

**Interconnected effects of water temperature and  
habitat on relative abundance of juvenile salmonids  
during the summer months in coastal urban streams**

**by  
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## Abstract

Juvenile Pacific salmon in urban streams face the double threat of habitat degradation and the effects of climate change. Increased water temperature and habitat alterations may compound to limit salmon distribution in fresh water. My thesis investigates which biotic and abiotic variables effect the abundance of anadromous and non-anadromous salmonid species in a dozen urbanized Metro Vancouver streams. I used Partial Least Squared Regression (PLSR) to elucidate which of the 31 habitat metrics measured were most strongly associated with relative abundance of coho (*Oncorhynchus kisutch*) and trout (*O. mykiss*, *clarkii*). Water temperature, even within an optimal range for growth, had a negative impact on relative abundance of coho and trout. Other key habitat metrics included a negative impact of fine sediments on coho, smaller channel sizes favored trout, and all juvenile salmonid abundance responded positively to a higher proportion of cobble. Non-anadromous trout may be disproportionately impacted by higher water temperatures and changes in substrate composition due to their resident life history tactic that limit their distributions options to their natal freshwater watershed. There was also a positive relationship between the percent impervious surface area of a watershed and higher values of temperature, canopy openness, and percent fines indicating that urbanization is likely a driver behind the lower relative abundance of these species in certain stream reaches. A better understanding of the interconnected mechanisms driving salmonid relative abundance in high-order urban streams has implications for prioritizing conservation strategies towards improving key habitat metrics relevant to urban ecosystems and for predicting salmonid species responses to future climate change.

**Keywords:** Juvenile salmonids; stream temperatures; urban streams; freshwater habitat; metro Vancouver; climate change

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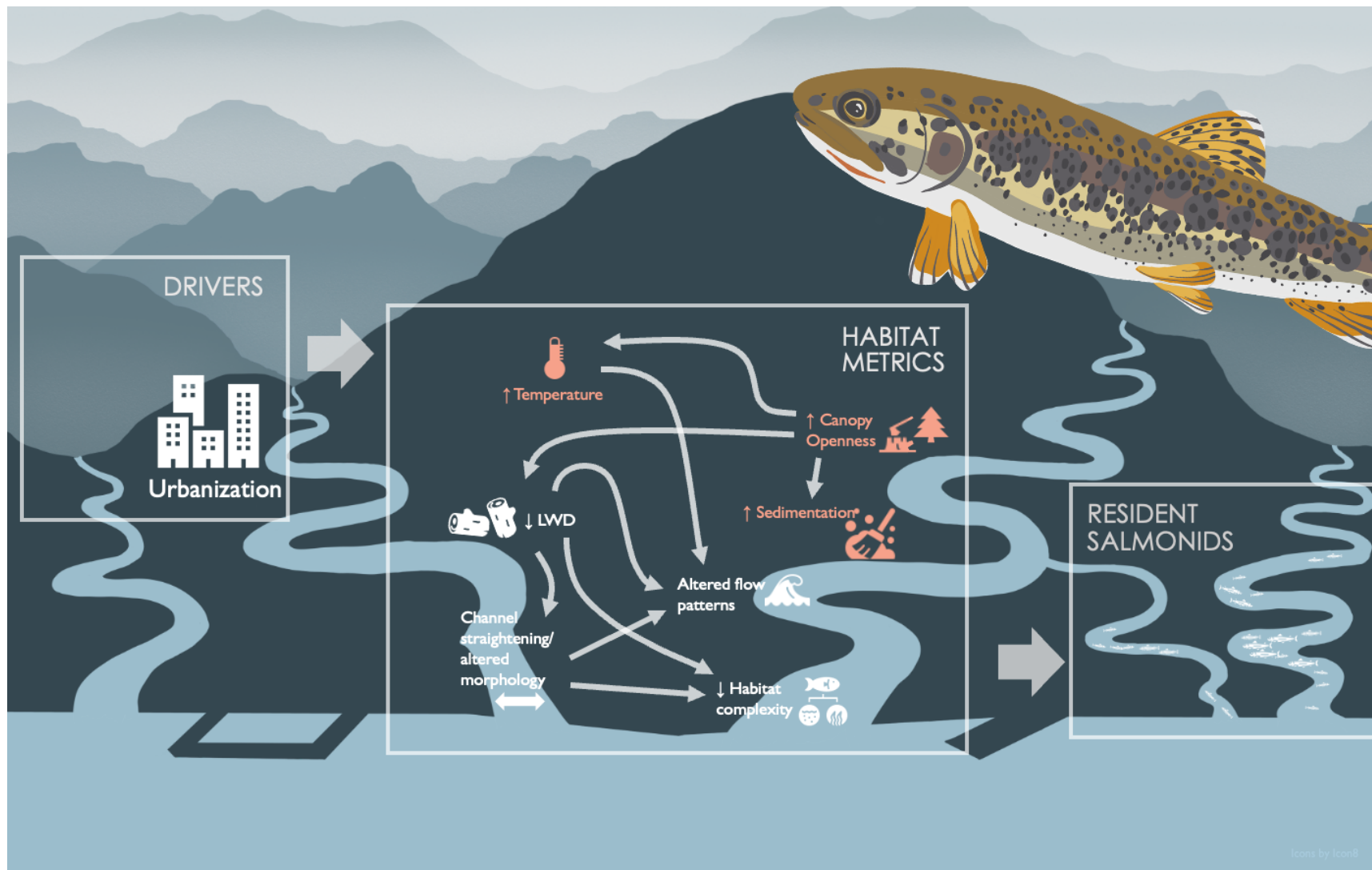
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# Introduction

Freshwater ecosystems support approximately 40% of the world's fish diversity for at least part of their lives (Balian et al., 2007). Unfortunately, these ecosystems are also considered to be the most threatened (Dudgeon et al., 2006; Hughes, 2021). Climate change and urbanization are two important pressures on some freshwater ecosystems. Watershed urbanization, even at low levels, has a variety of impacts on streams systems, including altering flow patterns, channel morphology, water quality, and biotic communities (Brown & American Fisheries Society Water Quality Section, 2005; Paul & Meyer, 2001; Settele et al., 2014; Spanjer et al., 2018; Walsh et al., 2005; Wang et al., 2001). Collectively, these impacts have been termed the "urban stream syndrome" (USS) and can also include warming water temperatures. Climate change is also increasing temperatures and altering flow regimes in streams. Changes in temperature can impact many aspects of fish biology, including competitive ability, survival, growth, distribution, and abundance (Spanjer et al., 2018; Torgersen et al., 2006). Various habitat features may buffer or exacerbate rising temperatures such as the presence of absence of shade from canopy cover (Schiemer et al., 1995). The relative importance of other habitat features in urban ecosystems can be challenging to predict, but likely varies across watersheds depending on species-specific habitat requirements (Bramblett et al., 2002; Kemp et al., 2011; Young, 2001, 2004). Understanding which temperature and habitat metrics interact to influence fish distribution in urban streams will help inform conservation efforts in terms of prioritizing habitat protection and restoration activities in these threatened freshwater ecosystems.

Salmonid species are at particularly high risk of experiencing negative effects from the projected thermal changes associated with urbanization and climate change because of their substantially lower thermal tolerance compared to many other freshwater fish species (Eaton et al., 1995; Ebersole et al., 2001; Kaylor et al., 2019; Lusardi et al., 2020; McCullough, 1999; Nielsen et al., 1994; Sloat & Osterback, 2013; Welsh et al., 2001). Generally, optimal temperatures for most juvenile salmonid species range between 13-19°C (Bear et al., 2007; Dockray et al., 1996; Kaylor et al., 2019; Lusardi et al., 2020; McCullough, 1999; Nielsen et al., 1994; Sullivan et al., 2000; Welsh et al., 2001; Wurtsbaugh & Davis, 1977). This range is influenced by various factors, including acclimation temperature, size, age, availability of cold-water refuges,

movement ability within a system, and additional stressors (Kahler et al., 2001; McCullough, 1999). Regardless, summer stream temperatures are predicted to meet or exceed salmonid thermal tolerance this decade in some inland watersheds draining into the Northeast Pacific, and coastal southwestern British Columbia streams are likely soon to follow with lower latitude systems being disproportionately affected (Mantua et al., 2010; Weller et al., 2023). As temperatures continue to increase, movement by salmonids into cooler, high-order streams, according to the Strahler method, or pockets of thermal refuge may become necessary for survival (Isaak et al., 2015). An understanding of these high-order stream habitats will become more crucial in the future especially in urban systems where the rate of increase in water temperature is expected to be higher because of habitat alteration (Settele et al., 2014; Spanjer et al., 2018).

Urbanization is an increasing stressor on freshwater ecosystems that could exacerbate pressure on salmonids during this period of climate warming. Urbanized watersheds experience decreased bank stability, reduced habitat complexity, lower amounts of large woody debris, and tend to have more water control structures (e.g. culverts) or other migration barriers that can decrease habitat connectivity (Hale et al., 2016; Walsh et al., 2005; Warkentin et al., 2019). In urban systems, forests and wetlands have been replaced by impervious surfaces, altering the magnitude of precipitation input and surface runoff into a watershed, and decreasing riparian cover and vegetation (Finkenbine et al., 2000; Wang et al., 2000, 2001; Warkentin et al., 2019). Impervious cover blocks ground water recharge resulting in lower base flows and altering water temperatures (Finkenbine et al., 2000). Understanding which specific habitat factors influence salmonid abundance in urban ecosystems is particularly relevant for restoration and management actions in urban ecosystems given that these systems are important touchpoints between human society and salmonids. Indeed, efforts to restore salmon habitat have had mixed results elucidating the need for continued research in important, but potentially misunderstood urban freshwater ecosystems (Cederholm et al., 1997; House, 1996; Roni & Quinn, 2001).

Here I examined the relative roles of water temperatures and other habitat attributes influencing salmonid abundance across a gradient of urbanized freshwater streams. Specifically, I studied salmonids in Metro Vancouver, British Columbia. I studied three salmonid species, coho salmon (*Oncorhynchus kisutch*), cutthroat trout (*O. clarkii*), and rainbow trout (*O. mykiss*). I asked the following research questions:

1. What are the effects of temperature and other habitat metrics on the relative abundance of juvenile salmonids in these urban streams?  
And;
2. How does upstream urbanization relate to important habitat metrics?

I used Partial Least Squared Regression (PLSR) to assess the effects of temperature and habitat metrics on relative abundance, and linear models to link seasonal temperature, canopy openness, and percent fines within a system to percent impervious surface; a surrogate for intensity of urbanization (Nelson & Palmer, 2007; Spanjer et al., 2018; Wang et al., 2000, 2001). A detailed rationale for the inclusion of each of the 31 temperature and habitat metrics used in the PLSR, as well as expected predictions are provided in Table 1. A better understanding of the mechanisms behind what factors are driving salmonid relative abundance in high-order urban streams has implications for predicting salmonid species responses to future climate change, and therefore inform conservation tactics.

**Table 1. Hypothesis table outlining the justification of inclusion for each predictor variable used in the Partial Least Squared Regression (PLSR) and the predicted response from the response variables where available in the literature.**

Metric	Justification For Inclusion	Predicted Response of Coho Salmon	Predicted Response of Cutthroat Trout	Predicted Response of Grouped Trout**
Seasonal Mean Weekly Minimum Temperature (MWMinT) (°C), Week Prior MWMinT (°C)	This metric detects chronic and acute low temperatures seen within a system without being overly influenced by a single low temperature. Systems of interest could be dropping below optimum temperatures for some species.	Juvenile coho rarely found where daily max exceeded 21°C (McCullough, 1999).	Juvenile Coastal cutthroat trout found in streams with a mean daily mean temperature of 9.1 °C to 14.8 °C (Kaylor et al., 2019).  The optimal growth temperature for westslope cutthroat trout was found to be 13.6 °C and lethal limit 19.6 °C (Bear et al., 2007), therefore we could expect Coastal cutthroat trout to have an optimal growth temperature similar to this.  ***	***  Optimal growth temperature for juvenile rainbow trout ( <i>O. mykiss</i> ) to be 13.1 °C and lethal limit 24.3 °C (Bear et al., 2007).
Seasonal Mean Weekly Maximum Temperature (MWMT) (°C), Week Prior MWMT (°C)	MWMT is a measure of both chronic and acute effects of temperature (Carter, 2005). It describes the maximum temperature seen in a system over a week but isn't overly influenced by one single daily maximum temperature. The highest probability of seeing the effects of high temperatures influencing relative abundance would be through MWMT due to the nature of the thermal regimes seen in the systems of interest.	Juvenile coho presence in sites less than 16.3°C MWMT, absence in sites MWMT of 18.0°C (Welsh et al., 2001).  Juvenile coho optimal growth between an MWMT of 13°C and 16.5°C (Sullivan et al., 2000).  Absolute growth rate of coho salmon to peak at an MWMT of 21.1 °C (Lusardi et al., 2020).  Juvenile coho rarely found where daily max exceeded 21°C (McCullough, 1999).		Expect rainbow trout growth to improve by increasing temperatures up until 16.5 °C and that no rainbow trout to be found in the streams of interest when the daily maximum temperature was more than 24 °C (McCullough, 1999).  Growth rate would be expected to be just barely positive for juvenile rainbow trout in an extended period temperature regime ranging between 21 °C to 26 °C (Dockray et al., 1996).
Seasonal Mean Weekly Average Temperature (MWAT) (°C), Week Prior MWAT (°C)	MWAT is one of the most universally used temperature metric and therefore will be the most comparable across different studies. It is a measure of chronic temperature exposure (Carter, 2005).	Juvenile coho presence in sites less than 14.5°C MWAT, absence in sites MWAT 16.7°C (Welsh et al., 2001).  Juvenile coho rarely found where daily max exceeded 21°C (McCullough, 1999).		Steelhead trout growth expected to improve by increasing temperatures to approximately 16.5°C if ample food present (Wurtsbaugh & Davis, 1977).  Optimal growth expected to be an MWMT between 14.5 °C and 21 °C for juvenile steelhead salmon (Sullivan et al., 2000).
Absolute Max (°C)	A measure of the most extreme, acute temperature experienced in a system. This metric would likely capture lethal temperatures if reached.	Juvenile coho rarely found where daily max exceeded 21°C (McCullough, 1999).		Steelhead salmon juveniles expected to actively avoid pools that have a daily maximum temperature greater than 22 °C by moving to cooler pools (Nielsen et al., 1994).  ***
Seasonal Standard Deviation (SD) (°C), Week Prior SD (°C)	Fluctuations in temperatures cause stress to fish and affects species to differing degrees. Climate change is projected to increase extreme weather events in the future resulting in larger fluctuations seen in stream temperatures (Fischer et al., 2021).	Coho Larger temperature fluctuations will result in lower abundance of coho due to their		
Boulder (%), Pool Boulder (%)	Boulders possess the potential to form pools and provide cover for predator avoidance or refuge from high flow conditions (Bisson et al., 1988).	More coho due to their habitat preference for pools (Bisson et al., 1988).	A significant increases in juvenile cutthroat trout densities was found after the placement of boulders and gabions (House, 1996). Therefore, we believe there will be a higher relative abundance of cutthroat trout when there is a higher percentage of boulder substrate present.	
Cobble (%), Pool Cobble (%)	Macrobenthos, many of which are juvenile salmonid food sources, have a preference for cobble type substrate which provide interstitial spaces for hiding and oviposit locations (Brusven & Prather, 1974). Interstitial spaces provided from this substrate type may also provide cover for small salmonids from predators.	Increased relative abundance of coho.	Increased relative abundance of cutthroat trout.	Increased relative abundance of trout.
Gravel (%), Pool Gravel (%)	Substrate deemed as potential spawning habitat.	Increased spawning habitat potential therefore higher probability of juvenile coho.	Increased spawning habitat potential therefore higher probability of juvenile cutthroat trout.	Increased spawning habitat potential therefore higher probability of juvenile trout.
Fines (%), Pool Fines (%)	An increased presence of fines within a system can be caused by wash off from impervious surfaces, construction, and streambank erosion, all of which are a result of urbanization (MacKenzie et al., 2022). Excessive sediment loading within a system can be detrimental to fish both directly and indirectly by interfering with photosynthesis, respiration, growth, and reproduction ultimately decreasing fitness(Hale et al., 2016; Paul & Meyer, 2001; Walsh et al., 2005).	As the level of fines within a system increase the relative abundance of coho will decrease.	As the level of fines within a system increase the relative abundance of cutthroat trout will decrease.	As the level of fines within a system increase the relative abundance of trout will decrease.

Metric	Justification For Inclusion	Predicted Response of Coho Salmon	Predicted Response of Cutthroat Trout	Predicted Response of Grouped Trout**
Bankfull Width (m)	A measure of maximum available habitat at the highest possible flow conditions. Salmonids have habitat preferences based on stream size.		Bankfull channel width is expected to be the single best predictor of cutthroat presence (Rosenfeld et al., 2000).	Juvenile rainbow trout will be found in high velocity waters such as the main stem of the watershed which in general will have larger channel widths (Bisson et al., 1988).
Wetted Width (m)	A measure of the current available habitat. Larger wetted widths have a greater buffering capacity against high water temperatures. Salmonids have habitat preferences based on stream size.	Coho were found to occur in their highest densities in streams with a channel width <5m (Rosenfeld et al., 2000).	Cutthroat were found to occur in their highest densities in streams with a channel width <5m (Rosenfeld et al., 2000).	Juvenile rainbow trout will be found in high velocity waters such as the main stem of the watershed which in general will have larger channel widths (Bisson et al., 1988).
Gradient (%)	Steeper gradient results in high flows which can buffer high water temperatures but are more energetically demanding. Body morphology evolved for these conditions may help mitigate this.	Coho salmon are morphologically adapted to slow water and likely will be found in pools with an average velocity less than 20cm/s (Bisson et al., 1988; Young, 2001). They will likely distribute up the lowest stream gradients.	Due to their lack of morphological adaptations and competitive displacement by Coho salmon and rainbow trout, cutthroat trout are predicted to occupy intermediate swiftly flowing habitat (Bisson et al., 1988; Young, 2001). However, in the absence of Coho salmon and rainbow trout, cutthroat trout have been known to move into deeper, slower moving areas (Bisson et al., 1988) and will likely distribute up intermediate stream gradients.	Juvenile rainbow trout will be found in high velocity waters such as the main stem of the watershed or in areas where riffles occur (Bisson et al., 1988; Bramblett et al., 2002; Young, 2001). This is likely due to morphological adaptations to fast water which results in superior swimming ability. They also possess the most potential to distribute up the steepest stream gradients due to their high critical swimming velocity (Hawkins & Quinn, 1996).
Pools per Meter (/m), Pool (%), Residual Depth (m)	A measure of available habitat at lowest possible flow conditions and a direct or indirect measure of a specific habitat type for which specific species prefer.	Higher relative abundance of coho due to their habitat preference for pools (Bisson et al., 1988).	When coho and rainbow are absent cutthroat prefer pool habitat (Bisson et al., 1988).	
Riffle (%)	Different species prefer or competitively excluded into specific habitat. Specific species may be competitively displaced into riffle habitat when in sympatry with other salmonids or may choose this habitat based on their body morphology	Lower relative abundance of coho due to their habitat preference for pools (Bisson et al., 1988).	Higher relative abundance of cutthroat due to competitive displacement when in sympatry with coho (Bisson et al., 1988).	Higher relative abundance of rainbow due to their habitat preference for riffles (Bisson et al., 1988).
Canopy Right Bank (%), Canopy Thalweg (%), Canopy Left Bank (%)	A measure of canopy openness. Decreased canopy openness within a watershed is known to regulate water temperature, provide instream cover, and contributes nutrients and terrestrial invertebrates as food (Herunter et al., 2004; Macdonald et al., 2003; Schiemer et al., 1995).	Higher relative abundance of coho would be expected with lower percentages of canopy openness.	Higher relative abundance of cutthroat trout would be expected with lower percentages of canopy openness.	Higher relative abundance of trout would be expected with lower percentages of canopy openness.
Large Woody Debris (LWD) Volume, LWD Count	In stream LWD is known to increase overall stream productivity (Bisson et al., 1988). Increases in the number of pools within a system, habitat complexity, and slow water sections have been well documented as a result of adding LWD to a system (eg. Riley & Fausch, 1995). However, evaluations of LWD restorations have produced variable results and urban systems on average have less LWD due to direct removal, reduced riparian area, and washout (Booth et al., 1997).	Juvenile coho densities expected to be higher in systems with more LWD (Roni & Quinn, 2001). Expect increased level of juvenile coho after LWD restoration (House, 1996). Expect no change in juvenile coho levels after restoration (Cederholm et al., 1997).	Expect juvenile cutthroat trout relative abundance not to change based on LWD levels (Roni & Quinn, 2001).	Juvenile steelhead trout and trout expected not to differ in systems with more LWD (Roni & Quinn, 2001). Expect decreased trout fry density after restoration (Cederholm et al., 1997; House, 1996). Expect no change in level of juvenile steelhead densities after LWD restoration (Cederholm et al., 1997; House, 1996).
Stocked	Cypress, Eagle, Lawson, Mossom, Nelson, Noons, and Rodgers Creek were stocked with hatchery raised Coho salmon (see Appendix Table A.1 for details). Stocked sites had consistently lower water temperatures and levels of impervious surface influence indicating non-random selection.	Higher relative abundance of coho expected at stocked sites.	Lower levels of cutthroat due to competitive displacement of cutthroat trout by coho salmon when in sympatry (Young, 2001, 2004)	

\*\* The grouped trout species grouping was composed of all cutthroat trout > 80 mm, rainbow trout > 80 mm, cutthroat/ rainbow hybrid trout, and all general trout < 80 mm. All information under the *Predicted Response of Cutthroat Trout* column is also applicable to this column.

\*\*\* The shared nature of the literature for the temperature metrics resulted in grouping of the predicted response sections by species to reduce excessive repetition of information.

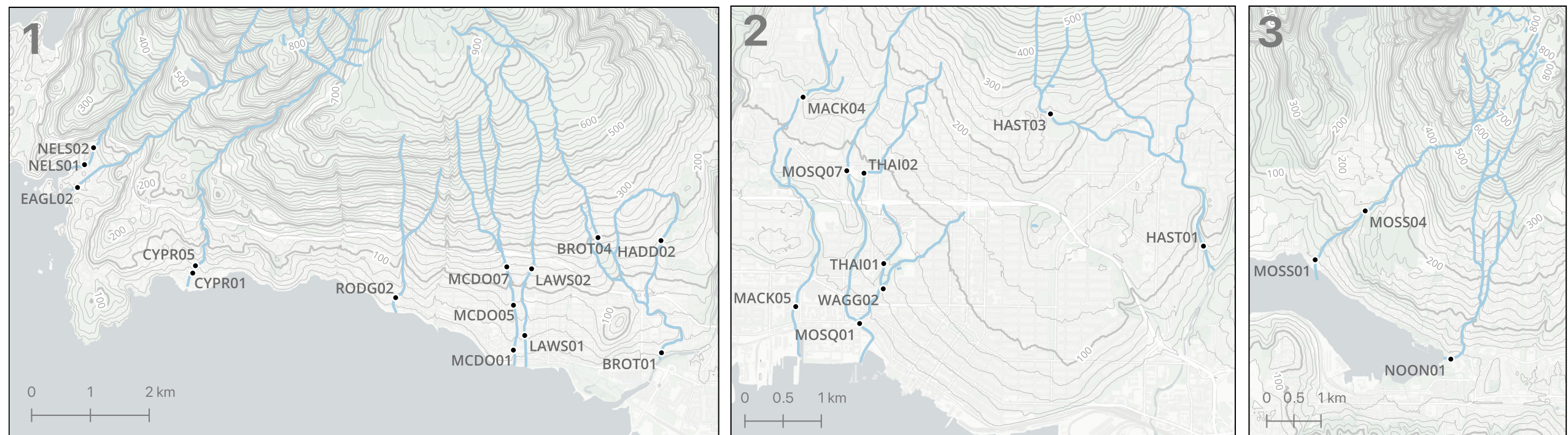
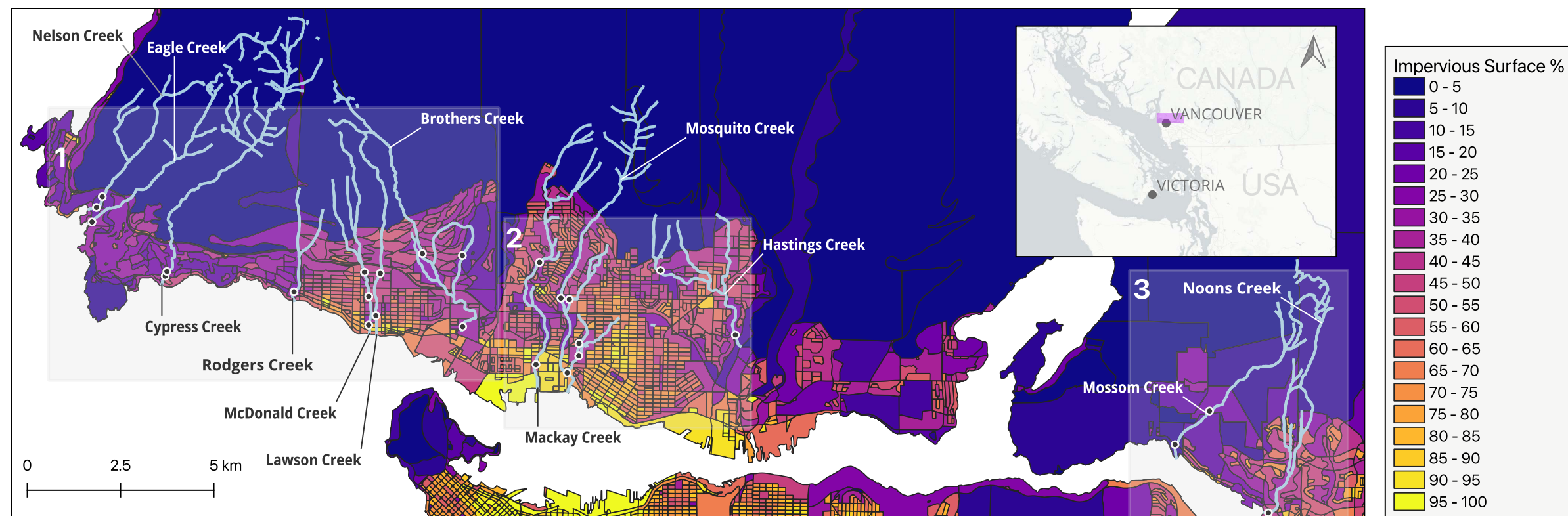
# Methods

## Study Location, Site Selection, and Species Groupings

Metro Vancouver sits in the West Coast climate region of Canada and is characterized by high precipitation rates for much of the year except for the summer months of July and August which are typically dry and warm, occasionally resulting in moderate drought conditions. Encompassing the lower Fraser River valley and smaller neighbouring watersheds, Metro Vancouver simultaneously possesses extensive urban development and complex fish communities. Approximately 40 fish species with varied habitat requirements and life histories exist in the 1200 streams that make up the lower Fraser Valley (Fisheries & Oceans Canada, 1998; McPhail, 2007). Unfortunately, many of the historical streams in the area have been substantially altered or lost all together through channelization, cleared riparian buffer zones, and pollution from urban development making the present available stream habitat now more critical than ever for future persistence of fish communities in Metro Vancouver (Finn et al., 2021).

This study was conducted in twelve unique watersheds in Metro Vancouver that span from West Vancouver to Port Moody, with most sites clustered across West and North Vancouver (Figure 1). Sites were sampled based on knowledge of previous fish presence and where there was a range in gradient, urbanization, and historical temperatures from temperature loggers deployed at each site. The initial study design intention was to sample pre and post maximum stream temperatures for the summer season, however, during the summer of 2021 Vancouver experienced an extreme heat dome from June 25th to July 1st which occurred just prior to the first planned sampling event ([Environment Climate Change Canada](#)). Many of these smaller systems experienced peak water temperatures during this time (Figure 3), 4-6 weeks earlier than typical peak temperatures seen in these systems. Because of this, both fish sampling events occurred after the heat dome. The second sampling period occurred after the normal peak temperatures in early August and serves as a repeat sampling of fish distribution after high temperature exposure to provide confidence in consistency of the July results, and to explore potential temporal differences across the summer period.





**Figure 1.** Map of Northern Metro Vancouver, highlighting the sampled watersheds shown in light blue and sample site locations within watersheds shown as black and white triangles. Percent impervious surface is represented by the colored polygons with colors associated with percent values.



I focused my study on coho salmon, cutthroat trout, and rainbow trout and their associated habitat requirements. Both trout species are optionally iteroparous and exhibit anadromous and non-anadromous life histories spending 2 to 5 years in freshwater (Trotter, 1989). Coho salmon are anadromous, semelparous, and spend 18 to 30 months in fresh water (Groot & Margolis, 1991) prior to undergoing smoltification and migrating to the marine environment. At the juvenile life stage these three species are known to occur in sympatry, often resulting in interspecific competition which can be avoided through habitat partitioning if ample habitat complexity is available (Young, 2001, 2004). Body morphology and territoriality also both play a role in species habitat preferences. Due to body morphology adaptations to slower moving water, coho salmon possess a strong preference for pool type habitat (Bisson et al., 1988; Young, 2001). In contrast, rainbow trout are more likely to be found in high velocity water such as riffles due to their superior swimming ability (Bisson et al., 1988; Bramblett et al., 2002; Young, 2001). Coho salmon and rainbow trout competitively exclude cutthroat from their preferred pool habitat (Bisson et al., 1988). When in allopatry cutthroat have been known to move into deeper, slower moving pools (Bisson et al., 1988). It is assumed that there are no competitive interactions between species groupings in the following study.

In the field, trout less than 80 mm were classified as general trout due to identification challenges below that size. Above 80 mm trout were classified as either cutthroat, rainbow, or cutthroat/rainbow hybrids based on identification traits as outlined by Watershed & Environmental Management, personal communications, July 2, 2021. To briefly summarize, fish were identified as cutthroat if they possessed a red slash on the jaw, 1-3 white interspaces between fin rays, their maxillary extended past the rear eye margin, and if their adipose fin had one or more breaks in the black outline. Fish were identified as rainbow trout if they did not have a red slash on the jaw, they had 3-5 white interspaces between fin rays, the maxillary did not extend past the rear eye margin, and there were 0 or 1 breaks in the black outline on the adipose fin. Hybrids were identified as fish possessing any combination of cutthroat or rainbow traits. See appendix Figure A.1 for a visual representation of the above information.

For analysis, species were separated into three groups to maximize the use of collected data. The coho salmon species grouping was composed of coho only,

cutthroat trout larger than 80mm made up the cutthroat trout species grouping, and the grouped trout species grouping was composed of all cutthroat trout > 80 mm, rainbow trout > 80 mm, cutthroat/ rainbow hybrid trout, and general trout. Due to likely shared life history traits, a grouped trout species grouping was justified.

Several watersheds had intentionally stocked coho salmon, most of which were unmarked in 2021 due to COVID related restrictions (see Appendix Table A.1 for release numbers). All fish stocked were fry except in Mossom and Noon's Creek where smolts were stocked. For my analysis, all sites within a watershed were identified as stocked if coho stocking occurred within the watershed, coho were present, and substantial barriers were not in place of potential migration. There were a few exceptions to the above criteria. Mossom and Noon's Creek were not identified as stocked because all the fish caught in these systems during the study period were below the smolt release size implying the hatchery smolts all left the system as expected. A high-water level event resulted in marked, fry sized coho to escape from the Noon's creek hatchery where they were being reared from future release as smolts. As a result, marked fish were captured at the NOON01 site and excluded from the analysis.

## **Fish Sampling**

Minnow traps were used to sample sites for species presence and relative abundance (Table 2; see Appendix Table A.7 for CPUE data by site) between July 19<sup>th</sup> and July 30<sup>th</sup> and August 23<sup>rd</sup> to August 27<sup>th</sup>, 2021. In this study relative abundance refers to sites based relative abundance. Five, coarse (1/4" mesh), galvanized steel minnow traps with 1" openings were deployed at each site. Traps were baited with preserved salmon roe and then set in the early afternoon. The following day (average soak time July = 20.08 hr +/- 1.67, August hr = 19.59 +/- 1.00), traps were checked and then removed and moved to another site. Latitude and longitude GPS coordinates, the habitat that each trap was set in, the date and time the trap was set, weather conditions, and water quality metrics were recorded at the time of setting.

Fish were removed from the traps and held in aerated 20 L buckets of stream water prior to being identified and counted. Length measurements were taken using a fish viewer. Fish were placed into a different aerated 20 L bucket of water and released at the same location that they were caught. This was repeated for each trap. Date and

time, number of traps recovered, and presence of an adipose fin (indicative of a hatchery release, as in stocked fish) was recorded when checking the traps.

Minnow trapping was the chosen methodology since electrofishing in an urban setting comes with an increased risk of electrocution of humans, dogs, and other animals. To account for varying trap-hours and soak times between sites, fish catch measurements were converted into Catch-per-Unit-Effort (CPUE). A CPUE value was calculated for each group at each site with the following formula:

$$CPUE = \frac{C}{s * t}$$

Where  $C$  is the number of fish caught at each site,  $s$  is the total soak time in hours, and  $t$  is the number of traps set. It should be noted that minnow trapping CPUE calculations typically requires a catchability estimate which was not available for this system. The only instances of a decreased sample size of traps set occurred at one site in Brother's, Mossom, and Mosquito creek in July where the sample size of traps set was decreased from five to four. In August, only one reduced sample size of traps occurred at a site in Hasting's creek when a citizen removed a trap from the stream. Trapping results were consistent with eDNA results sampled from the same sites, increasing confidence (unpublished data, appendix Table A.2). For example, the thirteen locations where coho were not trapped only three of them tested positive for coho eDNA, all three of which were downstream of known coho trapping locations. All nineteen locations where coho were trapped also tested positive for eDNA and the ten eDNA sites that had no coho eDNA detected also failed to trap coho.

**Table 2. Summary of sampling period specific response (i.e. CPUE variables) and predictor variables (i.e. weekly temperature metrics) and their description. The mean and range are derived from the mean and ranges of site means.**

Metric	Description	July		August	
		Mean	Range	Mean	Range
Week Prior MWMinT	Mean Weekly Minimum Temperature one week prior to trapping (°C)	15.32	12.18 - 16.62	15.34	14.64 - 16.34
Week Prior MWMT	Mean Weekly Maximum Temperature one week prior to trapping (°C)	16.95	14.40 - 19.09	16.47	15.31- 17.79
Week Prior MWAT	Mean Weekly Average Temperature one week prior to trapping (°C)	16.04	13.06 - 17.59	15.84	15.00 - 16.87
Week Prior SD	Mean Weekly Standard Deviation of Temperature one week prior to trapping (°C)	0.55	0.23 - 1.22	0.36	0.17 - 0.65
CPUE Coho Salmon	Catch per unit effort of Coho Salmon (#/hr)	0.13	0 - 0.71	0.10	0 - 0.64
CPUE Cutthroat Trout	Catch per unit effort of Cutthroat Trout (#/hr)	0.06	0 - 0.20	0.06	0 - 0.19
CPUE Grouped Trout	Catch per unit effort of Grouped Trout (#/hr)	0.02	0 - 0.05	0.02	0 - 0.06

## Stream Temperature

Stream temperatures were recorded hourly using temperature data loggers (Hobo Pendant; accuracy of  $\pm 0.5^{\circ}\text{C}$  from  $-20^{\circ}\text{C}$  to  $70^{\circ}\text{C}$  or Tidbit; accuracy of  $\pm 0.25^{\circ}\text{C}$  from  $-20^{\circ}$  to  $0^{\circ}\text{C}$  and  $\pm 0.2^{\circ}\text{C}$  from  $0^{\circ}$  to  $70^{\circ}\text{C}$ , Onset, Bourne MA, [www.onsetcomp.com](http://www.onsetcomp.com)). Loggers were encased in a protective, perforated white PVC housing. Loggers were deployed by either epoxying a housing to a large immobile boulder or by zip tying a small piece of PVC pipe to a stable root or human-made structure (U.S. EPA, 2014). In all cases, loggers were placed out of sight and reach from by-passers to avoid vandalism. Logger temperature data were plotted and systematically checked against a NIST approved thermometer in the field to ensure accuracy.

Given the unknown degree of movement of individuals within my study system, I calculated both week prior (See Table 2) and seasonal (See Table 3) temperature metrics. The week prior temperature metrics are assumed to be a more reliable indicator of temperatures experienced by individuals if movement was prevalent in the system. Mean Weekly Minimum Temperature (MWMinT), Mean Weekly Maximum Temperature (MWMT), Mean Weekly Average Temperature (MWAT), and Mean Weekly Standard Deviation (SD) was calculated for both the seasonal and week prior metrics and an

Absolute Maximum Temperature metric was calculated as a seasonal temperature metric only. More details on each temperature metric and their rationale can be found in Table 1.

Starting on June 21<sup>st</sup>, 2021, a 7-day running average was calculated for the daily maximum, average, minimum, and standard deviation of temperatures seen at each site. The highest value among each of these averages was selected to make the seasonal temperature metrics. The week prior temperature metrics were calculated by taking the average daily maximum, average, minimum, and standard deviation of temperatures exactly one week prior to each trapping date. More information on the temperature metrics included and predicted response of each species grouping can be found in Table 1 and a visual representation of the raw temperature data can be found in Figure 3. Temperature data by site can be found in Appendix Table A.5 and A.7.

## **Physical Habitat Features**

Data on habitat measurements were collected for each site where fish were trapped following various methodologies outlined below producing 22 habitat metrics (Table 3; see Appendix Tables A.4, A.5, A.6 for habitat data by site). For the purposes of this paper, the term reach and site are synonymous to one another. Reach length was 12 times the wetted width or the distance between the most upstream and downstream trap locations, as opposed to a commonly used standard 30 times the bankfull width described by Bain & Stevenson (2001), because this 12 times length typically included the location of all traps that were set at each site. The decided reach length was based off the Pacific Streamkeepers Handbook developed for Pacific salmon in British Columbia, which states that, on average, a pool-riffle sequence is repeated every six times the bankfull width and a full S-shaped meander every twelve (Taccogna & Munro, 1995). Hence, an appropriate range of habitats were captured within each reach by the presence of multiple habitat types within each sampling site.

Each reach was surveyed according to a stratified random sampling method. The reach was divided into four equal sections, within each section three transects were randomly placed totaling 12 transects per reach. At the reach level, macrohabitat, large woody debris, and gradient variables were measured. At each section channel morphology variables were measured. At each transect percent sediment cover and

canopy cover variables were measured. When the established reach length did not include the location of all the traps set at a particular site, macrohabitat, gradient, and total reach length measurements were extended to include the location of all traps.

### ***Reach Level***

Macrohabitats consisted of riffles and pools and were identified according to Bain & Stevenson (2001). Runs, glides, and cascades, as well as the total length of the undercut bank greater 30 cm in depth were measured but not included in the final models due to variance structure issues (i.e. inability to transform the data to meet assumptions of normality required in the analysis). The length, width, and depth of each habitat unit was measured to the nearest cm and then the area of each habitat unit was converted to a percentage of the total reach area. Pools were identified following the criteria described in Fausti et al. (2004) which were as follows: an identifiable upstream crest, a downstream tail, and the maximum depth of the pool being greater than 1.5 times deeper than the tail depth. The mean residual depth of pools was calculated using the following formula:

$$d_r = d_m - d_t$$

Where  $d_r$  = residual depth,  $d_m$  = maximum depth, and  $d_t$  = tail depth. The percent sediment cover was also recorded for each pool as finer sediments can often accumulate in pools from wash off from impervious surfaces, construction, and streambank erosion, all of which are a result of urbanization (MacKenzie et al., 2022). An explanation of the percent sediment classifications and methods can be found in the *Section Level* section.

Large woody debris (LWD) was measured following the protocol outlined by Roni & Quinn (2001). All pieces of wood > 1.5 m long and > 10 cm in diameter were included in the survey. All pieces of wood within the bankfull channel were measured and living trees were excluded from the count. Length and diameter measurements were taken from each piece of wood and then converted into volume ( $m^3$ ) using the following formula:

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

Where  $V$  = volume,  $r$  = radius, and  $h$  = height. The LWD volume metric was calculated by taking the sum of all LWD volumes and dividing by reach habitat area ( $m^2$ ).

LWD counts were simple a sum of all pieces of wood found within a site. The same two individuals conducted the LWD measurements for the entirety of the study to reduce samplers' bias.

Stream gradient was measured using a laser range finder ([TruPulse 360°R](#)) and a target taped to a wading staff at the eye level of the sighting individual. The distance from the target (accuracy  $\pm 0.2$  m) as well as the gradient (accuracy  $\pm 0.25^\circ$ ) was recorded. Multiple measurements were taken, if necessary, to accurately measure the meander of the system. The number of measurements required to obtain the total reach gradient varied depending on the degree that the stream meandered, how much vegetation was at the eye level of the sighter, and the presence of suitable standing locations to shoot from. A weighted average of the measurements was taken to create the gradient metric for the reach.

### ***Section Level***

Wetted width was measured at the water's surface perpendicular to flow from the water's edge on one side to the water's edge on the opposite side. Bankfull width was determined by looking for a change in vegetation from seasonal vegetation to permanent vegetation, such as trees and shrubs when present or from moss absence to presence. Once determined, bankfull width was measured from bank to bank at these transition points. If the system was separated by a vegetated island, then the sum of each channel width was used to calculate wetted width and bankfull width using the same methods described above. Both wetted width and bankfull width were measured using a measuring tape. Bankfull height was determined by placing a meter staff in the thalweg of the stream and, using a clinometer ([Suunto PM5/360PC](#), accuracy  $1/4^\circ$ , graduation interval  $0.5^\circ$ ), sighting to the transition point of the bankfull (Bain & Stevenson, 2001). All three of the above measurements were recorded at the start and end of the reach and at each section break for a total of five measurements per site.

### ***Transect Level***

Plot placement at each random transect alternated between left bank, thalweg, and right bank for a total of 12 estimates per reach. At each transect, percent sediment cover was visually estimated by classifying percent cover of boulder, cobble, pebble, gravel, and fine substrate types within a  $1\text{ m}^2$  quadrat. Substrate size classes were

adapted from a modified Wentworth classification described in Bain & Stevenson (2001) and were as follows: if the intermediate axis of the stone was > 256 mm it was classified as boulder, 64-256 mm as cobble, 4-64 mm as pebbles, 2-4 mm as gravel, and <2 mm as fines. For each pool habitat unit, a percent sediment cover estimate was recorded for the entire pool following the same substrate size class as above. Calipers were used to measure the intermediate axis of sediment types and percent cover reference diagrams were compared to visual estimates in the field to ensure accurate and consistent measurements across all sites. The mean percent cover of each substrate type was calculated for both the quadrat and pool estimates for each reach.

Three canopy openness measurements were taken at each transect using a convex spherical densiometer (Lemmon, 1956). While facing upstream, one reading was taken at the left bank, one in the thalweg, and one at the right bank for a total of 36 readings per site (12 readings per location within the stream). The densiometer was held at a 90° angle at elbow level. Each of the twenty-four ¼ in. squares etched into the mirror of the densiometer were scanned systematically. Within each of the 24 squares, four imaginary equally spaced dots were counted if they fell within an area of canopy opening (i.e. sky) for a total of 96 possible dots. The mean was calculated for each of the locations within the stream and multiplied by a correction factor of 1.04 to obtain the mean estimated percentage value of the left bank, thalweg, and right bank canopy opening per reach.



**Table 3. Summary of seasonal predictor variables and their description. The mean and range are derived from the mean and ranges of site means.**

Metric	Description	Mean	Range
Seasonal MWMinT (°C)	Mean Weekly Minimum Temperature for the summer season (June 21st-August 30th)	17.21	14.01 - 19.68
Seasonal MWMT (°C)	Mean Weekly Maximum Temperature for the summer season (June 21st-August 30th)	18.90	16.31 - 21.25
Seasonal MWAT (°C)	Mean Weekly Average Temperature for the summer season (June 21st-August 30th)	17.91	14.95 - 20.40
Seasonal SD (°C)	Mean Weekly Standard Deviation of Temperature for the summer season (June 21st-August 30th)	0.69	0.32 - 1.72
Absolute Max (°C)	Absolute Maximum Temperature for the summer season (June 21st-August 30th)	20.09	17.29 - 22.65
Bankful Width (m)	Average width perpendicular to the direction of flow of the dominant channel forming flow that reoccurs every few years	6.4	2.5 - 12.5
Wetted Width (m)	Average width perpendicular to the direction of flow of the wetted stream	4.2	1.7 - 8.6
Bank Height (m)	Average maximum vertical height of the dominant channel forming flow that reoccurs every few years within a watershed	0.7	0.3 - 1.0
Canopy Right Bank (%)	Average canopy opening estimate value of the right bank of the watershed	14	3 - 36
Canopy Thalweg (%)	Average canopy opening estimate value of the thalweg of the watershed	14	4 - 38
Canopy Left Bank (%)	Average canopy opening estimate value of the left bank of the watershed	11	2 - 35
Boulder (%)	Average percent cover of substrate >256 mm	34	0 - 68
Cobble (%)	Average percent cover of substrate 64-256 mm	25	0 - 47
Pebble (%)	Average percent cover of substrate 16-63 mm	27	9 - 52
Gravel (%)	Average percent cover of substrate 2-15 mm	6	2 - 19
Fines (%)	Average percent cover of substrate <2 mm	7	0 - 53
Gradient (%)	Average slope of the reach	3.3	0.2 - 8.5
Pool Area (%)	Average cover of pool habitat	33	3 - 77
Riffle Area (%)	Average cover of riffle habitat	33	0 - 82
LWD Volume	Average volume of large woody debris in m3/m2	0.0	0 - 0.1
LWD Count	Average number of pieces of large woody debris within a reach	8.3	0 - 29
Pool Boulder (%)	Average percent cover of substrate >256 mm in pool habitat	27	0 - 75
Pool Cobble (%)	Average percent cover of substrate 64-256 mm in pool habitat	22	0 - 35
Pool Pebble (%)	Average percent cover of substrate 16-63 mm in pool habitat	28	5 - 55
Pool Gravel (%)	Average percent cover of substrate 2-15 mm in pool habitat	9	3 - 22
Pool Fines (%)	Average percent cover of substrate <2 mm in pool habitat	15	1 - 62
Pools/ m	Average number of pools per meter	0.07	0.01 - 0.16
Residual Depth (m)	Average pool residual depth	0.38	0.13 - 0.83

## Urbanization

I used impervious surface as a surrogate to estimate the degree of urbanization that exists directly upstream of the sampling locations (Wang et al., 2000, 2001; Table 1). To estimate percent impervious surface of each watershed, I used Arcmap (ArcGIS, <https://pro.arcgis.com>). Percent impervious surface was generated from the 2014 Metro Vancouver Land Cover Classification shapefile dataset created using 5 m resolution land cover data. Data were downloaded from Metro Vancouver's Open Data Portal. The downloaded feature class summarizes the percent impervious surface by census block represented as polygons with impervious weightings being applied based on land cover using the following classifications: 100% for buildings, other built, paved and urban shadow; 75% for barren; 50% for soil and non-photosynthetic vegetation; 10% for modified or natural grass-herb; 0% for coniferous, deciduous, shrub, and non-urban shadow. Watershed boundary data and stream paths were downloaded from the Freshwater Atlas Watersheds dataset from the B.C. data catalogue and consisted of small polygons that represent base units of watersheds and lines to form the stream paths (GeoBC, 2019).

The watershed boundary was created by selecting and merging all watershed polygons draining into the stream of interest to form a single polygon per watershed. To separate out the upstream influence for each site a Digital Elevation Model (DEM) was downloaded from the province of British Columbia website and a watershed delineation was conducted to create individual drainage basin polygons per site. The drainage basins per site were then used in conjunction with the watershed boundary polygons to inform how the watershed boundary polygon be separated to populate each site's final drainage basin. Further details on the watershed delineation process are outlined on page 12-17 in ESS2222 Fall 2013 Participants (2013).

The populated polygons were then used to clip the percent impervious surface layer to determine the percent impervious surface influencing at each site using the following formula:

$$I = \frac{I_b * A}{A_w}$$

$I$  is the percent impervious surface of the drainage basis,  $I_b$  is the sum of all impervious surface percentages from each census block included in the drainage basis,  $A$  is the area of each census block included in the watershed boundary, and  $A_w$  is the sum of all census block areas included in the drainage basin.

## Analysis

To describe species relative abundance within the study setting, I used partial least squares regressions (PLSR) for each species grouping and month resulting in a total of six models. The models for July included 12 watersheds and 26 sites. August's models included the same watersheds sampled in July minus Mossom and Noon's creek for a total of 10 watersheds and 23 sites. Mossom and Noon's were excluded due to time constraints. The number of sites within a watershed ranged from 1-5 (See Figure 1 for the location of sites and Figure 3 for the number of sites per watershed). Separate models were created for each month to avoid pseudo replication from multiple sampling events in one season. Response variables differed between models, however, it should be noted that all models shared the same habitat metrics except for the week prior temperature metrics which differed between months. Watersheds were included as a variable, to account for similarities between sites in the same watershed and fish are assumed to be static in the model. The number of predictor variables varied between models and are outlined in full below. The following methodology was chosen due to the supervised nature of the questions posed, and the presence of numerous, highly correlated variables (Scott & Crone, 2021).

I used PLSR with a Poisson error distribution and an offset of log area sampled to collapse 24-31 scaled and centered predictor variables (see Table 2 and Table 3 for variables and descriptions) into multiple latent variables. Partial least squares regression is a dimension reduction strategy similar to Principle Component Analyses. However, PLS is more suitable for datasets with numerous multicollinear predictor variables because the model can split different proportions of each variable into different components such that the components are not colinear. Further, the components created explain the maximum variability in the response variable compared to a PCA which constructs composite variables to explain the maximum variability in all the

original predictor variables. Analysis followed methods from Bertrand & Maumy-Bertrand (2018) and Scott & Crone (2021).

I first used PLSR to fit all available variables, and pruned variables by variable importance. Average variable importance was determined through K-fold cross validation, whereby models were fit to 90% of data, and used to predict 10% of the data 1000 times. In this way, outliers had reduced influence on model fit and variable importance. All variables with less than 0.1 variable importance were excluded from the final model as they had low influence on the response variable. Pruned models were re-run and the number of latent variables, or hyperparameters, were determined using K-fold cross validation. The number of components was determined by comparing models with up to 20 components. I then compared the number of significant predictors in each model sequentially by number of components. The final model was the model with the most significant predictors before a large drop in significant predictors of the next sequential model. For example, the cutthroat in August model with 2 components had 15 significant predictors whereas the model with 3 components had 4, therefore the model with 2 components was selected. I report the top two latent variable loading in biplots as well as regression coefficients.

From the PLSR results, I report the top five metrics with the largest absolute value coefficients regardless of their significance status. I chose to do so since the sample size of the study was small and therefore likely does not have enough power to detect small effects. Variables within this top five list that were not significant have a potentially large, but uncertain effect. Refer to Figure 2 for a list of the included predictor variables and coefficient values in each model and Table 2 and Table 3 for variable descriptions.

Based on my PLSR results I used impervious surface percent, a proven driver of stream warming, channel degradation, and decreased species assemblage diversity (Nelson & Palmer, 2007; Spanjer et al., 2018; Wang et al., 2000, 2001) to predict a few of the key predictor metrics derived from the above PLSR, namely temperature, percent fines, and percent canopy openness. Only sites that were independent from one another were included in the following models. For example, for Brother's Creek, BROTH4 and HADD02 sites were used and BROTH1 site was excluded due to its downstream connection to both upstream sites. To predict the effect of impervious surface percent on

the seasonal MWAT, I used a linear model with a gaussian error distribution. I converted fines and canopy openness percentages to proportions by dividing by 100 and used a generalized linear regression with a beta error distribution and logit link function. Beta error distributions are used when data are bounded between 0 and 1.

Analysis was completed in R (R 4.2.1) using RStudio GUI (version 2023.06.1+524) and the following packages: plsRglm package (Bertrand & Maumy-Bertrand, 2022), glmmTMB (Brooks et al. 2017), and the base R stats package (R Core Team, 2022).

## Results

A total of 790 coho and 519 trout were captured in the summer of 2021 from the 12 study watersheds. Relative abundance of coho salmon in July was best described by a one-component PLSR model made up of 26 variables, which described 57% of observed variation (see Figure 2). The resulting top five predictor variables in order of absolute coefficient value and direction of effect on coho salmon relative abundance in July were as follows: Fines % (negative), Stocked (positive), Week Prior SD (negative), Week Prior MWMT (negative) and Pool Fines (negative). All variables were significant. See Appendix Table A.8 for coefficient values and rankings.

Relative abundance of Coho salmon in August was best described by a one-component partial least-squares regression model made up of 27 variables, which described 57% of observed variation (see Figure 2). The resulting top five predictor variables in order of absolute coefficient value and direction of effect on coho salmon relative abundance in August were as follows: Fines % (negative), Stocked (positive), Gravel (negative), Pool Fines (negative), and Seasonal MWMinT (negative). In this case, Seasonal MWMinT was not significant, indicating a potentially strong but uncertain effect on Coho relative abundance. See Appendix Table A.8 for coefficient values and rankings.

Unsurprisingly, the stocked variable was one of the top 5 predictor variables of Coho salmon relative abundance during both the July and August sampling periods. In addition, warmer and more variable water temperatures as well as high levels of fine sediment were negatively associated with juvenile coho salmon.

Relative abundance of cutthroat trout in July was best described by a one-component partial least-squares regression model made up of 26 variables, which described 48% of observed variation (see Figure 2). The resulting top five predictor variables in order of absolute coefficient value and direction of effect on cutthroat trout relative abundance in July were as follows: Wetted Width (negative), Canopy Thalweg (negative), Bankfull Width (negative), Cobble % (positive), and Canopy Left Bank (negative). All variables were significant. See Appendix Table A.9 for coefficient values and rankings.

Relative abundance of cutthroat trout in August was best described by a two-component partial least-squares regression model made up of 30 variables, which described 67% of observed variation (see Figure 2). The first component captured 34% of the variation, the second 35%. The resulting top five predictor variables in order of absolute coefficient value and direction of effect on cutthroat trout relative abundance in August were as follows: Wetted Width (negative), Gravel % (negative), Cobble % (positive), Bankfull Width (negative), and Week Prior MWMinT (negative). All variables were significant. See Appendix Table A.9 for coefficient values and rankings.

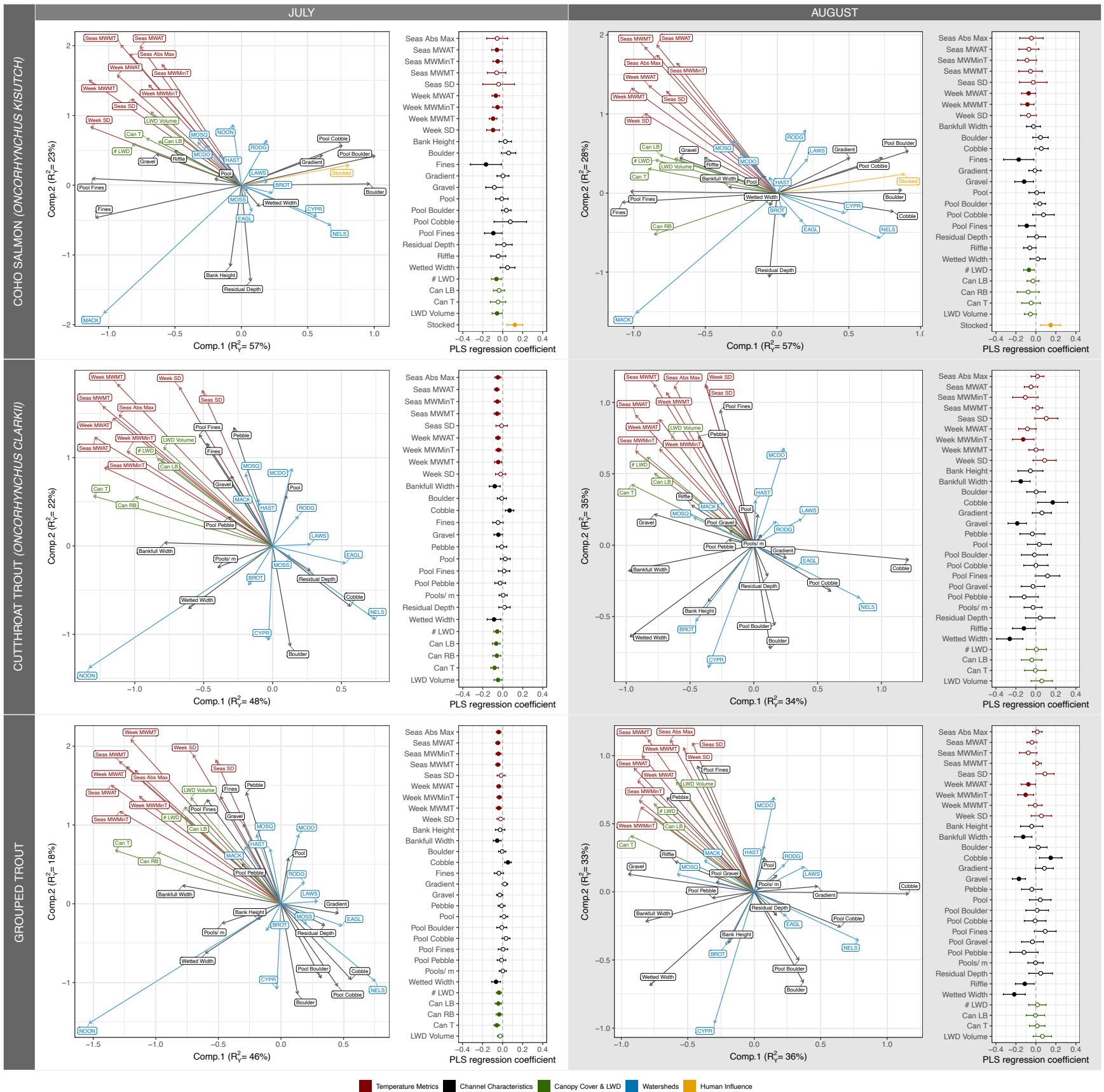
Relative abundance of Grouped trout in July was best described by a one-component partial least-squares regression model made up of 29 variables, which described 46% of observed variation (see Figure 2). The resulting top five predictor variables in order of absolute coefficient value and direction of effect on cutthroat trout relative abundance in July were as follows: Wetted Width (negative), Canopy Thalweg (negative), Bankfull Width (negative), Cobble % (positive), and Seasonal MWAT (negative). All variables were significant. See Appendix Table A.10 for coefficient values and rankings.

Relative abundance of Grouped trout in August was best described by a two-component partial least-squares regression model made up of 30 variables, which described 69% of observed variation (see Figure 2). The first component captured 36% of the variation, the second 33%. The resulting top five predictor variables in order of absolute coefficient value and direction of effect on Grouped Trout relative abundance in August were as follows: Wetted Width (negative), Gravel % (negative), Cobble % (positive), Bankfull Width (negative), and Pool Pebble (negative). All variables were significant except for Pool Pebble. See Appendix Table A.10 for coefficient values and rankings.

Channel characteristics were particularly influential to both the trout groups. Wetted width produced the largest coefficients for both the cutthroat trout and grouped trout species groupings during both the July and August sampling period indicating higher relative abundance of trout in smaller systems. The cobble metrics had a consistent positive effect across both trout groupings with at least one of the cobble metrics having a large positive effect on both species groups in both sampling periods.

A collection of habitat metrics displayed similar trends across species and months. The fines, canopy openness, large woody debris, and gravel metrics had a similar effect direction and clustering pattern to the temperature metrics for each species and month (see Figure 2 Biplots). The canopy openness metrics showed consistently negative effects to all groups in July with variable responses in August where there was little to no effect of canopy openness on both trout groups relative abundance. The cobble metrics had the opposite effect direction to the above-mentioned metrics and had a positive and significant effect on both trout groups and a positive effect on the coho. Gravel had a negative effect on all species groupings and months, significantly so for all except coho and grouped trout in July. Large woody debris metrics had variable effects across months for both trout groups but were consistently negative for coho in July and August and trout in July. In all cases, at least one temperature metric emerged in the top five list of important variables for every species grouping in at least one month.



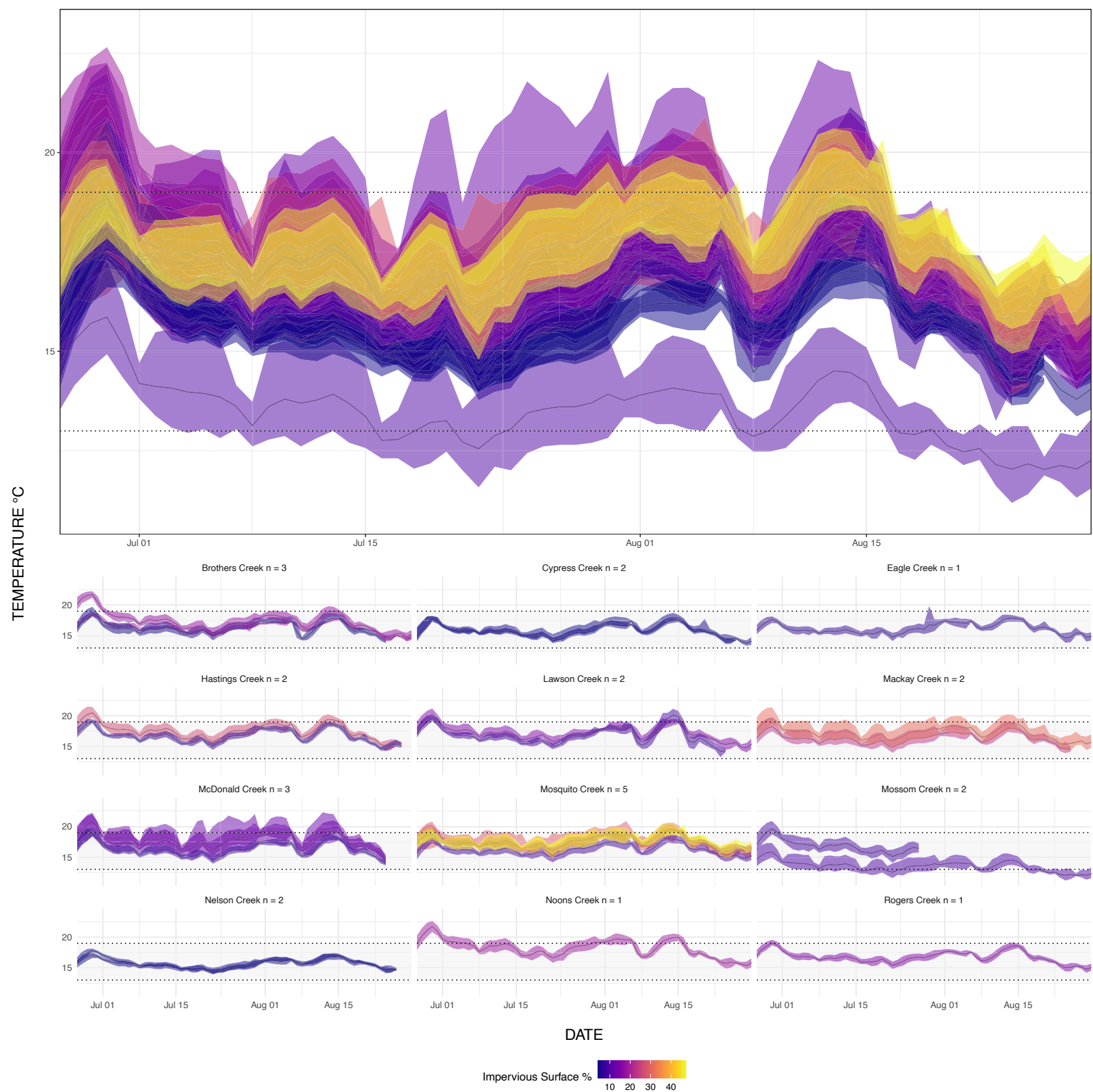


**Figure 2.** Partial Least-Squared Regression (PLSR) biplots and associated regression coefficients for each month and each species. Closed circles in the box plots indicate a significant value versus open indicate insignificant. For species and months that only had one significant component, a second component was included in the biplot for visual purposes only. CPUE data points excluded from the biplots to better visualize arrow, full extent provided in the appendix (Figure A.2, A.3, A.4).

The effect of temperature varied for different species. In both the July and August sampling periods, all temperature metrics had a negative effect on juvenile coho salmon relative abundance (Figure 2). Of these metrics all the week prior temperature metrics were significant in July and two out of three in August. The effect of the temperature metrics varied from July to August on the trout species groupings. In July, all average temperature metrics were significantly negative except for the standard deviation metrics. Only the week prior MWMInT metric for the cutthroat trout group and the week prior MWAT and week prior MWMInT metrics for the grouped trout group remained significantly negative in August. For both trout groups, the standard deviation metrics effect became positive in August, although not significantly so.

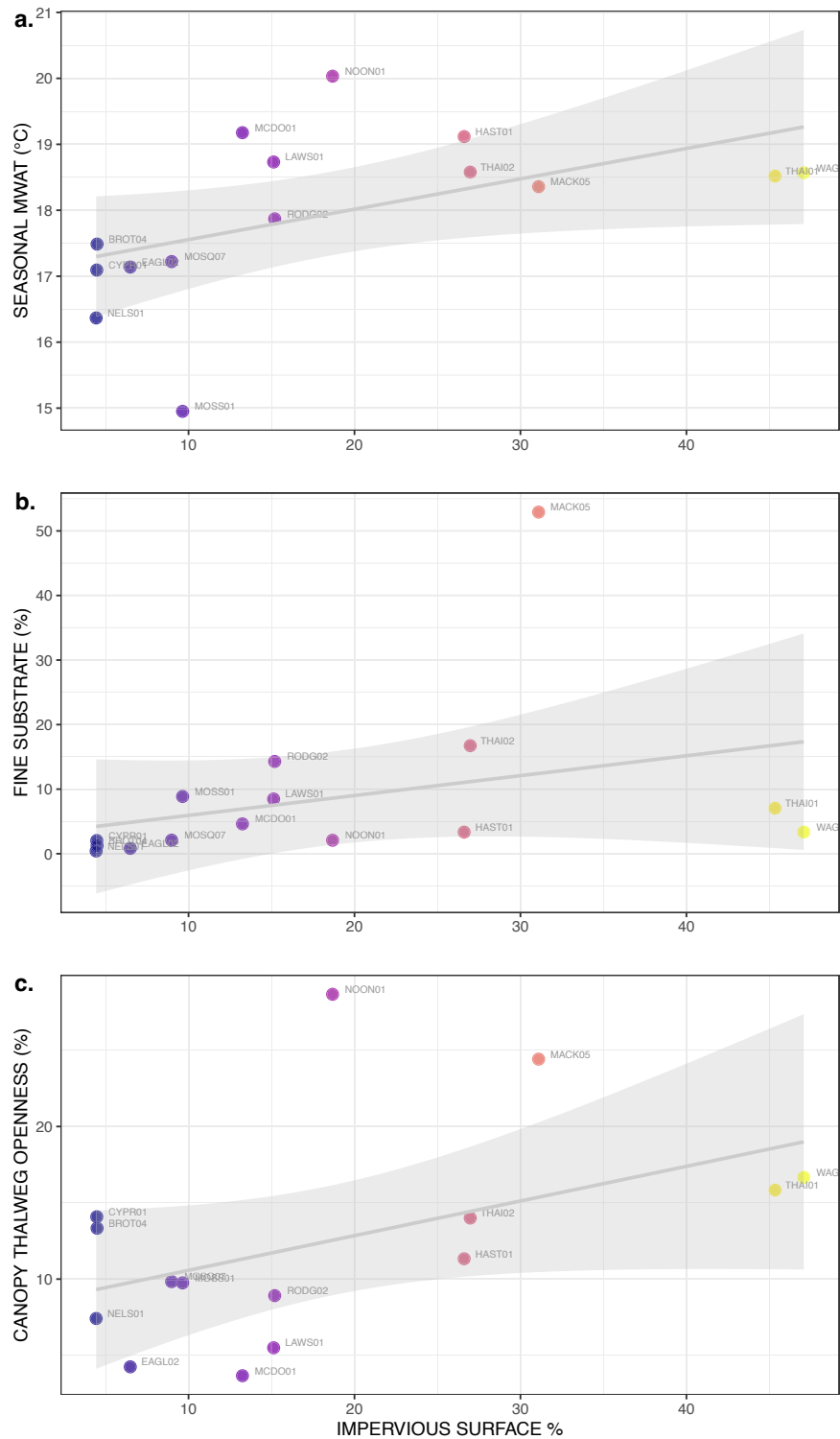
Water temperature varied across sites and watershed with most sites experiencing peak temperatures during the June heat dome (Appendix Table A.3). There was a range of variability between sites within the same watershed. Some watersheds had very similar temperature profiles across sites (e.g. Nelson Creek) and others had variable temperature profiles across sites (e.g. McDonald Creek). The highest temperatures were seen in Brothers Creek at HADD02, a site directly downstream of a golf course, where temperatures reached 22.65 °C and the lowest in Mossom creek at site MOSS01 where the average seasonal mean weekly minimum temperature was 14.01 °C.

Simultaneous temperature spikes in water temperatures were seen in Cypress and Eagle Creek on August 20<sup>th</sup> starting at 3PM for two hours. Another small spike was seen in Eagle on August 26<sup>th</sup> at 3 PM resolving within a few hours. All occurrences were paired with rain events which could be a result of the runoff impacts inflicted on these systems from urbanization. A substantial and puzzling spike in water temperature was observed in Eagle creek on July 29<sup>th</sup> starting at 10 AM. Temperatures reached 19.85 °C by 11 AM and resolving back down to 16.71 °C by 1 PM. The temperature spike was not paired with a rain event, however, an increase in paired depth data was observed indicating some sort of water input and could be the cause of the spike although the source remains unknown.



**Figure 3.** Daily average water temperatures by site (n = 26) and creek (n = 12) with daily range shown as coloured shaded area and sites coloured by upstream percent impervious surface. The temperatures between the dotted lines indicates a benchmark optimal temperature (13-19°C) for salmonid species growth. The top figure shows all sites together to visualize grouping of urbanization level by temperature. The bottom figure shows each watershed and the associated sites sampled within that watershed. The high peak at the beginning of the time series corresponds with the heat dome.

The results of the linear model indicate there was a positive relationship between percent impervious surface and seasonal MWAT ( $\beta = 0.0461$ , 95% CI [-0.0004, 0.0926]), fine sediment within a system ( $\beta = 0.0213$ , 95% CI [-0.0068, 0.0494]), and the proportion of canopy openness ( $\beta = 0.0199$ , 95% CI [0.0024, 0.0374]) (Figure 4). On average a 0.0461 °C increase in stream temperature was predicted for every 1% unit increase in percent impervious surface (Figure 4a). Therefore, water temperature was predicted to increase from 17.30 °C at the lowest impervious surface value to 19.26 °C at the highest impervious surface value, an increase of 1.96 °C on average. Fines were predicted to increase 0.2131% every 1% unit increase. At the lowest observed impervious surface value, a 6.8% cover of fines was predicted and at the highest impervious surface value 15.3%, resulting in an increase of 8.5% (Figure 4b). A 0.1994% increase in canopy openness was predicted to occur with every additional 1% impervious surface increase. At the lowest observed impervious surface value a percent canopy openness of 9.5% was predicted and at the highest impervious surface value a percent canopy openness of 19.7% was predicted, an increase of 10.2% (Figure 4c).



**Figure 4.** The impervious surface percentage of the drainage basin draining into each site by a. seasonal MWAT, b. percent fine substrate, and c. percent canopy openness at the thalweg. Points are coloured by percent impervious surface and labelled by site.

## Discussion

The results reported in this study highlight the complexity of anadromous and non-anadromous species and the interconnected impacts of water temperature, key habitat metrics, and urbanization on relative salmonid abundance. Warmer water temperatures, higher levels of percent canopy openness, and fine sediment were all negatively associated with relative abundance of cutthroat trout, general trout, and coho salmon to varying degrees during the warm and dry summer months. Moreover, the above mentioned variables are correlated with percent impervious surface, a metric of watershed urbanization (Hale et al., 2016; Walsh et al., 2005; Wang et al., 2000, 2001), highlighting the cumulative stressors from urbanization and climate change on juvenile salmon distribution in urban freshwater ecosystems.

The consistent negative impact of water temperatures on relative abundance of salmonids within this study underscores the biological significance of temperature to all ectotherms (Fry, 1971). Interestingly, the vast majority of daily water temperatures across all sites were in a temperature range that would be considered optimal for juvenile growth (Bear et al., 2007; Dockray et al., 1996; Lusardi et al., 2020; McCullough, 1999; Sullivan et al., 2000; Wurtsbaugh & Davis, 1977), suggesting that the negative impacts of higher temperatures can occur well below the commonly used physiological lethal limits (see review by Desforges et al., 2023). This is consistent with other ecological studies that found that changes to fish assemblages, diversity, and abundance have occurred at temperatures well below their proposed physiological tolerance limits (McCullough, 1999; Nielsen et al., 1994; Welsh et al., 2001). Of the species groups assessed, temperature affected coho the most, as evidenced by the presence of multiple temperature metrics in the top five absolute coefficient values from the PLSR model. Coho salmon have been observed to experience thermal stress at temperatures as low as 16°C and are recognized as the least temperature tolerant species of the Pacific salmonids (Brett, 1952; Richter & Kolmes, 2005) despite having a critical thermal limit of ~28 °C (Konecki et al., 1995). Differing life history traits could explain the varying relative importance of temperature across species if chronic warm temperature exposure has selected for more thermally tolerant non-anadromous trout versus anadromous coho.

Non-anadromous fish may experience chronic sub-optimal temperature exposure from inhabiting freshwater systems year-round and through multiple life stages (Groot & Margolis, 1991). This contrasts with anadromous species, such as coho, that only tolerate the warmest freshwater thermal regimes though a portion of their life cycle (Groot & Margolis, 1991). This could explain why temperature metrics closer in time to the trapping event performed better than seasonal temperature metrics for coho. Unfavorable thermal conditions affecting non-anadromous adults could result in negative carry-over effects into the juvenile life stage (Groot & Margolis, 1991). Likewise, chronic exposure of juveniles to suboptimal thermal conditions could reduce fitness through altered growth rate (Lusardi et al., 2020). Both scenarios would result in a decrease in trout numbers in the short term but could produce a greater tolerance to sub-optimal temperatures in the long term. Given the latter, by August a smaller number of higher temperature tolerant fish would remain, explaining the change in effect direction of some temperature metrics for trout from July to August. A difference in sample size between months could provide an alternative explanation to these results. Sites within the Mossom and Noons watersheds experienced the lowest and highest temperature regimes respectively. When these watersheds were not re-sampled in August it is likely that the model was unable to reliably assign an accurate effect size to the temperature metrics due to a lack of scope.

Habitat metrics other than water temperature can also play an influential role in species presence and abundance through various avenues (e.g. Bisson et al., 1988; Brusven & Prather, 1974; Hale et al., 2016; MacKenzie et al., 2022; Rosenfeld et al., 2000; Walsh et al., 2005). The negative association between stream width and relative abundance of the trout groups is consistent with my predictions, which were based on the species strong preference for smaller systems (Rosenfeld et al., 2000). Rosenfeld et al. (2000) found density of juvenile cutthroat trout to be significantly related to bankfull width with density generally being highest in the smallest streams. Possible reasons for this include lower flow typical of smaller systems may provide better conditions for rearing, trout may be competitively excluded into this specific habitat, and smaller systems may provide better overwinter conditions (Rosenfeld et al., 2000). Also consistent with the predicted response is the fine substrate metrics negative affect on coho salmon relative abundance. Literature has reported impacts to fish by fine sediment through increase turbidity, abrasion and scour, reduce benthic invertebrate

density, and reduced oxygen supply (Kemp et al., 2011). Due to coho's preference for pool habitat and the tendency for fines to settle in these areas it is logical that this metric would disproportionately affect this species over other more lentic trout species (Bisson et al., 1988; Kemp et al., 2011).

The lack of a positive association between higher levels of gravel substrate and LWD with fish abundance were two notable contradictions to my predicted responses. Rosenfeld et al. (2000) reported the highest abundance of cutthroat and coho in low-intermediate gradient reaches (0-5%) dominated by gravel substrate. However, the present study found a consistent negative effect of gravel on salmonids. High gravel content could be associated with higher energy streams with greater frequency of upstream bank failures and slides, although this was not tested. Large woody debris also produced unexpected results. Benefits of LWD presence are largely derived from physical modifications to a channel through increased habitat complexity, flow regulation, and pool formation (Bisson et al., 1988; Riley & Fausch, 1995). These physical modifications can also be accomplished by boulder presence. Typical of high-gradient mountain streams, study sites had a large portion of step-pool or cascade type habitat commonly formed by large, embedded substrate such as boulders (Bain & Stevenson, 2001). Montgomery et al. (1995) reported that pool spacing appeared to be independent of LWD loading in step-pool channels. Coupled with high removal rates of LWD from urban systems, the presence of boulders in high gradient systems may negating the need for LWD in fish bearing streams (Booth et al., 1997). Further work is warranted to unpack some of the puzzling associations.

Variables not directly measured in this work may provide the pathway of effect connecting key habitat variables and relative salmon abundance patterns. For example, prey availability was not measured but is an important component to salmonid persistence within a system, especially when temperatures approach the upper end of the optimal range for growth. In an *in-situ* enclosure experiment, Lusardi et al. (2020) found invertebrate prey abundance to be the predominant factor influencing age 0+ coho salmon growth, and growth rate was consistently higher in warmer reaches. Likewise, Spanjer et al. (2018) found that coho had stronger early season growth, although less efficient, in warmer, more urban streams. Taken together, the metabolic costs of elevated water temperatures in an urban stream may be mitigated if there is enhanced prey availability. However, relative prey availability among sites can be inferred from



known associations with the habitat metrics I measured. Macroinvertebrates, many of which are a juvenile salmonid food source, have a preference for cobble type substrate which provide interstitial spaces for hiding and oviposit locations (Brusven & Prather, 1974). Likewise, this interstitial space provides refuge for juvenile salmon from predators. Unsurprisingly, the presence of cobble had a positive effect on all species relative abundance, particularly trout. The consistent importance of this metric to trout specifically could again be due to life history events, given that food availability in freshwater habitat is influential to all life stages of non-anadromous trout, but only relevant to coho at the juvenile life stage. The high importance of canopy cover to trout could also be explained by this logic. Lower levels of canopy openness have been shown to regulate water temperature, provide instream cover, and contribute nutrients and terrestrial invertebrates as food, all of which are beneficial to trout fitness (Grunblatt et al., 2019; Herunter et al., 2004; Macdonald et al., 2003; Schiemer et al., 1995). Regardless of the pathway of effects, cobble abundance and canopy cover shows promise in helping combat detrimental effects of stream urbanization through restoration or protection actions.

Changes in flow, canopy cover, fine input, and water temperature through increased levels of urbanization has been well documented in the literature (Hale et al., 2016; Paul & Meyer, 2001; Walsh et al., 2005). In support of this previous work, I found that percent fine coverage, percent canopy openness, and water temperature were all positively correlated with impervious surface percent. The synchronous water temperature spikes observed in the temperature profiles in association with large basin-wide rain events are likely symptoms of the urbanized watershed, where partially regulated flows become more synchronized compared to natural watersheds (Moore et al., 2015). It is plausible that such synchrony would make it more difficult for juvenile salmon to seek temporary refuge in other systems if all are acting similarly. Previous work has demonstrated the negative impact of increasing urbanization to fish diversity, abundance, and biotic integrity (IBI) scores. Wang et al. (2001) identified a threshold value of 8-12% connected impervious surface levels, above which minor changes in urbanization could result in major changes to stream health. The extreme mid-point values exhibited by MACK05 fine sediment in Figure 4b and large canopy openness values of NOON01 and MACK05 in Figure 4c may be demonstrations of the major changes in stream health above a certain impervious surface threshold (Wang et al.,

2001). The predicted linear rate increase in water temperature with increased urbanization I found suggests a more insidious degradation of salmon habitat quality. However, these predicted rate increases can be used in conjunction with an upper temperature target by urban planners to set limits on impervious surface growth to meet specific habitat temperature requirements for fish. Determining the shape of how the key habitat metrics respond to increased levels of urbanization will be key to set urban development and restoration plans.

Improvements to the approach I have taken will need to focus on some of the key assumptions and limitations of this work. More highly urbanized areas were concentrated to lower gradient zones creating a non-random effect of the impervious surface metric. This factor was addressed by including gradient in the PLSR which would likely account for some of this misplaced variation, although not all. A substantial number of coho were stocked in these systems (Appendix Table A.1) often at sites with lower temperatures and urbanization levels indicating non-random selection. I attempted to account for this by including a stocked metric in the model however the effectiveness of this may be limited. The methodology used was also a limiting factor. The fact that all fish caught were less than 150 mm indicates the size selectivity of the minnow traps used. In general, it was not possible to include all potential metrics that are suspected to limit urban salmon populations, such as contaminants and migration barriers (Warkentin et al., 2019). Because of this a large amount of variance in fish abundance remain unexplained, a common issue expressed in the literature, specifically for coho (Rosenfeld et al., 2000). Notwithstanding the above limitations of this work, which are common among previous attempts to correlate habitat change to salmon distribution (e.g. Bradford & Irvine, 2000), these types of studies do help focus on those factors that require further investigation, especially those that have potential to be remediated or selected for protection.

This work provides empirical support to help prioritize which constellation of habitat metrics are best suited for conservation efforts to help vulnerable juvenile salmonids. Further disentangling the relative importance of each variable would be difficult, but addressing the sources for sediment input and improving riparian cover are tractable solutions for future conservation efforts. Results indicate that small streams are particularly important to trout at a juvenile life stage. Focused restoration effort on smaller systems could prove to be a useful mitigation option. The positive association

between cobble and the salmonid species investigated provides another mitigation opportunity. Influencing substrate type to have a larger proportion of cobble present could help buffer the negative effects of rising water temperatures by providing increased food resources and predator protection within a system (Lusardi et al., 2020). However, solely using cobble percentages to bolster available invertebrate prey levels should be cautioned against to avoid unintentional consequences such as less efficient growth and higher stress later in the season (Spanjer et al., 2018). Instead, these efforts should be used in conjunction with other mitigation efforts to bring temperatures down. Urban systems are experiencing compounding effects from climate change and a myriad of habitat alterations, and focusing on a single variable to reverse current salmon declines does not account for the complex ways that juvenile salmon in urban environments are responding to habitat change (Hale et al., 2016; Kemp et al., 2011; Paul & Meyer, 2001; Walsh et al., 2005; Wang et al., 2000, 2001). A multi-faceted approach is necessary and although complicated, stream health as a whole stands to benefit.

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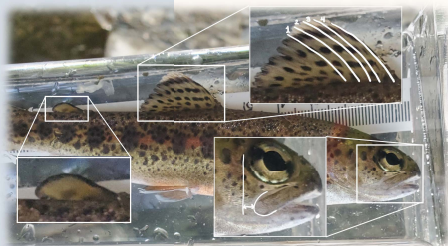
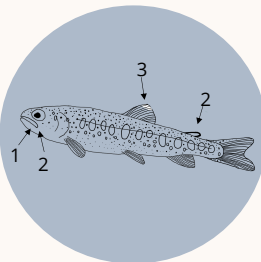
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## **Appendix. Supplemental Tables and Figures**

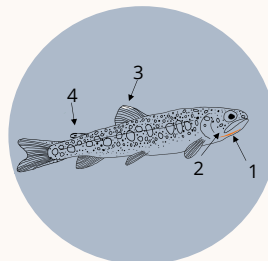
## RAINBOW TROUT (*Oncorhynchus mykiss*)

1. No red slash on jaw
2. Maxillary does not extend past rear eye margin
3. 3-5 white interspaces between fin rays
4. Adipose fin has 0 or 1 break



## CUTTHROAT TROUT (*Oncorhynchus clarkii*)

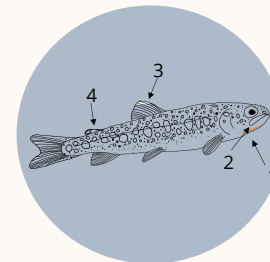
1. Red slash on jaw
2. Maxillary extends past rear eye margin
3. 1-3 white interspaces between fin rays
4. Adipose fin has 1+ breaks



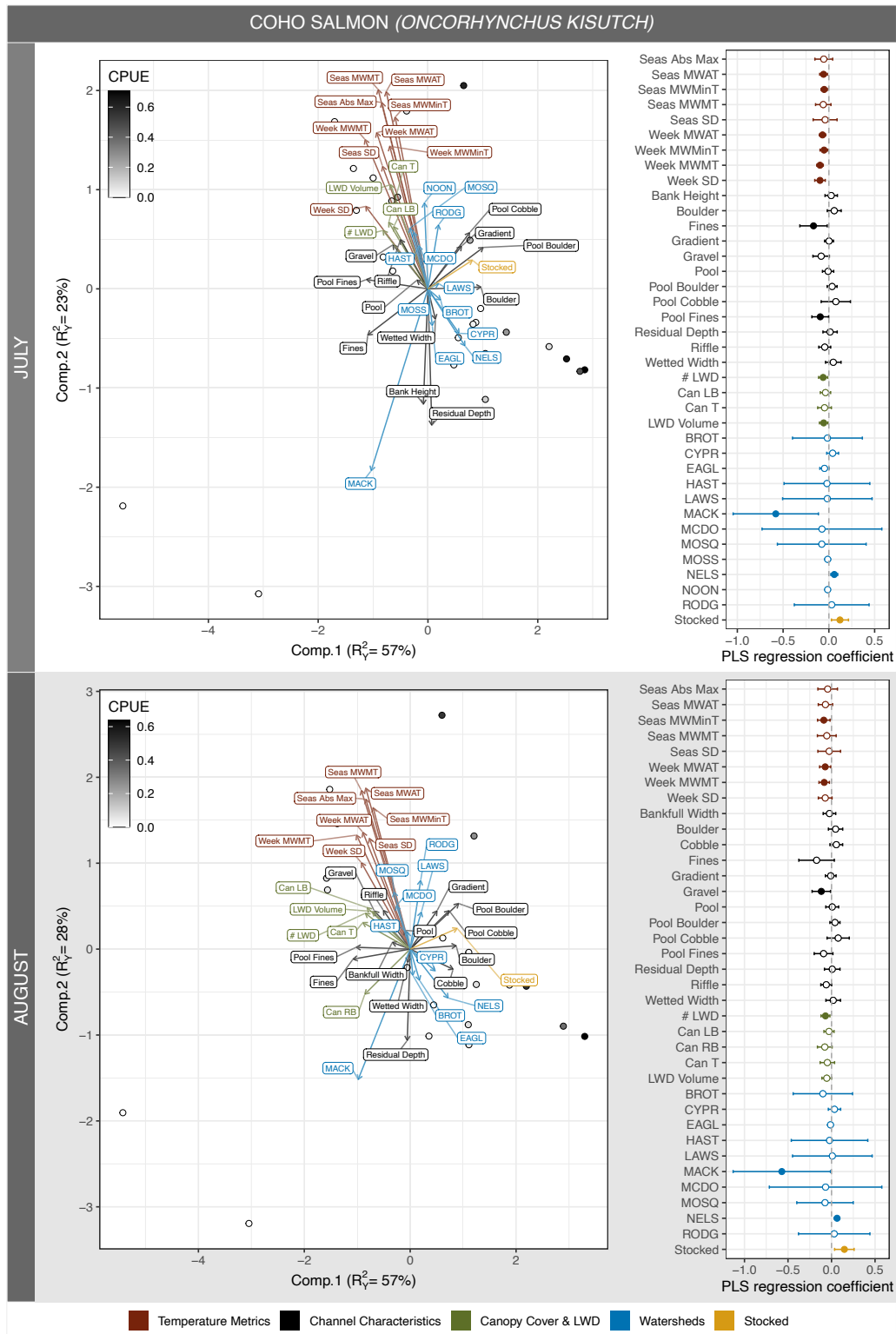
## RAINBOW/ CUTTHROAT HYBRID

### Any combination of rainbow and cutthroat traits

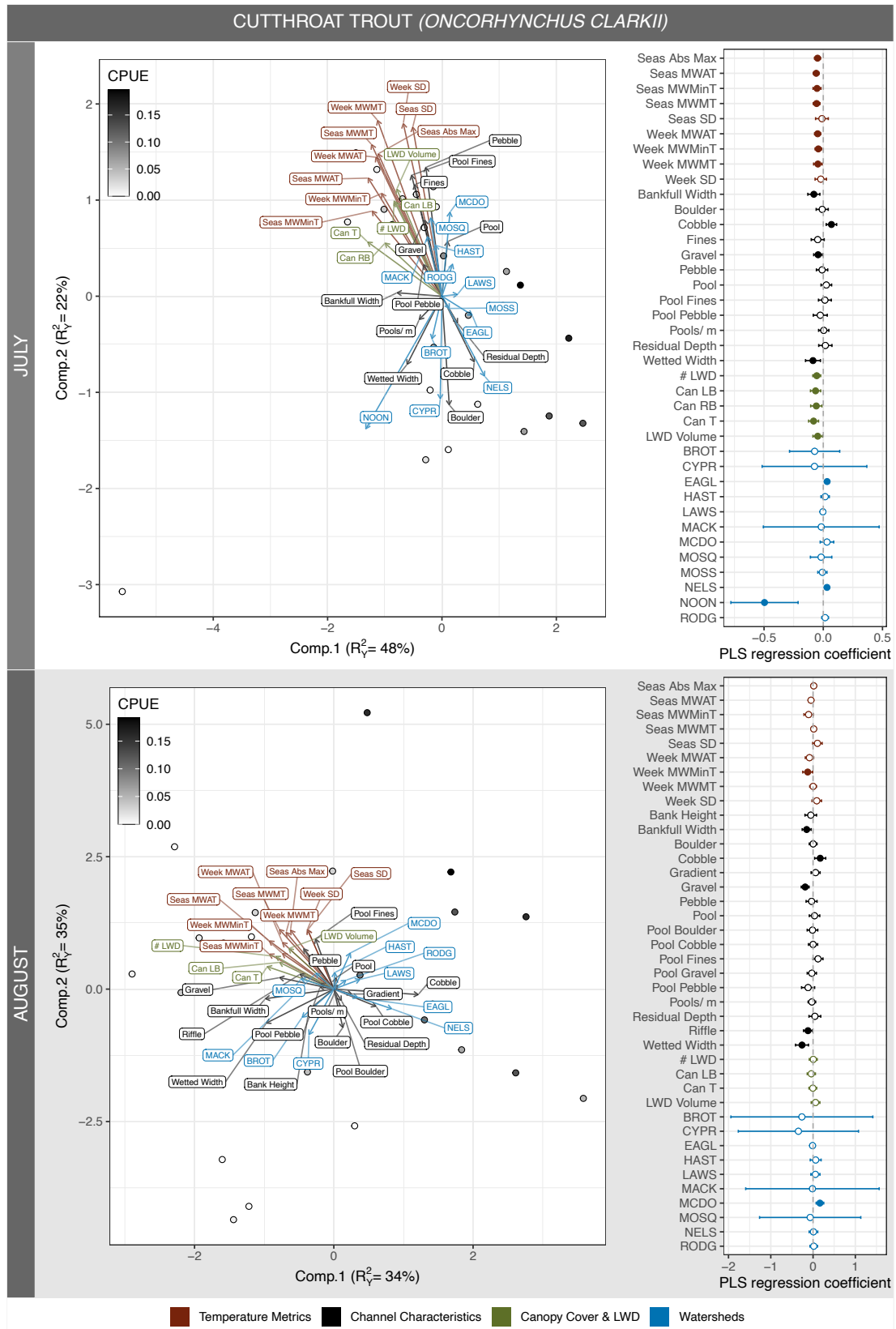
1. Red slash on jaw (cutthroat trait)
2. Maxillary does not extend past rear eye margin (rainbow trait)
3. 3 white interspaces between fin rays (shared trait)
4. Adipose fin has 1 break (shared trait)

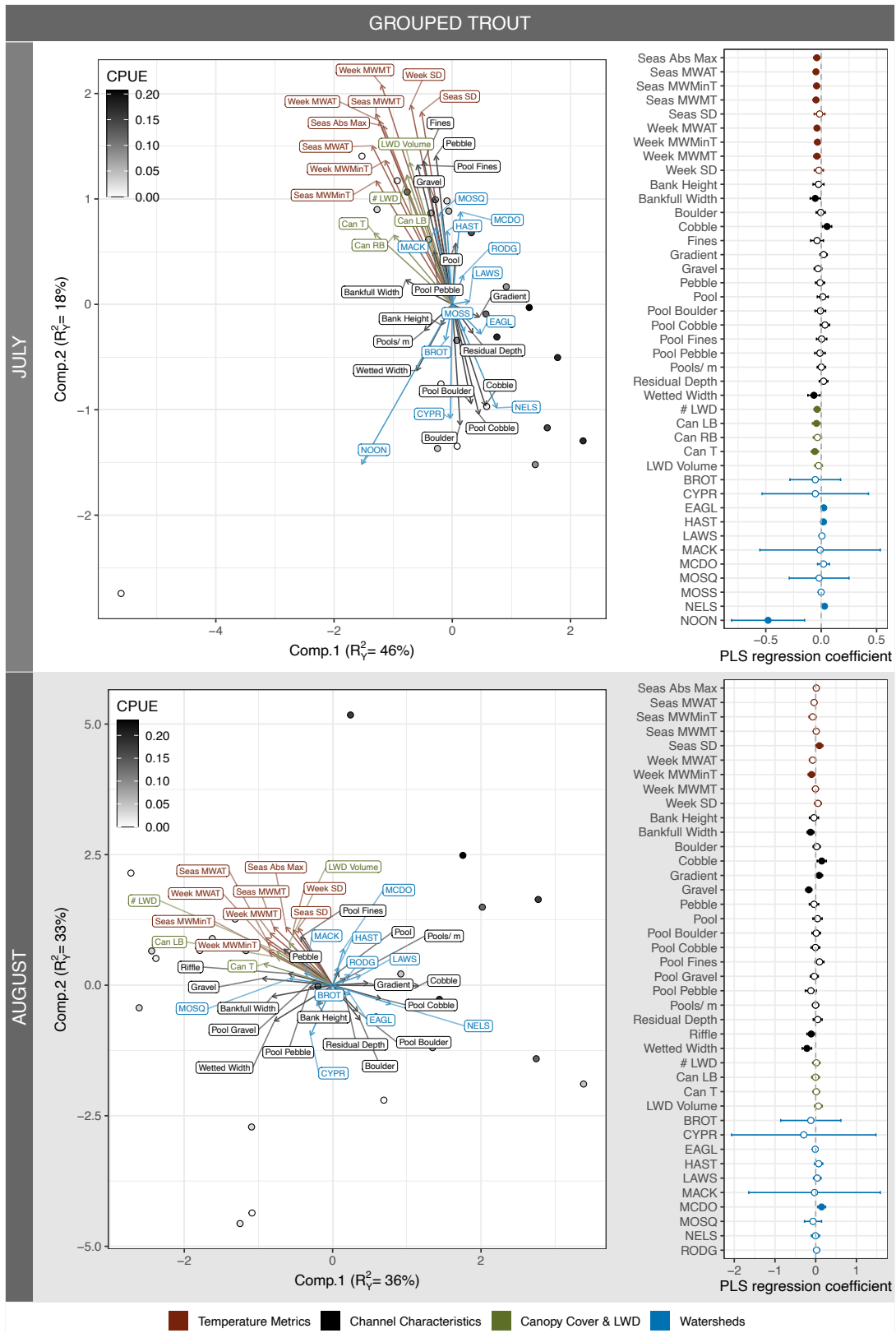


**Figure A.1. Trout identification based on key traits. Information from Watershed & Environmental Management, personal communications, July 2, 2021.**



**Figure A.2. PLSR results showing the full extent of data for coho salmon in July and August.**





**Table A.1.     Stocking Data by Watershed.**

<b>Watershed</b>	<b>Stocking Date</b>	<b># of Fish Stocked</b>	<b>Life Stage</b>	<b>Weight</b>	<b>Clipped?</b>
<i>Nelson Creek</i>	2021-06-30	8,142	fry	2.04 g	N
<i>Eagle Creek</i>	2021-06-30	1,401	fry	2.04 g	N
<i>Cypress Creek</i>	2021-06-30	6,147	fry	2.04 g	N
<i>Rodgers Creek</i>	2021-06-30	2,970	fry	2.04 g	N
<i>Lawson Creek</i>	2021-06-30	3,170	fry	2.04 g	N
<i>Mossom Creek</i>	2021-06-02	3,750	smolt	23.4 g	N
<i>Noons Creek</i>	2021-06-25 & 27	13,465	smolt	21.6 g	N



**Table A.2. eDNA hit rate by site and species. Triplicate qPCR results were ran from the field duplicate samples totalling 6 possible hits. A positive hit was established when a qPCR run returned a C<sub>t</sub> score.**

Site	Hit Rate Coho	Hit Rate Cutthroat	Hit Rate Rainbow
BROT01	6	6	6
BROT02	6	6	6
BROT03	0	1	0
BROT04	1	6	4
CYPR01	4	3	3
CYPR05	6	6	6
EAGL02	6	6	6
EAGL03	6	6	0
HADD02	0	6	6
HADD01	6	6	6
HAST01	6	6	6
HAST03	0	6	1
LAWS01	6	6	6
LAWS02	1	6	1
LYNN01	6	6	6
LYNN03	6	6	6
MACK04	6	6	6
MACK05	6	6	6
MCDO01	6	6	6
MCDO05	0	6	6
MCDO06	0	6	1
MCDO07	0	6	1
MOSQ01	6	6	6
MOSQ07	6	6	3
MOSS01	6	6	3
MOSS04	6	6	1
NELS01	6	6	5
NELS02	6	6	0
NOON01	6	6	5
NOON02	0	6	0
NOON03	0	6	0
RODG01	5	6	3
RODG02	1	6	6
THAI01	0	6	6
THAI02	6	6	0
WAGG02	1	6	5

**Table A.3. Seasonal temperature metrics and the week they occurred in 2021. Week 26 occurring from June 28<sup>th</sup> to July 4<sup>th</sup> and the heat dome occurred from June 25<sup>th</sup> to July 1<sup>st</sup>.**

Site	Seasonal MWMT	Week Occurred	Seasonal MWAT	Week Occurred
BROT01	17.92	26	17.33	26
BROT02	18.17	26	17.29	26
BROT04	18.08	26	17.49	31
CYPR01	17.83	31	17.09	33
CYPR05	17.14	31	16.74	33
EAGL02	17.76	31	17.14	33
HADD02	21.25	26	20.40	26
HAST01	20.24	26	19.12	26
HAST03	18.55	26	17.84	31
LAWS01	19.69	26	18.73	26
LAWS02	18.87	26	17.98	26
MACK04	18.47	26	17.12	31
MACK05	19.98	26	18.36	26
MCDO01	20.49	26	19.18	26
MCDO04	21.11	30	18.65	31
MCDO07	18.41	26	17.40	26
MOSQ01	20.08	31	18.72	31
MOSQ07	18.29	31	17.22	31
NELS01	16.83	33	16.37	33
NELS02	17.34	26	16.81	26
RODG02	18.56	26	17.87	26
THAI01	19.63	31	18.52	33
THAI02	19.23	31	18.58	31
WAGG02	19.33	33	18.57	33

Table A.4. Mean channel characterstics by site.

Site	Watershed	Bankfull Width (m)	Wetted Width (m)	Bank Height (m)	Reach Length (m)	Extended Reach Length (m)	Site Area (m²)	Gradient (%)	Cascade Area (%)	Glide Area (%)	Run Area (%)	Step Area (%)	Human Made Area (%)	Riffle Area (%)	Pool Area (%)	Pools/ m	Residual Depth (m)	Undercut Left Bank	Undercut Right Bank	Total Cutbank
BROT01	BROT	7.8	5.6	0.9	80	123.6	448	3*	15*	0*	0*	0*	0*	44*	40*	0.08*	0.30*	0.0	0.0	0.0
BROT02	BROT	7.8	5.6	0.9	76		422.56	4	8	0	0	0	0	64	28	0.08	0.31	0.0	0.0	0.0
BROT03	BROT	2.1	1.9	0.6	28		53.2	5	29	0	0	0	0	16	55	0.14	0.35	0.0	0.0	0.0
BROT04	BROT	8.9	5.5	0.7	48		263.04	9	40	0	0	0	0	0	60	0.08	0.82	0.0	0.0	0.0
CYPR01	CYPR	8.2	6.4	0.8	80	98.4	512	4*	13*	0*	0*	0*	0*	75*	9*	0.04*	0.32*	0.0	0.0	0.0
CYPR05	CYPR	9.7	8.6	0.9	88		758.56	3	79	0	0	0	0	0	21	0.02	0.56	1.0	0.0	0.5
EAGL02	EAGL	2.5	1.7	0.5	24		40.992	4	0	0	0	0	0	49	50	0.13	0.20	0.0	0.0	0.0
EAGL03	EAGL	2.9	2.3	0.4	24	39.4	56.16	4*	27*	0*	0*	0*	0*	55*	19*	0.08*	0.15*	0.0	1.4	0.7
HADD01	BROT	5.6	3.2	0.8	40		278.72	3	0	0	0	0	0	37	63	0.13	0.31	0.0	2.3	1.1
HADD02	BROT	6.4	5.4	0.6	52	104	129.76	5*	59*	0*	0*	0*	0*	20*	22*	0.05*	0.36*	0.0	0.0	0.0
HAST01	HAST	12.5	6.0	0.8	76		452.96	7	22	0	0	4	0	31	42	0.16	0.33	0.0	0.0	0.0
HAST03	HAST	6.1	2.9	0.9	48	78.2	141.12	3*	0*	0*	0*	0*	0*	77*	23*	0.06*	0.13*	0.3	0.1	0.2
LAWS01	LAWS	3.6	2.9	0.5	44		125.4	1	18	0	0	0	0	4	77	0.09	0.24	0.0	0.0	0.0
LAWS02	LAWS	5.7	3.2	0.6	44	72.6	140.8	6*	79*	0*	0*	0*	0*	0*	21*	0.07*	0.47*	0.0	0.4	0.2
MACK04	MACK	4.2	3.2	0.8	48		151.68	1	0	0	49	0	0	37	14	0.02	0.57	0.0	0.5	0.3
MACK05	MACK	10.2	6.2	1.0	88		547.36	0	0	3	7	0	0	21	69	0.05	0.54	2.5	1.1	1.8
MCDO01	MCDO	5.5	2.8	0.5	60	85.7	170.4	5*	0*	0*	0*	0*	21*	30*	10*	0.02*	0.32*	0.3	0.0	0.1
MCDO04	MCDO	6.4	3.3	0.6	40	55.7	133.6	3*	12*	0*	0*	0*	0*	41*	48*	0.05*	0.24*	1.5	0.0	0.8
MCDO06	MCDO	2.4	2.1	0.5	20	55.8	41.2	6*	18*	0*	0*	0*	43*	35*	5*	0.02*	0.31*	0.5	0.0	0.2
MCDO07	MCDO	4.1	3.1	0.6	24	75.9	73.44	6*	51*	0*	0*	0*	25*	10*	14*	0.01*	0.83*	0.0	0.0	0.0
MOSQ01	MOSQ	9.4	5.7	0.6	64	133.1	362.24	1*	0*	8*	0*	1*	0*	53*	39*	0.03*	0.39*	0.7	2.8	1.8
MOSQ07	MOSQ	7.5	5.1	0.7	60	65.7	306	2*	15*	21*	0*	1*	0*	15*	48*	0.05*	0.46*	0.0	0.0	0.0
MOSS01	MOSS	6.2	5.5	0.5	48	90.1	263.04	3*	0*	7*	5*	0*	22*	49*	17*	0.03*	0.43*	0.0	0.7	0.3
MOSS04	MOSS	4.3	3.2	0.5	44		139.92	2	59	10	0	0	0	7	24	0.05	0.26	0.0	0.0	0.0
NELS01	NELS	4.2	2.9	0.6	40	100.5	116.96	3*	0*	0*	8*	1*	57*	14*	20*	0.02*	0.55*	0.0	0.0	0.0
NELS02	NELS	4.7	3.5	0.7	44		154	3	53	11	0	0	0	0	35	0.07	0.33	0.0	0.4	0.2
NOON01	NOON	6.6	4.3	0.6	36	40.6	156.24	2*	39*	19*	0*	0*	0*	17*	25*	0.15*	0.22*	0.0	0.0	0.0
NOON02	NOON	4.9	4.0	0.5	32		129.28	6	41	0	0	0	0	0	59	0.16	0.44	0.0	0.0	0.0
RODG01	RODG	5.6	4.2	0.7	36		151.92	5	0	0	0	0	0	55	45	0.17	0.28	0.0	0.0	0.0
RODG02	RODG	4.8	2.6	0.7	44		116.16	6	27	0	0	0	1	18	54	0.16	0.22	0.0	0.0	0.0
THAI01	MOSQ	4.0	2.3	0.3	28	43.9	64.512	2*	0*	6*	0*	0*	0*	82*	12*	0.07*	0.24*	0.0	0.7	0.4
THAI02	MOSQ	7.7	4.8	0.6	40		193.6	1	6	0	0	0	0	54	40	0.08	0.41	0.3	0.0	0.2
WAGG02	MOSQ	5.9	3.9	0.7	48		185.28	3	45	14	0	0	0	38	3	0.02	0.41	0.0	0.0	0.0

\*When reach length did not include all trapping locations, measurements for these variables were extended to include all trapping locations.

Table A.5. Mean seasonal temperature metrics, canopy openness, and large woody debris by site.

Site	Watershed	Mean Seasonal Temperature Metrics					Mean Canopy Openness and Large Woody Debris				
		MWMinT (°C)	MWMT (°C)	MWAT (°C)	SD (°C)	Absolute Max (°C)	Canopy Right Bank	Canopy Thalweg	Canopy Left Bank	LWD Volume	LWD Count
BROT01	BROT	16.96	17.92	17.33	0.33	18.81	18	13	13	1.7E-03	6
BROT02	BROT	16.67	18.17	17.29	0.56	19.26	7	6	6	1.3E-03	7
BROT03	BROT						6	6	5	0.0E+00	0
BROT04	BROT	16.99	18.08	17.49	0.60	19.72	16	13	14	1.6E-02	9
CYPR01	CYPR	16.63	17.83	17.09	0.54	18.71	9	14	2	1.6E-03	2
CYPR05	CYPR	16.38	17.14	16.74	0.57	18.19	16	20	13	9.2E-05	1
EAGL02	EAGL	16.64	17.76	17.14	0.59	19.85	3	4	4	0.0E+00	0
EAGL03	EAGL						7	8	7	0.0E+00	0
HADD01	BROT						10	12	8	1.9E-02	10
HADD02	BROT	19.68	21.25	20.40	0.67	22.26	17	28	18	1.7E-02	11
HAST01	HAST	18.22	20.24	19.12	0.77	21.49	10	11	7	3.1E-02	24
HAST03	HAST	17.38	18.55	17.84	0.60	19.47	10	12	11	1.1E-01	29
LAWS01	LAWS	17.97	19.69	18.73	0.60	21.19	4	6	9	0.0E+00	0
LAWS02	LAWS	17.46	18.87	17.98	0.62	21.15	15	13	10	2.1E-02	6
MACK04	MACK	16.53	18.47	17.12	0.75	19.75	36	21	16	9.4E-04	3
MACK05	MACK	17.34	19.98	18.36	1.00	21.38	18	24	16	1.1E-02	21
MCDO01	MCDO	18.02	20.49	19.18	1.03	22.01	3	4	3	2.3E-02	8
MCDO04	MCDO	17.08	21.11	18.65	1.72	22.33	19	21	12	3.8E-02	22
MCDO06	MCDO						1	2	2	1.4E-03	3
MCDO07	MCDO	16.80	18.41	17.40	0.59	19.73	13	12	10	9.7E-04	4
MOSQ01	MOSQ	17.64	20.08	18.72	0.92	20.90	20	22	18	9.4E-04	4
MOSQ07	MOSQ	16.66	18.29	17.22	0.72	19.04	9	10	9	0.0E+00	0
MOSS01	MOSS	14.01	16.31	14.95	0.78	17.29	9	10	10	1.9E-02	17
MOSS04	MOSS	17.18	19.49	18.13	0.86	20.89	35	38	35	4.9E-02	14
NELS01	NELS	15.97	16.83	16.37	0.39	17.50	7	7	7	0.0E+00	0
NELS02	NELS	16.43	17.34	16.81	0.32	18.14	6	6	5	1.9E-03	3
NOON01	NOON	19.13	21.15	20.03	0.69	22.65	24	29	14	6.2E-03	3
NOON02	NOON						21	22	18	8.9E-04	2
RODG01	RODG	17.64	21.44	18.98	1.29	23.85	13	16	16	1.1E-02	5
RODG02	RODG	17.38	18.56	17.87	0.48	19.60	9	9	9	0.0E+00	0
THAI01	MOSQ	17.69	19.63	18.52	0.86	20.59	20	16	14	1.1E-02	5
THAI02	MOSQ	17.91	19.23	18.58	0.57	20.29	14	14	15	5.8E-02	18
WAGG02	MOSQ	17.97	19.33	18.57	0.60	20.33	14	17	18	5.4E-03	9

**Table A.6. Mean percent sediment type in the reach and in pools by site.**

Site	Watershed	Boulder (%)	Cobble (%)	Pebble (%)	Gravel (%)	Fines (%)	Pool Boulder (%)	Pool Cobble (%)	Pool Pebble (%)	Pool Gravel (%)	Pool Fines (%)
BROT01	BROT	59	12	19	6	4	43	23	24	6	4
BROT02	BROT	34	38	24	3	1	14	35	36	9	6
BROT03	BROT	39	21	25	10	5	41	13	20	15	11
BROT04	BROT	60	18	18	3	1	48	21	24	6	1
CYPR01	CYPR	68	16	9	5	2	61	21	13	4	1
CYPR05	CYPR	68	16	11	3	2	37	13	34	8	8
EAGL02	EAGL	20	32	37	10	1	17	20	32	22	10
EAGL03	EAGL	15	46	26	8	5	10	25	20	12	33
HADD01	BROT	14	33	28	6	18	13	21	19	11	36
HADD02	BROT	42	19	22	7	11	23	23	26	13	16
HAST01	HAST	63	9	19	6	3	32	17	34	8	9
HAST03	HAST	8	43	34	4	12	10	21	34	6	29
LAWS01	LAWS	37	32	20	2	8	33	28	26	6	8
LAWS02	LAWS	58	25	10	4	3	52	22	20	4	2
MACK04	MACK	23	25	41	6	6	5	20	55	10	10
MACK05	MACK	0	0	43	4	53	0	0	31	8	62
MCDO01	MCDO	33	12	33	18	5	13	5	53	18	13
MCDO04	MCDO	25	23	46	5	1	23	17	17	7	37
MCDO06	MCDO	32	13	42	6	8	20	10	10	30	30
MCDO07	MCDO	52	29	13	3	3	5	30	20	15	30
MOSQ01	MOSQ	7	22	52	6	14	13	33	32	4	18
MOSQ07	MOSQ	46	36	13	3	2	57	22	12	3	7
MOSS01	MOSS	9	31	43	8	9	15	33	18	15	18
MOSS04	MOSS	18	47	27	5	3	20	20	30	15	15
NELS01	NELS	28	45	24	3	0	25	35	30	5	5
NELS02	NELS	54	25	19	2	0	47	23	18	7	5
NOON01	NOON	38	33	24	4	2	18	27	33	6	17
NOON02	NOON	53	26	18	2	2	54	22	17	3	4
RODG01	RODG	47	24	25	3	1	27	38	27	6	3
RODG02	RODG	44	23	17	2	14	25	29	34	6	6
THAI01	MOSQ	18	14	52	9	7	20	20	42	8	10
THAI02	MOSQ	3	19	42	19	17	3	31	27	20	19
WAGG02	MOSQ	46	24	21	6	3	75	5	5	5	10

Table A.7. Mean week prior temperature metrics and CPUE by month and site.

Site	Watershed	JULY							AUGUST						
		Mean Week Prior Temperature Metric				Mean Catch Per Unit Effort (CPUE) x 100			Mean Week Prior Temperature Metric				Mean Catch Per Unit Effort (CPUE) x 100		
		MWMinT (°C)	MWMT (°C)	MWAT (°C)	SD (°C)	Coho Salmon	Cutthroat Trout	Grouped Trout	MWMinT (°C)	MWMT (°C)	MWAT (°C)	SD (°C)	Coho Salmon	Cutthroat Trout	Grouped Trout
BROT01	BROT	15.60	16.29	15.88	0.24	13.77	0.00	1.48	15.46	16.02	15.69	0.19	2.07	0.00	1.55
BROT02	BROT	14.52	15.46	14.93	0.31	26.60	1.02	0.77	14.64	15.53	15.02	0.28	4.16	0.00	0.78
BROT03	BROT					0.00	0.00	0.00					0.00	0.00	0.00
BROT04	BROT	15.12	16.52	15.76	0.49	0.00	5.29	2.11	15.11	15.96	15.49	0.28	0.00	4.44	2.22
CYPR01	CYPR	14.63	15.77	15.10	0.38	7.79	0.00	0.73	14.86	15.99	15.29	0.35	3.95	0.00	0.25
CYPR05	CYPR	14.43	15.28	14.80	0.26	66.93	1.94	2.18	14.67	15.31	14.98	0.20	50.18	1.00	1.25
EAGL02	EAGL	14.94	16.14	15.50	0.42	2.05	17.44	4.36	15.08	16.17	15.58	0.36	6.84	5.86	1.71
EAGL03	EAGL					54.00	4.00	8.50					8.48	0.94	5.65
HADD01	BROT					45.00	0.00	2.19					12.49	0.00	0.78
HADD02	BROT	16.08	17.43	16.76	0.46	0.00	1.02	0.77	15.55	16.47	15.98	0.30	0.00	1.04	0.78
HAST01	HAST	16.31	18.21	17.22	0.67	13.95	3.72	2.33	15.28	16.35	15.80	0.35	13.09	2.18	1.09
HAST03	HAST	16.01	17.11	16.52	0.39	0.00	12.00	3.46	14.89	15.82	15.30	0.29	0.00	19.18	5.82
LAWS01	LAWS	15.89	17.24	16.48	0.43	18.25	6.44	1.88	16.06	17.06	16.53	0.31	26.07	3.60	1.12
LAWS02	LAWS	15.26	16.22	15.68	0.29	0.00	5.30	2.91	14.68	16.24	15.45	0.51	0.00	12.16	3.26
LYNN01	LYNN					4.58	0.00	0.00							
LYNN03	LYNN					12.73	0.00	0.45							
MACK04	MACK	14.86	17.05	15.68	0.70	0.00	7.98	2.99	15.08	16.33	15.56	0.39	0.00	9.21	2.56
MACK05	MACK	15.60	18.39	16.72	0.95	0.00	0.00	0.00	15.60	17.38	16.28	0.56	0.00	0.00	0.00
MCDO01	MCDO	15.97	18.14	17.02	0.74	7.38	1.05	1.05	16.07	17.48	16.76	0.49	6.28	2.69	0.90
MCDO04	MCDO	15.13	19.01	16.51	1.22	0.00	12.78	3.46	15.69	17.79	16.44	0.65	0.00	15.79	4.39
MCDO06	MCDO					0.00	10.35	2.87					0.00	4.96	1.65
MCDO07	MCDO	14.76	15.90	15.25	0.37	0.00	19.64	5.18	15.14	16.07	15.54	0.29	0.00	15.60	4.33
MOSQ01	MOSQ	16.41	19.09	17.59	0.88	0.00	0.96	0.24	15.84	17.46	16.62	0.51	5.64	1.13	0.28
MOSQ07	MOSQ	15.10	17.16	15.90	0.71	3.31	17.66	4.42	15.02	16.40	15.50	0.43	6.60	16.51	4.68
MOSS01	MOSS	12.18	14.40	13.06	0.71	11.78	6.73	2.10							
MOSS04	MOSS	14.96	16.80	15.82	0.65	5.23	2.09	1.31							
NELS01	NELS	14.48	15.37	14.94	0.29	70.77	12.31	4.36	14.90	15.65	15.28	0.25	63.61	4.82	1.20
NELS02	NELS	14.67	15.39	14.97	0.23	45.57	12.15	3.54	15.07	15.61	15.33	0.17	34.42	12.43	3.35
NOON01	NOON	16.62	18.31	17.46	0.57	6.71	0.00	0.00							
NOON02	NOON					0.00	3.36	0.84							
RODG01	RODG	15.09	17.26	15.96	0.70	60.91	5.08	1.27	15.13	16.72	15.79	0.51	49.38	2.96	1.48
RODG02	RODG	15.45	16.56	15.96	0.37	0.00	11.99	3.50	15.39	16.26	15.80	0.29	0.00	9.79	3.18
THAI01	MOSQ	16.07	18.46	17.17	0.86	0.00	1.30	0.98	15.89	17.42	16.57	0.51	0.00	0.00	0.28
THAI02	MOSQ	16.28	17.83	17.07	0.56	15.63	5.58	2.51	15.93	17.06	16.44	0.36	4.38	3.29	0.82
WAGG02	MOSQ	16.45	18.15	17.21	0.60	0.00	0.00	0.00	16.34	17.55	16.87	0.40	0.00	0.00	0.55

**Table A.8. Ranked top five coefficient absolute values for Coho salmon by month.**

COHO				
Rank	July		August	
	Metric	Value	Metric	Value
1	Fines	-0.1655	Fines	-0.1716
2	Stocked	0.1187	Stocked	0.1477
3	Week SD	-0.0955	Gravel	-0.1174
4	Week MWMt	-0.0954	Pool Fines	-0.0916
5	Pool Fines	-0.0927	Seas MWMinT	-0.0887

**Table A.9. Ranked top five coefficient absolute values for Cutthroat trout by month.**

CUTTHROAT TROUT				
Rank	July		August	
	Metric	Value	Metric	Value
1	Wetted Width	-0.0870	Wetted Width	-0.2615
2	Can T	-0.0820	Gravel	-0.1865
3	Bankfull Width	-0.0796	Cobble	0.1687
4	Cobble	0.0687	Bankfull Width	-0.1511
5	Can LB	-0.0651	Week MWMinT	-0.1268

**Table A.10. Ranked top five coefficient absolute values for grouped trout by month.**

GROUPED TROUT				
Rank	July		August	
	Metric	Value	Metric	Value
1	Wetted Width	-0.0658	Wetted Width	-0.2161
2	Can T	-0.0575	Gravel	-0.1697
3	Bankfull Width	-0.0538	Cobble	0.1485
4	Cobble	0.0527	Bankfull Width	-0.1270
5	Seas MWAT	-0.0475	Pool Pebble	-0.1190