

ASSESSING PHYSICAL HABITAT AND JUVENILE SALMONID POPULATIONS IN  
SMALL COASTAL STREAMS IN SUMMER USING TRADITIONAL SAMPLING AND  
ADVANCED REMOTE SENSING APPROACHES

by

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B.Sc. (Honours), The University of British Columbia, 2019

A THESIS SUBMITTED IN PARTIAL FULFILLMENT OF  
THE REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

in

THE FACULTY OF GRADUATE AND POSTDOCTORAL STUDIES

(Forestry)

THE UNIVERSITY OF BRITISH COLUMBIA

(Vancouver)

January 2022

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SMALL COASTAL STREAMS IN SUMMER USING TRADITIONAL SAMPLING AND  
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submitted by Alyssa D. Nonis in partial fulfillment of the requirements for

the degree of Master of Science

in Forestry

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## Abstract

Freshwater habitat loss is a major threat to the persistence of salmonid (*Oncorhynchus* spp.) populations in the Pacific Northwest of North America. To better inform where and when to preserve this critical habitat, information on its extent and quality is required over spatial scales fish use for various life stages. Further, current and accurate habitat associations are required to make location-specific conservation recommendations. Given the limitations of fine-scale analysis due to sampling logistics and that land conversion activities and climate change affect stream salmonids on extremely large spatial scales, approaches to extend small-scale fish-habitat assessments accurately and cost-effectively to the watershed level are needed. To address this challenge, I investigated the habitat associations of juvenile rainbow trout (*Oncorhynchus mykiss*) and Dolly Varden char (*Salvelinus malma*) in the Nahmint watershed on Vancouver Island, British Columbia, during summer 2019. I also assessed whether 3-D Airborne laser scanning (ALS) derived measurements of small stream characteristics provided similar explanatory power in establishing habitat associations with physical characteristics as conventional field-based surveys. I surveyed 200 m of habitat across varying tributary stream sizes (5.1 m to 10.6 m) and implemented mark-recapture fish sampling using minnow traps and passive integrated transponder tags to establish habitat associations and characterize movement. The density of both species was significantly higher in pool habitats than in glides and riffles. Generally, restricted movement was observed by both species, with some Dolly Varden char moving in a primarily downstream direction. Further, ALS-measured and field-measured variables produced similar habitat associations, with fish presence and density associated with pool habitats classified by the ALS data. Overall, these results highlight the importance of habitat quality (such as abundant pools) and connectivity in small streams, while also illustrating the

utility of using ALS -derived measures of small stream habitat to establish fine-scale habitat associations of juvenile salmonids within a watershed. Moreover, ALS has the potential to be used as a predictive tool to assess juvenile salmonid distributions at a broad scale in a timely and cost-effective way.

## Lay Summary

Traditional approaches for quantifying habitat quality and characteristics for stream-dwelling salmonids can be time-consuming and challenging to obtain on large spatial scales. The potential to use 3-D airplane-derived ALS (Airborne Laser Scanning) data to characterize stream habitats could enable larger scale and more rapid planning for land-use decision-making. To investigate this, I surveyed juvenile salmonids (rainbow trout; *Oncorhynchus mykiss*; Dolly Varden char; *Salvelinus malma*) in five streams on Vancouver Island and developed relationships between stream characteristics and fish density. I then assessed whether ALS-derived measurements of small stream characteristics could establish habitat associations similarly to conventional field-based surveys. I found that ALS-measured and field-measured variables produced similar habitat associations. Overall, this research illustrates the utility of using ALS - derived fine-scale attributes to understand habitat associations of juvenile salmonids. Moreover, ALS has the potential to be employed as a tool to assess juvenile salmonid distributions in a timely and cost-effective way.

## **Preface**

I held primary responsibility for the study design, the collection of the biological data, the analysis, writing and the preparation of this manuscript. I received valuable supervision and guidance from my supervisor, Dr. Scott G. Hinch, and from my supervisory committee member, Dr. Nicholas C. Coops. British Columbia Timber Sales provided ALS acquisition data. Spencer Dakin Kuiper (IRSS) provided ALS-derived stream attribute data.

Chapter 2 has been submitted for publication (Authors: Alyssa D. Nonis, Scott G. Hinch, Nicholas C. Coops) and is currently under revision. With help from my co-authors, I developed and applied scientific methods to collect, analyze, synthesize, write, and prepare the manuscript for publication. Co-authors provided critical feedback and ideas for the manuscript.

Chapter 3 has been submitted for publication (Authors: Alyssa Nonis, Spencer Dakin Kuiper, Scott G. Hinch, Nicholas C Coops, Piotr Tompalski). With help from my co-authors, I developed and applied scientific methods to collect, analyze, synthesize, write, and prepare the manuscript for publication. Spencer Dakin Kuiper assisted by providing ALS data and contributed to writing. Co-authors provided critical feedback and ideas for the manuscript.

Fieldwork was led by myself with assistance from Andrew Lotto, Katie Zinn, Brian Hendriks, and Hunter Rigatti. All tagging and handling procedures were approved by the University of British Columbia Animal Ethics Committee (animal care permit: A190193). Another permit obtained for this research was Provincial fish collection permit (NA19-475941).

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## **Acknowledgements**

I would like to thank my supervisor, Dr. Scott G. Hinch, for the opportunity to be a part of his research group. Thank you for your unwavering support, mentorship, and encouragement over the past several years. Additionally, thank you Nicholas C. Coops for your support and mentorship, and for the opportunity to collaborate with the Integrated Remote Sensing Studio. The unparalleled dedication you both have to your students creates incredible research programs and opportunities for your students.

This research was supported by Natural Sciences and Engineering Research Council of Canada (NSERC, STPGP 517589) grant to Nicholas Coops and Scott Hinch and conducted in collaboration with British Columbia Timber Sales, the Canadian Forest Service, Pacific Forestry Centre, and the Integrated Remote Sensing Studio (IRSS; UBC). I also thank the Freshwater Fisheries Society of B.C. for a scholarship. I extended my gratitude to the Nuu-chah-nulth First Nations for allowing me to conduct my research on their traditional territory.

I was fortunate to be surrounded by many wonderful people from the PSEC and IRSS. To the PSEC lab friends and colleagues who helped in the field and office, I sincerely thank you: Andrew Lotto, Katie Zinn, Brian Hendricks, Hunter Rigatti, Dr. Arthur Bass, Adam Kanigan, Anna Smith, David Moulton, Steve Johnson, and Noah Kussin, thank you for your support and the for the fond memories that we have created over the past several years. A special thank you to Andrew Lotto for the effortless happiness he provided over two field seasons. I thank Spencer Dakin Kuiper, Chris Colton, Piotr Tompalski, and Shangrong Lin from the IRSS for field assistance that contributed to the ALS-data.

I thank my family and friends for their endless support. Thank you to my parents Andy and Irene, my brother Andrew, and my Uncle Peter for the encouragement to pursue a career in

science. To my close friends at UBC, Katie Zinn, Nat Benoit, Brian Hendricks, Liam Irwin, and Chris Colton, I could have not done this without the support and guidance you provided. I am excited to see where each of your academic pursuits take you.

## Chapter 1: Overview of the state of Pacific Salmonids

### 1.1 Ocean habitat conditions

Adult Pacific salmon (*Oncorhynchus* spp.) have reached record high abundance levels (Ruggerone & Irvine, 2018). This record high abundance is the result of a combined production of wild and hatchery pink salmon (*Oncorhynchus gorbuscha*), chum salmon (*Oncorhynchus keta*), and sockeye salmon (*Oncorhynchus nerka*) (Ruggerone and Irvine 2018). The hatchery production of these species from central Alaska and Japan, and wild production from western Alaska and Russia (Ruggerone & Irvine, 2018) raises concerns of potential competition for ocean resources in neighbouring jurisdictions (Holt, Rutherford, & Peterman, 2008).

Although record-high abundance levels of Pacific salmon are being documented, declines are evident in many stocks and populations across the Pacific Northwest (PNW) (Northwest Fisheries Science Center, 2015; Grant, MacDonald, & Winston, 2019). Specifically, in the continental American Northwest, many Chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*Oncorhynchus kisutch*), sockeye salmon, and chum salmon evolutionary significant units are listed as threatened or endangered (Northwest Fisheries Science Center, 2015). Furthermore, in southern British Columbia, multiple sockeye, Chinook, and coho designated units are threatened (Grant et al. 2019). In contrast, at northern latitudes (British Columbia and Yukon), populations are not in decline to the same extent (Grant, MacDonald, & Winston, 2019). Further, in Alaska ecosystems are still intact with populations of pink, chum and sockeye still relatively abundant (Ruggerone & Irvine, 2018; Oke et al., 2020), exemplifying that southern latitudes are not doing as well as northern ones.

This latitudinal shift coincides with changing ocean conditions across the northeast Pacific Ocean with growing conditions for marine rearing salmon being better in northern parts of the northeast Pacific ocean (Connors et al., 2020). Some of these ocean changes are associated with anthropogenic-induced climate change which has been increasing ocean temperatures and contributing to shifts in marine habitats and species distributions (IPCC, 2019) – sea surface temperatures in the North Pacific Ocean have risen 0.1°C to 0.3°C per year from 1950 – 2009 (Poloczanska et al. 2013; Holsman et al. 2018). Marine heatwaves have become an additional threat to Pacific salmon in recent years. Anomalies of warm water have developed along the Pacific Coast from 2013 onward, altering marine animal distributions, affecting competitive relationships (Cavole et al., 2016). Warmer ocean temperatures are causing prey species from southern regions to shift northward to occupy habitats previously too cold to inhabit (Mackas, Peterson, & Zamon, 2004). This shift in prey ultimately impacts oceanic salmon growth and survival, reducing abundances at southern latitudes (Cavole et al., 2016; Holsman et al., 2018).

## **1.2 Freshwater habitat conditions**

Southern latitudes are also faced with warming air temperatures, leading to increases in freshwater temperatures and hydrological shifts (Grant, MacDonald, & Winston, 2019). Air temperature changes in British Columbia's interior and north are causing glaciers to recede, shifting hydrological regimes resulting in earlier peak flows during the spring and marginally reduced summer low flows (Pike et al., 2008, 2010a). This raises a concern for the survival of stream-dwelling salmonids and migration, as outmigrations are timed in accordance with optimal freshwater and marine feeding conditions (MacDonald et al., 2018; Grant, MacDonald, & Winston, 2019). Further, British Columbia's coastal watersheds, typically rain-dominated systems, are also experiencing alterations in the magnitude and frequency of extreme conditions.



Seasonal droughts are becoming more common and contribute to a suite of problems for the persistence of stream-dwelling salmonids (Pike et al., 2010b; Quinn, 2018; Grant, MacDonald, & Winston, 2019). These droughts increase water temperatures, create migration barriers, compress habitat, and decrease the connectivity between seasonal refugia (Pike et al., 2010b; Quinn, 2018; Grant, MacDonald, & Winston, 2019; Sheldon & Richardson, 2022). Flash flood events are also increasing, likely leading to egg loss and juvenile displacement due to increased sediment transportation (Holtby & Healey, 1986; Lisle, 1989; Lapointe et al., 2000; Grant, MacDonald, & Winston, 2019). The degree to which these extreme conditions influence streams depends on stream size. Small streams may have less predictable or sufficient flow, leaving salmonids in these streams more susceptible to stochastic events caused by climate change (Cunjak, 1996; Rosenfeld, Porter, & Parkinson, 2000). Thus, species reliant on freshwater habitats are faced with many climatic-related challenges, already observed by the declining trends of many populations in southern British Columbia.

At southern latitudes, freshwater habitats are also being altered by land-use activities such as forestry, urbanization, mining, and agriculture (Slaney et al., 1996; Gregory & Bisson, 1997; Kaufmann & Hughes, 2006; Pike et al., 2010a, 2010b). These land conversion activities, like climate change, contribute to warming freshwater temperatures and changes to hydrological patterns due to the nature of deforestation and water extraction related to these activities (Pike et al., 2010a, 2010b). Although small streams are highly integrated into the landscape (Gomi, Sidle, & Richardson, 2002), and make up over 70% of freshwater habitat networks (Leopold, Wolman, & Miller, 1964), they are often not avoided during land conversion activities.

The integrity of small coastal streams is of utmost importance for the persistence of stream-dwelling salmonids. Adjacent riparian forests provide small streams with essential

contributions such as shading from radiation, stream bank stability, energy inputs (i.e., terrestrial insects), and structural complexity (i.e., in-stream functional wood) (Gregory et al., 1991; Fausch & Northcote, 1992; Richardson et al., 2005). Specifically, wood inputs contribute to structural complexity, by influencing pool habitat availability (Fausch & Northcote, 1992; Rosenfeld, Porter, & Parkinson, 2000). Many salmonids rely on pool habitats, especially during the summer low flow period (Everest & Chapman, 1972; Bisson, Sullivan, & Nielsen, 1988; Bjornn & Reiser, 1991; Rosenfeld, Porter, & Parkinson, 2000; Gonzalez et al., 2017). Therefore, alterations or removal of streamside riparian zones can have consequential impacts on aquatic ecosystems and the stream-dwelling salmonids that inhabit them (Mellina & Hinch, 2009).

To better understand how stream-dwelling salmonids use their habitats, relationships are established between measurable habitat variables to assist in predicting presence/density (Fausch, Hawkes, & Parsons, 1988; Rosenfeld, Porter, & Parkinson, 2000; Myrvold & Kennedy, 2015). Variables typically used in small-scale analysis to establish habitat associations include habitat type, water velocity and depth, physical cover, and substrate size (Hartman, 1965; Everest & Chapman, 1972; Nakano & Kaeriyama, 1995). For example, Fausch (1993) investigated the microhabitat selection of juvenile steelhead trout (*Oncorhynchus mykiss*), finding that this species favoured habitat with abundant overhead cover, structure, and velocity refuge areas. Thus, establishing links between fish abundance and small-scale variables help define species-specific habitat associations, which further provide insight on abiotic factors that contribute to distributions seen at broad scales (Magnan et al., 1994).

At broad scales, distributions of juvenile salmonids are often controlled by stream morphological characteristics such as stream gradient and bankfull channel width. These variables are often used to predict the presence of juvenile steelhead trout, cutthroat trout, and

coho salmon (McMillan et al., 2013; Ptolmey, 2013). In order to make broad-scale predictions, expansive, high-quality microhabitat and stream morphology data are required, due to the high variability of habitat associations within streams, between streams, and years, especially given rapid environmental changes (Fausch, Hawkes, & Parsons, 1988; McMillan et al., 2013).

Expanding the understanding of stream-dwelling salmonids and their stream habitats provides useful ecological information to inform rehabilitation efforts. By creating fish-habitat associations, habitat types can be used to establish carrying capacities and highlight areas of important habitat (Bond et al., 2019; Polivka & Claeson, 2020). Thus, this information can be applied to restoration efforts, by focusing on improving the components of physical habitat that increase habitat availability and quality (Polivka & Claeson, 2020). Rehabilitation efforts are often accomplished by constructing features out of in-stream wood or other materials to restore channel morphology and increase pool availability (Roni et al., 2015). For example, Roni and Quinn (2001) found that the addition of in-stream wood in small coastal streams resulted in an increase in pool availability, ultimately increasing juvenile coho densities. Thus, addressing stream habitat quality in tributary habitats is a crucial component to the recovery of many salmonid populations (Bernhardt, 2005; Walters, Copeland, & Venditti, 2013). Faced with current climate change challenges, resource managers require reliable and location-specific habitat-association information to ensure the success of restoration efforts.

### **1.3 Habitat surveys**

Typically, stream habitat assessments are conducted on small scales ranging from 50-500 m sections of stream (Fausch et al., 2002). The focus of these assessments is to quantify instream habitat characteristics, establish habitat associations, and draw inferences about larger populations throughout the system (Fausch, Hawkes, & Parsons, 1988; Fausch et al., 2002;

McMillan et al., 2013). Traditional approaches tend to focus on representative reaches rather than a comprehensive watershed census due to time and funding constraints. For example, a field survey typically involves a multi-person team walking representative streams for extensive periods in remote locations. When sampling these representative reaches within a watershed, estimates of fish abundance are usually accurate. However, it is challenging to make extrapolations to larger populations as small-scale sampling is inadequate at capturing the hierarchical and heterogeneous characteristics of stream habitat (Fausch et al., 2002). Further, broad-scale landscape approaches such as map applications and coarse resolution satellite imagery can only note watershed-scale variables in one dimension, missing critical information. Further, it is important to account for the spatial scale at which stream-dwelling salmonids carry out important life stages when considering habitat quality; however, this is often difficult to document as habitat use can range from 1 – 100 km (Fausch et al. 2002). Thus, to acquire detailed spatial data at an intermediate scale, methods such as imagery from a low flying aircraft could capture complex habitat features, ultimately distinguishing stream heterogeneity (Fausch et al., 2002).

Light Detection And Ranging (LIDAR), when flown on an aircraft, also known as airborne laser scanning (ALS), is a remote sensing technology that has demonstrated the capacity to penetrate forest canopies to create digital elevation models (DEM) from which riparian and stream attributes can be derived with high accuracy (O’Callaghan & Mark, 1984; Phinn et al., 2011; Tompalski et al., 2017). Studies have successfully derived attributes such as stream order and magnitude, gradient, channel width, and habitat type sequences (James, Watson, & Hansen, 2007; Cavalli et al., 2008; Phinn et al., 2011). Tompalski et al. (2017), using stream gradient as a proxy for fish occurrence in small streams on Vancouver Island, BC, classified the fish-bearing

potential of small streams across the landscape with an 82.9% accuracy rate. This evidence suggests that ALS has the potential to characterize and quantify critical salmonid habitat. While previous research has shown the successful application of ALS to derive physical stream features, there has been limited research on the characterization of smaller streams or the classification of specific channel units, which is needed to assess salmonid habitat conditions at an intermediate landscape-scale comprehensively. Incorporating remote sensing technology with field-acquired ecological data may provide an opportunity to allow resource managers to assess streams in a new, timely, and cost-effective way.

#### **1.4 Focal species**

Coastal rainbow trout (*O. mykiss*) is a salmonid species that heavily relies on freshwater habitat. This species historically ranged from coastal drainages of Baja California, Mexico, to Alaska, and across into Asia (MacCrimmon, 1971; Augerot & Foley, 2005). In British Columbia, Washington, and Oregon, endemic populations of coastal rainbow trout inhabit a variety of stream habitats (MacCrimmon, 1971). Rainbow trout are frequent inhabitants of large steep gradient streams, but can also be found in intermediate and small streams (Hartman & Gill, 1968; Roper, Scarnecchia, & Marr, 1994; Ptolmey, 2013). Typically, fry emerge in late spring or early summer (Quinn, 2018). Juveniles then seek rearing habitat with deep water, intermediate velocities, and cover present (Bisson, Sullivan, & Nielsen, 1988; Bjornn & Reiser, 1991; McMillan et al., 2013; Quinn, 2018). As they mature, this species can take various life-history forms, with two common distinctions: resident rainbow trout (resident) and steelhead (anadromous). Residents reside in freshwater for the entirety of their lives and may move to and from lakes or rivers to natal tributaries (Hodge et al., 2016). Steelhead spend one to three years in

freshwater, migrate to the ocean, then return to freshwater to spawn (Hartman, 1965; Quinn, 2018).

Although this species expresses high plasticity and individual variability in life history, many southern populations have experienced declines and extinctions due to anthropogenic causes (Augerot & Foley, 2005). For example, Kendall et al. (2017) observed over 80% of populations have declined across coastal British Columbia and Washington State. Furthermore, in British Columbia, many populations' status is unknown or thought to be in decline (Slaney et al., 1996). Within British Columbia, currently, two populations (Chilcotin and Thompson River) are designated as endangered under COSEWIC (COSEWIC, 2018a). No recent studies of populations in south coast BC have investigated habitat requirements of juvenile *O. mykiss* in coastal watersheds, specifically during the summer, an already vulnerable period for stream-dwelling salmonids (Harvey, Nakamoto, & White, 2006).

Dolly Varden char (*Salvelinus malma*) is a species of char that is distributed across the Pacific Ocean, ranging from Puget sound in Washington, U.S.A., to the Yukon and Northwest Territories, Canada, and Asia, consisting of four subspecies (Armstrong & Morrow, 1980). In the PNW, the southern (*S. malma lordi*) subspecies is found (Armstrong & Morrow, 1980). Like other salmonid species, Dolly Varden char can take on both anadromous and resident forms and inhabit all freshwater types (i.e., lakes, ponds, rivers, and streams) (Armstrong & Morrow, 1980). Specifically, non-anadromous Dolly Varden char occupy streams with abundant gravel and woody substrate (Armstrong & Morrow, 1980). This species is an opportunistic drift feeder in small streams (Bozeman & Grossman, 2019), that occupy benthic or deeper portions of the water column, primarily when interspecific competition arises with other salmonid species (Schutz & Northcote, 1992) —implying that water velocity and depth are key determinants of

Dolly Varden char occurrence (Bozeman & Grossman, 2019). Typically Dolly Varden char are fall spawners, with non-anadromous individuals being significantly smaller and younger than migratory forms (Koizumi, Yamamoto, & Maekawa, 2006).

In the PNW, basic biological information is lacking for many populations of Dolly Varden char (Bozeman & Grossman, 2019). Specifically, non-anadromous forms of this species are understudied, resulting in limited published information (Bozeman & Grossman, 2019). Currently, in southern British Columbia, Dolly Varden char is not a species of concern; however, there have been no recent broad population assessments conducted. Although, a detailed study focused on Campbell River Dolly Varden char, on Vancouver Island, BC, identified a decline in Dolly Varden char since the 1970s, supporting the case that this species requires an increased knowledge base (Michalski, 2006). It is essential to identify the current stream habitat requirements of this species as they typically inhabit streams with other salmonids present, thus isolating important physical factors that affect this species persistence contributes to larger conservation efforts.

## **1.5 Study overview**

The purpose of this thesis is twofold. In Chapter 2, I will examine habitat associations for stream-dwelling rainbow trout and Dolly Varden char in southern British Columbia. Previous studies have found that habitat preference is variable and highly dependent on location, season, and size class (Hartman, 1965; Armstrong & Morrow, 1980; Bramblett et al., 2002; Harvey, White, & Nakamoto, 2005; Reeves & Grunbaum, 2010). Therefore, this study is focused on the summer rearing habitat preference of individuals aged 1+ on Vancouver Island, BC. Obtaining this information is critical for guiding conservation efforts in this region, especially accounting for recent climate change challenges faced by coastal watersheds. In Chapter 3, using fish-habitat

relationships established in Chapter 2, I will utilize three-dimensional ALS data to measure small stream metrics at the habitat unit scale, and assess whether field-measured and ALS-measured variables can predict fish density in a similar way. This remote sensing technology may create a new tool for resource managers to more rapidly and cheaply assess fish abundance and habitat quality.

## 1.6 Study system

In the summer of 2019 I sampled five tributary streams of the Nahmint River watershed, located on the southwest coast of Vancouver Island, British Columbia (Figure 1.1). This watershed falls within the coastal western hemlock biogeoclimatic zone (CWH) dominated by sizeable western hemlock (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), and Douglas-fir (*Pseudotsuga menziesii*). The understory vegetation consists of deer fern (*Blechnum spicant*), salal (*Gaultheria shallon*), Alaskan blueberry (*Vaccinium alaskanense*), red elderberry (*Sambucus racemose*), and a variety of mosses on the forest floor (Meidinger & Pojar, 1991). The climate in this region is typically characterized by mild winters, cool summers, and high annual precipitation, with the majority of rainfall occurring between October and April (Meidinger & Pojar, 1991).

The Nahmint watershed is divided by Nahmint Lake into upper and lower sections. In the lower section, anadromous salmonids including chum (*Oncorhynchus keta*), Chinook (*Oncorhynchus tshawytscha*), coho (*Oncorhynchus kisutch*), sockeye (*Oncorhynchus nerka*), and pink salmon (*Oncorhynchus gorbuscha*). These species are not found in the upper portion of the watershed because of the presence of a 4.0 m tall waterfall located 4.8 km up the Nahmint River from the estuary (Brydges, Luzzi, & Burrell, 1999). This study was conducted in the upper portion of the watershed, located above the falls and Nahmint Lake. The upper portion of the



watershed supports resident rainbow trout, Dolly Varden char, and kokanee (*Oncorhynchus nerka*) (Narver, 1974). Summer run and winter-run steelhead trout can be found within the system, with only summer-run individuals passing above the waterfall (Ministry of Forests, 1990). However, populations of this anadromous life history form of rainbow trout in British Columbia have exhibited large declines (Kendall, Marston, & Klungle, 2017; COSEWIC, 2018b), suggesting that the abundance of steelhead trout is very low in this watershed. Therefore, *O. mykiss* captured during sampling were most likely resident rainbow trout.



**Figure 1.1** Map of the Upper Nahmint Watershed with five study streams, located on the southwest coast of Vancouver Island, BC. Sections of study reaches are defined by different colours.

## **Chapter 2: Summer physical habitat associations and movement of sympatric juvenile rainbow trout (*Oncorhynchus mykiss*) and Dolly Varden char (*Salvelinus malma*) in coastal streams of British Columbia**

### **2.1 Introduction**

Specific habitat features have been identified as important for stream-dwelling salmonid survival in terms of food, energetics, and shelter (Bjornn & Reiser, 1991). For example, pool habitats provide important rearing areas for juvenile coho (*O. kisutch*), Chinook (*O. tshawytscha*), cutthroat trout (*O. clarkii*), rainbow trout (*O. mykiss*) and Dolly Varden char (*Salvelinus malma*) (Hartman & Gill, 1968; Fausch, 1993; Rosenfeld, Porter, & Parkinson, 2000; Winkowski & Zimmerman, 2018; Bozeman & Grossman, 2019). Pool habitats provide ideal rearing conditions as they have reduced water velocities, cover associated with water depth, and often instream wood, an additional component of cover that is associated with pools (Rosenfeld, Porter, & Parkinson, 2000). Areas of faster-flowing water (i.e. riffles) are also utilized by some drift foraging species like rainbow trout (Everest & Chapman, 1972; Bjornn & Reiser, 1991). Streamside vegetation provides concealment and shelter (Fausch & Northcote, 1992; Harvey, White, & Nakamoto, 2005), and shade which along with tributary junctions and groundwater sources, provides cool water areas that are important for stream-dwelling salmonids particularly during the summer (Dugdale et al., 2015).

As water temperatures rise and upper thresholds of thermal tolerances are reached, stream-dwelling fish have been documented seeking refuge through reach-scale movements (Mellina et al., 2005; Dugdale et al., 2015; VerWey et al., 2018). Juvenile salmonids were previously thought to move very little in the summer, showing what has been termed 'restricted movement' (Edmundson et al. 1968; Bjornn 1971). Recent studies suggest that upstream (Khler,

Roni, & Quinn, 2001; VerWey et al., 2018) and downstream (Winkowski & Zimmerman, 2018) movements may be more prevalent during the summer low-flow period than previously believed, with movement linked to foraging and refuge locations (Gowan & Fausch, 2002). For example, juvenile steelhead trout and Chinook salmon were documented moving large distances downstream and upstream throughout the summer, suggesting that rearing habitat availability influences horizontal movements (Winkowski & Zimmerman, 2018). Further, diel horizontal migrations of juvenile salmonids have also been documented to exploit important feeding and thermal refuge areas (Armstrong et al., 2013).

Juvenile salmonid movement is a fundamental behaviour that influences ecological processes such as community structure and inter- and intra- specific competition (Schmetterling & Adams, 2004). Thus, understanding the scales at which fish move is an important component of the ecology of stream-dwelling fish, as individuals may not solely rely on one habitat type, but also spatially heterogeneous areas (Winkowski & Zimmerman, 2018). Movement and habitat use knowledge can be applied to better inform the scale at which habitat alterations may impact stream-dwelling fish. For example, as habitat availability in tributary streams is reduced and connectivity is lost due to reductions of summer flows, individuals may become concentrated in smaller habitats, increasing inter-and intra- specific competition for limited resources (Hilderbrand & Kershner, 2000), ultimately impacting population persistence (Rosenfeld, Porter, & Parkinson, 2000). Thus, there is a need to improve the understanding of the direction, spatial scales, timing and variability of salmonid movement and habitat use in tributary streams to better inform restoration and conservation efforts in fragmented and altered habitats (Stewart, Cayan, & Dettinger, 2005; Mantua, Tohver, & Hamlet, 2010). Further, by quantifying movement,

conservation efforts can be concentrated to the scale at which individuals are using habitats during critical periods.

Rainbow trout is a commonly occurring stream-dwelling salmonid that has, like many stream-dwelling fish in the Pacific northwest, been seriously affected by habitat alterations (Slaney et al., 1996; Gregory & Bisson, 1997; Behnke, 2002). This species has experienced numerous population extinctions and declines, with 80% of populations across coastal British Columbia and Washington state being impacted (Kendall, Marston, & Klungle, 2017). Often co-occurring with rainbow trout are Dolly Varden char. Their populations are also not doing well, but they are less well studied, and in many cases, their population status is unknown (Washington Department of Fish & Wildlife, 2000; Williams et al., 2015). The habitat requirements of allopatric populations of rainbow trout have been examined (Hartman, 1965; Everest & Chapman, 1972); however, habitat use across this species range, and particularly when co-occurring with Dolly Varden char, has not been thoroughly investigated. At southern latitudes, there has been very little study of habitat use of Dolly Varden (Bozeman & Grossman, 2019). Further, this species is often found in cold spring-fed tributaries (Fausch, Nakano, & Ishigaki, 1994; Taniguchi & Nakano, 2000), which could be compromised by climate change impacts. Given the joint concern of climate change and habitat degradation, it is essential to establish the habitat associations of these species to provide adequate conservation measures.

In this study, I aim to increase the understanding of habitat requirements and movement of sympatric juvenile rainbow trout and Dolly Varden char during the summer in tributary streams of the Nahmint River on Vancouver Island, British Columbia. When these species co-exist, interspecific competition often arises, with rainbow trout displacing Dolly Varden char and partitioning them into benthic locales (Bisson, Sullivan, & Nielsen, 1988; Nakano & Furukawa-

Tanaka, 1994; Nakano & Kaeriyama, 1995). Comparing the habitat associations and movement between these two species will increase the understanding of spatial overlap that may exist during the summer months. My first objective was to identify the associations between physical habitat characteristics and densities of each species. Based on previous work on the ecology of these species, I expected both species to be associated with habitat unit gradient, large substrate, cover in the form of overhanging vegetation and instream wood, and pool habitats (Hartman, 1965; Hartman & Gill, 1968; Everest & Chapman, 1972; Reeves & Grunbaum, 2010; McMillan et al., 2013; Ptolmey, 2013; Bozeman & Grossman, 2019). My second objective was to examine the frequency and magnitude of movement by each species at the reach scale. Previous work has documented a variety of rainbow trout directional and spatial movement (Bramblett et al., 2002; Mellina et al., 2005; Armstrong et al., 2013; Winkowski & Zimmerman, 2018), in contrast to Dolly Varden char that have been documented moving primarily in a downstream direction during the summer (Bramblett et al., 2002; Berger, 2013). Understanding the habitat use and behaviour of these species during the summer, an already vulnerable time for juvenile salmonids will help aid broad-scale conservation efforts by highlighting key stream characteristics.

## **2.2 Methods**

In the summer of 2019 (June 3<sup>rd</sup> to July 21<sup>st</sup>) I sample five tributary streams in the upper Nahmint watershed. To incorporate the effects of natural variability that exist among streams owing to differences in physical attributes, I selected five streams that represented a range of stream bankfull widths from 5.1 to 10.6 m average width (Table 2.1). Although Dolly Varden char were only present in four streams, I included all five streams to reflect a gradient of stream sizes. Within each stream, approximately 200 m was sampled to include all the habitat types typical for that stream.

### **2.2.1 Habitat assessment**

Physical habitat characteristics of study reaches were assessed using an approach modified from Johnston and Slaney (1996). Habitat units were identified and categorized into: pool, glide, riffle, or cascade. Each unit was numbered, their start and end spatial location denoted, and each was measured for length, bankfull and wetted channel width, and mean bankfull and wetted channel depth using a metered measuring tape. The gradient of each habitat unit was assessed using a Suunto clinometer. Dominant and subdominant substrate composition was visually estimated as a percentage based on four-particle size categories: fine ( $\leq 1$  mm), gravel (2 mm-64 mm), cobble (64 mm-256 mm), boulder ( $>256$  mm), and bedrock. The percentage of dominant and subdominant instream cover type was visually estimated based on the amount of small wood ( $<2$  m length and  $<10$  cm diameter), in-stream functional wood (dead wood  $\geq 2$  m length and  $>10$  cm diameter protruding into the stream), boulders ( $>0.5$  m diameter), undercut banks ( $>1$  m length), and overhanging vegetation (vegetation hanging within 1 m of the water surface) (Moore et al. 2010). Canopy cover percentage was visually estimated from the middle of each unit, and riparian vegetation was classified as shrub/herb or mixed conifer-deciduous forest. In-stream functional wood was counted and categorized as: small (10-20 cm diameter); medium (20-50 cm diameter); or large ( $>50$  cm diameter).

### **2.2.2 Fish sampling**

I used a mark-recapture design to estimate population sizes, abundance, and density of juvenile rainbow trout and Dolly Varden char over four sampling events, two in June and two in July. In each reach for each sampling event, approximately 50 Gee style minnow traps (0.25-inch mesh, 16.5 x 7.5, 2.3-inch opening) were baited with salmon roe and set every 5.0 m. If stream flows were too low to permit traps from being fully submerged, fewer traps were deployed.

Traps were placed in the same locales for each sampling event. After 24-hours, traps were emptied, and fish from each trap were placed into individual buckets with corresponding trap number tags to ensure fish were released back to their capture location. Each fish was anesthetized in a tricaine methanesulfonate buffered solution (MS-222, 0.1 g/liter) prior to measuring fork length (FL, mm), mass (g), and implanting a 8.4 mm passive integrated transponder (PIT) tag. A 16-gauge hypodermic needle was used to insert the tag into the body cavity. Only fish > 60.0 mm FL were implanted with tags; smaller fish were marked by removing a small tissue clip from the top of their caudal fin. In all subsequent sampling, smaller fish were visually inspected for caudal slips, and larger fish were scanned with a portable digital Biomark ® tag reader to identify previously PIT-tagged individuals, and if individuals were unmarked, a clip or tag was administered. To ensure a full recovery from the anesthetic, individuals were placed in buckets with fresh stream water covered with vegetation and deemed recovered once they had regained equilibrium and were able to swim. Fish were then returned to their initial trap location. I monitored temperature with a handheld thermometer to minimize handling stress and did not sample if water temperatures exceeded 18.0° C.

The population size in each sample stream, including PIT-tagged and caudal clipped individuals, was estimated using the Schnabel method (Schnabel, 1938) which involves multiple sampling events where individuals are marked and released. In subsequent sampling events, individuals were examined for marks (i.e. recaptures) to estimate a population size based on the ratio of marked and unmarked individuals. Estimates were calculated using the *FSA* (Ogle, Wheeler, & Dinno, 2021) and *Rcapture* (Rivest & Baillargeon, 2019) packages in R 3.6.1 (R Core Team, 2021).

For both species, only PIT-tagged age 1+ (>60.0 mm; Armstrong and Morrow 1980; McMillan et al. 2013) fish were used in habitat association analyses because young-of-year (YOY) fish were rarely captured due to the mesh size of the minnow traps used. Individual PIT-tagged age 1+ fish retention within a study reach (an assumption of mark-recapture population estimates) and general within-stream movement patterns were examined from individual recaptures among sampling periods. The movement (distance travelled longitudinally) of recaptured age 1+ rainbow trout and Dolly Varden char was calculated as the net distance moved between the first and last capture in either upstream (positive number) or downstream (negative number) directions. Net movement provides a picture of the total distance and direction of movement. To further assess movement, individuals were categorized as habitat movers, distance movers, or non-movers. If an individual moved one or more habitat units, they were considered habitat movers (Kahler, Philip, & Quinn, 2001). If they moved 20.0 m or more (calculated using the distances between traps) they were categorized as a distance mover (Rodríguez, 2002). Individuals that did not fit these criteria were classified as non-movers.

### **2.2.3 Data analysis**

Statistical analyses were conducted using R (R Core Team 2021) with Type 1 error rates set to 0.05. Assumptions of linear regression and ANCOVA (including normality and variances) were visually assessed. Further, to meet these assumptions, the response variable fish density was log-transformed ( $x + 0.01$ ).

Fish density (number of fish/habitat unit area -  $m^2$ ) was calculated for each species as the average density in each habitat unit over the four sampling events (June – July). To assess the association of fish density with habitat unit variables (gradient, dominant substrate, cover, and habitat type), I used linear mixed-effect regression and analysis of covariance (ANCOVA). A



linear mixed-effect regression was used to relate juvenile rainbow trout and Dolly Varden char density separately to gradient. The model included gradient as the fixed effect and stream as the random effect to control for variability that may arise in habitat conditions among streams. To examine if the density of rainbow trout and Dolly Varden char differed among cover (large wood, small wood, overhanging vegetation, undercut banks, and large boulders, and none), substrate (cobble, boulder, gravel, and bedrock), and habitat types (pool, glide, riffle) I used ANCOVA, controlling for stream. Both species were rarely found in habitat units with dominant 'none' cover, dominant fine substrate, and cascades; therefore, these cover, substrate, and habitat types were not included in analyses. Further, Dolly Varden char were rarely found in units with fine or gravel substrates, so these substrate types were excluded. Significant effects were analyzed using Bonferroni multiple comparison test.

To assess if the mean net distance moved by recaptured rainbow trout and Dolly Varden char was different among streams, I used a one-way analysis of variance (ANOVA). Further, I compared the mean net distance moved by rainbow trout and Dolly Varden char with an ANCOVA, including streams that had both species present to assess if one species moved more than the other.

Physical stream variables of interest were assessed for collinearity prior to including them in some analyses. Specifically, Pearson correlation was used to assess if gradient was associated with bankfull channel width and water depth, as the assumptions of normality and equal variance were met for this analysis. Further, Pearson correlation was also used to assess if bankfull channel width and wetted width were correlated to determine if bankfull channel width could be used as a proxy to calculate fish density based on area measurements.

## 2.3 Results

### 2.3.1 Stream habitat

Sample reaches ranged from 231.1 m to 116.2 m in length, with minnow trap density averaging one trap per 4.4 m of stream. Reach gradient was lowest at Headache Creek (0 to 2.5; mean = 1 % [ $\pm$  0.8 SD]), and highest at Head Water Creek (0 to 23; mean = 3.2 % [ $\pm$  4.7 SD]) (Table 2.1). Rainbow creek ranged from 1.8 to 8 m and had the narrowest mean bankfull channel width (mean = 5.1 [ $\pm$  2.1 SD]; Table 2.1). Headache Creek's widest mean width (mean = 10.6 [ $\pm$  3.2 SD]; Table 2.1) ranged from 6.1 to 17.4 m. View Creek, Head Water Creek, and Elk Creek were low gradient riffle-pool sequence streams that had more adjacent off-channel habitat available during high flows than streams of higher gradient restricted by bedrock creating a step-pool sequence (Rainbow Creek, Head Water Creek in higher elevations). Rainbow Creek was dominated by boulder substrate, whereas all other streams were dominated by cobble. Across all streams, overhanging vegetation was the dominant cover type. A total of 142 habitat units were sampled across the five study streams, with age 1+ rainbow trout present in 42 pools, 23 glides, and 18 riffles (Table 2.2). Dolly Varden char were present in 20 pools, 11 glides, and 10 riffles (Table 2.2). Bankfull channel width was strongly correlated with wetted width ( $r = 0.40$ ,  $P < 0.001$ ,  $n = 139$ ). Therefore, bankfull channel width was used as a proxy to calculate fish density ( $\text{density}/\text{m}^2 = \text{mean fish abundance} / (\text{unit length} \times \text{unit bfw})$ ). Depth was found to be correlated with gradient ( $r = -0.45$ ,  $P < 0.001$ ,  $n = 139$ ) therefore it was not used in subsequent analysis.

**Table 2.1** Summary of stream length, gradient, bankfull channel width (BFW), dominant substrate type, and dominant cover type from the Nahmint River Watershed located on Vancouver Island, British Columbia. Samples were taken approximately every 5 m. Mean  $\pm$  SD values are given.

Stream	Length (m)	Gradient (%)	BFW (m)	Dominant substrate type	Dominant cover type
Headache Creek	231.1	1.0 $\pm$ 0.8	10.6 $\pm$ 3.18	cobble	ov
View Creek	210.1	1.51 $\pm$ 1.3	9.22 $\pm$ 3.06	cobble	ov
Elk Creek	185.3	1.70 $\pm$ 1.92	6.89 $\pm$ 1.95	cobble	ov
Head Water Creek	220.3	3.23 $\pm$ 4.72	5.18 $\pm$ 1.91	Cobble/bo ulder	ov
Rainbow Creek	126.2	2.34 $\pm$ 2.93	5.05 $\pm$ 2.10	boulder	ov

\* ov = overhanging vegetation

**Table 2.2** Mean number/m<sup>2</sup> of age 1+ rainbow trout (*O. mykiss*) and Dolly Varden char (*Salvelinus malma*) in pool, glide, and riffle habitat types across all sample streams in the Nahmint watershed ( $\pm$  SD, n = total habitats each species was present).

Habitat Type	n	Rainbow trout/m <sup>2</sup>	n	Dolly Varden/m <sup>2</sup>
Pool	42	0.05 $\pm$ 0.05	20	0.03 $\pm$ 0.03
Glide	23	0.02 $\pm$ 0.01	11	0.01 $\pm$ 0.01
Riffle	18	0.02 $\pm$ 0.004	10	0.01 $\pm$ 0.004

### 2.3.2 Mark recapture, population estimates, and densities

Across the five study streams, I captured and PIT-tagged 260 age 1+ rainbow trout and recaptured 69. Fifty-two young-of-year (YOY) rainbow trout were captured, and one was recaptured. I captured Dolly Varden char in four streams, PIT-tagging 108 and recapturing 18 PIT-tagged individuals. Twenty-six YOY Dolly Varden char were captured, with no recaptures.

Rainbow trout FL ranged from 37.0 mm to 182.0 mm (mean = 88.7 [ $\pm$  24.9]), with 89% of individuals having lengths of >60.0 mm (age 1+). Dolly Varden char FL ranged from 50.0 mm to 160.0 mm (mean = 87.8 [ $\pm$  22.1]), with 90% of the individuals measuring >60.0 mm (age 1+).

Across all five study streams, age 1+ rainbow trout and Dolly Varden were observed in pool habitats more than any other habitat unit type. Rainbow trout were captured in 42 pools, and Dolly Varden in 20. Mean rainbow trout and Dolly Varden age 1+ density was also greatest in pool habitats (0.01 to 0.4, mean = 0.05 fish/m<sup>2</sup>, SD  $\pm$  0.05; 0.01 to 0.2, mean = 0.03 fish/m<sup>2</sup>, SD  $\pm$  0.03, respectively; Table 2.2).

Schnabel method (Schnabel, 1938) population estimates for all sizes of rainbow trout ranged from 21 (95% CI: 15-32, Rainbow Creek) to 307 (95% CI: 174-582, Headache Creek) (Table 2.3). Dolly Varden char population estimates, including all size classes, ranged from 17 (95% CI: 6-41, Headache Creek) to 355 (95% CI: 196-694, View Creek) (Table 2.3). Rainbow trout density was generally higher than Dolly Varden char, with the exception of View Creek. Head Water Creek had the highest rainbow trout unit density (0.002 to 0.27 fish/m<sup>2</sup>, mean = 0.05 fish/m<sup>2</sup>, SD  $\pm$  0.04 fish/m<sup>2</sup>), compared to Elk Creek with the lowest mean unit density (0.002 to 0.03, mean = 0.01 fish/m<sup>2</sup>, SD  $\pm$  0.01) (Table 2.3). Elk Creek had the lowest mean Dolly Varden char unit density (0.01 to 0.04, mean = 0.02 fish/m<sup>2</sup>, SD  $\pm$  0.01; Table 2.3). Dolly Varden char mean unit density was greatest in View Creek (0.002 to 0.20, mean = 0.05 fish/m<sup>2</sup>, SD  $\pm$  0.06; Table 2.3). In Head Water Creek this species was only present in one unit (0.04 fish/m<sup>2</sup>).

**Table 2.3** Summary of estimated juvenile populations using the Schnabel method (Schnabel 1938) with 95% CIs and mean the number of fish/m<sup>2</sup> in habitat units across all streams ( $\pm$  SD). Population estimates were calculated using the FSA (Ogle, Wheeler, & Dinno, 2021) and Rcapture (Rivest & Baillargeon, 2019) packages.

Stream	Species	Population estimate (95% CI)	Mean number of fish /m <sup>2</sup> $\pm$ SD
Headache Creek	Rainbow trout	307 (174, 582)	0.02 $\pm$ 0.02
	Dolly Varden	17 (6, 41)	0.001 $\pm$ 0.004
View Creek	Rainbow trout	181 (124, 274)	0.02 $\pm$ 0.03
	Dolly Varden	355 (196, 694)	0.02 $\pm$ 0.03
Elk Creek	Rainbow trout	86 (35, 211)	0.01 $\pm$ 0.01
	Dolly Varden	28 (11, 69)	0.02 $\pm$ 0.01
Head Water Creek	Rainbow trout	130 (88, 200)	0.04 $\pm$ 0.05
	Dolly Varden	19 (9, 44)	*0.02
Rainbow Creek	Rainbow trout	21 (15, 32)	0.04 $\pm$ 0.04

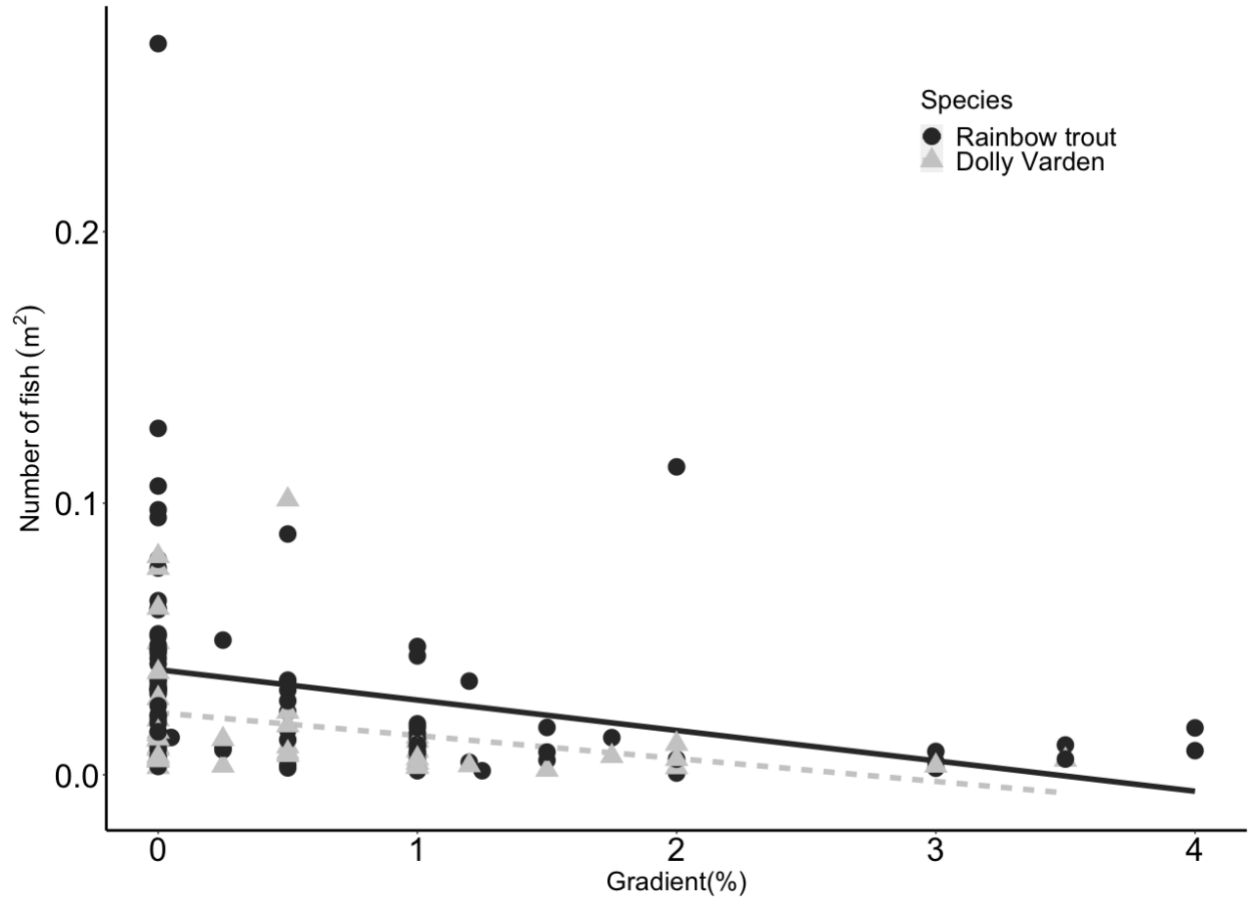
\*Present in one unit

### 2.3.3 Habitat associations

Rainbow trout density was significantly associated with habitat unit gradient ( $P < 0.001$ ). Further, no individuals were captured in locations with a gradient  $>4\%$  (Figure 2.2). Mean rainbow trout density was associated with substrate type (ANCOVA,  $F_{3,78} = 3.4$   $P = 0.021$ ), with mean fish density significantly higher in habitat units with dominant bedrock substrate compared to boulders (Bonferroni,  $P = 0.03$ ), no other comparisons were significant (Figure 2.3). Mean trout density was associated with habitat type (ANCOVA,  $F_{2,76} = 3.4$   $P < 0.001$ ). Further, density was significantly higher in pools than glides and riffles (Bonferroni,  $P = 0.01$  and  $P < 0.001$ , respectively) (Figure 2.4). There was no statistically significant difference in rainbow

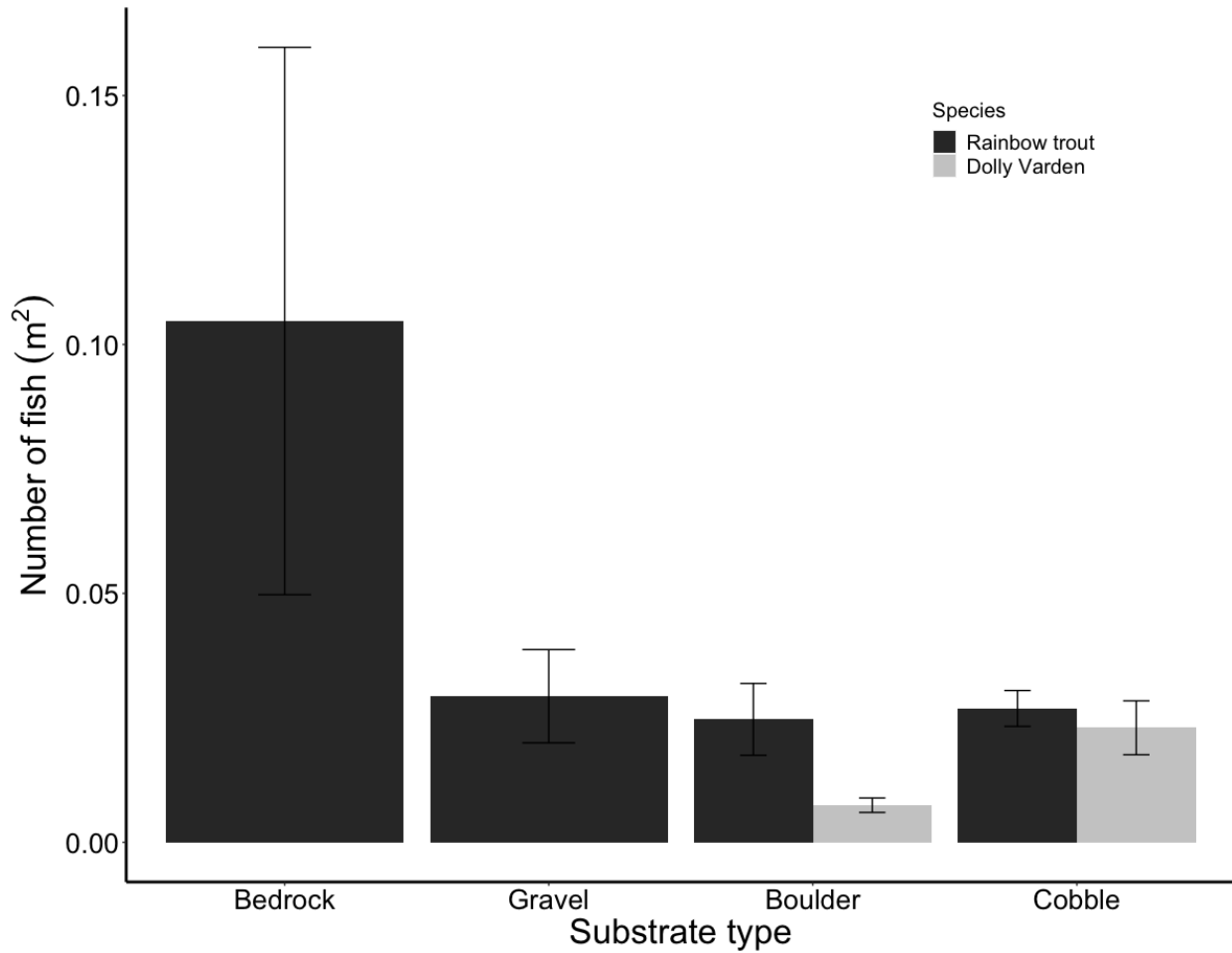
trout density between cover types. However, mean density was highest with small wood cover present (Figure 2.5).

Dolly Varden char density was significantly associated with habitat unit gradient ( $P = 0.006$ ). No individuals were captured in locations with a gradient  $>3\%$  (Figure 2.3). Mean Dolly Varden char density was associated with substrate type (ANCOVA,  $F_{1,31} = 5.4$   $P = 0.03$ ), with mean fish density significantly higher in habitat units with dominant cobble substrate compared to boulders (Bonferroni,  $P = 0.03$ ) (Figure 2.3). Dolly Varden char density was associated with habitat type (ANCOVA,  $F_{2,35} = 3.4$   $P = 0.02$ ). Density was significantly higher in pools than glides and riffles (Bonferroni,  $P = 0.01$  and  $P = 0.04$ , respectively) (Figure 2.4). There was no statistically significant difference in Dolly Varden char density between cover types. However, mean densities were generally higher with large wood and overhanging vegetation present (Figure 2.5).



**Figure 2.1** Relationship between the number/m<sup>2</sup> of aged 1+ rainbow trout (black, N = 88) and Dolly Varden char (grey, N = 43) and habitat unit gradient ( $P < 0.001$  and  $P = 0.006$ ,

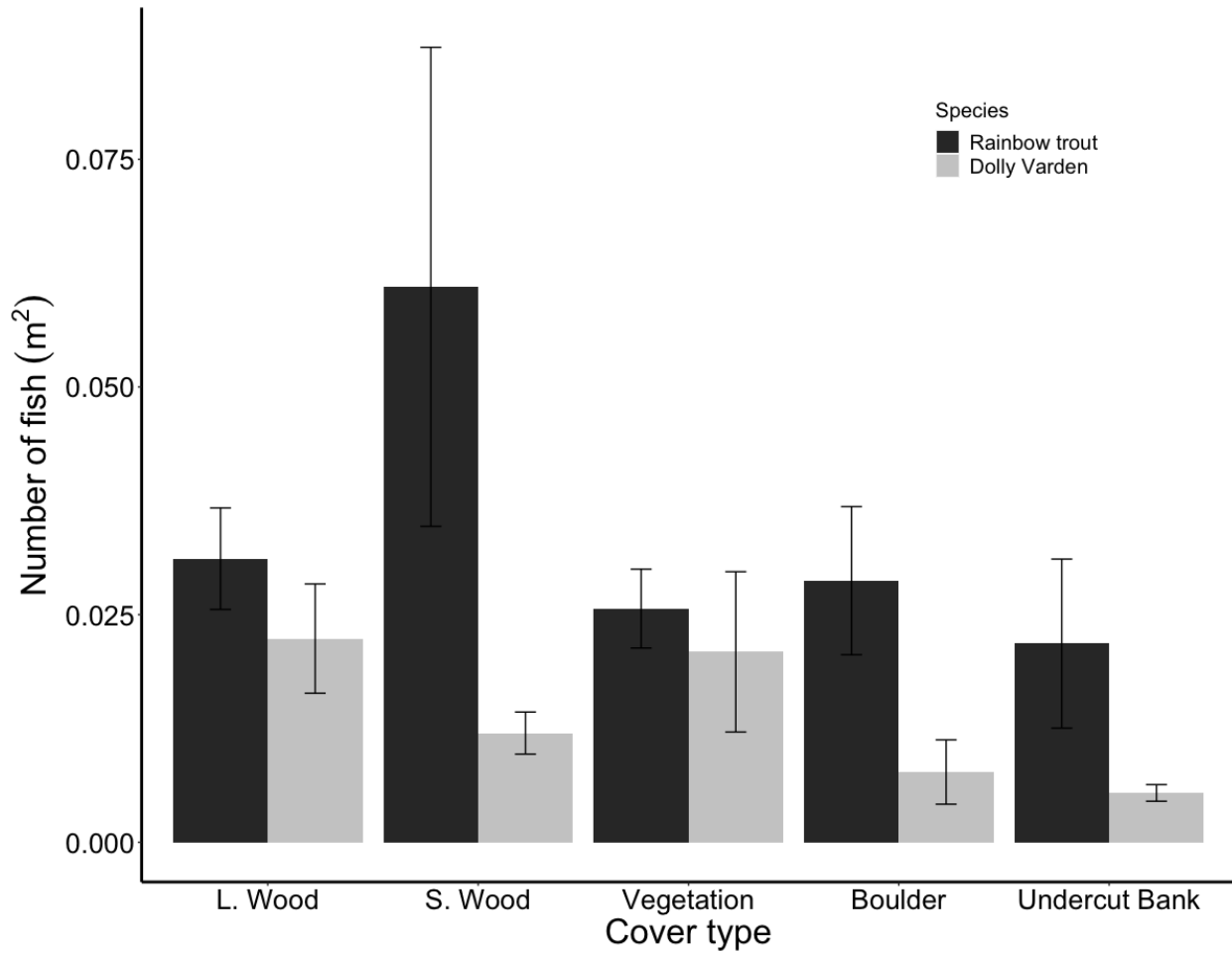
respectively). Points represent habitat units.



**Figure 2.2** The mean number/m<sup>2</sup> of age 1+ rainbow trout (black,  $N = 86$ ) and Dolly Varden char (grey,  $N = 36$ ) in habitat units with different dominant substrate classes, with error bars representing the SE of the mean. The mean number/m<sup>2</sup> of rainbow trout was significantly higher when bedrock was present ( $P = 0.03$ ), while the mean number/m<sup>2</sup> of Dolly Varden was

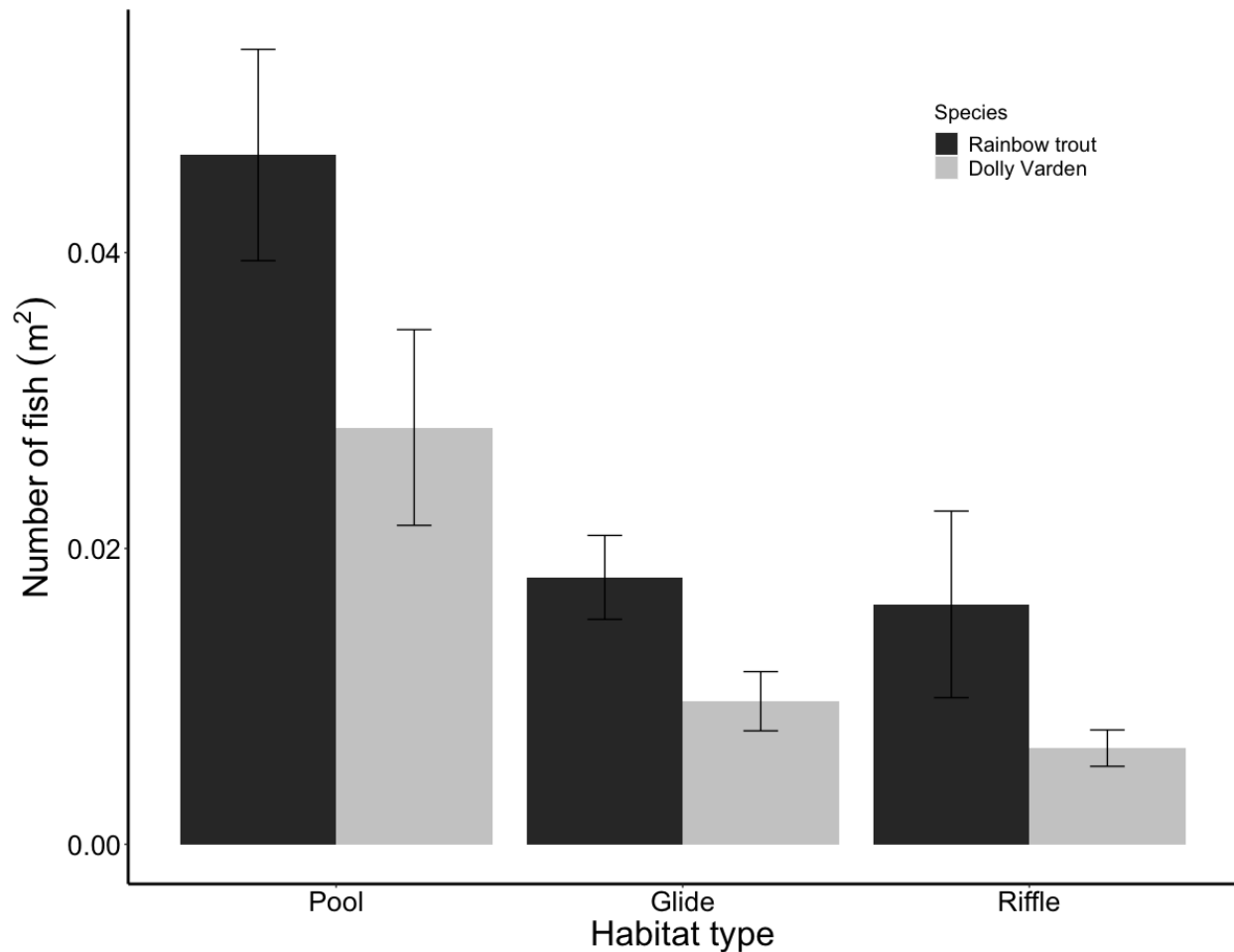


significantly higher when cobble was present ( $P=0.03$ ).



**Figure 2.3** The mean number/m<sup>2</sup> of age 1+ rainbow trout (black,  $N = 86$ ) and Dolly Varden char (grey,  $N = 43$ ) in habitat units with different cover types, with error bars representing the SE of

the mean.



**Figure 2.4** The mean number/m<sup>2</sup> of age 1+ rainbow trout (black,  $N = 86$ ) and Dolly Varden char (grey,  $N = 36$ ) in pools, glides and riffles, with error bars representing the SE of the mean. The mean number/m<sup>2</sup> of rainbow trout was significantly higher in pools than glides and riffles ( $P = 0.01$  and  $P < 0.001$ , respectively), while the mean number/m<sup>2</sup> of Dolly Varden was significantly higher in pools than glides and riffles ( $P = 0.01$  and  $P = 0.04$ , respectively).

#### 2.3.4 Fish movement

In total, 17% of recaptured rainbow trout moved at least one habitat unit (5 m), and 17% moved at least 20.0 m from their initial capture location during the 4 sampling events conducted from June 3<sup>rd</sup> to July 21<sup>st</sup>. Sixty-five percent of rainbow trout recaptured were non-movers.

Twenty-eight percent (5 of 18) of recaptured rainbow trout in Rainbow Creek, 16% (3 of 9) in Head Water Creek, 10% (2 of 20) in View Creek, and 22% (2 of 9) in Headache Creek were classified as habitat movers. No habitat movers were observed in Elk Creek. Twenty-two percent (4 of 18) of the recaptured trout from Rainbow Creek, 16 % (3 of 9) in Head Water Creek, 33% (1 of 3) in Elk Creek, 15 % (3 of 20) in View Creek, and 11% (1 of 9) in Headache Creek were classified as distance movers.

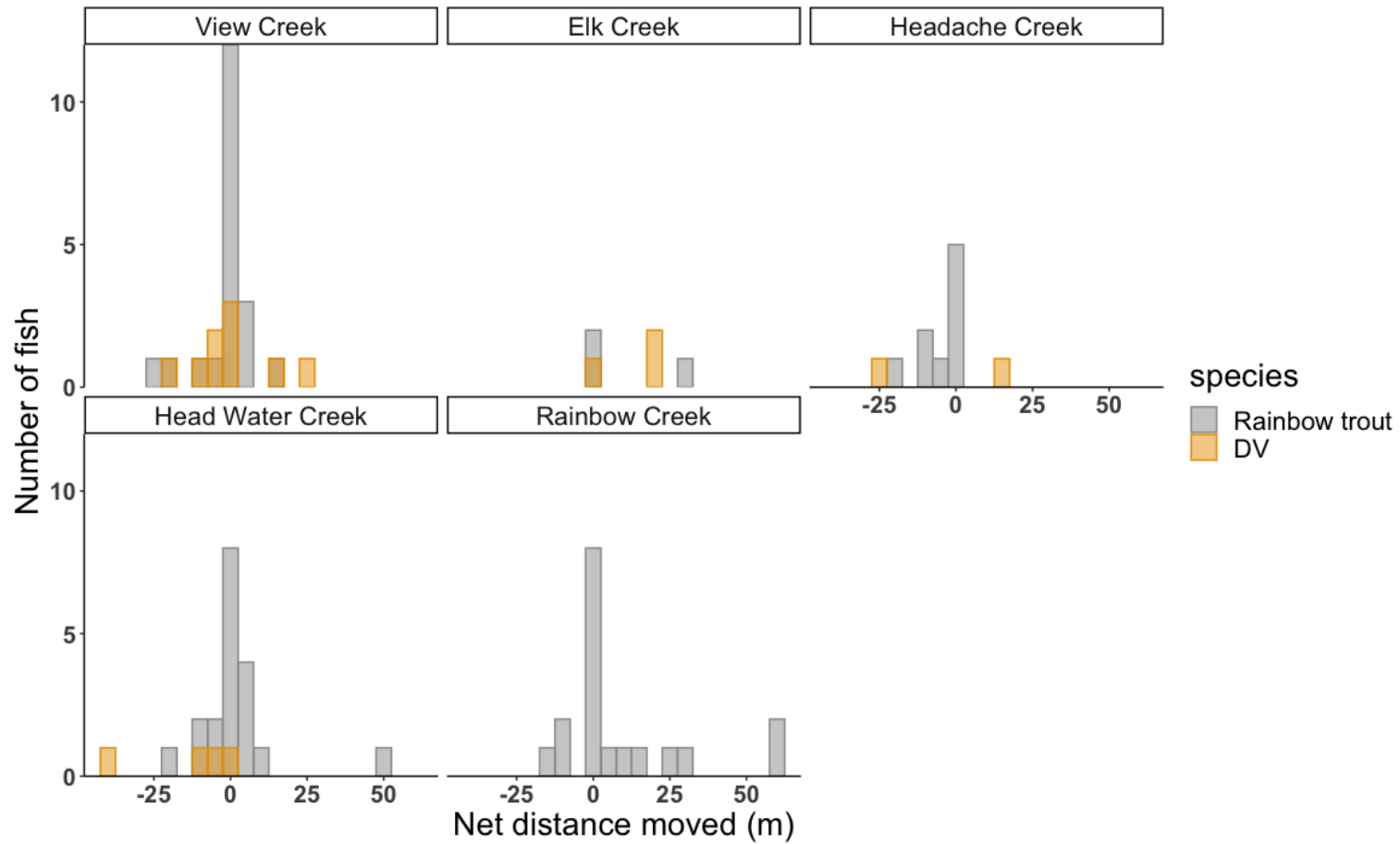
In total, 22% of recaptured Dolly Varden char moved at least one habitat unit, and 33% moved at least 20 meters from their initial capture location. Forty-four percent of Dolly Varden char recaptures were non-movers. Twenty-five percent (1 out of 4) of the recaptured Dolly Varden char in Head Water Creek, 22% (2 out of 9) in View Creek, and 50% (1 out of 2) in Headache Creek were classified as habitat movers. No habitat movers were observed in Elk Creek. Twenty-five percent (1 out of 4) of the recaptured char from Head Water Creek, 66% (2 out of 3) in Elk Creek, 22 % (2 out of 9) in View Creek, and 50 % (1 out of 2) in Headache Creek were classified as distance movers.

The net distance moved by rainbow trout was similar across each stream (ANOVA  $F_{4, 64} = 2.20$ ,  $P = 0.07$ ). Twenty-three percent of rainbow trout moved upstream, and 26% moved downstream. The net distance moved by rainbow trout in Rainbow Creek ranged from -15 to 60 (mean = 17.0 [ $\pm 27.1$ ]); 22.7 ( $\pm 18.1$ ) in Head Water Creek with a range of -20 to 50; -3.75 ( $\pm 13.8$ ) in View Creek with a range of -25 to 15; -11.25 ( $\pm 6.3$ ) in Headache Creek with a range of

-20 to -5. All individuals in Headache Creek moved downstream (Figure 2.6). In Elk Creek, an individual moved 30 m upstream.

Like rainbow trout, Dolly Varden char net distance movement was also similar across each stream (ANOVA  $F_{3,14} = 1.8$ ,  $P = 0.20$ ). Twenty-seven percent of Dolly Varden char moved upstream, and 44% moved downstream. The net movement of habitat units by char in Head Water Creek ranged from -40 to -5 (mean =  $-18.3 [\pm 18.9]$ );  $0 (\pm 16.7)$  in View Creek with a range of -20 to 25;  $-5 (\pm 28.2)$  in Headache Creek with a range of -25 to 15 (Figure 2.6). All individuals in Head Water Creek moved downstream. In Elk Creek, two individuals moved 20 m upstream.

In streams with both species present, there was no significant difference in net distance moved between rainbow trout and Dolly Varden char (ANCOVA  $F_{1,61} = 0.7$   $P = 0.41$ ) while accounting for stream.



**Figure 2.5** Frequency and net distance moved by recaptured rainbow trout (grey,  $N = 69$ ) and Dolly Varden (yellow,  $N = 18$ ) (yellow) in the Nahmint River watershed during summer 2019.

## **2.4 Discussion**

This study is one of the first to assess summer habitat associations and movement of co-occurring rainbow trout and Dolly Varden char in British Columbia. I found physical features measured at the habitat unit scale from my five study reaches associated with the density of both species, with some overlap. In addition, I observed generally restricted intra-seasonal movement by both species during the summer (June-July). Most rainbow trout stayed sedentary in their initial habitat locations, and if they did move, it was not in a consistent direction. In comparison, more Dolly Varden char moved both short and long distances, primarily in a downstream direction.

### **2.4.1 Habitat associations**

Rainbow trout and Dolly Varden char exhibited differences in abundance among study reaches. I observed rainbow trout abundance highest in reaches characterized by narrow bankfull channel widths and elevated gradients, indicative of swift-flowing water and step-pool sequences. This observation is similar to those reported for juvenile rainbow trout elsewhere in the Pacific Northwest (Hartman, 1965; Hartman & Gill, 1968; Johnson, Heifetz, & Koski, 1986; Bjornn & Reiser, 1991; Winkowski & Zimmerman, 2018). Further, Ptolmey (2013) quantitatively modelled juvenile steelhead as a function of stream size and gradient, finding juvenile steelhead trout dominant in third order and higher-order streams with adequate discharge. Although rainbow trout abundance was slightly higher in most streams, Dolly Varden char abundance was highest in wider reaches with lower gradients and long pool sections. Across this species native range, little published information is available pertaining to preferred tributary characteristics of stream-dwelling populations; however, some populations have been

documented in various stream types from high elevation mountain streams to large rivers (Armstrong & Morrow, 1980; Bramblett et al., 2002; Esin, 2015).

Further, both rainbow trout and Dolly Varden char were negatively correlated with habitat unit gradient. Rainbow trout were not found in habitats with a gradient  $>4\%$ , while Dolly Varden char were not found within units  $> 3\%$  gradient. Previous studies describing stream habitat associations for juvenile salmonids have established negative relationships between fish abundance and gradient. For example, juvenile steelhead are positively related to lower gradients at the habitat unit scale (i.e. step-pool sequences); however, they have been documented inhabiting reaches with gradients up to  $6\%$  (Roper, Scarnecchia, & Marr, 1994; Burnett et al., 2007). In comparison, species such as coho and cutthroat are more suited for long low-gradient sections in less steep streams (Rosenfeld, Porter, & Parkinson, 2000; Ptolmey, 2013). In coastal streams, there is often a longitudinal spatial overlap in species, but as the frequency of higher gradient of habitat units increases, patterns of species presence change. The frequency of habitat units with reduced gradients contributes to more favourable habitats, typically with deeper water. I found gradient to be negatively correlated with water depth, a key characteristic of pool habitats. Therefore, establishing a gradient threshold for the presence of a species within a tributary stream can help identify potential habitats related to the presence of a particular species without having to account for water depth.

Rainbow trout density was only associated with bedrock substrate. In contrast, previous studies have found that juvenile rainbow trout are often associated with large substrates (i.e., boulders), presumably as boulders provide cover, areas of refuge from swift water velocities, and optimal foraging microhabitats (Bjornn & Reiser, 1991). In the reaches examined in this research, juvenile rainbow trout density may have been higher in habitat units with dominant

bedrock substrate for several reasons. First, juvenile rainbow trout density was highest in the steeper-narrower streams (i.e., headwater streams) that were dominated by bedrock substrate, which morphologically influences the occurrence of step-pool sequences. Second, within the steeper-narrower streams, rather than seeking cover from large substrates like boulders or pieces of cobble, sufficient cover could have been provided from overhanging vegetation, and deep water found in pool habitats (Dolloff, 1986; Fausch, 1993; Boss & Richardson, 2002). For example, Lonzarich and Quinn (1995) found that water depth provided adequate cover and was a key variable in determining juvenile steelhead and coho salmon survival. Further, a lack of association with more substrate types (i.e. boulders) and specific cover types may have been the result of rainbow trout using all types of substrates and cover due to the high habitat quality I observed in my study reaches.

Alternatively, Dolly Varden char were significantly associated with cobble substrate. Similar to rainbow trout, a lack of association with boulder substrate was unexpected as boulders provide areas of refuge (Bjornn & Reiser, 1991). An association with cobble substrate could be explained by the abundance of Dolly Varden char being higher in wider, less-steep reaches, dominated by cobble substrate. Although no association was found between Dolly Varden char density and a specific cover type, mean density was highest when overhanging vegetation and large instream wood were present. Previous research has found overhanging vegetation and instream wood to be important cover types for many salmonids (Boss & Richardson, 2002; Gonzalez et al., 2017). For example, Saunders and Smith (1955) and Elliot and Waters (1969) concluded that brook trout (*Salvelinus fontinalis*) abundance depended upon cover availability. Similar to rainbow trout, the lack of relationship with a specific cover type may have been due to the abundance of cover provided by a variety of features.



Despite some differences in favoured habitat characteristics by rainbow trout and Dolly Varden char, similarities in habitat type preference were evident. Both species were positively associated with pool habitats. Across all sample reaches, both narrow and wide, pool habitats had the highest densities of both rainbow trout and Dolly Varden char. This observation is similar to other observations for both species (Hartman & Gill, 1968; Everest & Chapman, 1972; Fausch, 1993; Bozeman & Grossman, 2019), and likely because pool habitats provide important rearing conditions with slower water velocity, which minimizes the energetic costs of swimming (Fausch, 1984), while also providing cover (Lonzarich & Quinn, 1995) and thermal refugia during low flows (Baird & Krueger, 2003).

The overlap in the utilization of pool habitat by rainbow trout and Dolly Varden char suggests that interspecific competition may not be as prevalent as previously documented between trout and char species (Nakano & Kaeriyama, 1995; Jonsson et al., 2008). Typically, habitat partitioning occurs, with rainbow trout displacing Dolly Varden char due to their aggressive drift foraging behaviour (Everest & Chapman, 1972; Bisson, Sullivan, & Nielsen, 1988). However, when competition arises Dolly Varden char have the ability to shift between drift and benthic foraging (Nakano & Furukawa-Tanaka, 1994; Nakano & Kaeriyama, 1995) due to their capacity to see under low light conditions (Henderson & Northcote, 1985). In this study, the overlap in pool utilization suggests that interspecific food segregation behaviour may have occurred, allowing them to coexist in pools, with rainbow trout foraging within the water column or from the surface and Dolly Varden char occupying the benthic region (Nakano & Kaeriyama, 1995). Similarly, Naman et al. (2021) observed a less pronounced microhabitat partitioning of juvenile rainbow trout and bull trout (*Salvelinus confluentus*) among resting individuals in areas of reduced water velocities in the Skagit River, British Columbia. This study was observational;

therefore, I cannot make direct inferences on the occurrence of, and mechanisms related to competitive interactions between rainbow trout and Dolly Varden char in my study reaches. I did observe slight differences in stream size occupancy and cover association, but additional work is necessary to investigate further interactions between rainbow trout and Dolly Varden char and their habitat in southwest British Columbia.

#### **2.4.2 Fish movement**

Generally, minimal movement by age 1+ rainbow trout was observed in my small tributary streams during the summer (June 3<sup>rd</sup> - July 21<sup>st</sup>). Most fish stayed sedentary in their initial trap locations, and no clear direction of movement was detected. Previous studies have also identified no movement (Edmundson, Everest, & Chapman, 1968) or very little movement by juvenile salmonids during the summer low flow period (Khaler, Roni, & Quinn, 2001; Mellina et al., 2005; Armstrong et al., 2013). Specifically, rainbow trout may use flows as an environmental cue to halt movement (Bjornn, 1971). For example, Bramblett et al. (2002) found that juvenile steelhead emigration out of tributary habitats occurred primarily during the spring prior to low flows, resulting in low abundances and minimal movement during the summer in tributary streams of southern Alaska. In contrast, Winkowski and Zimmerman (2018) documented juvenile steelhead trout moving over 7 km in both up and downstream directions throughout summer in the Chehalis River, Washington state. Further, Northcote (1992) observed juvenile rainbow trout moving downstream in a lakeward direction in southern Interior British Columbia. These large differences reported among studies, including this one, suggests that movement may be system-specific with behaviour conforming to local environmental conditions experienced during the summer.

On the other hand, moderate movement by juvenile Dolly Varden char was observed, with slightly more recaptured individuals moving than staying sedentary in their initial habitat locations, with most individuals moving downstream. Similar to my results, Bramblett et al. (2002) observed primarily downstream summer movement patterns of Dolly Varden char in southeast Alaska, with relative abundance highest in tributary streams compared to the main stem. Further, small stream resident Dolly Varden char have been observed in high elevation streams of Alaska, Japan and Russia (Okada, 1960; Blackett, 1973; Esin, 2015). The Dolly Varden char that I captured ranged from 50 to 160 mm FL, and some individuals expressed milt, suggesting that the Dolly Varden char I studied are residents (Blackett, 1973; Schmidt & Robards, 1976; Esin, 2015). Thus, within my study system, resident Dolly Varden char may rely on the small tributary streams for year-round habitat.

Possible explanations for generally restricted movement by most juvenile rainbow trout and some Dolly Varden char are varied, as individuals have to ability to display sedentary or movement behaviours, allowing them to adapt to changes in environmental conditions (Reeb, 2003). Residency may have been favoured due to the energetic costs of movement to establish new territories, acclimation to new environments (i.e., due to low flows), and threats imposed by predators (Northcote, 1992). In addition, food abundance has also been documented to influence abundance levels and movement of trout during the summer (Wilzbach, 1985; Wilzbach & Hall, 1985). Therefore, restricted movement suggests that available habitat within my study reaches was sufficient and that my abundance estimates, which rely on the assumption of little immigration or emigration, are likely relatively accurate.

The net distance moved by rainbow trout and Dolly Varden char was similar across streams. Further, in streams with both species present, there was no significant difference in the

net distance moved between the two species. Notably, more rainbow trout moved in Rainbow Creek and Head Water Creek than in other reaches. Additionally, Dolly Varden char moved the longest distance downstream in Head Water Creek. Both of these reaches had abundant instream wood and allochthonous inputs that could have provided concealment from predators for short and long-distance movements (Boss & Richardson, 2002). Further, larger individuals were captured in these reaches and are often documented moving greater distances than smaller individuals (Young, 1994; Gowan & Fausch, 2002). In contrast, the least amount of movement was detected in Elk Creek and only in an upstream direction by one rainbow trout and two Dolly Varden char. In this stream, water levels decreased immensely as the summer season progressed, reducing habitat availability. Thus, a lack of recaptures and strictly upstream movement by both species suggests individuals emigrated out of this stream into the main stem where sufficient flow could be found. Movement upstream could have been into refuge habitats near edges of remaining habitat (VerWey et al., 2018), particularly sourced by groundwater inputs (observed in this reach). VerWey et al. (2018) observed similar trends in a stream in Oregon, USA experiencing drought, where cutthroat trout remained in their original capture location, and if an individual did move, it was upstream in search of refuge habitat. To further the understanding of movement behaviour of juvenile rainbow trout and Dolly Varden char in southwestern British Columbia, I suggest that further investigations on relationships between fish size, flow regimes, and movement on a broader scale are required.

### **2.4.3 Conclusion**

In this study, rainbow trout and Dolly Varden char were associated with pool habitats, highlighting the importance of these habitats during the summer low flow period, especially as areas for these species to coexist. Further, although both rainbow trout and Dolly Varden char

were not associated with overhanging vegetation and instream wood, these features are substantial components that contribute to high-quality habitat. As the frequency of extreme conditions increases due to anthropocentric causes, particularly reduced summer flows, pool habitats with physical cover will become increasingly crucial for stream-dwelling salmonid survival (VerWey et al., 2018). Considering ways to improve pool habitats and increase physical cover in small coastal streams must be considered as an approach to combat changes to hydrological regimes (Polivka & Claeson, 2020).

Generally, restricted movement was observed by both species, likely due to habitat availability in my study reaches. Previous work suggests that juvenile salmonids move during the summer in search of food and adequate shelter (Khaler, Roni, & Quinn, 2001). With changes to habitat complexity and hydrological regimes, reductions in movement and abundance of stream-dwelling salmonids could be expected. Typically, in the summer, habitat connectivity reduces, and pool habitats shrink. As hydrological shifts become more prevalent during the summer, populations may become stranded and confined to isolated pools, where they are faced with higher mortality rates (Rosenfeld, Porter, & Parkinson, 2000). However, as observed by the lack of recaptures in Elk Creek and generally low recapture rates overall, some populations of juvenile salmonids could be required to undertake long-distance movements in search of suitable habitat. Long-distance movements may lead to additional unintended consequences, with fish descending into areas below barriers, from which they cannot reascend. For many populations of juvenile salmonids, changes in movement patterns may become reflective of hydrological changes influenced by climate change (Rodríguez, 2002). Thus, it is essential for resource managers to obtain an understanding of location-specific habitat associations and movement to ensure the success of future conservation efforts.

## **Chapter 3: Assessing habitat associations of juvenile salmonids using stream metrics derived from Airborne Laser Scanning**

### **3.1 Introduction**

Typically, stream habitat assessments are conducted on small scales ranging from 50-500 m sections of stream (Fausch et al., 2002). The focus of these assessments is to quantify instream habitat characteristics, establish habitat associations, and draw inferences about larger populations at the landscape level (Fausch, Hawkes, & Parsons, 1988; Fausch et al., 2002; McMillan et al., 2013). When sampling these representative reaches within a watershed, features that influence abundance are quantified, such as pools or areas with low gradient (Rosenfeld, Porter, & Parkinson, 2000), instream wood (Gonzalez et al., 2017) and large substrate (Everest & Chapman, 1972) to establish associations. In addition, within representative reaches, the health of adjacent riparian forests is often assessed, as they heavily influence stream habitat integrity (Mellina and Hinch 2009; Boss and Richardson 2002; Roni et al. 2015). Thus, using these key stream characteristics, predictions about populations are ‘scaled up’ to larger spatial areas based on smaller-scale habitat associations (Beecher, Johnson, & Carleton, 1993; Knapp & Preisler, 1999). However, due to sampling logistics, traditional approaches tend to focus on representative reaches rather than a comprehensive watershed census due to time and funding constraints. For example, field surveys typically involve a multi-person team walking representative streams for extensive periods in remote locations. Given the limitations of fine-scale analysis due to these sampling logistics, and that land conversion activities and climate change affect stream-dwelling salmonids on extremely large spatial scales, approaches to extend in-depth, fine-scale assessments of fish habitat accurately and cost-effectively to the landscape level are needed.

Recent advances in remote sensing applications could expand field-based fish habitat assessments to empirically derive broad-scale remote assessments of fish habitat in a timely fashion (Fransen et al., 2006). Airborne Laser Scanning (ALS), is an active remote sensing technology that provides detailed three-dimensional point cloud data with high accuracy (Johansen, Phinn, & Witte, 2010; Tompalski et al., 2017). ALS data are collected by emitting laser pulses from a low-flying aircraft, from which the height of objects beneath are recorded by measuring the amount of time taken for reflected pulses to return to the aircraft (Lefsky et al., 2002). Due to the large amount of 3-D data that can be generated by this technology, ALS has been applied to understand and predict wildlife distributions and their habitats (Vierling et al., 2008). ALS is often utilized to assess forest vegetation attributes (Wulder & White, 2012; Racine et al., 2021), therefore research has focused on terrestrial species, such as bird-habitat associations (Bradbury et al., 2005; Glad et al., 2020) and small and large mammal-habitat associations (Nelson, Keller, & Ratnaswamy, 2005; Shanley et al., 2021).

However, interest in the use of ALS for aquatic environments has been growing. For example, the width of larger streams (Johansen, Phinn, & Witte, 2010), habitat sequences, gradient (Cavalli et al., 2008), and stream magnitude (James, Watson, & Hansen, 2007) have been examined with this technology. Other studies have used ALS to assess salmonid habitat specifically. For example, Jones (2006) utilized ALS in conjunction with aerial photography to map historical salmonid spawning channels in the Dosewallip Basin, Washington State, and identify areas most suitable for restoration efforts (Jones 2006). Further, the characteristics of small tributary streams related to salmonid habitat have been investigated using ALS, such as fish occurrence (Tompalski et al., 2017), and discrete habitat types and instream wood (Dakin Kuiper et al., In Review). However, to generate broad-scale habitat mapping of small streams,

ALS-derived attributes need to be examined to see if they can be linked with fish population characteristics. Therefore, in this chapter, based on the habitat associations established in Chapter 2, I aim to expand the applications of ALS related to juvenile salmonid habitat by linking high-resolution ALS-derived habitat-scale variables with field-measured fish data. My first objective was to utilize ALS to derive two stream variables: habitat unit gradient and canopy cover percentage. My second objective was to compare ALS-derived gradient and canopy cover percentage to field data and assess the accuracy of the approach. Lastly, my third objective was to assess whether field-measured and ALS-measured variables relate to fish density in a similar way.

Establishing relationships between ALS-derived attributes and field-collected fish density data will be important to examine the efficacy of ALS technology for habitat modelling and salmon conservation at the landscape scale. If successful, ALS could then be used as a tool for resource managers to assess streams habitat quality in a new, timely, and cost-effective way. Moreover, ALS could be employed as a predictive tool to seek out juvenile salmonid distributions based on what is known about historical habitat associations.

## **3.2 Methods**

### **3.2.1 Study site**

This study was conducted within the same streams as Chapter 2. However, I initially sampled a total of eight tributary streams of the Nahmint River watershed, located on the southwest coast of Vancouver Island, British Columbia (Figure 2.1). Three streams were removed from analysis because they were non-fish bearing. Further, Bug Creek and Steep Creek, were removed from analysis because they were either too narrow (Steep Creek) or had a canopy



cover >95%, which blocked ALS returns from the underlying terrain (Bug Creek), preventing the estimation of gradient.

### **3.2.2 Stream habitat and fish sampling**

The sampling protocol for physical habitat assessments (modified approach from Johnston and Slaney 1996) followed the same as Chapter 2, however, only primary habitat units are included in analysis. Fish sampling also followed the same protocol as Chapter 2. I used a mark-recapture design to assess juvenile rainbow trout and Dolly Varden char presence/absence and density over four sampling events (June 3<sup>rd</sup> to July 21<sup>st</sup>). For each sampling event, fish density/m<sup>2</sup> was calculated as the number of fish (both rainbow trout and Dolly Varden char) captured in a habitat unit/ habitat unit length x habitat unit bankfull channel width, to account for the variation in reach length and width between reproducible units. Both species were included in density estimates because, in Chapter Two, their habitat associations were similar (i.e. higher densities were found in pools). All sizes of individuals were also included to account for habitats used by varying size classes.

To accurately georeference field measured data, the location of the start and end of each habitat unit was determined using a Trimble Geo7X GNSS unit. Individual points were then connected into stream segments and joined with the field data to validate the ALS-derived stream attributes.

### **3.2.3 ALS data**

ALS data were acquired in the fall of 2015 using a Riegl Q 1560 sensor with an approximate point density of 25-35 points/m<sup>2</sup> (Table 3.1). Data were processed using the methods developed by Dakin Kuiper et al. (In Review) and are further described below (Table 3.2).

**Table 3.1** *Details of ALS data collection.*

<b>LiDAR Acquisition Parameters</b>	<b>Nahmint Watershed</b>
Sensor	Riegl Q1560 Dual-Channel
Point Density	25-35 points/m <sup>2</sup>
Date Flown	2015
Total Area	153.76 km <sup>2</sup>
Intensity	6651 - 65535
Scan Angle	58°
Flying Height	1600 m Above Ground Level
Flight Speed	115 kts
Scan Rate	533 kHz
Wavelength	1064 nm

**Table 3.2** *ALS-derived metrics used in this study.*

<b>Predictor</b>	<b>Unit</b>	<b>Description</b>	<b>Category</b>
<b>Random Forest Predictor Variables</b>			
Width	m	Bankfull width of the stream channel	Stream Structure
Instream wood	Count	A count of instream large and medium wood	Stream Structure
Canopy Height	m	The maximum height of returns per pixel	Vegetation Structure
15 <sup>th</sup> percentile of height	m	Height of the 15 <sup>th</sup> percentile of returns	Vegetation Structure
Normalized Elevation	Index between 0-1	Mean focal filter followed by a standard deviation filter of elevation	Topography
Edge Density/roughness	Index between 0-1	Surface roughness index	Topography
Intensity Class	Very Low, Low, Med, High, Very High	Equally distributed quantile classes of intensity from ALS ground returns	Stream Structure
<b>Additional ALS attributes</b>			
Percent Canopy Cover	Percentage	The percentage of ALS first returns above 8 m in a 1 x 1 m cell	Vegetation Structure

Stream Gradient	Percentage	The change in elevation between two set locations	Stream Structure
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### 3.2.4 ALS metric methodology

ALS light pulse returns classified as ground points were used to create a 1 m resolution Digital Elevation Model (DEM). Due to the size of the study streams the DEM was preprocessed to correct for imperfections and enforce flow paths using a breach depression and stream burning algorithm (Lindsay, 2016; Wu, 2019). Study streams were then delineated using a standard workflow consisting of calculating the flow direction and flow accumulation, followed by setting a stream initiation area of two ha, similar to Tompalski et al (2017) who analyzed a stream network in a similar Vancouver Island watershed. To determine the geographic bounds of the streams, stream width was extracted by expanding a stream centerline into an area of homogeneous elevation by calculating the accumulated cost of moving from each terrain slope pixel to the nearest stream source and setting a slope threshold of 18 degrees. ALS stream metrics were then derived within the geographical bounds of the estimated stream width. In total, 9 ALS metrics were derived (Table 3.3) seven of which were used as predictor variables to classify discrete habitat unit types using a Random Forest model (Dakin Kuiper et al., In Review; Breiman, 2001). Additional details on the seven ALS metrics and the prediction of discrete habitat units can be found in Dakin Kuiper et al. (In Review) while the additional two attributes are described below.

Individual unit gradient was calculated following a similar logic to Tompalski et al. (2017). The ALS derived stream habitat units were converted into points at 1.0 m intervals along the stream centerlines. Each point was assigned an elevation value from the ALS derived DEM.

Gradient was measured by calculating the slope of a smooth curve fit through the first and last elevation values of each individual habitat units using local polynomial regression. Percent canopy cover is calculated by dividing the number of ALS first returns above a height threshold by the total number of first returns in a defined area (Wulder, et al., 2008). In this study, due to the type of forest in the study area, the percentage of first returns above 8 m in a 1 m by 1 m grid along the stream centerlines were used to calculate canopy cover.

ALS processing was done using the *lidR* package (Roussel et al., 2020b, 2020a). Using a Random Forests model, Dakin Kuiper et al. (In Review) classified habitat units, reported an overall 86% accuracy in assigning habitat types with ALS. Dakin Kuiper et al. (In review) classified a total of 18 pools, 21 glides, 32 riffles, and 17 cascades were categorized, with misclassification between riffles and glides evident.

### **3.2.5 Data analysis**

I assessed the correspondence between the ALS estimated gradient and canopy cover versus field measurements using Pearson's correlation coefficient, RMSE, mean absolute deviation, and bias. Before constructing models, multicollinearity between habitat characteristics was assessed using correlation plots and a variance inflation factors (VIF) (Zuur, Ieno, & Elphick, 2010). The relationship between habitat unit characteristics (explanatory variables) and fish density (response) was examined using generalized linear mixed models (GLMM). The explanatory variables examined were habitat unit gradient, canopy cover, and habitat type. To control for the uniqueness of each stream and sampling event, I treated stream and habitat unit ID as random effects. I opted to use zero-inflated GLMMs to account for habitats units with fish absent, as it was likely that these habitats were not suitable for juvenile salmonids, thus important to include. Given the zero-inflated structure of the data, I employed a zero-inflated

Hurdle Model composed of a Binomial distribution with a log link function to model fish absence (zero) and fish presence (one) as a binary response, and a Gamma distribution with a log link function to model the positive, continuous response of fish/m<sup>2</sup> (Atkins et al., 2013; Zuur & Ieno, 2016). Both field measured and ALS measured gradient and canopy cover were standardized to a mean of zero and a Standard Deviation of one. I calculated 95% confidence intervals (CIs) for model coefficients. The parameters for which the 95% CIs did not include zero were considered to be associated with the response. To interpret the binomial model coefficients, I examined the odds ratio expressed as a percentage with 95% CIs. I also used the average-marginal-effects method to more intuitively interpret coefficients (Fernihough, 2011) from both binomial and gamma models. To compare the explanatory power of the field and ALS models, I evaluated the marginal and conditional  $r^2$  for each GLMM following Nakagawa & Schielzeth (2013). All statistical analyses were conducted in R version 4.1.1 (R Core Team, 2021). GLMMs were undertaken using the *glmer* function in the *lme4* package (Bates et al., 2015).

### **3.3 Results**

#### **3.3.1 Stream attributes**

##### **3.3.1.1 ALS measures**

ALS gradient was lowest at Headache Creek (0.38 to 2.3; mean = 1.5 % [ $\pm$  0.6 SD]; Table 3.3), and highest at Head Water Creek (2.28 to 8.17; mean = 4.6 % [ $\pm$  2.13 SD]; Table 3.3). Mean ALS canopy cover was highest at Headache Creek (36.0 to 100; mean = 80 % [ $\pm$  19.0 SD]; Table 3.3), and lowest at Elk Creek (15.3 to 95; mean = 48 % [ $\pm$  23 SD]; Table 3.3).

### 3.3.1.2 Field measures

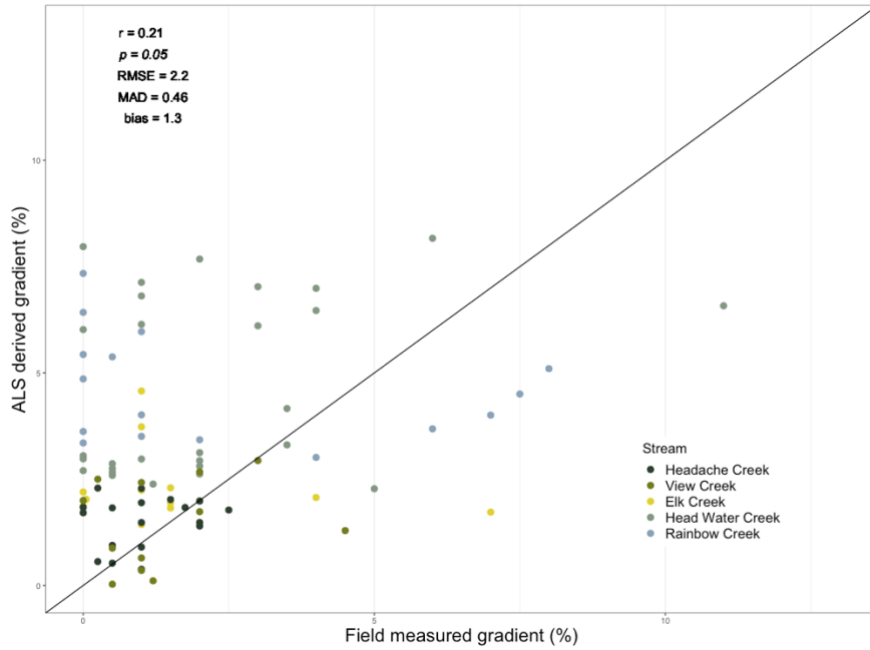
Stream gradient was lowest at Headache Creek (0 to 2.5; mean = 1.04 % [ $\pm$  0.8 SD]; Table 3.3), and highest at Head Water Creek (0 to 2.5; mean = 2.42 % [ $\pm$  2.80 SD]; Table 3.3). Mean field estimated canopy cover was highest at Head Water Creek (0 to 100; mean = 51.4 % [ $\pm$  37.7 SD]; Table 3.3), and lowest at Elk Creek (0 to 40; mean = 7.0 % [ $\pm$  13 SD]; Table 3.3). A total of 88 primary habitat types include: 17 pools, 21 glides, 34 riffles, 16 cascades.

**Table 3.3** Summary of field measured, and ALS estimated gradient, bankfull channel width and canopy cover of five fish-bearing streams in the Nahmint Watershed, located on Vancouver Island, BC. Samples were taken approximately every 5m. Mean  $\pm$  SD values are given.

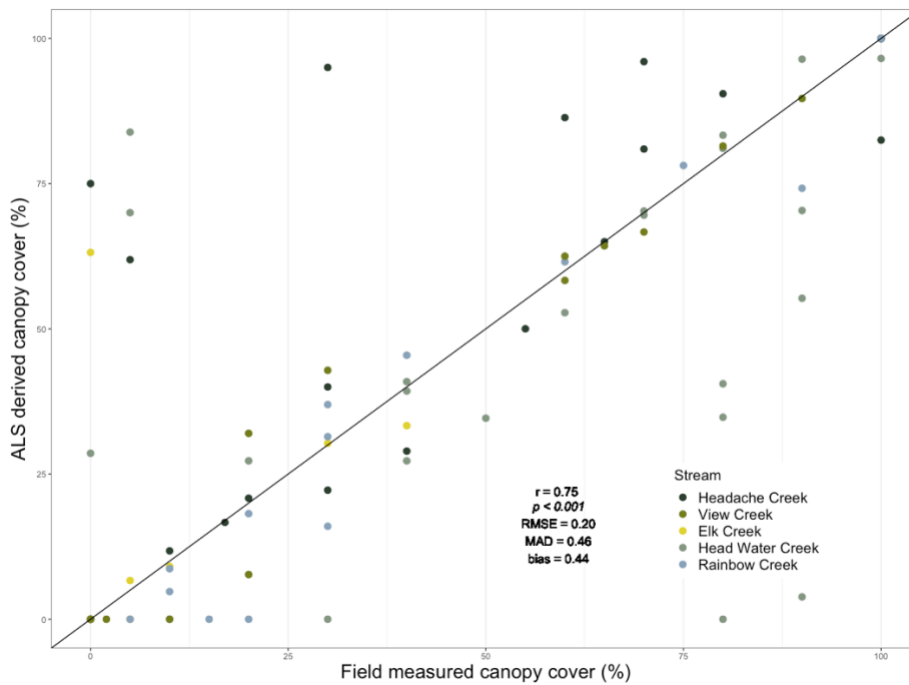
Stream	Length (m)	Gradient (%)	Canopy Cover (%)	ALS gradient (%)	ALS canopy cover (%)
Headache Creek	231.1	1.04 $\pm$ 0.8	49 $\pm$ 33.0	1.5 $\pm$ 0.60	80 $\pm$ 19
View Creek	210.1	1.41 $\pm$ 1.30	42.2 $\pm$ 31.9	1.5 $\pm$ 1.05	77 $\pm$ 29
Elk Creek	185.3	1.66 $\pm$ 1.92	7.0 $\pm$ 13.0	2.3 $\pm$ 0.90	48 $\pm$ 23
Head Water Creek	220.3	2.42 $\pm$ 2.80	51.4 $\pm$ 37.7	4.6 $\pm$ 2.10	69 $\pm$ 30
Rainbow Creek	126.2	2.38 $\pm$ 3.04	34.1 $\pm$ 31.0	4.6 $\pm$ 1.5	77 $\pm$ 29

### 3.3.2 ALS measures vs. field measures

Field-measured gradient and ALS-measured gradient were significantly correlated ( $r = 0.21$ ,  $P < 0.05$ ) (Figure 3.1). A bias of 1.3 % confirms that the approach to estimate gradient using ALS data slightly overpredicted gradient with a mean absolute deviation (MAD) of 0.46 % and an RMSE of 2.20 %. ALS estimated canopy cover and field-measured canopy cover were significantly correlated ( $r = 0.75$ ,  $P < 0.001$ ) (Figure 3.2). The approach to estimate canopy cover resulted in a 0.44 % bias, MAD of 0.46 %, and an RMSE of 0.20 %.



**Figure 3.1** ALS derived habitat unit gradient versus field measured gradient coloured by each stream.



**Figure 3.2** ALS derived canopy cover versus field measured habitat unit canopy cover coloured by each stream.

### 3.3.3 ALS vs. field measure habitat associations

All habitat characteristics included in the statistical models had a  $VIF < 3.0$ . In the Binomial GLMM, ALS classified pool and glide habitats were significantly associated with fish presence (Figure 3.3). Holding all other model variables constant, glides had an 83% (95% CI = 60-94%) greater probability of having fish present compared to riffles, while pools had an 89% (95% CI = 70-97 %) greater probability (Table 3.4). Expressed as the average marginal effect, there was a 29% chance of fish occurrences in glides and 38% chance of fish occurrence in pools. The marginal pseudo- $r^2$  for this model was 0.19, and the conditional was 0.50. Further, fish density was significantly associated with pool habitats in the Gamma model (Figure 3.4). On average, fish density was 3.1 fish/m<sup>2</sup> (95% CI = 1.8 – 5.5 fish/m<sup>2</sup>) higher in pool habitats compared to riffles (Table 3.5). Expressed as the average marginal effect, when pool habitats were present, the probability of fish density increasing was 5.7%. The marginal pseudo- $r^2$  for this model was 0.30, while the conditional was 0.30. I found no significant effect of ALS-derived gradient or canopy cover.

Comparing the results to the field-measures model, ALS performed with similar results. In the field measured Binomial GLMM with fish presence/absence as the response, pool and glide habitats were significantly associated with fish presence (Figure 3.3). Holding all other model variables constant, glides had an 84% (95% CI = 63-94%) greater probability of having fish present compared to riffles, while pools had an 88% (95% CI = 65-96%) greater probability (Table 3.4). Expressed as the average marginal effect, there was a 29% chance of fish occurrences in glides and 36% chance in pools. The marginal pseudo- $r^2$  for this model was 0.2, and the conditional was 0.5. In the Gamma GLMM, with fish density as the response, pool



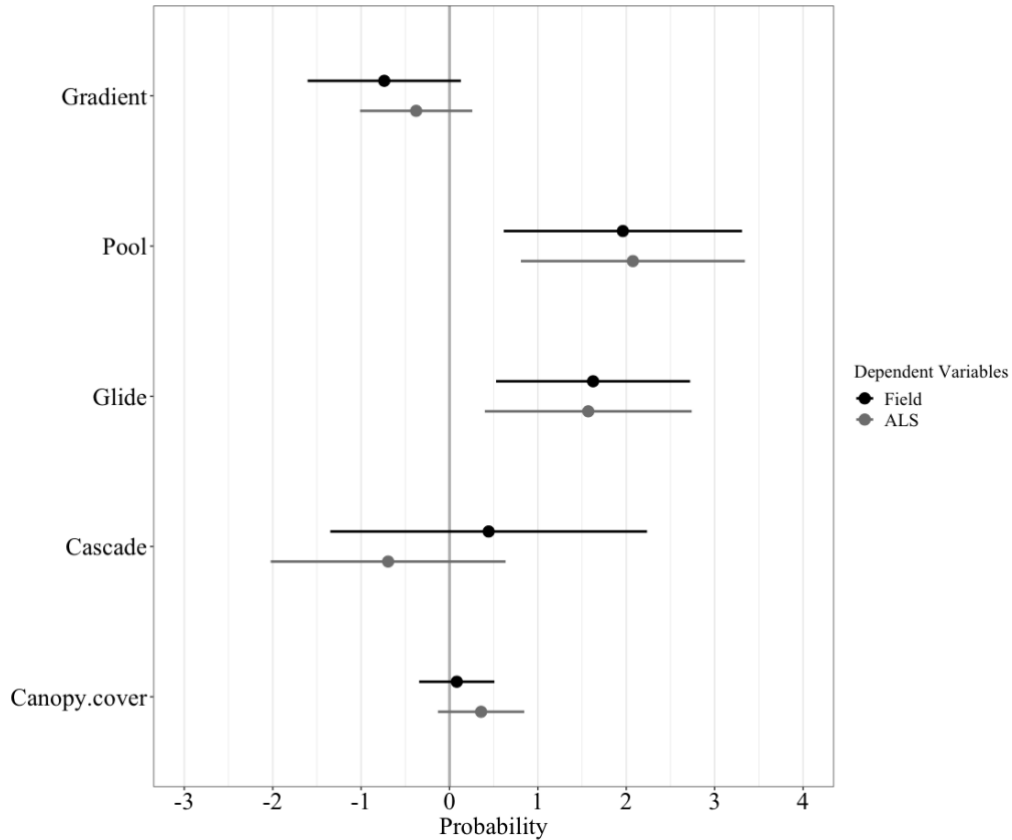
habitats were significantly associated with fish density compared to riffles (Figure 3.4). On average, there were 3.1 fish/m<sup>2</sup> (95% CI = 1.58 – 6.1 fish/m<sup>2</sup>) more in pool habitats compared to riffles (Table 3.5). The average marginal effect of pool habitat occurrence was a 5.5% increase in the probability of higher fish density. The marginal pseudo-*r*<sup>2</sup> for this model was 0.30, and the conditional was 0.64. I found no significant effect of gradient or canopy cover.

**Table 3.4** *Output of both ALS measured, and field measured Binomial generalized linear mixed model with a log link function. Values are expressed as the probability of fish presence.*

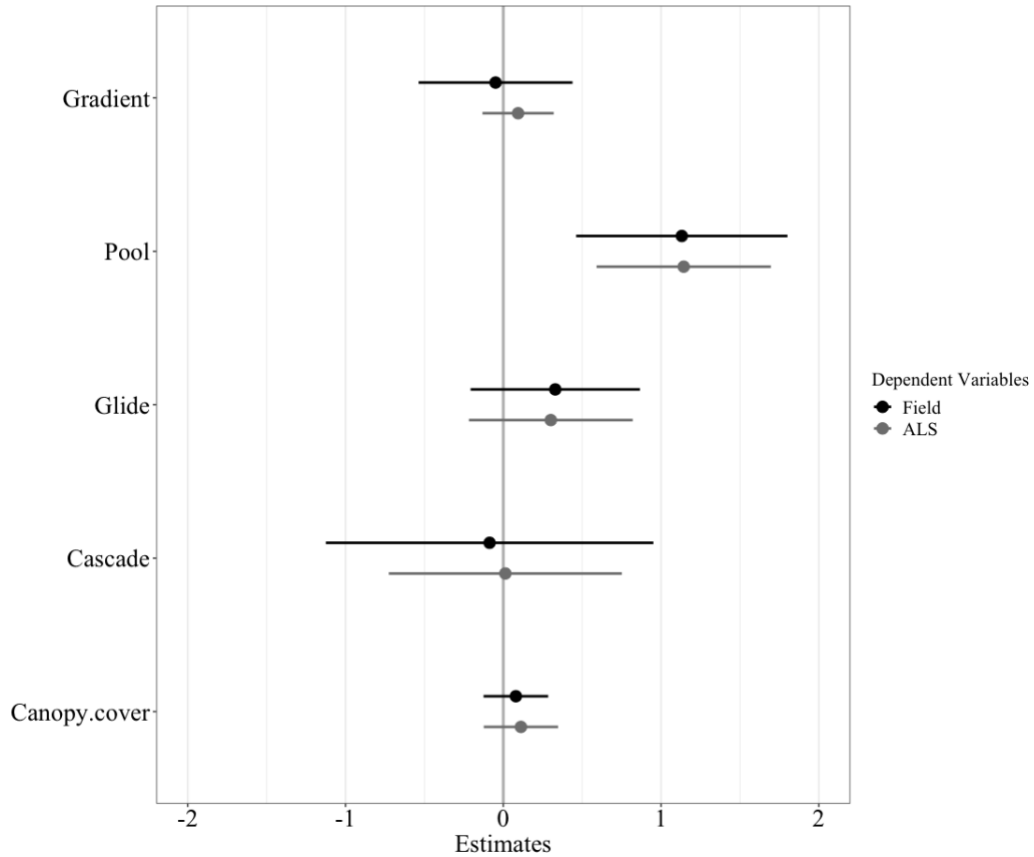
Binomial Models	Estimate	SE	95% CI
Field-measures			
Intercept	0.28	0.62	(0.13, 0.52)
Gradient	0.32	0.61	(0.17, 0.53)
<b>Pool</b>	<b>0.87</b>	<b>0.67</b>	<b>(0.65, 0.96)</b>
<b>Glide</b>	<b>0.84</b>	<b>0.64</b>	<b>(0.63, 0.90)</b>
Cascade	0.61	0.71	(0.20, 0.90)
Canopy cover	0.22	0.55	(0.41, 0.62)
ALS-measures			
Intercept	0.15	0.61	(0.15, 0.53)
Gradient	0.40	0.58	(0.27, 0.56)
<b>Pool</b>	<b>0.89</b>	<b>0.66</b>	<b>(0.69, 0.97)</b>
<b>Glide</b>	<b>0.82</b>	<b>0.64</b>	<b>(0.60, 0.94)</b>
Cascade	0.33	0.66	(0.12, 0.65)
Canopy cover	0.59	0.56	(0.47, 0.70)

**Table 3.5** *Output of both ALS measured, and field measured Gamma generalized linear mixed model with a log link function.*

Gamma Models	Estimate	SE	95% CI
Field-measures			
Intercept	0.02	1.21	(0.01, 0.03)
Gradient	0.95	1.28	(0.58, 1.55)
<b>Pool</b>	<b>3.10</b>	<b>1.41</b>	<b>(1.58, 6.05)</b>
<b>Glide</b>	<b>1.40</b>	<b>1.31</b>	<b>(0.81, 2.4)</b>
Cascade	0.92	1.69	(0.32, 2.60)
Canopy cover	1.08	1.11	(0.88, 1.3)
ALS-measures			
Intercept	0.02	1.21	(0.01, 0.03)
Gradient	1.09	1.10	(0.90, 1.40)
<b>Pool</b>	<b>3.10</b>	<b>1.30</b>	<b>(1.8, 5.50)</b>
<b>Glide</b>	<b>1.40</b>	<b>1.30</b>	<b>(0.80, 2.30)</b>
Cascade	1.01	1.46	(0.50, 2.12)
Canopy cover	1.11	1.12	(0.88, 1.40)



**Figure 3.3** Model standardized coefficients (probability) from the Binomial generalized linear mixed effect models of fish presence/absence in five tributary streams of the Nahmint River Watershed, Vancouver Island, BC, Canada sampled during summer 2019. Black circles denote the field measured model, while grey circles denote the ALS measures. Horizontal lines indicate 95% confidence intervals for each explanatory variable. Vertical line indicates zero, CIs that intercept zero are non-significant. Coefficients are recentred and standardized so that effect sizes are comparable among variables.



**Figure 3.4** Model standardized coefficients from the Gamma generalized linear mixed effect models of fish density/m<sup>2</sup> in five tributary streams of the Nahmint River Watershed, Vancouver Island, BC, Canada sampled during summer 2019. Black circles denote the field measured model, while grey circles denote the ALS measures. Horizontal lines indicate 95% confidence intervals for each explanatory variable. Vertical line indicates zero, CIs that intercept zero are non-significant. Coefficients are recentred and standardized so that effect sizes are comparable among variables.

### **3.4 Discussion**

In this study, I employed a novel investigation to link ALS-derived remote sensing data with small stream habitat characteristics important to juvenile salmonids. My results suggest that ALS measurements of small stream habitats provided a useful tool for modelling habitat associations of juvenile salmonids. Comparable results between the two models indicate that fish density is similarly related to ALS-derived metrics and field measures. Further, my results indicate a reasonable correlation between the ALS-derived gradient and field-measured gradient, increasing the capabilities of ALS to characterize attributes of small streams.

#### **3.4.1 ALS measures vs. field measures models**

A key result from this study is that both ALS-derived and field models indicated that fish presence and density were positively associated with pool habitats. Model parameters and fits assessed by pseudo marginal and conditional  $-r^2$  indicated that both ALS and field measures fit the data similarly. Further, the high accuracy achieved in the habitat type classification by Dakin Kuiper et al. (In Review) exemplifies the requirement of high quality and detailed remote sensing imagery required to assess fish habitat in small streams. When classifying habitat types using a Random Forest Model, Dakin Kuiper et al. (In Review) included ALS metrics such as roughness, elevation, canopy height, understory cover, and bankfull channel width, measured at one-meter intervals. Assessing stream metrics at this fine scale indicates the high amount of detail that ALS can provide (i.e. the number of attributes required for the RFM), while also suggesting that a close correspondence between ALS and true field measures is required to map small stream habitat types.

Using ALS-distinguished habitat types did not present any limitations although a few habitats were misclassified. Nonetheless, the habitat association between juvenile rainbow trout

and Dolly Varden char presence and density with pools habitats is generally consistent with prior knowledge and yielded a similar result achieved in Chapter 2. Pool habitats have been documented as a preferred habitat type by juvenile rainbow trout and steelhead, with larger individuals (age 1+) typically inhabiting deeper habitat (Hartman & Gill, 1968; Bisson, Sullivan, & Nielsen, 1988; Bjornn & Reiser, 1991; Lonzarich & Quinn, 1995; Reeves, Bisson, & Dambacher, 1998; Harvey, White, & Nakamoto, 2005). Further, Dolly Varden char and bull trout (*Salvelinus confluentus*) have also been reported using deep habitats with reduced water velocities (Al-Chokhachy & Budy, 2007; Bozeman & Grossman, 2019). However, younger individuals (age 0+) are sometimes observed inhabiting shallower habitat types (i.e., glides and riffles) (Bjornn & Reiser, 1991; McMillan et al., 2013), which often consists of small areas of pocket-pools or margins, reflecting the smaller three-dimensional space required by young-of-the-year fish (Grant & Kramer, 1990; Harvey, 1991). This could indicate why glide habitats were associated with fish presence in only the binomial GLMM - the densities of small individuals in glides were probably too small for an association to be established by the gamma GLMM. By including all size classes of fish in the model, it accounted for the variability in habitat associations by segregated size classes and represents all habitat types occupied.

The association found between pool habitats and fish density is not surprising because pools provide fish with ideal rearing conditions, consisting of slow water that minimizes the energetic cost of swimming (Fausch, 1984), cover provided by water depth and instream wood (Lonzarich & Quinn, 1995; Montgomery et al., 1995; Beechie & Sibley, 1997; Rosenfeld, Porter, & Parkinson, 2000), and thermal refuge during low flow periods (Baird & Krueger, 2003). The percentage of pools is an important explanatory variable in other studies which predicted fish density from habitat variables (Rosenfeld, Porter, & Parkinson, 2000; McMillan et al., 2013; See

et al., 2021). For example, Macmillan et al. (2013) found that pool habitat percentage positively influenced the density of juvenile coho salmon. Further, Myrvold & Kennedy (2016) found that pool proportion was a key determinant of juvenile steelhead movement, indicating that pool habitat provides structural complexity and reduced competition. Thus, with the capacity for ALS to map and quantify habitat units, there is the potential for this new technology to quantify habitat potential and be applied to many aspects of juvenile salmonid ecology.

No relationship was found between either ALS and field-derived gradient and juvenile salmonid habitat unit density. Previous studies describing stream habitat associations for juvenile salmonids have established negative relationships between fish abundance and gradient. For example, juvenile steelhead are positively related to lower gradients at the habitat unit scale (i.e. step-pool sequences) although they tend to inhabit steeper gradient streams (Roper, Scarnecchia, & Marr, 1994; Burnett et al., 2007). Further, in Chapter 2, both rainbow trout and Dolly Varden char habitat unit density was negatively associated with habitat unit gradient. In this Chapter a different result was yielded, likely due to the structure of the data. To match ALS and field measures, I included habitat units with fish absent, introducing a zero-inflated data structure. Thus, the abundance of units with zero fish likely nullified any relationship between fish density and gradient. Although no relationship was found between either ALS and field-derived gradient and juvenile salmonid abundance, estimating gradient contributes to the suite of attributes that can be derived with ALS that are related to juvenile salmonid habitat.

The lack of relationship between habitat unit density and canopy cover could be attributed to the nature of canopy cover not directly contributing to instream complexity. Although I found no direct link between fish abundance and canopy cover, riparian forests provide important structural components of stream habitat (Gregory et al., 1991; Fausch &

Northcote, 1992; Richardson et al., 2005). Riparian forests provide streams with shade, reducing solar radiation and water temperatures (Davies-Colley & Rutherford, 2005) while also contributing to bank stability and instream wood (Perry, Vellidis, & Lowrance, 1999). Wood inputs contribute to structural complexity and drastically influence pool habitat availability in coastal streams (Fausch & Northcote, 1992; Rosenfeld, Porter, & Parkinson, 2000). Nonetheless, ALS-derived riparian forest metrics are important to consider when quantifying fish habitat and these results indicate a reasonable correlation ( $r = 0.75$ ,  $P < 0.001$ ) between ALS-derived canopy cover and field-measured canopy cover.

Previous studies have demonstrated the ability of ALS data to measure stream gradient (see. Tompalski et al., 2017, Biron et al., 2013, Vocal Ferencevic & Ashmore., 2011). However, in this study, channel gradient was measured at varying intervals along the study stream reaches based on changes in discrete habitat units. Generally, measuring slope over such short distances can result in excessive detail and has the potential to magnify error (Vocal Ferencevic & Ashmore, 2012). However, calculating slope across too large an area can over-generalize stream gradient measurements and remove the local habitat unit scale variations important to juvenile salmonids. Therefore, in order to reduce the effect of short distance slope measurements a LOESS (Local Polynomial Regression) curve was fit through the maximum and minimum values of each habitat unit. These results indicate a reasonable correlation ( $r = 0.21$ ,  $P < 0.05$ ) between the ALS-derived gradient and field-measured gradient.

These results should be interpreted in light of the fact that fish and stream habitat sampling were collected within one watershed and in just one summer season. The habitat associations of juvenile salmonids are relatively consistent across literature; however, they can vary between streams, seasons, and years due to the heterogenic nature of stream habitats



(Fausch, Hawkes, & Parsons, 1988; McMillan et al., 2013). Recent studies have documented considerable movements of stream-dwelling salmonids across a range of stream environments (Kahler, Philip, & Quinn, 2001; Gowan & Fausch, 2002). For example, Winkowski and Zimmerman (2018) documented juvenile steelhead trout moving over 7 km in both up and downstream directions throughout summer, highlighting that spatial and temporal variability can arise. However, in Chapter 2, both species of fish displayed relatively sedentary behaviour. Thus, temporal, and spatial variability in habitat occupancy can influence the predictive accuracy of habitat models and should be considered when remotely assessing habitat. Accounting for temporal and spatial variability will likely be a challenge to incorporate with static ALS datasets. However, this study was focused on establishing the first link between fish data and ALS measures, bridging advanced remote sensing with juvenile salmonid stream ecology.

### **3.4.2 Conclusion**

This Chapter illustrates the utility of ALS habitat type classification to establish juvenile salmonid habitat associations. Models consisting of ALS and field measured variables were compared, showing similar results. With traditional fish survey approaches being time-consuming, costly, and often conducted over small areas, they limit the amount of available information pertaining to stream habitat. In contrast, ALS data provides information for the entire footprint of the acquisition area. Thus, many streams can be preliminarily assessed for habitat (i.e., habitat types) within a watershed before further infield surveys are conducted.

Given the concern for populations of salmonids and their stream habitats at southern latitudes, the development of a standardized protocol to incorporate remotely sensed data into the collection of habitat metrics would greatly increase the availability of habitat information at a broad scale, which could also be applied to habitat modeling. For example, See et al. (2021)

recently developed a quantile random forest model to provide estimates of juvenile chinook salmon carrying capacity during the summer at the reach and watershed scale. Variables that have the potential to be characterized with ALS, such as channel unit frequency, overhanging vegetation cover, instream wood, wood frequency in pools, and substrate composition, were included in the model (See et al., 2021). This study presents that stream attributes can be characterized with ALS and linked to fish ecology at the habitat unit scale, showing promising potential for other variables to be characterized at this scale, and broader scales.

## **Chapter 4: Summary and synthesis**

The purpose of this thesis was twofold. The first purpose was to broaden the understanding of the summer ecology of juvenile rainbow trout and Dolly Varden in small coastal streams, using traditional fish habitat assessments and sampling (Chapter 2). The second purpose of this thesis was to link traditional fish sampling methods with advanced remote sensing approaches to remotely assess juvenile salmonid habitat (Chapter 3).

Traditional fish sampling is time-consuming and costly, limiting the spatial extent of knowledge available on fish distributions and habitats. Thus, I used the habitat associations from Chapter 2, along with advanced remote sensing approaches to associate airborne laser scanning (ALS) derived metrics with fish density to determine if this technology can be utilized to map small stream habitats remotely (Chapter 3). While empirical studies should be used with caution, due to the variability that arises among streams, the observations and utilization of ALS in this thesis to assess fish habitat, provided evidence that this technology can be used to assess fish habitat in small coastal streams and could be implemented as a new tool for resource managers to determine stream habitats at the landscape level rapidly. Below I summarize my findings, discuss how these results could be implemented for the future management of salmonids, and suggest future applications of this new technology.

The persistence of stream-dwelling salmonids in the Pacific Northwest depends on the integrity of coastal streams. It is important to understand the spatial and temporal utilization of streams by stream-dwelling species to conserve and protect freshwater habitats. In Chapter 2, I found rainbow trout and Dolly Varden char to be associated with pool habitats, highlighting the importance of these habitats during the summer low flow period, especially as areas for these species to coexist. This finding enhances the importance of pool habitats as it is congruent with

previous research (Hartman & Gill, 1968; Everest & Chapman, 1972; Rosenfeld, Porter, & Parkinson, 2000). By creating fish-habitat associations, habitat types can be used to establish carrying capacities and highlight areas of important habitat (Bond et al., 2019; Polivka & Claeson, 2020). Further, I observed generally restricted movement by both species between June 3<sup>rd</sup> to July 21<sup>st</sup>. Previous work suggests that when juvenile salmonids move during the summer, they search for food and adequate shelter (Khaler, Roni, & Quinn, 2001). Thus, the streams that I surveyed most likely had sufficient summer rearing habitat conditions.

Observing higher fish densities in pool habitats combined with minimal movement has broader implications for the future management and conservation strategies of stream-dwelling salmonids. As thermal regimes shift, increasing the frequency and magnitude of droughts during the summer, fisheries managers should focus conservation efforts towards the maintenance or creation of pools and habitat connectivity in small streams of concern (Polivka & Claeson, 2020). As water levels deplete over the summer, populations are at risk of becoming stranded in isolated pools, increasing mortality rates (Rosenfeld, Porter, & Parkinson, 2000). Often, rehabilitation efforts are accomplished by constructing features out of in-stream wood to restore channel morphology and improve pool availability (Roni et al., 2015). For example, Roni and Quinn (2001) found that implementing additional in-stream wood in small coastal streams increased pool availability, which ultimately increased juvenile coho densities. Thus, improving the quality and quantity of tributary habitats of concern is a crucial component to the recovery and persistence of many salmonid populations (Bernhardt, 2005; Walters, Copeland, & Venditti, 2013).

Further, habitat connectivity requires consideration because, for many populations of juvenile salmonids, changes in movement patterns may become reflective of hydrological

changes (Rodríguez, 2002). This is reflected in Elk Creek, where recapture success was low, likely because individuals emigrated out of this study reach into the main stem due to the reduction of water in this reach as sampling progressed. Therefore, future studies should focus on surveying fish population characteristics and habitat quality and quantity (i.e. pools and connectivity) and movement of stream-dwelling fish in high-risk systems to understand the influences of climate-induced changes. Further, although not addressed in this research, streamflow and water temperature monitoring should also be conducted.

Traditional methods of obtaining and quantifying stream habitat information are time-consuming and limited to small spatial scales (50 – 500 m sections; Fausch et al., 2002), limiting the amount of available information about stream habitat in the Pacific Northwest. Given the concern for populations of salmonids and their stream habitats, the development of a standardized protocol to incorporate remotely sensed data into the collection of habitat metrics greatly increases the availability of habitat information. The results from Chapter 3 illustrate that ALS can map habitat units and establish habitat associations. Implementing remote mapping of stream habitat while considering fish-habitat associations can greatly benefit future management initiatives of stream-dwelling salmonids for several reasons.

First, remotely mapping habitat while also considering fish-habitat associations can help establish carrying capacities and highlight areas of important habitat in systems of high concern (Bond et al., 2019; Polivka & Claeson, 2020). For example, See et al. (2021) recently developed a quantile random forest model to provide estimates of juvenile chinook salmon carrying capacity during the summer at the reach and watershed scale. Variables that have the potential to be characterized with ALS, such as channel unit frequency, overhanging vegetation cover, instream wood, and wood frequency in pools, were included in the model (See et al., 2021).

Although in this study the sole ALS metric that was significantly associated with fish density was habitat type, it suggests that there is a promising potential for other variables to be characterized with ALS and linked to fish ecology, at both reach and habitat unit scales.

Second, remote mapping of stream habitats can greatly benefit habitat assessments and restoration activities. Within these areas of fisheries management, a-prior knowledge of habitat quality and quantity in highly managed systems, systems of concern, or systems in remote areas, can shape ground sampling logistics. For example, within British Columbia, measures of stream size (i.e. bankfull width) and fish presence (typically based on gradient) are used in stream protection guidelines, e.g. Forest and Range Practices Act (FRPA). The results from this study show that details at the habitat unit scale can be accounted for at the landscape scale, which helps account for stream complexity, an essential component of stream habitat quality (Fausch & Northcote, 1992; Rosenfeld, Porter, & Parkinson, 2000). Thus, by remotely observing habitat sequences, managers can assess fish habitat availability in addition to stream width and gradient, influencing which streams may require more habitat assessment attention or preservation based on the habitat presented. Further, once streams of high potential are identified, more intensive and targeted field sampling can be conducted. Extending this application and identifying a gradient of habitat potentials can also benefit restoration efforts. Although this is the first study establishing a link between fish data and ALS measures, it shows promising results for future bridging of advanced remote sensing with salmonid stream ecology.

There are several limitations to my thesis research that should be highlighted. The field sampling that I undertook was only conducted in one watershed, in one year, and in one season. Although underlying themes of habitat associations are consistent across the PNW, variability can arise in habitat associations between streams, seasons, years, and locations (geographic,

topographic, and climate) especially given rapid environmental changes (Fausch, Hawkes, & Parsons, 1988; McMillan et al., 2013). Thus, temporal and spatial variability in fish habitat utilization can influence the accuracy of ALS mapping applications. Further, accounting for temporal and spatial variability will be challenging to incorporate with static ALS datasets. Thus, implementing future applications of ALS mapping of stream habitat requires a more extensive investigation across variable conditions and should consider more species of stream-dwelling salmonids to account for the high heterogeneity that exists across small coastal streams in the PNW.

This is one of few studies contrasting both rainbow trout and Dolly Varden char habitat associations in southern British Columbia. Further, this is the first study linking juvenile salmonid ecology with LiDAR at the habitat unit scale. Given the concern of habitat degradation and unprecedented climate change consequences, stream-dwelling salmonids are highly vulnerable to anthropocentric influences. The persistence of all stream-dwelling salmonids will depend upon the management and restoration of freshwater habitat, making it essential to identify current stream habitat requirements and accurately map habitats. In conclusion, the results of this study provide forestry and fisheries managers tools to assess fish habitat using habitat unit scale measures, which can be expanded to the landscape scale, a scale that conservation decisions should be considered to ensure habitat connectivity.

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