Forestry influences on instream salmon habitat and the implications for watershed management

by

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Abstract

Freshwater ecosystems support important species, such as salmon, but can be degraded by various human pressures such as forestry. Forestry activities can alter the delivery and storage of water, nutrients, wood, and sediment in streams, resulting in changes to habitat. Forestry can also alter water temperatures, leading to changes in growth and survival of juvenile salmon. Previous research on forestry impacts on habitat has focused on small scale, intensively monitored coastal systems, challenging the application of this science to management decisions in larger watersheds. Here I examined forestry impacts, watershed characteristics, physical habitat, and stream temperature for 28 tributaries of the North Thompson River to examine relationships between forestry and juvenile coho stream habitat. For each watershed, I quantified natural components (elevation, area, and stream gradient), forestry impacts (harvest area, road density, and stream crossing density), and habitat (water level, temperature, pool cover, pool depth, woody debris abundance, sediment cover, and bank dimensions). 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 that were 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. Streams with 1% gradient had on average 23% pool area and 41% fine sediment cover, while streams with 3% gradient which had 12% pool area and 16% fine sediment cover. Taken together, these results indicate that watershed characteristics drive physical habitat, but forestry harvest can be a primary driver of water temperatures. This study advances understanding of impacts and monitoring of forestry on stream environments.

Keywords: Forestry, freshwater habitat, juvenile salmon, North Thompson, stream temperature

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Chapter 1. Introduction

Rivers connect landscapes and terrestrial ecosystems to freshwater ecosystems and ultimately to the ocean. Water moving through and over land delivers nutrients, wood, and sediment into downstream aquatic habitats (Poff et al., 2006; Seelbach, 2006). These components are transported downstream through freshwater systems such as streams, rivers, wetlands, and lakes, shaping physical habitat throughout this journey. Rivers themselves provide important habitat for many species, including Pacific salmon. As adult salmon return to freshwater to spawn, they rely on streams that provide enough flow to facilitate upstream passage and suitable water temperatures and substrate for spawning (Cooke et al., 2020; Crossin et al., 2008). Eggs and fry require appropriate substrate, water temperature, and oxygen levels to facilitate growth and development (Braun et al., 2013; Quinn, 2018). Juveniles need abundant food, cover from predators, and temperatures that support their metabolism (Ebersole et al., 2006; Rosenfeld et al., 2005). Stream temperature and physical freshwater habitat are important for the growth and survival of Pacific salmon in several life stages, however, the loss and degradation of freshwater habitat from human activities is widespread and represents a considerable threat to salmon (Andrew & Wulder, 2011; Finn et al., 2021; Reid et al., 2019).

Salmon are sensitive to changes in habitat quality, and many salmon populations are in decline, including those in the North Thompson River watershed, which spans the Thompson Plateau and Columbia Mountains of British Columbia (Bradford & Irvine, 2000; COSEWIC, 2017; Price et al., 2021; Wilson et al., 2022). The causes of the decline of North Thompson coho salmon are numerous, and include historic overfishing, climate change, and changes to stream habitat caused by human land use (Bradford & Irvine, 2000). By area, forest harvest is the most prevalent form of land use in the North Thompson. This mountainous watershed spans three main biogeoclimatic zones; Englemann Spruce – Subalpine Fir, Interior Cedar – Hemlock, and Interior Douglas Fir, with smaller patches of Montane Spruce and Sub-Boreal Spruce (GeoBC, 2022). The North Thompson is snowmelt dominated, forms part of the Fraser River watershed, and supports several species of salmon (coho, chinook, sockeye, and pink). In the context of declining populations of coho salmon in the North Thompson, it is important to improve our knowledge of how forest harvest influences salmon habitat.

Forest harvest alters land cover and watershed processes, changing physical habitat and stream temperature. Forest harvest alters physical habitat through changes in the delivery and storage of water, sediment, and woody debris in streams (Reid et al., 2019, 2020). Forest harvest can alter the infiltration and water storage capacity of soil, increasing runoff, which leads to increased fine sediment inputs (Smith & Redding, 2012). Increased runoff can also result in a higher frequency and magnitude of peak flows, causing a reduction in habitat complexity and a reduction in pool depth and pool habitat cover (Hartman et al., 1996; Tschaplinski & Pike, 2017). As the supply of sediment increases from overland transport, bank failures, and landslides caused by forestry, more fine sediment becomes available to fill pools, making them shallower (Hilton & Lisle, 1993; Lisle & Hilton, 1992). Forest harvest can also reduce the source of available large woody debris (LWD), leading to less instream LWD and fewer pools caused by LWD (Fausch & Northcote, 1992; Reid et al., 2020; Reid & Hassan, 2020). Removal of the forest canopy reduces shading, leading to increased solar radiation of streams and shallow groundwater, causing warming (Herunter et al., 2004; Macdonald et al., 2003b; Pollock et al., 2009; Tschaplinski & Pike, 2017). Forestry can also reduce summer baseflow and reduce channel stability, causing channels to become shallower and wider, making streams more sensitive to changes in air temperature and leading to warming (Gronsdahl et al., 2019; Tschaplinski & Pike, 2017). Reductions in physical habitat complexity and increased fine sediment cover caused by forestry can also cause warming by reducing hyporheic exchange (Herunter et al., 2004; Pollock et al., 2009; Story et al., 2003). Forestry alters habitat forming processes and alters the ability of forest cover to filter climate signals, ultimately leading to changes to physical habitat and stream temperatures.

Most of the existing research on the impacts of forestry on freshwater habitat has focused on small scale, intensively monitored watersheds. This does not necessarily inform practical management plans and techniques, especially in larger systems. The upstream watershed area of the streams featured in experimental watersheds is relatively small, most are smaller than 11 km^2, with many study watersheds smaller than 1 km^2 (Bladon et al., 2018; Macdonald et al., 2003b; Pollock et al., 2009; Tschaplinski & Pike, 2017). In addition to their small scale, many of these studies lack spatial replication which is required to make broader inference about forestry impacts on habitat. Furthermore, many existing studies are in coastal, rainfall dominated watersheds, leaving a knowledge gap for interior watersheds with snowmelt hydrographs. This leaves a need for research with spatial replication in larger watersheds to understand how the combined effects of landscape characteristics and forestry contribute to physical habitat and water temperature. There is also a need to understand how temperature varies across watersheds with a patchwork of forestry impacts, as opposed to below discrete cutblocks, and to tie harvest levels to changes in stream temperature to understand correlations for management decisions. Tributaries of the North Thompson River contain a gradient of forestry impacts and natural variation in watershed characteristics such as elevation and watershed size. This patchwork of forestry impacts and natural features provides a backdrop for this study.

In Chapter 2 of my thesis, I investigated how forestry activity and natural watershed components influence physical habitat and stream temperature. I used a geographic information system (GIS) to quantify forestry activities including forest harvest, road building, and stream crossings. I also quantified watershed characteristics such as watershed area, watershed elevation, and stream gradient. I installed water level loggers to record water level and temperature and conducted physical habitat surveys to quantify physical habitat components such as pool habitat cover, pool depth, sediment cover, and LWD abundance. I found that stream gradient explained most of the variation in physical habitat, with steeper streams associated with less pool cover, shallower pools, and less fine sediment cover. On average, low (1%) gradient stream reaches had 23% pool area and an average pool depth of 0.6 m, while high (3%) gradient reaches had 12% pool area and an average pool depth of 0.4 m. I also found a positive correlation between forest harvest and stream temperature. Stream temperature was higher in watersheds with a greater proportion of forest harvest, the average daily maximum temperature in the summer was 2°C higher in watersheds with high (35%) proportions of harvest than low (5%) proportions of harvest. This correlation was even stronger when looking at the proportion of the riparian area harvested, where the same increase from low to high was associated with an increase of 3.7°C. These results suggest that temperature increases associated with forest harvest could affect fish growth and survival.

Chapter 2. Forestry impacts on juvenile coho rearing habitat in the North Thompson, BC

2.1. Forestry and freshwater habitat

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 which take place on different spatial and temporal scales (Poff et al., 2006; Seelbach, 2006). Forestry can alter these processes, and in turn alter 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 & Pratt, 2007).

Forestry impacts on physical habitat, stream temperature, and streamflow can be transient or persistent. A 40-year study of stream habitat responses to forest harvest in Carnation Creek, British Columbia (BC), revealed physical habitat and temperature responses on different spatial and temporal scales (Hartman et al., 1996; Holtby, 1988; Tschaplinski & Pike, 2017). These studies found an immediate and persistent increase in stream temperature, and both immediate and delayed impacts to physical habitat structure, such as an increase in fine sediment and decreases in pool abundance and large woody debris (LWD) after extensive logging. Changes to physical habitat such as sediment and LWD are pulsed and can occur over decades, moving in a downstream direction (MacDonald & Coe, 2007; Reid et al., 2019, 2020). In headwater streams, temperature recovery after forest harvest can take 10 years or longer (Moore et al., 2005). Studies on the effects of forest harvest in the headwaters of Baptiste Creek, BC (Herunter et al., 2004; Macdonald et al., 2003a, 2003b; Story et al., 2003) found an immediate increase in stream sediment concentrations that declined to preharvest levels after three 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.

Watershed characteristics shape the state of the same variables that are used to observe impacts of forestry on fish. Watershed size, slope, elevation, gradient, and watershed geology have all been shown to influence habitat (Burnett et al., 2006; Carlier et al., 2018; Faustini et al., 2009; Isaak & Hubert, 2001; Richards et al., 1996). Stream and watershed gradient mediate many habitat-forming stream processes. As stream gradient increases so does hillslope connectivity, which means the sediment supply consists of larger sized particles, with steeper streams associated with larger mean particle size (Beechie & Sibley, 1997; Bracken et al., 2015; Lisle & Hilton, 1992). Stream gradient is also related to pool habitat cover and pool depth, with more pool habitat and deeper pools associated with lower gradient streams (Beechie & Sibley, 1997; Wohl et al., 1993). There are also relationships between gradient, LWD, and pool cover, and Beechie & Sibley (1997) found that pool habitat increased in medium and high gradient streams as LWD volume increased. Beyond physical habitat, natural components of watersheds such as elevation, upstream watershed area, lake cover, watershed aspect, and glacier cover regulate stream temperature (Isaak & Hubert, 2001; Moore, 2006; Subehi et al., 2009). Elevation moderates conditions such as air temperature, snowmelt, and glacier melt, and higher elevations are associated with lower temperatures (Beaufort et al., 2020; Caissie, 2006; Isaak & Hubert, 2001). Surface and bedrock geology regulates stream flow rates by influencing the infiltration rate (and therefore surface runoff and groundwater recharge) and storage capacity in the watershed, and by constraining subsurface flows (Carlier et al., 2018). 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). Upstream watershed area also has a direct influence on stream discharge, and there is a positive relationship between pool cover, pool depth, and watershed area (Burnett et al., 2006). 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.

The extent of forestry impacts could be modulated by watershed characteristics. Previous studies have found that stream temperature responses to forestry are greater in watersheds with more resistant lithologies (such as igneous rock) compared to watersheds with less resistant and more permeable lithologies (such as glacial till and sandstone) (Bladon et al., 2018). Similarly, the temperature in headwater streams is more responsive to harvesting in streams with fine textured substrate compared to those with coarse substrate (Janisch et al., 2012). In terms of physical habitat, forestry activities lead to a greater increase in suspended sediment in watersheds with less resistant lithologies (Bywater-Reyes et al., 2017). Thus, watershed characteristics have the potential to amplify or buffer forestry impacts.

Pacific salmon populations can be sensitive to changes in habitat quality and quantity across freshwater life stages. Physical habitat provides the structure needed to spawn, feed, and seek refuge from flows and predators, and stream temperature influences salmon life histories (Bailey et al., 2018; Braun et al., 2013; Hartman et al., 1996). For example, stream temperature regulates the amount of time it takes for eggs to hatch and fry to emerge (Braun et al., 2013; Quinn, 2018). For rearing juveniles, stream temperature regulates growth and influences summer and winter survival (Ebersole et al., 2006). High stream temperatures can also cause stress and mortality in spawning adults (Crossin et al., 2008; Martins et al., 2012a; Martins et al., 2012b). High streamflow caused by runoff can wash away eggs, crush redds with sediment, or deposit fine sediment overtop of redds, limiting water and oxygen exchange (Jensen et al., 2009; Montgomery et al., 1996; Smith & Redding, 2012). Both high and low flows can limit upstream migration (Bradford et al., 2011; Burnett et al., 2014; Rand et al., 2006). Large woody debris (LWD) provides juvenile salmonids with cover from predators and is a source of food for benthic invertebrates, which are an important source of food for juvenile salmonids (Roni & Quinn, 2001). Managing forestry activities to protect freshwater habitat for salmon is a lever available to land use managers that could help to mitigate climate change impacts on Pacific salmon (Schindler et al., 2008). Understanding how watershed characteristics and forest cover contribute to the state of physical habitat and stream temperature will allow management decisions to be made that could buffer the predicted impacts of climate change.

The objective of this study is to understand how watershed characteristics and forest harvest within watersheds influence habitat. I examined physical habitat features, stream temperature, and flow in 28 tributaries of the North Thompson River watershed, British Columbia, Canada. I also examined forestry activities and watershed characteristics for the entire study catchments. I hypothesized that the proportion of a watershed that has been harvested and the density of forestry activities will influence *i)* physical habitat metrics and *ii)* water temperature at each study site, while accounting for watershed metrics which have been shown to be key drivers of habitat. A list of *a priori* hypotheses and predictions can be seen in Table 2.1, which explains the proposed mechanisms and directions of each impact. I test multiple hypotheses with multiple statistical models that contain habitat metrics as response variables, and watershed and land use metrics as explanatory variables.

Table 2.1 Hypothesis table showing response variables, response metrics, explanatory variables, and the hypothesized impacts these will have.

Response variable	Metric	Explanatory variable	Hypothesized mechanism of impact on response variable	Reference
Fine sediment	1) % cover of fines in pools 2) % 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 & 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.	(Beechie & Sibley, 1997; Bracken et al., 2015; Buffington et al., 2004; Lisle & Hilton, 1992)
		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.	(Beechie and Sibley, 1997)

Response variable	Metric	Explanatory variable	Hypothesized mechanism of impact on response variable	Reference
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.	(Lassettre and Kondolf, 2012; Macdonald et al., 2003a)
		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.	(Beechie and Sibley, 1997; Ruiz-Villanueva et al., 2016)
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 & Northcote, 1992; Mellina & Hinch, 2009; Tschaplinski & 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; Mellina and Hinch, 2009; Macdonald et al., 2003a; Tschaplinski & Pike, 2017)

Response variable	Metric	Explanatory variable	Hypothesized mechanism of impact on response variable	Reference
		Road density	Roads can reduce pool cover and depth through increased peak flows.	(Fausch and Northcote, 1992; Mellina and Hinch, 2009; Macdonald et al., 2003a; Tschaplinski & Pike, 2017)
		Stream gradient	Low gradient streams are associated with higher pool cover and deeper pools.	(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 streambank	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 & 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.	(Macdonald et al., 2003a; Murphy, 1986; Tschaplinski & 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, 1986; Tschaplinski & 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 bank-full width to bank-full 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, 1986; Tschaplinski & Pike, 2017)

Response variable	Metric	Explanatory variable	Hypothesized mechanism of impact on response variable	Reference
		Stream crossings	Stream crossings can cause an increase in WDR through increased flow velocity which leads to erosion, reduced bank stability, and wider streams.	(Macdonald et al., 2003a; Murphy, 1986; Tschaplinski & 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.	(Macdonald et al., 2003a; Murphy, 1986; Tschaplinski & Pike, 2017)
		Stream gradient	Higher gradient is associated with shallow, narrow streams.	(Beechie and Sibley, 1997; Burnett et al., 2006; Faustini et al., 2009; Richards et al., 1996)
		Watershed size	Larger watersheds are associated with wider, deeper streams.	(Burnett et al., 2006; Faustini et al., 2009; Richards et al., 1996)
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.	(Bladon et al., 2018; Gronsdahl et al., 2019; Herunter et al., 2004; Macdonald et al., 2003b; Pollock et al., 2009; Tschaplinski & Pike, 2017)
		Watershed aspect	South facing streams and watersheds are associated with higher temperatures as they are exposed to more solar radiation.	(Smith & 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.	(Beaufort et al., 2020; Isaak & Hubert, 2001)
		Summer mean discharge	Lower discharge is associated with more thermal sensitivity.	(Caissie, 2006)

2.2. Methods

2.2.1. Study location and site selection

This study was conducted in 28 tributaries of the North Thompson, with sites located between Kamloops and Valemount in the interior of British Columbia (Figure 2.1 and Table 2.2). This snowmelt-dominated, mountainous watershed spans three main biogeoclimatic zones; Englemann Spruce – Subalpine Fir, Interior Cedar – Hemlock, and Interior Douglas Fir, with smaller patches of Montane Spruce and Sub-Boreal Spruce (GeoBC, 2022). By area, forestry is the most prevalent form of land use in these watersheds, in addition to 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 2.2 and 2.3). 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 to 6th Strahler stream order streams, mostly located in lower gradient reaches (<5%) where juvenile coho rear. I selected streams based on historical coho presence, site accessibility, the presence of juvenile rearing habitat, and local knowledge. Historical coho presence was based on recorded observations from the Interior Fraser Coho Recovery Team (Interior Fraser Coho Recovery Team, 2006), Fisheries and Oceans Canada (DFO) stock reports, and data sourced from the province of B.C. through the EcoCat Ecological Reports Catalogue (https://a100.gov.bc.ca/pub/acat/public/welcome.do). Thus, all sites were known to be current or historical coho 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 coho rearing habitat (which was visually assessed through field observations) 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). Following a stratified random sampling procedure, each reach was divided into four equal sections, and each section contained three randomly assigned transects.

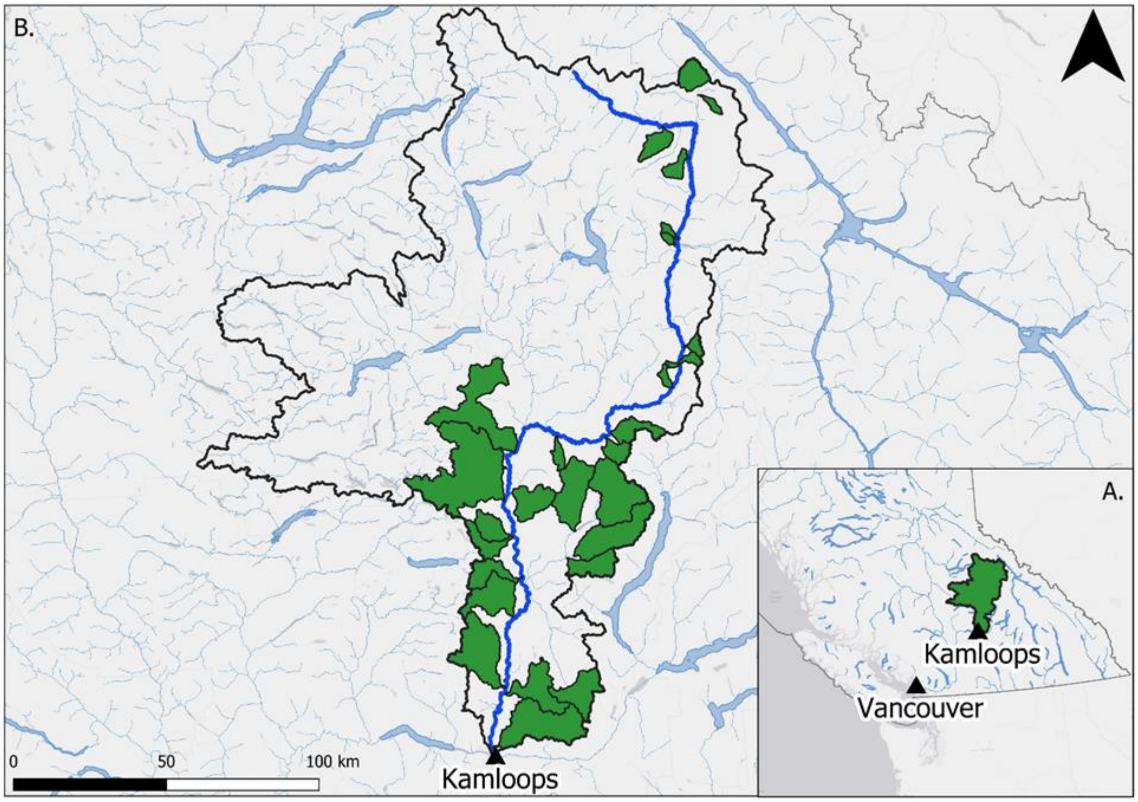


Figure 2.1 (A.) Map of the North Thompson watershed (shown in green) in British Columbia. (B.) The watersheds of the 28 tributaries of the North Thompson river that make up this study (shown in green), outlined by the North Thompson watershed in black, with the North Thompson River represented by the bold blue line.

2.2.2. Physical habitat

Physical habitat surveys took place during low flow periods (July to August 2019 and 2020) to measure a suite of habitat and explanatory variables (Table 2.1). I measured large woody debris (LWD) following protocols outlined by (Roni & Quinn, 2001). I counted all pieces of wood within the bankfull area of each channel with a length ≥ 1.5 m and a diameter ≥ 10 cm. The length and representative diameter of each piece of wood were recorded. These measurements were converted to volume (m^3) using the formula:

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

Where V is volume, r is radius, and h is height. LWD volume by area was measured as the sum of individual LWD volume divided by reach area (m^2). 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 calculate the distance (accuracy ± 0.2 m) and slope (accuracy ± 0.25°) to another surveyor with a surveying rod. This 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 PM5/360PC clinometer (Bain & Stevenson, 1999; Heitke et al., 2008). 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 meter in length and had 15 centimeters of overhanging bank (Moore et al., 2014).

Macrohabitat units were classified according to Bain & Stevenson (1999). Once classified, the length, width, and depth of each unit was 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 formula.

$$d_r = d_m - d_t$$

Where d_r is the residual depth, d_m is the maximum depth and d_t is the tail depth. Percent cover of sediment 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 twelve estimates per site (four from each bank and thalweg). Percent cover of sediment was also estimated for each unit of pool habitat. The percent 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 to percent cover reference diagrams, and sediment particles were measured in the field using calipers to ensure accurate measurements. Sediment particle size classes were adapted from a modified Wentworth classification described in Bain & 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 percent cover of fines and mean percent cover of fines in pools was calculated for each site.

Table 2.2 Stream characteristics. Streams are listed from North to South

Site	Reach Length (m)	Watershed Area (km²)	Stream Order	Bankfull Width (m)
Albreda	280	72	6	12
Dora	160	16	3	4
Canvas	500	61	4	15
Chappell	220	40	4	7
Cook	140	13	3	5
Cedar	120	11	3	5
Tumtum	140	23	4	8
Shannon	220	19	4	9
Avola	36	6	3	2
Wirecache	80	18	4	8
Foghorn	*	28	3	*
Reg Christie	200	76	4	7
Chuck	140	50	3	5
Mann	440	312	5	22
Dunn	300	109	5	13
Lemieux	500	532	6	20
Fennell	400	249	5	22
Darlington	180	72	5	11
Harper	200	186	5	25
Lindquist	160	69	5	7
East Barriere	240	209	4	9
Haggard	180	96	4	8
Peterson	180	84	4	6
Fishtrap	240	157	6	11
Jamieson	260	239	6	8
Louis	300	134	6	13
Heffley	160	147	5	5
Paul	140	278	5	5

* No physical habitat survey conducted at Foghorn.

2.2.3. Stream temperature and discharge

Stream temperature and water level were recorded hourly using Hobo U20 Water Level loggers with an accuracy of ±0.44 °C and ± 0.004 m. These loggers were suspended in ABS stilling wells fixed to large boulders or attached to steel T-posts.

(EPA, 2014). Additional temperature, water level, and discharge data for one site (Lemieux) was sourced from the Environment and Climate Change Canada Real-time Hydrometric Data website (station number 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 by plotting water temperature with air temperature and water level (see below) 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 – 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/s) was measured at each site 1 - 4 times annually using standard velocity-area methods (Bain & Stevenson, 1999; WMO, 2010b). Discharge was measured using a HACH FH950 portable velocity meter fitted with a EM950 electromagnetic velocity sensor (accuracy ± 0.015 m/s). Stage-discharge relationships were estimated using the formula:

$$Q = C(h - a)^n$$

where Q is discharge in m^3/s, h is stage in m, a is stage at zero flow in m, and C and n are coefficients (WMO, 2010a). a was estimated graphically by plotting instantaneous discharge measurements and corresponding stage heights, plotting a line of best fit and extending the line to the intercept; coefficients C and n were determined by fitting a non linear least squares regression in R (R Core Team, 2021). Once a stage-discharge relationship was established for each site, it was applied to the record of continuous hourly stage to estimate continuous hourly discharge. Mean discharge for the summer low-flow period was generated by averaging hourly estimates from July 15 – August 15, 2021. Continuous summer discharge could not be estimated for 4 sites (Avola, Foghorn, Jamieson, and Shannon); for these sites, mean discharge was calculated as the mean of instantaneous discharge measurements made in July and August of 2020 and 2021.

2.2.4. Land use and watershed characteristics

Watershed area and land use metrics were established using publicly available datasets published by the province of BC and the federal government. Watersheds were delineated using the Freshwater Atlas (FWA) Watersheds dataset (GeoBC, 2019). This dataset is derived from topographic base maps at a scale of 1:20,000, with watersheds delineated by elevation. I used QGIS to calculate the area of each watershed in km^2. Watershed elevation, slope, and aspect were calculated from a Digital Elevation Model (DEM) 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. Streams and lakes were mapped using the Stream Network and Lakes layers in the FWA.

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. I extracted the entire extent of cutblocks located within the watershed. I 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 & 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. 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.

Table 2.3 A list of the quantified response and explanatory variables, along with their description. The mean and range represent the mean and ranges of site means.

Metric	Description	Mean	Range
Response variables			
Undercut bank	% of streambank with at least 15 cm of overhanging bank	10	0 - 31
Mean bankfull width	Average perpendicular width of the stream at bankfull, in m	10.1	2.1 - 25.0
Bankfull depth	Average maximum vertical depth of a stream at bankfull, in m	0.9	0.5 - 1.5
Pool habitat cover	% cover of pool habitat	19	2- 39
% cover of fines in pools	% cover of sediment with a diameter of < 2 mm in pool habitat	42	0 - 100
Pool residual depth	Pool residual depth, in m	0.6	0.2 - 1.2
% cover of fines in reach	% cover of sediment with a diameter of < 2 mm in the sediment plots	31	0 - 99
Width to depth ratio	Ratio of stream bank-full width to bank-full height	11.6	2.9 - 22.2
LWD volume	Volume of large woody debris in stream area, in m^3/m^2	0.01	0 - 0.07
Average Daily Maximum (ADM)	Mean maximum temperature in for each day from July 15 – August 15, in °C	15.0	10.6 - 19.7
Average Daily Range (ADR)	Average temperature range (daily max - daily min) for each day from July 15 – August 15, in °C	2.6	1.3 - 4.7
Accumulated Thermal Units (ATU)	Cumulative daily mean for July 15 – August 15, in °C	437	290 - 563
Summer maximum	Maximum stream temperature in for July 15 – August 15, in °C	16.7	12.1 - 22.6
Summer mean	Mean stream temperature for July 15 – August 15, in °C	13.7	9.1 - 17.6
Summer range	The difference between the maximum temperature and minimum for July 15 – August 15, in °C	6.0	3.9 - 9.3
Explanatory variables - Watershed characteristics			
Reach gradient	Average gradient of the study reach, in %	1.6	0.1 - 4.5

Metric	Description	Mean	Range
Watershed area	The area of the upstream drainage basin for each site in km^2	121	6 - 532
Watershed elevation	The average elevation in the watershed in m	1357	1034 - 1833
Watershed aspect	The average direction the watershed slope is facing, in degrees.	160	101 – 179
Explanatory variables - Forest harvest			
Proportion of watershed harvested	% of each watershed that has been harvested from 1970 - 2019	24	1 - 59
Road density	The length of roads in each watershed divided by watershed area (km/km^2)	1.8	0.3 - 3.2
Crossings	Number of stream crossings in each watershed divided by watershed area ($crossings/km^2$)	1.4	0.2 - 2.7
Proportion of riparian area harvested	% of each 50 m stream buffer that has been harvested from 1970 - 2019	18	0 - 41

2.2.5. Analysis

Physical habitat

The relationship between forestry land use and physical habitat was explored for 25 sites. One site (Avola) was excluded from this analysis, as the length of the reach that was sampled was limited by the presence of a beaver pond upstream and a controlled pond downstream, resulting in a reach that lacked suitable replication and is influenced by the water level of the downstream pond. Another site (Wirecache) was excluded due to prominent and frequent influence from the mainstem of the North Thompson on the sampled reach. I did not survey physical habitat at Foghorn (site). For the remaining sites (n = 25) I tested for effects of watershed characteristics and forestry activities on habitat components using multiple linear regression models (Table 2.1 and Figure 2.2).

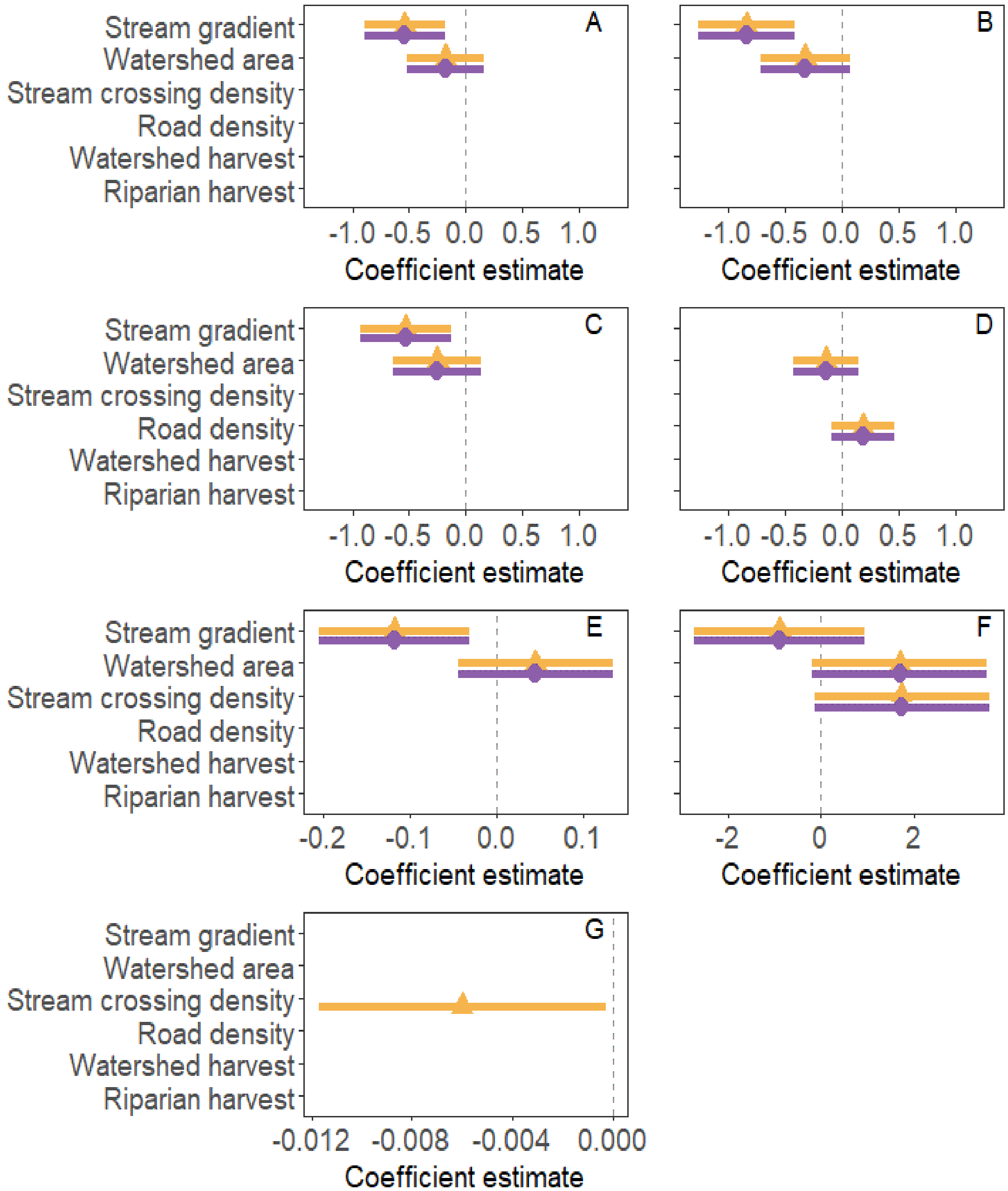


Figure 2.2 Model averaged coefficients from the physical habitat analysis for pool habitat cover (A), fine sediment cover in the reach (B), fine sediment cover in pools (C), undercut bank (D), pool residual depth (E), width to depth relationship (F), and LWD volume (G). Orange triangles are coefficients 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.

I 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 (Tables 2.1 and 2.3). 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. The explanatory variables were standardized by subtracting the mean and dividing by one standard deviation to allow for comparison of the coefficients (Cade, 2015). The response variables were: percent undercut bank, percent cover of fine sediment, percent cover of pool habitat, percent cover of fine sediment in pools, mean residual depth of pools, LWD m^3/m^2 of stream habitat, and mean width to depth ratio (Tables 2.1 and 2.4). The response variables for undercut bank, percent cover of fine sediment, percent cover of pool habitat, and percent cover of fine sediment in pools area were modeled on a beta distribution as they are proportional data (Douma & Weedon, 2019; Ferrari & 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 & Zeileis, 2010).

Temperature

I 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 21 sites. Fishtrap, Harper, Jamieson, and Peterson were excluded from this analysis due to missing data. Wirecache was excluded due to temperature influence from the mainstem of the North Thompson. Dunn and East Barriere were excluded due to the temperature influence of large lake outflows within two kilometers upstream. For the remaining sites (n = 21) I tested the hypotheses using multiple linear regression models. I built hypothesis-driven global models for the temperature response metrics of average daily mean, average daily maximum, average daily range, accumulated thermal units, summer mean, and summer maximum. Like the approach for physical habitat, I built global models to test temperature hypotheses that consisted of the response metric and a suite of explanatory land cover and forestry variables, with variables for riparian harvest and watershed harvest included in separate models (Figure 2.3).

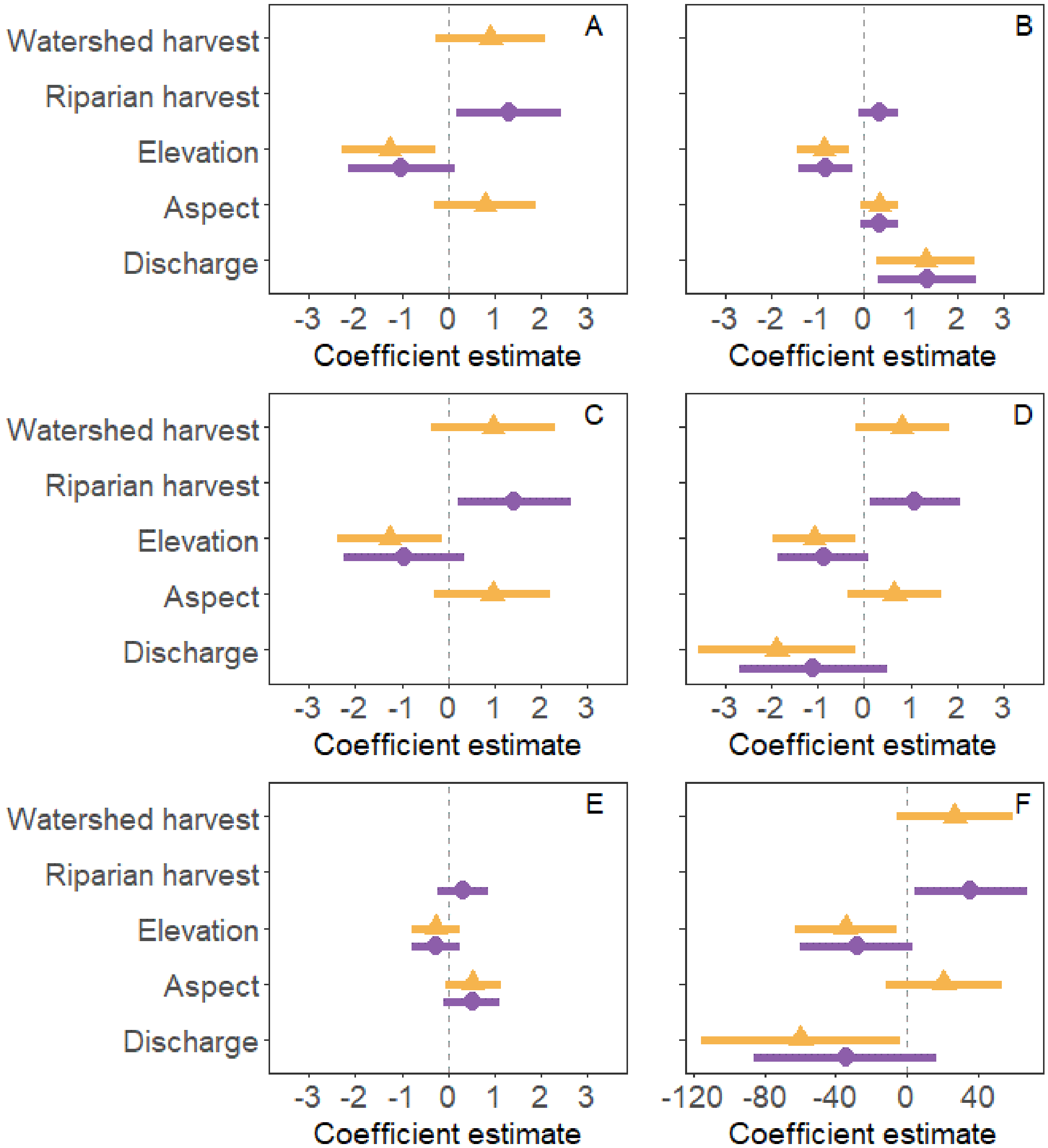


Figure 2.3 Model averaged coefficients from the temperature analysis for average daily mean (A), average daily range (B), summer maximum temperature (C), summer mean temperature (D), summer temperature range (E), and accumulated thermal units (F). Orange triangles are coefficients from models that included the proportion of the watershed harvested as an explanatory variable, while purple circles are from models that included the proportion of riparian area harvested.

Collinearity

I explored collinearity among explanatory variables using a correlation matrix and the Variance Inflation Factor (VIF). To avoid issues that may arise from collinearity I removed variables with a VIF greater than 5 from the global model (Zuur et al., 2010). I 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

I used Akaike's information criterion corrected for small sample sizes (AICc) to evaluate support for the two sets of different candidate models, one of which consisted of all combinations of variables for the watershed, the other for the riparian area (Burnham & Anderson, 2004). I used a score of Δ AICc ≤ 2 to identify the set of top models, as models with these values are roughly equal, and model averaged using the natural method to account for model uncertainty in estimates of coefficients (Barton, 2020; Burnham & Anderson, 2004). 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.

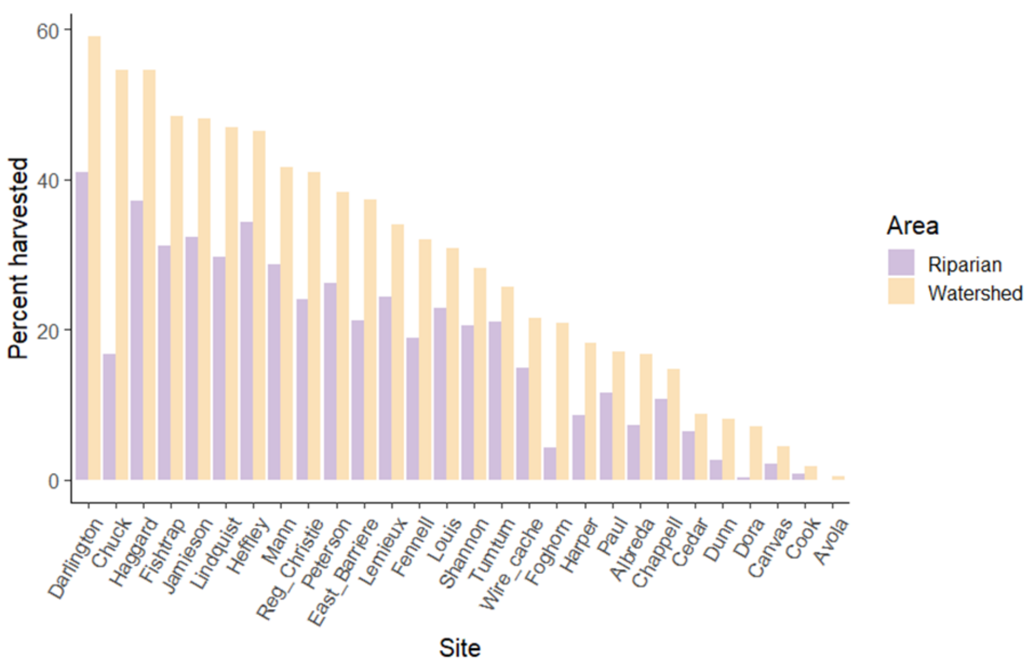


Figure 2.4 Total forest harvest from 1970 to 2019 represented as a percentage of each study watershed and each riparian area.

2.3. Results

Landscape and forestry variables varied widely among sites (Table 2.3). Forest harvest, measured as the proportion of the watershed harvested in the 50-year period between 1970 and 2019 ranged from 1 to 59% (Figure 2.4). Forest harvest ranged from 0to 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 also varied. Pool habitat cover ranged between 2 and 40% of the study reaches and average pool residual depth ranged from 0.2 to 1.2 m. Average fine sediment cover in the random plots ranged between 1 and 99% and average fine sediment cover in pools ranged from 0 to 100% at each site. LWD volume ranged from 0 to 0.7 m^3 of wood per m^2 of stream. Stream temperature varied between streams, average summer stream temperature ranged from 9 to 18°C, and the maximum summer stream temperature ranged from 12 to 23°C. The average daily maximum temperature ranged from 10.6 to 19.7°C and the average daily temperature range varied between 1 and 5°C, while accumulated thermal units from July 15th to August 15th ranged from 290 to 563°C. Thus, stream habitat characteristics as well as landscape and forestry intensity varied across the study sites, providing potential strong contrasts.

Table 2.4 AICc table showing each model for each physical habitat response variable. 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. Riparian harvest models are on top; watershed harvest models are below.

Response Variable	Int	Rip Har	Cros	Grad	Road	Wsd	R^2	df	Log Lik	Δ AICc	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
RD	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
WDR	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
Cut 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

Response Variable	Int	Wsd Har	Cros	Grad	Road	Wsd	R^2	df	Log Lik	Δ AICc	W_i
RD	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
WDR	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
Cut 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

Notes: Int is intercept, Rip Har is the percent of riparian area harvested, Wsd Har is the percent of watershed harvested, Cros is stream crossing density, Road is road density, and Wsd is watershed area.

Table 2.5 AICc table showing each model for each temperature response variable. 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. Riparian harvest models are on top; watershed harvest models are below.

Response variable	Int	Asp	Dis	Elev	Rip Har	R^2	df	Log Lik	Δ AICc	W_i
ADM	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
ADR	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
ATU	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 max	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

Response variable	**Int**	**Asp**	**Dis**	**Elev**	**Wsd Har**	R^2	**df**	**Log Lik**	**Δ AICc**	W_i
ADM	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
ADR	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
ATU	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 max	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

Notes: Int is intercept, Asp is aspect, Dis is discharge, Elev is elevation, Rip Har is the percent of riparian area harvested, and Wsd Har is the percent of watershed harvested.

2.3.1. Physical habitat

Natural landscape metrics were much stronger descriptors of physical habitat than metrics of forest harvest. Specifically, gradient was the best descriptor of most habitat metrics and explained between 18 and 32% of the variation in physical habitat metrics (Table 2.3). Steeper gradient stream reaches had less pool area, and pools were shallower with more fine sediment (Figure 2.2, Figure 2.5). Based on model average predictions, 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 (Δ AICc = 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 2.4). There was also a negative effect of gradient on the width to depth ratio, but the uncertainty was larger than the estimate (Figure 2.2). Watershed size was included in the top model set (Δ AICc < 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 (Figure 2.2).

The only clear effect of forestry on physical stream habitat was that of stream crossing density on LWD volume and channel width to depth ratio (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 (Δ AICc = 0) for LWD volume (R^2 = 0.16). Stream crossing density was also the only explanatory variable included in the top model

(Δ AICc = 0) for WDR (R^2 = 0.18). None of the top models (Δ AICc = 0) included watershed size, road density, or proportion of the watershed or riparian area harvested (Table 2.4).

2.3.2. 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 (Figure 2.3, Figure 2.6). 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 ADM 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 ADM 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 positive effect of riparian harvest was significant for average daily range, summer mean, summer range, and accumulated thermal units.

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 (Δ AICc = 0) candidate models the average daily maximum (ADM) temperature (R^2 = 0.27), the average daily range (ADR) in temperature (R^2 = 0.34), accumulated thermal units (ATU) (R^2 = 0.28), and summer maximum (R^2 = 0.21) and mean temperatures (R^2 = 0.28) (Table 2.5). For the model set that included riparian area harvest, elevation was only included in the top (Δ AICc = 0) model for ADR (R^2 = 0.34). Higher mean summer discharge was associated with higher daily temperature ranges. Discharge was included in the top (Δ AICc = 0) candidate models for ADR 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 (Δ AICc = 0) candidate models for summer temperature range (R^2 = 0.12).

2.4. Discussion

I examined 28 streams to understand how forest harvest and watershed characteristics influence physical habitat and temperature. Gradient explained more variation in physical habitat than any other natural or forestry related explanatory variable. In contrast, I found strong effects of both forestry and watershed characteristics on stream temperatures. Streams with higher proportions 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 are a primary driver of water temperatures.

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 that twice the effect of watershed size on these variables (Figure 2.2). These results were predicted based on previous findings of negative relationships between stream gradient and pool depth and cover (Beechie & Sibley, 1997; Wohl et al., 1993). 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 (Hupp & Osterkamp, 1996; Wohl et al., 1993). Pools also form from changes to water velocity caused by channel narrowing and stream slope (Chartrand et al., 2018). My findings match previous findings of positive relationships between gradient and sediment size (Beechie & Sibley, 1997; Buffington et al., 2004). Higher gradient streams are more connected to adjacent hillslope and have a sediment supply with larger grain sizes, while lower gradient streams have more fine sediment available to fill pools (Bracken et al., 2015; Lisle & Hilton, 1992; Montgomery & Buffington, 1997). Higher gradient streams have higher shear stress, which allows smaller sediment particles to be easily transported and deposited in lower gradient sections (Beechie & Sibley, 1997; Wohl et al., 1993). While past studies have found forestry has decreased pool cover and depth (Chen & Wei, 2008; Tschaplinski & Pike, 2017) and increased fine sediment cover (Herunter et al., 2004; Tschaplinski & Pike, 2017), I did not find any relationship between forestry and these metrics in my comparative study. My findings suggest that gradient is a key master factor in broadly determining physical habitat, and monitoring efforts that look to quantify forestry effects should account for this (Montgomery & Buffington, 1997).

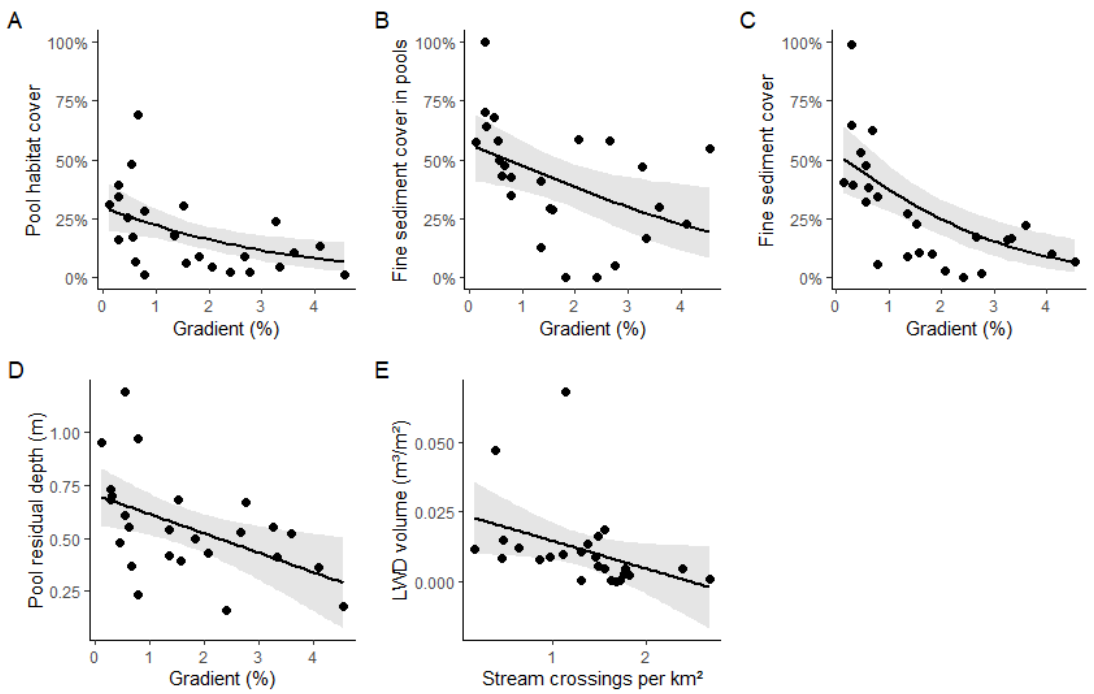


Figure 2.5 Observations and model predictions for the effect of gradient on (A) pool habitat cover, (B) fine sediment cover in pools, (C) fine sediment cover in random plots in the reach, and (D) residual depth in pools. (E) Observations and model predictions for the relationship between stream crossing density and LWD volume. The shaded regions represent 95% confidence intervals.

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. I 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. Indeed, many past studies (Macdonald et al., 2003b; Pollock et al., 2009; Tschaplinski & Pike, 2017) that found relationships between stream temperature and watershed harvesting took place in small watersheds, my research suggests this relationship may get weaker when the stressor is farther away. The effects of forestry on stream temperature were broadly consistent with the literature, where many studies (Macdonald et al., 2003b; Pollock et al., 2009; Story et al., 2003; Tschaplinski & 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. This study adds to the literature by offering spatial replication among midsize watersheds with different harvest levels and a patchwork of harvest dates (see Figure 2.4). These data advance understanding of how forestry can exacerbate temperature risks to thermally-sensitive fish, a critical issue in this warming world.

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 (BACI) studies that quantified the state of the physical habitat before forestry began, and continued monitoring for decades after the forestry impacts took place (Macdonald et al., 2003a, 2003b; Reid et al., 2019; Tschaplinski & Pike, 2017). My spatial comparative approach may miss episodic disturbances that could result from forest harvesting, making it challenging to link physical habitat to disturbances (Reid & Hassan, 2020). This was seen in Carnation Creek, where infrequent pulses of sediment inputs from landslides and bank failures have slowly 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 (Reid et al., 2019, 2020; Tschaplinski & Pike, 2017). Further, it is likely that watershed characteristics may vary so dramatically across different locations that it could be challenging to detect forestry effects. While I found watershed characteristics explained most of the variation in physical habitat, temperature was consistently explained by forest harvest. This indicates 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, while temperature may be more sensitive to stressors, and changes to stream temperature may be more persistent, and therefore a more reliable metric for detecting impacts.

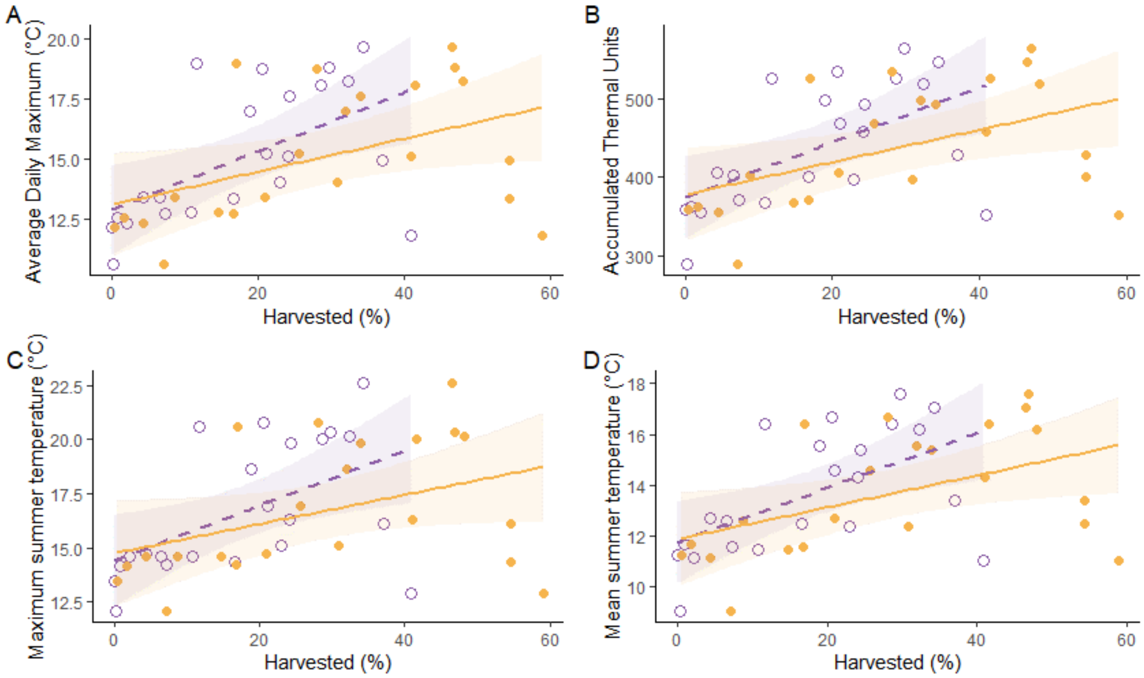


Figure 2.6 Observations and model predictions of the relationship between harvested area and (A) average daily maximum temperature, (B) Accumulated thermal units, (C) maximum summer temperature, and (D) mean summer temperature in the study streams (n = 22). The solid orange points and lines are for the 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.

The observed temperature increases associated with forest harvest were 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. This temperature is higher than the optimal rearing temperature of juvenile coho of 12°C to 15°C (Richter & 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 & 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 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. The population of coho 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 (Arbeider et al., 2020; Bradford & Irvine, 2000; COSEWIC, 2017). More broadly, Pacific salmon are threatened by climate change and freshwater habitat degradation (Andrew & Wulder, 2011; Finn et al., 2021; Munsch et al., 2022; Schindler et al., 2008). Stream temperatures are warming throughout the range of Pacific salmon, including in the Fraser watershed, and my research suggests that climate-induced temperature increases could be amplified by forest harvest or buffered by maintaining forest cover (Isaak et al., 2012; Islam et al., 2019; Kaushal et al., 2010; Mantua et al., 2010). While ocean health and climate change are long-term, international issues, managing watersheds for stream temperature and physical habitat is a management lever that can be pulled by federal and provincial government agencies.

In examining connections between forestry activity and stream habitat in 28 streams, I found stream gradient acts as a template for physical habitat. I also found that as forest harvest increases in watersheds and riparian areas, so do stream temperatures. In the context of declining salmon populations and increasing stream temperatures from climate change, the relationship between forest harvest and stream temperature presents both problems and opportunities. Beginning with the problem, 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 affected to some degree by forestry. The opportunity is that managing forests on a watershed scale is an available management lever that can possibly dampen climate signals and restore resilience to climate change by restoring the climate filtering functions of intact forests. This study offers insight on how watershed characteristics and forestry influence stream habitat. Future research should aim to develop temperature benchmarks and assess how distance mediates the impacts of forest harvest on stream temperatures.

Chapter 3. Conclusion

Forestry activity and watershed characteristics drive the contemporary state of physical instream habitat and stream temperature. In Chapter 2 of my thesis, I quantified these relationships and showed that on a watershed scale stream gradient drives physical habitat and forest harvest drives stream temperatures. In this chapter I present the implications of these findings in the context of forestry practices in British Columbia.

Maximum summer stream temperatures increased as the proportion of harvest area in a watershed increased. Increases in stream temperatures in salmon-bearing streams are concerning, especially in the context of climate change (Reid et al., 2019; Schindler et al., 2008). While the impact of forestry and changes in forest cover on stream temperatures threaten salmon, as discussed in Chapter 2, this relationship also offers opportunities. Climate change is a difficult problem to solve, one that will take international cooperation, resources, and time. Land cover and use is one management lever that can be pulled to buffer streams from climate stress; modified forestry practices could help offset some of the impacts of climate warming.

Forestry in British Columbia is currently managed on a large scale in the form of Timber Supply Areas (TSAs). The chief forester of BC determines the allowable annual cut for each TSA on a 10-year basis (Government of British Columbia, 2017). TSAs are large, there are 37 in BC, and a single TSA (Kamloops) is 6442 km^2 and contains the watersheds of all 28 of the streams from my thesis study (Figure 3.1). Forestry planning in BC is both broad and specific; the *Forest and Range Practices Act* (FRPA) provides regulations that cover individual cutblocks and entire TSAs; what is left aside is an intermediate management scale (Forest Practices Board, 2019; Government of British Columbia, 2021). This leads to the uneven distribution of forest harvest impacts in small and midsize watersheds. This uneven distribution is seen in the range of harvested area in the watersheds discussed in this study, which varies from 1% of the watershed to 59%. Planning at the scale of midsize watersheds (fish-bearing tributaries) could allow for more control over the distribution of forestry impacts, and ultimately over stream temperatures in the context of climate change.

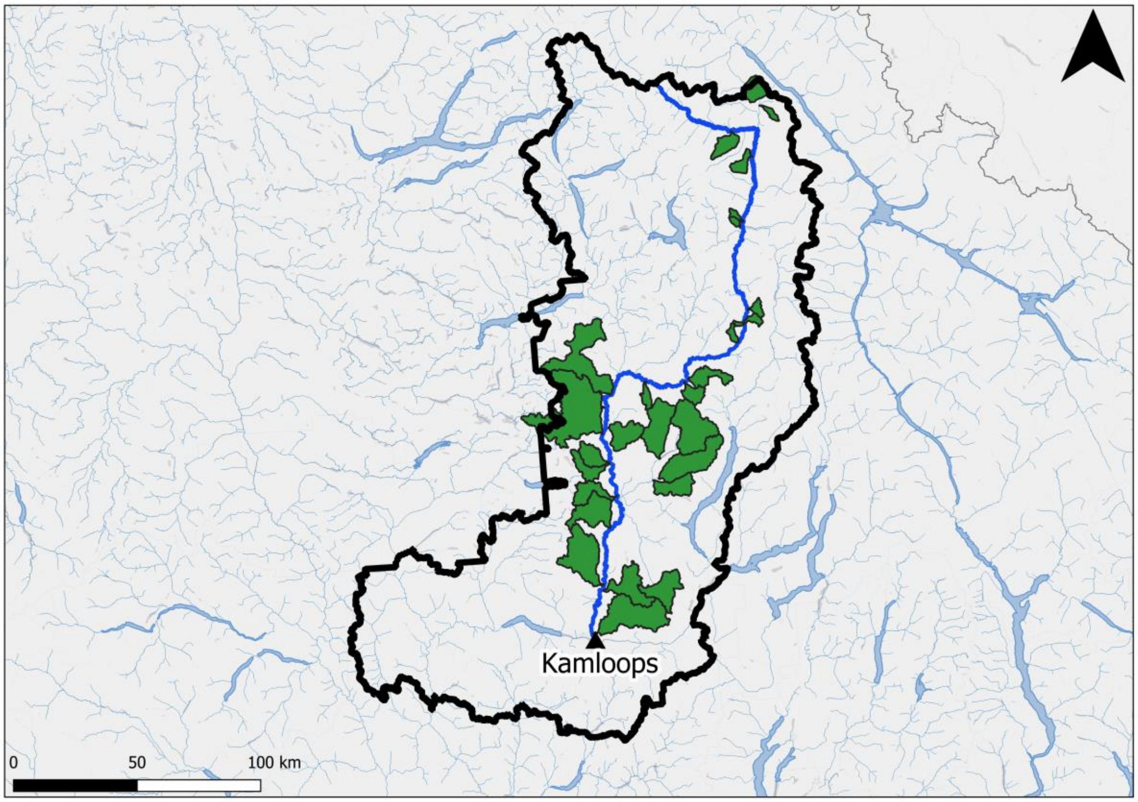


Figure 3.1 Map showing the Kamloops Timber Supply Area (TSA) outlined in by a bold black line, and the study watersheds in green.

The relationship between riparian area harvest and stream temperature is stronger than the relationship between watershed area harvest and stream temperature. The difference in average daily maximum summer stream temperatures between watersheds with 5% of the riparian area harvested and those with 35% harvested was 3.7°C, the same difference for the proportion of the entire watershed harvested was 2°C. The majority (72%) of the riparian harvest measured in this study was surrounding first-order streams (Figure 3.2). First-order streams are unlikely to be protected under current forestry regulations, which do not require any retention of trees around non fish-bearing streams that are less than 3 m wide or of fish-bearing streams that are less than 1.5 m wide (Forest Practices Board, 2018; Government of British Columbia, 2021). While past studies have found downstream cooling of headwater streams after re-entering shaded reaches, downstream shading before a confluence cannot be assumed in the mosaic of forest impacts in watersheds (MacDonald & Coe, 2007; Moore et al., 2005; Story et al., 2003). Protecting the riparian buffer around headwater streams could be a management lever to buffer stream temperatures. Research from the Baptiste watershed in the interior of British Columbia showed that a high retention riparian buffer separating a headwater stream from a cutblock was associated with a 1°C increase in average weekly summer temperatures, while low retention riparian areas were associated with temperature increases of 2°C to 4°C in the first three years after forest harvesting (Macdonald et al., 2003b). Further research into the cumulative impacts of headwater stream warming is required.

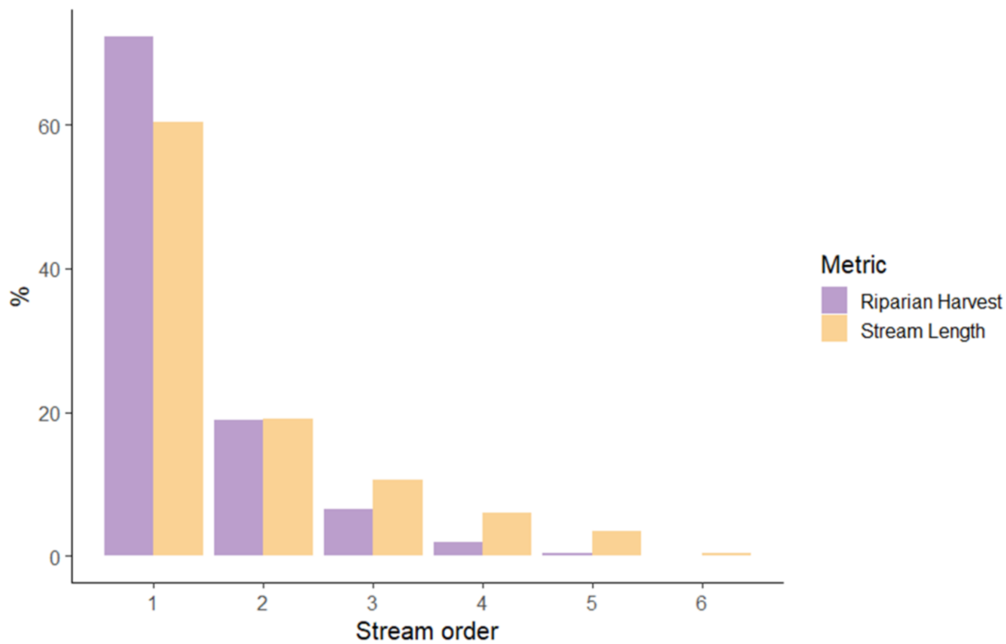


Figure 3.2 The percent of riparian harvest by stream order for the 28 streams from Chapter 2.

The impacts of forestry on stream temperature and physical habitat persist over decades (Macdonald et al., 2003b; Tschaplinski & Pike, 2017). Changes to physical habitat such as sediment and large woody debris are pulsed and can occur over decades, moving in a downstream direction (MacDonald & Coe, 2007; Reid et al., 2019, 2020). In headwater streams, temperature recovery after forest harvesting can take 10 years or longer (Moore et al., 2005). In the Baptiste watershed in central BC, stream temperatures did not return to pre-harvest levels in the 13 years of post-impact monitoring that occurred (Figure A1 and A2). I replicated this study 22 years after harvesting occurred and found that while stream temperatures were within the range of pre-harvest levels, there was still more seasonal variation than there was pre-harvest (Figure A1 and A2). As impacts to physical habitat and stream temperature are persistent and unevenly distributed across watersheds, the current framework of 10-year plans for annual allowable cut in large TSAs may not have enough temporal flexibility or spatial resolution to address forestry impacts on tributaries like those in this study. Long-term land use planning at the appropriate watershed scale will allow managers to offset impacts of climate change and land use stressors to promote freshwater salmon habitat.

References

Andrew, M. E., & Wulder, M. A. (2011). Idiosyncratic responses of Pacific salmon species to land cover, fragmentation, and scale. *Ecography*, *34*(5), 780–797. https://doi.org/10.1111/j.1600-0587.2010.06607.x

Arbeider, M., Ritchie, L., Braun, D., Jenewein, B., Rickards, K., Dionne, K., Holt, C., Labelle, M., Nicklin, P., Mozin, P., Grant, P., Parken, C., & Bailey, R. (2020). *Interior Fraser Coho Salmon Recovery Potential Assessment* (No. 2020/025; p. xi + 211). DFO Can. Sci. Advis. Sec. Res. Doc.

Bailey, C. J., Braun, D. C., McCubbing, D., Reynolds, J. D., Ward, B., Davies, T. D., & Moore, J. W. (2018). The roles of extrinsic and intrinsic factors in the freshwater life-history dynamics of a migratory salmonid. *Ecosphere*, *9*(9), e02397. https://doi.org/10.1002/ecs2.2397

Bain, M. B., & Stevenson, N. J. (Eds.). (1999). *Aquatic habitat assessment: Common methods*. American Fisheries Society.

Beaufort, A., Moatar, F., Sauquet, E., Loicq, P., & Hannah, D. M. (2020). Influence of landscape and hydrological factors on stream–air temperature relationships at regional scale. *Hydrological Processes*, *34*(3), 583–597. https://doi.org/10.1002/hyp.13608

Beechie, T. J., & Sibley, T. H. (1997). *Relationships between Channel Characteristics, Woody Debris, and Fish Habitat in Northwestern Washington Streams*. 13.

Bladon, K. D., Segura, C., Cook, N. A., Bywater-Reyes, S., & Reiter, M. (2018). A multicatchment analysis of headwater and downstream temperature effects from contemporary forest harvesting. *Hydrological Processes*, *32*(2), 293–304. https://doi.org/10.1002/hyp.11415

Bracken, L. J., Turnbull, L., Wainwright, J., & Bogaart, P. (2015). Sediment connectivity: A framework for understanding sediment transfer at multiple scales. *Earth Surface Processes and Landforms*, *40*(2), 177–188. https://doi.org/10.1002/esp.3635

Bradford, M. J., Higgins, P. S., Korman, J., & Sneep, J. (2011). Test of an environmental flow release in a British Columbia river: Does more water mean more fish? *Freshwater Biology*, *56*(10), 2119–2134. https://doi.org/10.1111/j.1365-2427.2011.02633.x

Bradford, M. J., & Irvine, J. R. (2000). *Land use, fishing, climate change, and the decline of Thompson River, British Columbia, coho salmon*. *57*, 4.

Braun, D. C., Patterson, D. A., & Reynolds, J. D. (2013). Maternal and environmental influences on egg size and juvenile life-history traits in Pacific salmon. *Ecology and Evolution*, *3*(6), 1727–1740. https://doi.org/10.1002/ece3.555

Buffington, J. M., Montgomery, D. R., & Greenberg, H. M. (2004). Basin-scale availability of salmonid spawning gravel as influenced by channel type and hydraulic roughness in mountain catchments. *Canadian Journal of Fisheries and Aquatic Sciences*, *61*(11), 2085–2096. https://doi.org/10.1139/f04-141

Burnett, K. M., Reeves, G. H., Clarke, S. E., & Christiansen, K. R. (2006). *Comparing Riparian and Catchment Influences on Stream Habitat in a Forested, Montane Landscape*. 18.

Burnett, N. J., Hinch, S. G., Braun, D. C., Casselman, M. T., Middleton, C. T., Wilson, S. M., & Cooke, S. J. (2014). Burst swimming in areas of high flow: Delayed consequences of anaerobiosis in wild adult sockeye salmon. *Physiological and Biochemical Zoology*, *87*(5), 587–598.

Burnham, K. P., & Anderson, D. R. (2004). Multimodel Inference: Understanding AIC and BIC in Model Selection. *Sociological Methods & Research*, *33*(2), 261–304. https://doi.org/10.1177/0049124104268644

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

Cade, B. S. (2015). Model averaging and muddled multimodel inferences. *Ecology*, *96*(9), 2370–2382. https://doi.org/10.1890/14-1639.1

Caissie, D. (2006). The thermal regime of rivers: A review. *Freshwater Biology*, *51*(8), 1389–1406. https://doi.org/10.1111/j.1365-2427.2006.01597.x

Carlier, C., Wirth, S. B., Cochand, F., Hunkeler, D., & Brunner, P. (2018). Geology controls streamflow dynamics. *Journal of Hydrology*, *566*, 756–769. https://doi.org/10.1016/j.jhydrol.2018.08.069

Chartrand, S. M., Jellinek, A. M., Hassan, M. A., & Ferrer-Boix, C. (2018). Morphodynamics of a Width-Variable Gravel Bed Stream: New Insights on Pool-Riffle Formation From Physical Experiments. *Journal of Geophysical Research: Earth Surface*, *123*(11), 2735–2766. https://doi.org/10.1029/2017JF004533

Chen, W., & Wei, X. (2008). Assessing the relations between aquatic habitat indicators and forest harvesting at watershed scale in the interior of British Columbia. *Forest Ecology and Management*, *256*(1), 152–160. https://doi.org/10.1016/j.foreco.2008.04.019

Cooke, S. J., Raby, G. D., Bett, N. N., Teffer, A. K., Burnett, N. J., Jeffries, K. M., Eliason, E. J., Martins, E. G., Miller, K. M., Patterson, D. A., Nguyen, V. M., Young, N., Farrell, A. P., & Hinch, S. G. (2020). On conducting management-relevant mechanistic science for upriver migrating adult Pacific salmon. In S. J.

Cooke, G. D. Raby, N. N. Bett, A. K. Teffer, N. J. Burnett, K. M. Jeffries, E. J. Eliason, E. G. Martins, K. M. Miller, D. A. Patterson, V. M. Nguyen, N. Young, A. P. Farrell, & S. G. Hinch, *Conservation Physiology* (pp. 35–56). Oxford University Press. https://doi.org/10.1093/oso/9780198843610.003.0003

COSEWIC. (2017, August 9). *Coho salmon (Oncorhynchus kisutch) interior Fraser population: COSEWIC assessment and status report 2016* [Assessments;research]. https://www.canada.ca/en/environment-climate-change/services/species-risk-public-registry/cosewic-assessments-status-reports/coho-salmon-interior-fraser-2016.html

Cribari-Neto, F., & Zeileis, A. (2010). Beta Regression in *R*. *Journal of Statistical Software*, *34*(2). https://doi.org/10.18637/jss.v034.i02

Crossin, G. T., Hinch, S. G., Cooke, S. J., Welch, D. W., Patterson, D. A., Jones, S. R. M., Lotto, A. G., Leggatt, R. A., Mathes, M. T., Shrimpton, J. M., Van Der Kraak, G., & Farrell, A. P. (2008). Exposure to high temperature influences the behaviour, physiology, and survival of sockeye salmon during spawning migration. *Canadian Journal of Zoology*, *86*(2), 127–140. https://doi.org/10.1139/Z07-122

Douma, J. C., & Weedon, J. T. (2019). Analysing continuous proportions in ecology and evolution: A practical introduction to beta and Dirichlet regression. *Methods in Ecology and Evolution*, *10*(9), 1412–1430. https://doi.org/10.1111/2041-210X.13234

Ebersole, J. L., Wigington Jr., P. J., Baker, J. P., Cairns, M. A., Church, M. R., Hansen, B. P., Miller, B. A., LaVigne, H. R., Compton, J. E., & Leibowitz, S. G. (2006). Juvenile Coho Salmon Growth and Survival across Stream Network Seasonal Habitats. *Transactions of the American Fisheries Society*, *135*(6), 1681–1697. https://doi.org/10.1577/T05-144.1

EPA. (2014). *Best Practices for Continuous Monitoring of Temperature and Flow in Wadeable Streams* (p. 129).

Fausch, K. D., & Northcote, T. G. (1992). Large Woody Debris and Salmonid Habitat in a Small Coastal British Columbia Stream. *Canadian Journal of Fisheries and Aquatic Sciences*, *49*(4), 682–693. https://doi.org/10.1139/f92-077

Faustini, J. M., Kaufmann, P. R., & Herlihy, A. T. (2009). Downstream variation in bankfull width of wadeable streams across the conterminous United States. *Geomorphology*, *108*(3), 292–311. https://doi.org/10.1016/j.geomorph.2009.02.005

Ferrari, S., & Cribari-Neto, F. (2004). Beta Regression for Modelling Rates and Proportions. *Journal of Applied Statistics*, *31*(7), 799–815. https://doi.org/10.1080/0266476042000214501

Finn, R. J. R., Chalifour, L., Gergel, S. E., Hinch, S. G., Scott, D. C., & Martin, T. G. (2021). Quantifying lost and inaccessible habitat for Pacific salmon in Canada's Lower Fraser River. *Ecosphere*, *12*(7), e03646. https://doi.org/10.1002/ecs2.3646

Forest Practices Board. (2018). *Special Report: Conserving Fish Habitats under the Forest and Range Practices Act—Part 1: A Review of the BC Government Approach* (p. 41). Forest Practices Board. https://www.bcfpb.ca/wp-content/uploads/2018/07/SR56-Fish-Habitat-Conservation.pdf

Forest Practices Board. (2019). *Tactical Forest Planning: The Missing Link Between Strategic Planning and Operational Planning in BC* (p. 17). Forest Practices Board. https://www.bcfpb.ca/wp-content/uploads/2019/07/SR58-Tactical-Forest-Planning.pdf

Forest Tenures. (2021). *Harvested Areas of BC (Consolidated Cutblocks)—Datasets—Data Catalogue*. https://catalogue.data.gov.bc.ca/dataset/harvested-areas-of-bc-consolidated-cutblocks-

Galipaud, M., Gillingham, M. A. F., & Dechaume-Moncharmont, F.-X. (2017). A farewell to the sum of Akaike weights: The benefits of alternative metrics for variable importance estimations in model selection. *Methods in Ecology and Evolution*, *8*(12), 1668–1678. https://doi.org/10.1111/2041-210X.12835

GeoBC. (2019). *Freshwater Atlas Watersheds—Datasets—Data Catalogue*. https://catalogue.data.gov.bc.ca/dataset/freshwater-atlas-watersheds

GeoBC. (2021). *Digital Road Atlas (DRA)—Master Partially-Attributed Roads—Datasets—Data Catalogue*. https://catalogue.data.gov.bc.ca/dataset/digital-road-atlas-dra-master-partially-attributed-roads

GeoBC. (2022). *BEC Map*. https://catalogue.data.gov.bc.ca/dataset/bec-map

Government of British Columbia. (2017). *Timber Supply Review Backgrounder* (p. 5). Government of British Columbia. https://www2.gov.bc.ca/assets/gov/farming-natural-resources-and-industry/forestry/stewardship/forest-analysis-inventory/tsr-annual-allowable-cut/tsr_backgrounder2.pdf

Government of British Columbia. (2021). *Forest Planning and Practices Regulation*. https://www.bclaws.gov.bc.ca/civix/document/id/lc/statreg/14_2004#section47

Gronsdahl, S., Moore, R. D., Rosenfeld, J., McCleary, R., & Winkler, R. (2019). Effects of forestry on summertime low flows and physical fish habitat in snowmelt-dominant headwater catchments of the Pacific Northwest. *Hydrological Processes*, *33*(25), 3152–3168. https://doi.org/10.1002/hyp.13580

Hartman, G. F., Scrivener, J. C., & Miles, M. J. (1996). *Impacts of logging in Carnation Creek, a high-energy coastal stream in British Columbia, and their implication for restoring fish habitat*. *53*, 15.

Heitke, J. D., Archer, E. J., Dugaw, D. D., Bouwes, B. A., Archer, E. A., Henderson, R. C., & Kershner, J. L. (2008). *Effectiveness monitoring for streams and riparian areas: Sampling protocol for stream channel attributes*. PACFISH/INFISH

Biological Opinion Effectiveness Monitoring Program (PIBO-EM) Staff - Multi-federal Agency Monitoring Program; Logan, UT. https://www.fs.fed.us/biology/resources/pubs/feu/pibo/pibo_2008_stream_sampling_protocol.pdf

Herunter, H. E., Macdonald, J. S., & MacIsaac, E. A. (2004). *Effectiveness of Variable-Retention Riparian Buffers for Maintaining Thermal Regimes, Water Chemistry, hemistry and Benthic Invertebrate Communities of Small Headwater Streams in Central British Columbia*.

Hilton, S., & Lisle, T. E. (1993). *Measuring the fraction of pool volume filled with fine sediment* (PSW-RN-414; p. PSW-RN-414). U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station. https://doi.org/10.2737/PSW-RN-414

Holtby, L. B. (1988). Effects of Logging on Stream Temperatures in Carnation Creek British Columbia, and Associated Impacts on the Coho Salmon (*Oncorhynchus kisutch*). *Canadian Journal of Fisheries and Aquatic Sciences*, *45*(3), 502–515. https://doi.org/10.1139/f88-060

Hupp, C. R., & Osterkamp, W. R. (1996). Riparian vegetation and fluvial geomorphic processes. *Geomorphology*, *14*(4), 277–295. https://doi.org/10.1016/0169-555X(95)00042-4

Interior Fraser Coho Recovery Team. (2006). *Conservation strategy for coho salmon (Oncorhynchus kisutch), interior Fraser River populations*. Fisheries and Oceans Canada. https://central.bac-lac.gc.ca/.item?id=Fs23-517-2007E&op=pdf&app=Library

Isaak, D. J., & Hubert, W. A. (2001). A Hypothesis About Factors That Affect Maximum Summer Stream Temperatures Across Montane Landscapes1. *JAWRA Journal* *of the American Water Resources Association*, *37*(2), 351–366. https://doi.org/10.1111/j.1752-1688.2001.tb00974.x

Isaak, D. J., Wollrab, S., Horan, D., & Chandler, G. (2012). Climate change effects on stream and river temperatures across the northwest U.S. from 1980–2009 and implications for salmonid fishes. *Climatic Change*, *113*(2), 499–524. https://doi.org/10.1007/s10584-011-0326-z

Islam, S. U., Hay, R. W., Déry, S. J., & Booth, B. P. (2019). Modelling the impacts of climate change on riverine thermal regimes in western Canada's largest Pacific watershed. *Scientific Reports*, *9*(1), 11398. https://doi.org/10.1038/s41598-019-47804-2

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

Jensen, D. W., Steel, E. A., Fullerton, A. H., & Pess, G. R. (2009). Impact of Fine Sediment on Egg-To-Fry Survival of Pacific Salmon: A Meta-Analysis of Published Studies. *Reviews in Fisheries Science*, *17*(3), 348–359. https://doi.org/10.1080/10641260902716954

Kaushal, S. S., Likens, G. E., Jaworski, N. A., Pace, M. L., Sides, A. M., Seekell, D., Belt, K. T., Secor, D. H., & Wingate, R. L. (2010). Rising stream and river temperatures in the United States. *Frontiers in Ecology and the Environment*, *8*(9), 461–466. https://doi.org/10.1890/090037

Lisle, T. E., & Hilton, S. (1992). THE VOLUME OF FINE SEDIMENT IN POOLS: AN INDEX OF SEDIMENT SUPPLY IN GRAVEL-BED STREAMS. *Journal of the American Water Resources Association*, *28*(2), 371–383. https://doi.org/10.1111/j.1752-1688.1992.tb04003.x

Macdonald, J. S., Beaudry, P. G., MacIsaac, E. A., & Herunter, H. E. (2003a). *The effects of forest harvesting and best management practices on streamflow and suspended sediment concentrations during snowmelt in headwater streams in sub-boreal forests of British Columbia, Canada*. *33*, 11.

Macdonald, J. S., MacIsaac, E. A., & Herunter, H. E. (2003b). *The effect of variable-retention riparian buffer zones on water temperatures in small headwater streams in sub-boreal forest ecosystems of British Columbia*. *33*, 12.

MacDonald, L. H., & Coe, D. (2007). Influence of headwater streams on downstream reaches in forested areas. *Forest Science*, *53*(2), 148–168.

Mantua, N., Tohver, I., & Hamlet, A. (2010). Climate change impacts on streamflow extremes and summertime stream temperature and their possible consequences for freshwater salmon habitat in Washington State. *Climatic Change*, *102*(1–2), 187–223. https://doi.org/10.1007/s10584-010-9845-2

Martins, E. G., Hinch, S. G., Cooke, S. J., & Patterson, D. A. (2012). Climate effects on growth, phenology, and survival of sockeye salmon (Oncorhynchus nerka): A synthesis of the current state of knowledge and future research directions. *Reviews in Fish Biology and Fisheries*, *22*(4), 887–914. https://doi.org/10.1007/s11160-012-9271-9

Martins, E. G., Hinch, S. G., Patterson, D. A., Hague, M. J., Cooke, S. J., Miller, K. M., Robichaud, D., English, K. K., & Farrell, A. P. (2012). High river temperature reduces survival of sockeye salmon (*Oncorhynchus nerka*) approaching spawning grounds and exacerbates female mortality. *Canadian Journal of Fisheries and Aquatic Sciences*, *69*(2), 330–342. https://doi.org/10.1139/f2011-154

Mellina, E., & Hinch, S. G. (2009). Influences of riparian logging and in-stream large wood removal on pool habitat and salmonid density and biomass: A meta-analysis. *Canadian Journal of Forest Research*, *39*(7), 1280–1301. https://doi.org/10.1139/X09-037

Montgomery, D. R., & Buffington, J. M. (1997). Channel-reach morphology in mountain drainage basins. *Geological Society of America Bulletin*, *109*(5), 596–611. https://doi.org/10.1130/0016-7606(1997)109<0596:CRMIMD>2.3.CO;2

Montgomery, D. R., Buffington, J. M., Peterson, N. P., Schuett-Hames, D., & Quinn, T. P. (1996). *Stream-bed scour, egg burial depths, and the influence of salmonid spawning on bed surface mobility and embryo survival*. *53*, 10.

Moore, K., Jones, K., Dambacher, J., & Stein, C. (2014). *Aquatic Inventories Project Methods for Stream Habitat Surveys*. Conservation and Recovery Program Oregon Department of Fish and Wildlife. https://odfw.forestry.oregonstate.edu/freshwater/inventory/pdffiles/hmethd14.pdf

Moore, R. D. (2006). Stream Temperature Patterns in British Columbia, Canada, Based on Routine Spot Measurements. *Canadian Water Resources Journal*, *31*(1), 41–56. https://doi.org/10.4296/cwrj3101041

Moore, R. D., Spittlehouse, D. L., & Story, A. (2005). RIPARIAN MICROCLIMATE AND STREAM TEMPERATURE RESPONSE TO FOREST HARVESTING: A REVIEW. *JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION*, 22.

Munsch, S. H., Greene, C. M., Mantua, N. J., & Satterthwaite, W. H. (2022). One hundred-seventy years of stressors erode salmon fishery climate resilience in California’s warming landscape. *Global Change Biology*, *28*(7), 2183–2201. https://doi.org/10.1111/gcb.16029

Murphy, M. L., Heifetz, J., Johnson, S. W., Koski, K. V., & Thedinga, J. F. (1986). Effects of Clear-cut Logging with and without Buffer Strips on juvenile Salmonids in

Alaskan Streams. *Canadian Journal of Fisheries and Aquatic Sciences*, *43*(8), 1521–1533. https://doi.org/10.1139/f86-190

Poff, N. L., Bledsoe, B. P., & Cuhaciyan, C. O. (2006). Hydrologic variation with land use across the contiguous United States: Geomorphic and ecological consequences for stream ecosystems. *Geomorphology*, *79*(3), 264–285. https://doi.org/10.1016/j.geomorph.2006.06.032

Pollock, M. M., Beechie, T. J., Liermann, M., & Bigley, R. E. (2009). Stream Temperature Relationships to Forest Harvest in Western Washington. *JAWRA Journal of the American Water Resources Association*, *45*(1), 141–156. https://doi.org/10.1111/j.1752-1688.2008.00266.x

Price, M. H. H., Moore, J. W., Connors, B. M., Wilson, K. L., & Reynolds, J. D. (2021). Portfolio simplification arising from a century of change in salmon population diversity and artificial production. *Journal of Applied Ecology*, *58*(7), 1477–1486. https://doi.org/10.1111/1365-2664.13835

Quinn, T. P. (2018). *The behavior and ecology of Pacific salmon and trout*. University of Washington press.

R Core Team. (2021). A language and environment for statistical computing. In *R Foundation for Statistical Computing* (Vol. 10, Issue 1).

Rand, P. S., Hinch, S. G., Morrison, J., Foreman, M. G. G., MacNutt, M. J., Macdonald, J. S., Healey, M. C., Farrell, A. P., & Higgs, D. A. (2006). Effects of River Discharge, Temperature, and Future Climates on Energetics and Mortality of Adult Migrating Fraser River Sockeye Salmon. *Transactions of the American Fisheries Society*, *135*(3), 655–667. https://doi.org/10.1577/T05-023.1

Reid, A. J., Carlson, A. K., Creed, I. F., Eliason, E. J., Gell, P. A., Johnson, P. T. J., Kidd, K. A., MacCormack, T. J., Olden, J. D., Ormerod, S. J., Smol, J. P., Taylor, W. W., Tockner, K., Vermaire, J. C., Dudgeon, D., & Cooke, S. J. (2019). Emerging threats and persistent conservation challenges for freshwater biodiversity. *Biological Reviews*, *94*(3), 849–873. https://doi.org/10.1111/brv.12480

Reid, D. A., & Hassan, M. A. (2020). Response of In-Stream Wood to Riparian Timber Harvesting: Field Observations and Long-Term Projections. *Water Resources Research*, *56*(8). https://doi.org/10.1029/2020WR027077

Reid, D. A., Hassan, M. A., Bird, S., & Hogan, D. (2019). Spatial and temporal patterns of sediment storage over 45 years in Carnation Creek, BC, a previously glaciated mountain catchment. *Earth Surface Processes and Landforms*, *44*(8), 1584–1601. https://doi.org/10.1002/esp.4595

Reid, D. A., Pike, R., Bird, S., Tschaplinski, P., & Wilford, D. (2020). Implications of legacy watershed disturbances for channel structure and salmon habitat availability under different low-flow levels: An analysis of 45 years of discharge–habitat relationships at Carnation Creek, British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences*, *77*(11), 1780–1793. https://doi.org/10.1139/cjfas-2020-0120

Richards, C., Johnson, L. B., & Host, G. E. (1996). *Landscape-scale influences on stream habitats and biota*. *53*, 17.

Richter, A., & Kolmes, S. A. (2005). Maximum Temperature Limits for Chinook, Coho, and Chum Salmon, and Steelhead Trout in the Pacific Northwest. *Reviews in Fisheries Science*, *13*(1), 23–49. https://doi.org/10.1080/10641260590885861

Roni, P., & Quinn, T. P. (2001). Density and size of juvenile salmonids in response to placement of large woody debris in western Oregon and Washington streams. *Canadian Journal of Fisheries and Aquatic Sciences*, *58*(2), 282–292. https://doi.org/10.1139/f00-246

Rosenfeld, J. S., Leiter, T., Lindner, G., & Rothman, L. (2005). *Food abundance and fish density alters habitat selection, growth, and habitat suitability curves for juvenile coho salmon (Oncorhynchus kisutch)*. *62*, 11.

Roy, S., Das, V. K., & Debnath, K. (2019). Characteristics of intermittent turbulent structures for river bank undercut depth increment. *CATENA*, *172*, 356–368. https://doi.org/10.1016/j.catena.2018.09.008

Schindler, D. E., Augerot, X., Fleishman, E., Mantua, N. J., Riddell, B., Ruckelshaus, M., Seeb, J., & Webster, M. (2008). Climate Change, Ecosystem Impacts, and Management for Pacific Salmon. *Fisheries*, *33*(10), 502–506. https://doi.org/10.1577/1548-8446-33.10.502

Seelbach, P. W. (2006). *Initial Classification of River Valley Segments across Michigan's Lower Peninsula*. 25.

Smith, R., & Redding, T. (2012). Cumulative Effects Assessment: Runoff Generation in Snowmelt- dominated Montane and Boreal Plain Catchments. *Aquatic Ecosystem Health and Management*, *15*, 24–34.

Story, A., Moore, R. D., & Macdonald, J. S. (2003). *Stream temperatures in two shaded reaches below cutblocks and logging roads: Downstream cooling linked to subsurface hydrology*. *33*, 14.

Subehi, L., Fukushima, T., Onda, Y., Mizugaki, S., Gomi, T., Terajima, T., Kosugi, K., Hiramatsu, S., Kitahara, H., Kuraji, K., & Ozaki, N. (2009). Influences of forested watershed conditions on fluctuations in stream water temperature with special reference to watershed area and forest type. *Limnology*, *10*(1), 33–45. https://doi.org/10.1007/s10201-008-0258-0

Sweeney, B. W., & Newbold, J. D. (2014). Streamside Forest Buffer Width Needed to Protect Stream Water Quality, Habitat, and Organisms: A Literature Review.

JAWRA Journal of the American Water Resources Association, *50*(3), 560–584. https://doi.org/10.1111/jawr.12203

Tschaplinski, P. J., & Pike, R. G. (2017). Carnation Creek watershed experiment—Long-term responses of coho salmon populations to historic forest practices. *Ecohydrology*, *10*(2), e1812. https://doi.org/10.1002/eco.1812

Wang, L., Seelbach, P., & Hughes, R. (2006). Landscape Influences on Stream Habitats And Biological Assemblages. *American Fisheries Society Symposium*, *48*.

Wilson, K. L., Bailey, C. J., Davies, T. D., & Moore, J. W. (2022). Marine and freshwater regime changes impact a community of migratory Pacific salmonids in decline. *Global Change Biology*, *28*(1), 72–85. https://doi.org/10.1111/gcb.15895

WMO. (2010a). *Manual on Stream Gauging Volume II - Computing of Discharge*. World Meteorological Organization.

WMO, N. (2010b). Manual on Stream Gauging Volume I - Fieldwork. *WMO No. 1044*.

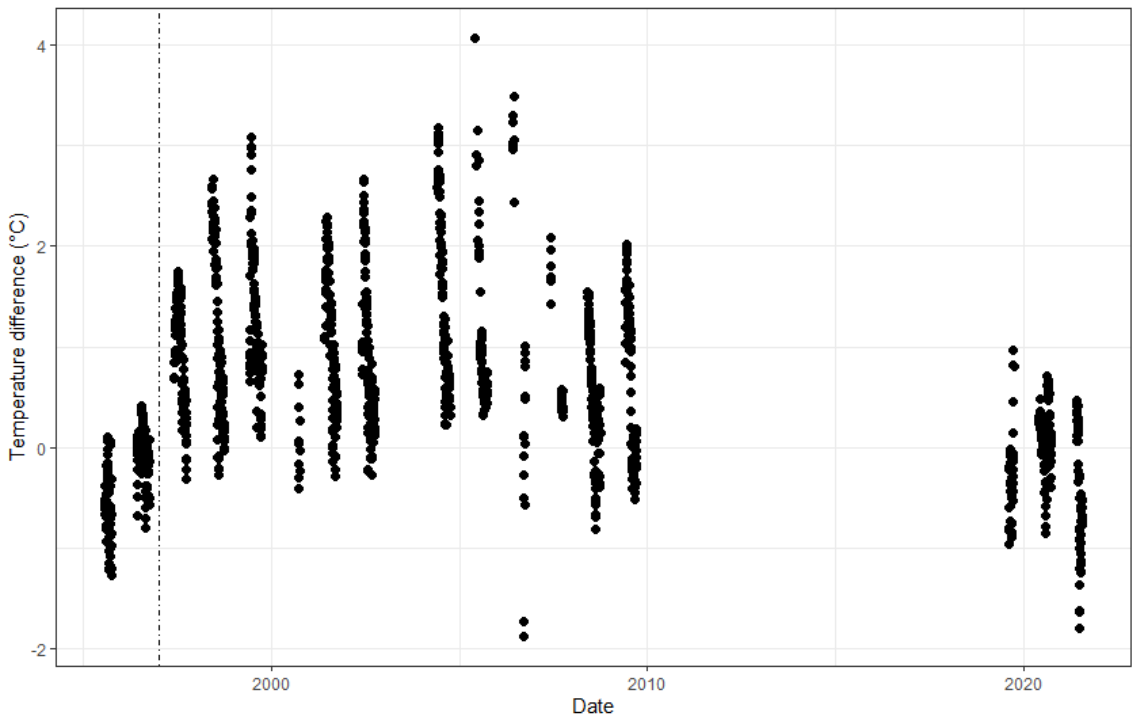
Wohl, E. E., Vincent, K. R., & Merritts, D. J. (1993). Pool and riffle characteristics in relation to channel gradient. *Geomorphology*, *6*(2), 99–110. https://doi.org/10.1016/0169-555X(93)90041-Y

Zuur, A. F., Ieno, E. N., & Elphick, C. S. (2010). A protocol for data exploration to avoid common statistical problems. *Methods in Ecology and Evolution*, *1*(1), 3–14. https://doi.org/10.1111/j.2041-210X.2009.00001.x

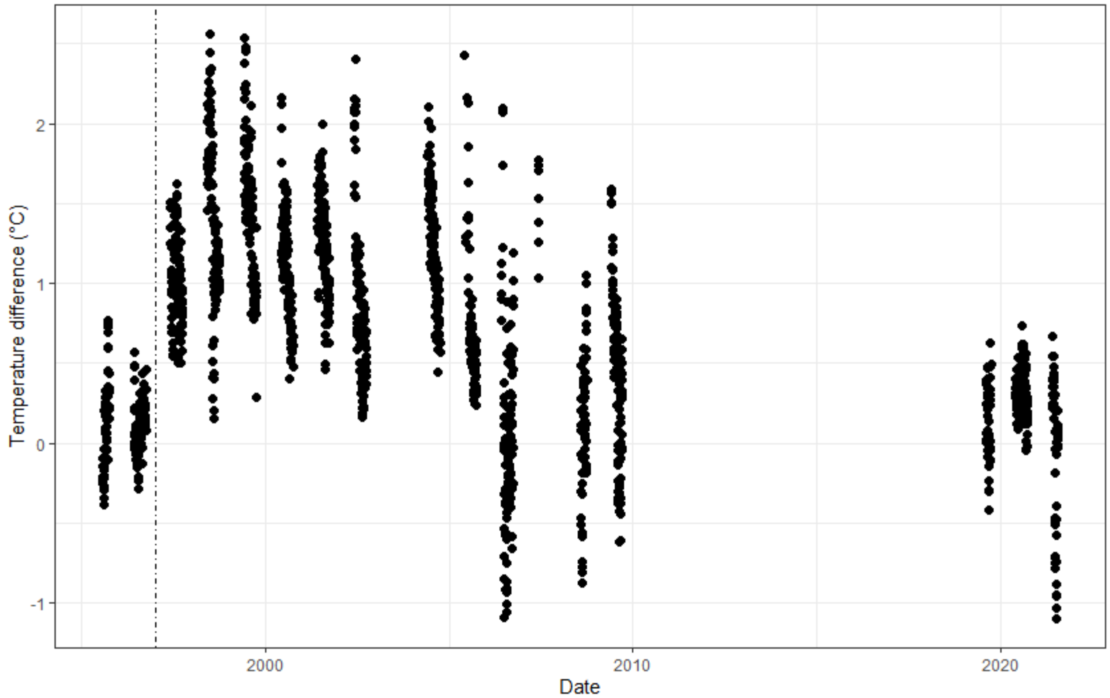
Zuur, A. F., Ieno, E. N., & Smith, G. M. (2007). *Analysing ecological data*. Springer.

Appendix

Baptiste Creek Temperature Study



A1 The difference in summer daily mean temperature at a harvested treatment stream compared to an unharvested reference stream. Negative temperatures signify the mean temperature is lower at the treatment stream, positive temperatures indicate the mean temperature is higher at the treatment stream. The dashed vertical line represents the period of forest harvest, data to the left of this line is pre impact, data to the right is post-impact. Harvesting took place in early 1997.



A2 The difference in summer daily mean temperature between the downstream harvested treatment station and the upstream unharvested control station. Negative temperatures signify the mean temperature is lower at the downstream impact station, positive temperatures indicate the mean temperature is higher at the downstream impact station. The dashed vertical line represents the period of forest harvest, data to the left of this line is pre impact, data to the right is post-impact. Harvesting took place in early 1997.