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Forestry influences on salmonid habitat in the North Thompson River watershed, British Columbia

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Abstract

Freshwater ecosystems that support juvenile salmonids can be degraded by human pressures such as forestry. Forestry activities can alter water temperatures and the delivery and storage of water, nutrients, wood, and sediment in streams, resulting in changes to the habitat, growth, and survival of juvenile salmon. Previous research on forestry impacts on habitat has focused on small, intensively monitored coastal systems. Here, we examined forestry activities, watershed characteristics, physical habitat, and stream temperature for 28 mid-sized tributaries of the North Thompson River to examine relationships between forestry and juvenile coho stream habitat in interior watersheds. Forest harvest had a positive correlation to maximum summer stream temperature. Streams with 35% of the riparian area harvested since 1970 had maximum summer temperatures 3.7 °C higher on average than those with 5% harvested. Stream gradient explained most of the variation in physical habitat and had negative correlations to pool cover, pool depth, and fine sediment cover. Taken together, these results indicate that watershed characteristics drive physical habitat, but forest harvest can be a primary driver of water temperatures.

Key words: forestry, freshwater habitat, riparian habitat, salmonids, stream temperature

Introduction

Forestry has the potential to degrade stream habitats for fish via several different pathways. Land cover regulates important watershed processes such as the delivery of water, sediment, nutrients, light, and wood into stream channels, processes that take place on different spatial and temporal scales (Poff et al. 2006; Wohl 2019). Forestry can alter these processes, influencing habitat components such as physical habitat structure, water temperature, and flow regime, with the potential to degrade the quality of fish habitat (Wang et al. 2006). Changes to habitat can have biological outcomes in fish populations, including changes to species composition and abundance, population density, and individual growth rates (Smokorowski and Pratt 2007). Pacific salmon populations can be sensitive to changes in habitat quality across all freshwater life stages (Roni and Quinn 2001; Beechie et al. 2013; Braun et al. 2013).

Forestry impacts on physical habitat, stream temperature, and stream flow can be transient or persistent and can occur on different spatial and temporal scales. Some impacts can be immediate such as elevated fine sediment inputs (Tschaplinski and Pike 2017), while others can occur over decades, such as impaired large woody debris (LWD) recruitment (Reid et al. 2020). For example, a 40-year study of stream habitat responses to forest harvest in Carnation Creek, British Columbia (BC), found an immediate and persistent increase in stream temperature, an immediate increase in fine sediment, and delayed decreases in both pool and LWD abundance after extensive logging (Hartman et al. 1996; Tschaplinski and Pike 2017). Changes to physical habitat such as sediment and LWD are pulsed and can occur over decades as they move downstream (MacDonald and Coe 2007; Reid et al. 2019, 2020). Studies on the effects of forest harvest in the headwaters of Baptiste Creek, BC (Macdonald et al. 2003*a*, 2003*b*; Story et al. 2003; Herunter et al. 2004), found an immediate increase in stream sediment concentrations that declined to preharvest levels after 3 years, persistent increases in temperature, and temperature increases associated with road construction. These studies have shown that while the effects on physical habitat and stream temperature are variable, increases in stream temperature and changes to physical habitat associated with forest harvest can be persistent.

Physical habitat and stream temperature vary naturally with watershed attributes. Habitat is influenced by gradient, watershed size, elevation, and watershed geology. Stream and watershed gradient influence many habitat-forming stream processes, and steeper stream gradients are associated with greater particle size (Lisle and Hilton 1992; Beechie and Sibley 1997), reduced pool cover, and reduced pool depth (Wohl et al. 1993; Beechie and Sibley 1997); however, these associations can be modulated by the presence of LWD (Montgomery et al. 1996; Beechie and Sibley 1997). Watershed area can regulate pool size, with larger upstream area associated with greater pool cover and depth (Burnett et al. 2006). Elevation regulates stream temperature, and higher elevations are associated with lower temperatures (Isaak and Hubert 2001; Caissie 2006; Beaufort et al. 2020). Surface and bedrock geology regulate stream flow through influencing the infiltration rate and storage capacity in the watershed (Carlier et al. 2018). These natural mediating factors can have stronger signals than land use. In fact, one study found that watershed area, slope, elevation, and geology accounted for more variation in physical habitat (including LWD abundance, fine sediment cover, bankfull width, and bankfull depth) than forest cover, agriculture, and other forms of land cover and land use (Richards et al. 1996). Thus, watershed characteristics shape the contemporary state of the same variables that are used to observe impacts of forestry on fish, potentially masking forestry impacts. Accounting for this variation is necessary to evaluate forestry impacts on fish habitat.

Given that both forestry activities and watershed characteristics can influence fish habitat, it remains challenging to understand the impacts of forestry activities on fish habitats across broader spatial scales, especially in watersheds with variable subbasin characteristics (Macdonald et al. 2003*b*; Pollock et al. 2009; Bladon et al. 2016; Tschaplinski and Pike 2017). Previous work has often taken place in small (>1–10 km^2), uniform catchments that are adjacent to forestry impacts. This study explores forestry activities across the extent of larger watersheds (6–532 km^2) with variable subbasin characteristics and a patchwork of forest harvest. Understanding how watershed characteristics and forest harvest contribute to the state of physical habitat and stream temperature can help inform planning and decision-making to what could protect fish habitat.

The objective of this study was to understand how watershed characteristics and forestry activities influence fish habitat in streams. We examined physical habitat features, stream temperature, and flow in 28 mid-sized tributaries of the North Thompson River watershed, BC, Canada. We also examined forest harvest, road density, stream crossings, and watershed characteristics for the entire study catchments. We hypothesized that the cumulative proportion of a watershed harvested between 1970 and 2019, and the cumulative extent of forestry activities within a watershed will influence (*i*) physical habitat metrics and (*ii*) water temperature across study sites, while accounting for watershed metrics that have been shown to be key drivers of habitat. A list of a priori hypotheses and predictions can be seen in Appendix A (Table A1), which explains the proposed mechanisms and directions of each impact. We test multiple hypotheses with multiple statistical models that contain habitat metrics as response variables, and watershed and land use metrics as explanatory variables.

Materials and methods

Study location and site selection

This study was conducted in 28 tributaries of the North Thompson River, with sites located between Kamloops and Valemount in the interior of BC (Fig. 1; Table S1). This snowmelt-dominated, mountainous watershed spans three main biogeoclimatic zones: Engelmann Spruce—Subalpine Fir, Interior Cedar—Hemlock, and Interior Douglas Fir, with smaller patches of Montane Spruce and Sub-Boreal Spruce (GeoBC 2022). Forestry is the most prevalent form of land use by area in these watersheds, with small amounts of agriculture and grazing. The 28 streams are geographically distributed throughout the watershed and represent a range of habitat metrics and forestry impacts (Tables S1 and S2). These streams support coho salmon (*Oncorhynchus kisutch*), rainbow trout (*Oncorhynchus mykiss*), bull trout (*Salvelinus confluentus*), Chinook salmon (*Oncorhynchus tshawytscha*), pink salmon (*Oncorhynchus gorbuscha*), sockeye salmon (*Oncorhynchus nerka*), and sculpin (*Cottus* spp.). The study reaches are 3rd–6th Strahler stream order streams, mostly located in lower gradient reaches (<5%) where juvenile coho salmon rear. We selected streams based on historical coho salmon presence, site accessibility, the presence of juvenile rearing habitat, and local knowledge. All sites were known to be current or historical coho salmon habitat. Streams were considered accessible if they could be reached by road and were wadeable and navigable by foot. Site reaches were placed in areas with potential juvenile coho salmon rearing habitat (which was visually assessed) and within or downstream of known coho salmon spawning reaches. Surveyed reaches were 30 times the average bankfull width and established following protocols outlined by Bain and Stevenson (1999). Study reaches were divided into four equal sections, and each section contained three randomly assigned transects, following a stratified random sampling procedure.

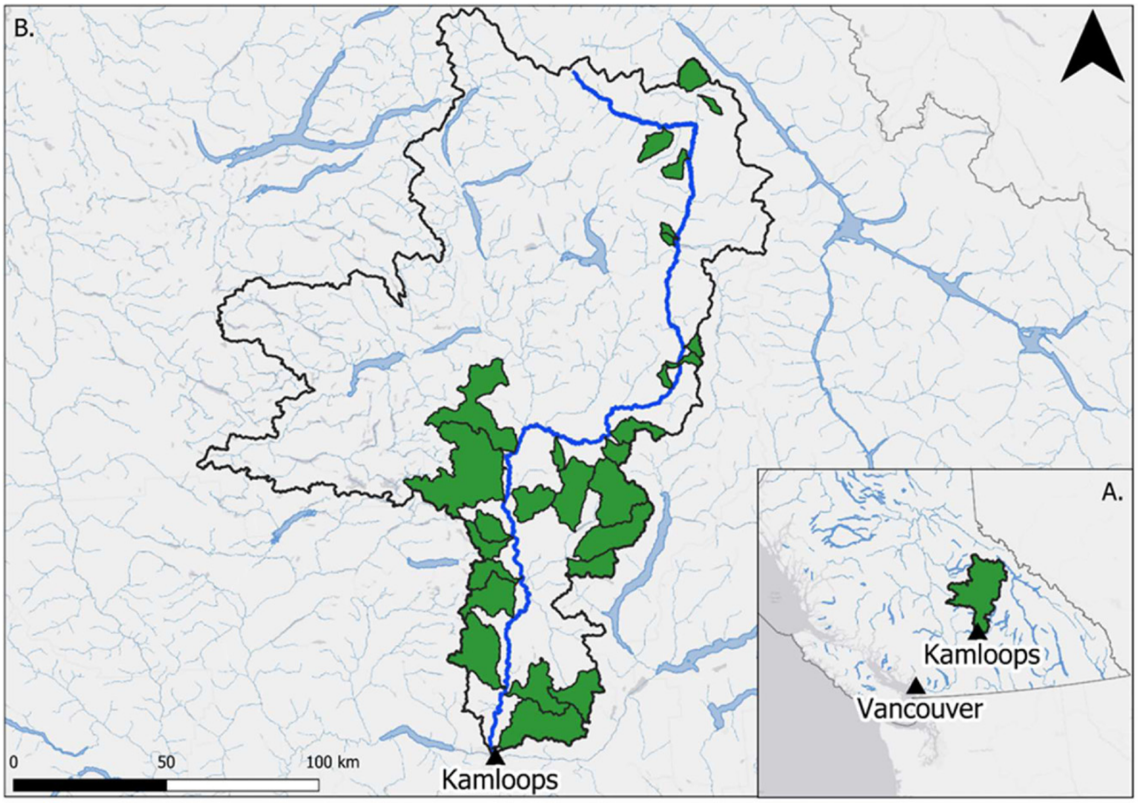
Physical habitat

Physical habitat surveys took place during low flow periods (July–August 2019 and 2020) to measure a suite of habitat and explanatory variables (Appendix A, Table A1). We measured LWD following protocols outlined by Roni and Quinn (2001). All LWD pieces within the bankfull area of each channel with a length $\geq$1.5 m and a diameter $\geq$0.10 m were counted. The length and representative diameter of each piece of wood were recorded. Large wood debris volume (m^3) was calculated using the following formula:

$$V = \pi r^2 h$$

where V is volume, r is radius, and h is length. LWD volume by area was measured as the sum of individual LWD volume divided by reach area (m^2). For larger log jams where individual pieces of wood could not be measured we estimated volume from the dimensions of the log jam. Gradient was measured at the edge of the stream, level with the water's surface. A surveyor used a TruPulse 360°R laser rangefinder to estimate the distance (accuracy $\pm$ 0.2 m) and slope (accuracy $\pm$ 0.25°) to another surveyor with a surveying rod. Surveyors identified points that were free from visual obstructions while also providing maximum distance from the surveyor with the rangefinder to limit the number of measurements and incorporate changes to channel gradi-

Fig. 1. (A) Map of the North Thompson watershed (shown in green) in British Columbia. (B) The watersheds of the 28 tributaries of the North Thompson River that make up this study (shown in green), outlined by the North Thompson watershed in black, with the North Thompson River represented by the bold blue line. Basemap from ESRI (2021); watershed and stream data from the BC Freshwater Atlas (GeoBC 2019*a*, 2019*b*).



ent. This process was repeated for the entire reach, and a weighted average was taken of the measurements to provide mean gradient for the entire reach. Wetted width, bankfull width, and bankfull height from the deepest point in the transect were measured at the start and end of the reach, as well as at each of the section breaks for a total of five measurements per site. Bankfull width and wetted width were measured with measuring tape, and bankfull height was measured using a wading rod and a Suunto PM5/360 PC clinometer (Bain and Stevenson 1999). The length of undercut bank was measured on both stream banks and converted to a percentage of the length of each bank, with the final metric being the mean of both banks. Undercut bank was measured if the undercut was at least 1 m in length and had 15 cm of overhanging bank (Moore et al. 2014).

Channel unit types (e.g., riffle, run, pool) were classified according to Bain and Stevenson (1999). Once classified, the length, width, and depth of each unit were measured. The area covered by each habitat unit was converted to a percentage of the entire reach. Pools were defined as low gradient features with an upstream crest, downstream tail, and a maximum depth of at least 1.5 times the tail depth (Heitke et al. 2008). The mean residual depth of pools was calculated for each site using the following formula:

$$d_{\mathrm{r}} = d_{\mathrm{m}} - d_{\mathrm{t}}$$

where d_{r} is the residual depth, d_{m} is the maximum depth, and d_{t} is the tail depth. Percent cover of particle classes was estimated in 1 m^2 plots that were placed on each of the random transects, with plot placement alternating between right bank, thalweg, and left bank, for a total of 12 estimates per site (four from each bank and thalweg). Percent cover of sediment was also estimated for each unit of pool habitat. The % cover of different size categories of sediment was visually estimated in the plots and pools based on methods adapted from Clapcott et al. (2011), giving measures of relative abundance. Visual estimates were compared with % cover reference diagrams, and sediment particles were measured in the field using calipers to ensure accurate measurements. One surveyor conducted all the estimates for consistency. Sediment particle size classes were adapted from a modified Wentworth classification described in Bain and Stevenson (1999). Particle size was classified as boulder (>256 mm), cobble (64–256 mm), pebble (4–64 mm), gravel (2–4 mm), and fines (<2 mm). Fines were included if they covered and embedded the sediment type underneath it (Moore et al. 2014). The mean % cover of fines and mean % cover of fines in pools were calculated for each site.

Stream temperature and discharge

Stream temperature and water level were recorded hourly using Onset HOBO U20L-04 Water Level loggers (accuracy of ±0.44 °C and ±0.004 m). These loggers were suspended in stilling wells (EPA 2014). Additional temperature, water level, and discharge data for one site (Lemieux) was sourced from the Environment and Climate Change Canada Hydrometric Data (station 08LB078). Water temperature was also taken with a calibrated thermometer at each site visit to compare to logger readings to ensure all differences were <0.2 °C. Temperature readings were checked for dewatering to identify any low-flow periods when water temperature loggers may have been recording air temperature, and these data were removed. Hourly temperature readings were recorded from July 15 to August 15 to coincide with summer high temperatures and low flows, a period during which juvenile salmonids are particularly vulnerable to high temperatures. Temperature readings from this period were converted into metrics for average daily maximum, accumulated thermal units, average daily range, summer mean, summer maximum, and summer range.

Instantaneous stream discharge ($m^3 \cdot s^{-1}$) was measured at each site 1–4 times annually using standard velocity-area methods (Bain and Stevenson 1999). Discharge was measured at transects near the water level loggers with approximately uniform substrate and flow using a HACH FH950 velocity meter (accuracy ± 0.015 $m \cdot s^{-1}$). Stage–discharge relationships were developed for each site and applied to the water level record to estimate continuous discharge (WMO 2010). Mean discharge for the summer low-flow period was generated by averaging hourly estimates from July 15 to August 15, 2021. When continuous summer discharge could not be estimated, the mean of the instantaneous discharge measurements made in July and August of 2020 and 2021 was used.

Land use and watershed characteristics

Watershed area and land use metrics were calculated using publicly available provincial (BC) datasets. Watersheds were delineated using the Freshwater Atlas Watersheds dataset (GeoBC 2019*a*). Watershed elevation, slope, and aspect were calculated from a digital elevation model at a 1:250 000 scale with 25 m resolution. The metrics for elevation, slope, and aspect all represent means for the entire watershed derived from the mean elevation, slope, and aspect of each 25 m pixel within each watershed.

Forest harvest was mapped using the Harvested Areas of BC (consolidated cutblocks) layer downloaded from the BC Data Catalogue (Forest Tenures 2021). This layer contains polygons of cutblocks derived from Forest Tenures applications and from satellite imagery and contains cutblocks that were harvested from 1970 to 2019. We delineated a 50 m riparian buffer on both sides of the entire stream network within the study watersheds and extracted the extent of cutblocks located within the riparian buffer. A distance of 50 m was chosen to capture the full extent of possible impacts within the riparian zone (Sweeney and Newbold 2014). The total area harvested from 1970 to 2019 was divided by watershed area to create a metric for proportion of watershed harvest, and the total area harvested within the 50 m riparian area was divided by the riparian buffer area to create a metric for proportion of riparian harvest. Other forestry activities included in this analysis were road density and stream crossing density. Roads were mapped from the Digital Road Atlas Master Partially-Attributed Roads dataset (GeoBC 2021). Stream crossings were identified at each location where a road crossed a stream. The total road length in each watershed was divided by the total area of each watershed to create a metric of kilometers of road length per square kilometer of watershed. The total number of stream crossings in each watershed was divided by watershed area to create a metric of crossings per square kilometer of watershed.

Analysis

Physical habitat

The relationship between forest harvest and physical habitat was explored for 25 sites. We excluded three sites from this analysis due to influence from the Thompson mainstem and wetlands, and incomplete survey data. We tested for effects of watershed characteristics and forestry activities on habitat components using multiple linear regression models (Appendix A, Table A1). The response variables were % undercut bank, % cover of fine sediment, % cover of pool habitat, % cover of fine sediment in pools, mean residual depth of pools, LWD $m^3 \cdot m^{-2}$ of stream habitat, and mean width-to-depth ratio (WDR; Table 1; Appendix A, Table A1).

We built two hypothesis-driven global linear models for each of the response metrics, both of which included a metric for forest harvest (proportion of watershed harvested or proportion of riparian area harvested), road density, stream crossing density, gradient, and watershed area as explanatory variables (Appendix A, Table A1; Table S2). Proportion of watershed harvested, and proportion of riparian area harvested, were not included in the same model as riparian area harvested is nested within watershed harvest. While there may be interactions between gradient and forestry variables, we took a conservative approach and did not include interactions in our analysis so as to not push the data beyond its inference. The explanatory variables were standardized by subtracting the mean and dividing by one standard deviation (SD) to allow for comparison of the coefficients (Cade 2015). Proportional response variables (e.g., undercut bank, % cover of fine sediment, % cover of pool habitat, and % cover of fine sediment in pools) were modeled using a beta distribution (Ferrari and Cribari-Neto 2004). All analyses were conducted in R (R Core Development Team 2021),and global models for the proportional response variables were made using the betareg package (Cribari-Neto and Zeileis 2010).

Temperature

We used multiple linear regression to test hypotheses about how forestry influences stream temperatures. The relationship between forestry land use and stream temperature was explored for 22 of our 28 sites. We excluded seven sites due to missing data and influence from the Thompson mainstem and nearby lake outflows. We built hypothesis-driven global models for the temperature response metrics of average daily mean, average daily maximum, average daily

Table 1. AICc table showing the top (ΔAICc $\leq$ 2) models for each physical habitat response variable.

(A) Response variable	Intercept	Riparian harvest	Crossings	Gradient	Roads	Watershed size	R^2	df	Log lik	ΔAICc	W*i*
LWD	0.01		−0.01				0.16	3	71.8	0.00	0.25
	0.01						0.00	2	69.7	1.62	0.11
Pool residual depth (m)	0.55			−0.12			0.25	3	4.2	0.00	0.30
	0.55			−0.11		0.04	0.29	4	4.8	1.78	0.12
Width-to-depth ratio	12.08		1.98				0.18	3	−71.3	0.00	0.15
	12.08					1.96	0.17	3	−71.3	0.11	0.14
	12.08		1.40			1.37	0.25	4	−70.2	0.65	0.11
	12.08			−0.90		1.69	0.21	4	−70.8	1.96	0.06
% Undercut bank	−2.02						0.00	2	31.7	0.00	0.20
	−2.03				0.19		0.06	3	32.4	1.10	0.12
	−2.02					−0.14	0.04	3	32.1	1.69	0.09
% Fines	−0.92			−0.80			0.32	3	11.2	0.00	0.27
	−0.95			−0.91		−0.32	0.38	4	12.4	0.49	0.21
% Pool fines	−0.36			−0.50			0.18	3	4.2	0.00	0.27
	−0.37			−0.59		−0.25	0.23	4	5.0	1.28	0.14
% Pool cover	−1.52			−0.52			0.26	3	20.8	0.00	0.32
	−1.52			−0.57		−0.18	0.28	4	21.2	1.99	0.12
(B) Response variable	**Intercept**	**Watershed harvest**	**Crossings**	**Gradient**	**Roads**	**Watershed size**	**R^2**	**df**	**Log lik**	**ΔAICc**	**W*i***
Pool residual depth (m)	0.56			−0.12			0.25	3	4.2	0.00	0.30
	0.56			−0.11		0.04	0.29	4	4.8	1.78	0.12
Width-to-depth ratio	12.08		1.98				0.18	3	−71.3	0.00	0.15
	12.08					1.96	0.17	3	−71.3	0.11	0.14
	12.08		1.40			1.37	0.25	4	−70.2	0.65	0.11
	12.08			−0.90		1.69	0.21	4	−70.8	1.96	0.06
% Undercut bank	−2.02						0.00	2	31.7	0.00	0.20
	−2.03				0.19		0.06	3	32.4	1.10	0.12
	−2.02					−0.14	0.04	3	32.1	1.69	0.09
% Fines	−0.92			−0.80			0.32	3	11.2	0.00	0.27
	−0.95			−0.91		−0.32	0.38	4	12.4	0.49	0.21
% Pool fines	−0.36			−0.50			0.18	3	4.2	0.00	0.27
	−0.37			−0.59		−0.25	0.23	4	5.0	1.28	0.14
% Pool cover	−1.52			−0.52			0.26	3	20.8	0.00	0.32
	−1.52			−0.57		−0.18	0.28	4	21.2	1.99	0.12

Note: Each response variable represents a separate model; the models are organized by the inclusion of (A) riparian harvest or (B) watershed harvest as an explanatory variable. Standardized coefficients are shown for each explanatory variable, as is R^2, degrees of freedom (df), log-likelihood (Log lik), delta (Δ, the difference from the lowest AICc score), and the model weight (W*i*).

range, accumulated thermal units, summer mean, and summer maximum (Table 2; Table S2). Like the approach for physical habitat, we built global models to test temperature hypotheses that consisted of the response metric and explanatory land cover and forestry variables (watershed harvest, riparian harvest, elevation, aspect, and summer discharge), with variables for riparian harvest and watershed harvest included in separate models.

Collinearity

We explored collinearity among explanatory variables using a correlation matrix and the variance inflation factor (VIF). To avoid issues that may arise from collinearity, we removed variables with a VIF greater than 5 from the global model (Zuur et al. 2010). We visually examined model diagnostic plots (residuals vs. fitted values, normal Q–Q, scale location, and Cook's distance) to test for assumptions of normality, homogeneity, independence, and to check for influential observations (Zuur et al. 2007).

Model selection and averaging

We used Akaike information criterion corrected for small sample sizes (AICc) to evaluate support for the two sets of candidate models, one of which consisted of all combinations of variables for the watershed, the other for the riparian area (Burnham and Anderson 2004). We used a score of ΔAICc $\leq$ 2 to identify the set of top models, as models with these values are roughly equal, and model averaged using the natural

Table 2. AICc table showing the top (ΔAICc ≤ 2) models for each temperature response variable.

(A) Response variable	Intercept	Aspect	Discharge	Elevation	Riparian harvest	R^2	df	Log lik	ΔAICc	W*i*
Average daily maximum	15.06				1.49	0.31	3	−49.3	0.00	0.24
	15.07			−0.76	1.03	0.37	4	−48.2	0.88	0.16
	15.09			−1.29		0.27	3	−49.9	1.21	0.13
Average daily range	2.92		1.39	−0.89		0.34	4	−27.2	0.00	0.26
	2.84	0.32	1.15	−0.84		0.41	5	−25.9	0.75	0.18
	2.93		1.47	−0.76	0.31	0.40	5	−26.0	1.04	0.15
Accumulated thermal units	436.90				42.49	0.32	3	−122.3	0.00	0.22
	437.26			−21.48	29.53	0.39	4	−121.1	0.74	0.15
	430.28		−33.97		32.18	0.38	4	−121.3	1.10	0.13
	437.70			−36.78		0.28	3	−122.9	1.31	0.12
Summer maximum	16.67				1.55	0.27	3	−52.0	0.00	0.29
	16.69			−0.66	1.15	0.31	4	−51.4	1.76	0.12
	16.70			−1.26		0.21	3	−53.0	1.84	0.12
Summer mean	13.67				1.31	0.32	3	−46.0	0.00	0.21
	13.68			−0.67	0.9	0.39	4	−44.9	0.71	0.15
	13.46		−1.09		0.98	0.38	4	−45.0	0.99	0.13
	13.69			−1.14		0.28	3	−46.6	1.12	0.12
Summer range	5.97	0.50				0.12	3	−36.3	0.00	0.20
	6.01					0.00	2	−37.8	0.18	0.19
	6.01				0.35	0.07	3	−37.0	1.29	0.11
	5.97	0.45			0.28	0.17	4	−35.8	1.88	0.08
	5.96	0.57		−0.26		0.17	4	−35.8	1.91	0.08

(B) Response variable	Intercept	Aspect	Discharge	Elevation	Watershed harvest	R^2	df	Log lik	ΔAICc	W*i*
Average daily maximum	15.09			−1.29		0.27	3	−49.9	0.00	0.24
	15.02	0.79		−1.43		0.34	4	−48.8	0.82	0.16
	15.08			−0.98	0.70	0.32	4	−49.1	1.35	0.12
	15.07				1.20	0.20	3	−50.9	1.94	0.09
Average daily range	2.92		1.39	−0.89		0.34	4	−27.2	0.00	0.31
	2.84	0.32	1.15	−0.84		0.41	5	−25.9	0.75	0.22
Accumulated thermal units	437.70			−36.78		0.28	3	−122.9	0.00	0.20
	437.55			−26.96	22.08	0.35	4	−121.8	0.82	0.13
	425.26		−60.84			0.24	3	−123.6	1.24	0.11
	437.22				35.93	0.23	3	−123.7	1.46	0.10
	436.20	17.37		−39.76		0.33	4	−122.2	1.64	0.09
	428.97		−41.82		23.96	0.32	4	−122.3	1.79	0.08
	420.67	24.16	−73.22			0.32	4	−122.4	1.87	0.08
Summer maximum	16.70			−1.26		0.21	3	−53.0	0.00	0.22
	16.62	0.94		−1.42		0.29	4	−51.8	0.63	0.16
	16.69				1.18	0.16	3	−53.6	1.34	0.11
	16.70			−0.95	0.69	0.25	4	−52.3	1.79	0.09
Summer mean	13.69			−1.14		0.28	3	−46.6	0.00	0.20
	13.69			−0.84	0.67	0.35	4	−45.5	0.95	0.12
	13.30		−1.90			0.25	3	−47.1	1.10	0.12
	13.68				1.10	0.23	3	−47.4	1.63	0.09
	13.65	0.54		−1.23		0.33	4	−45.9	1.66	0.09
	13.16	0.75	−2.29			0.32	4	−45.9	1.72	0.08
	13.42		−1.33		0.72	0.32	4	−46.0	1.82	0.08
Summer range	5.97	0.50				0.12	3	−36.3	0.00	0.24
	6.01					0.00	2	−37.8	0.18	0.22
	5.96	0.57		−0.26		0.17	4	−35.8	1.91	0.09

Note: Each response variable represents a separate model; the models are organized by the inclusion of (A) riparian harvest or (B) watershed harvest as an explanatory variable. Standardized coefficients are shown for each explanatory variable, as is R^2, degrees of freedom (df), log-likelihood (Log lik), delta (Δ, the distance from the lowest AICc score), and the model weight (W*i*).

method to account for model uncertainty in estimates of coefficients (Burnham and Anderson 2004; Barton 2020). The natural (also called subset or conditional) method of model averaging was used to avoid downward bias in parameter estimates (Galipaud et al. 2017). This was repeated for each of the global models.

Results

Landscape and forestry variables varied widely among sites (Table S2). Forest harvest, measured as the proportion of the watershed harvested in the 50-year period between 1970 and 2019, ranged from 1% to 59% (Fig. S1). Riparian harvest ranged from 0% to 41% of the 50 m riparian area. Road density ranged from 0.3 to 3.2 km·km^{-2}and stream crossing density ranged from 0.2 to 2.7 per km^2. Physical habitat components and stream temperature also varied between streams; a breakdown of this variation is shown in Table S1.

Physical habitat

Natural landscape predictor variables were much stronger descriptors of physical habitat than forest harvest predictor variables. Specifically, gradient was the best descriptor of most habitat response variables and explained between 18% and 32% of the variation in physical habitat response variables (Table S2). Steeper gradient stream reaches had less pool area, and pools were shallower with more fine sediment (Figs. 2 and 3). Based on model averaged coefficients, stream reaches with 1% gradient had 23% pool area, while streams with 3% gradient had 12% pool area. Similarly, stream reaches with 1% gradient had an average pool depth of 0.6 m, while streams with 3% gradient had an average pool depth of 0.42 m. Streams at low (1%) gradient had on average 50% fine sediment cover in pools and 41% fine sediment cover in the reach, while streams at high (3%) gradient had 31% fine sediment cover in pools and 16% fine sediment cover in the reach. Gradient was the only explanatory variable included in the top models of the candidate model sets (ΔAICc = 0) for pool habitat cover (pseudo $R^2 = 0.26$), pool residual depth ($R^2 = 0.25$), fine sediment cover in the reach (pseudo $R^2 = 0.32$), and fine sediment cover in pools (pseudo $R^2 = 0.18$) (Table 1). There was also a negative effect of gradient on the WDR, but the uncertainty was larger than the estimate (Fig. 2). Watershed size was included in the top model set (ΔAICc < 2) for every response variable other than LWD, however the effect of watershed area on response variables was smaller than the uncertainty except for pool residual depth where there was a slight positive effect (Fig. 2).

The only clear effect of forestry on physical stream habitat was that of stream crossing density on LWD volume and channel WDR. Streams with higher stream crossing densities had lower LWD volume. On average, streams with two crossings per km^2 had 97.5% less LWD volume than streams with no road crossings. Streams with higher stream crossing densities were also wider and shallower than streams with lower stream crossing densities; however, in the case of WDR, the uncertainty was greater than the effect size. Stream crossing density was the only explanatory variable included in the top model (ΔAICc = 0) for LWD volume ($R^2 = 0.16$). Stream crossing density was also the only explanatory variable included in the top model (ΔAICc = 0) for WDR ($R^2 = 0.18$). None of the top models (ΔAICc = 0) included watershed size, road density, or proportion of the watershed or riparian area harvested (Table 1).

Temperature

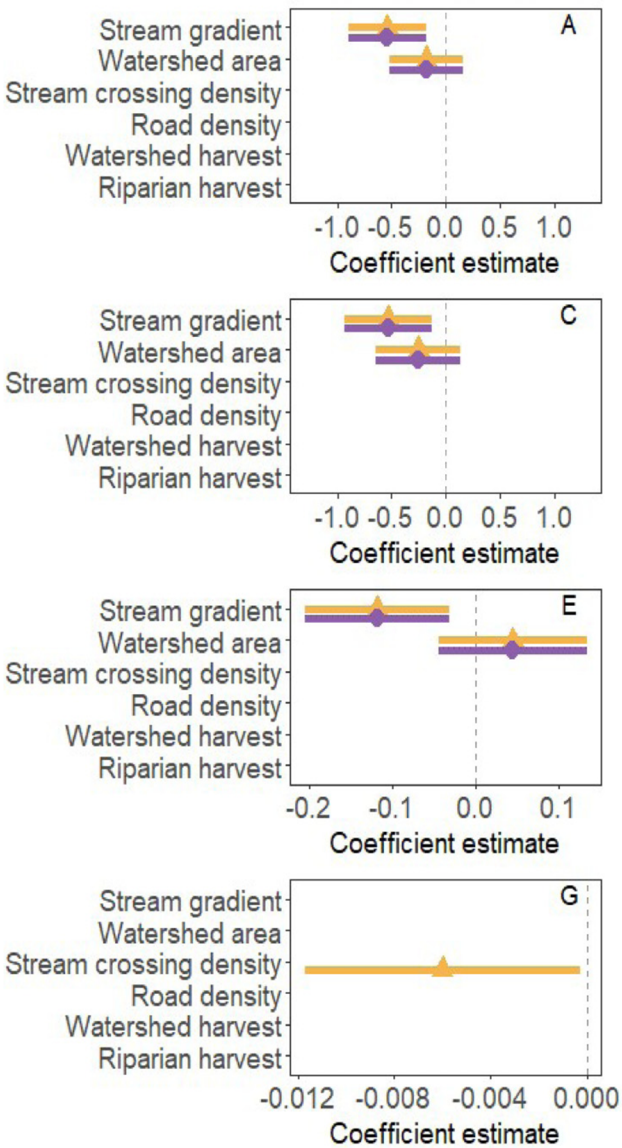
Riparian area harvest had a consistent and strong warming effect on stream temperature. Streams with higher proportions of riparian area harvested had higher stream temperatures (Figs. 4 and 5). Riparian area harvest consistently outperformed total harvest in describing each of the response variables. Specifically, models that included riparian area harvest had AICc values between 1.6 and 1.9 lower than the equivalent models where forestry effects were represented by watershed harvest. Holding other values constant at the mean, an increase from 5% to 30% of riparian area harvested (2 SD) increased the average daily maximum by 2.6 °C, the summer maximum by 2.9 °C, the summer mean by 2.2 °C, and the accumulated thermal units by 92. Likewise, an increase from 10% to 47% of watershed harvested (2 SD) increased the average daily maximum by 1.8 °C, the summer maximum by 1.9 °C, the summer mean by 1.6 °C, and the accumulated thermal units by 54. These relationships were approximately linear. The effect of watershed harvest had more uncertainty than the effect of riparian area harvest, with the uncertainty being greater than the estimate for each model, whereas the confidence intervals did not overlap zero for the positive effect of riparian harvest for average daily range, summer mean, summer range, and accumulated thermal units.

Watershed characteristics also influenced stream temperature. Higher elevation streams had lower average temperature, and smaller temperature ranges during the summer. For the model set that included watershed harvest, elevation was in the top (ΔAICc = 0) candidate models for the average daily maximum temperature ($R^2 = 0.27$), the average daily range in temperature ($R^2 = 0.34$), accumulated thermal units ($R^2 = 0.28$), and summer maximum ($R^2 = 0.21$) and mean temperatures ($R^2 = 0.28$) (Table 2). For the model set that included riparian area harvest, elevation was only included in the top (ΔAICc = 0) model for average daily range ($R^2 = 0.34$). Higher mean summer discharge was associated with higher daily temperature ranges. Discharge was included in the top (ΔAICc = 0) candidate models for average daily range in both model sets ($R^2 = 0.34$). Additionally, aspect had a positive association with summer range. For both model sets, aspect was the only explanatory variable in the top (ΔAICc = 0) candidate models for summer temperature range ($R^2 = 0.12$).

Discussion

We examined 28 streams to understand how forest harvest and watershed characteristics influence physical habitat and temperature in salmon-bearing rivers. Gradient explained more variation in physical habitat than any other natural or forestry-related explanatory variable. In contrast, we found strong effects of both forestry and watershed characteristics on stream temperatures. Streams with higher pro-

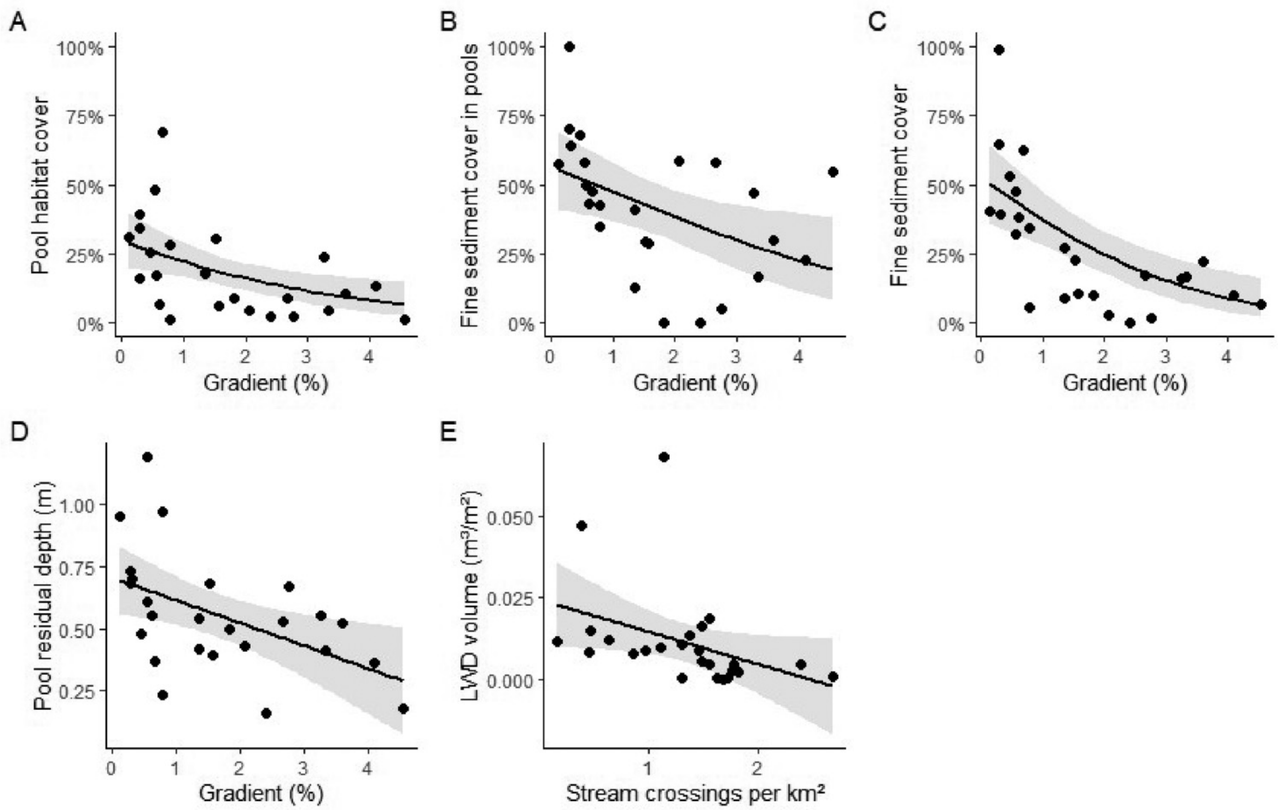
Fig. 2. Model averaged coefficients from the physical habitat analysis for pool habitat cover (A), fine sediment cover in the reach (B), fine sediment cover in pools (C), undercut bank (D), pool residual depth (E), width-to-depth relationship (F), and LWD volume (G). Orange triangles are coefficients from models that included the proportion of the watershed harvested as an explanatory variable, while purple circles are from models that included the proportion of riparian area harvested.



portions of harvested areas, in the watershed and in riparian areas, were consistently associated with higher and more variable stream temperatures. Elevation had a consistently negative relationship to temperature and temperature range. Thus, both forestry and landscape components contributed to the current state of stream habitat. Collectively, these results indicate that watershed characteristics take primacy in influencing physical habitat, but forestry activities strongly influence water temperatures.

Forest harvest had a consistent and positive linear relationship with stream temperature. Stream temperatures were higher in watersheds with higher levels of forest harvest, and the relationship was even stronger for forest harvest in the riparian area. We found that on average, the average daily maximum summer temperature of streams with low (5%) proportions of harvest in the riparian areas was 13.5 °C, while the average daily maximum in streams with high (35%) proportions of riparian harvest was 17.2 °C, a difference of 3.7 °C, larger than the 2 °C difference for watershed harvest. Thus, the proportion of riparian area harvest was a consistently stronger and more certain predictor of temperature metrics than the proportion of watershed harvest. The effects of forestry on stream temperature were broadly consistent with the literature, where many studies (Macdonald et al. 2003*b*; Story et al. 2003; Pollock et al. 2009; Tschaplinski and Pike 2017) have found higher stream temperatures associated with harvesting were caused by increased solar radiation, reduced hyporheic exchange, channel shallowing and widening, and a decline in summer baseflow. Our study advances understanding by offering spatial replication among mid-sized water-

Fig. 3. Observations and model predictions for the effect of gradient on (A) pool habitat cover, (B) fine sediment cover in pools, (C) fine sediment cover in random plots in the reach, and (D) residual depth in pools. (E) Observations and model predictions for the relationship between stream crossing density and LWD volume. The shaded regions represent 95% confidence intervals.

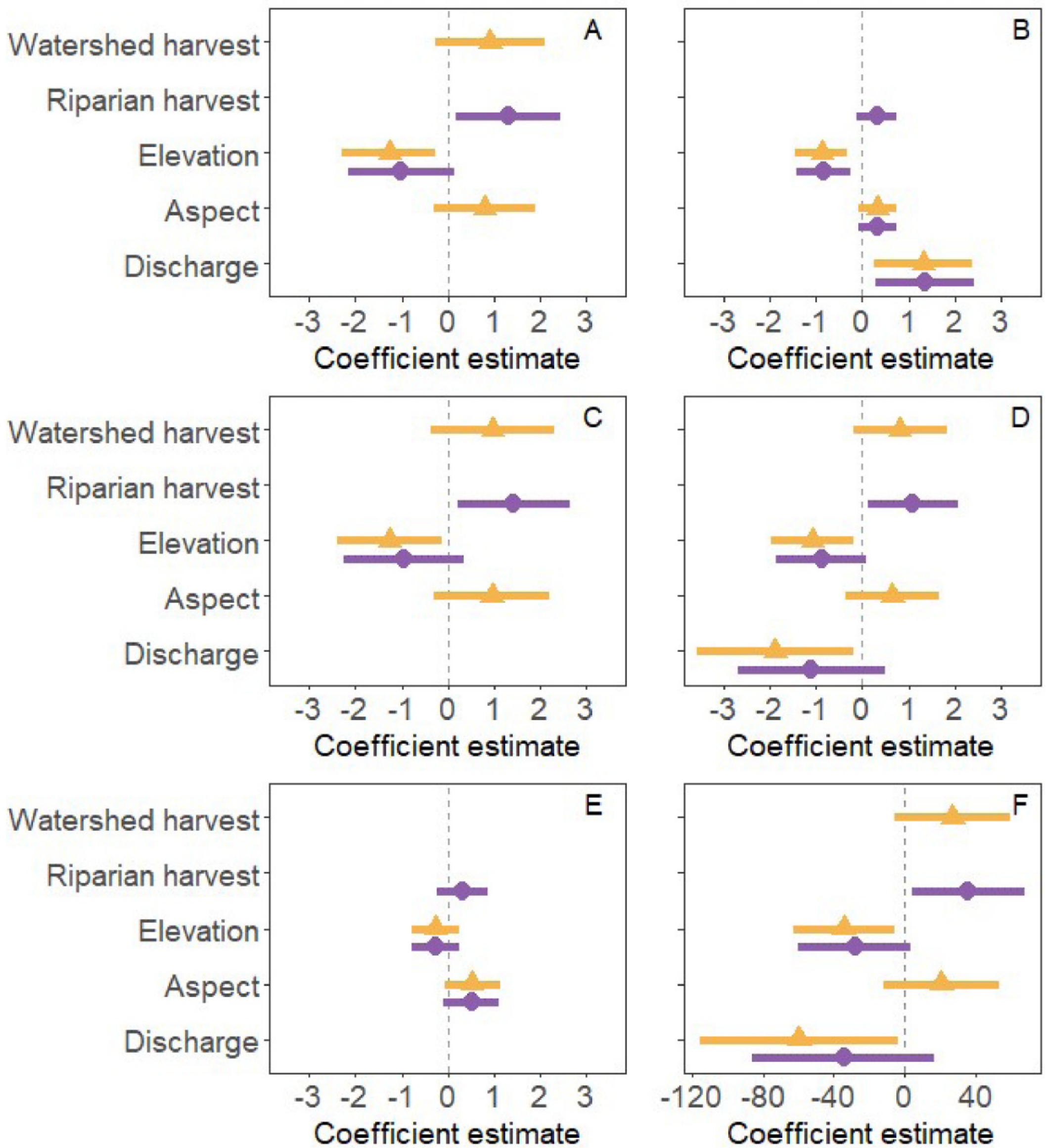


sheds with different harvest levels and a patchwork of harvest dates. Thus, forestry can exacerbate temperature risks to thermally sensitive fish (Pollock et al. 2009; Beechie et al. 2013), a critical issue in this warming world.

Gradient had the strongest effect of any explanatory variable on pool depth, pool habitat cover, and fine sediment cover in pools and in the reach. The effect of gradient was more than twice the effect of watershed size on these variables (Fig. 2). These results were predicted based on previous findings of negative relationships between stream gradient and pool depth and cover (Wohl et al. 1993; Beechie and Sibley 1997; Montgomery and Buffington 1997). Lower gradient streams tend to have less resistant channel boundaries and can more effectively scour the channel bed to form pools, and flatter surfaces provide more opportunities for pools to form (Wohl et al. 1993; Hupp and Osterkamp 1996; Montgomery and Buffington 1997). Pools also form from changes to water velocity caused by channel narrowing and stream slope (Chartrand et al. 2018). Our findings match previous findings of positive relationships between gradient and sediment size (Beechie and Sibley 1997; Buffington et al. 2004). Higher gradient streams are more connected to adjacent hillslopes and have a sediment supply with larger grain sizes, while lower gradient streams have more fine sediment available to fill pools (Lisle and Hilton 1992; Montgomery and Buffington 1997). Higher gradient streams have higher shear stress, which allows smaller sediment particles to be easily transported and deposited in lower gradient sections (Wohl et al. 1993; Beechie and Sibley 1997). While past studies have found that forestry has decreased pool cover and depth (Chen and Wei 2008; Mellina and Hinch 2009; Tschaplinski and Pike 2017) and increased fine sediment cover (Herunter et al. 2004; Tschaplinski and Pike 2017), we did not find any relationship between forestry and these metrics in our comparative study. Our findings suggest that gradient is a key factor in broadly determining physical habitat, and monitoring efforts that look to quantify forestry effects should account for this when selecting response variables by selecting variables that are process based and not pattern based (Montgomery and Buffington 1997).

The observed temperature increases associated with forest harvest could be large enough to impact fish growth and survival. At high (35%) levels of riparian area harvest, the average summer maximum temperature is 18.8 °C, which is higher than the optimal rearing temperature of juvenile coho of 12–15 °C (Richter and Kolmes 2005). Temperatures above 17 °C can cause stress for juvenile coho, with growth stopping at 20.3 °C (assuming a constant ration of food) and the upper lethal temperature is 25.9 °C (Richter and Kolmes 2005). While none of the study streams reached the upper lethal temperature during the sampling period, four streams reached temperatures above 20.3 °C. The predicted average

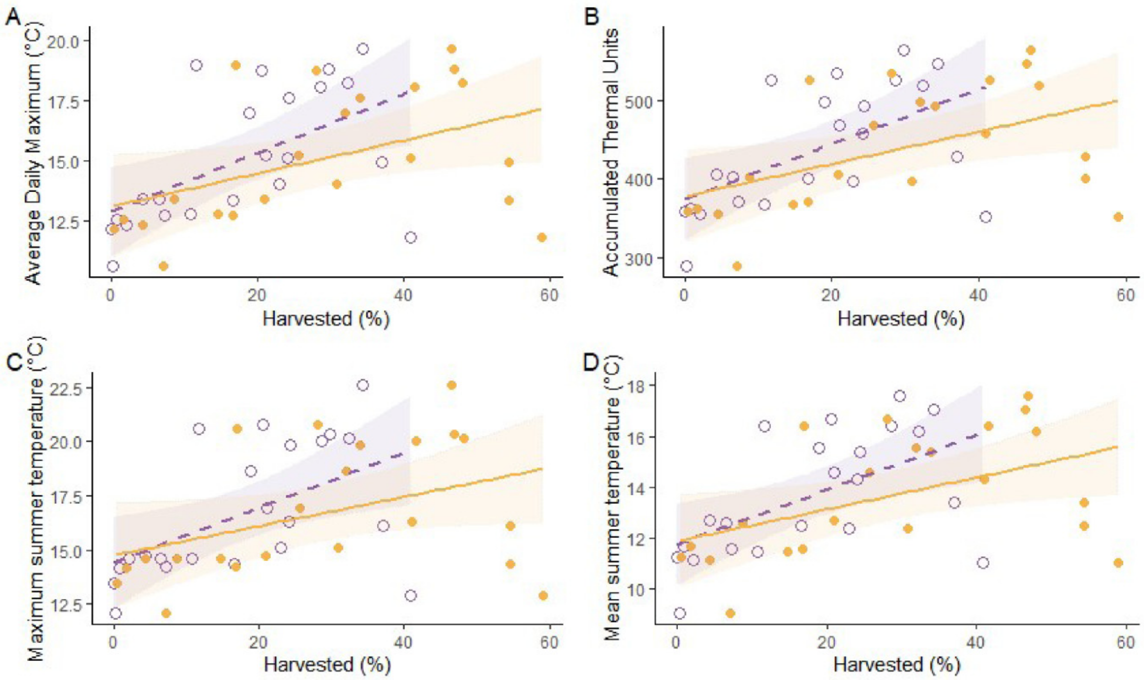
Fig. 4. Model averaged coefficients from the temperature analysis for average daily mean (A), average daily range (B), summer maximum temperature (C), summer mean temperature (D), summer temperature range (E), and accumulated thermal units (F). Orange triangles are coefficients from models that included the proportion of the watershed harvested as an explanatory variable, while purple circles are from models that included the proportion of riparian area harvested.



daily maximum and summer maximum at high (35%) levels of riparian harvest is above the optimal rearing temperature listed above. Similarly, the predicted summer maximum temperature at 30% riparian area harvest is within the range of temperatures that cause stress to fish. The population of coho salmon in the North Thompson is threatened, and the number of spawners has declined by over 60% since the early 1990s due to a combination of historic overfishing, changing ocean conditions, and freshwater habitat alteration (Bradford and Irvine 2000; Arbeider et al. 2020). More broadly, Pacific salmon are threatened by climate change and freshwater habitat degradation (Schindler et al. 2008; Munsch et al. 2022). Stream temperatures are warming throughout the range of Pacific salmon, including in the Fraser watershed, and our research suggests that climate-induced temperature increases could be amplified by forest harvest or buffered by maintaining forest cover (Mantua et al. 2010; Islam et al. 2019). While ocean health and climate change are long-term, international issues, managing watersheds for stream temperature and physical habitat is a management lever available to federal and provincial government agencies (Moore and Schindler 2022).

This study and previous research suggest that the detection of effects of forestry on physical habitat and stream temperature is dependent on study design and the scale of the underlying processes. This study was based on spatial but not temporal replication, whereas many previous studies are before-after control-impact studies that quantified the state of the physical habitat before forestry began (Macdonald et al. 2003*b*; Tschaplinski and Pike 2017). While we did not find

Fig. 5. Observations and model predictions of the relationship between harvested area and (A) average daily maximum temperature, (B) accumulated thermal units, (C) maximum summer temperature, and (D) mean summer temperature in the study streams ($n = 22$). The solid orange points and lines are for the relationship between the harvested proportion of the watershed and stream temperature metrics; the purple circles and dashed lines are for the relationship between the harvested proportion of the riparian area and stream temperature metrics.



evidence that forest harvest or road density explained variation in physical habitat, our spatial comparative approach may miss episodic disturbances that could result from forest harvesting as well as intrinsic dynamics of stream ecosystems, making it challenging to link physical habitat to forest harvest (Reid and Hassan 2020). This was seen in Carnation Creek, where infrequent pulses of sediment inputs from landslides and bank failures made their way downstream over decades, with a majority of small-scale inputs only evident for periods of 3–5 years, with larger inputs taking decades to show and move downstream (Tschaplinski and Pike 2017; Reid et al. 2019, 2020). In addition to higher magnitude episodic disturbances such as bank failure, more persistent natural, small disturbances from beaver activity, tree fall, grazing, and vegetation growth can cause alterations in channel condition and change or reset physical habitat conditions within a catchment (Roper et al. 2022). These natural disturbances should be considered alongside current and past land use when assessing land use impacts on habitat. Further, it is likely that watershed characteristics may vary so dramatically across different locations that it could be challenging to detect forestry effects. We found watershed characteristics explained most of the variation in physical habitat, however temperature was consistently explained by forest harvest. This suggests that the temporal and spatial scales of the processes that control physical habitat and stream temperature differ. Alterations to physical habitat may be more episodic and transient for large-scale disturbances that can be easily detected (Reid and Hassan 2020; Roper et al. 2022), while temperature may be more sensitive to forestry, and changes to stream temperature may be more persistent, and therefore a more reliable metric for detecting the impacts of forest harvest.

This study reveals how current forest management enables potentially harmful temperature impacts. Riparian harvest had a stronger effect on stream temperature than watershed harvest, with the vast majority of riparian harvest occurring in headwater streams. Specifically, over 91% of the riparian harvest measured in this study was in the riparian areas of first- and second-order streams (Fig. S2). Indeed, full retention riparian buffers are not required for nonfish-bearing streams that are <3 m in width or of fish-bearing streams that are <1.5 m in width given current forestry regulations in BC (Forest Practices Board 2018; Government of British Columbia 2021). Requiring riparian buffers on headwater streams is a management lever that could mitigate some of the effects of forestry on stream temperature, and past studies have demonstrated that buffers can limit the effects of forest harvest on stream temperature and lead to downstream cooling (Macdonald et al. 2003*b*; Bladon et al. 2018). Protecting riparian buffers in smaller streams would thus help provide climate resilience to BC's salmon streams, but would decrease the amount of area that is available for forestry—headwater streams constitute over 70% of a stream networks channel length (Wohl 2017). Additionally, the uneven distribution of forest harvest among the study watersheds (from 1% to 59% of each watershed) is also a product of forestry regulations. Forestry in BC is currently managed on a large scale (one timber supply area contains all 28 study watersheds); planning at the scale of midsize watersheds such as fish-bearing tributaries could allow for more control over the distribution of forestry impacts.

In the context of declining salmon populations and increasing stream temperatures from climate change, the relationship between forest harvest and stream temperature presents both challenges and opportunities for managers. The challenge is that forest harvest can increase stream temperatures to the point of causing stress to salmon, especially when paired with warming temperatures from climate change. Most salmon watersheds in BC have been exposed to some degree to forestry. The opportunity is that forestry management is an available management lever to cool water temperatures.

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Data availability

Data generated or analyzed during this study are available at the Dryad Digital Repository: https://doi.org/10.5061/dryad.d7wm37q54.

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Competing interests

The authors declare that there are no competing interests.

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Supplementary material

Supplementary data are available with the article at https://doi.org/10.1139/CJFAS-2022-0255.

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Appendix A

Table A1. Hypothesis tableshowing response variables, response metrics, explanatory variables, and the hypothesized influences explanatory variables will have on response variables.

Response variable	Metric	Explanatory variable	Hypothesized mechanism of impact on response variable	Reference
Fine sediment	1) % cover of fines in pools2) % cover of fines in stream reach	Proportion of watershed and riparian area harvested	Forest harvest can increase fine sediment inputs through increased runoff and more frequent landslides, scour, and sediment transport events	(Smith and Redding 2012; Tschaplinski and Pike 2017)
		Stream crossings	Stream crossings can increase fine sediment delivery to streams from direct inputs from drainage ditches and can cause constrictions in flow which increases erosion	(Macdonald et al. 2003a; Smith and Redding 2012)
		Road density	Roads increase fine sediment availability, transport, and inputs into streams through direct sediment contributions and by increasing peak flows from ditch runoff, causing erosion	(Macdonald et al. 2003a)
		Stream gradient	Higher gradient streams are more connected to the hillslope and have a sediment supply with larger grain sizes. Lower gradient streams have more fine sediment available to fill pools	(Lisle and Hilton 1992; Beechie and Sibley 1997; Buffington et al. 2004; Bracken et al. 2015) (Lisle and Hilton 1992; Beechie and Sibley 1997; Buffington et al. 2004; Bracken et al. 2015)
		Watershed size	Larger watersheds are associated with higher fine sediment cover due to wider streams and lower flow velocity. Smaller watersheds are associated with lower fine sediment cover due to narrower streams and higher flow velocity. Sediment availability is also influenced by bedrock geology	(Beechie and Sibley 1997)
Large wood	1) LWD volume	Proportion of riparian area harvested	Forest harvest can decrease the availability of LWD, leading to decreased inputs of LWD into streams	(Fausch and Northcote 1992; Mellina and Hinch 2009; Reid et al. 2020)
		Stream crossings	Culverts and bridges can decrease LWD by blocking the passage of LWD into downstream habitat	(Lassettre and Kondolf 2012)
		Road density	Roads can decrease LWD availability by causing increased peak flows and scour events, causing LWD transportation out of the stream	(Macdonald et al. 2003a; Lassettre and Kondolf 2012)
		Stream gradient	High gradient streams tend to be smaller and present more opportunities for LWD to become embedded, which can increase volume. Low gradient streams tend to be wider and have lower volumes of LWD	(Beechie and Sibley 1997; Ruiz-Villanueva et al. 2016)
		Watershed size	Streams in smaller watersheds tend to be smaller and present more opportunities for LWD to become embedded, which can increase volume. Larger streams in larger watersheds can have lower volumes of LWD as they tend to be wider	(Abbe and Montgomery 1996; Beechie and Sibley 1997; Ruiz-Villanueva et al. 2016)

Table A1. (*continued*).

Response variable	Metric	Explanatory variable	Hypothesized mechanism of impact on response variable	Reference
Pool cover and depth	1) Pool habitat % cover 2) Pool residual depth	Proportion of watershed and riparian area harvested	Forest harvest can decrease pool cover and depth through increased frequency and magnitude of peak flows which causes more sediment transport and downstream delivery, increasing the likelihood of pool filling	(Fausch and Northcote 1992; Montgomery et al. 1995; Mellina and Hinch 2009; Tschaplinski and Pike 2017)
		Stream crossings	Stream crossings can reduce pool cover and depth through increased flow velocity, leading to a reduction in pool area and depth	(Fausch and Northcote 1992; Macdonald et al. 2003a; Mellina and Hinch 2009; Tschaplinski and Pike 2017)
		Road density	Roads can reduce pool cover and depth through increased peak flows	(Fausch and Northcote 1992; Macdonald et al. 2003a; Mellina and Hinch 2009; Tschaplinski and Pike 2017)
		Stream gradient	Low gradient streams are associated with higher pool cover and deeper pools	(Montgomery et al. 1995; Beechie and Sibley 1997)
		Watershed size	Larger watersheds are associated with higher pool cover and deeper pools	(Burnett et al. 2006)
Undercut bank	1) % of undercut stream bank	Proportion of watershed and riparian area harvested	Forest harvest can reduce the amount of undercut bank through increased frequency and magnitude of peak flows that cause loss of stability in banks	(Murphy et al. 1986; Tschaplinski and Pike 2017)
		Stream crossings	Stream crossings can cause a reduction in undercut bank through increased flow velocity which leads to erosion and reduced bank stability	(Murphy et al. 1986; Macdonald et al. 2003a; Tschaplinski and Pike 2017)
		Road density	Roads can cause a reduction in undercut bank through increased peak flows which lead to erosion and reduced bank stability	(Macdonald et al. 2003a; Murphy et al. 1986; Tschaplinski and Pike 2017)
		Stream gradient	High gradient streams have higher shear stress and higher stream velocity, which are associated with increased undercut bank	(Beechie and Sibley 1997)
		Watershed size	Larger watersheds generally have higher flows and more potential for turbulent events which form undercut banks	(Roy et al. 2019)
Stream width-to-depth ratio	1) Ratio of stream bankfull width to bankfull height	Proportion of watershed and riparian area harvested	Forest harvest can increase WDR through increased frequency and magnitude of peak flows that cause loss of stability in banks	(Murphy et al. 1986; Tschaplinski and Pike 2017)
		Stream crossings	Stream crossings can cause an increase in WDR through increased flow velocity which leads to erosion, reduced bank stability, and wider streams	(Murphy et al. 1986; Macdonald et al. 2003a; Tschaplinski and Pike 2017)
		Road density	Roads can cause an increase in WDR through increased peak flows which leads to erosion, reduced bank stability, and wider streams	(Murphy et al. 1986; Macdonald et al. 2003a; Tschaplinski and Pike 2017)
		Stream gradient	Higher gradient is associated with shallow, narrow streams	(Richards et al. 1996; Beechie and Sibley 1997; Burnett et al. 2006; Faustini et al. 2009)
		Watershed size	Larger watersheds are associated with wider, deeper streams	(Richards et al. 1996; Burnett et al. 2006; Faustini et al. 2009)

Table A1. (*concluded*).

Response variable	Metric	Explanatory variable	Hypothesized mechanism of impact on response variable	Reference
Stream temperature	1) Average daily maximum 2) Average daily range 3) Accumulated thermal units 4) Summer maximum 5) Summer mean 6) Summer range	Proportion of watershed and riparian area harvested	Forest harvest can increase stream temperature and make the temperature range more variable through increased solar radiation, reductions in summer low flows, and alterations to hyporheic and groundwater exchange	(Macdonald et al. 2003b; Herunter et al. 2004; Pollock et al. 2009; Tschaplinski and Pike 2017; Bladon et al. 2018; Gronsdahl et al. 2019)
		Watershed aspect	South facing streams and watersheds are associated with higher temperatures as they are exposed to more solar radiation	(Smith and Redding 2012)
		Watershed elevation	Higher elevations are associated with lower stream temperatures as elevation moderates stream temperature through glacier melt, snowmelt, and lower air temperatures	(Isaak and Hubert 2001; Beaufort et al. 2020)
		Summer mean discharge	Lower discharge is associated with more thermal sensitivity	(Caissie 2006)

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