2368 analyses as an effect). The results from this study showed significantly higher mortality (based
2369 on cumulative stand mortality: downed tree counts divided by standing tree counts + downed tree
2370 counts by number/ha), significantly lower stand density (269 trees/ha in buffer units and 328
2371 trees/ha in reference units), and a significantly higher proportion of LW recruitment from the
2372 buffer zones of the treatment sites than in the reference sites. Also, results showed that mortality
2373 varied with distance to the stream. Differences in mortality for the treatment sites were similar to
2374 the reference sites for the first 0-10 m from the stream (only a 22% increase in the treated sites).
2375 However, mortality in the outer half of the stream buffers (10-20 m) across treatment sites was
2376 more than double (120% increase) that observed within the reference sites. The authors estimate
2377 that windthrow mortality was twofold and fivefold greater in the inner and outer halves of the
2378 treatment buffers than in the reference buffers, respectively.

2379
2380 Bahuguna et al. (2010) evaluated the difference in windthrow caused mortality between 10 m, 30
2381 m buffer widths (neither had thinning within the buffer and both had upland clear-cuts) and
2382 unharvested controls in the Coast Mountains, British Columbia. Following harvest, 11% of
2383 initially standing timber was blown down in the first and second years in the 10 m buffer,
2384 compared to 4% in the 30 m buffer, and 1% in the unharvested controls. However, after 8 years
2385 post-harvest, a significant amount of annual mortality occurred when winter storms brought
2386 down multiple trees in the unharvested control at 30%, compared to 15% in both 30 m and 10 m
2387 buffers. These results show evidence that timber harvest can increase windthrow caused
2388 mortality within protective buffers in the short term but can stabilize within a decade. Further,
2389 this study shows evidence that windthrow caused mortality is stochastic and large storm events
2390 can cause significant mortality within untreated riparian forests.

2391
2392 Schuett-Hames and Stewart (2019a) compared changes in stand mortality and LW recruitment
2393 between treated and untreated riparian areas along fish-bearing streams in eastern Washington.
2394 Treatments were prescribed under the Standard Shade Rule (SR), under the All-Available Shade
2395 rule (AAS), and unharvested reference sites. Both shade rules have a 30-ft no-cut buffer (core
2396 zone) immediately adjacent to the stream. The SR prescription allows thinning in the buffer zone
2397 30-75 feet (inner zone) from the stream while the AAS prescription requires retention of all
2398 shade providing trees in this area. Thinning non-shade providing trees within the inner zone is
2399 allowed under the AAS rule. Results from a mixed model comparison showed that the frequency
2400 of wood input from fallen trees was significantly greater in SR group compared to both the
2401 reference and AAS groups ($p < 0.001$), while the difference between reference and AAS groups

2402 was not significant. Over 60% of pieces recruited from AAS and SR fallen trees consisted of
2403 stems with attached rootwads (SWAR), double the proportion in the reference sites. The
2404 reference-AAS and reference-SR differences in recruitment of SWAR pieces were significant (p
2405 <0.001). The authors comment that the higher mortality and recruitment of LW in the SR sites
2406 was primarily due to windthrow.

2407 Schuett-Hames et al, (2011) compared tree mortality and LW recruitment between treated and
2408 untreated riparian stands along non-fish bearing streams in western Washington. Treated sites
2409 were prescribed a 50-foot-wide no-cut buffer. Annual fall rates of live and dead standing stems
2410 combined were over 8 times (by % of standing trees) and 5 times (by trees/acre/yr) higher in the
2411 50-foot buffers than in the reference buffers 3 years after treatment. These differences were
2412 significant for both metrics ($p < 0.001$). Over the entire five-year period, the percentages of
2413 standing trees that were uprooted and broken (as well as the combined total) were significantly
2414 greater in the 50-foot buffer. Wind was the dominant tree fall process, accounting for nearly 75%
2415 of combined fallen trees, 11% fell from other trees falling against them and 1.8% of fallen trees
2416 fell from bank erosion. Differences in mortality followed a similar pattern to tree fall rates. In the
2417 50-foot buffer sites mortality rates were significantly higher (3.5 times higher) than in the
2418 reference sites for the first three years following harvest. However, in years 4-5 mortality rates
2419 increased in the reference buffers after high-intensity storms resulting in non-significant
2420 differences in mortality during this period. The cumulative percentage of live trees that died over
2421 the entire five-year period was 27.3% in the 50-ft buffers compared to 13.6% in the reference
2422 reaches, but the difference was not statistically significant. The authors suggest that the lack of
2423 significance was likely due to the high variability in mortality between sites in the 50-foot
2424 buffers.

2425 In the follow-up study, Schuett-Hames & Stewart (2019b) reported that over a 10-year period,
2426 stand mortality in the 50-ft buffer treatment stabilized and showed a cumulative 14.1% reduction
2427 in live basal, while the reference stands showed a 2.7% increase in live basal area. The
2428 differences in these values were not significant. Cumulative LW recruited into stream channel
2429 over the 10-period was double in the 50-ft buffer treatment streams than in the reference streams.

2430 In general, the studies reviewed above show evidence that upland timber harvest with riparian
2431 retention buffers initially increases stand mortality within the buffers and increases LW
2432 recruitment relative to unharvested reference stands in the short-term. Hence, treated riparian
2433 forests appear to have a higher susceptibility to windthrow caused mortality, at least in the short
2434 term, compared to untreated stands. Depending on the streams in question, an increase in LW
2435 could be considered a positive or negative impact This increase in mortality and LW recruitment
2436 is attributed to an increase in the susceptibility to windthrow within the riparian buffers relative
2437 to the unharvested controls. Further, multiple studies (Liquori, 2006; Martin & Grotefendt, 2007,
2438 Schuett-Hames & Stewart 2019a) showed evidence that the increase in windthrow caused
2439 mortality is highest in the outer area of the riparian buffers (area closest to upland treatments).
2440 There is some evidence that thinning within the buffer can also affect mortality rates, but these
2441 studies are few. In the three studies that collected post-harvest data for 8 or more years
2442 (Bahuguna et al., 2010; McIntyre et al., 2021; Schuett-Hames & Stewart 2019b), there is
2443 indication that mortality in the riparian buffers and annual LW recruitment into adjacent streams

2444 stabilizes within 5-10 years. However, in the subsequent decades following treatments with
 2445 upland clearcuts there is evidence that LW recruitment rates can continue to decrease and in
 2446 stream wood loads may become depleted before recruitment rates can recover (Nowakowski &
 2447 Wohl, 2008; Reid & Hassan, 2020) depending on applied management practices (e.g., buffer
 2448 widths, road construction, etc.). For example, Teply et al. (2007) used simulation modeling to
 2449 estimate the effectiveness of Idaho Forest Practices for riparian buffers and found no significant
 2450 difference between predicted LW loads for harvested and unharvested sites 30-, 60-, or 100-years
 2451 post-harvest.

2452 *Nutrient*

2453 Table 17. Treatment and responses for selected publications investigating Nutrients relevant to
 2454 Q3.

Reference	Treatment	Response
Vanderbilt et al. (2003)	long-term datasets from six watersheds in the H.J. Andrews Experimental Watershed	<u>20-30 years of historical data</u> Total annual discharge was a positive predictor of annual dissolved organic nitrogen (DON) export in all watersheds with R ² values ranging between 0.42 to 0.79. No other nutrients nitrate (NO ₃ -N), ammonium (NH ₄ -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.

2455
 2456 Vanderbilt et al. (2003) analyzed long-term datasets (ranging 20-30 years for each watershed)
 2457 from six watersheds in the H.J. Andrews Experimental Watershed in the west-central Cascade
 2458 Mountains of Oregon to investigate patterns in dissolved organic nitrogen (DON) and dissolved
 2459 inorganic nitrogen (DIN) export with watershed hydrology. The researchers used regression
 2460 analysis of annual N inputs and outputs with annual precipitation and stream discharge to
 2461 analyze patterns. Their results showed that total annual discharge was a positive predictor of
 2462 annual DON export in all watersheds with R² values ranging between 0.42 to 0.79. In contrast,
 2463 relationships between total annual discharge and annual export of nitrate (NO₃-N), ammonium
 2464 (NH₄-N), and particulate organic nitrogen (PON) were variable and inconsistent across
 2465 watersheds. The authors speculate that different factors may control organic vs. inorganic N
 2466 export. The authors emphasize the importance of analyzing data from multiple watersheds in a
 2467 single climactic zone to make inferences about stream chemistry.

2468 Yang et al. (2021) investigated the effects of drought and forest thinning operations
2469 (independently and combined) on stream water chemistry in the Mediterranean climate
2470 headwater basins of the Sierra National Forest. The effects of drought alone were examined by
2471 comparing water samples collected from control watersheds for 2 years before and 3 years after
2472 drought. The effects of drought and thinning combined were examined by comparing water
2473 samples collected from treated sites to reference sites for three years post-harvest (all drought
2474 years). Drought alone altered the concentration of dissolved organic carbon (DOC) in stream
2475 water. Volume-weighted concentration of DOC was 62% lower ($p < 0.01$) and the ratio of
2476 dissolved organic carbon to dissolved inorganic nitrogen (DOC:DON) was 82% lower ($p =$
2477 0.004) in stream water in years during drought (WY 2013–2015) than in years prior to drought
2478 (WY 2009 and 2010). Drought combined with thinning altered DOC and DIN concentrations in
2479 stream. For stream water, volume-weighted concentrations of DOC were 66- 94% higher in
2480 thinned watersheds than in control watersheds for all three consecutive drought years following
2481 thinning. No differences in DOC concentrations were found between thinned and control
2482 watersheds before thinning. The authors conclude that their results showed evidence that the
2483 influences of drought and thinning are more pronounced for DOC than for DIN in streams.

2484 *Drought Frequency*

2485 Wise (2010) used reconstructed newly collected tree-ring data augmented with existing
2486 chronologies from sites at three headwater streams in the Snake River Basin to estimate
2487 streamflow patterns for the 1600-2005 time-period. Streamflow patterns derived from
2488 instrumental data and from reconstructed chronologies were compared with other streamflow
2489 previously reconstructions of three other western rivers (the upper Colorado, the Sacramento,
2490 and the Verde Rivers) in similar climates to examine synchronicity among the rivers and gain
2491 insight into possible climatic controls on drought episodes. The reconstruction model developed
2492 for the analysis explained 62% of the variance in the instrumental record after adjustment for
2493 degrees of freedom. Results showed evidence that droughts of the recent past are not yet as
2494 severe, in terms of overall magnitude, as a 30-year extended period of drought discovered in the
2495 mid-1600s. However, in terms of number of individual years of $< 60\%$ mean-flow (i.e., low-flow
2496 years), the period from 1977-2001 were the most severe. Considering the frequency of
2497 consecutive drought years, the longest (7-year-droughts), occurred in the early 17th and 18th
2498 centuries. However, the 5-year drought period from 2000-2004 was the second driest period over
2499 the 415-year period examined. The correlative analysis of the chronologies developed for the
2500 upper Snake River with other rivers of the West showed mixed results with periods of positive
2501 and negative correlations. The author interprets these results as evidence that drought frequency,
2502 in general, in this area appears to be increasing in severity and that mean annual flow appears to
2503 be reducing in the latter half of the 20th and the beginning of the 21st century. The exceptions
2504 being the 1930's dustbowl, and an unusually long dry period in the early 1600s.

2505 *Fire Frequency*

2506 Dwire & Kauffman (2003) in their reviewed and summarized the available conducted on fire
2507 regimes in forested riparian areas relative to uplands in the western United States. They
2508 summarized the distinctive features of riparian areas that can influence the properties of fire as

2509 (1) higher fuel loads because of higher net primary productivity, (2) higher fuel moisture content
2510 due to proximity to water, shallow water tables, and dense shade, (3) active channels gravel bars
2511 and wet meadows may act as fuel breaks, (4) topographic position (canyon bottoms, low point on
2512 landscape) leads to higher relative humidity, fewer lightning strikes, but more human-caused
2513 ignitions, (5) microclimate may lead to cooler temperatures and higher humidity that can lessen
2514 fire intensity and spread. They highlight a need for more extensive research on the history and
2515 ecological role of fire in the riparian areas of the western United States.

2516 There is a logical assumption that fire in riparian zones would be less frequent than in adjacent
2517 uplands because of its proximity to water. However, several studies have been conducted which
2518 reconstruct historical fire regimes in riparian areas relative to adjacent uplands and have
2519 provided varying results. Everett et al. (2003) used fire-scar and stand-cohort records to estimate
2520 the frequency and seasonality of fire in Douglas-fir dominated riparian areas and adjacent
2521 uplands. They sampled sites along 49 stream segments on 24 different streams in the Wenatchee
2522 (33 segments) and Okanogan (16 segments) National Forests. The data collected allowed for
2523 reconstruction of fire occurrence back to 1896. Their results showed that the mean count of fire
2524 scars was significantly fewer in riparian areas than in adjacent uplands regardless of valley type,
2525 aspect, or plant association group. However, the difference between riparian and upland fire scars
2526 was greatest for western aspects and least for northern aspects. Also, the differences were
2527 greatest for the ‘warm mesic shrub/herb’ plant association group (e.g., common snowberry), and
2528 least in the cool dry grass plant association group (e.g., pinegrass, or elk sedge).

2529 Prichard et al. (2020) evaluated drivers of fire severity and fuel treatment effectiveness at the
2530 2014 Carlton Complex in north-central Washington State. While this study's objective does not
2531 specifically evaluate differences in fire severity between riparian and upland forests, it did
2532 evaluate differences in fire severity based on variations in topographic and vegetation type
2533 variables. One vegetation variable was classified broadly as “riparian vegetation” from the
2534 publicly available data set LANDFIRE. The authors used a combination of simultaneous
2535 autoregression and random forests approaches to model drivers of fire severity. In the study
2536 area's southern section (1 of 2 designated study areas), the results showed cover type was a
2537 significant predictor with negative correlations with fire severity in non-forest types and riparian
2538 forests.

2539 Conversely, Olson & Agee (2005) provide evidence that fire return intervals in the riparian areas
2540 of the Umpqua National Forests, Oregon, may not have differed significantly from adjacent
2541 upland forests. They reconstructed historical fire return intervals from fire scar cross sections
2542 taken from 15 stream reaches and 13 paired upland forests. Sites were primarily dominated by
2543 Douglas-fir, western red cedar, and western hemlock. The number of fires per plot, maximum
2544 and minimum fire return intervals, and the Weibull median fire return interval (WMPIs) were
2545 compared between riparian and upland stands using the Wilcoxon signed rank test, the Mann-
2546 Whitney U-test for unmatched samples, and the Kruskal-Wallis one-way analysis of variance.
2547 The results showed that between 1650 and 1900, 43 fire years occurred on 80 occasions. Of these
2548 80 occasions, 33 were recorded in the riparian and adjacent upslope forest, 23 were recorded in
2549 only the riparian area, and 24 were recorded only in the upland forests. The riparian WMPIs

2550 were somewhat longer (ranging from 35-39 years, with fire return intervals ranging from 4-167
2551 years) than upslope WMPIs (ranging from 27-36 years, with fire return intervals ranging from 2-
2552 110 years), but these differences were not significant. The authors, Olson & Agee (2005),
2553 interpret these results as evidence that fires in this area were likely patchy and smaller in scale
2554 with a high incidence of fires occurring only in the riparian area or only in the upland forests,
2555 and less commonly in both. The authors also suggest that fire is a natural occurrence in the
2556 riparian areas of this area and should be restored to protect riparian forest health.

2557 Another study from the Klamath Mountains in northern California showed evidence that fires in
2558 riparian forests may have been more frequent than in adjacent upland forests (Skinner, 2003).
2559 Skinner (2003) used dendrochronological methods to construct fire return intervals for 5 riparian
2560 and adjacent upland forests sites, each between 1-2 hectares. Because of the small sample size,
2561 statistical analysis was not conducted, and their results are only descriptive. The ranges of fire
2562 return intervals (FRIs) were similar between riparian and upland forests. However, the median
2563 FRI for the riparian forests was nearly double that in adjacent uplands. The authors conclude that
2564 these limited data suggest fire in the riparian areas may be more variable than in the uplands in
2565 frequency and intensity.

2566 Yet, another study from Harley et al. (2020) showed evidence that the differential fire occurrence
2567 riparian and adjacent uplands may have been dependent on weather (i.e. drought). Harley et al.
2568 (2020) reconstructed low-severity fire histories from tree rings in 38 1-ha plots. This data was
2569 supplemented with existing fire histories from 104 adjacent upland plots. 2633 fire scars were
2570 sampled from 454 (127 riparian; 329 upland) trees from two sites in the Blue Mountains in
2571 north-eastern Oregon: One in the Wallowa-Whitman (WWNF) and one in the Malheur (MNF)
2572 National Forests. Fire-scar dates were used to construct plot composite fire chronologies,
2573 excluding fire dates recorded from only one tree. These were used to compute median fire
2574 intervals for riparian and upland forests for each site and for both sites combined. A mixed linear
2575 model with fire interval as a response and plot type (riparian vs. Upland) as a predictor was used
2576 to check for statistical difference in fire frequency. The influence of climate on fire occurrence
2577 was inferred by assessing whether the summer Palmer Drought Severity Index (PDSI) differed
2578 significantly during the fire year or preceding or following years (-3 to +1 years) using
2579 superimposed epoch analysis. Results showed that Fires burned synchronously in riparian and
2580 upland plots during more than half of the fire years at both WWNF and MNF (55%and57%,
2581 respectively). At WWNF, fires burned during 65 years of the analysis period (1650–1900); 36
2582 burned in both riparian and upland plots, 7 burned only in riparian plots and 22 burned only in
2583 upland plots. At MNF, fires burned during 74 years of the analysis period; 42 burned in both
2584 riparian and upland plots, 3 burned only in riparian plots and 29 burned only in upland plots. At
2585 both sites, average PDSI was significantly warm–dry during synchronous fire years. However,
2586 climate was not significantly cool–wet during non-synchronous fire years at either site. The
2587 authors interpret these results as evidence that historical synchronized fire occurrence was more
2588 likely during excessively dry or drought years.

2589 There is also evidence that riparian forest fire regimes have been altered in many areas from pre-
2590 Euro-American settlement due to fire suppression. Messier et al. (2012), used dendro-ecological

2591 methods to reconstruct pre-Euro-American settlement riparian forest structure and fire frequency
 2592 for comparison of changes post-settlement in the Rouge River of southwestern Oregon. Fire
 2593 events were dated from increment cores and fire-scar cross-sections back to the year 1600,
 2594 approximately. Changes in annual radial growth rates were used to infer changes in stand density
 2595 over time. Results showed the age distribution prior to 1850 followed a pulse pattern of
 2596 recruitment with recruitment peaks occurring around 1850, 1800, and between 1740-1770
 2597 (though this pulse was difficult to discern because the sample size of trees established prior to
 2598 1740 were relatively few). After 1900, many mixed conifer sites showed a dramatic increase in
 2599 the recruitment of more more-shade tolerant white fir (*Abies concolor*) compared to Douglas-fir
 2600 (*Pseudotsuga menziesii*). White fir comprised 51% of the live trees recruited after 1900, but only
 2601 18% of the live trees before 1900. Results from the 26 cross-dated fire scars spanned from 1748
 2602 – 1919 with the highest number of detected fires occurring in the early-settlement period (1850-
 2603 1900). The authors interpret these results as evidence that fire suppression over the last century
 2604 has changed the successional pathway and stand structure of riparian forests in this area.

2605 Van de Water & North (2011) found similar results from their study in the northern Sierra
 2606 Nevada. They compared current field data with reconstructed data to estimate changes in stand
 2607 structure, fuel loads, and potential fire behavior over time. Additionally, they estimated how
 2608 these conditions for riparian forests compared to adjacent upland forests during the reconstructed
 2609 and current periods. Data for current forest structure, species composition, and fuel loads were
 2610 collected from 36 adjacent riparian and upland sites (72 sites total). The reconstruction period
 2611 was set at the year of the last fire (ranging from 1848 – 1990), determined from fire-scar records.
 2612 Potential fire behavior, effects, and canopy bulk density were estimated for current and
 2613 reconstructed stand conditions for riparian and upland sites using Forest vegetation Simulator
 2614 (FVS). Stand structure (BA, stand density, snag volume, QMD, average canopy base height),
 2615 species composition, fuel load, potential fire behavior, canopy bulk density, and mortality were
 2616 compared between current and reconstructed periods for riparian and upland sites, and between
 2617 sampling areas (riparian vs. Upland) with an analysis of variance (ANOVA). Results showed that
 2618 under current conditions, riparian forests were significantly more fire prone than upland forests,
 2619 with greater stand density (635 vs. 401 stems/ha), probability of torching (0.45 vs. 0.22),
 2620 predicted mortality (31% vs. 16% BA), and lower quadratic mean diameter (46 vs. 55 cm),
 2621 canopy base height (6.7 vs. 9.4 m), and frequency of fire tolerant species (13% vs. 36% BA).
 2622 However, the reconstructed periods showed no significant difference between riparian and
 2623 upland forests for fuels and structure. The authors suggest that these results provide evidence that
 2624 the historic fire return intervals may not have differed significantly between riparian and upland
 2625 forests in this area.

2626 *Fire Effects on Function*

2627 *Litter and Nutrients*

2628 Table 17. Treatment and responses for selected publications investigating the relationship
 2629 between wildfire Litter and Nutrients relevant to Q3.

Reference	Treatment	Response
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Musetta-Lambert et al. (2017)	Buffer widths 30 m; wildfire	<u>Sampling began 7-17 years after harvest or 12 years after wildfire (n = 5 harvest, 7 fire, 6 reference)</u> Total leaf litter input was significantly higher at fire sites than at harvested or reference sites. 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
Rhoades et al. (2011)	Wildfire	<u>1- year pre- and 5-years following the 2002 Hayman Fire in Colorado</u> Cation concentrations and acid neutralizing capacity (ANC) increased immediately and significantly following fire that peaked at 4 months . 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.
Son et al. (2015)	Wildfire	<u>2-years pre- and immediately following wildfire</u> Results for turbidity showed no significant differences between pre- and post-fire ranges 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).

2630

2631 Musetta-Lambert et al. (2017) compared changes in leaf-litter inputs into streams following
 2632 adjacent riparian forest harvesting or wildfire to reference sites. This study took place in the
 2633 boreal forest of the White River Forest management Area in Ontario, Canada, ~75 km inland
 2634 from the northern shore of Lake Superior. This study is outside of western North America (the
 2635 focal area for this review), but it is the only study found that provides experimental evidence of
 2636 wildfire's effects on litter inputs. The study sites consisted of ~50 m reaches in 25 catchments, 10
 2637 that were harvested, 7 that experienced wildfire, and 8 references. Of these reaches a subset was
 2638 used to riparian forest structure, leaf litter inputs, and water chemistry (5 harvest, 7 fire, 6
 2639 reference). The harvested catchments were harvested 7-17 years prior to the study (minimum 30
 2640 m riparian buffers; specific harvest rules/methods not described). The wildfire catchments had
 2641 burned 12 years prior to the study and had no dead material removed. The reference catchments
 2642 had no fire or harvesting for a minimum of 40 years. Water grab samples were collected in
 2643 September, October and November 2010, and May, June and September of 2011 from the study
 2644 reaches.

2645 Water samples were analyzed to obtain measurements for pH, conductivity, dissolved organic
 2646 carbon (DOC) and dissolved inorganic carbon (DIC) concentrations, soluble reactive
 2647 phosphorous (SRP), along with a suite of other major elements and nutrient measurements (total
 2648 N, NH₄, total P, Ca, K, Mg, etc.). Vertical leaf litter traps consisting of plastic bins were placed at
 2649 10 locations along the bankfull width of each site. Lateral leaf fall was not collected or analyzed.
 2650 Leaf litter inputs were focused on leaves from deciduous trees and shrubs. Leaves were separated
 2651 to the lowest possible taxonomic level, dried and weighed for analysis.

2652 Univariate one-way ANOVA models were used to determine differences in water chemistry,
 2653 riparian forest characteristics of juvenile tree and shrub communities (richness, Shannon's

2654 diversity index, relative occurrence of individual taxa), mature tree communities (total basal
2655 area, stem density), and litter subsidies (richness, mass input). Results for water chemistry
2656 showed that Conductivity, pH, and dissolved inorganic carbon were significantly higher at fire
2657 sites than at reference sites ($p = 0.02$, $p = 0.04$, $p = 0.03$, respectively) but did not differ between
2658 harvested and fire sites or harvested and reference sites.

2659 Results for stand structure showed there was significantly higher taxa richness in fire sites than
2660 in reference sites or harvested sites ($p = 0.04$). Taxa richness did not differ significantly between
2661 reference and harvested sites. Reference sites had significantly higher total mean densities (# ha
2662 $^{-1}$) of mature riparian trees (>10 cm DBH) than fire ($p < 0.001$) and harvested sites ($p = 0.036$).
2663 Total mature tree densities in reference sites were 1.7x and 4x higher than in harvested and fire
2664 sites, respectively. Taxa richness in leaf litter subsidies did not significantly differ among
2665 disturbances ($p = 0.477$). Total leaf litter input (g m^{-1}) significantly higher at fire sites than at
2666 harvest ($p = 0.02$) or reference sites ($p = 0.02$). Fire sites had significantly greater leaf litter
2667 inputs of willow spp. ($p = 0.0002$, 0.006 , respectively), Atlantic ninebark ($p = 0.002$, 0.003 ,
2668 respectively) and speckled alder ($p = 0.02$, 0.04 , respectively) than in both reference and
2669 harvested sites. The authors interpret these results as evidence that natural fire disturbance in
2670 low-order boreal forest streams had higher leaf litter inputs, and different stand structures and
2671 composition than harvested or untreated riparian stands. They suggest that while harvested
2672 stands were more structurally similar to fire affected stands than reference stands, the future
2673 implementation of these treatments should intend to emulate the patchy nature of wildfire
2674 disturbance. This would enhance the diversity of riparian forest structure and increase litter
2675 subsidies into streams.

2676 *Nutrients*

2677 Rhoades et al. (2011) monitored stream chemistry and sediment 1-year before and for 5-years
2678 after the 2002 Hayman Fire in Colorado. Monthly water samples were collected from streams in
2679 three burned and three unburned watersheds. Pre-fire and post-fire water nitrate, cation
2680 concentration (Ca^{2+} , Mg^{2+} , K^{+}), acid neutralizing capacity (ANC) and turbidity were compared
2681 graphically and statistically between the three burned and unburned basins. Results for cation
2682 concentrations and ANC showed an immediate and significant increase that peaked during the 4-
2683 month period following the fire. The Ca^{2+} concentrations, ANC, and conductivity remained
2684 elevated in the burned streams for 2 years compared to pre-fire conditions, and unburned
2685 streams. Stream water nitrate and turbidity increased linearly with the proportion of a basin
2686 burned or burned at high severity. No other chemical analyte showed a significant response to
2687 fire severity or extent. Streams draining basins affected by extensive stand-replacement fires
2688 showed a 3.3-fold higher ($p = 0.000$) nitrate concentration than basins that burned less. Also,
2689 turbidity was 2.4-fold ($p = 0.000$) higher average turbidity compared to streams in basins burned
2690 less severely or extensively. In the extensively burned basins, stream water nitrate concentrations
2691 did not decline over the five years of the study and the mean concentrations of nitrate in the fifth
2692 year did not differ from the fourth year. The authors conclude that wildfire can have immediate
2693 and mid-term (up to 5 years) impacts on water chemistry and turbidity. Further, the magnitude

2694 and temporal increases of nitrate and turbidity, specifically, have a positive relationship with burn
2695 severity and extent.

2696 Son et al. (2015) compared stream water samples before and after an intense wildfire in the
2697 Cache la Poudre River basin in Colorado. Stream water samples for total phosphorus (TP) and
2698 total nitrogen (TN) were collected over 2 years (2010 – May 2012) before the fire in June 2012.
2699 Two post-fire water samples were taken: 1) immediately following containment of the fire (July
2700 4, 2012) and 2) twelve days after the fire was contained (July 16, 2012). For each pre- and post-
2701 fire sampling date water samples were collected at three randomly selected points at two sites.
2702 Riverbed sediments were also collected at each site and sieved through a 2 mm sieve to capture
2703 the geochemically reactive portion of the riverbed. The pre- and post-fire sediment and stream
2704 water quality were compared with t-test. Correlations of sediment and stream water quality with
2705 other factors (e.g., stream temperature, precipitation, streamflow) were evaluated with a
2706 Pearson's correlation at 0.05 and 0.1 significance levels. Results for turbidity showed no
2707 significant differences between pre- and post-fire ranges immediately following fire. However,
2708 after the first post-fire rainfall (2.5 mm) nephelometric turbidity ranged from 113.6 - 2099.4
2709 NTU (mean = 641.62 NTU), a considerable increase from pre-fire data (mean 11.3 NTU), and
2710 post-fire data before rainfall (47.3 NTU). Post-fire aqueous TP and TN loads ranged from 30.5 -
2711 56,086 and 45.4 - 1203 kg/day, respectively, and were significantly higher than pre-fire values
2712 (390 and 6 times higher than pre-fire values for TP and TN, respectively). The authors note that
2713 this is likely due to the transport and input of ash into the stream. After the first rainfall, all forms
2714 of P were significantly higher than pre-fire concentrations, such as soluble reactive phosphorus
2715 (SRP; $p = 0.000$), dissolved organic phosphorus (DOP; $p = 0.009$), and particulate phosphorus
2716 (PP; $p = 0.02$). Riverbed sediment equilibrium P concentrations increased significantly ($p =$
2717 0.007) from pre- to post-fire in all sites. The authors conclude that this study shows evidence that
2718 stream TP and TN, and riverbed sediment TP all increased significantly after the first rainfall,
2719 post-fire. They further suggest that the effects of wildfire on riverbed sorption mechanisms are
2720 very complex but further research would be valuable because fire impacted sediments highly
2721 concentrated P can become a long-term source of P.

2722 *LW*

2723 Bendix & Cowell (2010) investigated the effects of fire and flooding on LW input in two
2724 tributaries of Sespe Creek (Potrero John Creek and Piedra Blanca Creek) in the Los Padres
2725 national Forest in southern California. Both sites were located within the perimeter of the Wolf
2726 Fire that burned in June of 2002. Extensive flooding in the area occurred during January and
2727 February of 2005. The study area is characterized by chaparral dominated communities and a
2728 Mediterranean-type climate. While there is a scarcity of trees in the uplands, the riparian areas
2729 contained substantial growth of *Alnus rhombifolia* (white alder), *Populus fremontii* (Fremont
2730 cottonwood), *Quercus agrifolia* (coast live oak), *Quercus dumosa* (scrub oak) and *Salix* sp.
2731 (willows) on the valley floors. Thus, any change in in-stream or riparian area LW was sourced
2732 exclusively from the riparian area. Data for LW and standing live and dead stems in the riparian
2733 area were collected in July, of 2003 (1-year pre-fire) and again in July of 2005 (3-years post-fire,
2734 5-6 months after flood events). This data was used to answer 4 questions: 1) How many of the

2735 burned snags fell during this time, and what was the species composition?, 2) Did snags differ by
2736 species or size in the rate at which they fell?, 3) How did flooding after the fire affect the rate at
2737 which snags fell?, 4) How did flooding affect the mobilization of fallen snags? Questions 1 was
2738 analyzed by comparing descriptive data (i.e., no statistical analysis). A t-test was used to compare
2739 mean diameter of standing and fallen stems (question 2). T-tests were also used to analyze
2740 differences in mean flow depth for standing vs. fallen snags and for fallen snags still present vs.
2741 snags that had been transported after flooding (questions 4 and 5). Results showed high post-fire
2742 mortality (94%) with 339 of 362 stems killed. By 2005, 57 of the 339 snags had fallen (16.8%).
2743 The majority of fallen stems were either *Alnus* or *Salix* species. Standing snags varied in size
2744 from 3 cm to 69.2 cm, whereas those that had fallen ranged from 3 cm to 33 cm. Among the
2745 fallen snags, those <10 cm were not proportionate to the overall numbers, whereas snags between
2746 10 cm and 30 cm were disproportionately likely to fall. While fewer snags in the larger size
2747 classes the mean diameter of fallen snags was larger than the mean diameter of standing snags
2748 (11.4±10.9 cm vs. 11.0±8.0 cm) and did not differ significantly. The mean flood depth for fallen
2749 snags (1.05±0.68 m) was significantly greater than those still standing (0.40±0.56 m; $p < 0.0001$,
2750 $n=339$). The three species experiencing no snagfall at all (*Abies glauca*, *Rhamnus californica* and
2751 *Quercus agrifolia*) occurred only in higher quadrats, which had experienced virtually no
2752 flooding. Of the 57 snags that had fallen by July 2005, 43 (75%) were gone from the quadrats in
2753 which they had been recorded in 2003. The snags that had been mobilized were from quadrats
2754 that had experienced deeper flood depths (1.14±0.69 m) than those that had remained. (0.80±0.62
2755 m), but the difference is insignificant. The authors interpret these findings as an indication that
2756 short-term rates of snagfall following wildfire are influenced by the species composition of
2757 burned stems and by post-fire flood depth. Thus, although wildfire resulted in many burned snags
2758 across the valley floor, the rate at which these stems are recruited into the fluvial system as
2759 woody debris varies by the ecological characteristics and the geomorphic setting.

2760

2761 [Focal Question 4](#)

2762 *4. How do various treatments within riparian buffers relate to forest health and resilience to fire,*
2763 *disease, and other forest disturbances?*

2764 While there are several studies that discuss the frequency, dynamics, or potential for
2765 disturbances, especially fire, in riparian areas of the western United States (Dwire & Kauffman,
2766 2003; Everett et al., 2003; Merschel et al., 2014) there is a dearth of studies that investigate how
2767 treatments within the riparian area or in riparian buffers relate to the riparian area's resilience to
2768 disturbance. No studies found in our literature search and review were suitable for providing
2769 direct experimental evidence of the effects of riparian buffer treatments on riparian health and
2770 resilience to disturbance except for several studies that provide evidence that riparian harvest
2771 treatments have the potential to increase susceptibility to windthrow caused mortality. Post-
2772 harvest changes in windthrow susceptibility are discussed in focal question 3. One study used
2773 simulation modeling to estimate changes in health and susceptibility to disturbance with and
2774 without treatment.

2775 Ceder et al. (2018) used Forest Vegetation Simulator (FVS) to predict how treatment along fish-
 2776 bearing streams of eastern Washington affects riparian stand health and susceptibility to insects,
 2777 disease, and crown fire. The projected changes in susceptibility were produced for the low- and
 2778 mid-elevation regulatory zones for timber harvest. Models were run for 50 years with and
 2779 without application of prescribed treatments. Prescriptions for these zones include a buffer width
 2780 of 75-130 ft depending on stream width category. For all treatments, no harvest is allowed within
 2781 the first 30 feet from the bankfull channel. Timber harvest is allowed in the remaining width of
 2782 the buffer but must meet a minimum basal area based on the regulatory zone. The authors report
 2783 high variability in the data and the outputs of each modeling scenario. However, they report that
 2784 overall, as riparian zone growth was simulated with and without management, tree size and stand
 2785 density increased, along with some increases in insect and disease susceptibility and potential
 2786 fire severity without management and decreases with management.

2787 [Focal Question 5](#)

2788 *5. How do the functions provided by riparian stands change over time (e.g., large woody debris*
 2789 *recruitment from farther away from the stream)?*

2790 This question addresses a temporal and spatial component to changes in function. The question
 2791 specifies “change over time” but provides an example with a spatial component. While harvest is
 2792 not specified as a factor, studies that quantify changes to riparian function in harvested reaches
 2793 have been included. Studies that compare differences in one or more functions between
 2794 comparable sites in different successional stages (i.e., different mean age) are also included.
 2795 Papers that investigate the changes in LW source distance following harvest have been included
 2796 because of the given example (*large woody debris recruitment from farther away from the*
 2797 *stream*).

2798 *Shade*

2799 Table 18. Treatment and responses for selected publications investigating Shade relevant to Q5.

Reference	Treatment	Response
Kaylor et al. (2017)	old-growth (> 300 years old) and mid-successional (50-60 years old) Douglas-fir dominated forests	the authors estimate that 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)
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

2800

2801 Kaylor et al. (2017) compared canopy cover throughout stream networks adjacent to old-growth
 2802 (> 300 years old) and mid-successional (50-60 years old) Douglas-fir dominated forests in the
 2803 H.J. Andrews Experimental Forest in the Cascade Mountains of Oregon. Canopy openness was
 2804 quantified with a handheld spherical densiometer. Data was supplemented with a review of
 2805 literature studies conducted in the Pacific Northwest that reported stand age and canopy cover
 2806 over the stream. The combined datapoints for canopy openness (%) were plotted against stand
 2807 age and fit with a negative exponential curve. From the slope of the curve, the authors estimate
 2808 that canopy openness reaches its minimum value in regenerating forests at ~30 years and
 2809 maintains with little variability until ~100 years. Mean canopy openness in stands 30-100 years
 2810 old was 8.7% with a range from 1.2 to 32.0% (standard deviation = 5.7). Canopy openness over
 2811 streams in old-growth forests averaged 18.0% but was highly variable and ranged from 3.4 to
 2812 34.0% (standard deviation = 5 7.9).

2813 Warren et al. (2013) compared canopy cover between old-growth-forests (>500 years old) and
 2814 young second-growth stands (~40-60 years old) in the H.J. Andrews Experimental Forest in the
 2815 Cascade Mountains of Oregon. Canopy cover was estimated using a convex spherical
 2816 densiometer. Streams were paired based on reach length, bankfull width, and north (n =2), vs.
 2817 south (n=2) facing watersheds. Results showed significant differences in percent forest cover
 2818 between old-growth and second-growth reaches in both south-facing watersheds in mid-summer
 2819 ($p < 0.10$). For the north-facing watersheds, differences in canopy cover and light availability (p
 2820 < 0.10) were only significant at 1 of the two reaches. Overall, three of the four paired old-growth
 2821 reaches had significantly lower mean percent canopy cover. The authors interpret these results as
 2822 evidence that old-growth forest canopies were more complex and had more frequent gaps.

2823 *Litter*

2824 Table 19. Treatment and responses for selected publications investigating Litter relevant to Q5.

Reference	Treatment	Response
Kiffney & Richardson (2010)	Upland clearcuts with (1) no buffer, (2) 10 m buffer (~30 feet), (3) 30 m buffers	<u>8 years post-harvest data</u> Differences in litter flux relative to riparian treatment persisted through year 7, while 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.
Bilby & Heffner (2016)	Litter samples released from canopy height at one old-growth site and one young forest site.	<u>1-year of data actual age of stands not quantified, estimated by mean height (47.0 and 32.4 m)</u> Needles released at 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.

2825

2826 Kiffney & Richardson (2010) compared changes in litter input between riparian harvest
 2827 prescriptions that included clear-cut to stream edge, 10 m wide buffer reserve, 30 m buffer
 2828 reserves, and an uncut control over the course of 8 years. No thinning was applied within the
 2829 reserves. Upland treatment at all sites applied clearcut. Results showed differences in litter flux
 2830 relative to riparian treatment persisted through year 7, while a positive trend between reserve
 2831 width and litter flux remained through year 8. Needle inputs remained 6x higher in the buffer and
 2832 control sites through year 7, and 3-6x higher in year 8 than in the clearcut sites. Twig inputs into
 2833 the control and buffered sites were ~25x higher than in the clearcut sites in the first year after
 2834 treatment. The linear relationship between reserve width and litter inputs was strongest in the
 2835 first year after treatment, explaining ~57% of the variation, but the relationship could only
 2836 explain ~17% of the variation in litter input by buffer width by year 8 (i.e., the relationship
 2837 degraded over time). The authors interpret these results as evidence that litter flux from riparian
 2838 plants to streams, was affected by riparian reserve width and time since logging.

2839 Bilby & Heffner (2016) used linear mixed effects models developed for young and old-growth
 2840 forests of western Washington to estimate controls on litter delivery. Litter samples were released
 2841 from canopy height at one old-growth forest site and one young forest site. The mean age of
 2842 “mature” and “young” sites was not specified but the mean tree heights were 47.0 m and 32.4 m
 2843 for the mature and young sites, respectively. Results showed that needles released at mature sites
 2844 had a higher proportion of cumulative input from greater distances than needles or alder leaves
 2845 released at younger sites. The model estimated that the width of the contributing area for needles
 2846 was ~35% greater at older sites than at younger sites.

2847 *Source distance curves for LW*

2848 Table 20. Treatment and responses for selected publications investigating LW source distance
 2849 curves relevant to Q5.

Reference	Treatment	Response
Schuett-Hames & Stewart (2019a)	30-ft no-cut buffer width, and thinning 30-75 ft from the stream (SR); retention of all shade providing trees in this area (AAS)	<u>5-years post-harvest</u> Most recruited fallen trees originated in the core zone (0-30 feet; 76%, 72%, and 64% for the reference, AAS and SR groups, respectively), while the proportion from the inner zone (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	<u>6 years post-harvest</u> 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.

2850

2851 Schuett-Hames & Stewart (2019a) compared differences in LW recruitment between riparian
2852 management zones harvested under the current standard Shade Rules (SR), the All-Available
2853 Shade Rule (AAS), and unharvested references for fish-bearing streams in the mixed conifer
2854 habitat type (2500 - 5000 feet elevation) for eastern Washington. Both shade rules have a 30-ft
2855 no-cut buffer (core zone) immediately adjacent to the stream. The SR prescription allows
2856 thinning in the buffer zone 30-75 feet (inner zone) from the stream while the AAS prescription
2857 requires retention of all shade providing trees in this area. Results showed that cumulative wood
2858 recruitment from tree fall after the five-year post-harvest interval was highest in the SR group,
2859 lower in the AAS group and lowest in the REF group. The SR and AAS LW recruitment rates by
2860 volume were nearly 300% and 50% higher than the REF rates, respectively. Wood recruitment in
2861 the SR sites was significantly greater than in the AAS and reference sites. Conversely,
2862 differences in wood recruitment did not differ significantly between the AAS and reference sites.
2863 Considering the source distance of post-harvest recruited LW, most recruited fallen trees
2864 originated in the core zone (76%, 72%, and 64% for the REF, AAS and SR groups, respectively),
2865 while the proportion from the inner zone (30–75 feet from the stream) was ~10% greater for the
2866 SR group compared to the AAS and REF groups. These results provide evidence that the
2867 thinning treatments applied in the inner zone of the SR treatment changed the spatial pattern
2868 (source distance) of wood recruitment from fallen trees within 5 years post-harvest.

2869 Burton et al. (2016) examined the relationship between annual in-stream wood loading and
2870 riparian buffer widths adjacent to upland thinning operations. Buffer widths were 6, 15, or 70
2871 meters and upland thinning was to 200 trees per ha (tph), with a second thinning (~10 years later)
2872 to ~85 tph, alongside an unthinned reference stand ~400 tph. Data for LW in streams were
2873 collected for 6 years (5 years after the first harvest and 1 additional year after the second
2874 harvest). The results showed that between 82-85% of the wood with discernable sources (90%
2875 for wood in early stages of decay; 45% of wood in late stages of decay) came from within 15 m
2876 of the stream, and the relative contribution of wood to streams declined rapidly with increasing
2877 distance.

2878 Martin & Grotenfendt (2007) compared riparian stand mortality and in-stream LW recruitment
2879 characteristics between riparian buffer strips with upland timber harvest and riparian stands of
2880 unharvested watersheds using aerial photography. All buffer strips in this study were a minimum
2881 of 20 m wide and included selective harvest within the 20 m zone (thinning intensity not
2882 specified or included in the analyses as an effect). The results showed significantly higher
2883 mortality (based on cumulative stand mortality: downed tree counts divided by standing tree
2884 counts + downed tree counts), significantly lower stand density (269 trees/ha in buffer units and
2885 328 trees/ha in reference units), and a significantly higher proportion of LW recruitment from the
2886 buffer zones of the treatment sites than in the reference sites. LW recruitment based on the
2887 proportion of stand recruited (PSR) was significantly higher in the buffered units compared to
2888 the reference units. However, PSR from the inner 0-20 m was only 17% greater in the buffer
2889 units than in the reference units; while PSR of the outer unit (10 – 20 m) was more than double
2890 in the buffered units than in the reference units. From their analysis they also estimate that future

2891 potential supply of LW is diminished by ~10% in the buffered sites compared to the reference
2892 sites.

2893 *LW and stand age*

2894 Jackson and Wohl (2015) compared in-stream wood loads between old-growth (> 200 years) and
2895 young forests (age not reported). This study took place within the Arapaho and Roosevelt
2896 National Forests in Colorado. In-stream wood loads (m³/ha) were recorded for reaches in 10 old-
2897 growth forests and 23 young forests. Paired t- test or Kruskal-Wallis tests were used to check for
2898 significant differences in wood load. Results indicated that channel wood load (OG = 304.4 +
2899 161.1; Y = 197.8 + 245.5 m³ /ha), floodplain wood load (OG = 109.4 + 80; Y = 47.1 + 52.8 m³
2900 /ha), and total wood load (OG = 154.7 + 64.1; Y = 87.8 + 100.6 m³ /ha) per 100 m length of
2901 stream and were significantly higher in streams of old-growth forests than in young forests.
2902 Streams in old-growth forests also had significantly more wood in jams, and more total wood
2903 jams per unit length of channel than in younger forests (jam wood volume: OG = 7.10 +/- 6.9
2904 m³; Y = 1.71 +/- 2.81 m³)

2905 *Nutrient dynamics over time*

2906 Vanderbilt et al. (2003) investigated long-term datasets (ranging from 20-30 years) from six
2907 watersheds in the H.J. Andrews Experimental Watershed (HJA) in the west-central Cascade
2908 Mountains of Oregon. Their objective was to characterize long-term patterns of N dynamics in
2909 precipitation and stream water at the HJA. Patterns between nitrogen with precipitation and
2910 discharge were analyzed with logistic regression. Results showed that dissolved organic nitrogen
2911 (DON) concentrations increased in the fall in every watershed. The increase in concentration
2912 began in July or August with the earliest rain events, and peak DON concentrations occurred in
2913 October through December before the peak in the hydrograph. DON concentrations then
2914 declined during the winter months. However, other forms of N showed inconsistent patterns
2915 across all other watersheds. The authors conclude that total annual stream discharge was a
2916 positive predictor of DON output suggesting a relationship to precipitation. Also, DON had a
2917 consistent seasonal concentration pattern. All other forms of N observed showed variability and
2918 inconsistencies with annual and seasonal stream discharge. The authors speculate that different
2919 factors may control organic vs. inorganic N export. Specifically, DIN may be strongly influenced
2920 by terrestrial or in-stream biotic controls, while DON is more strongly influenced by climate.
2921 Last, the authors suggest that DON in streams may be recalcitrant, and largely unavailable to
2922 stream organisms.

2923

2924 *Focal Question 6*

2925 *6. Are there feedback mechanisms (e.g., microclimate changes within the riparian buffer) related*
2926 *to forest management that affect the recovery rates of riparian functions?*

2927 The studies considered appropriate for answering this question are those that quantify how forest
2928 management practices impact one or more factors that can in-turn impact the rate of recovery of
2929 riparian function. The regeneration, growth and development of vegetation within the riparian

2930 area following treatment can impact the rate of recovery of litter inputs, shade, sediment and
2931 nutrient filtration. Reduction in shade may affect the amount of light reaching the forest
2932 understory that then could impact productivity in the riparian area. Also, disturbance of soil and
2933 removal of vegetation during riparian management operations can impact streamflow and
2934 sediment supply, which in turn impacts sediment flux into streams. The studies summarized
2935 below provide experimental evidence in how these factors (e.g., vegetation productivity,
2936 streamflow discharge, sediment disturbance) are impacted by management.

2937 However, considering the second part of this question on how these feedback mechanisms affect
2938 the recovery rates of riparian function can only be inferred. To properly answer the full question
2939 a study would require an experimental design which 1) tracks the changes in site conditions (e.g.,
2940 microclimate, light availability to groundcover, exposed soil...) after treatment relative to
2941 untreated stands, 2) evaluates how these changes in site conditions lead to changes in stand
2942 development that can then impact function (e.g., vegetation), and finally 3) how these changes in
2943 development affect the recovery rates of function. This third step would require separating out
2944 the effect of these “feedback mechanism” so that the differences in recovery rates in treated
2945 stands with and without these effects (e.g., blocking newly available light to the understory) can
2946 be compared quantitatively. No studies that specifically, and entirely address these 3 objectives
2947 collectively could be found in the literature. Thus, the following reviewed studies provide
2948 evidence of how feedback mechanisms can affect function (e.g., increased light = increased
2949 primary productivity), but how these mechanisms affect the recovery rates of any particular
2950 function (e.g., timing of recovery with and without the feedback mechanism) can only be
2951 assumed.

2952 *Litter*

2953 Yeung et al. (2019) simulated post-harvest responses to leaf-litter derived coarse particulate
2954 organic matter (CPOM) quantity in a coastal rainforest stream in British Columbia. This study
2955 used a CPOM model that was calibrated using data from multiple published studies from,
2956 primarily the Pacific Northwest region, and several other North American regions. Calibration
2957 data included stream flow and temperature, and CPOM following different timber harvest
2958 intensities within 4 years of harvest. The model used estimated litterfall decreases of (-10%, -
2959 30%, -50%, -90%) for low, moderate, high, and very high basal area removal ; peak streamflow
2960 increases of +20%, +40%, +100%, +300%); and stream temperature increases of +1°C, +2°C,
2961 +4°C, and +6 °C. Treatment intensities in litterfall, peak flow, and stream temperature were
2962 modeled and analyzed individually and cumulatively to estimate their relative and combined
2963 effects on in-stream CPOM standing stocks. Results of the model showed that, in general, the
2964 standing stocks of CPOM decreased under the independent effects of reduced litterfall and
2965 elevated peak flows and increased with higher stream temperatures.

2966 Along the gradient of increasing timber removal, litterfall reductions on depleting CPOM
2967 standing stocks were at least an order of magnitude greater than those of elevated peak flows.
2968 The magnitude of CPOM changes induced by litterfall reductions was consistently greater than
2969 stream temperature increases, but their differences in magnitude became smaller at higher levels
2970 of disturbance severity. Only the effects of litterfall-temperature interactions on CPOM standing

2971 stocks were significant ($p < 0.001$). The authors interpret these results as evidence that litterfall
 2972 reduction from timber harvest was the strongest control on in-stream CPOM quantity for 4 years
 2973 post-harvest. However, the authors propose that the decreased activity of CPOM consumers
 2974 caused by increasing stream temperatures may be enough to offset the loss of litterfall inputs on
 2975 standing CPOM stocks. The caveat of this study is that it did not include LW dynamics in
 2976 preserving CPOM post-harvest. There is evidence that in-stream LW can act as a catchment for
 2977 CPOM (May & Gresswell, 2003; Richardson et al. 2007).

2978 *Sediment*

2979 Table 21. Treatment and responses for selected publications investigating Sediment relevant to
 2980 Q5.

Reference	Treatment	Response
Safeeq 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.	<u>1-year post-harvest data (n = 200 harvest units)</u> The authors conclude that in general, USFS riparian forest harvest practices are effective in reducing the development of sediment delivery pathways. They also interpret these results as evidence that skid trails should be directed away from streams , maintain surface roughness, and promptly decommissioned.

2981

2982 Safeeq et al. (2020) analyzed a long-term data set to changes in streamflow, and suspended
 2983 sediment load and sediment bedload in streams between two watersheds; one with a history of
 2984 timber management and one with no history of timber management. The two watersheds were
 2985 located in the H.J. Andrews Experimental Forest and were paired by size, aspect, and
 2986 topography. The treatment watershed was 100% clearcut during the period from 1962-1966,
 2987 broadcast burned in 1966 and re-seeded in 1968. Streamflow and sediment data were taken
 2988 intermittently; suspended sediment data after large storm events between 1952 (pre-harvest) and
 2989 1988; and sediment bedload in 2016. The researchers used a reverse regression technique to
 2990 evaluate the relative and absolute importance of changes in streamflow versus changes in
 2991 sediment supply from timber harvest on sediment transport. There were no significant changes in
 2992 precipitation patterns before or after harvest. The results for post-treatment sediment yields
 2993 showed suspended load declined to pre-treatment levels in the first two decades following
 2994 treatment and bedload remained elevated, causing the bedload proportion of the total load to
 2995 increase through time. Changes in streamflow alone account for 477 Mg/km² (10%) of the
 2996 suspended load and 113 Mg/km² (5%) of the bedload over the post-treatment period. Increase in
 2997 suspended sediment yield due to increase in sediment supply from timber harvest activities was
 2998 84% of the measured post-treatment total suspended sediment yield. The authors estimate that
 2999 following harvest, changes on streamflow alone was estimated in being responsible for < 10% of

3000 the resulting suspended sediment transported into streams, while the increase in sediment supply
3001 due to harvest disturbance was responsible for >90%. Thus, while timber harvest-induced
3002 increases in streamflow does increase sediment transport, it is negligible compared to the
3003 increase in sediment source created from management practices.

3004 Litschert & MacDonald (2009) investigated the frequency of sediment delivery pathways in
3005 riparian management areas and their physical characteristics and connectivity following harvest.
3006 In this study the authors describe sediment delivery pathways (“features”) as rills, gullies, and
3007 sediment plumes that form when excess sediment relative to overland flows transports sediment
3008 from the hillslope to the stream. The authors surveyed 200 riparian management areas (RMA) in
3009 four different National Forests of the Sierra Nevada and Cascade Mountains of California. USFS
3010 policy requires 90-m wide RMA along each side of perennial streams and 45-m wide RMA along
3011 each side of all ephemeral and intermittent streams. When features were found within an RMA,
3012 data for years since harvest, soil depth, soil erodibility (K), feature length, feature gradient,
3013 aspect, elevation, hillslope gradient, hillslope curvature, surface roughness, and connectivity
3014 were recorded for analysis. Association between these variables were analyzed with a
3015 Spearman’s rank correlation. The variables most strongly associated with feature length were
3016 used to develop a multiple linear regression model to predict feature length. Only 19 of the 200
3017 harvest units had sediment development pathways. Feature pathways ranged in age (time since
3018 harvest) from 2 to 18 years, and in length from 10 m to 220 m. Of the 19 feature pathways, only
3019 six were connected to streams, and five of those originated from skid trails. Feature pathway
3020 length was significantly related to mean annual precipitation, cosine of the aspect, elevation, and
3021 hillslope gradient ($R^2 = 64\%$, $p = 0.004$). The authors conclude that in general, USFS riparian
3022 forest harvest practices are effective in reducing the development of sediment delivery pathways.
3023 They also interpret these results as evidence that skid trails should be directed away from
3024 streams, maintain surface roughness, and promptly decommissioned.

3025

3026 *Impacts on Microclimate*

3027 Anderson et al. (2007) compared changes in understory microclimate above the stream, within
3028 the channel, and within the riparian area between thinned and unthinned riparian stands. The
3029 focus of this study was on second-growth (30- to 80-year-old) riparian Douglas-fir forests along
3030 headwater streams in the western Oregon Coast and Cascade Range. Stands were either thinned
3031 to approximately 198 trees per acre (TPA) or were left unthinned and ranged from 500-865 TPA.
3032 Streams within treated stands were surrounded by buffers of either 1) one site-potential tree
3033 averaging 69 m (B1, B1-T thinned and unthinned respectively), 2) variable width buffer
3034 averaging 22 m (VB, and VB-T), or 3) streamside retention buffer averaging 9 m (SR, and SR-
3035 T). Further, directly adjacent randomly selected B1-T and VB-T buffers patch openings (0.4 ha)
3036 were created (B1-P, VB-P). Microsite and microclimate responses were repeat sampled for each
3037 treatment and compared with untreated stands (UT). Within the riparian buffer zones, daily
3038 maximum temperatures were higher in all treated stands when compared to UT stands. The
3039 differences in daily maximum temperatures between treated and untreated stands ranged from
3040 1.1°C (B1) to 4.0°C (SR-T), but the difference was only significant in one SR-T stand. Daily
3041 maximum air temperature within buffer zones adjacent to patch openings were 3.5°C higher than

3042 in UT stands. Within patch openings daily maximum temperatures were on average 6 to 9°C
3043 higher than in UT stands. Soil temperature changes were only evident within patch openings
3044 ranging from 3.6 - 8.8°C higher than in UT stands. VB-T buffers that were 15 m wide or wider
3045 exhibited changes in daily maximum air temperature above stream centers <1°C and daily
3046 minimum relative humidity <5% lower than in untreated stands. The authors conclude that in
3047 general, thinned stands are warmer and drier than unthinned stands. However, the results for
3048 differences in microclimate were only significant in narrow (9 m) thinned buffers and patch
3049 openings.

3050 Anderson & Meleason (2009) conducted a companion study to Anderson et al. (2007) and
3051 compared changes in small (5-29 cm diameter) and large (≥ 30 cm diameter) downed wood
3052 abundance and understory vegetation between treated and untreated stands 5 years after harvest.
3053 Treatments compared were the same as those described in Anderson et al. (2007) discussed
3054 above. The results for small and large downed wood were highly variable between pre- and post-
3055 harvest periods and between treatments but the authors speculate from trends in the data that
3056 both wood and vegetation responses within buffers ≥ 15 m wide were insensitive to treatments.
3057 The strongest contrast in rate of change in herb cover was between the SR-T and VB-T buffers
3058 with higher herbaceous cover in the SR-T buffers and highest in SR-T buffers adjacent to patch
3059 openings. The authors conclude that in general these thinning treatments only led to subtle
3060 changes in understory vegetation cover and composition. Because of the high variability in
3061 responses among and between treatments significance could not be confirmed. The authors
3062 further conclude that a better functional understanding of the changes in ecological processes
3063 associated with changes in habitat characteristics following changes in understory wood and
3064 vegetation cover is needed to help discern ecological significance.

3065

3066 [Focal Question 7](#)

3067 *7. What major data gaps and uncertainties exist relative to effects of timber harvest (both*
3068 *riparian and adjacent upland) on the riparian functions?*

3069 No studies that provide experimental evidence that quantifies how specific treatments within the
3070 riparian area affect bank stability were found based on our search criteria (published after 2000,
3071 conducted in western North America). However, this may be because bank erosion relates
3072 directly to sediment transport and thus bank stability is inferred by the magnitude of change in
3073 sediment export. Furthermore, the importance of vegetation retention and equipment exclusion in
3074 areas closest to the stream for maintaining bank stability appears to be well understood
3075 considering its prevalence in riparian forest management plans ([WAC 222-30-022](#); WAC 22-30-
3076 021; 2022 [ODF](#); IDAPA 20.02.01).

3077 Our search of the literature focused on how treatments within or adjacent to forested riparian
3078 areas impact one or more of the riparian functions. Most of the studies found in our search focus
3079 on the impacts of riparian treatment on LW and shade (commonly coupled with stream
3080 temperature). There is also a significant body of research that considers the impact of harvest on

3081 nutrient and sediment flux into streams. Fewer studies could be found that quantify changes in
3082 litter input following riparian management.

3083 While few studies could be found that provide direct experimental evidence of how bank
3084 stability is affected by timber harvest, two studies were found that compared the relative
3085 influence of different factors on bank stability. Both of which showed evidence that bank
3086 stability is influenced by the type of vegetation dominating the riparian area. Rood et al. (2015)
3087 compared the relative erosion resistance of riverbanks occupied by forests versus grassland along
3088 the Elk River in British Columbia, Canada. This study used a combination of field sampling and
3089 aerial photo analysis from 1995 to 2013 to estimate the differences in channel migration between
3090 forest and grass dominated riparian areas. Relative tree cover was binned into 5 categories
3091 ranging from (1) no trees to (5) completely treed. Relative channel change was binned into 2
3092 categories as ‘moderate change’ for channels that migrated between 45 and 75 m, and as ‘major
3093 change’ for channels that migrated more than 75 m. Chi square analysis was used to assess the
3094 distributions of vegetation of channels with moderate and major changes. Results of the chi
3095 square analysis showed that the distribution of the observed vegetation types differed
3096 significantly ($p < 0.05$) by channel change categories. Of the 15 sites assessed with moderate or
3097 major erosion (changes), 7 were along banks dominated by grasslands without trees (‘1’), four
3098 were assessed as a ‘2’, with some trees, and three were in a ‘3’ with a mixed zone of similar
3099 proportions of trees and clearing. Only one site with a ‘4’ showed a moderate amount of change.
3100 The authors interpret these results as evidence that trees are better than grass at stabilizing banks,
3101 and that stability increases with tree cover.

3102 Outside of the U.S., Krzeminska et al. (2019), investigated the effect of different types of
3103 riparian vegetation on stream bank stability in a small agricultural catchment in South-Eastern
3104 Norway. The dominating soil type within the catchment is coarse moraine in the forested areas
3105 and marine deposits with silt loam and silty clay loam texture in agriculture areas. The
3106 researchers used a combination of field collected data with stream bank stability modeling using
3107 Bank-Stability and Toe-Erosion Modeling (BSTEM). Three experimental plots were established,
3108 one for each dominant vegetation type, grass dominated, shrub dominated, and tree dominated.
3109 Investigations of in-situ undrained shear strength of the root-reinforced soil were done with a
3110 Field Inspection Vane Tester. Additionally, potential changes in the bank profile were monitored
3111 with a series of erosion pins, 6 pins per each plot. Changes in root cohesion and % cover over
3112 time for each vegetation type were estimated using the RipRoots sub-model in BSTEM. Their
3113 results showed a difference in bank stability based on vegetation type, that varied seasonally with
3114 groundwater level and stream water level. The grass dominated and tree dominated plots,
3115 specifically, showed the lowest estimated stability during spring (March to April) and early
3116 autumn (September to November), and the highest estimated stability during the summer months
3117 (May-June). This seasonal trend was also observed for the shrub plots but not as strongly.
3118 Steeper slopes in the grass and shrub dominated plots showed a trend of reduced stability for
3119 plots 54° slopes showing potential for failure. The tree dominated plots showed a trend of lower
3120 stability for steeper slopes, however, it wasn’t as strong of a trend and the model did not predict
3121 potential for failure or ‘instability’. Regardless of season, groundwater levels, or slope steepness
3122 the tree plots showed the highest estimated bank stability overall.

3123 These two studies that investigate bank stability use methods which could be applied to an
3124 experimental design that also considers differences in stability between treated (harvested) and
3125 untreated stands. The combination of field observation and simulation modeling used by
3126 Krzeminska et al. (2019), especially, could be used to estimate how timber harvest affects bank
3127 stability (or erosion) while also accounting for geomorphic and hydrological differences.

3128 Considering the topics included in the focal questions, studies that investigate the effects of
3129 clearcut gaps, and studies that quantify how treatment within the riparian zone relates to
3130 resilience to fire had the fewest studies providing experimental evidence. Other than the Hard
3131 Rock and Soft Rock studies, only 2 other studies (Janisch et al., 2012, Swartz et al., 2020) were
3132 found that investigate the effects of similar buffer treatment designs (patched buffers and riparian
3133 canopy gaps). For how treatments within the riparian zone relate to resilience to fire, there were
3134 no studies that provide experimental evidence on this topic based on the search criteria. Some
3135 studies were found to quantify the probability of fire or fire severity within riparian zones in
3136 general (Reeves et al. 2006; Van de Water & North, 2011). However, none compares the
3137 resilience of riparian stands between treated and untreated stands after fire. One study, Ceder et
3138 al. (2018) used simulation modeling to compare fire susceptibility between managed and
3139 unmanaged stands and has been included in focal question 4.

3140 Indeed, Stone et al. (2010) surveyed fire management officers from 55 national forests across 11
3141 western states and found that fewer than half (43%) of them indicated that they were conducting
3142 fuel reduction treatments in riparian areas. The primary objective for most of these treatments
3143 involved some form of fuel reduction (83%), while others focused on multiple objectives such as
3144 ecological restoration and habitat improvement. Most of these treatments (93%) were of small
3145 extent (< 300 acres) and occurred in the wildland urban interface (73%). The authors conclude
3146 that these results are promising, but that well-designed monitoring programs are needed to
3147 estimate the consequences of these treatments on fire risk and other ecological effects.

3148 The study from Prichard et al. (2020), discussed in question 3, used a combination of
3149 simultaneous autoregression (SAR) and random forest (RF) modeling approaches to model the
3150 drivers of fire severity and the effectiveness of fuel treatments in mitigating fire severity in the
3151 2014 Carlton Complex. Results from this study provided evidence on how vegetation (based on
3152 broad LANDFIRE classifications), topography, and different fuel treatments (e.g., thinning only,
3153 thin and pile burn, thin and broadcast burn, etc.) related to fire severity and fire spread. This
3154 approach has potential to be used in riparian areas burned by wildfires. In terms of the topic of
3155 how various treatments relate to riparian forest resistance and resilience to fire would require
3156 using a dataset of riparian forest stand characteristics that includes information on fuel
3157 treatments, time since last fire, and basin characteristics. This information could be used along
3158 with spatial information of burn severity immediately following a fire.

3159 [References](#)

3160 Anderson, P. D., & Meleason, M. A. (2009). Discerning responses of down wood and understory
3161 vegetation abundance to riparian buffer width and thinning treatments: an equivalence–inequivalence
3162 approach. *Canadian Journal of Forest Research*, 39(12), 2470-2485.

- 3163 Anderson, P. D., Larson, D. J., & Chan, S. S. (2007). Riparian buffer and density management influences on
3164 microclimate of young headwater forests of western Oregon. *Forest Science*, 53(2), 254-269.
- 3165 Arkle, R. S., & Pilliod, D. S. (2010). Prescribed fires as ecological surrogates for wildfires: a stream and
3166 riparian perspective. *Forest Ecology and Management*, 259(5), 893-903.
- 3167 Bahuguna, D., Mitchell, S. J., & Miquelajauregui, Y. (2010). Windthrow and recruitment of large woody
3168 debris in riparian stands. *Forest Ecology and Management*, 259(10), 2048-2055.
- 3169 Benda, L. E., Litschert, S. E., Reeves, G., & Pabst, R. (2016). Thinning and in-stream wood recruitment in
3170 riparian second growth forests in coastal Oregon and the use of buffers and tree tipping as
3171 mitigation. *Journal of forestry research*, 27(4), 821-836.
- 3172 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., &
3173 Dunne, T. (2003, January). Wood recruitment processes and wood budgeting. In *American Fisheries*
3174 *Society Symposium* (pp. 49-74). American Fisheries Society.
- 3175 Beschta, R. L., Bilby, R.E.Brown, G.W., Holtby, L.B., and Hofstra., T.D., 1987. Stream Temperature and
3176 Aquatic Habitat: Fisheries and Forestry Interactions. In: *Streamside Management: Forestry and Fisheries*
3177 *Interactions*. E. O. Salo and T. W. Cundy (Editors). Contribution No. 57, University of Washington, Institute
3178 of Forest Resources, 471 pp.
- 3179 Bilby, R. E., & Heffner, J. T. (2016). Factors influencing litter delivery to streams. *Forest Ecology and*
3180 *Management*, 369, 29–37. <https://doi.org/10.1016/j.foreco.2016.03.031>
- 3181 Bilby, R. E., Sullivan, K., & Duncan, S. H. (1989). The generation and fate of road-surface sediment in
3182 forested watersheds in southwestern Washington. *Forest Science*, 35(2), 453-468.
- 3183 Bjornn, T. C., & Reiser, D. W. (1991). Habitat requirements of salmonids in streams. *American Fisheries*
3184 *Society Special Publication*, 19(837), 138.
- 3185 Bladon, K. D., Cook, N. A., Light, J. T., & Segura, C. (2016). A catchment-scale assessment of stream
3186 temperature response to contemporary forest harvesting in the Oregon Coast Range. *Forest Ecology and*
3187 *Management*, 379, 153-164.
- 3188 Bladon, K. D., Segura, C., Cook, N. A., Bywater-Reyes, S., & Reiter, M. (2018). A multicatchment analysis of
3189 headwater and downstream temperature effects from contemporary forest harvesting. *Hydrological*
3190 *Processes*, 32(2), 293-304.
- 3191 Brown, G. W. (1983). *Forestry and water quality*. *Forestry and water quality*, (Ed. 2).
- 3192 Brown, G. W., Gahler, A. R., & Marston, R. B. (1973). Nutrient losses after clear-cut logging and slash
3193 burning in the Oregon Coast Range. *Water Resources Research*, 9(5), 1450-1453.
- 3194 Burton, J. I., Olson, D. H., & Puettmann, K. J. (2016). Effects of riparian buffer width on wood loading in
3195 headwater streams after repeated forest thinning. *Forest Ecology and Management*, 372, 247–257.
3196 <https://doi.org/10.1016/j.foreco.2016.03.053>
- 3197 Bywater-Reyes, S., Bladon, K. D., & Segura, C. (2018). Relative influence of landscape variables and
3198 discharge on suspended sediment yields in temperate mountain catchments. *Water Resources*
3199 *Research*, 54(7), 5126-5142.

- 3200 Bywater-Reyes, S., Segura, C., & Bladon, K. D. (2017). Geology and geomorphology control suspended
3201 sediment yield and modulate increases following timber harvest in temperate headwater
3202 streams. *Journal of Hydrology*, 548, 754-769.
- 3203 Camp, A., C. Oliver, P. Hessburg, and R. Everett. (1997). Predicting late-successional fire refugia pre-dating
3204 European settlement in the Wenatchee Mountains. *Forest Ecology and Management* 95 63-77
- 3205 Chan, S., P. Anderson, J. Cissel, L. Lateen. and C. Thompson. 2004. Variable density management in
3206 Riparian Reserves: lessons learned from an operational study in managed forests of western Oregon,
3207 USA. *For. Snow Landsc. Res.* 78,1/2:151-172.
- 3208 Chapman, D. W., & Bjornn, T. C. (1969). Distribution of salmonids in streams. In *Symp. Salmon Trout*
3209 *Streams*. Institute of Fisheries, University of British Columbia, Vancouver (pp. 153-176).
- 3210 Chen, X., Wei, X., & Scherer, R. (2005). Influence of wildfire and harvest on biomass, carbon pool, and
3211 decomposition of large woody debris in forested streams of southern interior British Columbia. *Forest*
3212 *Ecology and Management*, 208(1-3), 101-114.
- 3213 Chen, X., Wei, X., Scherer, R., Luider, C., & Darlington, W. (2006). A watershed scale assessment of in-
3214 stream large woody debris patterns in the southern interior of British Columbia. *Forest Ecology and*
3215 *Management*, 229(1-3), 50-62.
- 3216 Chesney, C. (2000). Functions of wood in small, steep streams in eastern Washington: Summary of
3217 results for project activity in the Ahtanum, Cowiche, and Tieton basins. *Timber, Fish, Wildlife*.
- 3218 CMER 03-308 (2004) Review of the Available Literature Related to Wood Loading Dynamics in and
3219 around Stream in eastern Washington Forests
- 3220 Cole, E., & Newton, M. (2013). Influence of streamside buffers on stream temperature response
3221 following clear-cut harvesting in western Oregon. *Canadian journal of forest research*, 43(11), 993-1005.
- 3222 Cooper, J. R., Gilliam, J. W., Daniels, R. B., & Robarge, W. P. (1987). Riparian areas as filters for agricultural
3223 sediment. *Soil science society of America journal*, 51(2), 416-420.
- 3224 Crandall, T., Jones, E., Greenhalgh, M., Frei, R. J., Griffin, N., Severe, E., ... & Abbott, B. W. (2021).
3225 Megafire affects stream sediment flux and dissolved organic matter reactivity, but land use dominates
3226 nutrient dynamics in semiarid watersheds. *PloS one*, 16(9), e0257733.
- 3227 Cupp, C.E. and T.J. Lofgren. 2014. Effectiveness of riparian management zone prescriptions in protecting
3228 and maintaining shade and water temperature in forested streams of Eastern Washington. *Cooperative*
3229 *Monitoring Evaluation and Research Report CMER 02-212*. Washington State Forest Practices Adaptive
3230 *Management Program*. Washington Department of Natural Resources, Olympia, WA.
- 3231 Deval, C., Brooks, E. S., Gravelle, J. A., Link, T. E., Dobre, M., & Elliot, W. J. (2021). Long-term response in
3232 nutrient load from commercial forest management operations in a mountainous watershed. *Forest*
3233 *Ecology and Management*, 494, 119312.
- 3234 Devotta, D. A., Fraterrigo, J. M., Walsh, P. B., Lowe, S., Sewell, D. K., Schindler, D. E., & Hu, F. S. (2021).
3235 Watershed *Alnus* cover alters N: P stoichiometry and intensifies P limitation in subarctic
3236 streams. *Biogeochemistry*, 153(2), 155-176.

- 3237 Dwire, K. A., & Kauffman, J. B. (2003). Fire and riparian ecosystems in landscapes of the western USA.
3238 *Forest Ecology and Management*, 178(1-2), 61-74.
- 3239 Ebersole, J. L., Liss, W. J., & Frissell, C. A. (2001). Relationship between stream temperature, thermal
3240 refugia and rainbow trout *Oncorhynchus mykiss* abundance in arid-land streams in the northwestern
3241 United States. *Ecology of freshwater fish*, 10(1), 1-10.
- 3242 Ehinger, W.J., W.D. Bretherton, S.M. Estrella, G. Stewart, D.E. Schuett-Hames, and S.A. Nelson. 2021.
3243 Effectiveness of Forest Practices Buffer Prescriptions on Perennial Non-fish-bearing Streams on Marine
3244 Sedimentary Lithologies in Western Washington. Cooperative Monitoring, Evaluation, and Research
3245 Committee Report CMER 2021.08.24, Washington State Forest Practices Adaptive Management
3246 Program, Washington Department of Natural Resources, Olympia, WA.
- 3247 Everett, R., Schellhaas, R., Ohlson, P., Spurbeck, D., & Keenum, D. (2003). Continuity in fire disturbance
3248 between riparian and adjacent sideslope Douglas-fir forests. *Forest Ecology and Management*, 175(1-3),
3249 31-47.
- 3250 Fox, M. J. (2001). A new look at the quantities and volumes of instream wood in forested basins within
3251 Washington State (Doctoral dissertation, University of Washington).
- 3252 Fox, M., & Bolton, S. (2007). A regional and geomorphic reference for quantities and volumes of instream
3253 wood in unmanaged forested basins of Washington State. *North American Journal of Fisheries
3254 Management*, 27(1), 342-359.
- 3255 Fox, M., Bolton, S., & Conquest, L. (2003). 14Reference conditions for instream wood in western
3256 Washington. University of Washington Press: Seattle, WA, 361-393.
- 3257 Fratkin, M. M., Segura, C., & Bywater-Reyes, S. (2020). The influence of lithology on channel geometry
3258 and bed sediment organization in mountainous hillslope-coupled streams. *Earth Surface Processes and
3259 Landforms*, 45(10), 2365-2379.
- 3260 Fredriksen, R. L. (1975). Nitrogen, phosphorus and particulate matter budgets of five coniferous forest
3261 ecosystems in the western Cascades Range, Oregon.
- 3262 Gomi, T., Dan Moore, R., & Hassan, M. A. (2005). Suspended sediment dynamics in small forest streams
3263 of the Pacific Northwest 1. *JAWRA Journal of the American Water Resources Association*, 41(4), 877-898.
- 3264 Gomi, T., Sidle, R. C., Bryant, M. D., & Woodsmith, R. D. (2001). The characteristics of woody debris and
3265 sediment distribution in headwater streams, southeastern Alaska. *Canadian Journal of Forest
3266 Research*, 31(8), 1386-1399.
- 3267 Gravelle, J. A., & Link, T. E. (2007). Influence of timber harvesting on headwater peak stream
3268 temperatures in a northern Idaho watershed. *Forest Science*, 53(2), 189-205.
- 3269 Gravelle, J. A., Ice, G., Link, T. E., & Cook, D. L. (2009). Nutrient concentration dynamics in an inland
3270 Pacific Northwest watershed before and after timber harvest. *Forest Ecology and Management*, 257(8),
3271 1663-1675.
- 3272 Gregory, S. (1990). Riparian management guide: Willamette National Forest. US Department of
3273 Agriculture, Forest Service, Pacific Northwest Region.

- 3274 Gresswell, R. E., Barton, B. A., & Kershner, J. L. (Eds.). (1989). *Practical approaches to riparian resource*
 3275 *management: an educational workshop: May 8-11, 1989, Billings, Montana*. US Bureau of Land
 3276 Management.
- 3277 Groom, J. D., Dent, L., Madsen, L. J., & Fleuret, J. (2011b). Response of western Oregon (USA) stream
 3278 temperatures to contemporary forest management. *Forest Ecology and Management*, 262(8), 1618-
 3279 1629.
- 3280 Groom, J. D., L. Dent, and L. J. Madsen. (2011a). Stream temperature change detection for state and
 3281 private forests in the Oregon Coast Range, *Water Resour. Res.* 47, W01501
- 3282 Guenther, S. M., Gomi, T., & Moore, R. D. (2014). Stream and bed temperature variability in a coastal
 3283 headwater catchment: influences of surface-subsurface interactions and partial-retention forest
 3284 harvesting. *Hydrological Processes*, 28(3), 1238-1249.
- 3285 Harmon, M. E., Franklin, J. F., Swanson, F. J., Sollins, P., Gregory, S. V., Lattin, J. D., ... & Cummins, K. W.
 3286 (1986). Ecology of coarse woody debris in temperate ecosystems. *Advances in ecological research*, 15,
 3287 133-302.
- 3288 Hart, Stephanie K., David E. Hibbs, and Steven S. Perakis. (2013) "Riparian litter inputs to streams in the
 3289 central Oregon Coast Range." *Freshwater Science* 32.1 (2013): 343-358.
- 3290 Hartman, G. F., & Scrivener, J. C. (1990). Impacts of forestry practices on a coastal stream ecosystem,
 3291 Carnation Creek, British Columbia.
- 3292 Hassan, M. A., Church, M., Lisle, T. E., Brardinoni, F., Benda, L., & Grant, G. E. (2005). Sediment transport
 3293 and channel morphology of small, forested streams 1. *Jawra journal of the american water resources*
 3294 *association*, 41(4), 853-876.
- 3295 Hatten, J. A., Segura, C., Bladon, K. D., Hale, V. C., Ice, G. G., & Stednick, J. D. (2018). Effects of
 3296 contemporary forest harvesting on suspended sediment in the Oregon Coast Range: Alsea Watershed
 3297 Study Revisited. *Forest Ecology and Management*, 408, 238-248.
- 3298 Hoffmann, C. C., Kjaergaard, C., Uusi-Kämppe, J., Hansen, H. C. B., & Kronvang, B. (2009). Phosphorus
 3299 Retention in Riparian Buffers: Review of Their Efficiency. *Journal of Environmental Quality*, 38(5), 1942-
 3300 1955. <https://doi.org/10.2134/jeq2008.0087>
- 3301 Hough-Snee, N., Kasprak, A., Rossi, R. K., Bouwes, N., Roper, B. B., & Wheaton, J. M. (2016).
 3302 Hydrogeomorphic and Biotic Drivers of Instream Wood Differ Across Sub-basins of the Columbia River
 3303 Basin, USA. *River Research and Applications*, 32(6), 1302-1315. <https://doi.org/10.1002/rra.2968>
- 3304 Hunter, M. A., & Quinn, T. (2009). Summer water temperatures in alluvial and bedrock channels of the
 3305 Olympic Peninsula. *Western Journal of Applied Forestry*, 24(2), 103-108.
- 3306 Hyatt, T. L., & Naiman, R. J. (2001). The residence time of large woody debris in the Queets River,
 3307 Washington, USA. *Ecological Applications*, 11(1), 191-202.
- 3308 Jackson, C. R., C. A. Sturm, and J. M. Ward. 2001. Timber harvest impacts on small headwater stream
 3309 channels in the coast ranges of Washington. *JAWRA Journal of the American Water Resources*
 3310 *Association* 37(6):1533-1549.

- 3311 Jackson, K. J., & Wohl, E. (2015). Instream wood loads in montane forest streams of the Colorado Front
3312 Range, USA. *Geomorphology*, 234, 161-170.
- 3313 Janisch, J. E., Wondzell, S. M., & Ehinger, W. J. (2012). Headwater stream temperature: Interpreting
3314 response after logging, with and without riparian buffers, Washington, USA. *Forest Ecology and*
3315 *Management*, 270, 302-313.
- 3316 Johnson, S. L., & Jones, J. A. (2000). Stream temperature responses to forest harvest and debris flows in
3317 western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences*, 57(S2), 30-39.
- 3318 Karwan, D. L., Gravelle, J. A., & Hubbart, J. A. (2007). Effects of timber harvest on suspended sediment
3319 loads in Mica Creek, Idaho. *Forest Science*, 53(2), 181-188.
- 3320 Kaylor, M. J., Warren, D. R., & Kiffney, P. M. (2017). Long-term effects of riparian forest harvest on light in
3321 Pacific Northwest (USA) streams. *Freshwater Science*, 36(1), 1-13.
- 3322 Kibler, K. M., Skaugset, A., Ganio, L. M., & Huso, M. M. (2013). Effect of contemporary forest harvesting
3323 practices on headwater stream temperatures: Initial response of the Hinkle Creek catchment, Pacific
3324 Northwest, USA. *Forest ecology and management*, 310, 680-691.
- 3325 Kiffney, P., and J. Richardson. 2010. Organic matter inputs into headwater streams of southwestern
3326 British Columbia as a function of riparian reserves and time since harvesting. *Forest Ecology and*
3327 *Management*. 260:1931-1942.
- 3328 Knight, S.M. (1990). Forest harvesting impacts on coarse woody debris and channel form in central
3329 Oregon streams. M.S. Thesis. Oregon State University, Corvallis, Oregon.
- 3330 Lewis, D. D. (1998, April). Naive (Bayes) at forty: The independence assumption in information retrieval.
3331 In *European conference on machine learning* (pp. 4-15). Berlin, Heidelberg: Springer Berlin Heidelberg.
- 3332 Liquori, M. K. (2006). POST-HARVEST RIPARIAN BUFFER RESPONSE: IMPLICATIONS FOR WOOD
3333 RECRUITMENT MODELING AND BUFFER DESIGN 1. *JAWRA Journal of the American Water Resources*
3334 *Association*, 42(1), 177-189.
- 3335 Litschert, S. E., & MacDonald, L. H. (2009). Frequency and characteristics of sediment delivery pathways
3336 from forest harvest units to streams. *Forest Ecology and Management*, 259(2), 143-150.
- 3337 Macdonald, J. S., Beaudry, P. G., MacIsaac, E. A., & Herunter, H. E. (2003). The effects of forest harvesting
3338 and best management practices on streamflow and suspended sediment concentrations during
3339 snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada. *Canadian Journal of*
3340 *Forest Research*, 33(8), 1397-1407.
- 3341 Macdonald, J. S., MacIsaac, E. A., & Herunter, H. E. (2003b). The effect of variable-retention riparian
3342 buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of
3343 British Columbia. *Canadian journal of forest research*, 33(8), 1371-1382.
- 3344 Martin, D. J., & Grotefendt, R. A. (2007). Stand mortality in buffer strips and the supply of woody debris
3345 to streams in Southeast Alaska. *Canadian Journal of Forest Research*, 37(1), 36-49.

- 3346 Martin, D. J., Kroll, A. J., & Knoth, J. L. (2021). An evidence-based review of the effectiveness of riparian
3347 buffers to maintain stream temperature and stream-associated amphibian populations in the Pacific
3348 Northwest of Canada and the United States. *Forest Ecology and Management*, 491, 119190.
- 3349 May, C. L., & Gresswell, R. E. (2003). Large wood recruitment and redistribution in headwater streams in
3350 the southern Oregon Coast Range, USA. *Canadian Journal of Forest Research*, 33(8), 1352-1362.
- 3351 McDade, M, F. Swanson, W. McKee, J. Franklin and J. Van Sickle. (1990). Source distances for coarse
3352 woody debris entering small streams in western Oregon and Washington. *Canadian Journal of Forest*
3353 *Resources* 20, 326-330
- 3354 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn (technical
3355 coordinators). 2018. Effectiveness of Experimental Riparian Buffers on Perennial Non-fish-bearing
3356 Streams on Competent Lithologies in Western Washington. Cooperative Monitoring, Evaluation and
3357 Research Report CMER 18-100, Washington State Forest Practices Adaptive Management Program,
3358 Washington Department of Natural Resources, Olympia, WA.
- 3359 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D.E. Schuett-Hames, R. Ojala-Barbour, G. Stewart
3360 and T. Quinn (technical coordinators). 2021. Effectiveness of experimental riparian buffers on perennial
3361 non-fish-bearing streams on competent lithologies in western Washington – Phase 2 (9 years after
3362 harvest). Cooperative Monitoring, Evaluation and Research Report CMER 2021.07.27, Washington State
3363 Forest Practices Adaptive Management Program, Washington Department of Natural Resources,
3364 Olympia, WA.
- 3365 Meleason, M. A., Gregory, S. V., & Bolte, J. P. (2003). Implications of riparian management strategies on
3366 wood in streams of the Pacific Northwest. *Ecological Applications*, 13(5), 1212-1221.
- 3367 Merschel, A. G., Spies, T. A., & Heyerdahl, E. K. (2014). Mixed-conifer forests of central Oregon: effects of
3368 logging and fire exclusion vary with environment. *Ecological Applications*, 24(7), 1670-1688.
- 3369 Mueller, E. R., & Pitlick, J. (2013). Sediment supply and channel morphology in mountain river systems: 1.
3370 Relative importance of lithology, topography, and climate. *Journal of Geophysical Research: Earth*
3371 *Surface*, 118(4), 2325-2342.
- 3372 Murray, G. L. D., Edmonds, R. L., & Marra, J. L. (2000). Influence of partial harvesting on stream
3373 temperatures, chemistry, and turbidity in forests on the western Olympic Peninsula,
3374 Washington. *Northwest science.*, 74(2), 151-164.
- 3375 Naiman, R. J., Fetherston, K. L., McKay, S. J., & Chen, J. (1998). Riparian forests. *River ecology and*
3376 *management: lessons from the Pacific Coastal Ecoregion*, 289-323.
- 3377 Nowakowski, A. L., & Wohl, E. (2008). Influences on wood load in mountain streams of the Bighorn
3378 National Forest, Wyoming, USA. *Environmental Management*, 42(4), 557-571.
- 3379 Polyakov, V., Fares, A., & Ryder, M. H. (2005). Precision riparian buffers for the control of nonpoint source
3380 pollutant loading into surface water: A review. *Environmental Reviews*, 13(3), 129-144.
- 3381 Puntteney-Desmond, K. C., Bladon, K. D., & Silins, U. (2020). Runoff and sediment production from
3382 harvested hillslopes and the riparian area during high intensity rainfall events. *Journal of Hydrology*, 582,
3383 124452.

- 3384 Quinn, T., G.F. Wilhere, and K.L. Krueger, technical editors. (2020). Riparian Ecosystems, Volume 1:
3385 Science Synthesis and Management Implications. Habitat Program, Washington Department of Fish and
3386 Wildlife, Olympia.
- 3387 Rachels, A. A., Bladon, K. D., Bywater-Reyes, S., & Hatten, J. A. (2020). Quantifying effects of forest
3388 harvesting on sources of suspended sediment to an Oregon Coast Range headwater stream. *Forest*
3389 *Ecology and Management*, 466, 118123.
- 3390 Reid, D. A., & Hassan, M. A. (2020). Response of in-stream wood to riparian timber harvesting: Field
3391 observations and long-term projections. *Water Resources Research*, 56(8), e2020WR027077.
- 3392 Reiter, M., Bilby, R. E., Beech, S., & Heffner, J. (2015). Stream temperature patterns over 35 years in a
3393 managed forest of western Washington. *JAWRA Journal of the American Water Resources*
3394 *Association*, 51(5), 1418-1435.
- 3395 Reiter, M., Heffner, J. T., Beech, S., Turner, T., & Bilby, R. E. (2009). Temporal and Spatial Turbidity Patterns
3396 Over 30 Years in a Managed Forest of Western Washington 1. *JAWRA Journal of the American Water*
3397 *Resources Association*, 45(3), 793-808.
- 3398 Reiter, M., Johnson, S. L., Homyack, J., Jones, J. E., & James, P. L. (2020). Summer stream temperature
3399 changes following forest harvest in the headwaters of the Trask River watershed, Oregon Coast
3400 Range. *Ecohydrology*, 13(3), e2178.
- 3401 Roni, P., Beechie, T., Pess, G., & Hanson, K. (2015). Wood placement in river restoration: fact, fiction, and
3402 future direction. *Canadian Journal of Fisheries and Aquatic Sciences*, 72(3), 466–478.
3403 <https://doi.org/10.1139/cjfas-2014-0344>
- 3404 Roon, D. A., Dunham, J. B., & Groom, J. D. (2021a). Shade, light, and stream temperature responses to
3405 riparian thinning in second-growth redwood forests of northern California. *PloS One*, 16(2), e0246822–
3406 e0246822. <https://doi.org/10.1371/journal.pone.0246822>
- 3407 Roon, D. A., Dunham, J. B., & Torgersen, C. E. (2021b). A riverscape approach reveals downstream
3408 propagation of stream thermal responses to riparian thinning at multiple scales. *Ecosphere*, 12(10),
3409 e03775.
- 3410 Safeeq, M., Grant, G. E., Lewis, S. L., & Hayes, S. K. (2020). Disentangling effects of forest harvest on long-
3411 term hydrologic and sediment dynamics, western Cascades, Oregon. *Journal of Hydrology*, 580, 124259.
- 3412 Salo, E. O., & Cundy, T. W. (1986, February). Streamside management: forestry and fishery interactions.
3413 In *Proceedings of a conference sponsored by the College of Forest Resources, University of Washington*
3414 *and others, and held at the University of Washington*.
- 3415 Schuett-Hames, D., & Stewart, G. (2019a). Post-Harvest Change in Stand Structure, Tree Mortality and
3416 Tree Fall in Eastern Washington Riparian Buffers. Cooperative Monitoring Evaluation and Research
3417 Report. Washington State Forest Practices Adaptive Management Program. Washington Department of
3418 Natural Resources, Olympia, WA.
- 3419 Schuett-Hames, D. & Stewart, G. (2019b). Changes in stand structure, buffer tree mortality and riparian-
3420 associated functions 10 years after timber harvest adjacent to non-fish-bearing perennial streams in

3421 western Washington. Cooperative Monitoring Evaluation and Research Report. Washington State Forest
3422 Practices Adaptive Management Program. Washington Department of Natural Resources, Olympia, WA.

3423 Schuett-Hames, D., Martin, D., Mendoza, C., Flitcroft, R., & Haemmerle, H., (2015). Westside Type F
3424 Riparian Prescription Monitoring Project Technical Writing and Implementation Group (TWIG).
3425 Cooperative Monitoring Evaluation and Research Report. Washington State Forest Practices Adaptive
3426 Management Program. Washington Department of Natural Resources, Olympia, WA.

3427 Schuett-Hames, D., Roorbach, A., & Conrad, R. (2011). Results of the Westside Type N Buffer
3428 Characteristics, Integrity and Function Study Final Report. Cooperative Monitoring Evaluation and
3429 Research Report, CMER 12-1201, Washington Department of Natural Resources, Olympia, WA.

3430 Shah, N. W., & Nisbet, T. R. (2019a). The effects of forest clearance for peatland restoration on water
3431 quality. *Science of the Total Environment*, 693, 133617.

3432 Sievers, M., Hale, R., & Morrongiello, J. R. (2017). Do trout respond to riparian change? A meta-analysis
3433 with implications for restoration and management. *Freshwater Biology*, 62(3), 445–457.
3434 <https://doi.org/10.1111/fwb.12888>

3435 Six, L. J., Bilby, R. E., Reiter, M., James, P., & Villarin, L. (2022). Effects of current forest practices on
3436 organic matter dynamics in headwater streams at the Trask river watershed, Oregon. *Trees, Forests and
3437 People*, 8, 100233.

3438 Sobota, D. J., Gregory, S. V., & Sickle, J. V. (2006). Riparian tree fall directionality and modeling large
3439 wood recruitment to streams. *Canadian Journal of Forest Research*, 36(5), 1243–1254.
3440 <https://doi.org/10.1139/x06-022>

3441 Sugden, B. D., Steiner, R., & Jones, J. E. (2019). Streamside management zone effectiveness for water
3442 temperature control in Western Montana. *International Journal of Forest Engineering*, 30(2), 87-98.

3443 Sullivan, P. F., Neale, M. C., & Kendler, K. S. (2000). Genetic epidemiology of major depression: review
3444 and meta-analysis. *American journal of psychiatry*, 157(10), 1552-1562.

3445 Swanson, F. J., & Dyrness, C. T. (1975). Impact of clear-cutting and road construction on soil erosion by
3446 landslides in the western Cascade Range, Oregon. *Geology*, 3(7), 393-396.

3447 Swanson, F. J., Gregory, S. V., Sedell, J. R., & Campbell, A. G. (1982). Land-water interactions: the riparian
3448 zone.

3449 Swartz, A., Roon, D., Reiter, M., & Warren, D. (2020). Stream temperature responses to experimental
3450 riparian canopy gaps along forested headwaters in western Oregon. *Forest Ecology and
3451 Management*, 474, 118354.

3452 Sweeney, B. W., & Newbold, J. D. (2014). Streamside forest buffer width needed to protect stream water
3453 quality, habitat, and organisms: a literature review. *JAWRA Journal of the American Water Resources
3454 Association*, 50(3), 560-584.

3455 Teply, M., McGreer, D., & Ceder, K. (2014). Using simulation models to develop riparian buffer strip
3456 prescriptions. *Journal of Forestry*, 112(3), 302-311.

3457 Teply, M., McGreer, D., Schult, D., & Seymour, P. (2007). Simulating the effects of forest management on
3458 large woody debris in streams in northern Idaho. *Western Journal of Applied Forestry*, 22(2), 81–87.
3459 <https://doi.org/10.1093/wjaf/22.2.81>

3460 Vanderbilt, K. L., Lajtha, K., & Swanson, F. J. (2003). Biogeochemistry of unpolluted forested watersheds
3461 in the Oregon Cascades: temporal patterns of precipitation and stream nitrogen
3462 fluxes. *Biogeochemistry*, 62(1), 87-117.

3463 Warren, D. R., Keeton, W. S., Bechtold, H. A., & Rosi-Marshall, E. J. (2013). Comparing streambed light
3464 availability and canopy cover in streams with old-growth versus early-mature riparian forests in western
3465 Oregon. *Aquatic sciences*, 75(4), 547-558.

3466 Wing, M. G., & Skaugset, A. (2002). Relationships of channel characteristics, land ownership, and land
3467 use patterns to large woody debris in western Oregon streams. *Canadian Journal of Fisheries and*
3468 *Aquatic Sciences*, 59(5), 796-807.

3469 Wise, E. K. (2010). Tree ring record of streamflow and drought in the upper Snake River. *Water Resources*
3470 *Research*, 46(11).

3471 Wissmar, R.C., Beer, W.N. & Timm, R.K. Spatially explicit estimates of erosion-risk indices and variable
3472 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)
3473 [004-0714-9](https://doi.org/10.1007/s00027-004-0714-9)

3474 Yang, Y., Hart, S. C., McCorkle, E. P., Stacy, E. M., Barnes, M. E., Hunsaker, C. T., ... & Berhe, A. A. (2021).
3475 Stream water chemistry in mixed-conifer headwater basins: role of water sources, seasonality,
3476 watershed characteristics, and disturbances. *Ecosystems*, 24(8), 1853-1874.

3477 Yeung, A. C., Stenroth, K., & Richardson, J. S. (2019). Modelling biophysical controls on stream organic
3478 matter standing stocks under a range of forest harvesting impacts. *Limnologica*, 78, 125714.

3479 Zhang, X., & Shu, C. W. (2010). On positivity-preserving high order discontinuous Galerkin schemes for
3480 compressible Euler equations on rectangular meshes. *Journal of Computational Physics*, 229(23), 8918-
3481 8934.

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3503 Appendix I

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3505 Table A-1.. List of treatments, variables, metrics, and results from publications reviewed for information on litter, organic matter, and
 3506 nutrient inputs.

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

					during the post-road and PH-I treatment periods. However, a statistically significant increase in TP concentrations ($p < 0.001$) occurred at all sites, including the downstream cumulative sites, during PH-II. Generally, OP concentrations throughout the study remained near the minimum detectable concentrations
Gravelle et al., 2009	clearcut to stream, 50% shade retention, uncut reference	Changes in nitrogen and phosphorus compounds.	monthly grab samples from multiple flume sites pre- and post-harvest, laboratory chemical analysis	Data was compared in three treatment periods: pre-harvest, under road construction, post-harvest.	Results showed significant increases in monthly mean NO 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 (28.6–191.6) 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	<u>1) unharvested reference, 2) 100% treatment, a two-sided 50-ft riparian buffer along the entire RMZ, 3) FP treatment a two-sided 50-ft riparian buffer along at least 50% of the RMZ, (4) 0% treatment, clearcut to stream edge (no-buffer).</u>	stream discharge, nitrogen export		Type N (non-fish-bearing streams). Hard-Rock study.	Discharge increased by 5-7% on average in the 100% treatments while increasing between 26-66% in the FP and 0% treatments Results for harvest effects on total Nitrogen export showed significant (P <0.05) treatment effects were present in the FP treatment and in the 0% treatment in the post-harvest (2-years immediately following harvest) and extended periods (7 and 8 years post-harvest) relative to the reference sites, Analysis showed an increase in total-N export of 5.73 (P = 0.121), 10.85 (P = 0.006), and 15.94 (P = 0.000) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments, respectively, and of 6.20 (P = 0.095), 5.34 (P = 0.147), and 8.49 (P = 0.026) kg/ha/yr in the extended period. The authors conclude that the 100% treatment was generally the most effective in minimizing changes in total-N from pre-harvest conditions, the FP was intermediate, and the 0% treatment was least effective. At the end of the study (8 years), only one site had recovered to pre-harvest nitrate-N levels.

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

Yang et al., 2021	Young stands with high shrub cover (> 50%) masticated to < 10% shrub cover. trees removed to a target basal area range of 27–55 m ² 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.

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3508 Table A-2. List of treatments, variables, metrics, and results from publications reviewed for information on large wood (LW), wood loads, and wood recruitment.

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

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

Chen et al., 2005	All harvested streams were clearcut to stream edge. Wildfire streams had no post-fire harvest	Instream wood load, biomass, carbon pool	LW count, volume, decay class, size		LW volume, biomass, and carbon pools were significantly higher in streams adjacent to areas recently disturbed by timber harvest (~10 years) or wildfire (~40 years) than in streams passing through old-growth forests. There was no significant difference in in-stream LW between old-growth riparian areas and areas harvested > 30 years ago. The wildfire sites had significantly higher LW values than both the harvested sites. The authors conclude: (1) LWD input in old growth forested streams was relatively stable based on statistical significance. They also speculate: (1) timber harvesting activities would cause a short-term increase of LWD stocks and might greatly reduce LWD loadings over a long-term, and (2) wildfire disturbance would delay LWD recruitment because not all burnt trees would fall in the stream immediately after the wildfire, based on trends in, and extrapolation of the data.
Chen et al., 2006	A total of 35 sites with stream orders ranging from 1-5 (grouped into 4 stream size categories (I = first order; II = second to third order; III = third to fourth order; IV = fourth to fifth order) were selected to measure spatial distribution and variability of LW characteristics	LW, defined as having a diameter of > 0.1 m and a length > 1.0 m.	LW size, volume, density, and biomass. Multiple stream channel features obtained from readily available physiographic and forest cover data.	Study sites were selected based on the following criteria. (1) the streams were in areas of intact mature riparian forests (>80 years); (2) the stream side forests were not disturbed by human activities, such as harvesting, road building; (3) the streams were not salvaged.	Results from this study show that LW size, volume, and biomass generally increased with increasing stream size. For example, the mean LWD diameter in stream size I (16.4 cm) was lower than that in stream size III (20.6 cm) and IV (20.5 cm), respectively. Mean LW length also increases with stream size from 2.3 m in size I, 2.9 m in size II, 3.1 m in size III, and 3.9 m in size IV. Stream IV had the highest mean volume (0.18 m 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 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 9.7 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., 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.

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

			disturbance history.	23 younger forests.	1.71 + 2.81 m ³). 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.
Jackson et al., 2001	3 unthinned riparian buffers; 1 with a partial buffer; 1 with a buffer of non-merchantable trees; and 6 were clearcut to the stream edge. Buffers ranged from 15 to 21 m wide, partial buffers were as thin as 2.3 m.	Instream LW, particle size, surface roughness	LW as functional and nonfunctional (not altering flow hydraulics). Particle size distributions.	Data collected for only 1-year pre- and 1-month post-harvest. These results only describe immediate effects of harvest on stream conditions.	Increased slash debris (LW) provided shade for the harvested streams but trapped sediments and prevented fluvial transport. The percentage of fine particles increased from 12 to 44% because of bank failure and increased surface roughness. This was a short-term study on small headwater streams. Sediment and LW conditions in the unharvested and buffered streams remained relatively unchanged during the study.
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 3 times greater than competition induced mortality for 3 years following treatment Tree fall direction was heavily biased towards the channel regardless of channel or buffer orientation and tree fall probability was highest in the outer areas of the buffers (adjacent to the harvest area). Tree fall rates and direction were also heavily biased by species with western hemlock and Pacific silver fir having the highest fall rates compared to Douglas-fir, western red cedar, and red alder.

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

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

					(1.1 m 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, 2019b	Clearcut to stream with 30-foot equipment exclusion zone, and 50-foot no-cut buffers	LW, mortality, stand structure, canopy cover	QMD, basal area, tree fall rates, instream LW counts and volume, canopy percentage from densiometer.	1) Substantial variability among sites. 2) Due to scale of study, results only applicable to immediate vicinity of buffer treatment.	10 years post treatment, 50-foot buffer mortality stabilized, cumulative 14.1% reduction in basal area; Reference stands increased in basal area by 2.7% over the 10 years. 10-year cumulative LW recruitment into channels were double that of the reference stands 10-year canopy cover of the 50-foot buffer recovered to similar percentages as the reference stands 10-year cumulative canopy cover of CC was 71.5% due to ingrowth of dense shrubs, saplings and herbaceous plants.

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

					gradient of 4.9%. The streams with a gradient < 4.9% averaged 0.5 key LW pieces per reach while streams with higher gradients averaged 0.9 key LW pieces per reach; in this model stream gradient explained 8% of the variation in key LW pieces found in streams. Lithology caused second, third or fourth level splits after stream gradient or BFW.
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3510 Table A-3 List of treatments, variables, metrics, and results from publications reviewed for information on sediment inputs and source.

Reference	Treatment	Variables	Metrics	Notes	Results
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 Northeastern Oregon and included data from harvested and unharvested sub-catchments underlain by heterogeneous lithologies.	Results from this study indicate that site lithology was a first order control over suspended sediment yield (SSY) with SSY varying by an order of magnitude across lithologies observed. Specifically, SSY was greater in catchments underlain by Siletz Volcanics ($r = 0.6$), the Trask River Formation ($r = 0.4$), and landslide deposits. In contrast, the site effect had a strong negative correlation with percent area underlain by diabase ($r = 0.7$), with the lowest SSY associated with 100% diabase independent of whether earthflow terrain was present. Sites with low SSY and underlain by more resistant lithologies were also resistant to harvest-related increases in SSY. The authors conclude that sites underlain with a friable lithology (e.g., sedimentary formations) had SSYs an order of magnitude higher, on average, following harvest than those on more resistant lithologies (intrusive rocks).
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 ($\sim 0.2 - \sim 953$ t/km ²) indicates that the factors tested in this study should be tested more broadly to investigate their utility to forest managers.	The results of this study show that watershed slope variability combined with cumulative annual discharge explained 67% of the variation in annual sediment yield across the approximately 60-year data set. The results, however, show that annual sediment yields also moderately correlated with many other physiographic variables and the authors caution that the strong relationship with watershed slope variability is likely a proxy for many processes, encompassing multiple catchment For the relationships between disturbance and sediment yield the authors conclude that the few anomalous years of high sediment yield occurred in watersheds with high slope variability and within a decade of forest management and a large flood event.

<p>Hatten et al., 2018</p>	<p>Data from pre restriction and post Oregon BMPs prescriptions for non-fish bearing streams. BMPs: no buffer in non-fish-bearing streams with equipment exclusion zones, and a 15 m no-cut-buffer in fish-bearing streams</p>	<p>suspended sediment concentrations (SSC)</p>	<p>suspended sediment, stream discharge, and daily precipitation</p>	<p>Phase I harvest: 2009 harvest of upper half of watershed. Phase II harvest: 2015 harvest of lower half of watershed.</p>	<p>Methods used in 1966 to harvest the same watershed (no buffer, road construction, broadcast burning) resulted in an approximate 2.8-fold increase in SSC from pre- to post-Harvest. In the contemporary study both the mean and maximum SSC were greater in the reference catchments (FCG and DCG) compared to the harvested catchment (NBLG) across all water years. In NBLG the mean SSC was 32 mg L⁻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.</p>
<p>Karwan et al., 2007</p>	<p>clearcut of the watershed area of by 50%, partial cut of 50% canopy removal, timber road construction Riparian zone harvest followed Idaho FPA rules.</p>	<p>Total suspended solid (TSS) yields</p>	<p>Monthly total suspended solid readings from multiple flume locations for pre-, and post-harvest, and pre- and post-road construction.</p>		<p>A significant and immediate impact of harvest on monthly sediment loads in the clear-cut watershed ($p = 0.00011$), 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 impacts on monthly sediment loads in either treated watershed during the immediate or recovery time intervals.</p>

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

	stream edge (no-buffer).				
Mueller & Pitlick, 2013	The study used sediment concentration data from 83 drainage basins in Idaho and Wyoming.	Sediment concentration, basin lithology, geomorphology	Sediment concentration distribution, geomorphology, and weather data from multiple sources.		The strongest correlation of in stream sediment supply was with lithology relative softness. Bankfull sediment concentrations increased by as much as 100-fold as basin lithology became dominated by softer sedimentary and volcanic rock. Relief (elevation), basin sideslope, and drainage density showed little correlation strength with bankfull sediment supply.
Puntenney-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 + 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 + 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).

Safaeq 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, 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.

3512 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 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 ± 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 catchments with a high percentage of harvest that were underlain by permeable geology

<p>Cole & Newton, 2013</p>	<p>clearcut to stream, partial buffer (12 m width on predominant sun-side), Oregon state BMP (15-30 m no-cut buffer both sides)</p>	<p>Stream temperature</p>	<p>Controlled for yearly fluctuations in temperatures by analyzing the difference in stream temperature entering and exiting the reach with digital temperature data loggers</p>	<p>Stream temperature data collected for 2 – years prior and 4 to 5 years following harvest. Unharvested control sites were located downstream of treatment sites. Treatment applied to four small fish-bearing streams.</p>	<p>Results showed the most significant increases in daily maximum, and mean, and diel fluctuations in temperatures post-harvest for all no tree buffers. Changes to daily maxima ranged from -0.11 to 3.84 °C, and changes to daily minimum ranged from -1.12 to 0.49 °C. The no tree buffers also showed small but significant changes below predicted summer minima between -1.12 and -0.49 °C. The partial buffer units varied in their response to treatment exhibiting increases, decreases, and no change from preharvest trends.</p>
<p>Cupp & Lofgren, 2014</p>	<p>the “all available shade” rule (ASR), and the standard rule (SR) in eastern WA. ASR: requires retention of all available shade within 75 feet of the stream. SR: some harvest is allowed within the 75-foot buffer depending on elevation and pre-harvest canopy cover.</p>	<p>Canopy closure, shade measurements, stream temperature</p>	<p>Hand-held densiometer (canopy closure), self-leveling fisheye lens digital camera (shade), temperature data loggers</p>	<p>Sites were between 65-100 years old and were situated along second to fourth order streams with harvest-regenerated or fire-regenerated forests. Reference reaches were located upstream from treatment reaches where harvest was applied.</p>	<p>Results showed post-harvest shade values decreased in SR sites (mean effect of -2.8%, $p = 0.002$), 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.</p>
<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)</p>

					and by 0.1°C at another (at year 5 post-harvest). None of the three REF sites exceeded 16°C during the study.
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.

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

					Differences in treatment streams and reference stream temperatures were less than 1.1°C pre-treatment and 30-years post-treatment.
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 20-30m of stream, and 3) Patch-cut removal of all vegetation in the upper 40% of the watershed.	Stream temperature	Temperature data were recorded with Vemco dataloggers. Canopy cover was estimated with densiometers.		Significant increase in stream temperatures ranging from 4 – 6 °C at five years post-harvest, and increased ranges of diurnal temperature fluctuations for all treatment streams relative to the reference streams. Streams that had summer maximum mean weekly temperatures of 8°C before harvesting had maximum temperatures near 12°C or more following harvesting. Daily ranges of 1.0–1.3°C before harvesting became 2.0–3.0°C following harvesting, high-retention buffer treatment mitigated temperature increases for the first three years. Still, increased mortality (attributed to windthrow) caused a reduction in the canopy that, thus, led to increased stream temperatures equivalent to other treatment streams by year five.

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

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

					shade recovery from initial harvest (indexed by ACD). Significant correlations were found for monthly temperature metrics that were adjusted for climate, for all basins. The authors conclude that the results of this study show evidence that implementation of protection buffers in this area were sufficient in maintaining stream temperatures. Conversely, this study also shows evidence that despite these protections from land management induced stream temperature changes, these protections have been somewhat offset by the warming climate conditions.
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 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 (MWT) 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 T 7DayMax or T 7DayMean significantly across the 6 study sites. However, reaches showed a statistically significant effect of the gap for average daily maximums (p < 0.01) and for average daily means (p = 0.02). The regression comparison reveals there will be on average an additional 0.12 °C/°C increase in daily maximum temperature in the reach with a gap.

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

3514

3515 Appendix II

3516

3517 **Shade and LW**

3518

3519 Anderson & Meleason, 2009

3520

3521 Anderson, P.D., Meleason, M.A., 2009. Discerning responses of down wood and understory
3522 vegetation abundance to riparian buffer width and thinning treatments: an equivalence-
3523 inequivalence approach. *Can. J. For. Res.* 39, 2470–2485 <https://doi.org/10.1139/X09-151>

3524

3525 The purpose of this study was to determine the effect of buffer width on understory vegetation
3526 and down woody responses both within the unthinned buffer and in the adjacent thinned stand. A
3527 secondary objective of this study was to explore the ability of equivalence-nonequivalence
3528 statistical tests at assessing the degree of similarity between stands. The focus of this study was
3529 on second-growth stands dominated by Douglas-fir at multiple sites along the coast and Cascade
3530 Range in western Oregon. Six combinations of buffer width and upslope density management
3531 prescription were evaluated: one site potential tree height buffer averaging 69 m adjacent to
3532 thinning and a 0.4 patch opening; variable width buffer averaging 22 m adjacent to thinning and
3533 a 0.4 patch opening; streamside retention width averaging 9 m adjacent to thinning; and an
3534 unthinned stand serving as a reference. Pearson correlation and multivariate analysis of variation
3535 were used to examine data on percent cover of small and large down wood, and percent cover of
3536 shrubs, herbs, and moss. Inferences on buffer performance were generated using linear mixed
3537 model analysis, equivalence-inequivalence tests, and two post-hoc comparisons. The results from
3538 this study show upland thinning led only to subtle changes in understory vegetation cover and
3539 composition with vegetation responses most prevalent with narrow buffer widths and particularly
3540 when adjacent to patch openings. There was a lack of significant change in down wood response
3541 to treatments.

3542

3543 **Shade**

3544

3545 Anderson et al., 2007

3546

3547 Anderson, P.D., Larson, D.J., Chan, S.S., 2007. Riparian buffer and density management
3548 influences on microclimate of young headwater forests of western Oregon. *For. Sci.* 53, 254–
3549 269. <https://doi.org/10.1093/forestscience/53.2.254>

3550
3551 The purpose of this study was to characterize variation in overstory density, canopy closure, and
3552 microclimate as a function of distance from headwater streams, and (2) determine differences in the
3553 ability of thinned stands and unthinned stands to maintain understory microclimate above the stream
3554 channel and in the riparian zone. The focus of this study was on second-growth (30- to 80-year-old)
3555 Douglas-fir forests characteristic of western Oregon. The study was located at four sites along the
3556 Oregon coast and at one site on the western Oregon Cascade Range. Stands were either thinned to
3557 approximately 198 trees per acre (TPA) or were left unthinned and ranged from 500-865 TPA. Within
3558 thinned stands, 10% of the area was harvested to create patch openings and 10% was left as clusters of
3559 “leave islands”. Streams within treated stands were surrounded by buffers of either (1) one site-potential
3560 tree averaging 69 m (B1), (2) variable width buffer averaging 22 m (VB), or (3) streamside retention
3561 buffer averaging 9 m (SR-T). These six combinations of buffer width and adjacent density management
3562 were evaluated using univariate linear modeling and compared with untreated (UT) stands. Microsite
3563 and microclimate data were obtained through repeated transect measurements extending laterally from
3564 stream center and into the riparian zone and upland treated stand 2-5 years after treatment. The stand
3565 basal area was determined through variable radius plot sampling. Canopy cover was estimated through
3566 photographic techniques during the summer leaf-on period. The results from this study show that the
3567 ability of narrow streamside buffers (SR-T) at moderating stream microclimate in treated stands was
3568 questionable. Visible sky at stream center only differed significantly between SR-T (9.6%) and UT
3569 (4.2%) stands. The SR-T stands showed a +4.5°C difference in daily maximum temperatures just above
3570 stream center when compared to the UT stands. However, this difference was not statistically significant.
3571 The researchers report that SR-T had a weak temperature gradient (tested at 0-10 m and 10-30 m
3572 increments from stream center) indicating the stream center and buffer microclimates were nearly the
3573 same as upslope in the thinned stand. Within the riparian buffer zones daily maximum temperatures
3574 were higher in all treated stands when compared to UT stands. The differences in daily maximum
3575 temperatures ranged from 1.1°C (B1) to 4.0°C (SR-T), but the difference was only significant in one
3576 SR-T stand. The maximum air temperature within buffer zones adjacent to patch openings was 3.5°C
3577 higher than in UT stands. Soil temperature changes were only evident within patch openings ranging
3578 from 3.6 - 8.8°C higher than in UT stands. The researchers of this study conclude by saying that buffers
3579 with widths defined by the transition of riparian to upslope vegetation or significant topographic slope
3580 breaks appear sufficient at mitigating effects from upslope harvests on the above-stream microclimate.
3581 Their suggestions for further study center around cross-disciplinary research into the relationships
3582 between forest structure, microclimate, and habitat suitability on headwater riparian organisms.

3583

3584 **Stream Temperatures**

3585

3586 Cole & Newton, 2013

3587

3588 Cole, E., & Newton, M. (2013). Influence of streamside buffers on stream temperature response
3589 following clear-cut harvesting in western Oregon. *Canadian journal of forest research*, 43(11), 993-1005.
3590 <https://doi.org/10.1139/cjfr-2013-0138>

3591

3592 This study compares the changes in stream temperatures following a clearcut with three different buffer
3593 treatments – no tree buffer, predominantly sun-sided 12 m wide partial buffer, and a two-sided 15-30 m
3594 buffer (BMP for this area). The study was conducted on four small fish bearing streams in the area
3595 surrounding Corvallis, Oregon. Streams were dominated by both hardwood and conifers and were
3596 located at low- and mid-elevations. Each treatment alternated with unharvested reference sections along
3597 study reaches spanning 1800-2600 meters. Stream temperature data adjacent to treatment and
3598 downstream of treatment were collected for 2 –years prior and 4 to 5 years following harvest. Time-
3599 series regression analysis was used to evaluate the change in temperatures between pre- and post-
3600 harvest. The researchers controlled for yearly fluctuations in temperatures by analyzing the difference in
3601 stream temperature entering and exiting the experimental reaches. Results showed significant increases
3602 in daily maximum, mean, and diel fluctuations in temperatures post-harvest for all no tree buffers (up to
3603 3.8 °C). The no tree buffers also showed small but significant changes below predicted summer minima
3604 by as much as 1.2°C. The partial buffer units varied in their response to treatment exhibiting increases,
3605 decreases, and no change from preharvest trends. For example, at one site, there were no detectable
3606 changes in means, minima, or diel fluctuations but significantly lower maximum temperatures post-
3607 harvest ($p = 0.0021$; actual temperatures not reported). Partial buffers at another site reported lower
3608 trends in mean, maxima, and diel fluctuations in temperature post-harvest, and no difference in minima.
3609 Only one partial buffer site showed increases in all recorded trends (mean, minima, maxima, diel
3610 fluctuations). The BMP buffered treatment sites also showed variation in results. One site showed no
3611 detectable changes, one site showed small but significant ($p < 0.0350$; actual temperatures not reported)
3612 decreases in downstream temperatures. Only two BMP buffered sites showed significant ($p < 0.0499$)
3613 increases in mean, maxima, and diel fluctuations in temperatures. The highest increase in maxima for
3614 any BMP buffered site was 5.3°C. Changes in temperature trends in uncut reference post-treatment were
3615 minimal and attributed to downstream effects from the treatment reaches. However, when post-harvest
3616 trends in upstream treated sites were higher than pre-harvest temperatures tended to fall below pre-
3617 harvest values when passing through the unharvested downstream units. For within-unit trends,
3618 unharvested units downstream from no tree and partial buffers showed trends of significantly decreasing
3619 daily maximum temperatures. When the data was analyzed by 7-day moving mean maximum
3620 temperatures, the no tree buffers showed significant increases after harvest. The authors report that most
3621 partial and BMP buffers resulted in minimal increases or negligible changes to the 7-day moving mean
3622 maximum temperatures (actual values not reported). Significant changes in one or more temperature
3623 trends (mean, minima, maxima, diel fluctuations) were detected in all treatment stream post-harvest with
3624 only one exception at a BMP buffered site This was a well planned and executed experimental design
3625 that shows how changes in stream temperatures post-harvest are directly related to residual buffer
3626 treatment while also showing evidence that many other factors such as stream features (orientation,

3627 topography, ground water source) can compound or ameliorate these effects (I.e., changes in temperature
3628 were highly affected by site factors).

3629

3630 **Stream Temperature**

3631

3632 Johnson & Jones, 2000

3633

3634 Johnson, S. L., & Jones, J. A. (2000). Stream temperature responses to forest harvest and debris flows in
3635 western Cascades, Oregon. *Canadian Journal of Fisheries and Aquatic Sciences*, 57(S2), 30-39.
3636 <https://doi.org/10.1139/f00-109>

3637

3638 This paper is a study of the changes in mean stream temperature minimum, maximum, diurnal
3639 fluctuation, and interannual and seasonal variability following harvest in three small basins of the
3640 H.J. Andrews experimental watershed between 1962 and 1966. The experimental design used
3641 historic stream temperature data to examine changes in stream temperature following clear-cut
3642 (no buffer) and burning in one watershed; patch cutting and debris flows (resulted in the removal
3643 of all streamside vegetation 3 years after cut) treatments in another watershed; and one old-
3644 growth uncut reference watershed. All watersheds were dominated by 450-year-old Doug-fir
3645 forests prior to harvest. Data was analyzed for the period 1959-1997. Mean weekly temperature
3646 maximum, minimum, and annual fluctuations were compared between all three watersheds using
3647 a complete factor analysis of variance (ANOVA). The experiment also involved long-term
3648 monitoring to evaluate time until recovery of pre-treatment temperature fluctuations. Results
3649 showed a significant increase in stream temperatures in both treatment watersheds after treatment
3650 compared to the unharvested site. The unharvested watershed showed higher interannual
3651 variability in maximum stream temperatures ranging from 15 to 19°C. The two treatment
3652 watersheds, despite differences in disturbances, (clear-cut and burn vs. Patch cut and debris-
3653 flow) followed similar trajectories from 1966-1982. Stream temperature summer maximums
3654 reached 23.9°C and 21.7°C 1-2 years post-harvest (clear-cut/burn and patch-cut/debris flow
3655 respectively) and returned to pre-harvest summer temperatures by 1980 (~15 years post-harvest).
3656 Both treatment watersheds exhibited significant increases in mean weekly minimum and
3657 maximum stream temperatures in the summer months immediately following harvest and for at
3658 least 3 years compared to the unharvested reference. The clear-cut and burn watershed's
3659 weekly maximum summer temperatures ranged between 5.4 and 6.4°C higher, and mean weekly
3660 minimum ranged 1.6-2.0°C higher than the reference streams for 4 years post-harvest. The patch-
3661 cut and debris-flow watershed exhibited mean weekly maximum stream temperatures 3.5-5.2°C
3662 higher than in the reference stream for 3 years following harvest/disturbance. Prior to harvest and
3663 30 years post-harvest the mean weekly maximum and minimum stream temperatures for both
3664 treatment streams differed less than 1.1°C from the reference stream. These differences in stream
3665 temperatures from treated and untreated sites were amplified during periods of high solar inputs

3666 and reduced during periods of cloud cover. Differences in stream temperatures were greatest
3667 during the end of July and beginning of June. Diurnal fluctuations in stream temperatures were
3668 also significantly higher in both treatment watersheds (6-8 °C in the clearcut, and 5-6 °C in the
3669 patch-cut) relative to the reference stream (1-2 °C). Stream temperatures returned to pre-harvest
3670 levels after 15 years of growth.

3671

3672 **Large Wood (LW)**

3673

3674 Bahuguna et al., 2010

3675

3676 Bahuguna, D., Mitchell, S.J., Miquelajauregui, Y., 2010. Windthrow and recruitment of large woody
3677 debris in riparian stands. *Forest Ecology and Management* 259, 2048–2055.
3678 <https://doi.org/10.1016/j.foreco.2010.02.015>

3679

3680 The purpose of this paper was to evaluate the effect of riparian buffer width on windthrow and LW
3681 recruitment and to contrast data with unharvested controls. This paper also seeks to document the
3682 geometry of post-harvest windthrow from buffers of varying widths and to develop a model framework
3683 for incorporating supply of LW originating from windthrow to streams from riparian buffers. The focus
3684 of this paper is on dense young conifer-dominated forests originating from harvest followed by wildfire.
3685 This study is located in the Coast Mountains, approximately 60 km east of Vancouver, BC. Two buffer
3686 widths on each side of the stream (10 m and 30 m) along with an unharvested control were each
3687 replicated three times in the experiment. The researchers used a strip plot sampling method running
3688 parallel to the stream to collect data on species, diameter, height, and status (standing live/dead)
3689 beginning in the year prior to harvest and annually thereafter for seven years. A General Linear Model
3690 Procedure was used to determine the significance of variables. The Pearson correlation coefficient was
3691 used to assess correlations and potential predictor variables. Multiple linear regression was then used to
3692 determine the utility of the variables at determining LW height above the stream. Following harvest,
3693 11% of initially standing timber was blown down in the first and second years in the 10 m buffer,
3694 compared to 4% in the 30 m buffer, and 1% in the unharvested controls. Following 8 years post-harvest,
3695 a significant amount of annual mortality occurred in the unharvested control at 30%, compared to 15%
3696 in both 30 m and 10 m buffers. 77% of LW was in the 10 cm - 20 cm diameter class while the mean
3697 diameter of standing trees in riparian buffers was 30 cm indicating small diameter trees were
3698 significantly more represented in streams. Only 3% of windthrown logs fell perpendicular to the stream
3699 with the majority falling diagonal-perpendicular relative to the stream. The researchers of this study
3700 conclude that recruitment of logs into streams lags behind the post-harvest pulse of windthrow by
3701 several years. The lag depends on the size, species, and condition of logs, and their direction of fall
3702 relative to stream valley geometry.

3703

3704 **Species Richness**

3705

3706 Baldwin et al., 2012 (Removed from focal list)

3707

3708 Baldwin, L.K., Petersen, C.L., Bradfield, G.E., Jones, W.M., Black, S.T., Karakatsoulis, J., 2012.
3709 Bryophyte response to forest canopy treatments within the riparian zone of high-elevation small streams.
3710 *Can. J. For. Res.* 42, 141–156. <https://doi.org/10.1139/x11-165>

3711

3712 The purpose of this study was to examine the influence of forest harvesting practices and distance from
3713 the stream on riparian-bryophyte communities. The experiment was limited to the montane spruce forest
3714 type which is considered moderately open and dominated by lodgepole pine in the uplands and by
3715 hybrid spruce in well-developed riparian areas. The study took place at five different watersheds located
3716 approximately 70 km from Kamloops, BC. Three primary treatments: clear-cut (n=7), two-sided buffer
3717 averaging approximately 15 m on both sides (n=10), and a continuous forest (n=6) were used to sample
3718 numerous environmental variables including elevation, aspect, slope, buffer width, and CWD decay
3719 class. Bryophytes (classified into life history strategies), stand structure, and microhabitat were also
3720 measured 1, 5, and 10 m from the streams edge. Additionally, the DBH of all conifer stems as well as
3721 percent vegetation cover were measured along transects. All data were collected in July-August of 2007
3722 and 2008. Minimum time since disturbance for clearcut sites was 13 years versus a minimum of 5 years
3723 in buffered sites. An analysis of variance was used to compare environmental, stream, and stand
3724 structure characteristics among canopy treatments. Mean values were calculated for stand structure and
3725 substrate variables recording in transects. Bryophytes were analyzed within functional groups based on
3726 growth form, substrate affiliations, and life history. Linear models were used to evaluate the effects of
3727 distance to stream, forest canopy treatment, and their interaction on response variables. Overall CWD
3728 did not differ significantly among treatments, although buffer treatment sites had significantly higher
3729 volume of CWD in early decay classes compared to clearcut and continuous forests. The researchers
3730 suggest the early decay class CWD in buffer treated sites was likely the result of increased stem
3731 breakage. After accounting for distance from the stream, the richness and frequency of bryophyte
3732 functional communities was intermediate to continuous and clearcut sites. Compared to continuous sites,
3733 buffered sites featured significantly lower richness and frequency of many forest-associated groups.
3734 Furthermore, buffered sites also did not support increased richness or frequency of disturbance-
3735 associated species. Clearcut treatments featured higher levels of disturbance associated species including
3736 colonists, canopy species, and species typically found on mineral soil. Data from this study also showed
3737 bryophyte species richness and frequency decline with increasing distance from the stream. The authors
3738 conclude by noting that while bryophyte communities in buffered sites are significantly more diverse
3739 than communities in clearcut sites, reductions in forest-associated species as well as in the bryophyte
3740 mat as a result of large-scale forestry indicate that the ecological function of buffer-dwelling bryophyte
3741 communities may be hindered and could benefit alongside large uncut forest reserves.

3742

3743 **Sediment**

3744

3745 Mueller & Pitlick, 2013

3746

3747 Mueller, E. R., & Pitlick, J. (2013). Sediment supply and channel morphology in mountain river
3748 systems: 1. Relative importance of lithology, topography, and climate. *Journal of Geophysical*
3749 *Research: Earth Surface*, 118(4), 2325-2342. <https://doi.org/10.1002/2013JF002843>

3750

3751 This study used correlation analysis to assess the relative impact of lithology, basin relief, mean basin
3752 slope, and drainage density on in stream sediment supply defined by the bankfull sediment concentration
3753 (bedload and suspended load). The study used sediment concentration data from 83 drainage basins in
3754 Idaho and Wyoming. Lithologies of the study area were divided into four categories ranging from
3755 hardest to softest- granitic, metasedimentary, volcanic, and sedimentary. The results showed the
3756 strongest correlation of bankfull sediment concentration was with basin lithology, and showed little
3757 correlation strength with slope, relief and drainage density. As lithologies become dominated by softer
3758 parent materials (volcanic and sedimentary rocks), bankfull sediment concentrations increased by as
3759 much as 100-fold. These results suggest that lithology can be more important in estimating sediment
3760 supply than topography. The authors discuss using a correlative analysis but give little description of
3761 what that analysis was or how they compare the values of each correlation strength to see if the
3762 differences were significant.

3763

3764 **CWD Modeling**

3765

3766 Benda et al., 2016

3767

3768 Benda, L.E., Litschert, S.E., Reeves, G., Pabst, R., 2016. Thinning and in-stream wood
3769 recruitment in riparian second growth forests in coastal Oregon and the use of buffers and tree
3770 tipping as mitigation. *J. For. Res.* 27, 821–836. <https://doi.org/10.1007/s11676-015-0173-2>

3771

3772 The purpose of this study was to develop a model which examines the effects of riparian thinning
3773 on in-stream wood recruitment in second growth stands. A secondary objective of this study was
3774 to model how manual felling of trees in no-harvest buffer zones impacts the effects of thinning.
3775 The study site was located within the Alcea watershed in central coastal Oregon. Silvicultural
3776 simulation treatments used the reach scale wood model (RSWM) and included: (1) no harvest
3777 control; (2) single entry thinning from below (thinning from below removes the smallest trees to
3778 simulate suppression mortality) with and without a 10 m width no-cut buffers; (3) double entry

3779 thinning from below with the second thinning occurring 25 years after the first with and without
3780 10 m no-cut buffers (4) Each thinning treatment was also combined with some mechanical
3781 introduction of thinned trees into the stream encompassing a range between 5 and 20 % of the
3782 thinned trees. . The simulation model RSWM was run for 100 years in 5-year time steps. In the
3783 no-harvest control, the model output shows the density of live trees declines from 687 trees-per-
3784 hectare (tph) in 2015 to 266 tph in 2110 due to natural suppression mortality (-61 % from initial
3785 conditions). The single-entry thin reduces stand density to 225 tph in 2015 (-67 %) and declines
3786 further to 160 tph by 2110 (-77 %). The double entry thinning resulted in 123 tph after the
3787 second thinning in 2040 (-82%) and maintained that density until 2110. Both thinning treatments
3788 resulted in a substantial reduction of dead trees that could contribute to in-stream wood over
3789 time. The model output for single entry thinning treatments predicts a 33% or 66% reduction of
3790 in-stream wood over a century relative to the unharvested reference for harvest on one side or
3791 both sides of the stream, respectively. Adding the 10-m no cut buffer reduced total loss to 7 and
3792 14%. Including mechanical tipping of 5,10,15, and 20% of cut stems without a buffer in the
3793 single entry thinning treatment changes the relative in-stream percentages of wood relative to the
3794 reference stream to -15, -6, +1, and +6%, respectively. To completely offset the loss of in stream
3795 wood due to single entry thinning mechanical tipping of 14 and 12% were required without and
3796 with buffers. Double entry thinning treatments without a buffer predicted further reduction in
3797 wood recruitment over a century of simulation with 42 and 84% reduction of in stream wood
3798 relative to the reference stream when one side and both sides of the channel were harvested.
3799 Adding a 10 m buffer reduced total reduction of in stream wood to 11 and 22% for thinning on
3800 one and both sides of the channel. To offset the predicted changes of in stream wood volume
3801 following double entry harvest would require tipping of 10 and 7% of cut stems without and with
3802 the 10-m buffer. The authors conclude that thinning without some mitigation efforts resulted in
3803 large losses of in stream wood over a century. However, by including a 10-m no cut buffer or a
3804 practice of mechanical tipping can offset these losses Although predictions from this study
3805 contribute to the in-stream wood recruitment conversation moving forward, the model contained
3806 limitations such as utilizing data from FIA plots which only approximate riparian forest
3807 conditions.

3808

3809 **Modeling Stream Litter Delivery**

3810

3811 Bilby & Heffner, 2016

3812

3813 Bilby, R.E., Heffner, J.T., 2016. Factors influencing litter delivery to streams. *Forest Ecology and*
3814 *Management* 369, 29–37. <https://doi.org/10.1016/j.foreco.2016.03.031>

3815 The purpose of this study was to understand the relative influence of wind speed and direction,
3816 topography, litter type, species, and stand conditions on the distance from which litter is
3817 delivered to streams. This study utilized a combination of field experiments, literature, and

3818 simple models to estimate the width of a delivery areas. The effects of wind speed on litter
3819 delivery distance were measured on litter samples from two common species of the Pacific
3820 Northwest, Douglas-fir and red alder by releasing litter from a riparian tree canopy at various
3821 wind speeds and recording the distances traveled for each litter type at each wind speed. The
3822 relationship between distance of litter recruitment area and variables of interest (e.g., wind speed,
3823 topography, litter type...) were determined with a linear mixed effects model Data for wind speed
3824 and direction was recorded for one year in 30 min intervals along Humphrey Creek in the
3825 Cascade Mountains of western Washington. Results showed that under the wind conditions
3826 recorded at Humphrey Creek the majority of the litter recruited into the stream originated from
3827 within 10 m of the stream regardless of litter or stand type. No difference was found in delivery
3828 distance and litter type (needles or broadleaf) at young sites. However, needles released at mature
3829 sites had a higher proportion of cumulative input from greater distances than needles or alder
3830 leaves released at younger sites. This is likely due to the higher canopy and thus higher release
3831 position. Litter travel distance was linearly related to wind speed ($p < 0.0001$) Doubling wind
3832 speed at one site led to a 67-87% expansion of the riparian contribution zone in the study area.
3833 The results reveal a trend that suggests slope also contributes to the width of the litter
3834 contributing area. However, the authors did not apply statistical analysis to these values and only
3835 speculate that increasing the slope from 0-45% would increase the width of the litter contributing
3836 area by 70%. Overall, the results of this study show evidence that wind speed has a strong effect
3837 on the width of litter delivery areas within riparian areas, but that relationship is also affected
3838 stand age and litter type. Trends in the data also suggest that topography is an important factor,
3839 but it was not quantified.

3840

3841 **Stream Temperature**

3842

3843 Bladon et al., 2016

3844

3845 Bladon, K.D., Cook, N.A., Light, J.T., Segura, C., 2016. A catchment-scale assessment of stream
3846 temperature response to contemporary forest harvesting in the Oregon Coast Range. *Forest
3847 Ecology and Management* 379, 153–164. <http://dx.doi.org/10.1016/j.foreco.2016.08.021>

3848

3849 The purpose of this study was to compare the effects of contemporary riparian forest harvest
3850 treatments under the Oregon Forest Practices Act (15 m riparian management area with a
3851 minimum of ~ 3.7 m² conifer basal area retained for every 300 m length of stream) with historical
3852 riparian forest harvest practices (no maintenance of streamside vegetation) on stream
3853 temperatures. This study took place in the Siuslaw National Forest in the Oregon Coast Range
3854 as part of the Alsea Watershed Study Revisited. Historical records of stream temperatures were
3855 sourced from the original Alsea Watershed Study that monitored stream temperature changes
3856 from 1958-1973, before and after streamside timber harvesting in 1966. Stream temperature data

3857 was collected for contemporary forest practices over a 6-year period (3 years pre- and 3 years
3858 post-harvest; 2006-2012). Data for the contemporary harvest was also compared with stream
3859 temperature changes in unharvested reference streams to support a Before-After-Control Impact
3860 (BACI) design. Stream temperature thermistors were installed, and data was taken at 30-minute
3861 intervals at three sections of both the harvested (2 within harvest boundary and 1 downstream)
3862 and reference sites. Mean canopy closure, as measured with a densiometer, along the stream
3863 channel in the harvested portion of Needle Branch was reduced from ~96% in the pre-harvest
3864 period to ~89% in the post-harvest period. Comparatively, mean canopy closure along the stream
3865 channel in the reference sites were ~92% in the pre-harvest period and 91% in the post-harvest
3866 period. Data was analyzed to assess whether there were changes in the 7-day moving mean of
3867 daily maximum stream temperature, mean daily stream temperature, and diel stream temperature
3868 following harvest. The results showed no significant changes in any of the three parameters
3869 measured following contemporary forest harvesting practices when analyzed across all
3870 catchments for all summer months (July to September). When the mean 7-day moving maximum
3871 temperature was constrained to the summer period between July 15 – August 15 across all sites
3872 there was a significant increase in stream temperatures in the harvested sites by $0.6 \pm 0.2^{\circ}\text{C}$
3873 following harvest. However, when the data was arranged for individual pair-wise comparisons
3874 with the unharvested sites, and intrinsic annual and site variability was accounted for, the
3875 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
3876 any site. The only comparison made in the study to the original Alsea Watershed study was with
3877 the single day maximum stream temperatures for pre- and post-harvest. The contemporary
3878 practices showed a change of single day maximum stream temperatures from 15.7°C to 14.7°C
3879 (a reduction) from pre- to post-harvest. In contrast, the historical stream temperature data showed
3880 an increase in single day maximum stream temperatures from 13.9°C (pre-harvest) to as much
3881 as 29.4°C (2-years post-harvest). The authors caution that while these results support the
3882 conclusion that contemporary forest practices in Oregon are sufficient in maintaining stream
3883 temperatures after riparian forest harvest, and much more efficient than historical practices; these
3884 results should not be generalized to areas outside of coastal Oregon. The authors caution that the
3885 streams in this study have potential for a muted stream temperature response following harvest
3886 relative to other regions because of the (1) north-south stream orientation, which would
3887 maximize RMA effectiveness (2) steep catchment and channel slopes that can increase stream
3888 velocity and hyporheic exchange, (3) potential increases in groundwater contributions after
3889 harvest.

3890

3891 **Stream temperature**

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3893 Bladon et al., 2018

3894

3895 Bladon, K.D., Segura, C., Cook, N.A., Bywater-Reyes, S., Reiter, M., 2018. A multicatchment
3896 analysis of headwater and downstream temperature effects from contemporary forest harvesting.
3897 *Hydrological Processes* 32, 293–304. <https://doi.org/10.1002/hyp.11415>

3898

3899 The purpose of this study was to (1) examine the effects of contemporary forest harvesting
3900 practices on headwater stream temperature, (2) determine if increased temperatures from
3901 harvesting was detectable in downstream fish-bearing streams, and (3) examine the relative role
3902 of geology and forest management on influencing the differential stream temperature responses
3903 in both headwater and downstream reaches. This study took place at three paired watershed
3904 studies, of which two (Alesa, Trask) were located in the Oregon coast range, and one (Hinkle)
3905 was located in the western Cascades of Oregon. This study featured pre- and post-harvest
3906 measurements, as well as measurements within and downstream from harvested and reference
3907 sites. Buffer widths at harvested sites varied but averaged 20 m on either side of streams.
3908 Statistical models were generated which analyzed whether (a) the 7-day moving average of daily
3909 maximum stream temperature (7daymax) changed between pre- and post-harvest sites, and (b)
3910 whether post-harvest changes in 7daymax were detectable downstream. A regression analysis
3911 was also performed to assess the relative relationship between catchment lithology and percent
3912 catchment harvested on temperature at all sites. Statistical models were generated for each
3913 harvest site and reference pair. The pre-harvest relationship in stream temperatures for paired
3914 sites were used to create predicted changes in stream temperatures post-harvest. The post-harvest
3915 stream temperatures were then compared to the predicted values and the 95% prediction
3916 intervals. If post-harvest values of the 7daymax were outside the prediction interval the authors
3917 referred to these observations as statistical “exceedances”. Results showed that the 7daymax
3918 exceeded the predictive interval at 7 of the 8 harvested headwater sites (within the harvested
3919 boundary) when analyzed across all harvest years. The exceedances were largest in the first year
3920 after harvest but diminished in the second and third year at two treatment sites. However, at one
3921 site, the elevated 7daymax continued for three years post-harvest. In 4 of the 7 harvested sites
3922 with exceedances, the exceedances were recorded between 22 and 100% of the time. Smaller
3923 increases in stream temperatures were detected in the other 3 streams with exceedances, the
3924 exceedances occurred < 15% of the time. There was no evidence of elevated stream temperatures
3925 beyond the predicted intervals in any of the downstream sites following harvesting. The
3926 magnitude of change in stream temperature and transmission of warmer water downstream were
3927 a function of percentage of catchment harvested and the underlying geology. Although, these
3928 relationships were scale dependent. At the upstream, harvested sites there was a strong
3929 relationship between stream temperature increases and catchment lithologies, but no statistically
3930 significant relationship between stream temperature changes and percent of catchment harvested.
3931 Sites downstream from harvested areas showed a strong relationship with the interaction of
3932 percentage of catchment harvested and the underlying lithologies. The greatest temperature
3933 increases at downstream sites were in areas with a higher percentage of catchment harvested and
3934 were underlain by more resistant lithologies. There was no evidence for increases in stream
3935 temperatures in catchments with a high percentage of harvest that were underlain by permeable
3936 geology. The authors suggest that this relationship may be due to the buffering effect of increases

3937 in summer low flows and greater groundwater or hyporheic exchange. They conclude that the
3938 variability of rock permeability and the relative contribution of groundwater during summer
3939 months, and their effect on stream temperatures following harvest should be investigated further.

3940

3941 **Wood Loading**

3942

3943 Burton et al., 2016

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3945 Burton, J.I., Olson, D.H., Puettmann, K.J., 2016. Effects of riparian buffer width on wood
3946 loading in headwater streams after repeated forest thinning. *Forest Ecology and Management*
3947 372, 247–257. <https://doi.org/10.1016/j.foreco.2016.03.053>

3948

3949 The purpose of this study was to examine the relationship between in-stream wood loading and
3950 riparian buffer width in thinned stands in conjunction with several stand, site, and stream
3951 variables. This study is a part of a larger density management study which covered 6 sites along
3952 the coastal and western Cascade Range of Oregon. The sites used for this study were dominated
3953 by Douglas-fir and ranged in age from 30-70 years old. Two consecutive thinning treatments
3954 took place on a portion of each site, while the other portions were designated as an unthinned
3955 control. Treated sites featured one of four buffer width prescriptions: (1) ~ 70-m buffer
3956 representative of one site potential tree, (2) ~15-m buffer, (3) a 6-m buffer representative of trees
3957 immediately adjacent to the stream. Wood surveys were carried out at four times during the
3958 study: (1) prior to the first thinning, (2) five years after the first thinning, (3) 9-13 years after the
3959 first thinning and just prior to the second thinning, and (4) one year after the second thinning. At
3960 each site, the first thinning was to 200 trees per ha (tph), the second thinning (~10 years later)
3961 was to ~85 tph, alongside an unthinned reference stand ~400 tph. Spatial and geomorphic
3962 characterization were measured using a combination of field and geospatial data. Hierarchical
3963 linear mixed models were developed with repeated measures using a multi-step process to
3964 examine relationships between large wood volume in headwater streams over time and in-stream
3965 wood characteristics (decay stage, zone), buffer width, time since thinning, and reach and
3966 geomorphology (drainage basin area, width:depth ratio, gradient). Wood volume was found to
3967 increase exponentially with drainage basin area; for every 1-ha increase in drainage basin area,
3968 wood volume increased by 0.63%. Slightly higher volumes of wood were found in sites with a
3969 narrow 6-m buffer, as compared with the 15-m and 70-m buffer sites in the beginning 5 years
3970 after the first harvest and maintained through year 1 of the second harvest (end of study). The
3971 authors attributed this difference to a higher likelihood of logging debris and/or windthrow but
3972 was not analyzed. Low volumes of wood from stands in the stem-exclusion phase were found to
3973 contribute to overall in-stream wood. The results showed that between 82-85% of the wood with
3974 discernable sources (90% for wood in early stages of decay; 45% of wood in late stages of
3975 decay) came from within 15 m of the stream, and the relative contribution of wood to streams

3976 declined rapidly with increasing distance. The authors hypothesize that this finding in
3977 conjunction with their results, which show a positive relationship between basin area and wood
3978 volume suggests a greater role for other large wood recruitment processes such as creep,
3979 landslides, and debris flow.

3980

3981 **Sediment**

3982

3983 Bywater-Reyes et al., 2018

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3985 Bywater-Reyes, S., Bladon, K.D., Segura, C., 2018. Relative Influence of Landscape Variables
3986 and Discharge on Suspended Sediment Yields in Temperate Mountain Catchments. *Water*
3987 *Resources Research* 54, 5126–5142. 10.1029/2017WR021728

3988

3989 The purpose of this paper was to improve our ability to predict suspended sediment yields by
3990 quantifying how sediment yields vary with catchment lithography and physiography, discharge,
3991 and disturbance history. This study took place at the HJ. Andrews Experimental Site in the
3992 Western Cascade Range of Oregon. The questions this paper sought to answer were (1) What is
3993 the relative association between discharge and catchment setting (i.e., lithology and
3994 physiography) and suspended sediment yields over an ~60-year period (2) Is there an
3995 association between historical forest management activities (i.e., forest harvesting and road
3996 building) or extreme hydrologic events and the spatial and temporal trends in suspended
3997 sediment yield Data was collected from 10 catchments, 8 within the Lookout Creek Watershed, 1
3998 just below the Lookout Creek Watershed, and 1 that drains to the adjacent Blue River. The data
3999 set spanned a 60-year period from 1955-2015 Methods for determining suspended sediment
4000 concentration involved using either vertically integrated storm-based grab samples, or discharge-
4001 proportional composite samples where composite samples were collected every three weeks at
4002 the outlet of each catchment. A linear mixed effects model (log transformed to meet the
4003 normality assumption) was used to predict annual sediment yield. In this model, site was treated
4004 as a random effect while discharge and physiographic variables were treated as fixed variables.
4005 This allowed for the evaluation of the relationships between sediment yield and physiographic
4006 features (slope, elevation, roughness, and index of sediment connectivity) while accounting for
4007 site. To account for the effect of disturbance history a variable was added to the model when the
4008 watershed had a history of management or natural disturbances. If the models for the disturbed
4009 watersheds significantly underpredicted the sediment discharge, the timing of the sudden
4010 increases were further examined to assess whether it correlated with a disturbance event. Last,
4011 the authors considered changes in stage derived from comparing measured historic stage values
4012 to those predicted from current rating curves. Changes in stage were interpreted as a relative bed-
4013 elevation change resulting from changes in scour and deposition of material likely moved as
4014 bedload. The results of this study show that sediment yield varied greatly across space and time

4015 with the lowest annual yield occurring in 2001 (~ 0.2 t/km²) at one catchment, and the highest
4016 annual yield (~ 953 t/km²) occurring in 1969 at another catchment. Annual suspended sediment
4017 yield was most strongly correlated with the standard deviation of watershed slope ($r = 0.72$), Only
4018 moderately correlated with slope ($r = 0.32$), and with drainage area ($r = 0.38$). Standard deviation
4019 of slope was also strongly correlated with TPI (a surface roughness index), and standard
4020 deviation of index of connectivity. When considering disturbance, the largest magnitude changes
4021 in bed-elevation (I.e., sediment movement), were after floods with a ≥ 30 -year return interval.
4022 The authors conclude that variability in watershed slope was the best predictor of annual
4023 suspended sediment yield relative to other physiographic variables. The authors report that the
4024 variability in watershed slope combined with cumulative annual discharge explained 67% of the
4025 variation in annual sediment yield across the 60-year data set. The results, however, show that
4026 annual sediment yields also moderately correlated with many other physiographic variables and
4027 caution that the strong relationship with watershed slope variability is likely a proxy for many
4028 processes, encompassing multiple catchment characteristics. For example, the strong relationship
4029 between watershed slope standard deviation and surface roughness. For the relationships
4030 between disturbance and sediment yield the authors conclude that the few anomalous years of
4031 high sediment yield occurred in watersheds with high slope variability and within a decade of
4032 forest management and a large flood event. The authors further caution that the high variability
4033 of sediment yield over space and time indicate that the factors tested in this study should be
4034 tested more broadly to investigate their utility to forest managers.

4035

4036

4037 **LW, Wildfire**

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4039 Chen et al., 2005

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4041 Chen, X., Wei, X., Scherer, R., 2005. Influence of wildfire and harvest on biomass, carbon pool,
4042 and decomposition of large woody debris in forested streams of southern interior British
4043 Columbia. *Forest Ecology and Management* 208, 101–114. doi:10.1016/j.foreco.2004.11.018

4044

4045 The purpose of this study was to compare the components of in-stream LW features between
4046 wildfire and forest harvesting disturbances. This study focuses particularly on the change in
4047 biomass and carbon pool among LW under different disturbances. This study was located in the
4048 central Okanagan Valley, Kelowna, British Columbia. A total of 19 forest streams, first and
4049 second order, within the study area were divided into four categories based on disturbance
4050 history of the adjacent upland forest and included: (1) riparian forest harvested 10 years ago; (2)
4051 riparian forest harvested 30 years ago; (3) riparian forest burnt ~ 40 years ago; and (4)
4052 undisturbed old-growth riparian forests that had a mean forest age of 163 years.. All harvested
4053 streams were clear-cut to the stream edge. New trees had established on these sites within 1-3

4054 years of harvest (planted or natural growth) and resulted in lodgepole pine being the dominant
4055 species. The wildfire streams included those that had been burnt ~40 years ago with no post-fire
4056 harvest or salvage logging. In stream LW was recorded for analysis if it had a minimum diameter
4057 of 10 cm and length of 1.0 m and were situated within the bankfull width. LW biomass was
4058 determined through the conversion of wood density and wood volume. LW was also categorized
4059 by decay class (3 classes), species, orientation submergence, and distance from the beginning of
4060 the study reach. Sampling took place during the period between July and October 2003 along a
4061 150 m study reach for each stream. An analysis of variance was used to determine the
4062 relationships between the chosen variables. When significant differences were found, the data
4063 was further analyzed with the data was fitted with a linear regression model to obtain
4064 correlations between the three variables (volume, biomass, and carbon). Results from this study
4065 show that on average the riparian sites disturbed by wildfire had the highest biomass, volume,
4066 and carbon content for individual LW pieces, followed by the 10-year harvest, then the old-
4067 growth forest; the 30-year harvest had the lowest of all streams for all parameters. Mean LW
4068 biomass of each individual piece of wood was significantly higher in sites which had been
4069 burned than in harvested sites. Biomass values were, on average, 31 kg in the wildfire sites,
4070 compared to 21 kg and 19 kg for sites harvested 10 years ago and 30 years ago, respectively. The
4071 volume of individual pieces in wildfire sites was significantly higher than in old-growth sites,
4072 and nearly significantly higher than in sites harvested 30 years ago. No statistical significance
4073 was found comparing piece volume in wildfire sites to sites harvested 10 years ago. The average
4074 carbon content of individual pieces of wood was also highest in the wildfire sites but the
4075 differences were not significant. The authors present data that the LW found in the wildfire and
4076 30-year harvest sites was mostly in the third decay class (most decayed), with less than 1% of
4077 LW in the class 1 decay class. Statistical significance was not discussed in the results for
4078 differences in decay class. The authors conclude that streams adjacent to wildfire disturbed and
4079 recently harvested (10-years post-harvest) forests contained significantly higher LW individual
4080 pieces and total volume than old-growth and 30-year post-harvest sites. Further because biomass,
4081 volume, and carbon were significantly higher in the 10-year post harvest sites, but there was no
4082 difference in the 30-year post-harvest sites and the old-growth sites; the authors speculate that
4083 harvest can increase the abundance of LW in the short-term from leaving harvest residues but
4084 reduces the abundance of LW over the long-term (~30 years post) due to a lack of recruitment
4085 from the young forests, and loss of in-stream LW from decomposition. The three main takeaways
4086 presented by the authors for this paper were (1) LWD input in old growth forested streams was
4087 relatively stable, (2) timber harvesting activities would cause a short-term increase of LWD
4088 stocks and might greatly reduce LWD loadings over a long-term, and (3) wildfire disturbance
4089 would delay LWD recruitment because not all burnt trees would fall in the stream immediately
4090 after the wildfire.

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4092

4093 **LW**

4094

4095 Chen et al., 2006

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4097 Chen, X., Wei, X., Scherer, R., Luider, C., Darlington, W., 2006. A watershed scale assessment of
4098 in-stream large woody debris patterns in the southern interior of British Columbia. *Forest*
4099 *Ecology and Management* 229, 50–62. <https://doi.org/10.1016/j.foreco.2006.03.010>

4100

4101 The purpose of this study was to (1) determine the spatial distribution and variation of LW
4102 characteristics (size, amount, volume, mass, orientation, position) within different order streams
4103 of forested watersheds; (2) to examine the relationship between LW characteristics and stream
4104 features through channel networks; and (3) to estimate the total density, volume and mass of LW
4105 at the watershed scale using a combination of field surveys and GIS data. This study took place
4106 at three different watersheds located in the south-central interior of British Columbia near
4107 Kelowna. A total of 35 study reaches with stream orders ranging from first- through fifth-order
4108 were selected to measure spatial distribution and variability of LW characteristics. Data collected
4109 for each reach was binned into 4 stream size categories (I = first order; II = second to third order;
4110 III = third to fourth order; IV = fourth to fifth order). Study sites were selected based on the
4111 following criteria. (1) the streams were in areas of intact mature riparian forests (>80 years); (2)
4112 the stream side forests were not disturbed by human activities, such as harvesting, road building;
4113 (3) the streams were not salvaged. Therefore, the results from this study provide a baseline of
4114 LWD characteristics in intact mature riparian forests in the southern interior of British Columbia.
4115 LW in this study is defined as having a diameter of > 0.1 m and a length > 1.0 m. LW
4116 characteristics (decay class, orientation, position within channel, distance from downstream end
4117 of channel) were recorded for any piece of LW that was within or above the bankfull width of the
4118 channel. Watershed features and the distribution of stream orders were derived from remotely
4119 sensed data. Mean values of LW density, volume, and biomass were compared between stream
4120 size classes with an analysis of variance (ANOVA). Results from this study show that LW size,
4121 volume, and biomass generally increased with increasing stream size. For example, the mean
4122 LWD diameter in stream size I (16.4 cm) was lower than that in stream size III (20.6 cm) and
4123 size IV (20.5 cm), respectively. Mean LW length also increases with stream size from 2.3 m in
4124 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
4125 volume (0.18 m³), significantly higher than stream size I (0.06 m³). LW volume was also
4126 significantly lower than in stream sizes II, and III. LW density (pieces per 100 m² of stream
4127 area), however, decreased as stream size increased. For example, LW density (defined as piece
4128 numbers per 100 m²) numbers were 19, 17, 12, and 4 for stream size I, II, III, and IV
4129 respectively. Increases in channel bankfull width ($R^2 = 0.52$) and stream area ($R^2 = 0.58$) was
4130 found to be strongly inversely correlated with LW density. Taken together, this study shows that
4131 spatial variation and distribution of LW characteristics vary as a function of stream size. From
4132 their results the authors conclude that in small sized streams, LW exhibit high density (number of
4133 pieces per 100 m²), low volume and biomass per unit area of stream. While in large sized
4134 streams, LW number, volume and biomass per unit of stream area are low but mean individual
4135 LW size was high.

4136

4137 **Stream Temperature Response to Harvesting**

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4139 Gravelle & Link, 2007

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4141 Gravelle, J.A., Link, T., 2007. Influence of Timber Harvesting on Headwater Peak Stream
4142 Temperatures in a Northern Idaho Watershed. *Forest Science* 53, 189–205.

4143

4144 The purpose of this study was to examine the effects of clearcutting and partial cutting on
4145 summer peak water temperatures in downstream fish-bearing streams, and to measure direct
4146 harvesting impacts on peak water temperature within headwater catchments. This study took
4147 place at the Mica Creek Experimental Watershed in Northern Idaho. Three headwater drainages
4148 were used to assess harvesting impacts on stream temperatures: (1) Watershed 1 which had 50%
4149 of the drainage area clearcut in 2001; (2) Watershed 2 which was thinned to a 50% target shade
4150 removal in Fall 2001; (3) and an unimpacted control. Riparian buffers were applied adjacent to
4151 the streams under the Idaho Forest Practices Act. This means, for fish-bearing streams the
4152 riparian management area must be at least 75 ft (22.9 m) wide on each side of the ordinary high-
4153 water mark (definable bank). Harvesting is still permitted, but there is a restriction where 75% of
4154 existing shade must be left. There are also leave tree requirements, which is a target number of
4155 trees per 1,000 linear feet (305 m), depending on stream width. For non-fish-bearing streams
4156 there is a 30 ft (9.1 m) equipment exclusion zone on each side of the ordinary high-water mark
4157 (definable bank). There are no shade requirements and no leave tree requirements, but skidding
4158 logs in or through streams is prohibited. Stream temperature data and canopy cover percentage
4159 data were collected at multiple sites within and downstream of treatment areas between 1992-
4160 2005. However, for the non-fish-bearing, headwater sites pre-treatment data was only collected
4161 one season prior to treatment. Temperature data was summarized as maximum daily temperature
4162 and was analyzed using simple linear regression to estimate changes in stream temperature
4163 following harvest during the summer months (July 1 – September 1). Results from this study
4164 show that there is no strong evidence of a posttreatment increase in stream temperature at long-
4165 term downstream sampling points for each harvest treatment. In general, the downstream sites
4166 showed a cooling effect between -0.2 and -0.3°C . The estimated cooling effect could not be
4167 attributed to any cause (e.g., increase in water yield), but the authors conclude that there was no
4168 post-harvest increase in peak summer temperatures at the downstream sites. For streams
4169 immediately adjacent to the clearcut treatment (headwater streams) a significant increase in
4170 temperature was detected at 2 sites ranging between 0.4 and 1.9°C , while a marginally
4171 significant decrease in temperature was detected at the third site (-0.1°C , $p = 0.06$). At the sites
4172 located immediately adjacent to partial cuts, results showed mixed results with decreases in
4173 temperature (-0.1°C ; non-significant) at one site and significant but minimal changes at another
4174 site (0.0 - 3.0°C) across the individual post-harvest years. Overall, there were minimal to no
4175 changes in stream peak temperatures following treatment in the partial-cut riparian areas. The

4176 authors go on to point out that headwater stream temperatures were highly variable, and that the
4177 shade value of understory vegetation may be an important factor contributing to results.

4178

4179 **SED**

4180

4181 Bywater-Reyes et al., 2017

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4183 Bywater-Reyes, S., Segura, C., Bladon, K.D., 2017. Geology and geomorphology control
4184 suspended sediment yield and modulate increases following timber harvest in temperate
4185 headwater streams. *Journal of Hydrology* 548, 754–769.
4186 <https://doi.org/10.1016/j.jhydrol.2017.03.048>

4187

4188 The purpose of this study was to assess the influence of natural controls (basin lithology and
4189 physiography) and forest management on suspended sediment yields in temperate headwater
4190 catchments. The study sought to achieve three objectives: (1) Quantify how suspended sediment
4191 yield varies by catchment setting in forested headwater catchments, (2) Determine whether
4192 contemporary forest management practices impact annual suspended sediment yield (SSY) in
4193 forested headwater catchments (3) Determine whether there are natural catchment settings that
4194 result in different levels of vulnerability or resilience to increases in suspended sediment yield
4195 associated with disturbances (e.g., harvest activities). This study analyzed 6 years of data from
4196 the Trask River Watershed in Northeastern Oregon and included data from harvested and
4197 unharvested sub-catchments underlain by heterogeneous lithologies. Baseline SSY data collection
4198 began in water year 2010 and continued through water year 2015, with road upgrades (July–
4199 August 2011) and harvest (May–November 2012) occurring in the middle of the study period.
4200 Generalized least square candidate models quantifying the parameters from each site were used
4201 to test differences in the relationship between suspended sediment yield and catchment setting.
4202 Results from this study indicate that site lithology was a first order control over SSY with SSY
4203 varying by an order of magnitude across lithologies observed. Specifically, SSY was greater in
4204 catchments underlain by Siletz Volcanics ($r = 0.6$), the Trask River Formation ($r = 0.4$), and
4205 landslide deposits ($r = 0.9$) and displayed an exponential relationship when plotted against
4206 percent watershed area underlain by these lithologies, combined. In contrast, the site effect had a
4207 strong negative correlation with percent area underlain by diabase ($r = 0.7$), with the lowest SSY
4208 associated with 100% diabase independent of whether or not earthflow terrain was present.
4209 Following timber harvest (water year 2013), increases in SSY occurred in all harvested
4210 catchments. The SSY in both PH4 (clearcut with buffers) and GC3 (clearcut without buffers)
4211 declined to pre-harvest levels by water year 2014. Interestingly, the SSY in UM2 (clearcut
4212 without buffers) increased annually throughout the post-harvest period, ultimately resulting in
4213 the highest SSY of all catchments during the final two years of the study after producing the
4214 lowest SSY in the pre-harvest period. Catchment physiographic variables (hypsoetry, slope,

4215 standardized topographic position index (SD TPI), and sediment connectivity (IC)) appeared to
4216 be good indicators of the underlying lithology of each site. Principle component analysis
4217 constructed from physiographic variables separated sites underlain by resistant diabase from
4218 those underlain by mixed lithologies along the PC1 axis. While sites along the second axis (PC2)
4219 were separated by relative values of earthflow terrain (high proportion vs. Little to none). Sites
4220 with low SSY and underlain by more resistant lithologies were also resistant to harvest-related
4221 increases in SSY. The authors conclude that sites underlain with a friable lithology (e.g.,
4222 sedimentary formations) had SSYs an order of magnitude higher, on average, following harvest
4223 than those on more resistant lithologies (intrusive rocks). In general, sites with higher SSY also
4224 had 1) lower mean elevation and slope, 2) greater landscape roughness, and 3) lower sediment
4225 connectivity (potential for sediment transport based on physiography). The authors suggest that
4226 their research be undertaken in different regions with different disturbance types to broadly apply
4227 their findings.

4228

4229 **Plant Communities**

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4231 D'Souza et al., 2012

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4233 D'Souza, L.E., Six, L.J., Bakker, J.D., Bilby, R.E., 2012. Spatial and temporal patterns of plant
4234 communities near small mountain streams in managed forests. *Can. J. For. Res.* 42, 260–271.
4235 <https://doi.org/10.1139/x11-17>

4236

4237 The purpose of this study was to examine spatial and temporal patterns in plant communities
4238 along fish-bearing streams in western Washington. The focus of this study is on areas which were
4239 harvested to the streambank within the last 100 years. The study took place in the western
4240 Cascade Mountains of Washington. Sites were randomly selected using a geographic information
4241 system. Stands that had been impacted by road development were excluded. Stands were
4242 stratified into a chronosequence of age classes: young (31-51 years), mature (52-70 years), old
4243 (>100 years). Due to availability, the sample sizes included 11 young stands, 10 mature stands,
4244 but only 4 old stands. Vegetation characteristics were captured in each stand using 0.16 ha plots
4245 located 30 m from stand edges to limit the influence of adjacent stands. Transects perpendicular
4246 to the stream were used 10 m apart and extended 80 m upslope. Vegetation and physical features
4247 along each transect were sampled using a series of subplots at 10 m intervals from the channel.
4248 The authors found little variation in riparian landform type and or canopy cover and were not
4249 included in the analysis for their effect on vegetation. Plant communities were examined
4250 spatially as a function of distance to stream and temporally by using the chronosequence of stand
4251 ages. Three distinct plant communities were observed in the shrub and herb layer (riparian: 0-9
4252 m; transitional: 10-29 m; and upslope: 30-80 m) and their composition differed significantly
4253 between communities. A total of 12 species were identified as indicators of these communities.

4254 For the shrub layer, community composition differed between old stands and young and mature
4255 stands. In the herb layer, community composition differed between all age classes. The results
4256 from this study suggest that plant communities along small fish-bearing streams have distinct
4257 changes in community with distance to stream, but also reflect successional status in nearby
4258 forests. The authors conclude by suggesting increased research in understanding the effects of
4259 forest management on streamside vegetation.

4260

4261 **LW Residence Time**

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4263 Hyatt & Naiman, 2001

4264

4265 Hyatt, T.L., Naiman, R.J., 2001. The Residence Time of Large Woody Debris in the Queets
4266 River, Washington, Usa. *Ecological Applications* 11, 191–202. [https://doi.org/10.1890/1051-
4267 0761\(2001\)011\[0191:TRTOLW\]2.0.CO;2](https://doi.org/10.1890/1051-0761(2001)011[0191:TRTOLW]2.0.CO;2)

4268

4269 The purpose of this study was to determine the depletion rate of LW by examining differences in
4270 size and species composition in the Queets River compared to the adjacent forest. This study
4271 took place in the Queets River Watershed located on the west slope of the Olympic Mountains in
4272 Washington. Field sampling was carried out at 25 transects and four different sites. Increment
4273 cores from in-stream LW were cross-dated against cores from riparian conifers to estimate the
4274 time which LW was recruited into the channel. LW pieces which were in a heightened state of
4275 decay were dated using carbon-dating techniques. the most common tree species (> 30 cm
4276 diameter) in the riparian zone is red alder, followed by Sitka spruce and western hemlock,
4277 whereas the most common species of LWD (> 30 cm diameter) is Sitka spruce, followed by red
4278 alder and western hemlock. Each of the hardwood species is better represented among standing
4279 trees than among LWD, and each of the conifers are better represented as LW than among trees
4280 in the riparian zone. The depletion curve developed in the results was based only on conifer LW
4281 because hardwood LW was either too small or too young to provide accurate estimates of
4282 residence time in the stream. Based on the depletion curve developed for all available LW
4283 showed that wood typically disappears from the active channel within the first 50 years, while
4284 some pieces may remain for several hundred years. By cross-referencing the LW depletion
4285 curves with field notes the authors suggest that the longer residence time, beyond 50 years, was
4286 dependent on more than one process such as burial. Decay class was not an accurate predictor of
4287 LW age. Also, Dependent vegetation on or around LWD was a poor and often misleading
4288 indicator of residence time. Many LWD pieces that had 1–5 year old vegetation growing on
4289 or around them were discovered to have died and presumably recruited to the channel 20 years
4290 previous. The authors conclude that LW originating from hardwoods is depleted faster than
4291 conifers. Considering the depletion rate curve, the authors speculate that the majority of LW is
4292 transported out of the system within 50 years, while pieces of LW that are buried or jammed in

4293 the river floodplain may remain for hundreds of years. Overall, ~80% of LW residing in the
4294 active channel were living within 50 years of the study. The authors explain there are several
4295 caveats to the depletion curve created for this study (1) the depletion constant was developed for
4296 a large, mostly alluvial river and should probably not be applied to smaller streams (mean
4297 bankfull width at study transects on the Queets is 165 m and the range is 51–398 m; mean key
4298 LWD length is 23.4 m, and the range is 5.3–69.0 m). Also, from the data the authors infer that
4299 alluvial channel trap wood from upstream, and constrained channels export LWD downstream,
4300 so it is not to be expected that the LWD resident in a channel was recruited from the riparian
4301 zone in that reach. In general, the authors conclude that for this study the depletion curve shows
4302 that the half-life of LW is ~20 years and thus all resident LW will be exported, buried, or broken
4303 down within 3-5 decades. Also, hardwood LW will be depleted from the channel more rapidly
4304 than conifers.

4305

4306 **Litter Input**

4307

4308 Hart et al., 2013

4309

4310 Hart, S.K., Hibbs, D.E., Perakis, S.S., 2013. Riparian litter inputs to streams in the central
4311 Oregon Coast Range. *Freshwater Science* 32, 343–358. <https://doi.org/10.1899/12-074.1>

4312

4313 The purpose of this study was to understand how riparian vegetation composition, understory
4314 density, and topography affect the quantity and quality of litter input to streams throughout the
4315 annual cycle. This study took place within 5 contiguous watersheds located in the central Coast
4316 Range of Oregon. At each of the study sites uniform areas along a ≤ 300 m stream reach, 3 plots
4317 were delineated on 1 side of the stream, each 8x 25 m along the stream. Three treatments were
4318 applied: (1) a no cut or fence control; (2) cut and remove a 5 x 8 m section adjacent to stream
4319 plants < 10 cm DBH and >12 cm height every 2 months; and (3) 5 m fence extending
4320 underground and parallel to the stream to block litter moving downslope from reaching stream.
4321 Vertical and lateral litter traps were installed at each site and collected monthly between August
4322 2003-August 2004. Variation of riparian vegetation and woody debris characteristics were
4323 analyzed with a 3-way ANOVA using overstory, treatments, and sections and their interactions.
4324 Two-way ANOVA with repeated measures was used to compare seasonal and monthly control
4325 and treatment inputs for different overstory and litter types. 1-way ANOVA was used to test for
4326 differences in nutrient concentration flux between overstory type. Results from this study show
4327 that deciduous forests dominated by red alder delivered significantly greater vertical and lateral
4328 inputs to stream than did coniferous forests dominated by Douglas-fir. Deciduous-site vertical
4329 litter input (mean, 95% CI; 504 g m⁻¹ y⁻¹, 446.6–561.9) exceeded that from coniferous sites
4330 (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
4331 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

4332 sites (63 g m⁻¹ y⁻¹, 28.9– 96.6). Lateral inputs calculated for a 3-m-wide stream accounted for
4333 9.6% (5.4–12.5) of total annual inputs at coniferous sites and 12.7% (10.2–14.5) of total inputs at
4334 deciduous sites. Composition of litter also differed significantly by overstory type. Annual lateral
4335 inputs at coniferous sites were dominated by deciduous leaves (,33%), twigs (,23%), and leftover
4336 (,18%) litter types, whereas annual lateral inputs at deciduous sites were deciduous leaves (,61%)
4337 and leftover (,15%) litter types. Leftover litter types were defined as those that were too small or
4338 decayed to identify, bark, moss, or lichens. Vertical litter inputs at deciduous sites were
4339 dominated by deciduous leaves (,65%) and deciduous-other (,15%) litter types. While deciduous
4340 leaves (,33%), coniferous needles (,24%), and twigs (,21%) composed the annual vertical litter
4341 inputs at coniferous sites. The strongest deciduous inputs to streams occurred in November.
4342 Annual lateral litter input increased with slope at deciduous sites ($R^2 = 0.4073$, $p = 0.0771$), but
4343 showed no strong relationship at coniferous sites ($R^2 = 0.1863$, $p = 0.2855$). Total nitrogen flux
4344 to streams at deciduous sites was twice as much as recorded at coniferous sites. However, there
4345 was seasonal effect where the N fluxes in deciduous sites was only higher in autumn. The
4346 authors of this study conclude by suggesting management in riparian areas consider utilizing
4347 deciduous species such as red alder for greater total N input to aquatic and terrestrial ecosystems
4348 along with the increased shade and large woody debris provided by coniferous species.

4349

4350 **Effect of Contemporary Management on Nutrient Concentration and Cycling**

4351

4352 Gravelle et al., 2009

4353

4354 Gravelle, J.A., Ice, G., Link, T.E., Cook, D.L., 2009. Nutrient concentration dynamics in an
4355 inland Pacific Northwest watershed before and after timber harvest. *Forest Ecology and*
4356 *Management* 257, 1663–1675. <https://doi.org/10.1016/j.foreco.2009.01.017>

4357

4358 The purpose of this study was to assess the effects of contemporary forest harvesting practices on
4359 nutrient cycling and concentrations. This study took place at the Mica Creek Experimental
4360 Watershed in Northern Idaho. Seven steel Parshall flumes were installed at select locations
4361 within the watershed to assess the effects of clearcut to stream and partial cut (50% shade
4362 retention) harvesting practices. All harvesting was conducted in compliance with the Idaho
4363 Forest Practices Act. Within fish-bearing streams (Class I) Harvesting is permitted, but 75% of
4364 existing shade must be retained. There are also leave tree requirements for a target number of
4365 trees per 1000 linear feet (305 m), depending on stream width. In Mica Creek, this was roughly
4366 200 trees in the 3–12 in. (8–30 cm) diameter class per 305 m of the riparian management zone
4367 (RMZ). Along non-fish-bearing streams (Class II) the RMZ is 30 feet (9.1 m) of equipment
4368 exclusion zone on each side of the ordinary high-water mark (definable bank); skidding logs in
4369 or through streams is prohibited. There are no shade requirements and no requirements to leave
4370 merchantable trees. Two-sided riparian buffers were left on all Class I streams during harvest

4371 operations. Timber was removed from both sides of the Class II streams. In the post-harvest and
4372 post-burn conditions, Class II streams in clearcut treatments had only a small amount of green
4373 tree retention within the riparian zone, while in partial cut treatments equal amounts of canopy
4374 cover (approximately 50%) were removed from both sides of the stream. This study followed the
4375 BACI design and featured a pre-treatment measurement phase (1992-1997), a post-road
4376 construction phase (1997-2001), and a post-harvest phase (2001-2006). A students t-test was
4377 used to analyze the data between the observed and predicted values of post-treatment sites for
4378 several nitrogen and phosphorus compound concentrations (Kjeldahl nitrogen (TKN), nitrate +
4379 nitrite (NO₃ + NO₂), TP, total ammonia nitrogen (TAN) consisting of unionized (NH₃) and
4380 ionized (NH₄⁺) ammonia, and unfiltered orthophosphate (OP) samples). Results from the post-
4381 road construction period showed no significant changes in concentrations of any nutrients
4382 analyzed. Results from this study show statistically significant increases in NO₃ and NO₂
4383 concentrations following clearcut and partial harvest cuts in headwater streams. Increases at the
4384 clearcut treatment site were greatest, where mean monthly concentrations increased from 0.06
4385 mg-N L⁻¹ during the calibration and post-road periods to 0.35 mg-N L⁻¹. There was also an
4386 observable seasonal effect on NO₃ + NO₂ concentrations with the peak concentration of 0.89
4387 mg-N L⁻¹ occurred at F1 in April 2004, with mean monthly concentrations of 0.43 mg-N L⁻¹
4388 and 0.59 mg-N L⁻¹ in water years (October–September) 2004 and 2005, respectively. Similar
4389 results were also observed at sites further downstream although changes were smaller which, the
4390 authors point out this may be due to in-stream uptake and/or dilution. No significant changes of
4391 in-stream concentration of any other nutrient recorded were found between time periods and
4392 treatments except for one downstream site that showed a small increase in orthophosphate by
4393 0.01 mg P L⁻¹. In general, the results of this study show that forest management influences in-
4394 stream NO₃ + NO₂ immediately adjacent to treatment and downstream of treatment. The authors
4395 conclude by suggesting future research in understanding variability in nutrient concentrations
4396 and cycling as affected by seasons and storm runoff events.

4397

4398 **Organic Matter Inputs**

4399

4400 Kiffney & Richardson, 2010

4401

4402 Kiffney, P.M., Richardson, J.S., 2010. Organic matter inputs into headwater streams of
4403 southwestern British Columbia as a function of riparian reserves and time since harvesting.
4404 *Forest Ecology and Management* 260, 1931–1942. <https://doi.org/10.1016/j.foreco.2010.08.016>

4405

4406 The purpose of this paper was to assess how differences in riparian buffer width and timing since
4407 harvest affect terrestrial particulate organic matter flux into streams. The focus of this paper was
4408 on 1st and 2nd order headwater streams located approximately 45 km east of Vancouver in
4409 British Columbia, Canada. Sites were measured over an 8-year period and included clear-cut

4410 (n=3), 10-m buffered reserve (n=3), 30-m buffered reserve (n=2), and uncut control (n=2)
4411 treatments. For streams receiving a 10 or 30-m reserve, there was no logging on either side of the
4412 stream within these reserves. Study reaches were approximately 200m long. Vertical litter inputs
4413 were collected monthly and at approximately 6–8-week intervals during each season for years
4414 1,2,6,7, and 8 years after harvest. Litter was separated into broadleaf deciduous, twig, needles,
4415 and other (seeds, cones, and moss) categories following collection and subsequently dried and
4416 weighed using a microbalance. A mixed-model analysis of covariance was used for Fall data
4417 with riparian treatment as a fixed effect and year as a covariate. Secondly, ordinary least
4418 squares regression was used to quantify the functional relationship between reserve width and
4419 litter flux within each year. Results show riparian treatments having significant effects on the
4420 quantity and composition of litter input into streams. Inputs consisting of needles and twigs were
4421 significantly lower while deciduous inputs were higher in clearcuts compared to other
4422 treatments. Differences in litter flux relative to riparian treatment persisted through year 7, while
4423 a positive trend between reserve width and litter flux remained through year 8. For example, one-
4424 year post-treatment, needle inputs were 56x higher during the Fall into control and buffered
4425 treatments than into the clearcut. Needle inputs remained 6x higher in the buffer and control sites
4426 through year 7, and 3-6x higher in year 8 than in the clearcut sites. Twig inputs into the control
4427 and buffered sites were ~25x higher than in the clearcut sites in the first year after treatment.
4428 There was no significant difference in treatment for deciduous litter but a trend of increasing
4429 deciduous litter input in the clear cut was observed in the data. For example, one-year post-
4430 treatment deciduous litter was lowest in the clearcut, but by year 8 deciduous litter was highest in
4431 the clearcut sites relative to control and buffered sites. The linear relationship between reserve
4432 width and litter inputs was strongest in the first year after treatment, explaining ~57% of the
4433 variation, but the relationship could only explain ~17% of the variation in litter input by buffer
4434 width by year 8 (i.e., the relationship degraded over time). The authors interpret these results as
4435 evidence that riparian reserves showed a similar litter flux to streams when compared to uncut
4436 controls. They also conclude that litter flux from riparian plants to streams, was affected by
4437 riparian reserve width, time since logging, and potentially channel geomorphology.

4438

4439 **In-stream Wood Loads**

4440

4441 Jackson & Wohl, 2015

4442

4443 Jackson, K.J., Wohl, E., 2015. Instream wood loads in montane forest streams of the Colorado
4444 Front Range, USA. *Geomorphology* 234, 161–170.
4445 <http://dx.doi.org/10.1016/j.geomorph.2015.01.022>

4446

4447 The purpose of this study was to examine in-stream wood loads and geomorphic effects between
4448 stands of different ages and stands with different disturbance histories The first objective of this

4449 study was to determine whether instream wood and geomorphic effects differ significantly
4450 among old-growth, younger, healthy, and beetle-infested forest stands. The second objective of
4451 this study was to determine whether instream wood loads correlate with valley and channel
4452 characteristics. The authors hypothesized that streams in old-growth montane forests have (1)
4453 significantly larger in stream and floodplain wood loads than those in younger stands, (2) greater
4454 frequency of volume of jams than those in younger forests, and (3) more wood created
4455 geomorphic effects. They also hypothesized that instream wood loads in healthy montane forests
4456 are significantly smaller than in beetle-infested forests. Last, they hypothesized that instream
4457 wood load correlates with lateral valley confinement, with unconfined valleys having the greatest
4458 in-stream and total wood loads. This study took place within the Arapaho and Roosevelt National
4459 Forests in Colorado. Sediment storage, channel geometry, in-stream wood load, and forest stand
4460 characteristics were measured along 33 pool-riffle or plane-bed stream reaches (10 located in
4461 old-growth (> 200 years); 23 located in younger forests (age range not reported)). LW
4462 characteristics were recorded for all in-stream wood ≥ 10 cm diameter and ≥ 1 m in length. Pair-
4463 wise t-test or Kruskal-Wallis tests were used to check for significant differences in wood load,
4464 logjam volume, and logjam frequencies. To test for significant differences in wood created
4465 geomorphic effects a principal component analysis was used. Results indicated that channel
4466 wood load (OG = 304.4 ± 161.1 ; Y = 197.8 ± 245.5 m³ /ha), floodplain wood load (OG = 109.4
4467 ± 80 ; Y = 47.1 ± 52.8 m³ /ha), and total wood load (OG = 154.7 ± 64.1 ; Y = 87.8 ± 100.6 m³
4468 /ha) per 100 m length of stream and per unit surface area were significantly larger in streams of
4469 old-growth forests than in young forests. Streams in old-growth forests also had significantly
4470 more wood in jams, and more total wood jams per unit length of channel than in younger forests
4471 (jam wood volume: OG = 7.10 ± 6.9 m³; Y = 1.71 ± 2.81 m³). When standardized to stream
4472 gradient, old-growth streams had significantly greater pool volume and significantly greater
4473 sediment volume than younger stands. No significant difference was detected in in-stream wood
4474 loads between healthy and beetle-infested stands. Although wood load in streams draining from
4475 pine beetle infested forests did not differ significantly from healthy forests, best subset regression
4476 (following principal component analysis) indicated that elevation, stand age, and pine beetle
4477 infestation were the best predictors of wood load in channels and on floodplains. The authors
4478 speculate that beetle infestation is affecting in-stream wood, but perhaps not enough time has
4479 passed since the infestation for the affected trees to fall into the stream. Time since beetle-
4480 infestation was not reported.

4481

4482 **LW Recruitment**

4483

4484 May & Gresswell, 2003

4485

4486 May, C.L., Gresswell, R.E., 2003. Large wood recruitment and redistribution in headwater
4487 streams in the southern Oregon Coast Range, U.S.A. *Can. J. For. Res.* 33, 1352–1362.

4488 <https://doi.org/10.1139/x03-023>

4489

4490 The purpose of this study was to understand the relative influence of processes that recruit and
4491 redistribute wood into channels and to understand how these processes vary spatially. Specific
4492 research questions included the following:(i) Do processes that deliver and redistribute wood
4493 differ in small colluvial channels compared with larger alluvial channels? (ii) Do proximal and
4494 distal controls on wood delivery differ for colluvial and alluvial channels? (iii) How do input and
4495 redistribution processes influence the functional role of wood in the channel? The focus of this
4496 research is specifically on differences between small colluvial channels and large alluvial
4497 channels in the southern Oregon Coast Range. All downed wood exceeding 20 cm mean
4498 diameter and 2 m in length, and in contact with the bank-full channel were measured in three
4499 second order and one third-order stream. Large wood was categorized based on the various
4500 mechanisms delivering it to the stream channel. Categories included (i) direct delivery from local
4501 hillslopes and riparian areas, (ii) fluvial redistribution, (iii) debris flow transported, or (iv) an
4502 unidentified source. Results from this study show that stream size and topographic position
4503 strongly influence processes that recruit and redistribute wood in channels. Processes of slope
4504 instability were shown to be important conveyors of wood from upland forests to small colluvial
4505 channels. In the larger alluvial channels, windthrow was found to be the dominant recruitment
4506 process from adjacent riparian area. Results showed that Wood derived from local hillslopes and
4507 riparian areas accounted for the majority of pieces (63%) in small colluvial channels. The larger
4508 alluvial channel received wood from a greater variety of sources, including recruitment from
4509 local hillslopes and riparian areas (36%), fluvial redistribution (9%), and debris flow transported
4510 wood (33%). However, because pieces recruited from local sources (hillslope and riparian area)
4511 were larger, these sources of wood had a disproportionately large contribution to volume of wood
4512 in the stream. For example, wood recruited from the local hillslopes and riparian areas accounted
4513 for 36% of wood pieces in the alluvial stream, which accounted for 74% of the total volume of
4514 wood. Slope instability and windthrow were the dominant mechanisms for wood recruitment into
4515 small colluvial channels. Windthrow was the dominant recruitment mechanism for wood
4516 recruitment into larger alluvial channels. Distributions of the source distance of wood pieces
4517 were significantly different between colluvial and alluvial channels. In colluvial streams, 80% of
4518 total wood and 80% of total wood volume recruited originated from trees rooted within 50 m of
4519 the channel. In the alluvial channel, 80% of the pieces of wood and 50% of the total volume
4520 originated from trees which came from 30 m of the channel. The primary function of wood in
4521 smaller colluvial channels was sediment storage (40%) and small wood storage (20%). The
4522 primary function of wood in larger alluvial channels is bank scour (26%), stream bed scour
4523 (26%), and sediment storage (14%). Recruitment and redistribution processes were shown to
4524 affect the location of the piece relative to the channel/flow direction, thus influencing its
4525 functional role. The authors conclude that wood recruited from local sources is variable by
4526 position in the stream network because of differences in recruitment processes, degree of
4527 hillslope constriction, and slope steepness.

4528

4529 **Sediment**

4530

4531 Macdonald et al., 2003a

4532

4533 Macdonald, J. S., Beaudry, P. G., MacIsaac, E. A., & Herunter, H. E. (2003). The effects of forest
4534 harvesting and best management practices on streamflow and suspended sediment concentrations
4535 during snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada.
4536 Canadian Journal of Forest Research, 33(8), 1397-1407. <https://doi.org/10.1139/x03-110>

4537

4538 (BACI, only single year pre-harvest)

4539

4540 This study investigates the changes in suspended sediment concentration and stream discharge
4541 during freshet (spring snowmelt) at two harvest intensities relative to each other and an
4542 unharvested control watershed, pre- and post-harvest. The design included three small sub-
4543 boreal, first order, forest streams (<1.5 m width) in the central interior of British Columbia
4544 (Baptiste watershed). Both treatment streams received a 55% harvest treatment; one (low-
4545 retention) removed all merchantable timber >15 cm DBH for pine and > 20 cm DBH for spruce
4546 within 20 m of the stream; the other treatment (high-retention) removed all merchantable timber
4547 > 30 cm within 20 m of the stream; and an un-harvested control. Data for stream flow and total
4548 suspended sediments (TSS) was collected using Parshall flumes downstream from the treatment
4549 and control sites for one-year pre- and four-years post-harvest during snowmelt periods.

4550 Regression analysis was used to analyze relationships between treatment and control reaches pre-
4551 and post-treatment to estimate and compare predicted changes in TSS. The results showed an
4552 increase in freshet discharge for both treatments above predicted values for the entirety of the
4553 study. During the year prior to treatment, TSS relationships of both treatment watersheds during
4554 freshet closely matched those of the control. Immediately following harvest TSS concentrations
4555 increased above predicted values for both treatment streams. Increased TSS persisted for two-
4556 years post-harvest in the high-retention treatment, and for 3-years in the low-retention. The
4557 authors speculate that the treatment areas may have accumulated more snow (e.g., more exposed
4558 area below canopy) than in the control reaches leading to the increase in discharge. This study
4559 shows evidence that harvest intensity (low vs. high retention) is proportional to the increase in
4560 stream discharge, TSS, and recovery time to pre-harvest levels.

4561

4562 **LW**

4563

4564 Fox & Bolton, 2007

4565

4566 Fox, M., & Bolton, S. (2007). A regional and geomorphic reference for quantities and volumes of
4567 instream wood in unmanaged forested basins of Washington State. *North American Journal of*
4568 *Fisheries Management*, 27(1), 342-359. <https://doi.org/10.1577/M05-024.1>

4569

4570 This study uses in-stream LW values from 150 stream segments located in unmanaged
4571 watersheds, across all of Washington State, to investigate the relationships between
4572 geomorphology, forest zone, and disturbance regimes with LW recruitment. The purpose of this
4573 study was to create a base-line value of central tendency for in-stream LW values in “natural”
4574 streams for which salmonids are theoretically adapted. The authors define natural and
4575 unmanaged as streams that (1) had no part of the basin upstream of the survey site ever logged
4576 using forest practices common after European settlement and (2) the basin upstream of the
4577 survey site contains no roads or human modifications to the landscape that could affect the
4578 hydrology, slope stability, or other natural processes of wood recruitment and transport in
4579 streams. Sites were stratified to capture the variations in forest types, channel morphologies, and
4580 hydrological origins. The authors used descriptive statistics to establish and evaluate correlations
4581 between wood loading and watershed characteristics to reveal the highest valued variables
4582 influencing wood loading. Following this analysis, the variables with the highest mechanistic
4583 values in determining wood loading were evaluated and compared using simulation modeling.
4584 Results showed that in-stream wood volume increased with drainage area and as streams became
4585 less confined. However, bank full width (BFW) was a significantly better predictor of wood
4586 parameters than basin size. There was observational evidence that alluvial channels contained
4587 more wood volume on average than bedrock channels. However, due to limits in sample size
4588 following stratification, statistical analysis could not be completed. Sample sizes for isolating
4589 gradient and confinement were also too small to apply statistical analyses. Fire was found to
4590 influence in-stream wood quantities and volumes west of the Cascade crest; In-stream wood
4591 volume increased with adjacent riparian timber age as determined by the last stand replacing fire.
4592 Other disturbances such as debris flow, snow avalanche, and flooding were too few in frequency
4593 in the study area to be analyzed statistically. From these results the authors developed thresholds
4594 for expected “key piece volume (m^3) (pieces with independent stability) of wood for three BFW
4595 classes (20-30 m, >30 – 50 m, > 50 m width) per 100 m stream length for streams with BFW
4596 greater than 20 m. From percentile distributions the authors recommend minimum volumes,
4597 defined by the 25th percentiles, of approximately 9.7 m^3 for the 20- to 30-m BFW class, 10.5 m^3
4598 for the 30- to 50-m³ BFW class, and 10.7 m^3 for channels greater than 50 m BFW per 100 m
4599 length of stream. The results of this study suggest that BFW is the single greatest predictor of in-
4600 stream wood quantity and volume relative to other predictor variables. However, this result
4601 comes with the caveat that other processes and geomorphologies (e.g., channel bed form,
4602 gradient, confinement) are also important in the mechanisms for wood recruitment, modeling in
4603 this study showed too much inconsistency with these predictor variables too draw strong
4604 conclusions. Further the authors warn that these values for reference conditions are only
4605 applicable to streams with bank-full widths between 1 and 100 m, gradients between 0.1% and
4606 47%, elevations between 91 and 1,906 m, drainage areas between 0.4 and 325 km^2 , glacial and
4607 rain- or snow-dominated origins, forest types common to the Pacific Northwest.

4608

4609 **LW and sediment**

4610

4611 Gomi et al., 2001

4612

4613 Gomi, T., Sidle, R. C., Bryant, M. D., & Woodsmith, R. D. (2001). The characteristics of woody
4614 debris and sediment distribution in headwater streams, southeastern Alaska. *Canadian Journal of*
4615 *Forest Research*, 31(8), 1386-1399. <https://doi.org/10.1139/x01-070>

4616

4617 This study investigated different riparian conditions related to harvest and disturbance
4618 (landslides), their influence on woody debris and sediment distributions, and their related
4619 functions in headwater streams. This study examined the effects of recent and past timber
4620 harvests on woody debris abundance and distribution, landslides and debris flow on woody
4621 debris abundance and sediment accumulations, and the function of in-stream woody debris on
4622 sediment storage. The researchers examined 15 steep headwater streams in the Maybeso
4623 Experimental Forest and Harris River basin in the Tongass National Forest, Prince of Wales
4624 Island, southeastern Alaska. Treatments of headwater streams included five management or
4625 disturbance regimes: old growth (OG), recent clear-cut (CC; 3 years), young growth conifer
4626 forest (YC; 37 years after clear-cut), young growth alder (YA; 30 years after clear-cut), and
4627 recent landslide and debris flow channels (LS). Three headwater streams were sampled for each
4628 of the 5 treatments, 15 streams total. Analysis of covariance (ANCOVA) was used to compare
4629 LW quantity and distribution, and sediment quantity and distribution, across plots nested within
4630 each treatment site. Results showed in-channel numbers of LW pieces were significantly higher
4631 in YC and CC sites when compared to OG, YA, and LS sites. The number of LW pieces was
4632 highest in YC streams even though logging concluded 3 decades prior to sampling. No
4633 significant differences in LW volume were found among OG, CC, and YC streams. However,
4634 LW volume per 100 m of stream length in YC was twice that in OG. The total volume of LW per
4635 100 m associated with CC channels was half that in OG channels. However, the majority of the
4636 LW volume in OG systems was outside of the bank-full area. When the data was stratified by
4637 channels that experienced landslides (LS and YA), the number of LW pieces among OG, YA, and
4638 LS was not statistically significant. However, the in-channel volumes of LW in LS and YA
4639 channels were significantly lower than in OG sites because individual LW pieces in the OG sites
4640 were relatively larger than in the LS and YA sites. There was high variability among sites in the
4641 amount of sediment stored within streams. The authors conclude that timber harvesting and
4642 related landslides and debris flows affect the distribution and accumulation of LW and related
4643 sediment accumulation in headwater streams. These effects are summarized as (i) inputs of
4644 logging slash and unmerchantable logs significantly increase the abundance of in-channel woody
4645 debris; (ii) in the absence of landslides or debris flows, these woody materials remain in the
4646 channel 50–100 years after logging; (iii) relatively smaller woody debris initially stores
4647 sediment; (iv) when landslides and debris flows occur 3–15 years after logging because of

4648 intensive rain and weakening of root strength (Sidle et al. 1985), woody debris is evacuated from
4649 headwater streams and deposited in downstream reaches; (v) although less woody debris remains
4650 in the scour zone, woody debris pieces and jams contribute to sediment storage in both the scour
4651 and deposition zones of landslide and debris flow channels; (vi) red alder stands actively
4652 recolonize riparian zones of headwater streams for 20–50 years after mass movement and recruit
4653 woody debris and organic materials, which in turn provide sediment storage sites; and (vii)
4654 subsequent sediment movement after landslides and debris flows are affected by residual woody
4655 debris and newly introduced debris.

4656

4657 **LW and sediment**

4658

4659 Johnson et al., 2000 (removed from focal list)

4660

4661 Johnson, S. L., Swanson, F. J., Grant, G. E., & Wondzell, S. M. (2000). Riparian forest
4662 disturbances by a mountain flood—the influence of floated wood. *Hydrological processes*,
4663 14(16-17), 3031-3050. [https://doi.org/10.1002/1099-1085\(200011/12\)14:16/17<3031::AID-
4664 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)

4665

4666 This study examined the differences in riparian forest responses to a 100-year flood event along
4667 eight third- to fifth-order streams in the Cascade Mountain Range of Oregon. Disturbance
4668 intensities were grouped into three categories: purely fluvial (high water flow only), fluvial with
4669 uncongested wood transport, and fluvial with congested wood transport. Riparian forest
4670 responses were heavily influenced by pre-flood forest structure and disturbance/harvest history,
4671 especially the characteristics of LW presence within streams and along channels. The quantity
4672 and severity of toppled trees (fully uprooted vs. partially uprooted) during the flood event was
4673 proportional to the quantity and congestion of LW already present (i.e., higher volumes of LW
4674 already present during the flood event increased the frequency of toppled trees and newly
4675 deposited LW in streams). Further, stands that experienced higher frequencies of toppled trees
4676 also showed higher frequencies and magnitudes of debris flow. The authors concluded that the
4677 land use practices, and disturbance histories influenced the age and structure of the riparian
4678 forests, but also the availability of the agents of disturbance (presence of LW) during the 100-
4679 year flood event. This paper is a good discussion of how pre-disturbance structure affects the
4680 response of riparian forests to disturbances (in this case, flood), however, there is no statistical
4681 analysis discussed in the methods. This is purely descriptive science that involves an intensive
4682 survey of before and after riparian forest structures.

4683

4684 **Sediment**

4685

4686 Yang et al., 2022 (removed from focal list)

4687

4688 Yang, Y., Safeeq, M., Wagenbrenner, J. W., Asefaw Berhe, A., & Hart, S. C. (2022). Impacts of
4689 climate and forest management on suspended sediment source and transport in montane
4690 headwater catchments. *Hydrological Processes*, 36(9), e14684.
4691 <https://doi.org/10.1002/hyp.14684>

4692

4693 This paper investigates the changes in annual hysteresis patterns for in-stream suspended
4694 sediment in 10 headwater streams at 2 sites, Providence Creek (rain-snow-dominated,
4695 transitional), and Kings River Experimental Watershed (snow-dominated). Aside from
4696 precipitation pattern differences in the two catchments, the researchers also compared differences
4697 in hysteresis patterns for forested riparian control, burn-only, thin-only, and thin-and-burn
4698 combined areas. The differences in the proportion of clockwise-loop hysteresis patterns for
4699 suspended sediments in the warmer rain-snow-transition sites compared to the colder snow-
4700 dominated sites suggests that warming temperatures may cause the snow-dominated basins to
4701 receive sediment from extended source areas and for longer periods if they transition to rain
4702 dominated catchments. The results found no discernable difference in hysteresis loops between
4703 the control, burn-only, thin-only, and thin-and-burn combined areas. Further, there seemed to be
4704 little change in the hysteresis loops during drought, average, and excessively wet years. The
4705 authors speculate that local conditions will be more important in understanding the impacts of
4706 climate change than changes in precipitation patterns or average annual temperatures alone.
4707 Mainly, there is evidence that if snow-dominated watersheds become warm enough to transition
4708 to rain-dominated, there is potential for disruption to sediment discharge frequency, rates, and
4709 source distance. The indiscernible difference in hysteresis loops for the different treatments also
4710 suggests that management practices imposed to ameliorate these changes may not be completely
4711 effective.

4712

4713 **Nutrients**

4714

4715 Vanderbilt et al., 2003

4716

4717 Vanderbilt, K. L., Lajtha, K., & Swanson, F. J. (2003). Biogeochemistry of unpolluted forested
4718 watersheds in the Oregon Cascades: temporal patterns of precipitation and stream nitrogen
4719 fluxes. *Biogeochemistry*, 62(1), 87-117. DOI:10.1023/A:1021171016945

4720

4721 This study uses long-term datasets (ranging from 20-30 years) from six watersheds in the H.J.
4722 Andrews Experimental Watershed (HJA) in the west-central Cascade Mountains of Oregon to
4723 investigate patterns in dissolved organic nitrogen (DON) and dissolved inorganic nitrogen (DIN)
4724 export with watershed hydrology. The objectives of this study were to 1) characterize long-term
4725 patterns of N dynamics in precipitation and stream water at the HJA, 2) analyze relationships
4726 between annual output of N solutes and annual stream discharge, 3) analyze relationships
4727 between seasonal stream water N solute concentrations and precipitation and stream discharge,
4728 and 4) compare results with those from other forested watersheds. Precipitation data were
4729 collected at three-week intervals from 10/1/1968 until 5/24/1988 and at one-week intervals
4730 thereafter. Stream chemistry samples were collected weekly for the entirety of the study. Stream
4731 discharge was measured continuously throughout the study. The researchers used regression
4732 analysis of annual N inputs and outputs with annual precipitation and stream discharge to
4733 analyze patterns. The results showed DON was the largest component of N input at the low-
4734 elevation collector, followed by PON (particulate organic N), NO₃-N, and NH₄-N. At the high-
4735 elevation collector, NO₃-N input was higher than at low elevation and was the largest component
4736 of N in bulk and wet-only inputs, followed by NH₄-N, DON, and PON. For annual stream
4737 outputs, DON was the largest fraction of annual N output, followed by PON, NH₄-N and then
4738 NO₃-N. Total annual discharge was a positive predictor of annual DON export in all watersheds
4739 with r² values ranging from 0.42 to 0.79. In contrast, significant relationships between total
4740 annual discharge and annual export of NO₃-N, NH₄-N, and PON were not found in all
4741 watersheds. No systematic long-term average seasonal trends were observed for NO₃-N or PON
4742 concentrations. Elevated concentrations of NH₄-N occurred in spring and early summer in all
4743 three watersheds, although they are not convincingly synchronous. DON concentrations
4744 increased in the fall in every watershed. The increase in concentration began in July or August
4745 with the earliest rain events, and peak DON concentrations occurred in October through
4746 December before the peak in the hydrograph. DON concentrations then declined during the
4747 winter months. The authors conclude that total annual stream discharge was a positive
4748 predictor of DON output suggesting a relationship to precipitation. Also, DON had a consistent
4749 seasonal concentration pattern. All other forms of N observed showed variability and
4750 inconsistencies with annual and seasonal stream discharge. The authors speculate that different
4751 factors may control organic vs. Inorganic N export. Also, DIN may be strongly influenced by
4752 terrestrial or in-stream biotic controls, while DON is more strongly influenced by climate. Last,
4753 the authors suggest that DON in streams may be recalcitrant, and largely unavailable to stream
4754 organisms. The authors emphasize the importance of analyzing data from multiple watersheds in
4755 a single climactic zone to make inferences about stream chemistry.

4756

4757 **Stream temperature**

4758

4759 Roon et al., 2021b

4760

4761 Roon, D. A., Dunham, J. B., & Torgersen, C. E. (2021). A riverscape approach reveals
4762 downstream propagation of stream thermal responses to riparian thinning at multiple scales.
4763 *Ecosphere*, 12(10), e03775. <https://doi.org/10.1002/ecs2.3775>

4764

4765 This study uses a riverscape approach to evaluate the effects of streamside forest thinning on
4766 stream temperatures at multiple spatiotemporal scales. This study addresses the question of how
4767 thinning second-growth riparian forests influences local and downstream temperatures at
4768 watershed extents. This study attempts to answer this question by addressing four objectives: (1)
4769 quantify pretreatment spatial and temporal variability in stream temperature conditions; (2)
4770 evaluate local responses in stream temperature to riparian thinning; (3) assess the spatial extent
4771 and temporal duration of downstream effects to local responses in temperature; and (4)
4772 characterize local and downstream responses to thinning with a conceptual framework based on
4773 waveforms. The researchers compared upstream, local, and downstream, stream temperature
4774 fluctuations following different intensities of streamside forest thinning at 10 treatment reaches
4775 across three watersheds in the redwood forests of northern California. Treatments varied by
4776 landowners. In two watersheds thinning treatments were intended to reduce 50% of canopy
4777 closure within the riparian zone along a 200 m reach on both sides of the active channel. This
4778 treatment resulted in a reduction in effective shade over the stream between 19-30%. In the other
4779 treatment watershed, thinning treatments reduced basal area by as much as 40% on both sides of
4780 the active channel along a 100 m long reach. Reductions in effective shade over the stream in
4781 these sites ranged from 4-5%. The analysis considered each reach both individually and
4782 collectively to understand how site and treatment heterogeneity may affect thermal responses at
4783 local and watershed extents. Temperature data were collected before, during, and after treatment
4784 and in the thinned experimental reaches and in adjacent unthinned control reaches with digital
4785 temperature sensors. Temperature data was collected for only 1-year pre-treatment and 1-year
4786 post-treatment. For data analysis, semivariograms of summer degree days were used to
4787 determine the presence of spatial autocorrelation. To control temporal variations in local and
4788 downstream responses summer cumulative degree-days were plotted for pre- and post- treatment
4789 temperatures and along a longitudinal gradient. A Lagrangian framework was used to track
4790 changes in temperature through space and time. Results showed that increases in thermal
4791 heterogeneity occurred in the treatment reaches, in the year following treatment (20° to 139°C),
4792 compared to the pre-treatment year (66° to 112°C). Local changes in stream temperature were
4793 dependent on thinning intensity, with higher levels of canopy cover reduction leading to higher
4794 increases in local stream temperatures. In the reaches with higher reductions in shade (19-30%)
4795 there was accumulation of 45° to 115°C additional degree days from pre- to post treatment years,
4796 while the reaches with lower reductions in shade (4-5%) only accumulated 10° to 15°C
4797 additional degree days. Travel distance of increased stream temperatures also appeared to be
4798 dependent on thinning intensity. The lower shade reduction reaches had an increased temperature
4799 effect downstream with travel distance of 75-150 m, while the high shade reduction sites had a
4800 downstream travel distance of 300- ~1000 m. In the high shade reduction sites, treatment reaches
4801 that were further apart (> 400 m) showed dissipation in increased stream temperatures
4802 downstream, while in parts of the stream where treatments were <400 m apart, temperature

4803 increases did not always dissipate before entering another the next treatment reach. The analyses
4804 with the conceptual framework based on waveforms showed there was no evidence of
4805 cumulative watershed effects at the downstream extent. The authors conclude that their results
4806 show evidence that riparian forest management impacts may extend beyond local stream
4807 environments. Further, the authors propose that riparian forest management that uses a holistic
4808 approach may be more effective in preserving some functions (e.g., shade).

4809

4810 **Sediment**

4811

4812 Wissmar et al., 2004

4813

4814 Wissmar, R.C., Beer, W.N. & Timm, R.K. (2004) Spatially explicit estimates of erosion-risk
4815 indices and variable riparian buffer widths in watersheds. *Aquat. Sci.* 66, 446–455 . DOI:
4816 10.1007/s00027-004-0714-9

4817

4818 The purpose of this study is to use management records, the spatial distribution, and the
4819 variability of different landcover types that can contribute to unstable conditions to develop
4820 erosion-risk indices and variable riparian buffer widths in watersheds of different drainages in
4821 the State of Washington. The objectives of this study were to 1) define erosion risk indices based
4822 on “different land cover types,” 2) evaluate erosion risk indices with sediment inputs into
4823 streams, 3) use erosion risk categories to define locations of stream reaches that are susceptible
4824 to different levels of erosion 4) use categories to identify distribution of channels requiring
4825 variable width buffers for protection 5) Test procedure by applying ground-truthed data from the
4826 upper Cedar River drainage near Seattle, Washington. The land cover types used to assess risk
4827 included unstable soils, immature forests, roads, critical slopes for land failure, and rain-on-snow
4828 events. Based on available data, the researchers developed a map of these land cover features
4829 with sediment input values to define erosion risk indices. The indices were used to categorize the
4830 landscape into 6 levels of erosion risk. Results of the mapped erosion risk categories explained
4831 65% of the variation associated with sediment inputs. The highest-risk areas contained a
4832 combination of all landscape cover factor combinations (rain-on-snow zone, critical failure
4833 slope, unstable soil, immature forests, and roaded areas). The lowest risk categories contained
4834 only rain-on-snow zones, and critical failure slopes. Roaded areas and unstable soils were only
4835 present in risk categories 3-6. This paper shows the importance of investigating multiple factors
4836 when evaluating the controls on sediment discharge and stream inputs. Further, when factors
4837 influencing erosion combine in an area, their effects are compounded.

4838

4839 **Nutrient and forest structure**

4840

4841 Devotta et al., 2021 (removed)

4842

4843 Devotta, D. A., Fraterrigo, J. M., Walsh, P. B., Lowe, S., Sewell, D. K., Schindler, D. E., & Hu,
4844 F. S. (2021). Watershed *Alnus* cover alters N: P stoichiometry and intensifies P limitation in
4845 subarctic streams. *Biogeochemistry*, 153(2), 155-176. DOI:10.1007/s10533-021-00776-w

4846

4847 This study investigates how coverage of alder species affects the aquatic N and P availability
4848 across a natural alder coverage gradient in 26 streams of southwestern Alaska. Alder coverage in
4849 the Alaskan streams was inversely related to elevation (i.e., lower coverage at higher elevations).
4850 To identify the presence of alder as the N and p contributing factor, the researchers analyzed
4851 resin lysimeter samples from select watershed soils supporting variable percent coverages of
4852 alder. Soils supporting alders leached, on average, three times more N and two times more P than
4853 soils not containing alders. The relationship between alder coverage and N and P values was not
4854 linear. Still, the authors identified 30% alder coverage as a transitional threshold from low to
4855 markedly higher soil N and p availability. The higher soil N and P resulted in higher dissolved N
4856 in streams, but the higher soil P under alder coverage did not translate to higher stream P
4857 availability. The authors speculate that soil chemistry or local soil biota may be immobilizing the
4858 soil P from transport into the streams. This led to a high N:P ratio in the spring and summer
4859 stream chemistry of reaches supporting >30% alder coverage. As climate change causes
4860 increasing temperatures, alder may begin to expand its range into higher elevations. This, in turn,
4861 may lead to increased N availability, but higher P limitations in high-elevation montane streams.

4862

4863 **Sediment and lithology**

4864

4865 Fratkin et al., 2020 (removed from focal, scope and results not relevant to review)

4866

4867 Fratkin, M. M., Segura, C., & Bywater-Reyes, S. (2020). The influence of lithology on channel
4868 geometry and bed sediment organization in mountainous hillslope-coupled streams. *Earth
4869 Surface Processes and Landforms*, 45(10), 2365-2379. <https://doi.org/10.1002/esp.4885>

4870

4871 This study compares the differences in channel form patterns, sediment flow, grain size, and
4872 shear stress thresholds between two gravel-bed streams, one on basalt and one on sandstone
4873 parent material in the Oregon Coast Range. Study sites were in a region where widespread
4874 landslides and debris flows occurred in 1996. The researchers compared channel
4875 geomorphologies (e.g., slope, valley width, channel geometry, etc.) to evaluate thresholds and

4876 channel bed adjustments since the 1996 events. The results showed similar sediment coarsening
4877 patterns in the first several kilometers indicating hillslope influence, but downstream fining was
4878 lithology dependent. The authors hypothesized threshold channel conditions in the basalt basin,
4879 and non-threshold conditions in the sandstone basin with a tendency to expose bedrock, based on
4880 the relative competencies (i.e., basalt = high-competency, sandstone = low-competency).
4881 However, results showed evidence of threshold conditions for over 60% of the streams in both
4882 basins. The authors inferred a cycle adjustment to correct the assumed sediment delivery from
4883 the 1996 flood season. The authors speculate that the basalt basins would act as threshold
4884 channels over longer time periods despite a higher debris flow frequency. This paper provides
4885 some evidence that lithologies impose control on channel adjustments driven by different rock
4886 competencies. This difference in rock competency ultimately controls the grain size fining rates
4887 and bed load transport (sediment availability).

4888

4889 **Nutrient and species composition**

4890

4891 Whigham et al., 2017 (removed from focal)

4892

4893 Whigham, D. F., Walker, C. M., Maurer, J., King, R. S., Hauser, W., Baird, S., ... & Neale, P. J.
4894 (2017). Watershed influences the structure and function of riparian wetlands associated with
4895 headwater streams—Kenai Peninsula, Alaska. *Science of the Total Environment*, 599, 124-134.
4896 <https://doi.org/10.1016/j.scitotenv.2017.03.290>

4897

4898 This field study was designed to test the hypothesis that alder cover in watersheds influences the
4899 structure and function of riparian wetlands adjacent to headwater streams. The researchers
4900 compared biomass production, biomass distribution (aboveground vs. belowground),
4901 decomposition rates, and chemical characteristics of interstitial groundwater, between watersheds
4902 with and without alder coverage. Study sites were located on two headwater streams located in
4903 the Kenai Peninsula in south-central Alaska. The results showed that aboveground biomass was
4904 higher in watersheds with alder cover, but the largest differences were in the litter layer and the
4905 belowground biomass. Watersheds without alder had significantly higher belowground root
4906 biomass. The litter overhanging the stream was higher in N content at the alder sites than in the
4907 no-alder sites. The quantity of litter overhanging the stream was higher in the no-alder sites.
4908 Interstitial groundwater was significantly higher in dissolved N at the alder sites. The results of
4909 this study show that species composition within the riparian area can have a considerable effect
4910 on nutrient concentrations which consequently affect stream chemistry, biomass production,
4911 vegetation structure, and decomposition rates.

4912

4913 **LW**

4914

4915 Wing & Skaugset, 2002

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4917 Wing, M. G., & Skaugset, A. (2002). Relationships of channel characteristics, land ownership,
4918 and land use patterns to large woody debris in western Oregon streams. *Canadian Journal of*
4919 *Fisheries and Aquatic Sciences*, 59(5), 796-807. <https://doi.org/10.1139/f02-052>

4920

4921 This study investigated the relationships of land use, land ownership, and channel and habitat
4922 characteristics with LW quantity and volume in 3793 stream reaches in western Oregon State
4923 (west of Cascade crest). This study analyzed an extensive spatial database of aquatic habitat
4924 conditions created for western Oregon using stream habitat classification techniques and a
4925 geographic information system (GIS). The overall objectives of this study were to identify the
4926 database factors most strongly related to LWD abundance and to determine whether ownership
4927 and land use patterns are related to LWD abundance. Regression tree analysis is an exploratory
4928 regression analysis that allows for the inclusion of multiple explanatory variables. LW counts (by
4929 piece, and by key pieces (logs at least 0.60 m in diameter and 10 m long)) and volume were used
4930 as the response variables and explanatory variables included morphology of active channel
4931 (hillslope, terrace, terrace hillslope, unconstrained), lithology (e.g., alluvium, basalt, etc.), Land
4932 use and land cover (e.g., young timber, old timber, rural resident, agriculture, etc.), ownership
4933 (private industrial (PI), private non-industrial (PNI), state, federal (BLM, USFS)), vegetation
4934 type, and other channel characteristics. The analysis was run at the reach scale. Results showed
4935 that the most important predictor for LW volume was land ownership with PNI split from all
4936 other ownership types. Mean LW volumes in stream reaches with PNI ownership were 3.1 m³
4937 while mean volume of LW in reaches in all other ownerships (PI, state, BLM, USFS) were 17.9
4938 m³. However, this was likely because the PNI lands held a disproportionately higher percentage of
4939 unforested lands compared to all other ownership types. When the ownership and land use
4940 variables were removed, stream gradient became the most important explanatory variable for LW
4941 volume. The split for stream gradient occurred for reaches with < 2.3% gradient averaged 5.8 m³
4942 while higher gradient streams averaged 17.9 m³ per reach. When ownership and land use were
4943 included but non-forested lands were removed, stream gradient again was the most important
4944 predictor with the split occurring for stream reaches with gradients less than 4.7% averaging 11.5
4945 m³, which was less than half of the average found at higher gradient reaches (25.2 m³); in this
4946 model the stream gradient split explained 11% of the variation observed of instream LW volume.
4947 For LW pieces in forested stream reaches bankfull channel width was the most important
4948 explanatory variable with the split occurring for streams channels less than 12.2 m wide. LW
4949 pieces for streams <12.2 m wide averaged 11.1 LW pieces per reach while larger channels
4950 averaged 4.9 pieces per reach; in this model the BFW split explained 7% of the variation in LW
4951 pieces found in forested streams. For key LW pieces (logs at least 0.60 m in diameter and 10 m
4952 long) in forested reaches, stream gradient was again the most important explanatory variable
4953 with the split occurring at a slope of 4.9%. The streams with a gradient < 4.9% averaged 0.5 key
4954 LW pieces per reach while streams with higher gradients averaged 0.9 key LW pieces per reach;

4955 in this model stream gradient explained 8% of the variation in key LW pieces found in streams.
4956 For forested streams, lithology caused second, third or fourth level splits after stream gradient or
4957 BFW. In three of these four splits, Mesozoic sedimentary and metamorphic geologies, located in
4958 southern Oregon stream reaches, were grouped and split from basalt, cascade, and marine
4959 sedimentary geologies. In stream reaches in Mesozoic sedimentary and metamorphic geologies,
4960 the quantity of LWD was roughly half the amount found in other geologies. The only exception
4961 to this grouping was for LW volume in larger stream reaches, where basalt and marine
4962 sedimentary geologies were grouped separately from all other geologies in a fourth-level split
4963 and contained more LW volume. The authors conclude that the geomorphic characteristics of
4964 stream reaches, in particular stream gradient and bankfull width, in forested areas correlated best
4965 with LW presence.

4966

4967

4968 **LW and plant communities**

4969

4970 Rot et al., 2000 (removed from focal list)

4971

4972 Rot, B. W., Naiman, R. J., & Bilby, R. E. (2000). Stream channel configuration, landform, and
4973 riparian forest structure in the Cascade Mountains, Washington. *Canadian Journal of Fisheries
4974 and Aquatic Sciences*, 57(4), 699-707. <https://doi.org/10.1139/f00-002>

4975

4976 This study investigates the hierarchical relationships between the “five key elements”, valley
4977 constraint, riparian landform, riparian plant community, channel type, and channel configuration.
4978 for 21 sites in mature old-growth riparian forests of the western Cascade Mountains in
4979 Washington State. The objective of this article is to expand this perspective over several spatial
4980 scales and the temporal life span of a conifer by examining how channel configuration interacts
4981 with valley constraint, streamside landform, channel bedform, and successional processes within
4982 the riparian forest. Stepwise regression was used to examine the relationship between physical
4983 and biological characteristics and the individual elements of channel configuration. Channel
4984 configuration is the channel elements at the habitat unit scale, including channel units (total
4985 number of pool–riffle habitat units per 100 m of channel length), LW pieces (per 100 m of
4986 channel length), LW volume (cubic meters per 100 m of channel length), pool spacing, percent
4987 pools, and percent LW-formed pools. Results showed that significantly more total LW pieces
4988 were found in forced pool–riffle channels than in the bedrock and plane-bed channels (Kruskal–
4989 Wallis, $p < 0.05$). Forced pool–riffle channels averaged 16.4 pieces per 100 m, bedrock 10.8
4990 pieces, and plane-bed 10.1 pieces. The volume of LW (cubic meters per 100 m) followed a
4991 similar trend. The percentage of deep pools (>0.5 m) formed by LW increased with stand age (r^2
4992 = 0.36). LW diameters were significantly smaller for ages 55–220 than for ages 333–727
4993 (Kruskal–Wallis, $p = 0.01$). The authors conclude that scale is an important consideration for

4994 management of aquatic habitat. At the largest spatial scale, results showed valley constraint
4995 significantly influenced off-channel habitat (plant communities associations and landform
4996 categories) and in-stream LW volume within forced pool-riffle channels. At the smallest scale,
4997 channel type (bedrock, plane-bed, and forced pool-riffle) was most closely related to LW
4998 volume, density, and the number of LW-formed pools. The diameter of the in-channel LW
4999 increased with riparian forest stand age. Streams adjacent to old-growth forests in-channel LW
5000 diameter were equivalent to or greater than the average standing riparian tree diameter at all
5001 sites. In younger stands, the relationship of in-stream LW diameter had a mixed relationship with
5002 riparian tree average diameters. The authors speculate this may be due to many in-stream LW
5003 pieces being relics from previous old-growth communities. In this area, four landform classes
5004 differentiated the riparian communities (floodplain, low terrace, high terrace, slope). Most were
5005 dominated by conifers, except the floodplain landforms, which supported a higher density of
5006 deciduous species, but a higher basal area of conifer species. The results of this study provide
5007 more evidence, similar to other studies, that channel geomorphology and valley constraint are
5008 important predictors of LW abundance (quantity and volume) in streams. The novelty in this
5009 study is how the riparian area landforms lead to different riparian plant communities, which
5010 consequently affect the input of LW.

5011

5012 **Nutrients**

5013

5014 Yang et al., 2021

5015

5016 Yang, Y., Hart, S. C., McCorkle, E. P., Stacy, E. M., Barnes, M. E., Hunsaker, C. T., ... & Berhe,
5017 A. A. (2021). Stream water chemistry in mixed-conifer headwater basins: role of water sources,
5018 seasonality, watershed characteristics, and disturbances. *Ecosystems*, 24(8), 1853-1874.
5019 DOI:10.1007/s10021-021-00620-0

5020

5021 This study investigated the effects of drought and forest thinning operations (independently and
5022 combined) on water chemistry from multiple basin water sources (snowmelt, soil solution,
5023 stream water) in the Mediterranean climate headwater basins of the Sierra National Forest. Data
5024 on water chemistry was taken 2 years prior and 3 years following drought and thinning
5025 operations in two watersheds, each with thinned and control stands. This data was analyzed to
5026 answer 3 questions: 1. How does the chemistry of different water sources (that is, snowmelt, soil
5027 solution at two depths, stream water) vary monthly and interannually prior to drought and
5028 thinning? 2. How does drought alone and drought combined with thinning impact water
5029 chemistry? 3. Can watershed characteristics predict stream water chemistry over contrasting
5030 water years? The authors used general linear models to analyze differences in chemistry by water
5031 source, repeated measures analysis of variance for effects of drought and thinning on water
5032 chemistry, and linear regression to predict water chemistry based on watershed characteristics.

5033 Results showed that monthly concentrations of dissolved C and N varied among different water
5034 sources prior to drought and thinning. For dissolved organic carbon (DOC) soil solution at 13 cm
5035 depth (mean \pm SE of $25.97 \pm 2.75 \text{ mg l}^{-1}$, across months for 2 years) had higher monthly
5036 concentrations than soil solution collected at 26 cm depth ($16.93 \pm 1.55 \text{ mg l}^{-1}$). Snowmelt (9.67
5037 $\pm 0.89 \text{ mg l}^{-1}$) and stream water ($5.33 \pm 0.52 \text{ mg l}^{-1}$) had the lowest concentrations. For total
5038 dissolved Nitrogen (TDN) and dissolved organic nitrogen (DON), soil solution at 13 cm depth
5039 (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
5040 $0.92 \pm 0.32 \text{ mg l}^{-1}$), and snowmelt (0.94 ± 0.17 and $0.73 \pm 0.18 \text{ mg l}^{-1}$) had higher
5041 concentrations than stream water (0.11 ± 0.02 and $0.08 \pm 0.01 \text{ mg l}^{-1}$). For dissolved inorganic
5042 nitrogen (DIN), snowmelt ($0.25 \pm 0.05 \text{ mg l}^{-1}$) had the highest concentration followed by the soil
5043 solution at 13 cm depth ($0.06 \pm 0.01 \text{ mg l}^{-1}$). Soil solution at 26 cm depth ($0.03 \pm 0.01 \text{ mg l}^{-1}$)
5044 and stream water had the lowest values ($0.04 \pm 0.01 \text{ mg l}^{-1}$). For pH, snowmelt (pH 6.09 ± 0.06)
5045 was more acidic than soil solutions at both depths (7.52 ± 0.23 at 13 cm depth and 7.79 ± 0.11 at
5046 26 cm depth) and stream water (7.37 ± 0.07). Drought alone altered DOC in stream water, and
5047 DOC:DON in soil solution in unthinned (control) watersheds. Volume-weighted concentration of
5048 DOC was 62% lower ($p < 0.01$) and DOC:DON was 82% lower ($p = 0.004$) in stream water in
5049 years during drought (WY 2013–2015) than in years prior to drought (WY 2009 and 2010).
5050 Drought combined with thinning altered DOC and DIN in stream water, and DON and TDN in
5051 soil solution. For stream water, volume-weighted concentrations of DOC were 66- 94% higher in
5052 thinned watersheds than in control watersheds for all three consecutive drought years following
5053 thinning. No differences in DOC concentrations were found between thinned and control
5054 watersheds before thinning. Watershed characteristics explained inconsistently the variation in
5055 volume-weighted mean annual values of stream water chemistry among different watersheds.
5056 The authors conclude that their results showed evidence that the influences of drought and
5057 thinning are more pronounced for DOC than for N in streams.

5058

5059 **Geology**

5060

5061 Kusnierz and Sivers, 2018 (removed from focal)

5062

5063 Kusnierz, P.C., Sivers, E., 2018. How important is geology in evaluating stream habitat? *J Soils*
5064 *Sediments* 18, 1176–1184. DOI:10.1007/s11368-017-1885-z

5065

5066 The purpose of this study was to assess the importance of considering geology when evaluating
5067 stream habitat conditions. Stream habitat data were collected from 424 sites on federally
5068 managed lands in western Montana, USA. These sites represented a variety of ecoregions, stream
5069 types, management practices, and geologies. The importance of accounting for geology in data
5070 analysis was evaluated using five sediment-related habitat variables and three analyses that
5071 examined (1) differences across geology for the entire dataset and for sites in reference and

5072 managed watersheds; (2) differences between reference and managed sites within geologies; and
5073 (3) the relative strength of geology as a factor when accounting for the effects of management,
5074 stream type, and ecoregion. This objective was pursued by using five sediment-related habitat
5075 variables (Log instability index, Log roughness-corrected index of relative bed stability, Median
5076 substrate size, Percent pool tail fines < 6 mm, Percent stable banks). Five sediment-related
5077 habitat variables were collected from 424 sites on federally managed lands between 2009-
5078 2012. Factorial ANOVA on ranks was performed to evaluate the relative importance of geology
5079 when other factors were taken into account. Results from this study show that differences in
5080 sediment-related habitat variables did not differ significantly according to geology; however,
5081 observed differences were typically drawn from managed sites. The authors conclude by
5082 advising against using geology as the sole means of stratifying habitat data when attempting to
5083 account for between-site variability.

5084

5085 **Stream Temperatures**

5086

5087 Leach et al., 2017 (removed from focal list)

5088

5089 Leach, J.A., Olson, D.H., Anderson, P.D., Eskelson, B.N.I., 2017. Spatial and seasonal variability
5090 of forested headwater stream temperatures in western Oregon, USA. *Aquat Sci* 79, 291–307.
5091 DOI:10.1007/s00027-016-0497-9

5092

5093 This study is a case study of thermal regimes for headwater streams in the Keel Mountain Study
5094 area. This study examined (1) forested headwater stream temperature variability in space and
5095 time; (2) relationships between stream temperature patterns and weather, above-stream canopy
5096 cover, and geomorphic attributes; and (3) the predictive ability of a regional stream temperature
5097 model to account for headwater stream temperature heterogeneity. Stream temperature data was
5098 collected at 48 sites within a 128-ha watershed in western Oregon between 2012 and 2013.
5099 Spatial statistical modeling was used to relate stream temperature patterns to site characteristics
5100 (elevation, stream width, catchment area, slope, aspect, channel substrate, and terrain shading), a
5101 cluster analysis was used to capture the full variability in annual stream temperatures. Results
5102 from this study show considerable variability in stream temperature over relatively small areas,
5103 and between seasons. The greatest spatial variability existed during summer (up to 10 Celsius)
5104 and during cold and dry winter periods (up to 7.5 Celsius). Geomorphic attributes typically used
5105 in stream temperature models were not good predictors of variability at headwater scales.

5106

5107 **Stream Temperatures**

5108

5109 Groom et al., 2011b

5110

5111 Groom, J.D., Dent, L., Madsen, L.J., Fleuret, J.(2011b). Response of western Oregon (USA)
5112 stream temperatures to contemporary forest management. *Forest Ecology and Management* 262,
5113 1618–1629. <https://doi.org/10.1016/j.foreco.2011.07.012>

5114

5115 The objective of this paper was to assess the riparian characteristics that best predict shade, and
5116 to determine the stream temperature changes that result following harvest. This study took place
5117 in the Oregon Coastal Range at 33 sites (15 state-owned and 18 private-owned). The 33 sites
5118 studied were approximately 50-70 years old and predominately composed of Douglas-fir and red
5119 alder. Private sites (n = 18) followed FPA rules whereby the riparian management area (RMA)s
5120 are 15 and 21 m wide on small and medium fish-bearing streams, with a 6 m no-cut zone
5121 immediately adjacent to the stream. Harvesting is allowed in the remaining RMA to a minimum
5122 basal area of 10.0 (small streams) and 22.9 (medium streams) m²/ha. State sites (N = 15)
5123 followed the state management plan whereby a 52 m wide buffer is required for all fish-bearing
5124 streams, with an 8 m no cut buffer immediately adjacent to the stream. Limited harvest is
5125 allowed within 30 m of the stream only to create mature forest conditions. Harvest operations
5126 within this zone must maintain 124 trees per hectare and a 25% Stand Density Index. Additional
5127 tree retentions of 25–111 conifer trees and snags/hectare are required between 30 and 52 m. A
5128 site's control reach was located immediately upstream of its treatment reach. The control reaches
5129 were continuously forested to a perpendicular slope distance of at least 60 m from the average
5130 annual high-water level. Reach lengths varied from 137 m to 1,829 m with means of 276 m and
5131 684 m for the control and treatment reaches, respectively. Temperature recording stations were
5132 located upstream and downstream of both control and treatment sites. Stream temperature data
5133 was summarized to provide daily minimum, maximum, mean, and fluctuation for analysis. The
5134 temperature data was modeled using mixed-effects linear regression. Shade analysis included
5135 trees per hectare, basal area per hectare, vegetation plot blowdown, and tree height. A linear
5136 regression analysis of shade data (n = 33) was performed and compared small-sample AIC values
5137 to determine relative model performance among 8 a priori models. Results showed that average,
5138 minimum, and diel stream temperatures increased on private sites following harvest, suggesting a
5139 relationship between decreased shade derived from buffer width and an increase in stream
5140 temperature. Outputs from the model predicted an increase of ~2 °C for minimum shade
5141 conditions and a decrease of ~ -1 °C for maximum shade conditions. For sites that exhibited an
5142 absolute change of shade > 6% from pre-harvest to post-harvest experienced an increase in
5143 maximum temperatures. Further, the model predicted an increase in stream temperature
5144 proportional to treatment reach length. The authors estimate an increase in maximum and
5145 minimum temperatures of 0.73 and 0.59 °C per km, respectively. Following harvest, maximum
5146 temperatures at private sites increased relative to state sites on average by 0.71 °C. Similarly,
5147 mean temperatures increased by 0.37 °C (0.24 - 0.50), minimum temperatures by 0.13 °C (0.03 -
5148 0.23), and diel fluctuation increased by 0.58 °C (0.41 - 0.75) relative to state sites. A comparison
5149 of within site changes in maximum temperatures pre-harvest to post-harvest showed an overall

5150 increase at private sites, but not all sites behaved the same and some had decreases in maximum
5151 temperatures. The average of maximum state site temperature changes = 0.0 °C (range = -0.89 to
5152 2.27 °C). Observed maximum temperature changes at private sites averaged 0.73 °C (range = -
5153 0.87 to 2.50 °C) and exhibit a greater frequency of post-harvest increases from 0.5 to 2.5 °C
5154 compared to state sites. Private site shade values also appeared to decrease pre-harvest to post-
5155 harvest. Private post-harvest shade values differed from pre-harvest values (mean change in
5156 Shade from 85% to 78%); however, no difference was found for state site shade values pre-
5157 harvest to post-harvest (mean change in Shade from 90% to 89%). They did not find evidence
5158 that shade differed if one or both banks were harvested for private sites although the sample size
5159 for single sided harvests was low. Similarly, private site shade values did not appear to differ
5160 between medium or small streams. Results from this study also show that between 68% and
5161 75% of variability in post-harvest shade may be accounted for by basal area within 30 m of the
5162 stream, tree height, and potentially blowdown. The authors speculate that their results suggest
5163 sites with shorter trees have higher post-harvest shade and this may be due to the negative
5164 correlation between crown ratios and tree heights. Overall, this study shows that buffers managed
5165 by state sites were sufficient at mitigating the effects of upland harvesting on stream temperature.
5166 Increases in stream temperature on private sites were related to decreases in shade, which were
5167 related to decreases in basal area on sites with greater tree heights. The authors suggest that their
5168 results are likely relevant to other high-rainfall low-order Douglas-fir dominated streams in the
5169 Pacific Northwest that are subject to similar harvest practices.

5170

5171 **Litter**

5172

5173 Yeung et al., 2019

5174

5175 Yeung, A. C., Stenroth, K., & Richardson, J. S. (2019). Modelling biophysical controls on stream
5176 organic matter standing stocks under a range of forest harvesting impacts. *Limnologia*, 78,
5177 125714. <https://doi.org/10.1016/j.limno.2019.125714>

5178

5179 This study investigates the relative impact of major biophysical controls (stream temperature,
5180 riparian litterfall, and stream discharge) on in-stream CPOM (coarse particulate organic matter)
5181 quantity across a variety of streamside timber harvest intensities using simulation modeling. The
5182 CPOM model used was developed by Stenroth et al., 2014, for similar stream types and
5183 conditions of coastal rainforest streams of British Columbia. The model was calibrated using
5184 data from multiple published studies from, primarily the Pacific Northwest region, and several
5185 other North American regions, that quantified stream flow, temperature, and CPOM following
5186 different timber harvest intensities within 4 years of harvest. The model used an estimated
5187 response of low, moderate, high, and very high severity timber harvest for litterfall (-10%, -30%,
5188 -50%, -90%), peak flows (+20%, +40%, +100%, +300%), and stream temperature (+1°C, +2°C,

5189 +4°C, +6 °C). These changes in litterfall, peak flow, and stream temperature were modeled and
5190 analyzed individually and cumulatively to estimate their relative and combined effects on in
5191 stream CPOM standing stocks. Results of the model showed that in general the standing stocks
5192 of CPOM decreased under the independent effects of reduced litterfall and elevated peak flows
5193 and increased with higher stream temperatures. Along the gradient of harvest severities, litterfall
5194 reductions on depleting CPOM standing stocks were at least an order of magnitude greater than
5195 those of elevated peak flows. At low severity, litterfall reductions led to a 13.5% reduction of
5196 CPOM stocks while peak flow increases at high severity harvest only led to a 5% reduction in
5197 CPOM stocks. The magnitude of CPOM changes induced by litterfall reductions was
5198 consistently greater than stream temperature increases, but their differences in magnitude became
5199 smaller at higher levels of disturbance severity. For example, at low severity, stream
5200 temperatures only led to an increase on CPOM stocks by 1.1% while litter fall reductions led to a
5201 reduction of CPOM by 13.5%. However, at the high intensity treatment CPOM stocks changed
5202 by -90.24%, and +72.07% for litterfall, and stream temperature respectively. For scenarios
5203 involving perturbations of multiple model drivers (combined effects), the effect size of
5204 disturbance was significantly negative (indicating significantly lower CPOM standing stocks
5205 than in undisturbed conditions) whenever litterfall reductions reached 50% or above (i.e., high
5206 severity). When litterfall reductions were 30% or below, the effect size of disturbance varied with
5207 the relative changes in peak flows and stream temperature. Only the effects of litterfall-
5208 temperature interactions on CPOM standing stocks were significant ($p < 0.001$). The authors
5209 interpret these results as evidence that litterfall reduction from timber harvest was the strongest
5210 control on in-stream CPOM quantity for 4 years post-harvest. Further, the authors propose that
5211 the decreased activity of CPOM consumers caused by increasing stream temperatures may be
5212 enough to offset the loss of litterfall inputs on CPOM stocks. The caveat of this study is that it
5213 did not include LW dynamics in preserving CPOM post-harvest. As other studies have shown,
5214 harvest can increase in-stream LW, and in-stream LW can act as a catchment for CPOM.

5215

5216 **Drought Frequency**

5217

5218 Wise, 2010

5219

5220 Wise, E. K. (2010). Tree ring record of streamflow and drought in the upper Snake River. *Water*
5221 *Resources Research*, 46(11). <https://doi.org/10.1029/2010WR009282>

5222

5223 This study used newly collected tree-ring data augmented with existing chronologies from sites
5224 at three headwater streams in the Snake River Basin to estimate streamflow patterns for the
5225 1600-2005 time-period. The reconstructed chronologies were tested for significant correlations
5226 with streamflow patterns during the 1911-2005 time period prior to extrapolation. Streamflow
5227 patterns derived from instrumental data and from reconstructed chronologies were compared

5228 with other streamflow reconstructions of three other western rivers in similar climates to
5229 examine synchronicity among the rivers and gain insight into possible climatic controls on
5230 drought episodes. The reconstruction model developed for the analysis explained 62% of the
5231 variance in the instrumental record after adjustment for degrees of freedom. Results showed
5232 evidence that droughts of the recent past are not yet as severe, in terms of overall magnitude, as a
5233 30-year extended period of drought discovered in the mid-1600s. However, in terms of number
5234 of individual years of < 60% mean-flow (i.e., low-flow years), the period from 1977-2001 were
5235 the most severe. Considering the frequency of consecutive drought years, the longest (7-year-
5236 droughts), occurred in the early 17th and 18th centuries. However, the 5-year drought period from
5237 2000-2004 was the second driest period over the 415-year period examined. The author explains
5238 that the area has continued to experience a drought period, but its severity could not be
5239 calculated as it hadn't ended by the time of the study (2010). The correlative analysis of the
5240 chronologies developed for the upper Snake River with other rivers of the West (the upper
5241 Colorado, the Sacramento, and the Verde Rivers) showed mixed results with periods of positive
5242 and negative correlations. The author interprets these results as evidence that drought frequency
5243 in general, in this area appears to be increasing in severity and that mean annual flow appears to
5244 be reducing in the latter half of the 20th and the beginning of the 21st century. The exceptions
5245 being the 1930's dustbowl, and an unusually long dry period in the early 1600s.

5246

5247 **Shade and structure**

5248

5249 Warren et al., 2013

5250

5251 Warren, D. R., Keeton, W. S., Bechtold, H. A., & Rosi-Marshall, E. J. (2013). Comparing
5252 streambed light availability and canopy cover in streams with old-growth versus early-mature
5253 riparian forests in western Oregon. *Aquatic sciences*, 75(4), 547-558. DOI:10.1007/s00027-013-
5254 0299-2

5255

5256 This study investigates the differences in canopy cover and streambed light availability between
5257 paired reaches in old-growth (> 500 years old) and secondary-growth (~40-60 years old) riparian
5258 forests on canopy cover and streambed light exposure in four second order fish-bearing streams
5259 in the H.J. Andrews Experimental Forest. Streams were paired based on reach length and
5260 bankfull width and north (n=2), and south (n=2) facing watersheds. The overall mean percentage
5261 of canopy cover was estimated using a convex spherical densiometer every five meters along the
5262 thalweg of each stream reach. At each point densiometer readings were taken from four
5263 directions (upstream, downstream, left bank, right bank) The amount of light reaching the bottom
5264 of the stream was estimated every five meters using fluorescent dye that degrades overtime from
5265 light exposure. Differences in light availability and canopy cover were analyzed separately for
5266 each of the four reaches using a single factor ANOVA. To avoid the inclusion of overlapping

5267 canopy images from adjacent densiometer sampling locations, the canopy cover data from sites
5268 every 15 m (rather than every 5 m) were used in the comparison of canopy cover between the
5269 two age classes along each reach. Linear regression was used to compare values from mean
5270 densiometer readings with mean dye photodegradation site (every 5 meters). To evaluate the
5271 hypothesis that light availability in old-growth forested streams would be more variable than in
5272 second-growth forested streams, the standard deviations of the mean densiometer readings and
5273 mean photodegradation values were compared between old-growth and second-growth forested
5274 streams with an ANOVA. Results showed that the differences in stream light availability and
5275 percent forest cover between old-growth and second-growth reaches were significant in both of
5276 the south-facing watersheds in mid-summer at an alpha of 0.01 for the dye results and 0.10 for
5277 the cover results. For the north-facing watersheds differences in canopy cover and light
5278 availability (alpha = 0.01, and 0.10 respectively) were only significant at 1 of the two reaches.
5279 Overall, three of the four paired old-growth reaches had significantly lower mean percent canopy
5280 cover, and significantly higher mean decline in fluorescent dye concentrations The authors
5281 interpret these results as evidence that old-growth forest canopies were more complex and had
5282 more frequent gaps allowing for more light availability and lower mean canopy cover, on
5283 average, than in adjacent mature second-growth forests.

5284

5285 **LW**

5286

5287 Teply et al., 2007

5288

5289 Teply, M., McGreer, D., Schult, D., & Seymour, P. (2007). Simulating the effects of forest
5290 management on large woody debris in streams in northern Idaho. *Western Journal of Applied*
5291 *Forestry*, 22(2), 81–87. <https://doi.org/10.1093/wjaf/22.2.81>

5292

5293 This paper uses simulation modeling to estimate the effects of timber harvest, under the Idaho
5294 Forest Plan (IFP), on in-stream LW loading for Class I streams (fish-bearing streams) of the
5295 Priest Lake Watershed in northern Idaho relative to unharvested riparian forest streams. Under
5296 the IFP, class one streams have a 25-foot no-cut-buffer that extends out from the high-watermark,
5297 and an additional 50 feet beyond the edge of the no-cut-buffer where harvest requires retention of
5298 88-trees-per-acre that are greater than 8-in diameter at breast height (DBH). This study used the
5299 Riparian Aquatic Interaction Simulator (RAIS) to estimate the potential wood loading for 58
5300 randomly selected north Idaho stream segments with and without harvest. Stream segments were
5301 measured in the field along the stream centerline from the upstream starting point (0 ft) to a
5302 downstream ending point (200 ft). Riparian stand conditions were measured within 75 ft-long by
5303 10-ft-wide strips oriented perpendicular to the stream at 25, 75, 125, and 175 ft downstream of
5304 the upstream starting point on each side of the stream segment to provide a total of eight strips
5305 for each stream segment. Along each strip, live trees and snags greater than 8 in dbh within the

5306 strip were located and measured. Three circular subplots, each 10 ft in diameter, were located
5307 along each 75-foot strip plot at 12.5, 37.5, and 62.5 ft from the stream edge. Within the subplots,
5308 smaller live trees (less than 8-in. dbh) were tallied by 1-in. dbh classes. Instream LW loads were
5309 surveyed along the same 200-ft stream segments located for measuring riparian stand conditions.
5310 Qualifying LW (greater than 4-in diameter and longer than 6.6 ft) occurring within the high-
5311 water mark along the entire extent of the segment was tallied. Observed instream LW loads
5312 ranged from 10 to 710 pieces per 1,000 ft of stream. Stream size measured by bank full width
5313 covered a wide range (1 ft to 190 ft), averaging 32.5 ft (SD = 28.1). The authors determined that
5314 active streambank erosion was uncommon in the study area and did not include it as a LW
5315 recruitment mechanism in their analysis. Simulation was based on a four-step process applied to
5316 each riparian stand: 1) Harvest the stand according to riparian management prescriptions, 2)
5317 Predict stand characteristics using growth and yield simulators, 3) Estimate the number of trees
5318 that fall due to mortality in each time step, 4) Calculate the probability that a tree would deliver
5319 LWD to the stream. The simulation evaluated both a harvest and a no-harvest scenario to predict
5320 mean in-stream LW loads after 30, 60, and 100 years. The results predicted mean LW loads at 30
5321 years for the 58 segments studied were 151.1 pieces per 1,000 ft for the no-harvest scenario (SD
5322 = 76.2) and 145.1 pieces per 1,000 ft for the harvest scenario (SD = 75.6), which were not
5323 significantly different ($P = 0.67$). However, on a pairwise basis, loads predicted for these
5324 segments using the harvest scenario were significantly lower by an average of about 6.0 pieces
5325 per 1,000 ft than those predicted via the no-harvest scenario ($P < 0.001$). Compared to the initial
5326 surveyed LW loads, LW loads at 30 years predicted in the no-harvest scenario decreased by an
5327 average of 19.5 pieces per 1,000 ft, representing a significant ($P < 0.007$) downward shift in the
5328 distribution. Predicted mean LW loads at 60 years were 136.1 pieces per 1,000 ft in the no-
5329 harvest scenario (SD = 49.2) and 128.3 pieces per 1,000 ft under the harvest scenario (SD =
5330 48.3). At 100 years, predicted mean LW loads were 122.5 (SD = 35.4) and 116.7 (SD = 35.8),
5331 respectively. Based on 20-piece LW classes, the frequency distributions of predicted loads
5332 between the scenarios were not significantly different at either time step. However, on a pairwise
5333 basis, predicted loads for the harvest scenario were significantly lower than the no-harvest
5334 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
5335 100 years, respectively. Compared to LW loads predicted at 30 years and 60 years, LWD loads
5336 decreased significantly on a pairwise basis by an average of 15.1 ($P < 0.001$) and 13.6 ($P <$
5337 0.001) at 60 and 100 years, respectively. The authors note that the collective effect of the
5338 assumptions made for the simulation is likely to underestimate the number and variability of LW
5339 pieces recruited and retained in the streams sampled. The authors interpreted these results as
5340 evidence that the IFP prescriptions for class I Idaho streams were sufficient in maintaining LW
5341 recruitment potential.

5342

5343 **Shade**

5344

5345 Swartz et al., 2020

5346

5347 Swartz, A., Roon, D., Reiter, M., & Warren, D. (2020). Stream temperature responses to
5348 experimental riparian canopy gaps along forested headwaters in western Oregon. *Forest Ecology*
5349 *and Management*, 474, 118354. <https://doi.org/10.1016/j.foreco.2020.118354>

5350

5351 This study tested the effects of adding canopy gaps within young, regenerating forests of western
5352 Oregon on stream light availability and stream temperatures. The addition of gaps in the young
5353 regenerating forests were used to theoretically mimic the natural disturbance regimes and the
5354 higher canopy complexity of late-successional forests. The researchers used a before-after-
5355 control-impact design on six replicated streams within the McKenzie River Basin. In the
5356 experimental reaches 30 m gaps were created, centered on a tree next to the stream and at least
5357 30 m in from the beginning of the reach. The study reaches were located on second- and third-
5358 order fish-bearing steep step-pool and cascade dominated headwater streams with boulder
5359 substrate that ranged from 2.2 to 6.4 m in bankfull width and were lined by 40- to 60-year-old
5360 riparian forests. Study sites in each stream encompassed two 120 m reaches with no large
5361 tributary inputs within or between the study reaches, and reference and treatment reaches were
5362 separated by a buffer section of 30–150 m. In each treatment reach, gaps were designed to create
5363 openings in the canopy that were approximately 20 m in diameter. Gaps were centered on a tree
5364 next to the stream at approximately meter 30 along each reach. The gap sizes were intended to
5365 mimic naturally occurring gaps from an individual large tree mortality or small-scale disturbance
5366 events found in these systems which range from 0.05 to 1.0 gap diameter to tree height ratio with
5367 smaller gaps occurring more frequently. Using the Douglas-fir canopy height of 50 m, gaps were
5368 created in the 0.4–1.0 gap diameter to tree height ratio range (approximately 314 m² – 1,963
5369 m²). Actual gap sizes varied across sites from approximately 514 m² to 1,374 m² (0.45 – 0.74
5370 gap ratios) with a mean of 962 m² (mean gap ratio 0.61). Riparian shade was quantified with
5371 hemispherical photos. Light reaching the stream was quantified using photodegradation of
5372 fluorescent dyes placed at 5 m intervals, over a 24 -hour period. Stream temperature was
5373 recorded continuously, at 15-minute intervals, using HOBO sensors to quantify the seven-day
5374 moving average of mean and maximum temperatures. Data was collected for one year pre-
5375 harvest, during harvest year (harvest took place in late fall 2017), and one-year post-harvest. To
5376 determine the effects of experimental canopy gaps on stream light as well as reach responses a
5377 linear mixed-effects model was fit to the data. The results showed that after gaps were cut, the
5378 BACI analysis showed strong evidence for significant increase in mean reach light ($p < 0.01$) to
5379 a mean of 3.91 (SD \pm 1.63) moles of photons m⁻² day⁻¹. overall resulting in a mean change in
5380 light of 2.93 (SD \pm 1.50) moles of photons m⁻² day⁻¹. Mean stream shading could not be
5381 evaluated in the full BACI analysis because post-treatment hemispherical photographs could not
5382 be taken at all sites due to fire impeding access in 2018. For the remaining sites, the areas
5383 beneath each gap had notable localized declines in shade, through the entirety of the treatment
5384 reach mean shading declined by only 4% (SD \pm 0.02%). Overall, the gap treatments did not
5385 change summer T 7DayMax or T 7DayMean significantly across the 6 study sites. The mean
5386 response (change in reach difference before and after the cut) indicated an increase on average
5387 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
5388 °C); however, there was not statistical support of the BACI effect for either metric. The light

5389 response was not correlated with T 7DayMax responses ($r^2 < 0.01$, $p = 0.69$), nor was gap area
5390 ($r^2 = 0.01$, $p = 0.63$), but there was a significant relationship between discharge ($r^2 = 0.73$, $p =$
5391 0.03), and bankfull width ($r^2 = 0.93$, $p < 0.01$) and the T7DayMax response. Wetted width was
5392 also highly correlated with T 7DayMax responses, but the relationship was not as strong with
5393 this stream size metric as with discharge or bankfull width ($r^2 = 0.65$, $p = 0.05$). In contrast to the
5394 summary values, results from the analysis of individual days throughout the full 40-day summer
5395 period identifying differences in the relationships of daily maximums and daily means between
5396 reaches showed a statistically significant effect of the gap for average daily maximums ($p < 0.01$)
5397 and for average daily means ($p = 0.02$). The regression comparison reveals there will be on
5398 average an additional $0.12\text{ }^\circ\text{C}/^\circ\text{C}$ increase in daily maximum temperature in the reach with a gap.
5399 Likewise, for the daily mean, for every degree increase in the shaded reference reach, an average
5400 additional increase of $0.05\text{ }^\circ\text{C}$ in a reach with a small gap is expected. The authors conclude that
5401 adding gaps to young regenerating forests only minimally increases temperatures, dependent on
5402 stream size, and that riparian canopy gaps may be a viable management strategy that can be
5403 implemented with minimal effects on stream temperatures. This paper does not quantify changes
5404 in stream productivity, also expected from the increase in available light.

5405

5406 **Shade**

5407

5408 Sugden et al., 2019

5409

5410 Sugden, B. D., Steiner, R., & Jones, J. E. (2019). Streamside management zone effectiveness for
5411 water temperature control in Western Montana. *International Journal of Forest Engineering*,
5412 30(2), 87-98. <https://doi.org/10.1080/14942119.2019.1571472>

5413

5414 This study investigates the effects of riparian forest timber harvest, under the Montana
5415 Streamside Management Zone (SMZ) laws, on stream temperature in Class 1 streams (fish-
5416 bearing, or flow more than 6 months per year and are connected to downstream waters).
5417 Montana state law requires timber be retained within a minimum of 15.2 m of the class 1
5418 streams, with equipment exclusion zones extended on steep slopes for up to 30.5 m. Within the
5419 SMZ no more than half the trees greater than 204 mm (8 in) diameter at breast height (DBH) can
5420 be removed, and trees retained must be representative of the pre-harvest stand. In no case,
5421 however, can stocking levels of leave trees be reduced to less than 217 trees per hectare. The
5422 objectives of this study were to fill the information gap in this region by: (1) evaluating the
5423 performance of 15.2 m SMZs retained during harvest activities for protecting against adverse
5424 changes in summer maximum stream temperatures, (2) quantifying the level of timber removal
5425 occurring within operational SMZs that may help explain any observed changes, and (3)
5426 Evaluating fish response that may be associated with a stream temperature change. Data for
5427 stream temperature and fish population response was collected for 30 harvest reaches in western

5428 Montana (northern Rocky Mountain Region), for a minimum of one-year pre- and one-year post-
5429 harvest. Data for stream temperatures and fish populations were also collected from unharvested
5430 reaches upstream from the harvest sites as a control. Temperature data was collected
5431 with Optic StowAway™ and StowAway TidBit™ digital temperature loggers manufactured by
5432 Onset Computer Corporation. Shade over the stream surface was not directly measured in this
5433 study. Canopy cover was estimated using a combination of simulation modeling and using a
5434 concave spherical densiometer. Fish populations were estimated for 100 m reaches at study sites
5435 using an electro-fishing pass of capture method. Linear mixed effects models were used to
5436 analyze the relationship between year, stream position, harvest, fish populations and stream
5437 temperatures. The results showed that within harvest areas, the mean basal area (BA) declined
5438 from 30.2 m²/ha pre-harvest to 26.4 m²/ha post-harvest (mean = -13%, range from -32% to
5439 0%). Windthrow further reduced the mean BA to 25.9 m²/ha (mean = -2%, range = -32% -0%).
5440 Changes in mean canopy cover were not significant based on the simulation modeling (-3%), or
5441 densiometer readings (+1%). Results of the model for the effect of harvest on stream
5442 temperature showed no detectable increase in treatment streams relative to control streams. The
5443 estimated mean site level response in maximum weekly maximum temperatures (MWMT)
5444 varied from -2.1 °C to +3.3 °C. Overall, 20 of 30 sites had estimated site level response within
5445 ±0.5 °C. There were five sites that had an estimated site-level response greater than 0.5 °C (i.e.
5446 warming) and five sites that had an estimated site level response less than -0.5 °C (i.e. cooling).
5447 Results for the fish population showed approximately 7% increase in trout population from pre-
5448 harvest to post-harvest, but this difference was not significant. The authors conclude that the
5449 results suggest that Montana's 15.2 m SMZs retained during timber harvest activities are highly
5450 protective (change <0.5°C) of stream temperatures.

5451

5452 **LW**

5453

5454 Sobota et al., 2006

5455

5456 Sobota, D. J., Gregory, S. V., & Sickle, J. V. (2006). Riparian tree fall directionality and
5457 modeling large wood recruitment to streams. *Canadian Journal of Forest Research*, 36(5), 1243–
5458 1254. <https://doi.org/10.1139/x06-022>

5459

5460 The objectives of this study were to evaluate patterns of riparian tree fall directions in diverse
5461 environmental conditions and evaluate correlations with tree characteristics, forest structural
5462 variables, and topographic features. Specifically, the authors were interested in correlations
5463 between fall directionality and tree species type, tree size, riparian forest structure, and valley
5464 topography (side slope). Data was collected from 21 field sites located west of the Cascade
5465 Mountains crest (11 sites: Coast Range and west slopes of the Cascades), and in the interior
5466 Columbia Basin (10 sites: east slopes of the Cascades, Blue Mountains, and Northern Rockies)

5467 of Oregon, Washington, Idaho, and Montana, USA. Streams were second- to fourth-order
5468 channels and had riparian forests that were approximately 40 to >200 years old. The location of
5469 specific study reaches (200–300 m stream length) on each stream were selected randomly.
5470 Minimum size criteria for a fallen tree in this study were diameter at breast height (DBH) of 0.1
5471 m and height of 5 m. All fallen trees up to 50 m slope distance from stream or the first 100 trees
5472 were measured at all sites. Tree fall direction was standardized among sites by streamside
5473 location (upstream = 0° and 360°; toward stream = 90°; downstream = 180°; away from stream =
5474 –90° and 270°). Spearman rank correlations were used to compare site level statistics of tree fall
5475 directions with physical and riparian forest characteristics. Then trees were pooled among sites
5476 and classified by species for analysis of species, tree size, and valley side slope effects. To avoid
5477 small sample sizes species were grouped by side slope categories (< 40%, >40%). Average
5478 direction of tree fall by site was significantly correlated with valley constraint (Spearman $r = -$
5479 0.53 ; $P = 0.02$). Average direction of tree fall by site was weakly correlated with active channel
5480 width, tree stem density, and basal area ($P > 0.05$), with Spearman r coefficients of 0.22, -0.21 ,
5481 and 0.39, respectively. Trees on valley side slopes >40% for each species had a 95% CI that only
5482 included falls directly towards the stream channel; trees on side slopes <40% had a 95% CI for
5483 mean fall direction that included directly upstream, downstream, away from the stream, towards
5484 the stream, or all four directions simultaneously (consistent with random fall directions),
5485 depending on species. Tree size was only different between side slope categories for coastal
5486 Douglas fir on >40% side slopes which had a median DBH 1.2 to 1.9 times greater than trees on
5487 <40% side slopes. Also, red alder trees on side slopes > 40% had a median DBH 1.1 to 1.6 times
5488 greater than on side slopes < 40%. Model projections of LW recruitment calibrated with the
5489 results of the spearman rank correlations estimated that sites with uniform steep side slopes
5490 (>40%) produced between 1.5 (first resolution) to 2.4 (second resolution) times more in stream
5491 LW by number of tree boles than sites with uniform moderate side slopes (< 40%). The authors
5492 interpret their results as evidence that edaphic, topographic, and hydrologic characteristics are
5493 related to greater variability of tree fall directions on moderate slopes than on steep slopes. The
5494 authors conclude that models that use tree fall directions in predictions of LW recruitment should
5495 consider stream valley topography. The authors warn that while side slope categories (>40%,
5496 <40%) was the strongest predictor of tree fall direction in this study, they believe the differences
5497 in tree fall direction between these categories mainly characterized differences between fluvial
5498 (88% of moderate slope sites) and hillslope landforms (71% of steep slope sites). They suggest
5499 that the Implications from this study are most applicable to small- to medium-size streams
5500 (second- to fourth-order) in mountainous regions where sustained large wood recruitment from
5501 riparian forest mortality is the significant management concern.

5502

5503 **LW**

5504

5505 Schuett-Hames & Stewart, 2019a

5506

5507 Schuett-Hames, D., & Stewart, G. (2019a). Post-Harvest Change in Stand Structure, Tree
5508 Mortality and Tree Fall in Eastern Washington Riparian Buffers: Comparison of the Standard
5509 and All Available Shade Rules for the Fish-Bearing Streams in the Mixed Conifer Timber Habitat
5510 Type Under Washington's Forest Practices Habitat Conservation Plan. Cooperative Monitoring
5511 Evaluation and Research Report CMER. Washington State Forest Practices Adaptive
5512 Management Program. Washington Department of Natural Resources, Olympia, WA.

5513

5514 This report is a comparative analysis of the differences in stand structure, tree fall, and LW
5515 recruitment between riparian sites of eastern Washington harvested under the current Standard
5516 Shade Rule (SR), under the All-Available Shade rule (AAS), and unharvested reference sites
5517 (REF). Both shade rules have a 30-ft no-cut buffer (core zone) immediately adjacent to the
5518 stream. The SR prescription allows thinning in the buffer zone 30-75 feet (inner zone) from the
5519 stream while the AAS prescription requires retention of all shade providing trees in this area.
5520 Post-harvest surveys were completed at each site one–two years and five years post-harvest. A
5521 census was done of all standing trees ≥ 4 inches diameter at breast height (DBH) within 75 feet
5522 (horizontal distance) of the channel on both sides of the stream in each treatment and reference
5523 reach. The condition (live or dead), species, canopy class, and DBH were recorded for each tree.
5524 Dead or fallen trees with a decay class of 1 or 2 were classified as post-harvest mortality and a
5525 mortality agent was recorded (e.g. wind, erosion, suppression, fire, insects, disease, and physical
5526 damage). Metrics were calculated separately for regulatory zones defined by horizontal distance
5527 from the channel, including the core zone (0–30 feet) and inner zone (30–75 feet) and the
5528 combined core and inner zone (the full RMZ). Mixed model analysis was used to evaluate
5529 differences in treatment response. Results showed Cumulative wood recruitment from tree fall
5530 over the five-year post-harvest interval was highest in the SR group, lower in the AAS group and
5531 lowest in the REF group. The SR and AAS rates by volume were nearly 300% and 50% higher
5532 than the REF rates, respectively. The mixed model comparisons indicated that the frequency of
5533 wood input from fallen trees was significantly greater in SR group compared to both the REF
5534 and AAS groups ($p < 0.001$), while the difference between REF and AAS groups was not
5535 significant. Over 60% of pieces recruited from AAS and SR fallen trees consisted of stems with
5536 attached rootwads (SWAR), double the proportion in the REF sites. The REF-AAS and REF-SR
5537 differences in recruitment of SWAR pieces were significant ($p < 0.001$). Most recruiting fallen
5538 trees originated in the core zone (76%, 72%, and 64% for the REF, AAS and SR groups,
5539 respectively), while the proportion from the inner zone (30–75 feet from the stream) was ~10%
5540 greater for the SR group compared to the AAS and REF groups. The authors interpret the results
5541 and conclude that harvest of the adjacent stand outside the RMZ appeared to alter the spatial
5542 pattern of wood recruitment from fallen trees, increasing recruitment from trees located farther
5543 from the stream. Recruitment of fallen trees from the inner zone of the AAS and SR sites were
5544 two and four times the rate for the inner zones of the unharvested reference sites due to increased
5545 tree fall from wind disturbance in the buffers after harvest of the adjacent stand, as reported in
5546 other studies. It is important to note that this was a short-term study (5 years). The authors note
5547 that LW recruitment is a process that can change over decadal time scales. Adding that thinning

5548 and post-harvest mortality also reduced the standing stock of trees available for wood
5549 recruitment in the SR and AAS RMZs compared to unharvested REF RMZs.

5550

5551 **Litter and LW**

5552

5553 Six et al., 2022

5554

5555 Six, L. J., Bilby, R. E., Reiter, M., James, P., & Villarin, L. (2022). Effects of current forest
5556 practices on organic matter dynamics in headwater streams at the Trask River watershed,
5557 Oregon. *Trees, Forests and People*, 8, 100233. <https://doi.org/10.1016/j.tfp.2022.100233>

5558

5559 This study investigates the effects of different riparian timber harvest intensities on changes in
5560 canopy cover, and litter input into streams and litter transport downstream. The objective of this
5561 study was to investigate whether differing levels of tree retention adjacent to the channel altered
5562 coarse particulate organic matter (CPOM) delivery, retention, and transport. The authors
5563 hypothesized an inverse relationship between tree removal and litter delivery (i.e., increase in
5564 tree removal adjacent to the channel would result in a reduction of litter delivery). Data was
5565 collected for leaf litter in streamside litter traps, canopy cover percentage using hemispherical
5566 photos in-stream LW, and litter retention in stream flume litter traps pre- and post-treatment at
5567 five watersheds of the Trask River in the northern Oregon Coast range. The experimental design
5568 included three treatment watersheds: clearcut with no leave trees or retention buffer (CC),
5569 clearcut with leave trees (CC w/LT; retention of 5 trees per hectare/2 trees per acre), and clearcut
5570 with 15 m wide retention buffer (CC c/B) and two uncut references (REF 1, and 2) along
5571 headwater streams. Because there were no replication sites for treatments, data was analyzed
5572 using descriptive and graphical summaries of the data (i.e., no quantitative statistical analysis).
5573 Results showed a reduction of canopy cover from 91.4% to 34.4% in the clearcut treatment with
5574 no leave trees, from 89.8% to 76.1% in the clearcut treatment with leave trees, and from 89.5%
5575 to 86.9% in the clearcut treatment with the 15 m retention buffer. Change in canopy cover in the
5576 reference streams was < 1% for both reaches. Post harvest litter delivery decreased for the
5577 clearcut with no leave trees but increased for both the clearcut with leave tree and clear cut with
5578 retention buffer. The number of logjams, the total weight of logjams, and the volume of LW in
5579 streams increased for all treatment sites. The results of this study were consistent with similar
5580 studies and provide supporting evidence that riparian timber harvest can affect litter and LW
5581 delivery into and retention in streams.

5582

5583 **Shade and LW**

5584

5585 Schuett-Hames et al., 2011

5586

5587 Dave Schuett-Hames, Ashley Roorbach, Robert Conrad. 2011. Results of the Westside Type N
5588 Buffer Characteristics, Integrity and Function Study Final Report. Cooperative Monitoring
5589 Evaluation and Research Report, CMER 12-1201. Washington Department of Natural Resources,
5590 Olympia, WA.

5591

5592 This report presents the results from the Washington State Westside Type N Buffer
5593 Characteristics, Integrity and Function (BCIF) study. The purpose of the study was to evaluate
5594 the effects of westside riparian timber harvest prescriptions for Type Np (perennial non-fish-
5595 bearing) streams on resource objectives (riparian stand tree mortality, wood recruitment, channel
5596 debris, shade, and soil disturbance) described in the Forest and Fish Report of 1999. Three
5597 treatment prescriptions were evaluated, 1) clearcut harvest to the edge of the stream (CC) at eight
5598 sites, 50-foot-wide no-cut-buffers (50-ft) at 13 sites, and 56-foot radius circular no-cut-buffer at
5599 the perennial initiation point (PIP) at three sites (not used in statistical analysis due to small
5600 sample sizes). Each treatment site was paired with an uncut reference site as a control. The CC
5601 and 50-ft treatments were compared with treatment sites at three time periods (the first 1-3 years,
5602 years 4-5, and the whole 5-year period). Differences in variable mean values were checked for
5603 statistical significance between treatment and reference streams using non-parametric Mann-
5604 Whitney U tests. Tree fall rates (annual fall rates of live and dead standing stems combined) was
5605 over 8 times and 5 times higher in the 50-foot buffers than in the reference buffers 3 years after
5606 treatment when compared as a percentage of standing trees and as trees/acre/yr, respectively.
5607 These differences were significant for both metrics ($p \leq 0.001$). In the period 4-5 years post
5608 treatment rate of tree uprooting decreased but rate of stem breakage increased in the 50-foot
5609 buffer. For this period only the percentage of broken trees were significantly different (higher)
5610 than what was observed in the reference buffers. Over the entire five-year period, the percentages
5611 of standing trees that were uprooted and broken (as well as the combined total) were
5612 significantly greater in the 50-foot buffer. Wind was the dominant tree fall process, accounting
5613 for nearly 75% of combined fallen trees, 11% fell from other trees falling against them and 1.8%
5614 of fallen trees fell from bank erosion. Differences in mortality followed a similar pattern to tree
5615 fall rates. In the 50-foot buffer sites mortality rates were significantly higher (3.5 times higher)
5616 than in the reference sites for the first three years following harvest. However, in years 4-5
5617 mortality rates increased in the reference buffers after high-intensity storms resulting in non-
5618 significant differences in mortality during this period. The cumulative percentage of live trees
5619 that died over the entire five-year period was 27.3% in the 50-ft buffers compared to 13.6% in
5620 the reference reaches, but the difference was not statistically significant. This was likely because
5621 of the high variability in mortality between sites in the 50-foot buffers. LW recruitment into the
5622 channel after treatment was higher in the 50-ft buffers than in the reference patches during the
5623 first three years after harvest, over 8 times higher in pieces/acre/yr and over 14 times higher in
5624 volume/acre/yr. In years 4-5 after harvest LW recruitment decreased in the 50-ft buffers and
5625 increased in the reference patches, and the number of recruited LW pieces/acre/yr was greater in

5626 the reference patches, although the volume of LW recruited was greater in the 50-ft buffers. For
5627 the entire first 5 years after harvest, the 50-ft buffers recruited about twice the number of LW
5628 pieces recruited in the reference patches, and over 3 times the volume. The CC treatment,
5629 unsurprisingly, had significantly lower LW recruitment following harvest relative to the reference
5630 streams. Mean overhead shade (from trees and tall shrubs) was 13% lower in the 50-ft treatment,
5631 and 77% lower in the CC treatment relative to reference streams. The CC treatment, however,
5632 increased by 25% five years after harvest relative to values recorded 1-year following harvest.
5633 The implications of these results suggest that immediate and direct changes in stand structure,
5634 canopy cover, and LW are most severe for clear-cut treatments, but that the 50-foot buffer
5635 treatment showed an increase in LW and stand mortality, and a decrease in shade over the five-
5636 year period. Limitations of this study were the lack of pre-harvest data and the relatively short
5637 time-period (5-years) in evaluating impacts that may last for several decades.

5638

5639 Schuett-Hames & Stewart, 2019b (BCIF)

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5641 Schuett-Hames, D & Stewart, G. (BCIF), (2019). Changes in stand structure, buffer tree
5642 mortality and riparian-associated functions 10 years after timber harvest adjacent to non-fish-
5643 bearing perennial streams in western Washington. Cooperative Monitoring Evaluation and
5644 Research Report. Washington State Forest Practices Adaptive Management Program. Washington
5645 Department of Natural Resources, Olympia, WA.

5646

5647 This paper presents a 10 -year follow-up to the results of the BCIF report (Schuett-Hames et al.,
5648 2012) that originally presented 5-year post-treatment results. Over the 10-year period stand
5649 mortality in the 50-ft buffer treatment stabilized and showed a cumulative 14.1% reduction in
5650 live basal, while the reference stands showed a 2.7% increase in live basal area. The differences
5651 in these values were not significant. Cumulative LW recruited into stream channel over the 10-
5652 period was double in the 50-ft treatment streams than in the reference streams. However, the
5653 majority of the LW recruited in the 50-ft treatment streams came to rest above the streams,
5654 providing shade but not affecting streamflow, pool formation, or sediment storage. Further, while
5655 the 50-ft buffer treatment provided more LW recruitment in the short-term (10-years), the authors
5656 speculate there is a reduction in future LW recruitment potential given the removal of trees
5657 outside the 50-ft buffer. Canopy cover in the 50-ft treatment streams recovered to similar
5658 percentages as the reference's streams by the end of the 10-year period. The authors speculate
5659 that the 50-ft buffer was better at maintaining resource objectives than the clearcut but propose
5660 that the narrow buffers presented variable increases in mortality (specifically increased
5661 susceptibility to windthrow) and recommend further research before drawing definitive
5662 conclusions.

5663

5664 **Riparian thinning effects on shade, light, and temperature**

5665

5666 Roon et al., 2021a

5667

5668 Roon, D.A., Dunham, J.B., Groom, J.D., 2021. Shade, light, and stream temperature responses to
5669 riparian thinning in second-growth redwood forests of northern California. PLOS ONE 16,
5670 e0246822. <https://doi.org/10.1371/journal.pone.0246822>

5671

5672 The purpose of this study was to evaluate the effects of riparian thinning on shade, light, and
5673 temperature in three watersheds located in second-growth redwood stands in northern California.
5674 The objectives of this study were to evaluate: 1) the effects of experimental riparian thinning
5675 treatments on shade and light conditions; 2) how changes in shade and light associated with
5676 thinning affected stream temperatures at a reach-scale both locally and downstream; 3) how
5677 thermal responses varied seasonally; and 4) how these thermal responses were expressed across
5678 the broader thermal regime to gain a more complete understanding of thinning on stream
5679 temperatures in these watersheds. This study took place between 2016 and 2018 with thinning
5680 treatments applied during 2017 giving 1-year pre-treatment and 1-year of post-treatment data.
5681 Two study sites prescribed treatment on one side of the stream of a 45 m buffer width with a 22.5
5682 m inner zone with 85% canopy retention and a 22.5 m outer zone that retained 70% canopy
5683 cover (Green Diamond Resource Company, Tectah watershed). At the third treatment site
5684 thinning prescriptions included removal of up to 40% of the basal area within the riparian zone
5685 on slopes less than 20% on both sides of the channel along a ~100–150 m reach (Lost Man
5686 watershed, Redwood national park). Control reaches were located upstream from treatment
5687 reaches. Data analysis was conducted separately for each experimental watershed (i.e., 1 Lost
5688 man site, 2 Tectah sites). Stream temperature was collected using digital sensors; solar radiation
5689 was measured using silicon pyranometers; riparian shade was measured using hemispherical
5690 photography. A classical BACI analysis was performed to test the effects of riparian thinning on
5691 shade, light, and stream temperature using linear-effects models. Results for the Tectah
5692 watershed showed a significant reduction in canopy closure by a mean of 18.7%, (95% CI: -21.0,
5693 -16.3) and a significant reduction of effective shade by a mean of 23.0% (-25.8, -20.1) one-year
5694 post treatment. In the Lost man watershed, a non-significant reduction of mean shade by 4.1% (-
5695 8.0, -0.5), and mean canopy closure by 1.9% was observed in 2018. Results for below canopy
5696 light availability showed significant increases by a mean of 33% (27.3, 38.5) in the Tectah
5697 watershed, and non-significant increases in Lost man watershed of 2.5% (-1.6, 5.6) by 2018.
5698 Results for stream temperature changes showed variation seasonally and between watersheds.
5699 The Lost Man watershed showed no significant changes in average daily maximum, maximum
5700 weekly average of the maximum (MWMT), average daily mean, or maximum weekly average of
5701 the mean (MWAT). In the Tectah watershed, MWMT increased during spring by a mean of 1.7°C
5702 (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)
5703 and increased in downstream reaches during spring by a mean of 1.0°C (0.0, 2.0) and summer by
5704 a mean of 1.4°C (0.3, 2.6). Thermal variability of streams in the Tectah watershed were most
5705 pronounced during summer increasing the daily range by a mean of 2.5°C (95% CI:

5706 1.6, 3.4) and variance by a mean of 1.6°C (0.7, 2.5), but also increased during spring (daily
5707 range: 0.5°C; variance: 0.3°C) and fall (daily range: 0.4°C; variance: 0.1°C). Increases in thermal
5708 variability in downstream reaches were limited to summer (daily range: 0.7°C; variance: 0.5°C).
5709 Again, no significant changes in stream and downstream temperature variability were detected in
5710 the Lost Man watershed. In the Techtah watersheds the frequency of days with temperatures
5711 greater than 16°C increased in summer by a mean of 42.9 more days (95% CI: 31.5, 53.8) in
5712 thinned reaches and a mean of 16.3 more days (6.1, 27.4) in downstream reaches. Temperatures
5713 greater than 16°C persisted for a mean duration of 31.1 more consecutive days (21.0, 41.1) in
5714 thinned reaches and 11.6 more consecutive days (3.9, 20.0) in downstream reaches under the
5715 BACI analysis. The authors conclude that responses to the experimental riparian thinning
5716 treatments we evaluated differed greatly depending on treatment intensity. For example, they
5717 interpret their results as evidence that that changes in shade of 5% or less caused minimal
5718 changes in temperature while reductions in shade of 20–30% resulted in much larger increases in
5719 temperature. However, the authors warn that their data only evaluated immediate (1-year-post-
5720 treatment) changes in stream shade and temperatures. Also, the study was conducted in relatively
5721 small (< 10 km²) coastal watersheds and may not apply to larger watersheds of different regions.

5722

5723 **Sediment**

5724

5725 Safeeq et al., 2020

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5727 Safeeq, M., Grant, G.E., Lewis, S.L., Hayes, S.K., 2020. Disentangling effects of forest harvest
5728 on long-term hydrologic and sediment dynamics, western Cascades, Oregon. *Journal of*
5729 *Hydrology* 580, 124259. <https://doi.org/10.1016/j.jhydrol.2019.124259>

5730

5731 The purpose of this study was to separate and investigate the effects of changes in streamflow
5732 and sediment supply due to disturbances (specifically timber harvest), on sediment transport into
5733 streams. Timber harvest affects both streamflow and sediment supply simultaneously. The
5734 researchers used a reverse regression technique to evaluate the relative and absolute importance
5735 of changes in streamflow versus changes in sediment supply on sediment transport. The
5736 technique was applied to long-term data collected from two paired experimental watersheds in
5737 the H.J. Andrews Experimental Forest, Oregon. The two watersheds were paired by size, aspect,
5738 and topography. The treatment watershed was 100% clearcut during the period from 1962-1966,
5739 broadcast burned in 1966, and re-seeded in 1968. Streamflow, and sediment data were taken
5740 intermittently, and after large storm events from 1952 (pre-harvest) through 1988 for suspended
5741 sediment data, and 2016 for sediment bedload. The control watershed was forested, and had no
5742 treatments (e.g., harvest) during the study period. The results that considered the effects of
5743 harvest on streamflow alone showed an increase in annual water yield in the treatment watershed

5744 by 10% (136 mm/year) over the 51-year post-treatment period. There were no significant
5745 changes in precipitation patterns before or after harvest. Further, the patterns of streamflow in the
5746 control watershed showed diverging patterns in streamflow after the harvest period. The authors
5747 state that these patterns strongly suggest that the increase in streamflow in the treatment
5748 watershed was caused by timber harvest. The results for post-treatment sediment yields showed
5749 suspended load declined to pre-treatment levels in the first two decades following treatment,
5750 bedload remained elevated, causing the bedload proportion of the total load to increase through
5751 time. Changes in streamflow alone account for 477 Mg/km² (10%) of the suspended load and
5752 113 Mg/km² (5%) of the bedload over the post-treatment period. Increase in suspended sediment
5753 yield due to increase in sediment supply is 84% of the measured post-treatment total suspended
5754 sediment yield. In terms of bedload, 93% of the total measured bedload yield during the
5755 posttreatment period can be attributed to an increase in sediment supply. The authors interpret
5756 these results as evidence that while streamflow alone can cause a modest increase in sediment
5757 transport, it is negligible compared to the increases in sediment transport following harvest.
5758 Following harvest, changes on streamflow alone was estimated in being responsible for < 10% of
5759 the resulting suspended sediment transported into streams, while the increase in sediment supply
5760 due to harvest disturbance was responsible for >90%. The authors suggest these results provide
5761 evidence for a need to investigate thresholds for specific watershed management regimes to
5762 ameliorate these impacts following harvest, or thinning treatments. Also, the sharp increases in
5763 sediment transport following logging can be confidently attributed to the increase in sediment
5764 supply and delivery to streams due to the ground disturbances associated with logging rather than
5765 increased streamflow.

5766

5767 **Stream Temperature**

5768

5769 Reiter et al., 2020

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5771 Reiter, M., Johnson, S. L., Homyack, J., Jones, J. E., & James, P. L. (2020). Summer stream
5772 temperature changes following forest harvest in the headwaters of the Trask River watershed,
5773 Oregon Coast Range. *Ecohydrology*, 13(3), e2178. <https://doi.org/10.1002/eco.2178>

5774

5775 This paper investigates the effects of different riparian forest harvest treatments on stream
5776 temperature. Stream temperature data was collected from 2006 to 2016 for multiple small (<50
5777 ha), non-fish-bearing headwater stream watersheds in the Trask River Watershed of the
5778 northwestern Oregon Coast range. The experiment followed a BACI design with four treatments,
5779 1) clearcut, no buffer (CC_NB; n = 4), 2) clearcut with 10-m no cut buffer (CC_B; n = 3), 3)
5780 Thinning with 10 m no-cut buffer (TH_B; n = 1), and 4) unharvested, reference streams (REF; n
5781 = 7). Temperature data was collected at 30-minute increments for all streams using continuously
5782 recording thermistors. Harvest operations occurred in the Summer of 2012 giving 6 summers of

5783 pre-treatment and 4 summers of post-treatment data collection. Temperature data was separated
5784 into 5th, 25th, 50th, 75th, and 95th percentiles, with each percentile being treated as independent
5785 response variables in a linear mixed model. Treatments were compared to reference watersheds
5786 to check for significant differences in temperature percentiles. For ecological context, the
5787 researchers also quantified the percentage of summer where temperatures were above 16 and 15
5788 °C, the preferred thermal regime limits for two local amphibian larvae (coastal tailed frog,
5789 coastal giant salamander). Results showed that even the small (10 m buffer; CC_B, TH_B) buffer
5790 was efficient in maintaining similar temperature changes throughout the summers compared to
5791 reference streams. There were no significant changes in the buffered watersheds with
5792 temperature responses in these watersheds ranging from negative values to negative values close
5793 to zero. The treatments with no buffer (CC_NB), however, showed significant increases in
5794 temperature for all percentiles with the greatest increases occurring in the 95th percentile,
5795 showing a mean increase of 3.6 °C (SE = 0.4). For the 5th percentile, the CC_NB also showed a
5796 mean temperature response 1.7°C (SE = 0.3; range from 1.5 - 2.8°C). Temperature changes were
5797 more severe in the CC_NB watersheds with no leave trees (4.2 and 4.4°C), however, this
5798 difference was not analyzed. The percentage of time the post-harvest, no-buffer treatments spent
5799 above the 16 and 15 °C thresholds were 1.3% and 4.7%, respectively. This was an increase from
5800 pre-harvest values that showed no instances of temperatures above 16°C, and only 0.2% of the
5801 recorded time above 15°C. The authors conclude that their evaluation of temperature responses
5802 as potential biologically significant changes adds context to the changes and fluctuations
5803 observed in each harvest design. While significant changes in mean and percentile changes in
5804 temperature were observed, the amount of time spent above critical temperature thresholds for
5805 important amphibian species was minimal.

5806

5807 **SHD, Stream temperature**

5808

5809 Chan et al., 2004 (Removed from focal list, significant results only apply to fauna)

5810

5811 Chan, S.S., Anderson, P.D., Cissel, J., Larsen, L., Thompson, C., 2004. Variable density
5812 management in Riparian Reserves: lessons learned from an operational study in managed forests
5813 of western Oregon, USA. USDA Forest Service. <https://doi.org/10.1016/j.foreco.2013.06.055>

5814

5815 The purpose of this study was to assess the ability of variable retention thinning and riparian
5816 buffers at accelerating late-seral habitat, facilitating rare species management, and maintaining
5817 ecological functions within riparian zones of 40–70-year-old headwater forests in western
5818 Oregon. This study evaluated 13 separate sites each averaging ~ 100 ha whereby 4 buffer width
5819 treatments adjacent to variable retention thinning prescriptions were assessed. Buffer treatments
5820 include: (1) one site potential tree; (2) two-site potential trees; (3) variable buffer width based on
5821 vegetation and/or topographic site factors; (4) streamside buffer of only the first tree whereby

5822 thinning treatments applied up to 6 m of stream. Thinning treatments included: (1) Unthinned
5823 control - 500-750 trees per hectare; (2) High density retention - 70-75% of area thinned to 300
5824 TPH, 25-30% unthinned riparian reserves or leave islands; (3) Moderate density retention - 60-
5825 65% area thinned to 200 TPH, 25-30% unthinned riparian reserves or leave islands with 10%
5826 circular patch openings; (4) Variable density retention - 10% area thinned to 100 TPH, 25-30%
5827 thinned to 200 TPH, 25-30% thinned to 300 TPH, 20-30% unthinned riparian reserves or leave
5828 islands with 10% circular patch openings. Variables measured include stand development
5829 metrics, understory vegetation, microclimate, aquatic ecology, invertebrates, lichens, and
5830 bryophytes. Early findings from this study show that relatively small changes in the riparian
5831 environment are attributed to different residual thinning densities and different buffer widths.
5832 According to the results, the most suitable habitat for many species of fauna is consistently found
5833 within 5 m of the stream. The largest changes in relative humidity in warm and dry summer
5834 conditions occur within 15 m of the stream channel and begin to stabilize at 25 m. In summary,
5835 the early findings of this study indicate the near-stream riparian environment provides critical
5836 functions and habitat for a wide variety of organisms.

5837

5838 **Sediment**

5839

5840 Reiter et al., 2009

5841

5842 Reiter, M., Heffner, J. T., Beech, S., Turner, T., & Bilby, R. E. (2009). Temporal and Spatial
5843 Turbidity Patterns Over 30 Years in a Managed Forest of Western Washington 1. *JAWRA Journal*
5844 *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)
5845 [1688.2009.00323.x](https://doi.org/10.1111/j.1752-1688.2009.00323.x)

5846

5847 This study evaluates the efficacy of the changes in a forest practices plan developed in 1974 to
5848 reduce sediment inputs into streams in the Deschutes River watershed of western Washington. To
5849 test this, the researchers analyzed 30 years of data (1975-2005) on water levels, discharge,
5850 suspended sediment, turbidity, and water and air temperature from four permanent sampling sites
5851 representing a range of basin sizes from small tributary headwaters to the mainstem of the
5852 Deschutes River. In the 1970s roughly 30% of the watershed had been harvested and
5853 approximately 63% of the existing road network had been constructed. Timber harvest continued
5854 until the early 1990s and the road network was completed in the late 1970s but updated to
5855 include culverts and sediment traps in the early 2000s. The researchers used turbidity as a proxy
5856 for suspended sediment correlation and corrected for typical seasonal increases in streamflow.
5857 The results showed a declining trend in turbidity at all permanent sampling sites during the study
5858 period even with active forest management. Following the road construction and harvest
5859 activities of the 1980s turbidity levels continued to decline until the year 2000 when they

5860 returned to pre-logging levels. The authors interpret these results as evidence that management's
5861 increased attention to reducing sediment is responsible for the reduction in sediment transport.

5862

5863 **Effect of debris torrents on shade, vegetation, and stream temperature**

5864

5865 D'Souza et al., 2011

5866

5867 D'Souza, L.E., Reiter, M., Six, L.J., Bilby, R.E., 2011. Response of vegetation, shade and stream
5868 temperature to debris torrents in two western Oregon watersheds. *Forest Ecology and*
5869 *Management* 261, 2157–2167. <https://doi.org/10.1016/j.foreco.2011.03.015>

5870

5871 The purpose of this study was to examine the effects of debris torrents on vegetation, shade, and
5872 stream temperature eight years after an extreme storm-related disturbance. This study examined
5873 two separate managed watersheds which were affected by storm-related debris torrents in 1996.
5874 This study addressed several questions regarding the patterns and rate of vegetation, shade and
5875 water temperature change post-disturbance: (1) What is the relationship between vegetation and
5876 local landform and substrate types along the study streams? (2) Does vegetation composition and
5877 structure, stream shade and water temperature in debris torrented streams differ between the two
5878 watersheds? and (3) How does recovery of stream temperature relate to vegetation and shade
5879 recovery and does this differ through time between watersheds? Data was gathered from
5880 multiple headwater streams following the disturbance in 1996 at 2 managed watersheds: the
5881 Williams River watershed (WRW), and the Calapooia River watershed (CRW). Data for stream
5882 temperature, to analyze stream temperature recovery, was collected immediately following the
5883 disturbance event in 5 streams, 3 at the CRW (2 disturbed; 1 reference), and 3 at the WRW (1
5884 disturbed, 1 reference) and for 8 years through the summer of 2004. Eight years post-disturbance
5885 12 disturbed streams ($n = 6$ for each watershed) were selected for data collection to examine the
5886 relationships between riparian vegetation, shade, and stream temperatures. Data on landform,
5887 substrate, and vegetation (density, species, and seedlings) were collected at each stream. Stream
5888 shade was estimated using hemispherical photographs taken 1 m above the stream center during
5889 summer and winter months and compared using t-tests. Stream temperature data was collected
5890 using continuously recording thermistors. Data were averaged and analyzed using t-tests, chi-
5891 square tests, simple linear regression, Pearson's correlation coefficient, and analysis of
5892 covariance. Results from this study show early successional species red alder and willow species
5893 dominated areas affected by debris torrents. All red alder variables (density, basal area, and
5894 height) showed a significant relationship with vegetation-related shade. Red alder showed a
5895 significantly higher density ($p = 0.0277$) and basal area ($p = 0.0367$) in the WRW sites. While
5896 stem density of red alder was similar in both watersheds, the size of the trees differed suggesting
5897 that colonization and/or growth of red alder in the WRW occurred more rapidly than in the CRW.
5898 However, there was no statistical difference in landforms or site factors between watersheds that

5899 explained these differences. The only correlations found were a negative relationship between
5900 alder density and rock; and a positive relationship between alder basal area and moss suggesting
5901 a relationship between moisture availability and red alder establishment and growth. The authors
5902 note that the WRW sites experienced greater precipitation in the years following disturbance and
5903 may have contributed to the greater growth rates of red alder, but no analysis was conducted.
5904 Total shade was also significantly higher in the WRW ($p = 0.0049$). Mean maximum daily
5905 temperature fluctuations ($p = 0.0483$), and 7-day maximum temperatures ($p=0,0483$) were also
5906 significantly lower in the WRW streams. Mean max daily stream temperatures were lower in the
5907 WRW streams but the difference was not significant ($p = 0.0779$). The authors conclude that
5908 even though the debris torrents resulted in poor soil conditions, the ability of red alder to thrive
5909 in these conditions resulted in rapid recovery of shade and thermal control.

5910

5911 **Stream temperature, shade and climate**

5912

5913 Reiter et al., 2015

5914

5915 Reiter, M., Bilby, R. E., Beech, S., & Heffner, J. (2015). Stream temperature patterns over 35
5916 years in a managed forest of western Washington. *JAWRA Journal of the American Water*
5917 *Resources Association*, 51(5), 1418-1435. <https://doi.org/10.1111/1752-1688.12324>

5918

5919 This study was an analysis of long-term stream temperature data in a western Washington
5920 watershed to evaluate the effects of forest management, before and after implementation of
5921 riparian forest best management practices, and climate change on stream temperatures. Stream
5922 temperature data from four permanent sampling stations in the Deschutes River Watershed.
5923 Stream and air temperature data was analyzed on a monthly basis from 1975-2009. This long-
5924 term dataset allowed for the examination of changes in stream temperature in four basins of
5925 varying size across a period from before stream buffers were implemented, during their
5926 implementation, and several instances of buffer expansion. Because the study period covered
5927 such a long time the changes in stream temperature based on climate change needed to be
5928 accounted for as well. The recovery of shade was estimated using the shade recovery function
5929 developed by R. Summers of Oregon State University (1983), whereby stream shade is estimated
5930 by angular canopy density (ACD) as a function of the age of stream-adjacent harvest units. To
5931 detect correlations of stream and air temperature change with land management activity
5932 separately from climate changes the data was fit to a model that included the effects of climate.
5933 The researchers accomplished this with a technique for deriving the residuals between stream
5934 temperature and climate called locally weighted scatterplot smoothing (LOWESS). The four
5935 watersheds varied in size from small (2 sites: Hard Creek, 2.4 km²; Ware Creek, 2.9 km²),
5936 medium (1 site: Thurston Creek, 9.3 km²), and large (1 site: The Deschutes River Station, 150
5937 km²). In the 1970s nominal buffer widths were required along fish-bearing streams, which

5938 expanded in the 1980s (requirements not listed), again in the mid-1990s to 23 m, and again to 30
5939 m in 2001. Methods for stream temperature data collection varied at different periods resulting in
5940 a margin of error for monthly temperatures of 0.14°C for 1975 - 1983, 0.09°C for 1984 – 1999,
5941 and 0.02°C. for 2000 – 2009. Because these margins of error were smaller than what the authors
5942 expected from climate and management, they were not accounted for in confidence intervals and
5943 p-values. The results for air temperature changes showed a statistically significant ($p \leq 0.05$)
5944 increasing trend in regional air temperatures for July TMAX_AIR and June and July
5945 TMIN_AIR. The trend for TMAX_AIR for July resulted in a trend magnitude of +0.07°C per
5946 year, for a total increase of 2.45°C over the 35-year record. For minimum air temperatures the
5947 magnitude of the June trend was +0.03°C per year while July TMIN_AIR had a trend magnitude
5948 of +0.04°C per year. The resulting increases in minimum temperatures for the period of record
5949 are 1.05°C and 1.40°C for June and July TMIN_AIR, respectively. Results for trends in stream
5950 temperature over the 35-year study period without adjustment for climate change showed no
5951 statistically significant trend in water temperature changes for the large watershed, while the
5952 medium watershed (Thurston Creek) showed decreasing trends in TMAX_WAT for June, July,
5953 and August, ranging in magnitude from 0.05°C (August) to 0.08°C (July) per year. For the
5954 smaller watershed, Hard Creek (Ware Creek was not included in this analysis), had significant
5955 decreasing trends in TMAX_WAT for July, August, and September. The magnitude of these
5956 trends was yearly decreases of TMAX_WAT by 0.05, 0.08, and 0.05°C, for July, August, and
5957 September, respectively. Significant changes in trends for TMIN_WAT were only found for the
5958 large basin site with yearly increases of 0.04, 0.03, and 0.04°C for July, August, and September,
5959 respectively. Results for stream temperature trends after adjusting for changes in air temperature
5960 (climate) showed significant decreasing trends in TMAX_WAT for the large basin by 0.04, 0.03,
5961 and 0.04°C yearly, for July, August, and September, respectively. For the medium basin, trends
5962 showed yearly decreases in TMAX_WAT of 0.07, 0.08, 0.06, and 0.03 for June, July, August,
5963 and September, respectively. For the small basin, climate adjusted trends in TMAX_WAT
5964 showed significant decreases in yearly trends by 0.05, 0.08, and 0.05 for July, August, and
5965 September, respectively. When stream temperature was examined with its correlation with
5966 estimated annual shade recovery from initial harvest (indexed by ACD). Significant correlations
5967 were found for monthly temperature metrics that were adjusted for climate, for all basins. The
5968 strongest correlations were for the smallest basin (Ware Creek) with correlation coefficients for
5969 climate adjusted maximum water temperatures (CTMAX_WAT) with ACD valuing -0.66, -.078,
5970 -0.65, and - 0.69 for June, July, August, and September, respectively. Correlation coefficients for
5971 Ware Creek CTMIN_WAT with ACD were -0.46, -0.64, -0.71, and -0.52 for June July, August,
5972 and September respectively. The largest basin (The Deschutes River) only showed significant
5973 correlations of CTMAX_WAT with ACD with July (-0.39) and August (-0.25); and only showed
5974 significant correlations of CTMIN_WAT with ACD for the months of August (+0.27), and
5975 September (+0.37). The authors interpret their results as evidence that following canopy
5976 recovery after implementation of riparian harvest rules the larger mainstem of the Deschutes
5977 River decreased in average maximum temperatures by approximately 1.3 °C when accounting for
5978 climate driven changes. The effects of canopy closure cooling were even more dramatic in the
5979 smaller headwater streams by 2.67 and 1.6 °C during the study period when accounting for
5980 climate driven changes (this includes a 0.5 °C correction based on climate warming). However,

5981 following re-initiation of timber harvest in 2001 for the area, when riparian protection buffers of
5982 30 m minimum were required, there was no detectable change in stream temperatures. The
5983 authors conclude that the results of this study show evidence that implementation of protection
5984 buffers in this area were sufficient in maintaining stream temperatures. Conversely, this study
5985 also shows evidence that despite these protections from land management induced stream
5986 temperature changes, these protections have been somewhat offset by the warming climate
5987 conditions.

5988

5989 **Overstory structure effects on understory light and vegetation**

5990

5991 Giesbrecht et al., 2017 (removed from focal, not relevant to questions, essentially a case study)

5992

5993 Giesbrecht, I.J.W., Saunders, S.C., MacKinnon, A., Lertzman, K.P., 2017. Overstory structure
5994 drives fine-scale coupling of understory light and vegetation in two temperate rainforest
5995 floodplains. *Can. J. For. Res.* 47, 1244–1256. [dx.doi.org/10.1139/cjfr-2016-0466](https://doi.org/10.1139/cjfr-2016-0466)

5996

5997 The purpose of this paper was to characterize the overstory structure and understory light
5998 regimes of temperate rainforest floodplains, and to assess the role of light and other site variables
5999 in driving stand vegetation patterns and processes. This study took place along two 1-ha coastal
6000 BC, Canada floodplain sites. These sites were selected as representative examples of floodplain
6001 forests in the Coastal Temperate Rainforest (CTR) as part of a larger network of long-term, old-
6002 growth monitoring plots. These sites were in the submontane variant of the very wet maritime
6003 subzone of the Coastal Western Hemlock zone (CWHvm1) of the B.C. coast. In each stand, the
6004 largest overstory trees are *Picea sitchensis* (Bong.) Carr., with several individuals taller than 60 m
6005 in height (maximum of 62 to 93 m). Based on coring a sample of main canopy trees, stand age at
6006 Kitlope is at least 95 years. Stand age at Carmanah is at least 350 years, based on a core from a
6007 50 m tall *P. sitchensis*. All trees ≥ 5 cm were measured along with all understory vegetation
6008 within 25 2m x 2m subplots. Stand characteristics were recorded as well as information on gap
6009 origins. Hemispheric canopy photographs were taken to estimate understory light penetration.
6010 Visual estimations of organic material, mineral layer, CWD, and other substrates were taken in
6011 each vegetation subplot. Relationships among measures of light transmission, vegetation
6012 structure, and diversity were analyzed with linear correlation analysis. Nonmetric
6013 multidimensional scaling was used to describe variation in species composition on multivariate
6014 axes. Results from this study show both sites as having a relatively high degree of canopy
6015 openness (11-11.6%) and light transmission (median 18% full sun) compared to many other
6016 tropical and temperate forests. Light transmission at both sites is however significantly lower
6017 than a number of old-growth sites in Quebec and northern BC. The origins of canopy openness
6018 and stand shade differ between both sites indicating distinct stand processes and different stages
6019 of stand development. Further, light levels vary substantially within short distances at each site

6020 reflecting a complex overstory structure. Although results from this study are reflective
6021 specifically of the coastal temperate rainforests of BC, the descriptive assessment of these two
6022 separate floodplain forests reveal a natural disturbance history which fostered a high degree of
6023 canopy openness and structural heterogeneity which may ultimately aid in informing future
6024 temperate rainforest floodplain restoration efforts.

6025

6026 **LW**

6027

6028 Reid & Hassan, 2020

6029

6030 Reid, D. A., & Hassan, M. A. (2020). Response of in-stream wood to riparian timber harvesting:
6031 Field observations and long-term projections. *Water Resources Research*, 56(8),
6032 e2020WR027077. <https://doi.org/10.1029/2020WR027077>

6033

6034 This paper proposes a conceptual model of wood storage response to different harvesting
6035 intensities. The model predicts how LW in streams is expected to change spatially and
6036 temporally following three different harvest patterns. The model was developed with 45 years of
6037 LW data retrieved from the Pacific coastal region of Vancouver Island, British Columbia. The
6038 Carnation Creek watershed, which supports gravel bed forested streams, contains riparian forests
6039 that have received a wide range of harvest plans implemented. During logging in the 1970s and
6040 '80s riparian forests of one region were harvested with buffer widths ranging from 1 – 70 meters
6041 in upstream reaches, and another region with near complete or complete removal of vegetation to
6042 the streams edge in downstream reaches. In-stream wood volume and characteristics data has
6043 been collected in eight of these study reaches since 1973 (pre-harvest). The researchers used this
6044 data with simulation modelling to develop a reach-scale wood budget model that predicts wood
6045 loss and recover patterns for 300 years (1900-2200). This paper has two objectives: (i) to use this
6046 field data and modeling approach to examine LW storage changes, the time to minimum wood
6047 load, and wood load recovery times as a result of riparian timber harvesting and forest
6048 regeneration, and (ii) to describe the characteristics of in stream wood, with particular focus to
6049 spatial and temporal patterns in wood storage over the multidecade scale following harvesting in
6050 riparian areas. The model was based upon the proposed response outlined by Murphy and Koski
6051 (1989). Wood budget responses were estimated using three management scenarios. Scenario 1 is
6052 a no harvest scenario, in this configuration, the loss of wood supply from the landscape has little
6053 to no impact on input from wood mortality or bank erosion, and therefore in-stream storage,
6054 decay, and transport of wood is not affected. Scenario 2 represents partial loss of forested area in
6055 the riparian zone, which will lead to a near-immediate reduction in wood recruitment to the
6056 channel from mortality and bank erosion along harvested areas. Wood decay and other
6057 components of wood loss will exceed rates of input, leading to a reduction in storage until time
6058 T_{min} , the point where wood recruitment equals losses as the forest regrows in riparian areas and

6059 the greatest overall reduction in storage has occurred (ΔS_{max}). Wood storage increases
6060 thereafter, eventually recovering to preharvest levels after time T_{rec} . Scenario 3 represents an
6061 intensive harvest scenario where most of the riparian area has undergone harvesting over a short
6062 period of time, a major reduction of input from bank erosion and mortality occurs. This greater
6063 reduction leads to a much larger ΔS_{max} than in Figure 1b as wood losses exceed recruitment.
6064 However, as the dominant wood sources recover at the same rate, the time to T_{min} and T_{rec} is
6065 similar under both the moderate and intensive harvest scenarios. Results of the model show
6066 evidence that wood storage in streams of harvested reaches, hits its minimum value in 50 years
6067 or more following loss of LW input, decay, and export of current stock. Recovery of LW volume
6068 in-streams following harvest is estimated to take approximately 150-200 years. The pattern and
6069 intensity of the harvesting operation had little effect on LW loss and recovery times but did affect
6070 the estimated magnitude of LW volume loss in the first 50 – 80 years. These results show
6071 evidence that timber harvest has a long-term effect on LW storage and loading dynamics even
6072 with protective buffers. However, buffers can ameliorate the magnitude of LW loss during the
6073 recovery period. The one caveat of this model is it doesn't account for as much variability on
6074 stream configuration or valley morphologies that are likely to affect LW storage.

6075

6076 **Buffers and LW Recruitment**

6077

6078 Grizzel et al., 2000 (Removed)

6079

6080 Grizzel, J., McGowan, M., Smith, D., Beechie, T., 2000. STREAMSIDE BUFFERS AND
6081 LARGE WOODY DEBRIS RECRUITMENT: EVALUATING THE EFFECTIVENESS OF
6082 WATERSHED ANALYSIS PRESCRIPTIONS IN THE NORTH CASCADES REGION
6083 (Timber/Fish/Wildlife Monitoring Advisory Group and the Northwest Indian Fisheries
6084 Commission). fp_tfw_mag1_00_003

6085

6086 This study analyzed the effectiveness of the Washington Watershed Analysis (WWA)
6087 prescriptions at recruiting large woody debris. This study took place at 10 riparian sites
6088 distributed across 5 watershed administrative units in the Northern Cascades of Washington. Ten
6089 sites were randomly chosen with gradients and buffer width classes in compliance with WWA
6090 indices. To analyze WWA effectiveness, debris frequency and size at each site were compared to
6091 targets derived from WWA. In addition, debris recruitment was compared between three buffer
6092 width classes. Geometric mean diameter and geometric mean length of debris was calculated
6093 based on measurements of midpoint diameter and total lengths. This data was then compared to
6094 targets derived from a channel width-dependent regression. Results show post-harvest mortality
6095 substantially decreasing stand density at several sites. In stream frequency targets were met at
6096 most sites; however, debris categorized as "good" for habitat was only achieved at four out of ten
6097 sites. At the time of data collection, a large portion of debris recruited from buffers was either

6098 above or outside the bankfull flow zone. The authors point out that the degree to which the debris
6099 will influence fluvial processes in the future will depend on whether or not they are recruited into
6100 the stream and will also depend on the size and state of decay. The size of debris recruited from
6101 buffers was significantly smaller than recruited from unmanaged old-growth stands.
6102 Interestingly, data shows recruitment occurring from the outermost margins of the widest buffers
6103 (20-30 m, >30 m), suggesting narrow buffers may limit recruitment. The authors point out that
6104 the large degree of variability in recruitment from site to site suggests windthrow as an important
6105 causal factor. In channels oriented perpendicular to damaging winds (east-west), there was a
6106 higher likelihood of potential recruitment as compared to channels oriented parallel to damaging
6107 winds. The authors conclude with multiple recommendations for future study. First, they suggest
6108 integrating habitat inventory with recruitment to achieve a better understanding of relationships.
6109 Second, they suggest future study into the fate of debris suspended above channels given much
6110 of our current understanding is based on assumptions of decay and breakage. Finally, they
6111 recommend study into factors influencing windthrow in riparian buffers.

6112

6113 **Sediment**

6114

6115 Rachels et al., 2020

6116

6117 Rachels, A. A., Bladon, K. D., Bywater-Reyes, S., & Hatten, J. A. (2020). Quantifying effects of
6118 forest harvesting on sources of suspended sediment to an Oregon Coast Range headwater stream.
6119 *Forest Ecology and Management*, 466, 118123. <https://doi.org/10.1016/j.foreco.2020.118123>

6120

6121 This study uses sediment source fingerprinting techniques to quantify the proportional
6122 relationship of sediment sources (hillslope, roads, streambanks) in harvested and un-harvested
6123 watersheds of the Oregon Coast Range. The researchers used sediment traps, and chemical
6124 analysis to estimate the origin of suspended sediment in the stream and to quantify magnitude of
6125 sediment stored in protection buffers. The study included one catchment (Enos Creek) that was
6126 partially clearcut harvested in the summer of 2016 and an unharvested reference catchment
6127 (Scheele Creek) located ~3.5 km northwest of Enos Creek. The paired watersheds had similar
6128 road networks, drainage areas, lithologies and topographies. The treatment watershed was
6129 harvested with a skyline buffer technique in the summer of 2016 under the Oregon Forest
6130 practices Act policy that requires a minimum 15 m no-cut buffer. The proportion of suspended
6131 sediment sources were similar in the harvested ($90.3 \pm 3.4\%$ from stream bank; $7.1 \pm 3.1\%$ from
6132 hillslope) and unharvest ($93.1 \pm 1.8\%$ from streambank; $6.9 \pm 1.8\%$ from hillslope) watersheds.
6133 However, the harvested watershed contained a small portion of sediment from roads ($3.6 \pm$
6134 3.6%), while the unharvested reference watershed suspended sediment contained no sediment
6135 sourced from roads. In the harvested watersheds the sediment mass eroded from the general
6136 harvest areas (96.5 ± 57.0 g) was approximately 10 times greater than the amount trapped in the

6137 riparian buffer (9.1 ± 1.9 g), and 4.6 times greater than the amount of sediment collected from
6138 the unharvested hillslope (21.0 ± 3.3 g). These results suggest that the riparian buffer was
6139 efficient in reducing sediment erosion relative to the harvested area. The caveat of this study was
6140 the limited sample size (1 treatment, 1 paired reference watershed) and does not incorporate the
6141 effects of different watershed physiography on sediment erosion.

6142

6143 **SED**

6144

6145 Puntteney-Desmond et al., 2020

6146

6147 Puntteney-Desmond, K. C., Bladon, K. D., & Silins, U. (2020). Runoff and sediment production
6148 from harvested hillslopes and the riparian area during high intensity rainfall events. *Journal of*
6149 *Hydrology*, 582, 124452. <https://doi.org/10.1016/j.jhydrol.2019.124452>

6150

6151 This study uses simulation modeling to evaluate the differences in run-off rates, sediment
6152 concentrations, and sediment yields between watershed harvested areas, along the interface of
6153 harvested areas and riparian buffers, and within riparian buffers during periods of high-intensity
6154 rainfall events. The model simulations were calibrated with soil and watershed characteristic data
6155 collected from the Star Creek catchment located in southeastern Alberta. Fifteen plots were
6156 selected for rainfall simulations along three transects on a north facing hillslope (aspect: $\sim 358^\circ$)
6157 and along two transects on a southeast facing hillslope (aspect: $\sim 129^\circ$). Each transect consisted
6158 of three plots that were spaced ~ 20 m apart along the planar hillslopes. Each plot was one
6159 square-meter, which was bounded by a three-sided steel frame that was inserted into the soil with
6160 the open side facing down the slope. The plots were located either (a) within the general harvest
6161 area, (b) along the edge of the riparian buffer at the interface with the harvested area, or (c)
6162 within the riparian buffer. The high-intensity rainfall events were calibrated to mimic 100-year,
6163 or greater, storm events of the Northern Rocky Mountains (1-hour high intensity rainfall). The
6164 results showed runoff rates and surface and shallow subsurface were greatest in the buffer areas
6165 than in the harvested areas or in the harvest-buffer interfaces especially during dry conditions.
6166 During the dry condition rainfall simulations, the general pattern of runoff rates (surface/shallow
6167 subsurface flow) was riparian buffer (175.6 ± 17.3 [SE] ml min^{-1}) > harvest-riparian edge
6168 (125.8 ± 18.2 ml min^{-1}) > general harvest area (37.2 ± 8.5 ml min^{-1}). Mean runoff rates within
6169 the riparian buffer plots were greater than within the general harvest area plots ($t = 2.90$, $p = .03$).
6170 Runoff ratios were only statistically greater in the riparian buffer plots ($13.9 \pm 3.1\%$) relative to
6171 the general harvest area ($2.9 \pm 1.5\%$) during the dry conditions. All runoff ratios declined during
6172 the wet condition rainfall simulations relative to the dry condition simulations with no evidence
6173 for differences between any of the plot positions ($p > .27$ for all pairwise comparisons). During
6174 the dry condition rainfall simulations, the general patterns of sediment concentrations and
6175 sediment yields were opposite of the runoff rates, with the general harvest area > harvest-riparian

6176 edge > riparian buffer. The sediment concentration was (a) 424.8 mg l⁻¹ (151.0–1195.3 mg l⁻¹)
6177 in the general harvest area, (b) 100.9 mg l⁻¹ (45.8–222.1 mg l⁻¹) along the harvest riparian
6178 edge, and (c) 26.9 mg l⁻¹ (12.2–59.1 mg l⁻¹) in the riparian buffer. Statistically, there was
6179 strong evidence for differences in sediment concentrations between the general harvest area and
6180 along the harvest-riparian edge ($t = 3.21$, $p = .01$) and between the harvest area and the riparian
6181 buffer ($t = 6.17$, $p < .001$). Statistically, there was no evidence for differences in sediment yields
6182 between any of the plot positions. Sediment concentration among plot positions remained the
6183 same during the wet rainfall simulations as the dry rainfall simulations—general harvest area >
6184 harvest-riparian edge > riparian buffer. The geometric mean and 95% confidence intervals (back-
6185 transformed) for the sediment concentration was (a) 285.7 mg l⁻¹ (67.9–1201.5 mg l⁻¹) in the
6186 general harvest area, (b) 79.6 mg l⁻¹ (36.5–173.5 mg l⁻¹) along the harvest-riparian edge, and
6187 (c) 22.3 mg l⁻¹ (3.5–141.7 mg l⁻¹) in the riparian buffer. However, while sediment
6188 concentrations differed most strongly between the general harvest area and the riparian buffer (t
6189 $= 3.51$, $p = .01$), other pairwise comparisons were not significant ($p > .20$). Statistically, there
6190 was no evidence for differences in sediment yields between any of the plot positions for rainfall
6191 simulations during wet conditions. The authors speculate this was likely due to the greater soil
6192 porosity in the disturbed, harvested areas. Sediment concentration in the runoff, however, was
6193 approximately 15.8 times higher for the harvested area than in the riparian buffer, and 4.2 times
6194 greater than in the harvest-buffer interface. Total sediment yields from the harvested area (runoff
6195 + sediment concentration) were approximately 2 times greater than in the buffer areas, and 1.2
6196 times greater in the harvest-buffer interface (however, these proportions were not statistically
6197 different). Replication of the model showed high levels of variability in total run off rate,
6198 sediment concentrations, and sediment yields but the relationships between timing and relative
6199 magnitudes between the three experimental areas were consistent. The authors speculate that
6200 these results will become more relevant as climate change is expected to increase the frequency
6201 of high-intensity rainfall events following dry periods in this area. They suggest expanding
6202 similar methods to understand these effects in areas of different hydro-climatic settings.

6203

6204 **Stream Temperature**

6205

6206 Pollock et al., 2009

6207

6208 Pollock, M. M., Beechie, T. J., Liermann, M., & Bigley, R. E. (2009). Stream temperature
6209 relationships to forest harvest in western Washington 1. *JAWRA Journal of the American Water*
6210 *Resources Association*, 45(1), 141-156. <https://doi.org/10.1111/j.1752-1688.2008.00266.x>

6211

6212 This study investigates the effect of watershed harvest percentage, and time since harvest on
6213 summer stream temperatures at different scales in the Olympic Peninsula, Washington. The
6214 researchers examined recorded stream temperature data in 40 small watersheds that experienced

6215 a range of harvest from 0 – 100% (7 unharvested, 33 harvested between 25-100%), with
6216 regrowth age groups binned for analysis as recently clear cut (< 20 years old) and less recently
6217 clearcut (mostly < 40 years old). Unharvested sites were estimated as being >150-years old.
6218 Clearcut is defined in this paper as removing any protective canopy cover for streams. This study
6219 tested 3 hypotheses: (1) the condition of the riparian forest immediately upstream of a site
6220 primarily controls stream temperature, (2) the condition of the entire riparian forest network
6221 affects stream temperature, and (3) the forest condition of the entire basin affects stream
6222 temperature. These hypotheses were test by examining correlations of stream temperature with
6223 the condition of the immediate upstream riparian forest, or more correlated with forest conditions
6224 more spatially distant and on a coarser scale, such as the entire upstream riparian forest network
6225 or the forest condition of the entire basin. To avoid site effects in their analysis sites were chosen
6226 from a narrow range of subbasin sizes (approximately 1-10 km²) and elevation (75-400 m).
6227 Further, all sites were underlain by sedimentary rock and had perennial flow. Each hypothesis
6228 was tested with linear regression to evaluate the correlations of each age group at each scale with
6229 stream temperature data. The researchers also used AIC value comparisons for model selection to
6230 assess the correlation of other physiographic features (elevation, basin area, aspect, slope, or
6231 geologic composition) with stream temperatures. Results of general temperature patterns showed
6232 that average daily maximum (ADM) were strongly correlated with average diurnal fluctuations
6233 ($r^2 = 0.87$, $p < 0.001$, $n = 40$), indicating that cool streams also had more stable temperatures. For
6234 basin-level harvest effects on stream temperatures. The percentage of the basin harvested
6235 explained 39% of the variation in the ADM among subbasins ($r^2 = 0.39$, $p < 0.001$, $n = 40$) and
6236 32% of variation in the average daily range (ADR) ($r^2 = 0.32$, $p < 0.001$, $n = 40$). The median
6237 ADM for the unharvested subbasins was 12.8 °C (mean = 12.1 °C), which was significantly
6238 lower than 14.5 °C, the median (and average) ADM for the harvested subbasins ($p < 0.001$).
6239 Likewise, the median (and average) ADR for the unharvested subbasins was 0.9 °C, which was
6240 significantly lower than 1.6 °C, the median ADR (average = 1.7 °C) for the harvested subbasins
6241 ($p < 0.001$). Results for the correlations between the riparian network scale forest harvest and
6242 stream temperature showed that the total percentage of the riparian forest network upstream of
6243 temperature loggers harvested explained 33% of the variation in the ADM among subbasins ($r^2 =$
6244 0.33 , $p < 0.001$, $n = 40$) and 20% of variation in the ADR ($r^2 = 0.20$, $p = 0.003$, $n = 40$).
6245 However, the total percentage of upstream riparian forest harvested within the last 20 years was
6246 not significantly correlated to ADM or ADR. Results for near upstream riparian harvest and
6247 stream temperature showed either non-significant, or very weakly significant correlations. For
6248 example, there were no significant correlations between the percentage of near upstream riparian
6249 forest recently clear-cut and ADM temperature ($r^2 = 0.03$, $p = 0.79$, $n = 40$), the ADR of stream
6250 temperatures ($r^2 = 0.02$, $p = 0.61$, $n = 40$) or any other stream temperature parameters. The
6251 proportion of total harvested near upstream riparian forest (avg = 0.66, SD \pm 0.34, range = 0.0-
6252 1.0) was weakly correlated with ADM ($r^2 = 0.12$, $p = 0.02$, $n = 40$) and not significantly
6253 correlated with ADR ($r^2 = 0.07$, $p = 0.06$, $n = 40$). Even when the upstream riparian corridor
6254 length was shortened to 400 m and then to 200 m, and the definition of recently harvested was
6255 narrowed to <10 year, no significant relationships between temperature and the condition of the
6256 near upstream riparian forest was found. Results for the effect of physical landscape variables on
6257 stream temperature found that the variables of elevation, slope, aspect, percent of the basin with

6258 a glacial surficial geology, upstream distance of the site to sedimentary (bedrock) geology, and
6259 the percent of sedimentary surficial geology in the basin individually explain between 5% and
6260 14% more of the variability relative to basin harvest. Adding any one of these variables to the
6261 model increases the r^2 from 0.40 up to between 0.48 and 0.51. However, the coefficient for
6262 percent of basin harvested and its standard error stay essentially the same, thus the authors
6263 concluded that adding additional variables to the model did not change the basic finding that
6264 there is a strong relationship between ADM and total amount of harvest in a basin. Thus, for
6265 these models, the percentage of basin area harvested was the best predictor of variation in mean
6266 maximum stream temperatures. The probability of stream temperatures increasing beyond DOE
6267 standards (16 °C for seven-day average of maximum temperatures) increased with percent
6268 harvest. Nine of the 18 sites with 50-75% harvest and seven of the nine sites with >75% harvest
6269 failed to meet these standards. The authors interpret these results as evidence that the total
6270 amount of forest harvested within a basin, and within a riparian stream network are the most
6271 important predictors of changes in summer stream temperatures. They conclude that watersheds
6272 with 25-100% of their total area harvested had higher stream temperatures than watersheds with
6273 little or no harvest. Furthermore, they speculate that past basin-wide timber management can
6274 impact stream temperatures over long periods of time in a way that riparian buffer treatments
6275 cannot entirely ameliorate.

6276

6277 **Stream Temperature**

6278

6279 Groom et al., 2011a

6280

6281 Groom, J.D., Dent, L., Madsen, L.J., 2011. Stream temperature change detection for state and
6282 private forests in the Oregon Coast Range. *Water Resources Research* 47.
6283 <https://doi.org/10.1029/2009WR009061>

6284

6285 The purpose of this study was to evaluate the effectiveness of private and state forest buffer rules
6286 on state water quality stream temperature antidegradation standards in the Oregon Coast Range.
6287 According to the Department of Environmental Quality (DEQ), under the Protecting Cold Water
6288 (PCW) criterion, anthropogenic activities are not permitted to increase stream temperature by
6289 more than 0.3 °C above its ambient temperature. In addition, the cumulative amount of
6290 anthropogenic temperature increase allowed in streams with temperature total maximum daily
6291 loads (TMDLs) is 0.3 °C for all sources combined. Stream temperature and riparian stand
6292 conditions were measured pre- and post-harvest between 2002 and 2008 at 33 sites (18 private-
6293 owned, 15 state-managed). Treatment stands included 26 clear-cuts and 7 partial cuts (leave tree
6294 requirements not specified), all of which were harvested in adherence to FPA (private) and FMP
6295 (state) standards. Private sites followed FPA rules whereby the riparian management area
6296 (RMA)s are 15 and 21 m wide on small and medium fish-bearing streams, respectively, with a 6

6297 m no-cut zone immediately adjacent to the stream. State sites followed the state management
6298 plan whereby a 52 m wide buffer is required for all fish-bearing streams, with an 8 m no cut
6299 buffer immediately adjacent to the stream. Stream temperature data was collected for at least 2
6300 years prior to harvest. Reference reaches were located immediately upstream from the harvested
6301 reaches. Generalized least square regression was used to model ambient conditions while
6302 accounting for temporal autocorrelation. The authors examined prediction intervals to assess the
6303 rule exceedance (>0.3 °C increase in temperature). Results indicate that sites harvested according
6304 to FPA standards exhibited a 40.1% probability that a pre harvest to post harvest comparison of
6305 2 years of data will detect a temperature change of > 0.3 °C. Conversely, harvest to state FMP
6306 standards resulted in an 8.6% probability of exceedance that did not significantly differ from all
6307 other comparisons. The a priori and secondary post hoc multimodel comparisons did not indicate
6308 that timber harvest increased the probability of PCW exceedance at state sites. The authors point
6309 out that the 0.3 °C change threshold still lies 1 or 2 orders of magnitude lower than previous
6310 findings from studies which took place prior to the enactment of the riparian protection
6311 standards. The authors recommend further research looking into the potential persistence of
6312 stream temperature change downstream after harvest. In addition, they recommend looking into
6313 the biological significance of increases in stream temperature change particularly to aquatic life.

6314

6315 **Stream and subsurface water temperature**

6316

6317 Guenther et al., 2014

6318

6319 Guenther, S.M., Gomi, T., Moore, R.D., 2014. Stream and bed temperature variability in a
6320 coastal headwater catchment: influences of surface-subsurface interactions and partial-retention
6321 forest harvesting. *Hydrological Processes* 28, 1238–1249. <https://doi.org/10.1002/hyp.9673>

6322

6323 This study documented changes in stream and subsurface water temperature in response to forest
6324 harvesting in two paired headwater catchments. Specifically, the researchers hypothesized that
6325 post-logging changes in bed temperatures should be greatest in locations experiencing hyporheic
6326 downwelling (DW) and least in areas with lateral inflow/groundwater discharge. This study took
6327 place in the University of British Columbia Malcolm Knapp Research Forest near Vancouver,
6328 Canada. As a part of an ongoing study into the effects of riparian buffers on stream ecology. The
6329 catchments of 3 southerly-aspect first order streams were harvested using partial retention (50%
6330 removal of basal area including riparian zone) methods resulting in approximately 14% reduction
6331 in canopy cover on average; 3 other southerly-aspect streams served as unharvested controls.
6332 Before thinning treatments, the harvested riparian forests were dominated by western hemlock,
6333 (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), and Douglas-fir (*Pseudotsuga*
6334 *menziesii*). The forests were mature second growth forests with trees approximately 30-40 m tall,
6335 and canopy closure than 90%. Harvest operations began in September 2004 and completed in

6336 November of 2004. Temperature data was summarized from 10-minute intervals to daily
6337 minimum, maximum, and mean temperatures for stream and bed temperatures for one-year prior
6338 to, and one year following harvest. An analysis of the post-harvesting effects was conducted
6339 using a paired-catchment analysis. Results from this study show treatment sites resulted in higher
6340 daily maximum stream and bed temperatures after harvest but smaller changes in daily minima.
6341 Daily maximum post-harvest stream temperatures averaged over July and August ranged from
6342 1.6°C to 3°C at different locations. Post harvest changes in bed temperature at the lower reaches
6343 were smaller than changes in stream temperature, but was greater at sites with downwelling (DF)
6344 flow, and decreased with depth at upwelling (UW) and DF sites dropping to approximately 1°C
6345 at a depth of 30 cm. Changes did not vary significantly with depth at the middle reach, and
6346 averaged approximately 1°C change in daily maximum bed temperature over July and August. In
6347 summary, stream temperature responses differed at different locations within the cutblock. Bed
6348 temperatures also differed between UW and DW zones as well as between reaches with different
6349 contributions of lateral inflow. Given evidence that stream/bed temperature is shown to change
6350 spatially and with differences in hyporheic exchange and lateral inflow, the authors conclude by
6351 suggesting further research into the how these results might impact biological and ecological
6352 processes.

6353

6354 **Stream Temperature and evaporation/wind speed**

6355

6356 Guenther et al., 2012 (not in focal, does not separate the effects of shade reduction from wind
6357 speed/)

6358

6359 Guenther, S. M., Moore, R. D., & Gomi, T. (2012). Riparian microclimate and evaporation from
6360 a coastal headwater stream, and their response to partial-retention forest harvesting. *Agricultural
6361 and Forest Meteorology*, 164, 1-9.

6362

6363 The purpose of this study was to (1) develop and test an evaporimeter designed specifically to
6364 measure stream surface evaporation from headwater streams; (2) fit a wind function for
6365 computing evaporation from meteorological observations, and to compare it to previously
6366 published wind functions for evaporation from streams; and (3) quantify the influence of partial-
6367 retention forest harvesting on riparian microclimate and evaporation. This study was conducted
6368 in the University of British Columbia Malcom Knapp Research Forest (MKRF), approximately
6369 60 miles east of Vancouver, Canada and focused on the headwater stream of Griffith Creek. The
6370 harvesting treatment involved removal of 50% of the basal area from within the cut block,
6371 including the riparian zone. Smaller stems were removed, leaving the larger stems for harvest at
6372 a later date. creek. Analysis of paired pre- and post-logging hemispherical photographs indicated
6373 that canopy closure decreased by about 14% due to the logging treatment. Air temperature and
6374 relative humidity were measured by a Campbell Scientific CS500 sensor with stated accuracies

6375 of ± 0.5 °C for temperature and ± 3 –6% for relative humidity. Wind speed was measured with a
6376 Met One anemometer with a stall speed of 0.447 m s^{-1} . Instruments were scanned every 10 s by
6377 a Campbell Scientific CR10x data logger; observations were averaged and stored every 10
6378 minutes. Evaporation was measured using four specially designed evaporimeters comprising an
6379 evaporation pan connected to a Mariotte cylinder. Results showed that Daily mean wind speeds
6380 increased following harvest, but were still consistently lower than wind speeds at the control site,
6381 with a maximum of 1.09 m s^{-1} . Vapor pressure was generally lower after harvesting. Vapor
6382 pressure deficit (vpd) increased following harvesting, but tended to remain lower than vpd
6383 measured at the control site. After harvesting, the relatively high wind speeds in the afternoon
6384 generally coincided with higher water temperatures, which in turn are associated with higher vpd
6385 at the water surface and a stronger vapor pressure gradient to drive evaporation. After harvest,
6386 wind speeds and vapor pressure gradients were higher and stability was weaker, consistent with
6387 the observed increase in evaporation. The authors conclude that the generally stronger relations
6388 between riparian and open microclimate variables after harvesting suggest that the riparian zone
6389 became more strongly coupled to ambient climatic conditions after harvesting as a result of
6390 increased ventilation. Further, that stream evaporation increased markedly as a result of partial
6391 retention harvest, consistent with the decrease in atmospheric vapor pressure, the increase in
6392 stream vapor pressure, the increase in wind speed and the decreased stability. In fact, prior to
6393 harvest, vapor pressure gradients often favored condensation rather than evaporation.

6394

6395 **LW**

6396

6397 Opperman, 2005 (Not in focal)

6398

6399 Opperman, J. J. (2005). Large woody debris and land management in California's hardwood-
6400 dominated watersheds. *Environmental Management*, 35(3), 266-277. DOI:10.1007/s00267-004-
6401 0068-z

6402

6403 The purpose of this paper was to evaluate the effects of stream and riparian area characteristics
6404 (bankfull width, gradient, basal area), and land ownership (public vs. private) on LW loading,
6405 and frequency, and debris jam frequency (response variables) in 21 hardwood-dominated forests
6406 of a Mediterranean climate region of northern California. The relationship between the stream
6407 and riparian area characteristics (explanatory variables: basal area of riparian trees, bankfull
6408 width, and gradient), and the response variables (woody debris loading and frequency, and
6409 debris-jam frequency) were evaluated with linear regression. The characteristics were then
6410 combined with ownership categories and their relative weight in explaining LW loading,
6411 frequency and pool frequency were assessed with a multi-variate analysis. Debris jam frequency
6412 was also analyzed by channel position with a chi-square. Results showed that debris jam
6413 frequency in the 21 reaches analyzed were strongly influenced by living standing trees rooted at

6414 the margins of the bank, especially in channel positions near the stream bank, but also spanning
6415 the channel partially, or completely. In general, LW loading was significantly higher in reaches
6416 adjacent to public lands (104 ± 13 m³/ha) than in those adjacent to private lands (46 ± 8 m³/ha;
6417 $P = 0.0015$). The strongest relationship for LW loading was with bankfull width ($r^2 = 0.32$; $p =$
6418 0.0006), and riparian basal area ($r^2 = 0.22$; $p = 0.006$) riparian basal area. This is likely the cause
6419 of the difference in public vs. private, as the public lands had significantly higher basal area in
6420 the riparian areas at distances >5 m from the stream, than the private lands. Debris jam frequency
6421 was also significantly influenced by riparian area gradient ($r^2 = 0.14$; $p = 0.03$) and basal area (r^2
6422 $= 0.11$; $p = 0.05$). The author concludes that landownership, and thus, land-management
6423 practices are driving factors in LW dynamics in this region.

6424

6425 **LW**

6426

6427 Nowakowski & Wohl, 2008

6428

6429 Nowakowski, A. L., & Wohl, E. (2008). Influences on wood load in mountain streams of the
6430 Bighorn National Forest, Wyoming, USA. *Environmental Management*, 42(4), 557-571.
6431 DOI:10.1007/s00267-008-9140-4

6432

6433 The purpose of this paper is to evaluate the relationship between riparian area characteristics, and
6434 land management practices with in-stream wood-loads in the Bighorn National Forest of
6435 northern Wyoming. The authors hypothesized that 1) valley geometry correlates with wood load,
6436 2) stream gradient correlates with wood load, 3) wood loads are significantly lower in managed
6437 watersheds than in similar unmanaged watersheds. The study analyzed data from 19 conifer
6438 dominated, forested headwater reaches in the bighorn mountains. Study reaches were separated
6439 by two watersheds, managed and unmanaged, with similar drainages, elevation, and lithology.
6440 Unmanaged watersheds were defined as having a history of minimal anthropogenic influences.
6441 The managed watershed had a history of different harvest prescriptions from unregulated in the
6442 late 1800s, clearcutting in the mid-1900s with tie floating practices. The relationship between in-
6443 stream wood loads (m³/ha) was analyzed with 11 valley-scale (elevation, forest type, forest stand
6444 density, etc.) and 13 channel-scale (reach gradient, channel width, etc.) variables with linear
6445 regression. Results support the first and third hypotheses. Across all streams, the highest
6446 explanatory power of all models tested produced land use (managed vs unmanaged), and basal
6447 area as a significant predictor of wood loads ($r^2 = 0.8048$). For the unmanaged watershed the
6448 model produced stream valley sideslope gradient as the single best predictor of wood load ($r^2 =$
6449 0.5748) supporting the first hypothesis. Shear stress was the best predictor of wood load in the
6450 managed watersheds ($r^2 = 0.2403$), These results did not directly support the second hypothesis.
6451 The authors suggest that while shear stress is a function of stream gradient (shear stress and
6452 stream gradient were significantly correlated, $r^2 = 0.9392$), gradient itself did not have the

6453 highest explanatory power of wood load in any of the models tested. Valley characteristics
6454 consistently explained more of the variability in wood load (42-80%) than channel characteristics
6455 (21-33%). When land use (managed vs. Unmanaged) effect on wood loads was analyzed the
6456 number of wood pieces per 100 m of stream was marginally significant ($p = 0.0565$), and the
6457 difference in wood volume per channel was significant ($p = 0.0200$) supporting the third
6458 hypothesis. When the significant valley and channel characteristics of the managed and
6459 unmanaged watersheds were controlled for, the significant difference in wood loads between
6460 managed and unmanaged watersheds were enhanced ($p = 0.0006$). Managed watersheds (1.1
6461 $m^3/100 m$) had, on average, 2-3 times lower in-stream wood loads than unmanaged (3.3 $m^3/100$
6462 m) watersheds. These results suggest watersheds with a history of timber harvest have a decrease
6463 in stream wood loads than unmanaged watersheds, and that wood load dynamics can be driven
6464 by valley morphology, specifically, slope.

6465

6466 **Harvesting Practices on Suspended Sediment Yields**

6467

6468 Hatten et al., 2018

6469

6470 Hatten, J.A., Segura, C., Bladon, K.D., Hale, V.C., Ice, G.G., Stednick, J.D., 2018. Effects of
6471 contemporary forest harvesting on suspended sediment in the Oregon Coast Range: Alsea
6472 Watershed Study Revisited. *Forest Ecology and Management* 408, 238–248.
6473 <https://doi.org/10.1016/j.foreco.2017.10.049>

6474

6475 The objectives of this study were to (1) determine the effects of contemporary harvesting
6476 practices on suspended sediment yields and concentration, and (2) determine if contemporary
6477 harvesting practices produce lower sediment yields than historic practices. This study took place
6478 in the central Oregon Coast Range and consisted of a paired watershed study whereby Flynn
6479 Creek (FC) served as a reference watershed and Needle Branch (NB) served as a treatment
6480 watershed. A third watershed, Deer Creek (DC) served as a secondary control to compare
6481 historical vs contemporary harvest practices. The upper section of the treatment watershed was
6482 clearcut harvested using contemporary harvest practices (no buffer in non-fish-bearing streams
6483 with equipment exclusion zones, and a 15 m no-cut-buffer in fish-bearing streams) adhering to
6484 BMP's. Daily precipitation, discharge, and suspended sediment were collected at all three
6485 watersheds from October 2005 to June 2016. The upper half of the treatment watershed, (35 ha;
6486 measured at the Needle Branch Upper Gage or NBUG) was harvested in 2009 (Phase I) and the
6487 lower half (NBLG) was harvested in the fall of 2014 and mid-summer 2015 (Phase II). A model
6488 was developed using step wise linear regression to compare suspended sediment concentration
6489 (SSC). Differences in SSC among downstream sites and across harvest entries were compared
6490 utilizing an analysis of covariance. Results of the stepwise multiple linear regression showed
6491 strong evidence ($p < .001$) that all covariates (hydrograph limb, cumulative area discharge within

6492 water year, day of water year, daily precipitation, previous day's precipitation) were related to
6493 SSC across all watersheds. Both the mean and maximum SSC were greater in the reference
6494 catchments (FCG and DCG) compared to the harvested catchment (NBLG) across all water
6495 years. In NBLG the mean SSC was 32 mg L⁻¹ (~63%) lower after the Phase I harvest and 28.3
6496 mg L⁻¹ (~55%) lower after the Phase II harvest when compared to the pre-harvest
6497 concentrations. Compared to the reference watersheds, the mean SSC was 1.5-times greater in
6498 FCG (reference) compared to NBLG during the pre-harvest period. After the Phase I harvest the
6499 mean SSC in FCG (reference) was 3.1-times greater and after the Phase II harvest was 2.9-times
6500 greater when compared to the SSC in NBLG, the harvested watershed. Data from historical and
6501 contemporary harvests indicate contemporary practices are more effective at mitigating
6502 sedimentation. Historical data from the original study show harvesting without buffers, road
6503 building, and slash burning resulted in ~2.8 times increase in annual sediment yields and aquatic
6504 ecosystem degradation. The authors conclude that contemporary harvesting practices (i.e., stream
6505 buffers, smaller harvest units, no broadcast burning, leaving material in channels) using buffers
6506 were shown to sufficiently mitigate sediment delivery to streams, especially when compared to
6507 historic practices.

6508

6509 **Riparian Vegetation Removal Effects on Inputs and Production.**

6510

6511 Hetrick et al., 1998 (Removed, outside of timeline)

6512

6513 Hetrick, N.J., Brusven, M.A., Meehan, W.R., Bjornn, T.C., 1998. Changes in Solar Input, Water
6514 Temperature, Periphyton Accumulation, and Allochthonous Input and Storage after Canopy
6515 Removal along Two Small Salmon Streams in Southeast Alaska. *Transactions of the American*
6516 *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)
6517 [8659\(1998\)127<0859:CISIWT>2.0.CO;2](https://doi.org/10.1577/1548-8659(1998)127<0859:CISIWT>2.0.CO;2)

6518

6519 The purpose of this study was to assess whether or not the removal of second growth riparian
6520 vegetation would affect the production of juvenile coho salmon. In addition, this study aims to
6521 understand whether perceived effects are due to changes in habitat or food availability. This
6522 study took place in the Tongas National Forest on Prince of Wales Island, Alaska. Experimental
6523 reaches were divided into untreated and treated sections whereby treated sections had all
6524 vegetation on both sides of the streambank 6-15 m back removed. Stream discharge, water
6525 temperature, periphyton accumulation, allochthonous inputs, and storage of benthic organic
6526 matter were assessed during the summer and fall of 1988-1989. Differences in measured
6527 variables were assessed with a split-block analysis of variance. Results from this study show
6528 average light intensities reaching the water surface was significantly greater ($P < 0.01$) in the
6529 open canopy block than in the closed canopy block and was influenced significantly by weather
6530 conditions. Removal of riparian vegetation in both sections of the study significantly increased

6531 the accumulation of periphyton biomass and chlorophyll a ($P < 0.01$), and significantly decreased
6532 the amount of allochthonous organic inputs to streams ($P < 0.01$). Average daily allochthonous
6533 input rates for closed and open canopy conditions at Eleven creek were 789 and 6 mg AFDM/m²
6534 respectively, while input rates for closed and open canopy conditions at Woodsy creek were 805
6535 and 6 mg AFDM/m². Average daily water temperatures in open and closed canopy blocks at
6536 Eleven Creek were similar in 1988 but were significantly higher in the open blocks than in the
6537 closed blocks in 1989 ($P < 0.01$). The authors conclude by suggesting a thorough investigation
6538 into the interactions and responses of higher trophic levels to increases in periphyton biomass
6539 production and decreases in allochthonous inputs resulting from removal of riparian vegetation.
6540 Furthermore, the authors point out that the ability of stream segments to retain organic inputs
6541 through in-stream large woody debris may be a more important factor for allochthonous input
6542 processing by stream biota than the amount of allochthonous inputs entering a stream.

6543

6544 **Wood Recruitment and Retention**

6545

6546 Hough-Snee et al., 2016

6547

6548 Hough-Snee, N., Kasprak, A., Rossi, R.K., Bouwes, N., Roper, B.B., Wheaton, J.M., 2016.
6549 Hydrogeomorphic and Biotic Drivers of Instream Wood Differ Across Sub-basins of the
6550 Columbia River Basin, USA. *River Research and Applications* 32, 1302–1315.
6551 <https://doi.org/10.1002/rra.2968>

6552

6553 The purpose of this study was to understand the hydrogeomorphic and ecological processes
6554 which lead to wood recruitment and retention in seven sub-basins of the interior Columbia River
6555 Basin (CRB), USA. To achieve this, in-stream wood volume and frequency are quantified across
6556 sub basins. Following this, the riparian, geomorphic, and hydrologic attributes which are most
6557 strongly correlated to in-stream wood loads were determined. Random forest models were used
6558 to identify relationships between ecological and hydrogeomorphic attributes that influence in-
6559 stream wood within each sub-basin. Non-metric multidimensional scaling was performed on a
6560 matrix of hydrogeomorphic and forest cover variables, excluding instream wood frequency and
6561 volume to visualize reaches and sub-basins' relative similarity. To determine how wood
6562 predictors differed between sub-basins, ordinary least squares regression models of wood volume
6563 and frequency were built within each sub-basin. Results from this study show that in stream
6564 wood volume and frequency were distinctly different across all seven sub-basins. Across the
6565 CRB, wood frequency ranged from 0 to 2117.0 pieces km⁻¹, while volume ranged from 0 to 539
6566 m³ km⁻¹. Large wood volume (PERMANOVA $F = 5.1$; $p = 0.001$) and frequency
6567 (PERMANOVA $F = 5.4$; $p = 0.001$) differed significantly between sub-basins. According to
6568 random forest (RF) models, mean annual precipitation, riparian large tree cover, and individual
6569 watershed were the three most important predictors of wood volume and frequency. Watershed

6570 area was the fourth strongest predictor of wood frequency, while catchment-scale and reach-scale
6571 forest cover were the fourth and fifth strongest predictor of wood volume. In contrast, sinuosity
6572 and measures of streamflow and stream power were relatively weak predictors of wood volume
6573 and frequency. Taken together, wood volume and frequency increased with precipitation and
6574 large riparian tree cover and decreased with watershed area. Final RF models explained 43.5% of
6575 the variance in volume and 42.0% of the variance in frequency of in stream wood loads. Results
6576 for drivers of wood frequency and volume between sub-basins were highly variable either
6577 showing no relationship between candidate models and predictive power (e.g., $r^2 \leq 0.12$; Entiat
6578 sub-basin). The highest predictive models for wood volume ($r^2 > 0.55$) and wood frequency (r^2
6579 ≤ 0.45) were for the John Day sub basin. Depending on the sub basin wood volume and
6580 frequency was positively correlated with forest cover, watershed area, large tree cover, 25-year
6581 flood event stream power, riparian conifer cover, and precipitation. Negative correlations,
6582 depending on sub basin, of wood volume and frequency with baseflow discharge, riparian woody
6583 cover, watershed area, and large tree cover. Given the heterogeneous results across all sub-basins
6584 studied, the authors conclude by emphasizing the importance of incorporating local data and
6585 context when building wood models to inform future management decisions.

6586

6587 **Stream Temperature**

6588

6589 Hunter, 2010 (not in focal, treatments and results not relevant to questions)

6590

6591 Hunter, M.A., 2010. Water Temperature Evaluation of Hardwood Conversion Treatment Sites
6592 Data Collection Report (Data Collection Report). Cooperative Monitoring, Evaluation, and
6593 Research (CMER). Fp_cmer_05_513

6594

6595 The purpose of this study is to evaluate the response of stream temperature to changes in canopy
6596 cover using a before-after-control-impact design. This study took place along nine hardwood-
6597 dominated riparian stands in Western Washington. Variables measured among locations and years
6598 include riparian conditions, canopy cover, channel dimensions, substrate, flow and stream
6599 temperature. Results from this study show that hardwood conversion buffers (HCB -
6600 approximately 15 m width) intended to convert hardwood-dominated riparian areas to conifer-
6601 dominated riparian areas usually resulted in decreased canopy cover of streams. Mean Global
6602 Site Factor (GSF - the proportion of global radiation under a plant canopy relative to the amount
6603 in an open area) increased in most study sites with HCB's. However, mean GSF did not change
6604 substantially at sites with buffers closer to standard (~ 18 – 45 m) non-hardwood conversion
6605 buffers. Temperature was highly variable over time and among locations suggesting stream
6606 temperature is affected by many factors that might differ among locations and throughout time.
6607 Longitudinal patterns of warming and cooling were consistent at all sites indicating the potential

6608 importance of careful site selection to account for changes in the longitudinal distribution of
6609 temperatures.

6610

6611 **Influence of Stream Geomorphology on Water Temperature**

6612

6613 Hunter & Quinn, 2009

6614

6615 Hunter, M.A., Quinn, T., 2009. Summer Water Temperatures in Alluvial and Bedrock Channels
6616 of the Olympic Peninsula. *Western Journal of Applied Forestry* 24, 103–108.
6617 <https://doi.org/10.1093/wjaf/24.2.103>

6618

6619 The purpose of this study was to understand how stream geomorphology influences water
6620 temperature in managed stands on the Olympic Peninsula, Washington. Sites chosen for this
6621 included an alluvial study site and a bedrock study site whose overall characteristics were
6622 otherwise comparable apart from geomorphology. The alluvial study site was a 1.6-km reach of
6623 Thorndyke Creek. The bedrock study site was a 1.4-km reach of the South Fork Pysht River.
6624 Both channels were located in 35–50-year-old managed forests dominated by Douglas-fir
6625 (*Pseudotsuga menziesii*) in the uplands and red alder (*Alnus rubra*) in the riparian zone. Surface
6626 substrate at the alluvial channel was composed mostly of gravel, whereas the bedrock channel
6627 was composed of mostly bedrock, boulder, and cobble. The mean solar input (GSF: global site
6628 factor) did not differ between streams. Water temperature was recorded at 75-m intervals along
6629 each channel during the summers of 2003 and 2004. Results from this study show consistent
6630 differences in stream temperature response in alluvial versus bedrock channels. Seasonal
6631 maximum and minimum average daily temperatures varied less at the alluvial site compared to
6632 the bedrock site. This, the authors suggest may be due to hyporheic exchange in alluvial channels
6633 helping to buffer surface water temperatures from gaining or losing heat. In addition,
6634 groundwater may also contribute to the increased stability at the alluvial site. Two same-day
6635 measurements at each site showed the alluvial site gaining 8% of its flow, as compared to the
6636 bedrock site whose flow decreased by approximately 15%. The bedrock site was also shown to
6637 have the highest variation in reach-scale water temperatures during low flow. The authors
6638 conclude that stream geomorphology may have profound impacts on spatial and temporal
6639 patterns of channel water temperature. The authors suggest temperature reading from a single
6640 location may not accurately represent the entire channel. Additional research involving collection
6641 of temporal and longitudinal data will be needed to tailor riparian buffers to channel type.

6642

6643 **Stream temperature, sediment, nutrient**

6644

6645 Murray et al., 2000

6646

6647 Murray, G. L. D., Edmonds, R. L., & Marra, J. L. (2000). Influence of partial harvesting on
6648 stream temperatures, chemistry, and turbidity in forests on the western Olympic Peninsula,
6649 Washington. *Northwest science.*, 74(2), 151-164. Handle: <https://hdl.handle.net/2376/1065>

6650

6651 This study investigates the effects of partial watershed harvest (7-33%) on stream temperature,
6652 chemistry, and turbidity relative to an unharvested old-growth watershed in the western Olympic
6653 Peninsula, Washington. Both harvested watersheds (Rock and Tower creeks) originally contained
6654 old-growth forests. Rock Creek had 7% of its watershed harvested in 1981, and Tower Creek had
6655 33% of its watershed harvested between 1985 and 1987. Logging extended to the stream edge
6656 near the in-stream monitoring sites. Data for stream daily maximum, minimum, and mean
6657 temperatures, chemistry, and turbidity was recorded and monitored from June 1996 to June 1998
6658 (10-15 years post-harvest). Differences in variables between treatment and reference watersheds
6659 were compared with a one-way ANOVA with a posthoc Tukey HSD test. Results showed higher
6660 maximum summer stream temperatures (15.4 °C), and lower winter maximum stream
6661 temperatures (3.7 °C) in the two treatment watersheds compared to the unharvested reference
6662 watershed (12.1 °C and 6.0 °C for summer max, and winter max, respectively). Winter minimum
6663 temperatures for one of the harvested watersheds reached 1.2 °C (Rock Creek) compared to a
6664 winter minimum of 6 °C Thus, seasonal variation of stream maximum temperatures and winter
6665 minimum temperatures were more extreme in the treatment watershed than in the control. There
6666 were no seasonal patterns or significant differences between watersheds in stream chemistry
6667 except for calcium and magnesium concentrations being consistently higher in the unharvested
6668 watersheds. Turbidity was low and not significantly different between watersheds. The authors
6669 interpret these results as evidence of partial harvest having minimal impact on stream
6670 temperatures, chemistry, and turbidity long-term (after 10-15 years). The stream temperature
6671 changes were significant but did not exceed the 16 °C threshold used as a standard for salmonid
6672 habitat. However, there was no data collection during the first decade following harvest.

6673

6674 **Channel Habitat, Particle Size, Stream Temperature, and Woody Debris Response to**
6675 **Harvest**

6676

6677 Jackson et al., 2001

6678

6679 Jackson, C.Rhett., Sturm, C.A., Ward, J.M., 2001. Timber Harvest Impacts on Small Headwater
6680 Stream Channels in the Coast Ranges of Washington1. *JAWRA Journal of the American Water*
6681 *Resources Association* 37, 1533–1549.

6682 <https://doi.org/10.1111/j.1752-1688.2001.tb03658.x>

6683

6684 The purpose of this study was to evaluate changes in stream temperature, particle size
6685 distributions of bed material, and channel habitat distributions in 15 first- or second order
6686 streams located on the Coast Range of Western Washington. Four of the fifteen stream basins
6687 were not harvested and served as references; three streams were cut with unthinned riparian
6688 buffers; one with a partial buffer; one with a buffer of non-merchantable trees; and six were
6689 clearcut to the stream edge. Buffer widths varied by operation; the average buffer width varied
6690 from 15 – 21 meters. The narrowest buffer measured on one side of the stream was 2.3 meters.
6691 Data for woody debris, sediment concentrations, turbidity, and stream temperatures were
6692 recorded for one-year prior to harvest (1998). Harvest was conducted in the spring and early
6693 summer of 1999, and post-harvest data was collected for about a month after operations were
6694 complete. Thus, the results presented in this study represent changes in stream attributes and
6695 characteristics immediately following harvest. Results from this study show that logging without
6696 buffers had immediate and dramatic effects on channel morphology. Without buffers, and the
6697 relatively steep topography of the study sites logging debris tended to accumulate at the bottom
6698 of slopes thereby burying or covering many headwater streams. Covered channels were defined
6699 in this study as having flow completely obscured by organic debris, but a recognizable channel
6700 still exists below the debris. Buried channel was defined as having so much organic detritus in
6701 the flow cross-section that the channel was no longer definable. Needles, twigs, whole branches,
6702 and logs buried headwater streams with a mean depth of 0.94 meters of organic debris (range:
6703 0.5 - 2.0 meters). Of the clearcut streams the percent of stream buried with organic matter ranged
6704 from 6 to 90%, and the percent covered by organic matter ranged from 8 to 85%. The sum of
6705 buried and covered for each stream ranged from 72 to 100%. On the other hand, most buffered
6706 streams had 0% covered or buried by organic matter post-harvest with the only exception being
6707 one stream that experienced blowdown post-harvest that covered 29% of the stream. While
6708 debris accumulation tended to protect streams from the effects of solar radiation, organic logging
6709 debris was also shown to trap fine sediment in the channels which, in the near term, greatly
6710 reduced downstream sediment movement. As a result of increased roughness and additional bank
6711 failures within the clearcut sites, sediment size shifted towards finer particles growing from 12 to
6712 44 percent. In contrast, particle size distributions continued nearly unchanged in buffered and
6713 reference sites. In the first summer after logging, significant increases were detected in overall
6714 macroinvertebrate densities, collector densities, shredder abundance and biomass, and organic
6715 and inorganic matter accretion. However, these responses were not detected one year following
6716 logging. For stream temperature changes, because the data collection was for such a short period
6717 of time (1-year pre- and 1-month post-harvest), and because the summer of 1999 was much
6718 cooler than 1998, the assessment of harvest effects on stream temperature changes was difficult.
6719 Thus, to interpret significant changes in stream temperatures from pre- to post- harvest, daily
6720 maximum temperatures were plotted against the appropriate reference stream, and a regression
6721 equation was calculated. The slopes of the regression lines were compared with a student's t-test
6722 to determine significant differences. Of the seven clearcut streams, three showed no significant
6723 changes in temperature, one became cooler (-1.1 °C), one became slightly warmer (+0.8 °C), and

6724 the other 2 became warmer or colder depending on location with decreases in temperature
6725 upstream (-2.2 and -1.7 °C) and increases in temperature downstream (+5.2 and +15.1 °C). The
6726 buffered streams had significant but less dramatic changes in temperature with one decreasing in
6727 temperature (-0.3 °C), and 2 increasing in temperature (+1.6 and +2.4 °C). The one site with the
6728 non-merchantable buffer had much higher temperature increases (+3.7 and +6.6 °C). The authors
6729 posit that sites which retained riparian buffers succeeded in keeping debris out of streams as well
6730 as served to protect streambanks from failure or erosion. Some mature trees left within buffers
6731 experienced blow down and spanned the channel. While the clearcut streams had nearly all
6732 canopy cover removed, the buildup of slash and LW in the stream also provided shade and
6733 insulation that caused reductions in stream temperatures, or slight increases with one exception
6734 (+15.1 °C) The authors point out that this study only served to point out immediate effects of
6735 logging on physical channel conditions. Although important, there are still many questions about
6736 how channel conditions will evolve over time.

6737

6738 **LW**

6739

6740 Meleason et al., 2003

6741

6742 Meleason, M. A., Gregory, S. V., & Bolte, J. P. (2003). Implications of riparian management
6743 strategies on wood in streams of the Pacific Northwest. *Ecological Applications*, 13(5), 1212-
6744 1221. <https://doi.org/10.1890/02-5004>

6745

6746 This study used simulation modeling to evaluate the potential effects of three different riparian
6747 and watershed harvest scenarios on the standing stock of large wood in a hypothetical stream in
6748 the Pacific Northwest. The three scenarios involved harvest 1) clearcut to the streambank, 2)
6749 riparian management buffer widths ranging from 6-75 m, and 3) riparian buffers of various
6750 widths with upland forest plantation. The effects of each scenario on wood load dynamics were
6751 simulated with OSU STREAMWOOD for four harvest rotation periods (no harvest, 60, 90, and
6752 120 years) over the course of 720 years. Results for scenario one (clear-cut to stream) showed
6753 minimal accumulation of wood into the stream with little change over time due to the lack of a
6754 forested riparian management zone. Results for scenario two showed the maximum standing
6755 stock of in-stream wood loads required ≥ 30 m no-cut buffer zones for 500-year-old forests.
6756 Wood loads in streams with 6 m wide buffers showed 32% of standing wood load stocks after
6757 240 years. Results from scenario three showed minimal amounts of wood contributed into
6758 streams from forest plantations when > 10 m wide buffers were used. The authors interpret these
6759 results as evidence that riparian buffer widths and forest age are more important for estimating
6760 changes in wood loads over time than the harvest rotation age of plantation forests.

6761

6762 LW

6763

6764 Martin & Grotefendt, 2007

6765

6766 Martin, D. J., & Grotefendt, R. A. (2007). Stand mortality in buffer strips and the supply of
6767 woody debris to streams in Southeast Alaska. *Canadian Journal of Forest Research*, 37(1), 36-49.
6768 <https://doi.org/10.1139/x06-209>

6769

6770 This study compared riparian stand mortality and in-stream LW recruitment characteristics
6771 between riparian buffer strips with upland timber harvest and riparian stands of unharvested
6772 watersheds using aerial photography. This study was conducted in the northern and southern
6773 portions of Southeast Alaska at multiple sites in nine timber harvest areas. All study sites were
6774 along moderate- and low-gradient streams with channel widths ranging from 5 m to 30 m wide.
6775 All buffer strips were conifer dominated and a minimum of 20 m wide that included selective
6776 harvest within the 20 m zone. Reference sites were along unharvested reaches in the same area.
6777 Stand mortality was estimated by the proportion of downed trees within a buffer strip.
6778 Differences in downed tree proportions relative to reference streams were assumed to be caused
6779 by timber harvest, accounting for selective in-buffer harvests. A one-tailed paired t-test or a
6780 Wilcoxon signed rank test was used to check for statistical differences between treatment and
6781 reference sites. Results showed significantly higher mortality (based on cumulative stand
6782 mortality: downed tree counts divided by standing tree counts + downed tree counts),
6783 significantly lower stand density (269 trees/ha in buffer units and 328 trees/ha in reference units),
6784 and a significantly higher proportion of LW recruitment from the buffer zones of the treatment
6785 sites than in the reference sites. Densities within all units ranged from 0 – 1334 trees/ha
6786 depending on location. Overall, mean stand density in the buffer units was 18% lower than in the
6787 reference units. Results also showed that mortality varied with distance to the stream.
6788 Differences in mortality for the treatment sites were similar to the reference sites for the first 0-
6789 10 m from the stream (only a 22% increase in the treated sites). However, mortality in the outer
6790 half of the buffers (10-20 m) from the stream in the treatment sites was more than double (120%
6791 increase) what was observed in the reference sites. This caused a change in the LW recruitment
6792 source distance curves, with a larger proportion of LW recruitment coming from greater
6793 distances in logged watersheds. LW recruitment based on the proportion of stand recruited (PSR)
6794 was significantly higher in the buffered units compared to the reference units. However, PSR
6795 from the inner 0-20 m was only 17% greater in the buffer units than in the reference units, while
6796 PSR of the outer unit (10 – 20 m) was more than double in the buffered units than in the
6797 reference units. The researchers conclude that the increase in mortality was caused by an
6798 increased susceptibility to windthrow. They estimate that future recruitment potential from the
6799 logged sites diminished by 10% relative to the unlogged reference sites.

6800

6801 **Stream temperatures**

6802

6803 Macdonald et al., 2003b

6804

6805 Macdonald, J. S., MacIsaac, E. A., & Herunter, H. E. (2003). The effect of variable-retention
6806 riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest
6807 ecosystems of British Columbia. *Canadian journal of forest research*, 33(8), 1371-1382.
6808 <https://doi.org/10.1139/x03-015>

6809

6810 This study investigates the impacts of forest harvest on stream temperatures under three variable
6811 retention buffer treatments in headwater streams of the interior sub-boreal forests of British
6812 Columbia. Temperature data were recorded for two years pre- and five years post-harvest from
6813 five harvested streams and two unharvested reference streams. Differences between pre- and
6814 post-harvested stream temperatures were compared with the paired reference streams using
6815 repeated measures ANOVA. Treatment riparian areas were harvested with the following
6816 prescriptions: 1) low-retention – removal of all merchantable timber >15 or >20 cm DBH for
6817 pine or spruce respectively, within 20 m of the stream 2) high-retention – removal of
6818 merchantable timber >30 cm DBH within 20-30 m of the stream, and 3) Patch-cut – high
6819 retention for the lower 60% of watershed approaching streams and removal of all vegetation in
6820 the upper 60% of the watershed. Eight first-order streams were included in this study: two

6821 in the Gluskie Creek watershed (G5, G7) and six in the Baptiste Creek watershed (B1–B6). Five
6822 of these streams were within the harvested boundaries (2 high-retention, 2 low-retention, and 1
6823 patch cut), and 3 reaches outside of the harvest boundary served as controls. Results showed a
6824 significant increase in stream temperatures ranging from 4 – 6 °C at five years post-harvest, and
6825 increased ranges of diurnal temperature fluctuations for all treatment streams relative to the
6826 reference streams. Streams that had summer maximum mean weekly temperatures of 8°C before
6827 harvesting had maximum temperatures near 12°C or more following harvesting. Daily ranges of
6828 1.0–1.3°C before harvesting became 2.0–3.0°C following harvesting, Greater temperature ranges
6829 occurred in low-retention and patch treatments than the high-retention or control treatment
6830 locations. The high-retention buffer treatment mitigated temperature increases for the first three
6831 years. Still, increased mortality (windthrow) caused a reduction in the canopy that increased
6832 stream temperatures equivalent to other treatment streams by year five. The results of this study
6833 show evidence that high-retention buffers are no more effective in preserving stream temperature
6834 changes than small retention buffers when treatment areas have a high susceptibility to
6835 windthrow.

6836

6837 **Sediment delivery pathways**

6838

6839 Litschert & MacDonald, 2009

6840

6841 Litschert, S. E., & MacDonald, L. H. (2009). Frequency and characteristics of sediment delivery
6842 pathways from forest harvest units to streams. *Forest Ecology and Management*, 259(2), 143-
6843 150. <https://doi.org/10.1016/j.foreco.2009.09.038>

6844

6845 This study investigates the frequency of sediment delivery pathways (“features”) in riparian
6846 management areas and measures the physical characteristics and connectivity of these pathways
6847 following timber harvest. The results of this study were then used to develop models for
6848 predicting the length and connectivity of pathways formed from harvest units. Data was collected
6849 from over 200 harvest units with riparian management areas in the Eldorado, Lassen, Plumas,
6850 and Tahoe National Forests in the Sierra and Cascade mountains of northern California. Riparian
6851 buffer widths for this area are 90 m and 45 m for perennial and annual streams respectively. No
6852 machinery is allowed in the riparian management areas. Data collected and analyzed for the
6853 pathways included years since harvest, mean annual precipitation, soil depth, soil erodibility,
6854 hillslope gradient, aspect, and elevation. Characteristics of pathway length, gradient, and
6855 roughness were also collected. Relationships between site variables and pathway variables were
6856 assessed using linear regression. The site variables with the most significant relationships with
6857 the pathway variables were used in a multivariate regression model to predict pathway length.
6858 Only 19 of the 200 harvest units had sediment development pathways. Pathways ranged in age
6859 (time since harvest) from 2 to 18 years, and in length from 10 m to 220 m. Of the 19 pathways,
6860 only six were connected to streams, and five of those originated from skid trails. Pathway length
6861 was significantly related to mean annual precipitation, cosine of the aspect, elevation, and
6862 hillslope gradient. The authors conclude that timber prescription practices for these National
6863 Forests are effective in reducing sediment delivery pathways. The authors interpret these results
6864 as evidence that skid trails should be directed away from streams, maintaining surface roughness,
6865 and promptly decommissioning skid trails.

6866

6867 **LW**

6868

6869 Liquori, 2006

6870

6871 Liquori, M. K. (2006). POST-HARVEST RIPARIAN BUFFER RESPONSE: IMPLICATIONS
6872 FOR WOOD RECRUITMENT MODELING AND BUFFER DESIGN 1. *JAWRA Journal of the*
6873 *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)
6874 [1688.2006.tb03832.x](https://doi.org/10.1111/j.1752-1688.2006.tb03832.x)

6875

6876 This study investigates the differences in treefall characteristics in riparian management areas
6877 based on ecological and physiographic variables to give insight on the variables important for
6878 wood recruitment modeling. Data were collected from 20 riparian buffer sites that had all been
6879 clearcut within three years of sampling with standard no-cut buffers 25 ft. An additional 50-100
6880 ft buffer was applied to fish-bearing streams depending on stream type, in a managed tree farm in
6881 the Cascade Mountains of western Washington. These riparian buffers generally consisted of
6882 naturally regenerated, second-growth conifer stands about 45 to 70 years old. "Very modest"
6883 thinning was applied to some stands to meet wildlife objectives and any downed wood not
6884 affecting the channel was removed. Tree characteristic data collected included tree size (DBH
6885 and height), species, fall direction, tree fall angles, estimated cause of mortality, and distance to
6886 the stream. Site characteristics included stream gradient, valley morphology, and time since
6887 harvest. Tree recruitment probability curves were developed as a function of tree height using
6888 methods described by Beschta, (1990). Results showed that wind-caused mortality and tree fall
6889 rates were significantly higher, up to three times higher, than competition-induced mortality
6890 within buffers for three years following treatment. The median observed treefall per site was
6891 15% of all trees in each buffer, ranging from 1 to 57%. total treefall at each site for one, two, and
6892 three years since harvest was $16 \pm 10\%$, $28 \pm 21\%$, and $10 \pm 10\%$, respectively. Total treefall
6893 percentage for each site was not correlated to years since harvest (Spearman $R = 0.11$; $p = 0.34$).
6894 The mean and standard deviation of the total normalized treefall for one-year old sites was $405 \pm$
6895 394 trees/km ($n = 9$), for two-year old sites was 264 ± 280 trees/km ($n = 7$), and for three-year
6896 old sites was 556 ± 316 trees/km ($n = 4$). Treefall varied significantly by species. Downed red
6897 alder (*Alnus rubra*), western red cedar (*Thuja plicata*), and Douglas-fir (*Pseudotsuga menziesii*)
6898 comprised 3 percent to 8 percent of all downed trees; these species had treefall rates ranging
6899 from 5 percent to 9 percent of the total number of trees of the same species. By contrast, treefall
6900 rates for western hemlock (*Tsuga heterophylla*) and Pacific silver fir (*Abies amabilis*) ranged
6901 from 23 percent to 26 percent. Treefall rates also varied somewhat by size, with the 31 to 41 cm
6902 (12 to 16 in) diameter class having the greatest treefall rates (All trees were grouped into size
6903 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
6904 than 20 in). Treefall following harvest greatly exceeded the expected competition induced
6905 mortality rates (posited by Franklin, 1970) of 0.5%, and the model of average competition
6906 mortality used in Rainville et al. (1985), which ranged from 0.7 - 1.6%, and 2% per year for bank
6907 undercutting. Treefall direction was heavily biased towards the channel regardless of channel or
6908 buffer orientation and tree fall probability was highest in the outer areas of the buffers (adjacent
6909 to the harvest area). Fall direction bias increased significantly in the inner portions of the buffer.
6910 Within the 0 to 7 m zone and 7 to 15 m zone, 68% and 67% of the trees, respectively, fell toward
6911 the channel ($n = 125$ and 153 , respectively). Only 44% of the outer zone (> 15 m) downed trees
6912 fell toward the channel ($n = 403$). Generally, recruitment was negatively correlated to buffer
6913 width ($r^2 = 0.40$). Treefall was generally highest at the outside edges of buffers (50+ feet),
6914 representing about 60% of the total observed treefall, while the 0–25-foot zone represented
6915 ~18%, and the 25–50-foot zone represented ~22%. The authors interpret their results as evidence
6916 that tree fall models that use a random fall direction may underrepresent the probability of LW
6917 recruitment into streams. Further, they suggest that the increase in windthrow mortality and the
6918 probability of tree fall with increasing distance from the stream should be considered.

6919

6920 **LW**

6921

6922 Lininger et al., 2021 (removed from focal list, this is a case study)

6923

6924 Lininger, K. B., Scamardo, J. E., & Guiney, M. R. (2021). Floodplain large wood and organic
6925 matter jam formation after a large flood: Investigating the influence of floodplain forest stand
6926 characteristics and river corridor morphology. *Journal of Geophysical Research: Earth Surface*,
6927 126(6), e2020JF006011. <https://doi.org/10.1029/2020JF006011>

6928

6929 This study examines how river corridor morphology and forest stand density influence LW and
6930 coarse particulate matter (CPOM) deposition patterns in the flood plain resulting from a 400-year
6931 flood event in West Creek in the Colorado Front Range in 2013. The researchers tested the
6932 hypothesis that if river corridor geomorphology affects LW and CPOM deposition then there
6933 should be an inverse relationship between elevation above and distance from the stream's edge.
6934 Further, that deposition frequency would be higher in unconfined portions of the corridor.
6935 Considering forest stand structure, the researchers hypothesized that LW/CPOM jams would be
6936 pinned by trees, higher in intermediate forest densities, and decrease in size with increasing
6937 forest stand density. Field data of LW/CPOM jams were analyzed with non-parametric Spearman
6938 correlation tests to determine the strength of their relationship with channel and stand
6939 characteristics. Results showed support for most of the hypotheses. LW accumulations did
6940 decrease in size with distance from the stream, but CPOM did not. Confined channels (steeper
6941 reaches) contained fewer LW/CPOM loads per unit area. The authors speculate that these reaches
6942 had higher flow rates and thus lower deposition during the flood. CPOM jams increased in
6943 number per area with increasing stand density with most jams pinned against live trees. The
6944 authors conclude that the effect of riparian forest stand density is evidence that riparian forests in
6945 the floodplains should be preserved to increase LW and CPOM trapping probability.

6946

6947 **Stream Temperature**

6948

6949 Janisch et al., 2012

6950

6951 Janisch, J.E., Wondzell, S.M., Ehinger, W.J., 2012. Headwater stream temperature: Interpreting
6952 response after logging, with and without riparian buffers, Washington, USA. *Forest Ecology and*
6953 *Management* 270, 302–313. <https://doi.org/10.1016/j.foreco.2011.12.035>

6954

6955 The purpose of this study was to assess the stream temperature response to three different
6956 harvesting treatments in small, forested headwater catchments in western Washington. The pre-
6957 logging calibration period lasted 1–2 summers and stream temperatures were monitored for two
6958 or more summers after logging. Harvest treatments occurred between September 2003 and July
6959 2005; catchments were clustered by harvest year for analysis. A before-after-control-impact
6960 study design was used to contrast stream temperature responses for three forest harvest
6961 treatments: clearcut logging to the stream (n=5), a continuous buffer (n=6) with widths 10-15 m
6962 on each side of the channel, and a patched buffered (n=5) where portions of the riparian forests
6963 ~50-110 m long were retained in distinct patches along some portion of the channel with the
6964 remaining riparian area clearcut. For the patch buffers there was no standard width, the buffer
6965 spanned the full width of the floodplain area and extended well away from the stream. Upland
6966 areas adjacent to buffers were clearcut. Regression relationships were developed between
6967 temperatures measured in the treatments and corresponding reference catchments. A simple
6968 ANOVA model was used that only included fixed effects for treatment, years since treatment,
6969 and day of year. Because of the unbalanced experimental design and variation in time of harvest,
6970 clustering of treatments caused the sample sizes to become too small to apply a more complex
6971 nested, repeated measures ANOVA could not be used. Correlation analysis was conducted
6972 between post-harvest stream temperatures and descriptive variables on a subset of catchments to
6973 examine possible factors that might control post-harvest thermal responses. Results from this
6974 study show significant increases in stream temperature in all treatments. Although temperature
6975 responses were highly variable within treatments, July and August daily maximum temperatures
6976 increased in clearcut catchments during the first year after logging by an average of 1.5°C (range
6977 0.2 to 3.6°C), in patch-buffered catchments by 0.6°C (range – 0.1 to 1.2°C), and in continuously
6978 buffered catchments by 1.1°C (range 0.0 to 2.8°C). Canopy cover in all streams averaged 95%
6979 prior to harvest and did not differ between treatment and reference streams. Following treatment,
6980 canopy cover in the clearcut catchments averaged 53%, canopy cover in the patch buffer
6981 treatment averaged 76%, and canopy cover in the continuous buffer treatment averaged 86%.
6982 Following treatment, the canopy cover of the clearcut and patch buffer treatments were
6983 significantly lower than in the reference streams. The continuous buffer treatments did not differ
6984 significantly from the reference streams for canopy cover. Further analyses which attempted to
6985 identify variables responsible for controlling the extent of stream temperature responses showed
6986 the amount of cover retained in the riparian buffer was not a strong explanatory variable. Post-
6987 treatment temperature changes suggested that treatments ($p = 0.0019$), the number of years after
6988 treatment ($p = 0.0090$), and the day of the year ($p = 0.0007$) were all significant effects
6989 explaining observed changes in temperature. Wetland area ($r^2 = 0.96$, $p < 0.01$) and length of
6990 surface flow ($r^2 = 0.67$, $p = 0.05$) were strongly correlated with post-logging temperature
6991 changes. Regression analysis of these variables showed streams with fine-textured substrates
6992 responded differently than coarse textured substrates. The authors speculate this is possibly due
6993 to groundwater interactions which can buffer thermal responses of small streams. In summary,
6994 the authors conclude that their results suggest small headwater streams may be fundamentally
6995 different than larger streams partly because factors other than canopy shade can greatly influence
6996 stream energy budgets to moderate stream temperatures despite changes and/or removal of the
6997 overstory canopy.

6998

6999 **Large woody debris**

7000

7001 Jones et al., 2011 (Removed from focal list, study not relevant to focal questions)

7002

7003 Jones, T.A., Daniels, L.D., Powell, S.R., 2011. Abundance and function of large woody debris in
7004 small, headwater streams in the Rocky Mountain foothills of Alberta, Canada. *River Research*
7005 *and Applications* 27, 297–311. <https://doi.org/10.1002/rra.1353>

7006

7007 The purpose of this study was to assess LW abundance in the upper foothills of the Rocky
7008 Mountains in Alberta, Canada. This study also sought to understand key processes that underlie
7009 changes in LW function. Finally, this study used results to develop a LW recruitment, decay and
7010 interaction model. This research was conducted in 21 headwater streams spanning two
7011 watersheds. At each site, all LW was sampled and was classified according to decay, orientation,
7012 position and function. LW frequency, total volume, and total in-stream volume were calculated
7013 and analyzed for differences using a one-way ANOVA followed by a Tukey post hoc test to
7014 differentiate among significant classes. Results show LW frequency was greater in the Alberta
7015 foothills (64.0 ± 3.3 LW 100 m¹) than in many small, headwater streams in mountain (46.2 ± 3.6),
7016 coastal (47.6 ± 3.8), mixed broad-leaf (47.0 ± 4.2) and boreal (31.0 ± 3.0) streams. This, the
7017 authors suggest, is likely due to the narrow bankfull width channels characteristic of the Alberta
7018 foothills which are less able to transport LW downstream. LW with ≥ 20 cm was more frequent in
7019 coastal streams, and overall LW volume was also greatest in coastal streams (721.0 ± 99.9 m³ ha⁻¹).
7020 The authors note that large LW volumes in coastal streams are likely due to geomorphic
7021 disturbances alongside large, long-lived, decay resistant tree species. According to Harmon et al.
7022 1986, much of the variation in LW recruitment is due to differences in species life history and
7023 forest type which together govern log size and decay rates.

7024

7025 **Suspended Sediment**

7026

7027 Karwan et al., 2007

7028

7029 Karwan, D., Gravelle, J., Hubbart, J., 2007. Effects of timber harvest on suspended sediment
7030 loads in Mica Creek, Idaho. *Forest Science* 53, 181–188.
7031 <https://doi.org/10.1093/forestscience/53.2.181>

7032

7033 The purpose of this study was to examine the effects of forest road construction and timber
7034 harvest on total suspended solids (TSS) in a forested watershed. This study took place at the
7035 Mica Creek Experimental Watershed in northern Idaho. The study area consisted of dense,
7036 naturally regenerated, even-aged stands ~65 years old and ~300 trees per acre. Timber harvesting
7037 and heavy road use began in 2001. Treatments in the paired-watershed experiment consisted of
7038 (1) commercial clearcut of the watershed area of 50%, and was broadcast burned and replanted
7039 by the end of May 2003, (2) partial cut in which half the canopy was removed in 50% of the
7040 watershed in 2001, with final 10% of log processing and hauling in early summer of 2002. and
7041 (3) a no-harvest control. All harvests were carried out according to best management practices
7042 and in accordance with the Idaho Forest Practices Act. At the time of the study this involved a
7043 22.86 m (75 ft) stream protection zones (SPZs) on each side of fish-bearing (Class I) streams.
7044 The inner 50 ft is an equipment exclusion zone where no ground-based skidding machinery is
7045 allowed. Timber harvesting is allowed in Class I SPZs, but 75% percent of existing shade must
7046 be retained. Along non-fish-bearing (Class II) streams, harvesting equipment was excluded from
7047 entering within 9.14 m (30 ft) of definable stream channels and any cut trees were felled away
7048 from the stream; however, there were no tree retention requirements. In the clearcut and partial
7049 cut units, line skidding was used on slopes in the watershed exceeding approximately 20%, while
7050 tractor skidding was used on the lower gradient slopes. On all skid trails, drainage features, such
7051 as water bars, were installed for erosion control at the end of the harvest period. Time series data
7052 were compiled for all measured TSS values from 1991 through 2004. Data was collected via
7053 seven stream monitoring flumes located within the Mica Creek Watershed. Monthly TSS loads
7054 were compared across watersheds for five time intervals: (1) pretreatment: ~6 years, (2)
7055 immediate post-road construction: ~1 year, (3) recovery post-road construction: ~3 years, (4)
7056 immediate post-harvest: ~1 year, and (5) recovery post-harvest: ~3 years. Trends in the
7057 relationship between treatment and control watersheds were statistically examined for each of the
7058 time intervals. Treatments in the paired-watershed experiment consisted of (1) commercial
7059 clearcut of the watershed area of 50%, and was broadcast burned and replanted, (2) partial cut in
7060 which half the canopy was removed in 50% of the watershed (3) a no-harvest control. All
7061 harvests were done according to best management practices and the Idaho Forest Practices Act.
7062 This included equipment exclusion zones of 50- and 30-feet for fish- and non-fish-bearing
7063 streams, respectively. On all skid trails, drainage features, such as water bars, were installed for
7064 erosion control at the end of the harvest period. Analysis of covariance was used for each
7065 treatment-control watershed pair. Results show monthly TSS loads from watersheds 1 (clearcut),
7066 2 (partial cut), and 3 (no-harvest) ranged from 0.4 kg km⁻² to above 10,000 kg km⁻², with a
7067 maximum in the spring months and minimum in the winter and late summer months similar to
7068 intra-annual trends in water yield. Road construction in both watersheds did not result in
7069 statistically significant impacts on monthly sediment loads in either treated watershed during the
7070 immediate or recovery time intervals. A significant and immediate impact of harvest on monthly
7071 sediment loads in the clear-cut watershed ($p = 0.00011$), and a marginally significant impact of
7072 harvest on monthly sediment loads in the partial-cut ($p = 0.081$) were observed. Total sediment
7073 load from the clearcut over the immediate harvest interval exceeded predicted load by 152%
7074 (6,791 kg km⁻²); however, individual monthly loads varied around this amount. The largest
7075 increases in percentage and magnitude occurred during snowmelt months, namely April 2002

7076 (560%, 2,958 kg km⁻²) and May 2002 (171%, 3,394 kg km⁻²). Neither treatment showed a
7077 statistical difference in TSS during the recovery time (clearcut: p = 0.2336; partial-cut: p =
7078 0,1739) compared to calibration loads (pre-treatments). The authors conclude that best
7079 management practices for road construction, including improvement of existing roads, did not
7080 produce significant changes in TSS. Significant changes in TSS only occurred immediately after
7081 harvest. However, after one year, the TS load became statistically indistinguishable from the
7082 control.

7083

7084 **Harvest effects on Instream light**

7085

7086 Kaylor et al., 2017

7087

7088 Kaylor, M.J., Warren, D.R., Kiffney, P.M., 2017. Long-term effects of riparian forest harvest on
7089 light in Pacific Northwest (USA) streams. *Freshwater Science* 36, 1–13.
7090 <https://doi.org/10.1086/690624>

7091

7092 The purpose of this study was to evaluate relationships between riparian forest stand age and
7093 stream light availability. The specific goals dealt with evaluating characteristics of late-
7094 successional forest light regimes, and whether canopy openness and light differed between
7095 streams flowing through harvested units and late-successional forest units. This study took place
7096 at the HJ Andrews Experimental Forest in the Cascade Mountain, Oregon. Approximately 11.5
7097 km of stream length were sampled in the McCrae Basin which consists mostly of old-growth
7098 forests Douglas-fir forests with small patch clear cuts. All treatment sites were harvested within
7099 50 to 60 years before the study. Clearing up to both stream banks occurred at two of seven
7100 treated sites and clearing up to one bank occurred on all other treated sites. Stream bank-full
7101 width, wetted width, canopy openness, % red alder, and estimated photosynthetically active
7102 radiation (PAR) were quantified at 25-m intervals to evaluate relationships between channel and
7103 riparian characteristics and stream light. Results from this study show mean estimated PAR
7104 reaching the streams was lower in the recovering harvested units (50-year post-treatment) than
7105 in up and downstream reaches bordered by old growth for all comparisons (n=14), while only 6
7106 were significant (p<0.05). All in all, old growth reaches averaged 1.7 times greater PAR values
7107 than in nearby harvested units with the greatest differences occurring when harvest was
7108 implemented on both banks. Mean canopy openness was higher in late-successional forests (>
7109 300 years old) than in young second growth forests (30–100-year-old forests), 18% and 8.7%
7110 respectively. Results also indicate the relationship between canopy openness and PAR was
7111 stronger at the reach scale than at individual locations with mean canopy openness explaining
7112 78% of the variance in mean PAR estimates. The researchers also conducted a review of
7113 available literature of studies that contained information on the effects of Northwest Douglas-fir
7114 forest growth dynamics on canopy cover and light availability. The researchers concluded from

7115 this review that canopy closure, and thus lower light availability, occurs approximately 30 years
7116 after growth and maintained until after 100 years of growth when the canopy structure begins to
7117 open and produce gaps. Altogether, this study suggests stream light regimes are affected by
7118 initial canopy removal and subsequent recovery. Depending on forest type, dominant species and
7119 the age of the stand, different stages of stand development may reflect complex overstory
7120 structures allowing variable levels of light to the stream.

7121

7122 **Stream Temperatures**

7123

7124 Kibler et al., 2013

7125

7126 Kibler, K.M., Skaugset, A., Ganio, L.M., Huso, M.M., 2013. Effect of contemporary forest
7127 harvesting practices on headwater stream temperatures: Initial response of the Hinkle Creek
7128 catchment, Pacific Northwest, USA. *Forest Ecology and Management* 310, 680–691.

7129 <https://doi.org/10.1016/j.foreco.2013.09.009>

7130

7131 The purpose of this study was to investigate the effects of contemporary forest harvesting
7132 practices on headwater stream temperatures using a BACI design. This study was conducted as
7133 part of the Hinkle Creek paired Watershed Study (HCPWS). This study consisted of a nested,
7134 paired watershed study in which harvesting treatments in accordance with the Oregon Forest
7135 Practices Act (FPA) were applied to four headwater catchments in southern Oregon. Oregon FPA
7136 does not require retention of fixed-width buffer strips adjacent to non-fish-bearing streams. Thus,
7137 as a part of the harvest activities, fixed-width buffer strips containing merchantable overstory
7138 conifers were not left adjacent to the non-fish-bearing streams. Clearcut harvest took place
7139 between August 2005 and May 2006. Streamflow and temperature were measured at 8 locations
7140 within the basin from autumn 2002 until autumn of 2006 giving 3 years of pre-harvest data and
7141 <1 year of post-harvest data. Treatment and reference catchments were paired based on similarity
7142 in catchment area, aspect, stream orientation, stream length, and discharge. Significant
7143 differences between pre- and post-harvest daily max temperature measurements were detected
7144 across all sites, however, magnitude and direction of changes were inconsistent. Results for daily
7145 mean maximum stream temperatures show a variable response across all four harvested streams
7146 ranging from 1.5°C cooler to 1.1°C warmer relative to pre-harvest years. No statistically
7147 significant changes in max, mean, or minimum daily stream temperatures to timber harvest were
7148 observed. The authors suggest possible explanations for lack of consistent temperature increases
7149 to shading provided by logging slash. Interestingly, statistically significant changes to
7150 relationship between treatment and reference site pairs with respect to minimum and mean
7151 stream temperatures resulted in decreased minimum daily stream temperatures on days where
7152 high temperatures were observed in reference streams. At one treatment site, mean minimum
7153 temperatures across the warm season decreased 1.9°C relative to pre-harvest years, and the

7154 minimum temperature on the warmest day decreased by 2.8°C relative to pre-harvest years.
7155 Except for one treatment-reference pair, highly significant changes to slope and intercept
7156 parameters of minimum daily stream temperatures were detected for each stream pair ($p < 0.001$).
7157 The authors suggest decreases in daily minimum stream temperature is a likely consequence of
7158 timber harvest.

7159

7160 **Shade and Stream temperature**

7161

7162 Cupp & Lofgren, 2014

7163

7164 Cupp, C.E. & Lofgren, T.J. (2014). Effectiveness of riparian management zone prescriptions in
7165 protecting and maintaining shade and water temperature in forested streams of Eastern
7166 Washington. Cooperative Monitoring Evaluation and Research Report CMER 02-212.
7167 Washington State Forest Practices Adaptive Management Program. Washington Department of
7168 Natural Resources, Olympia, WA.

7169

7170 The purpose of this study was to assess the percent reduction in canopy cover, and the response
7171 in stream temperatures following riparian timber harvest under the “all available shade” rule
7172 (ASR), and the standard rule (SR) in eastern Washington. The ASR is applied to areas in the Bull
7173 Trout Habitat Overlay (BTO; map of bull trout habitat) that requires retention of all available
7174 shade within 75 feet of the stream. Under the standard shade rule (SR) some harvest is allowed
7175 within the 75-foot buffer depending on elevation and pre-harvest canopy cover. The primary
7176 objectives of this study were to (1) Quantify and compare differences in post-harvest canopy
7177 closure between the SR and the ASR riparian prescriptions of eastern Washington; and (2)
7178 Quantify and compare differences in stream temperature effects of the two riparian prescriptions:
7179 the SR and the ASR. This study was conducted at 30 sites in eastern Washington. Sites were
7180 between 65-100 years old and were situated along second to fourth order streams with harvest-
7181 regenerated or fire-regenerated forests. Reference reaches were located upstream from treatment
7182 reaches where harvest was applied. Eighteen sites were located on state owned and managed
7183 forests and 12 sites were located on private industrial forests. Prior to harvest treatments, canopy
7184 closure measurements ranged from 89% to 97%, with a mean of 93%. The riparian management
7185 zone (RMZ) consists of three zones: The core zone is nearest to the edge of the stream and
7186 extends out 30 feet horizontally from the bankfull edge or outer edge of the channel migration
7187 zone (CMZ), whichever is greater. The inner zone is situated immediately outside of the core
7188 zone. For streams with a bankfull width of less than or equal to 15 feet wide, the inner zone
7189 width is 45 feet wide. All streams assessed in this study were less than or equal to 15 feet wide.
7190 The outer zone of the RMZ is the zone furthest from the water and its width varies according to
7191 stream width and site class for the land. The specific site class (a measure of site productivity) at
7192 each treatment site would vary the outer zone width from 0 to 55 feet wide. Seven sites had up to

7193 four years pre-harvest temperature data with only two years post-harvest data. Nine sites had
7194 three years pre-harvest data and one site had only one year pre-harvest data. The remaining 13
7195 sites had two years pre-harvest data. Following harvest treatments, all 30 sites had at least two
7196 years post-harvest temperature data collection, although 21 of the 30 sites had at least three years
7197 post-harvest monitoring. Data collection included twice hourly stream and air temperature data
7198 during each sample period. Canopy, shade, riparian, and channel data were collected during the
7199 first-year pre-harvest and the first year post-harvest. Stream temperature data were collected at
7200 30-minute intervals between 1 July and 15 September for a total of 77 days each year a site was
7201 investigated. Stream canopy closure and shade were quantified at 75-ft intervals within each
7202 reach using a hand-held densiometer (for canopy closure measurements) and a self-leveling
7203 fisheye lens digital camera (for shade measurements). A t-test was used to evaluate differences in
7204 pre-harvest canopy cover between reference and treatment reaches, and between ASR and SR
7205 sites. A correlation analysis between post-harvest change in shade and the descriptive riparian
7206 and channel values (e.g., trees per acre, basal area, channel gradient, etc.) was also used to
7207 examine possible factors that may control post-harvest changes in shade. A linear mixed effects
7208 model was used to quantify and compare differences in daily max stream temperatures (DMAX)
7209 between no harvest, ASR and SR prescriptions. Results showed post-harvest shade values
7210 decreased in SR sites (mean effect of -2.8%, $p = 0.002$), as did the canopy closure values (mean
7211 effect of -4.5%, $p < 0.001$). Shade and canopy closure values did not significantly change in the
7212 treatment reaches of the ASR sites. Mean shade reduction in the SR treatment sites exceeded the
7213 mean shade reduction in the ASR sites by 3%. Canopy closure reduction was also greater in the
7214 SR sites than in the ASR sites by a mean of 4%. Specifically, the mean shade reduction in ASR
7215 sites was 1% with a maximum reduction of 4%. The mean reduction of shade in the SR sites was
7216 4% with a maximum reduction of 10%. Mean shade contribution of upland trees (trees outside of
7217 the RMZ) per study site was calculated as $< 1\%$. Shade reduction levels did not differ between
7218 the sites receiving RMZ-harvest only and the sites receiving standard operational upland harvest.
7219 Site seasonal means of daily maximum stream temperature treatment responses in the first two
7220 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
7221 the SR reaches. Site seasonal mean post-harvest background responses in reference reaches
7222 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
7223 stream temperature increased $0.16\text{ }^{\circ}\text{C}$ in the SR harvest reaches, whereas stream temperatures in
7224 both the ASR sites and in the no-harvest reference reaches increased on average by $0.02\text{ }^{\circ}\text{C}$.
7225 Seasonal mean stream temperature responses of up to $0.5\text{ }^{\circ}\text{C}$ in the no-harvest references were
7226 common during the post-harvest test period. Sample period means of daily maximum
7227 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
7228 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
7229 authors interpret these results as evidence that temperature effects of the SR, and ASR were
7230 similar to reference conditions along sampled reaches for small streams in the mixed fir zone
7231 mid-successional forests of eastern Washington. Further, that processes not directly related to
7232 canopy cover alteration over streams may be primarily responsible for the small variations
7233 observed in stream temperatures following harvest.

7234

7235

7236 Ehinger et al., 2021

7237

7238 Ehinger, W.J., W.D. Bretherton, S.M. Estrella, G. Stewart, D.E. Schuett-Hames, and S.A. Nelson.
7239 2021. Effectiveness of Forest Practices Buffer Prescriptions on Perennial Non-fish-bearing
7240 Streams on Marine Sedimentary Lithologies in Western Washington. Cooperative Monitoring,
7241 Evaluation, and Research Committee Report CMER 2021.08.24, Washington State Forest
7242 Practices Adaptive Management Program, Washington Department of Natural Resources,
7243 Olympia, WA.

7244

7245 The purpose of this study was to assess the effectiveness of riparian management zone
7246 prescriptions in maintaining functions and processes in headwater perennial, non-fish-bearing
7247 streams in incompetent (easily eroded) marine sedimentary lithologies in western Washington.
7248 Specifically, this study used a multiple before after control impact (MBACI) design to compare
7249 unharvested reference sites to sites harvested under the western Washington Forest Practices for
7250 non-fish-bearing streams to assess the effects of these rules on riparian vegetation and wood
7251 recruitment, canopy closure and stream temperature, stream discharge and downstream transport
7252 of suspended sediment and nitrogen, and benthic macroinvertebrates. The Forest Practices rules
7253 for non-fish-bearing streams in the study area includes clearcut harvest with a two-sided 50-foot-
7254 wide riparian buffer along at least 50% of the riparian management zone, including buffers
7255 prescribed for sensitive sites and unstable slopes. Ten study sites were chosen with first-, second-
7256 , and third-order non-fish-bearing streams. Data was collected for 1-2 years of pre-harvest,
7257 during the harvest period (2012 – 2014), and at least 2 years post-harvest at all sites. Because of
7258 unstable slopes, total buffer area was 18 to 163% greater than the 50-foot-buffer. This resulted in
7259 4 different buffer types 1) Buffers encompassing the full width (50 feet), 2) <50ft buffers, 3)
7260 Unbuffered, harvested to the edge of the channel, and 4) Reference sites in unharvested forests.
7261 Because of the separation into multiple treatments, sample sizes became small and unbalanced.
7262 Thus, no statistical analyses were conducted, and only descriptive statistics were applied for
7263 changes in stand structure and wood loading. Density decreased by 33 and 51% and basal area
7264 by 26 and 49% in the full and <50ft buffers, respectively, with high variability among sites.
7265 Nearly all trees were removed from Unbuffered sites during harvest (>99% of basal area). In the
7266 reference plots, cumulative post-harvest mortality during the 3-year post-harvest interval was
7267 only 6.5% of live density. In contrast, mean post-harvest mortality in the full buffer sites and the
7268 <50 ft buffer sites were 31 and 25% of density, respectively. However, there was considerable
7269 variation in mortality among sites exceeding 65% in two full buffer treatment sites. Windthrow
7270 and physical damage from falling trees accounted for ~75% of mortality in the full and <50 ft
7271 buffers. In contrast to the treated sites, <10% of trees died due to wind or physical damage in the
7272 reference sites. There was little post-harvest large wood input in reference sites: an average of
7273 4.3 pieces and 0.34 m³ of combined in- and over-channel volume per 100 m of channel. In
7274 contrast, the full buffer sites and <50 ft buffer sites received an average of 23 and 10 pieces/100

7275 m and 2.3 and 0.7 m³/100 m of large wood, respectively. The majority of recruited large wood
7276 pieces had stems with roots attached (SWRW); 60, 70, and 100% in the reference, full buffer,
7277 and <50 ft buffer types, respectively. Pre-harvest channel large wood loading ranged from 55.8 to
7278 111 pieces/100 m and from 9.8 to 25.2 m³/100 m among buffer types. Piece counts remained
7279 stable in the reference sites through year 3 post-harvest, increased in the full buffer and
7280 unbuffered sites (8 and 13%, respectively), and decreased in the <50 ft buffers (15%). For effects
7281 of treatment on shade, data was analyzed with generalized linear mixed-effects models. For
7282 effects of treatment on stream temperature, data was analyzed for the seven-day average in a
7283 linear-mixed-effects model analysis of variance. Mean canopy closure decreased in the treatment
7284 sites from 97% in the pre-harvest period to 75%, 68%, and 69% in the first, second, and third
7285 post-harvest years, respectively, and was related to the proportion of stream buffered and to post-
7286 harvest windthrow within the buffer. The seven-day average temperature response increased by
7287 0.6°C, 0.6°C, and 0.3°C in the first, second, and third post-harvest years, respectively. During
7288 and after harvest, mean monthly water temperatures were higher, but equaled or exceeded
7289 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
7290 three REF sites exceeded 15°C during the study. Predictive models could not be fitted to the
7291 temperature data for statistical analysis. Results for changes in nutrient concentrations post-
7292 harvest were highly variable. Harvest treatment effects on nutrient concentrations, discharge, and
7293 suspended sediment export could not be calculated because prediction equations could not be
7294 developed.

7295

7296

7297 McIntyre et al., 2018

7298

7299 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D. Schuett-Hames, and T. Quinn
7300 (technical coordinators). 2018. Effectiveness of Experimental Riparian Buffers on Perennial
7301 Non-fish-bearing Streams on Competent Lithologies in Western Washington. Cooperative
7302 Monitoring, Evaluation and Research Report CMER 18-100, Washington State Forest Practices
7303 Adaptive Management Program, Washington Department of Natural Resources, Olympia, WA.

7304

7305 The purpose of the study was to evaluate the effectiveness of forest management prescriptions in
7306 maintaining aquatic conditions and processes for small non-fish-bearing (Type N) headwater
7307 stream basins underlain by competent “hard rock” lithologies (i.e., volcanic or igneous rock) in
7308 western Washington. Specifically, this study quantified and compared the effects of timber
7309 harvest adjacent to Type N streams on riparian stand structure and tree mortality, in stream wood
7310 loading and recruitment, stream temperature and canopy cover, stream discharge, turbidity, and
7311 suspended sediment export, nitrogen export, and response of stream associated amphibians. This
7312 study used a before-after control-impact (BACI) study design. This involved evaluation of four
7313 experimental treatments: (1) unharvested reference (n = 6), (2) 100% treatment (n = 4), a two-

7314 sided 50-ft riparian buffer along the entire Riparian Management Zone (RMZ), (2) FP treatment
7315 (n = 3), a two-sided 50-ft riparian buffer along at least 50% of the RMZ, consistent with the
7316 current Forest Practices buffer prescription for Type N streams, This treatment also included a
7317 circular buffer protecting the uppermost points of perennial flow (PIP), (3) 0% treatment (n = 4),
7318 clearcut to stream edge (no-buffer). The upland forests of all treatments were clearcut harvested.
7319 The study design included data collection for at least two years pre-harvest (2006 –2008), and
7320 three years of post-harvest data (2009 – 2011). Results for stand structure and tree mortality
7321 showed that in the RMZs, the proportional changes in stem count (dstems) and basal area (dBA)
7322 were similar for the reference (mean dstems: -11.8, SE 5.3; dBA: -6.9, SE 5.4) and 100% (mean
7323 dstems: -3.8, SE 5.9; dBA -6.7, SE 6.0) treatment. In contrast, the magnitude of decrease was
7324 significantly greater in the FPB (portion of FP containing trees; mean dstems: -29.6, SE 6.5; dBA
7325 124.4, SE 6.7) treatment than in either the reference or 100% treatment. The pattern was similar
7326 in the PIPs. 2 years post-harvest tree mortality was mostly (70%) attributed to wind/mechanical
7327 agents (pre-harvest wind/mechanical agent caused mortality was 70%). In the reference sites,
7328 trees that died post-harvest had smaller diameters (mean 10.3 in) and fewer came from the
7329 overstory crown class (59.0%) than the other treatments. In contrast, in the 100% and FPB
7330 treatments, ~70% of trees that died were from the overstory crown class and their mean
7331 diameters were 1 (11.2 in) and 2 (12.2 in) in greater than those in the reference sites,
7332 respectively. Results for wood recruitment and loading showed that tree fall rates were highly
7333 variable during the pre-harvest period between sites ranging from 0 to 239.9 trees/ha/yr. Large
7334 wood (LW) recruitment rates in the pre-harvest period were also highly variable ranging from 0
7335 to 121.6 pieces/ha/yr, along with recruitment volume (0-16.2 m³/ha/yr). 2 years post-harvest
7336 recruitment rates in the reference riparian management zones (RMZs) were lower and less
7337 variable (5.9 to 37.3 trees/ha/yr) than in buffer treatments. Tree fall rates for the 100% treatment
7338 ranged from 7.7 to 76.4 trees/ha/yr, and for the FPB treatments tree fall rates ranged from 4.2 to
7339 152.2 trees/ha/yr. Post-harvest LW recruitment volumes in reference RMZs were relatively low,
7340 ranging from 0.7 to 2.2 m³/ha/yr. Post-harvest LW recruitment volumes were generally higher
7341 and more variable in the 100% and FPB RMZs, ranging from 0.3 to 14.0 m³/ha/yr in the 100%
7342 treatment and 0 to 7.6 m³/ha/yr in the FPB. Because of the high variability between sites in all
7343 treatments the p values for comparisons between treatments were generally high ($p \geq 0.35$),
7344 except for the FPB vs. reference comparison for piece count which was nearly significant ($p =$
7345 0.13). The only significant differences were for the 0% treatments which had significantly lower
7346 LW recruitment by volume than the Reference RMZ ($P = 0.02$). For PIPs, LW recruitment in the
7347 100% treatment was over 12 times the reference rate by piece count ($P = 0.03$) and 30 times the
7348 reference rate by volume ($P = 0.04$). Recruitment in the FPB PIPs was also high, over nine times
7349 the reference rate by piece count ($P = 0.08$) and 18 times the reference rate by volume ($P = 0.11$).
7350 The amount of change in the number of LW pieces per meter from pre-harvest to post-harvest
7351 depended on treatment ($P < 0.01$). Analysis estimated the changes in 100%, FP and 0% treatments
7352 to be different from the change in the reference ($P < 0.001$, 0.03 and < 0.01 , respectively). The
7353 percentage of the stream channel length covered by newly recruited wood in the second post-
7354 harvest year ranged from 0 to 11% in the reference, 1 to 15% in the 100% treatment and 0 to
7355 10% in the FP treatment and was 0% in all four of the 0% treatments. The percent of stream
7356 channel covered by new wood differed between the 0% treatment and the reference ($P = 0.03$),

7357 100% ($P < 0.01$), and FP treatments ($P = 0.03$). Overall, the authors estimated a mean between-
7358 treatment increase of 60% (95% CI: 0–150%), 70% (95% CI: 0–190%) and 170% (95% CI:
7359 80–330%) in the number of SW pieces per stream meter in the 100%, FP and 0% treatments
7360 compared with the reference, respectively. Also, a between-treatment increase of 60% (95% CI:
7361 30–110%), 40% (95% CI: 0–100%) and 50% (95% CI: 10–90%) in the number of LW pieces
7362 per stream meter in the 100%, FP and 0% treatments compared with the reference, respectively.
7363 The authors conclude that windthrow was responsible for much of the increase in LW. However,
7364 they also posit that the timing and magnitude of wood inputs was inconsistent, resulting in
7365 considerable variability between and within sites, especially in the FP treatment. Results for
7366 shade response to treatments post-harvest was greatest in the 0% treatment than in either the
7367 100% or the FP treatment. Effective shade decreased to 77, 52, and 14% 2 years post-treatment,
7368 in the 100%, FP, and 0% buffer treatments, respectively. Canopy and Topographic Density
7369 (CTD), defined as the percentage of the photograph obscured by vegetation or topography
7370 decreased from an average of 95% pre-harvest to 86, 71, and 43% 2 years post-harvest in the
7371 100%, FP, and 0% buffer treatments, respectively. All were significantly lower than the reference
7372 (92% 2 years post-treatment). Results for stream temperature showed maximum daily water
7373 temperatures increased post-harvest in all but one of the harvested sites and was elevated over
7374 much of the year at most of the sites. Daily temperature response (TR) increased in late winter or
7375 early spring, reached a maximum in July–August and was still elevated well into the fall. This
7376 pattern was observed at most of the sites. For the Buffer Treatment locations, 94 of the 131
7377 calculated mean monthly temperature responses (MMTRs) were significant and 91 of these
7378 significant responses were positive. In comparison, only 52 of 156 MMTR values calculated for
7379 the reference sites were significant and these were nearly evenly split with 25 positive and 27
7380 negative responses. This strongly suggests that the pattern of post-harvest increases in daily
7381 maximum water temperature is real even though the magnitude of some of the individual
7382 MMTRs is relatively small ($< 0.5^{\circ}\text{C}$). Warming tended to be greatest in July or August with
7383 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
7384 in the 0% treatments. Post-harvest, Max7D (seven-day-average maximum stream temperature)
7385 was higher at 36 of the 40 locations within the harvest units across all 11 buffer treatment sites
7386 regardless of presence or absence of a buffer, buffer width, and longitudinal location along the
7387 stream. Relative to the unharvested sites, there were summertime temperature increases
7388 throughout the stream length and across all buffer treatment sites. The authors conclude that none
7389 of the buffer treatments were successful in preventing significant increases in maximum stream
7390 temperature. The generalizable conclusions made by the authors from this portion of the study
7391 are that 1) Buffer widths greater than 50 ft (15.2 m) are needed to prevent shade loss and (2)
7392 Maximum water temperature decreased below the harvest unit after flowing through
7393 approximately 100 m of intact forest but was still elevated compared to pre-harvest conditions.
7394 Results for nitrogen and phosphorus concentrations showed that post-harvest changes for total-N
7395 or total-P were not significant for any of the treatments relative to the Reference. The only
7396 significant difference detected within 2 years post-harvest was for nitrate-N concentration
7397 between the 0% buffer treatment and all other treatments. However, for annual export, total-N
7398 and nitrate-N export increased post-harvest at all sites, with the smallest increase in the 100%
7399 treatment and the largest in the 0% treatment. Compared to the reference sites, the GLMM

7400 analysis showed a relative increase in total-N export post-harvest of 5.52 ($P = 0.051$), 11.52 ($P =$
7401 0.0007), and 17.16 ($P < 0.0001$) $\text{kg ha}^{-1} \text{ yr}^{-1}$ in the 100%, FP, and 0% treatments. The GLMM
7402 analysis showed a relative increase in nitrate-N export post-harvest of 4.83 ($P = 0.048$), 10.24 (P
7403 $= 0.001$), and 15.35 ($P < 0.0001$) $\text{kg ha}^{-1} \text{ yr}^{-1}$ in the 100%, FP, and 0% treatments, respectively,
7404 only slightly less than the changes in total-N. Total-P export increased post-harvest by a similar
7405 magnitude in all treatments: 0.10 ($P = 0.006$), 0.13 ($P = 0.001$), and 0.09 ($P = 0.010$) $\text{kg ha}^{-1} \text{ yr}^{-1}$
7406 in the 100%, FP, and 0% treatments, respectively. The increase in N, total-N and nitrate-N, from
7407 the treatment watersheds post-harvest was strongly correlated with the increase in annual runoff
7408 ($R^2 = 0.970$ and 0.971 ; $P = 0.001$ and 0.001 , respectively) and with the proportion of the basin
7409 harvested ($R^2 = 0.854$ and 0.852 ; $P = 0.031$ and 0.031 , respectively). The correlation with the
7410 proportion of stream length buffered was weaker ($R^2 = 0.761$ and 0.772 ; $P < 0.079$ and 0.072 ,
7411 respectively). In contrast, total-P export was uncorrelated with all three variables. Overall, the
7412 authors concluded that mean flow-weighted concentration of total-N and nitrate-N increased at
7413 all buffer treatment sites post-harvest, however the magnitude was variable and significant only
7414 for the 0% treatment. However, the export of total-N increased in the FP and 0% treatments and
7415 nitrate-N increased in all buffer treatments. Increases in N export was correlated with increased
7416 stream discharge and the proportion of the site that was harvested. Pre-harvest total-P
7417 concentration was low and remained so post-harvest, although P export increased slightly post-
7418 harvest in all treatments due to the increase in discharge. Results for changes in water turbidity
7419 and suspended sediment concentrations (SSC) showed both turbidity and SSC increased with
7420 increasing discharge during storm events but then rapidly fell off. Analysis of treatment effects
7421 revealed no significant effects of harvest and no clear pattern regarding the relative effectiveness
7422 of buffer treatments at mitigating the effects of clearcut harvests on suspended sediment export
7423 (SSE). The general conclusions made by the authors were that all sites appeared to be supply
7424 limited both pre- and post-harvest. Results for litterfall input showed a decrease in TOTAL
7425 litterfall input in the FP ($P = 0.0034$) and 0% ($P = 0.0001$) treatments between pre- and post-
7426 treatment periods. LEAF litterfall (deciduous and conifer leaves combined) input decreased in
7427 the FP ($P = 0.0114$) and 0% ($P < 0.0001$) treatments in the post-treatment period. In addition,
7428 CONIF (conifer needles and scales) litterfall input decreased in the FP ($P = 0.0437$) and 0% (P
7429 < 0.0001) treatments, DECID (deciduous leaves) in the 0% ($P < 0.0001$) treatment, WOOD (twigs
7430 and cones) in the FP ($P = 0.0044$) and 0% ($P = 0.0153$) treatments, and MISC (e.g., moss and
7431 flowers) in the 0% ($P = 0.0422$) treatment. Results for comparison of the post-harvest effects
7432 between treatments showed LEAF litterfall input decreased in the 0% treatment relative to the
7433 reference ($P = 0.0040$), 100% ($P = 0.0008$), and FP ($P = 0.0267$) treatments. Likewise, there was
7434 a decrease in DECID litterfall input in the 0% treatment relative to the Reference ($P = 0.0001$),
7435 100% ($P < 0.0001$), and FP ($P = 0.0015$) treatments. Results for detritus with comparisons
7436 between the pre- and post-treatment periods showed an increase in TOTAL detritus export in the
7437 100% treatment ($P = 0.0051$) and a decrease in the 0% treatment ($P = 0.0046$; Table 12-9).
7438 Likewise, there was an increase in CPOM, WOOD, MISC, and FPOM detritus export in the
7439 100% treatment ($P < 0.05$), but a decrease in the 0% treatment ($P < 0.05$) The authors for this
7440 portion of the study conclude that overall, total litterfall input was slightly higher after harvest in
7441 the 100% treatment, lower in the FP treatment and lowest in the 0% treatment; however,
7442 statistical differences were only detected for deciduous inputs between the 0% treatment and the

7443 other treatments. Total detritus export decreased in the 0% treatment relative to the reference,
7444 and in the FP and 0% treatments relative to the 100% treatment.

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7447 McIntyre et al., 2021

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7449 McIntyre, A.P., M.P. Hayes, W.J. Ehinger, S.M. Estrella, D.E. Schuett-Hames, R. Ojala-Barbour,
7450 G. Stewart and T. Quinn (technical coordinators). 2021. Effectiveness of experimental riparian
7451 buffers on perennial non-fish-bearing streams on competent lithologies in western Washington –
7452 Phase 2 (9 years after harvest). Cooperative Monitoring, Evaluation and Research Report CMER
7453 2021.07.27, Washington State Forest Practices Adaptive Management Program, Washington
7454 Department of Natural Resources, Olympia, WA.

7455

7456 This study was a follow-up study to the hard-rock Phase 1 study (McIntyre et al., 2018) to assess
7457 changes over longer time periods (up to 9 years post-harvest). The purpose of the study was to
7458 evaluate the effectiveness of forest management prescriptions in maintaining aquatic conditions
7459 and processes for small non-fish-bearing (Type N) headwater stream basins underlain by
7460 competent “hard rock” lithologies (i.e., volcanic or igneous rock) in western Washington.
7461 Specifically, this study quantified and compared the effects of timber harvest adjacent to Type N
7462 streams on riparian stand structure and tree mortality, in stream wood loading and recruitment,
7463 stream temperature and canopy cover, stream discharge, turbidity, and suspended sediment
7464 export, nitrogen export, and response of stream associated amphibians. This study used a before-
7465 after control-impact (BACI) study design. This involved evaluation of four experimental
7466 treatments: (1) unharvested reference (n = 6), (2) 100% treatment (n = 4), a two-sided 50-ft
7467 riparian buffer along the entire Riparian Management Zone (RMZ), (2) FP treatment (n = 3), a
7468 two-sided 50-ft riparian buffer along at least 50% of the RMZ, consistent with the current Forest
7469 Practices buffer prescription for Type N streams, (3) 0% treatment (n = 4), clearcut to stream
7470 edge (no-buffer). The upland forests of all treatments were clearcut harvested. The study design
7471 included data collection for at least two years pre-harvest (2006 –2008), and up to nine years
7472 post-harvest from 2009 (harvest began in 2008) until 2016 or 2017 depending on the variable
7473 (e.g., wood loading, shade, etc.). Results for stand structure showed that in the buffered portions
7474 of the FP treatments (FPB) density, basal area and relative density (RD) decreased by 59%, 55%
7475 and 54%, respectively, 8 years after harvest. For the same variables, reductions in the 100%
7476 RMZs were 30%, 14%, and 17%, respectively. In contrast, stand structure in the reference RMZs
7477 was more stable, with a 17% decrease in density and little change in basal area or RD. Change in
7478 live basal area did not differ statistically between 100% and REF RMZs for any time interval
7479 although the differences increased over time. The FPB–REF contrast was not significant in the
7480 first interval (years 1 and 2 post-harvest), but it was in subsequent intervals (5- and 8-years post-
7481 harvest) as the magnitude of change in FPB RMZs increased over time. The FPB–100% contrast

7482 was not significant until the last interval when basal area stabilized in the 100% treatment but
7483 continued to decline in FPB. Between treatment comparison of cumulative change in live basal
7484 area (m²/ha) between the 100% treatment and the Reference was -2.9 (CI: -16.9, 11.0), -6.0 (CI:
7485 -20.0, 8.0), and -6.8 (CI -20.8, 7.1) for the first-, second-, and third-time intervals respectively
7486 (none were significant). Comparison between the FPB and Reference were -10.2 (CI: -25.5, 5.2),
7487 -16.1 (CI: -31.4, -0.8), and -21.1 (CI: -36.4, -5.8) for the first-, second-, and third-time intervals
7488 respectively (differences for intervals 2 and 3 were significant). For tree mortality, results
7489 showed that by year 8 post-harvest mortality as a percentage of pre-harvest basal area was lower
7490 in the reference (16.1%) than in the 100% (24.3%) and FPB (50.8%). The FPB–Reference
7491 contrast was not significant 2 years post-harvest, but it was at 5- and 8-years post-harvest as
7492 mortality in FPB increased relative to the reference. The contrast between the 100% and Ref
7493 were not significant for any time interval 8 years post-harvest. The contrasts 100% vs. REF and
7494 FPB vs. 100%—were not significant for any time interval. This may have been because of the
7495 high variability in the data. There was a temporal pattern to mortality in 100% and FPB RMZs.
7496 Annual rates of mortality as percentage of live basal area and density were highest in the first
7497 two years after harvest, then decreased. Wind/physical damage was the primary cause of
7498 mortality. In the 100% treatment it accounted for 78% and 90% of the loss of basal area and
7499 density, respectively; in FPB it accounted for 78% and 65% of the loss. Wind accounted for a
7500 smaller proportion of mortality in reference RMZ (52%). Large wood recruitment to the channel
7501 was greater in the 100% and FPB RMZs than in the reference for each pre- to post-harvest time
7502 interval. Eight years post-harvest mean recruitment of large wood volume was two to nearly
7503 three times greater in 100% and FPB RMZs than in the references. Large wood recruitment rates
7504 were greatest during the first two years, then decreased. However, these differences were not
7505 significant between any treatment comparisons, again, likely due to the high variability in the
7506 data. Mean large wood loading differed significantly between treatments in the magnitude of
7507 change overtime. Results showed a 66% (P < 0.001), 44% (P = 0.05) and 47% (P = 0.01) increase
7508 in mean large wood density in the 100%, FP and 0% treatments, respectively, in the first 2 years
7509 post-harvest compared with the pre-harvest period and after controlling for temporal changes in
7510 the references. Five years post-treatment the mean LW density in the FP continued to increase
7511 42% (P = 0.08), and again 8 years post-treatment (41%; P = 0.09). Results for canopy cover
7512 showed that riparian cover declined after harvest in all buffer treatments reaching a minimum
7513 around 4 years post-harvest. The treatments, ranked from least to most change, were REF, 100%,
7514 FP, and 0% for all metrics and across all years. Effective shade results showed decreases of 11,
7515 36, and 74 percent in the 100%, FP, and 0% treatments, respectively. Significant post-harvest
7516 decreases were noted for all treatments and all years. Results for stream temperature showed that
7517 within treatment mean post–pre-harvest difference in the REF treatment never exceeded 1.0°C.
7518 In contrast, the mean within treatment difference in the 100% treatment was 2.4°C in 2009 (Post-
7519 harvest year 1) but never exceeded 1.0°C in later years. The mean difference in the FP treatment
7520 exceeded 1.0°C immediately after harvest then again in 2014–2016 (post-harvest years 6–9)
7521 while in the 0% treatment the mean difference was 5.3°C initially, then decreased over time to
7522 near, but never below, 0.9°C. Stream temperature increased post-harvest at most locations within
7523 all 12 harvested sites and remained elevated in the FP and 0% treatments over much of the nine
7524 years post-harvest. Temperature responses varied by treatment, by season, and over the years. In

7525 three out of the first four post-harvest years there was, at least, a weak ($r < -0.48$) negative
7526 correlation between July monthly mean temperature response (MMTR) and the change in
7527 riparian cover based on each of the four shade metrics. The correlation was generally weaker
7528 ($-0.4 < r$ and $P > 0.10$) after post-harvest year 4, except for post-harvest year 9 ($-0.6 < r < -0.4$).
7529 However, there were only eight data pairs available for Post 9, compared to ten to twelve for the
7530 other years, which affected the correlation coefficient and p-value. However, there was a great
7531 deal of variability in the correlation coefficient of July MMTR with shade across post-harvest
7532 years among sites and treatments with some sites showing negative correlations and others
7533 positive for some treatments in some years. Considering site characteristics, aspect showed an
7534 influence on stream temperature response. In the first five post-harvest years and in Post 7 the
7535 highest MMTR in each treatment was nearly always the site with a southern (SE or SW) aspect.
7536 No significant correlation between July MMTR and either mean July discharge or the post-
7537 harvest difference in discharge was observed. For the effects of harvest on stream discharge,
7538 cumulative results of regression analysis (forward and reverse regression approaches) indicated
7539 that discharge did increase following harvest. In relative terms, discharge increased by 5-7% on
7540 average in the 100% treatments while increasing between 26-66% in the FP and 0% treatments.
7541 The change in discharge following harvest was also affected by climate, weather, and physical
7542 hydrology of the watershed. In all basins, discharge varied with precipitation, but this was a
7543 complex relationship showing lag time between precipitation events and discharge rate response
7544 in some watersheds. This indicated a potential relationship with physical hydrology at some
7545 watersheds. Results for water turbidity and suspended sediment export (SSE) were stochastic in
7546 nature and the relationships between SSE export and treatment effects were not strong enough to
7547 confidently draw conclusions. Results for harvest effects on total nitrogen export following a
7548 generalized linear mixed effects model, however, showed significant ($P < 0.05$) treatment effects
7549 were present in the FP treatment post-harvest and in the 0% treatment in the post-harvest (2-
7550 years immediately following harvest) and extended periods (2015 – 2017; 7 and 8 years post-
7551 harvest) relative to the reference sites, but there were no significant differences in total-N export
7552 between the treatments. Analysis showed an increase in total-N export of 5.73 ($P = 0.121$), 10.85
7553 ($P = 0.006$), and 15.94 ($P = 0.000$) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments,
7554 respectively, and of 6.20 ($P = 0.095$), 5.34 ($P = 0.147$), and 8.49 ($P = 0.026$) kg/ha/yr in the
7555 extended period. Results for nitrate-N export showed changes similar to but slightly less than
7556 those seen in the total-N analysis with a relative increase in nitrate-N export of 4.79 ($P = 0.123$),
7557 9.63 ($P = 0.004$), and 14.41 ($P < 0.001$) kg/ha/yr post-harvest in the 100%, FP, and 0% treatments,
7558 respectively. None of the changes in the extended period were significant. However, the authors
7559 note that there was high variability in the data for the extended period and nitrate-N export only
7560 returned to pre-harvest levels in one watershed. The increase in total-N and nitrate-N export
7561 tended to be highest during the high flow months in the fall and early winter. The authors
7562 conclude that the 100% treatment was generally the most effective in minimizing changes from
7563 pre-harvest conditions, the FP was intermediate, and the 0% treatment was least effective. The
7564 collective effects of timber harvest were most apparent in the 0% treatment in the two years
7565 immediately post-harvest.

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7567 **LW**

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7569 Johnston et al., 2011

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7571 Johnston, N. T., Bird, S. A., Hogan, D. L., & MacIsaac, E. A. (2011). Mechanisms and source
7572 distances for the input of large woody debris to forested streams in British Columbia, Canada.
7573 Canadian journal of forest research, 41(11), 2231-2246. <https://doi.org/10.1139/x11-110>

7574

7575 The purpose of this study was to determine whether the processes and source distances from
7576 which LW entered streams differed among channel types and sizes, to describe LW source
7577 distance curves for a wide range of undisturbed stream and forest types, and to characterize the
7578 relationships between LW input mechanism, source distance, and piece size. Input processes,
7579 source distances, and physical characteristics of approximately 2100 pieces of LW at 51
7580 anthropogenically undisturbed stream reaches throughout south and central British Columbia
7581 were determined. Large wood (LW) was defined in this study as pieces within or suspended
7582 above the active channel, with a minimum length of 1 m. and capable of inducing sediment scour
7583 or deposition. A delivery mechanism was assigned to each LW piece, when it could be
7584 determined, as bank erosion, landslide, windthrow of live trees, stem snap, or standing dead tree
7585 fall. Differences in the frequencies of count data among LW delivery mechanisms, LW positions,
7586 or LWD functions were assessed using chi-square tests. The effects of channel (type, width) and
7587 forest (maximum tree height) characteristics on the proportions of LWD pieces entering the
7588 channel by a given input mechanism were examined using ANCOVA. Channel type for this
7589 study was grouped into 3 categories; riffle-pool (RP), cascade-pool (CP), and step-pool (SP).
7590 Results showed that tree mortality was the most common entry mechanism at all channel types
7591 and width categories and accounted for 65% of all LW pieces sampled. Both channel and
7592 riparian forest characteristics influenced the proportion of LW pieces that entered streams by tree
7593 mortality ($P < 0.05$) but did not vary significantly among channel types ($P = 0.13$). The
7594 proportion of LW pieces recruited by tree mortality decreased with increasing channel width and
7595 with increasing maximum tree height. Bank erosion inputs accounted for 20%–25% of all LW
7596 pieces at the lower-gradient RP and CP sites but were much less important at the SP channels.
7597 Erosion inputs increased with increasing stream size within all channel types ($P = 0.0004$). Wind-
7598 induced inputs (windthrow and stem snap) accounted for 13%–20% of inputs over the channel
7599 types and generally increased in importance in the smaller channels. The proportion of LW
7600 recruited to the stream by stem breakage increased with increasing tree height ($P < 0.0001$) and
7601 varied among channel types ($P = 0.040$), being about twice as prevalent at SP channels as
7602 elsewhere. Landslide inputs of LWD were a minor delivery mechanism. There was considerable
7603 variability in distances from which LW entered the stream. However, based on the cumulative
7604 distributions over sites, 90% of the LW pieces or volume entering the channels originated within
7605 18 m of the stream in 90% of all cases (between 2 and 23 m in all cases). The distances from
7606 which LW entered the streams differed significantly among the various input mechanisms ($P <$

7607 0.001), the rank ordering of the mean source distances being bank erosion < tree mortality <
7608 stem breakage < windthrow < landslides. Bank erosion and landslides delivered the largest LW
7609 pieces and tree mortality and stem breakage the smallest. In general, source distances increased
7610 with increasing tree height, with the effect being stronger in the steeper channel types and
7611 weaker in the wider channels for LW pieces and volume. However, all two-way interactions
7612 among variables were significant implying that the mechanisms through which vegetation and
7613 stream geomorphology influenced LW source distance were complex. Maximum tree height in
7614 the adjacent forest accounted for the greatest variance in in-stream LW source distance for all
7615 models.

7616

7617 **Nutrient**

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7619 Deval et al., 2021

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7621 Deval, C., Brooks, E. S., Gravelle, J. A., Link, T. E., Dobre, M., & Elliot, W. J. (2021). Long-
7622 term response in nutrient load from commercial forest management operations in a mountainous
7623 watershed. *Forest Ecology and Management*, 494, 119312.
7624 <https://doi.org/10.1016/j.geomorph.2013.11.028>

7625

7626 The purpose of this study was to quantify and compare the differences in nitrogen and
7627 phosphorus concentrations and loads between pre-disturbance, post road construction (post-
7628 road), post experimental harvest (PH-I), and post operational harvest (PH-II) from both a
7629 hydrological yield and nutrient concentration perspective. This study was carried out in the Mica
7630 Creek Experimental Watershed in Northern Idaho. For this analysis time periods have been
7631 broken into four distinct phases: Pre-disturbance (1992–1997), Post-road (1997–2001),
7632 experimental-harvest Phase I (PH-I) (2001–2007), and operational sequential harvest Phase II
7633 (PH-II) when the extent and frequency of harvests increased (2007–2016). PH-I represents an
7634 experimental treatment phase during which harvest activities were experimentally controlled
7635 (only upstream headwater watersheds were harvested and mature vegetation removal ranged
7636 between 24% and 47%) followed by site management operations including broadcast burning
7637 and replanting. PH-II represents the post-experimental phase where the study area transitioned to
7638 operational treatments that consisted of additional road construction and timber harvest, with site
7639 management operations including pile burning and competition release herbicide application.
7640 During this operational phase, the mature vegetation removal in the upstream and cumulative
7641 downstream watersheds ranged between 36% and 50% and 17–28%, respectively. Monthly
7642 annual grab samples of stream water were collected from seven flumes over the course of 25
7643 years (from pre- to post-treatments). The samples were analyzed for six parameters, specifically
7644 nitrate + nitrite (NO₃ + NO₂), total Kjeldhal nitrogen (TKN), total ammonia nitrogen (TAN)
7645 containing un-ionized (NH₃) and ionized (NH₄⁺) ammonia, total nitrogen (TN), total

7646 phosphorus (TP), and orthophosphate (OP). This study used a before-after, control-impact paired
7647 series design (BACIPS) to evaluate direct and cumulative effects of forest management practices
7648 on stream nutrient concentrations in paired and nested watersheds. Results for long-term trends
7649 in stream flow showed a statistically significant increasing trend in all the watersheds during the
7650 fall and winter seasons. Significant increases in summer streamflow only occurred in the control
7651 watersheds. There were minimal changes in TKN concentration with a slight observed reduction
7652 in long-term TKN loads. Overall, the cumulative mean TAN loads from all watersheds did not
7653 show large variations with sequential varying treatments over time. In contrast to TAN, there was
7654 a significant response in NO₃ + NO₂ following timber harvest. The response in NO₃ + NO₂
7655 concentrations was negligible at all treatment sites following the road construction activities.
7656 However, NO₃ + NO₂ concentrations during the PH-I period increased significantly ($p < 0.001$)
7657 at all treatment sites. Similar to the PH-I period, all watersheds experienced significant increases
7658 in NO₃ + NO₂ concentration during the PH-II treatment period. Overall, the cumulative mean
7659 NO₃ + NO₂ load from all watersheds followed an increasing trend with initial signs of recovery
7660 in one treatment watershed after 2014. Mean monthly TP concentrations showed no significant
7661 changes in the concentrations during the post-road and PH-I treatment periods. However, a
7662 statistically significant increase in TP concentrations ($p < 0.001$) occurred at all sites, including
7663 the downstream cumulative sites, during PH-II. Generally, OP concentrations throughout the
7664 study remained near the minimum detectable concentrations. A statistically significant increase
7665 in mean monthly OP concentrations occurred only at the cumulative downstream treatment site
7666 during both Post-road (p -value = 0.021) and PH-I (p -value < 0.001) treatment periods,
7667 respectively. The largest cumulative increase in mean annual loads was largely attributed to
7668 increased flow. The authors conclude that only relatively small increases in nutrient loads were
7669 detected suggesting that Idaho Forest Practices Act regulations and BMPs are effective in
7670 minimizing the delivery of particulate-bound pollutants. Forest management activities increased
7671 stream NO₃ + NO₂ concentrations and loads following timber harvest activities, but these effects
7672 were also attenuated in downstream reaches and reduced through time as vegetation regrowth
7673 occurred.

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7675 Quinn, T., G.F. Wilhere, and K.L. Krueger, technical editors. 2020. Riparian Ecosystems, Volume
7676 1: Science Synthesis and Management Implications. Habitat Program, Washington Department
7677 of Fish and Wildlife, Olympia.

7678 This publication is a synthesis of scientific literature concerning riparian areas (function, process,
7679 characteristics, etc.) for the purpose of informing management and the development of policies
7680 related to management of riparian areas and watersheds of Washington State. The most relevant
7681 information in the publication to this review are in chapters 3 (large wood), 4 (stream
7682 temperature), and 6 (Nutrient dynamics in Riparian ecosystems).

7683 The main conclusions from chapter 3 (large wood) state that the successful conservation, or
7684 restoration, of fish habitats in forested areas requires management practices that deliver adequate
7685 wood into aquatic systems. They purpose the main scientific uncertainties, from a management
7686 perspective, is (1) the shape of the wood recruitment curves under different watershed and site-

7687 level conditions. These curves describe the function of wood input into streams from greater
7688 distances from the stream (source distance curves) based on stand structure (e.g., young-old, tree
7689 height variability, density metrics, etc.), species compositions (especially conifer vs. hardwood),
7690 and site conditions (e.g., slope, moisture availability, soils, site index). The second uncertainty is
7691 the effects of the potential wood delivery mechanisms that occur outside of the riparian area
7692 (e.g., landslides, debris flows). They posit that management objectives for large wood
7693 recruitment potential should aim to restore site composition and structure that is similar to
7694 unmanaged riparian forests. The authors suggest that much is known about large wood
7695 recruitment potential from within the riparian forests based on site potential and stand structure.
7696 Previous work and the development of source distance curve equations show the range of
7697 “effective” tree heights (trees with the bulk of stem > 10 cm, functionally classified as large
7698 wood) is between 85 and 230 feet. This means 100% of a sites wood recruitment potential is
7699 within 85 -230 feet of the stream. However, these equations do not account for the presence of
7700 smaller trees or the potential of tree recruitment from outside of the riparian area, or from
7701 extreme channel migration events.

7702 The main conclusions from chapter 4 identify that the science surrounding stream thermal
7703 regimes is uneven. Scientists are certain that stream temperatures and thermal regimes are
7704 important to aquatic species, and thus it is important for management practices to restore and
7705 conserve these conditions. However, while the general conclusions of most studies show that
7706 land use changes (urbanization, and agriculture) and forest management within riparian areas
7707 leads to warmer stream temperatures, the spatial and temporal effects of any specific riparian
7708 management action remain uncertain. Recovery rates for stream temperature post-treatment vary
7709 greatly based on site location because of differences in stream size (width and depth), and
7710 physiography (climate, physical geography). Shade from the adjacent riparian area is widely
7711 accepted as the most important, and most directly manageable, factor affecting stream
7712 temperature. However, because of the variability in other factors affecting stream temperature,
7713 predicting changes in stream temperature from shade removal will likely always suffer from
7714 imprecision.

7715 The main conclusions from chapter 6 (Nutrient dynamics in Riparian ecosystems) list land use,
7716 forest age and composition, Climate and seasonality, elevation and topography, hydrology,
7717 nutrient concentrations, forms and inputs, soil properties and geology, and biota as the major
7718 factors influencing nutrient dynamics in riparian ecosystems. Riparian areas that are structurally
7719 diverse in physiography and soil are most likely to support diverse biota (vegetation, animals,
7720 and microbial communities. More diverse riparian communities are considered best in processing
7721 and assimilating nutrient loads. The authors identify headwater streams as important zones for
7722 active nutrient processing because they affect downstream nutrient loads. Also, maintaining the
7723 connection between the aquatic and terrestrial environments via floodplain conservation and
7724 restoration is important. While there is still a lot of uncertainty involved in the mechanisms
7725 responsible for nutrient transport through the system, it is clear that riparian areas are vital for the
7726 not only providing nutrients to stream, but also in filtering, processing, and storing nutrients in
7727 the short and long term. The results of most studies indicate that the storage and filtering of
7728 nitrogen is most effective in areas with wide vegetated buffers compared to narrower buffers or

7729 unvegetated riparian areas regardless of the type of vegetation present. However, the type of
7730 vegetation directly impacts the quality and quantity of nutrients available. Deciduous trees
7731 generally provide more litter with higher nutrient content. Coniferous trees, on the other hand,
7732 life longer and provide more shade and large wood input potential. Thus, the authors conclude
7733 that riparian management should consider both the structural and food web roles of each species
7734 present in a forested riparian area.

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