Effects of logging on summertime low flows and fish habitat in small, snowmelt-dominant catchments of the Pacific Northwest

by

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The following individuals certify that they have read, and recommend to the Faculty of Graduate and Postdoctoral Studies for acceptance, the thesis entitled:

**Effects of logging on summertime low flows in small, snowmelt-dominant catchments of the Pacific Northwest**

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Abstract

Reductions in forest cover due to logging and natural disturbance often result in an earlier peak flow and an increase in annual yields. Most studies that consider the hydrological impacts of forest harvesting have focused on these changes to peak flows and annual yields, and not summertime low flows, even though summertime low flows are critical for instream rearing fish. This study specifically considered how forestry has impacted summertime low flows and fish habitat through time in three catchments in the southern interior of BC, two with an area of about 5 km², and one with an area of 36 km². To assess the hydrological impacts of logging, a paired-catchment analysis was applied to each day-of-year, as well as the July 1-September 30 yield. The post-harvest time series was divided into treatment periods, each about 6-10 years. To quantify the effects on altered flows on fish habitat, an eco-hydraulic instream flow habitat assessment using a physical habitat simulation (PHABSIM)-style approach was coupled to the results of the paired-catchment analysis. In the two catchments with the longest post-harvest time series, beginning approximately two decades after the onset of harvesting, summertime low flows were consistently less than the predictions from the pre-harvest regression. Reductions in summertime low flows were less pronounced in the third catchment, where there are only 10 years of post-harvest data. The eco-hydraulic habitat modelling indicated that these reductions in streamflow often corresponded to reductions in fish habitat that typically ranged from about 20-50%. The delay between forest harvesting and reductions in streamflow is speculated to be due to regenerating forests that produces greater transpiration loss than the original, mature forests. These results begin to fill an important gap in our knowledge about longer-term impacts of logging on summertime low-flows and fish habitat in small snowmelt dominated hydrological systems of the Pacific Northwest.
Lay Summary

Relatively few studies have assessed how logging impacts the timing and magnitude of water flowing in rivers during the late summer period. This study, which was conducted in three small catchments in the southern interior of BC, assessed 1) how the impacts of logging on these summertime flows evolved through time and 2) what the corresponding effects on fish habitat were. Results show that these summertime flows were reduced starting approximately two decades after the onset of forest harvesting and these reductions corresponded to decreases in modelled fish habitat in two of the catchments. The lag-time between forest harvesting and reductions in flow may be due to regenerating forests that use more water than the mature forests that were there before. These results begin to fill a knowledge gap about the longer-term impacts of logging on summertime flows and fish habitat in small, high elevation streams located in the interior regions of the Pacific Northwest.
Preface

This thesis is the original and unpublished work of the author, Stefan Gronsdahl. The paired catchment analyses was conducted under the guidance of Dan Moore. Jordan Rosenfeld, Brett Eaton, Rich Mc Cleary, and Rita Winkler provided guidance in relation to study design and execution. The normalized difference vegetation index (NDVI) component of the chronological analysis of clearcuts in Greata Creek (Appendix A) was conducted by Jordyn Carss.

Preliminary results from this study were previously reported in:


Results from this study were presented at the following conferences:

Gronsdahl, S, Moore, RD, Eaton, BC, and Rosenfeld, JS. 2018. Effects of forestry on summertime low-flows and physical fish habitat in snowmelt-dominated catchments of the Pacific Northwest. Oral presentation at the 2018 Joint Meeting of the CGU, CSSS, CIG, ES-SSA, and CSFAM, June 10-14, Niagra Falls, ON.

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1 Introduction

1.1 The Streamflow Regime in Snow-Dominated Catchments

A river’s streamflow regime describes the characteristic pattern of the timing, magnitude, and variability of the discharge it conveys. In the Pacific Northwest, river systems are typically classified into one of three regimes: rainfall-dominant, found in coastal watersheds, where snow does not typically accumulate throughout the winter; snowmelt-dominant, which occur in higher-elevation coastal or interior watersheds where snowpacks persist through much of the winter; and hybrid, which occur mainly in coastal catchments spanning a broad elevation range (Eaton and Moore, 2010). In snowmelt-dominant watersheds, the spring freshet is the primary hydrological event of the year, and is generally the cause of annual peak flows (Eaton and Moore, 2010).

Following the snowmelt freshet, streamflow declines into a period of low flows that persist throughout the summer and into autumn (Pike and Scherer, 2004). Post-freshet low flows are periodically disrupted by increases in streamflow due to rainstorms, especially in autumn, prior to the onset of the winter snow accumulation. Following the autumn rainstorms, low flows continue through the winter months, when precipitation falls as snow and streamflow is sustained by the depletion of water stored elsewhere within the catchment (Pike and Scherer, 2004; Eaton and Moore, 2010).

The general pattern of streamflow in snowmelt-dominant streams is broadly consistent from year to year, but the timing and magnitude can vary substantially (Eaton and Moore, 2010). Over decadal time scales, streamflow variability reflects
changes in climate (Hatcher and Jones, 2013) and changes to land cover, such as logging and natural disturbance. Previous studies of the impacts of forestry on streamflow have mostly focused on peak flows and annual water yield; in comparison, much less work done has assessed changes in low flows, particularly in snowmelt-dominant catchments (Moore and Wondzell, 2005). However, low flows are critically important because, as discharge declines, there is limited physical habitat available for fish (Bradford and Heinonen, 2008). In addition, when flows decline, the ability of a stream to facilitate the downstream transportation of aquatic invertebrates, which are an important food source for fish populations, is reduced (Rosenfeld and Ptolemy, 2012).

### 1.2 Forestry Effects on Hydrologic Processes

For headwater catchments without glaciers or surface water bodies, low flows are sustained by discharge contributions that originate from subsurface water (Winkler et al., 2010a). Depending on the local geology, subsurface contributions can occur from deeper groundwater aquifers as well as from shallow subsurface flow paths located above relatively impermeable soil or bedrock layers (Winkler et al., 2010a). The time lag between recharge of subsurface storage and discharge of water to the stream depends on catchment area, the hydraulic characteristics and spatial organization of aquifers, and the distribution and physiology of vegetation on the land surface (Smakhtin, 2001; McGlynn et al., 2004). Forestry operations potentially influence low flows by changing the magnitude and timing of soil moisture recharge, by removing trees that uptake and transpire subsurface water, and by modifying the pathways by which water flows to the stream channel.

Predicting how forestry may impact the hydrology of a stream requires an understanding of how the different hydrological processes that are acting within
a watershed are affected. The impacts of forestry on the various hydrological processes relevant to summertime low flows in snowmelt dominant catchments are reviewed through this section and are summarized in Table 1.1.

<table>
<thead>
<tr>
<th>Forestry Activity</th>
<th>Affected Hydrological Process</th>
<th>Expected Process Change</th>
<th>Effect on Summertime Low Flows</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tree removal</td>
<td>Interception of rain and snow / transpiration</td>
<td>Immediate reduction followed by a gradual increase through time as trees regrow</td>
<td>Immediate increase, followed by a gradual decrease through time as trees regrow</td>
</tr>
<tr>
<td>Tree removal</td>
<td>Snowmelt</td>
<td>Occurs earlier and at an increased rate. Return to pre-harvest conditions as trees regrow.</td>
<td>Possible spring-ward extension of summertime low flows.</td>
</tr>
<tr>
<td>Road and ditch construction/ operation of heavy machinery</td>
<td>Groundwater recharge and discharge</td>
<td>Reduction</td>
<td>Decrease that is consistent through time.</td>
</tr>
</tbody>
</table>

Forest harvesting increases winter snow accumulation by reducing interception loss. In the interior of BC, approximately 30% to 40% more snow is typically found in clear-cuts than under mature forest (Winkler et al., 2010b; Dickerson-Lange et al., 2017). In addition to the effects of harvesting on accumulation, removal or reduction of forest canopy increases incident solar radiation at the snow surface and increases wind speed, both of which increase the sensible and latent heat exchanges. Snowmelt in open areas tends to begin earlier in the spring and occur at a higher rate than under a forest canopy (Pike and Scherer, 2004; Moore and Wondzell, 2005; Winkler et al., 2017). In some cases, accelerated snowmelt results in an earlier disappearance of snow in openings (Moore and Scott, 2005; Winkler et al., 2017); however, in other cases the increased accumulation in openings is associated with a delay in snow disappearance despite the accelerated melt (Dickerson-Lange et al., 2017).
Interception loss from a forest canopy reduces the amount of precipitation that infiltrates the soil (Jost et al., 2007), and transpiration further reduces subsurface storage by removing soil moisture. In addition, when soil moisture falls below field capacity, water infiltrating the soil will tend to be held within the soil rather than recharging deeper groundwater or travelling downslope to the stream channel as throughflow. Both interception loss and transpiration are greatly reduced immediately following logging and tend to increase as forests regenerate.

Following logging, there is considerable uncertainty regarding how transpiration rates evolve in regenerating forests at the stand-level. Some studies suggest that transpiration rates increase asymptotically as forests age (Winkler et al., 2010b). This assertion is supported by a study of predominantly Douglas fir (*Pseudotsuga mensiezzii*) forests that found seven-year-old stands produced approximately two-thirds the evapotranspiration of nearby 19 and 58-year old stands (Jassal et al., 2009).

In contrast, other studies have suggested that transpiration rates reach a peak and gradually decline as forests mature (Vertessy et al., 2001). A geographically diverse body of work supports this hypothesis. In coastal Oregon, a 40-year-old forest transpired 21% more than a nearby, 450-year-old forest (Moore et al., 2004). Similar work in the interior of Oregon found that 40-year-old Ponderosa pine (*Pinus ponderosa*) forests transpired approximately 45% more than 290-year old forests over the June 1–August 31 period (Ryan et al., 2000) and that transpiration in replanted lodgepole pine (*Pinus contorta*) forests in the southern interior of BC steadily increased in forests up to 25 years old, with a 25-year-old stand having roughly the same transpiration as a mature stand (Winkler et al., 2010b). In Australia, increases in transpiration in relatively young, regenerating mountain ash (*Eucalyptus regnans*) forests have been linked to long-term decreases in water yield (Vertessy et al., 2001) and in France, maritime pine
(Pinus pinaster) forests were found to transpire less as they aged (Delzon and Loustau, 2005).

The effect of forest harvesting on low flows can depend on where it occurs within a watershed. For example, changes in transpiration due to harvesting upslope areas can take weeks to years to affect streamflow, whereas harvest within the riparian area can cause variations in streamflow noticeable over the course of a day (Moore et al., 2011). Studies of transpiration typically focus on individual trees and are not usually scaled up to the stand or catchment level, making it challenging to interpret how different transpiration rates affect the water balance of a catchment because tree densities and species compositions vary with forest age (Asbjornsen et al., 2011)

The use of heavy machinery like skidders during logging operations compacts soil, reducing its hydraulic conductivity and infiltration capacity (Moore and Wondzell, 2005; Zhang and Wei, 2012). Logging also commonly involves road construction and the creation of culverts and ditches that intercept shallow subsurface flow, more quickly conveying it to the stream network (Moore and Wondzell, 2005; Winkler et al., 2010b). Thus, logging activities typically result in a ‘flashier’ hydrological response (Wemple and Jones, 2003), with less water available to recharge groundwater flow paths that lag behind snowmelt and contribute to summertime low flows. These changes are expected to be relatively immediate and are expected to be consistent through time unless roadways are ditches reclaimed.
1.3 Effects of Forest Harvesting on Summertime Low Flows

The impacts of forestry on peak flows and annual yield are relatively well understood compared to the impacts on low flows (Moore and Wondzell, 2005). A review of available studies relevant to snowmelt-dominant catchments in the Pacific Northwest indicates that low flows are typically unaffected or increased slightly by logging (Table 1.2). Of particular relevance to this study, results from the 241 Creek sub-basin of Upper Penticton Creek in the southern interior of BC showed declines in July (-17%) and August (-20%), although these differences were not significant (Winkler et al., 2017).

A limitation of existing studies is that they typically evaluated the impacts of forestry on low flows for about 5-10 years following harvest (Pike and Scherer, 2003; Moore and Wondzell, 2005). These relatively short post-harvest time series stem, in large part, from the difficulty and time necessary to develop, plan, and execute paired catchment studies. Although some of the sites detailed in Table 1.1 do have extended post-treatment records, the analyses did not focus on the evolution of low flows through time. Another limitation is that the studies employed a range of low-flow metrics, such as seasonal water yield, which are not well linked to aquatic habitat conditions.
Table 1.2 Summary of paired catchment studies that consider low flows in snowmelt-dominant watersheds of the Pacific Northwest.

<table>
<thead>
<tr>
<th>Study Watershed</th>
<th>Author</th>
<th>Forest Type</th>
<th>Drainage Area (km²)</th>
<th>Elevation Range (masl)</th>
<th>Harvest method</th>
<th>Forest cover removed (%)</th>
<th>Post Harvest Period Studied (yr)</th>
<th>Low flow metric</th>
<th>Magnitude of change in low flow metric</th>
</tr>
</thead>
<tbody>
<tr>
<td>Camp Creek, BC</td>
<td>(Moore and Scott, 2005; Cheng, 1989)</td>
<td>Lodgepole and ponderosa pine</td>
<td>33.9</td>
<td>1070 - 1920</td>
<td>Clearcut</td>
<td>27</td>
<td>18</td>
<td>Monthly, 7-day low flows</td>
<td>None</td>
</tr>
<tr>
<td>Upper Penticton Creek, BC</td>
<td>(Winkler et al., 2017)</td>
<td>Lodgepole pine, Engelman spruce</td>
<td>4.9</td>
<td>1614 - 2021</td>
<td>Clearcut</td>
<td>47</td>
<td>7</td>
<td>Monthly</td>
<td>July (-17% not significant), Aug (-20% not significant), Sep (none), Oct (+45%)</td>
</tr>
<tr>
<td>Mica Creek Experimental Watershed - C1, Idaho</td>
<td>(Hubbart et al., 2007)</td>
<td>Mixed coniferous</td>
<td>1.4</td>
<td>1205 - 1528</td>
<td>Clearcut</td>
<td>48</td>
<td>4 (after road construction), 4 (after harvest)</td>
<td>July - Oct yield</td>
<td>Road construction (none), Harvest: (+5% - considered negligible)</td>
</tr>
<tr>
<td>Mica Creek Experimental Watershed - C2, Idaho</td>
<td>(Hubbart et al., 2007)</td>
<td>Mixed coniferous</td>
<td>1.8</td>
<td>1201 - 1612</td>
<td>Partial-cut</td>
<td>24</td>
<td>4 (after road construction), 4 (after harvest)</td>
<td>July - Oct yield</td>
<td>Road construction (+28%), Harvest (none).</td>
</tr>
<tr>
<td>Marmot Creek Research Basin - Cabin Creek, AB</td>
<td>(Swanson et al., 1986; Harder et al., 2015;)</td>
<td>Mixed coniferous - alpine</td>
<td>2.4</td>
<td>1700 - 2824</td>
<td>Clearcut</td>
<td>23</td>
<td>33</td>
<td>Monthly, Frequency of flows &lt; Q10¹</td>
<td>Aug (+4% not significant), Sep (+11% not significant), no change in frequency &lt;Q10</td>
</tr>
<tr>
<td>Marmot Creek Research Basin - Twin Creek, AB</td>
<td>(Harder et al., 2015)</td>
<td>Mixed coniferous - alpine</td>
<td>2.8</td>
<td>1700 - 2824</td>
<td>Honeycomb</td>
<td>17</td>
<td>42</td>
<td>Frequency of flows &lt; Q10</td>
<td>None</td>
</tr>
<tr>
<td>Wagon Wheel Gap, Colorado</td>
<td>(Van Haveren, 1988)</td>
<td>-</td>
<td>0.8</td>
<td>2818 - 3338</td>
<td>Clearcut</td>
<td>100</td>
<td>7</td>
<td>Annual 30 day low flow</td>
<td>None</td>
</tr>
<tr>
<td>Coon Creek, Wyoming</td>
<td>(Troendle et al., 2001)</td>
<td>Engleman spruce, subalpine fir, lodgepole pine</td>
<td>16.8</td>
<td>2682 - 3322</td>
<td>Clearcut</td>
<td>24</td>
<td>5</td>
<td>Monthly</td>
<td>Jul (-1% not significant), Aug (+14% not significant), Sep (+20% not significant)</td>
</tr>
</tbody>
</table>
Recent work has evaluated the hydrological effects of forest harvesting through time in eight rainfall-dominant and hybrid catchments in the HJ Andrew Research Forest of coastal Oregon (Perry and Jones, 2017). The sub-catchments have 40-50 years of post-harvest streamflow data. For the 5-10 years immediately following forest harvesting, the eight catchments typically showed an initial increase in summertime (July-Sept) water yield, followed by a gradual return to pre-harvest water yield. However, as forests regrew with time, summertime water yield fell below pre-harvest conditions. After a period of approximately 25–30 years, all five basins that were completely logged experienced a decrease in July-September water yield of around 30–50%. Basins that were not fully clearcut, or were subject to a different harvest method, tended to experience less pronounced effects on summer flows (Jones and Post, 2004; Perry and Jones, 2017). This longer term reduction in low flows has been attributed to the recovery of interception loss and increasing transpiration rates in younger, regenerating forests (Hicks et al., 1991; Moore et al., 2004).

The hydrological characteristics of the catchments of the HJ Andrews Research Forest are markedly different from snowmelt-dominant systems. Further work is required that specifically considers how forestry impacts low flows through time in snowmelt-dominant watersheds, and how these changes affect aquatic habitat.

1.4 Management of Summertime Low Flows

The summertime low flow period coincides with high demand of fresh water for purposes such as irrigation, power generation, and drinking water consumption (Bradford and Heinonen, 2008). Because the summertime low flow period is also a critical time for the aquatic ecology of these streams, there are often competing interests for the use of these water resources, necessitating their careful management. Resource management decisions that evaluate changes to the
streamflow regime during low-flow periods are often made by considering environmental flow needs (EFN), which is a framework that describes the quantity, timing, and quality of water necessary to support stream ecology and aquatic habitat (Jowett, 1997). Previous EFN assessments have mainly focused on the ecological impacts associated with hydrological alteration by dams and diversions (Tharme, 2003). Even though land-use changes that occur through forestry have considerable impacts on the natural flow regime, they have historically not been considered in the context of EFN. In BC, EFN assessments are legislated through the Environmental Flow Needs Policy (Water Sustainability Act, SBC 2014, Ch. 15) that requires that the EFN of streams are considered when water allocation decisions are made.

Various methods for determining the EFN of streams are discussed by Jowett (1997) and Tharme (2003). Generally, the most simple or basic approaches for setting EFN are considered ‘desktop,’ ‘historic,’ or ‘hydrological’ methods that determine streamflow thresholds for water withdrawals based on statistical analysis of available hydrological measurements. These methods implicitly assume that decreasing flows are detrimental to aquatic habitat (Jowett, 1997) – which is an assumption that most often holds true in small streams (Hatfield and Bruce, 2000). One of the earliest and most popular desktop methods, the ‘Tennant Method,’ sets EFN as a percentage of mean annual discharge (MAD) with different thresholds for summer and winter (Tennant, 1976). Local desktop methods specific to BC, such as The BC Modified Tennant Method and the BC Instream Flow Thresholds, were created to reflect the local variation of different hydrological regimes and aquatic habitats found in BC (Hatfield et al., 2003). While these desktop hydrological methods are simple and cost effective to execute, they are overly simplistic and may not reflect the many biological complexities of streams.
More complex assessments utilize ‘habitat’ or ‘eco-hydraulic’ approaches for setting EFN (Jowett, 1997; Tharme, 2003; Lancaster and Downes, 2010). In contrast to desktop approaches, eco-hydraulic methods generally involve a field campaign to characterize the hydraulic conditions of rivers. The intent of these approaches is to quantify available fish habitat at different flow levels using a measure of aquatic habitat called weighted usable area (WUA) (Payne, 2003). The output of these models is a relation between WUA and discharge. There are two main components for determining the WUA of a stream: (1) hydraulic data that characterize the distributions of depth, velocity, and substrate characteristics of the stream, and (2) biological data that describe the affected species’ preference for those hydraulic characteristics, known as habitat suitability curves.

The hydraulic distributions used to develop WUA curves are derived from a hydraulic model that describes how depths and velocities vary with streamflow. The habitat suitability curves, are statistical abundance-environmental relationships that describe how hydraulic characteristics (depth and velocity) correspond to observed fish densities (Ahmadi-Nedushan et al., 2006; de Kerckhove et al., 2008). Habitat suitability curves are usually structured so that the values that are assigned to the hydraulic parameters range from 0 (not suitable) to 1 (preferred habitat) and are determined for a range of values.

In North America, the most common habitat or eco-hydraulic method has historically been the physical habitat simulation (PHABSIM) model for many decades (Bovee et al., 1998). More recently, the System for Environmental Flow Analysis (SEFA), which differs from PHABSIM with only a few subtle differences to its hydraulic modelling routine, has become more common (Jowett et al., 2016). Although more contemporary approaches for setting EFN, such as the ecological limits of hydrological alteration (Poff et al., 2010), have recently been developed, they are slow to be adopted and eco-hydraulic approaches remain commonly applied by practitioners. This is likely due, at least in part, because the WUA
curves generated through eco-hydraulic methods are simple and powerful in an incremental decision-making context because they show how inferred habitat availability changes as a function of streamflow. Eco-hydraulic methods are also commonly viewed as being a comprehensive and quantitatively defensible way to make decisions about EFN (Tharme, 2003), even despite numerous systemic flaws (Mathur et al., 1985; Lancaster and Downes, 2010) and the common lack of validation of WUA curves (Railsback, 2016).

1.5 Study Objectives

The regulation of logging in forested watersheds often relies on simplified indices that only consider impacts to peak flows or annual water yields (Winkler and Boon, 2017). In the short term, the effect of logging on summertime low flows is often considered to be negligible or even favourable (i.e. higher summer flows) (Pike and Scherer, 2003), and so they are not often considered in a regulatory context. However, as reviewed in Section 1.3, there is evidence from Australia and coastal Oregon that summer low flows can become more severe in the medium-long term in association with forest growth (Vertessy et al., 2001; Perry and Jones, 2017). No previous studies have examined the medium to longer term effects of forestry on low flows in snowmelt-dominated catchments. Therefore, the objectives of this study were (1) to use a paired-catchment approach to quantify changes in summertime low flows following forest harvesting over two or more decades, and (2) to quantify the resulting changes in physical fish habitat.
2 Methods

2.1 Study Locations

This research focused on three snowmelt-dominant catchments located in the southern interior of BC. These catchments are detailed in the below sections.

2.1.1 Upper Penticton Creek

The Upper Penticton Creek watershed experiment includes three catchments located in the uplands east of Okanagan Lake: 240 Creek, 241 Creek and Dennis Creek (Figure 2.1). The catchments of 240 and 241 Creeks are located between elevations of 1,600–2,000 m, while Dennis Creek’s catchment lies between elevations of 1,700–2,140 m (Winkler et al., 2015). Catchment areas are 4.94 km², 4.50 km², and 3.73 km² for 240, 241, and Dennis Creeks, respectively. The catchments for 240 and 241 Creeks are south facing, while the Dennis Creek catchment is west facing.

The Upper Penticton Creek site is located within the dry Engelmann spruce subalpine fir biogeoclimatic zone. The Upper Penticton Creek catchment receives approximately 700 mm of mean annual precipitation, just over half of which falls as rain (Winkler et al., 2017). The 240 and 241 Creek catchments are predominantly covered with lodgepole pine stands, while Dennis Creek’s catchment is covered by a mix of Engelmann spruce (Picea engelmannii), lodgepole pine, and subalpine fir (Abies lasiocarpa) stands (Winkler et al., 2015). The soils in the Upper Penticton Creek area generally shallow, and consist of sandy loam and loamy sand that overlie glacial till and granitic rocks (Winkler, 2010; Winkler et al., 2015).
The Water Survey of Canada has maintained streamflow gauging stations at all three creeks since 1983. The presence of rainbow trout (*Oncorhynchus mykiss*) has been identified throughout the Upper Penticton Creek watershed through both visual observation during the field program for this study, as well as through previous observations catalogued by the BC Conservation Data Centre (2018).

In the Upper Penticton Creek experiment, 240 Creek was designated the control catchment, while the catchments of 241 Creek and Dennis Creek were subject to logging treatments. Although 240 Creek is the experimental control, salvage logging occurred over 4% of the catchment in 1992 following a windthrow event.
(Winkler et al., 2017). In 241 Creek, 47% of the catchment was logged between 1993 and 2007, and in Dennis Creek 52% of the catchment was logged between 1995 and 2000 (Winkler, personal communication, 2017; Figure 2.2). Logging primarily occurred over the lower portions of both the 241 Creek and Dennis Creek watersheds, and was conducted using skidders (Winkler et al., 2015, 2017).

![Graph showing cumulative clearcut area through time at 241 Creek and Dennis Creek.](image)

**Figure 2.2:** Cumulative clearcut area through time at 241 Creek and Dennis Creek.

### 2.1.2 Camp Creek

Camp Creek is a 35.9 km² watershed that drains the uplands west of Okanagan Lake near Summerland, BC. Elevations within the Camp Creek catchment range from 1,070 m to 1,920 m above sea level (Figure 2.3). Camp Creek is bordered to the northeast by the Greata Creek catchment, which has an area of 43.5 km² and an elevation range of 880 to 1,620 m above sea level, and has been used as a control to Camp Creek for previous paired catchment analyses (Cheng, 1989; Moore and Scott, 2005). Camp Creek is mostly south facing while Greata Creek is mostly east facing. Both watersheds are covered by a mixture of lodgepole pine,
with some ponderosa pine and Douglas-fir stands at lower elevations (Cheng, 1989). These watersheds receive an average of approximately 600 mm of precipitation annually, with over half falling as snow (Cheng, 1989). The geology of the Camp and Greata Creek catchments consists of granitic rocks overlain by a thin layer of glacial till and colluvial soils (Cheng, 1989; Moore and Scott, 2005). The Water Survey of Canada has maintained streamflow gauging stations since 1965 for Camp Creek and 1970 for Greata Creek. The presence of rainbow trout has been identified in this watershed through both visual observation during the field program for this study, as well as through previous observations catalogued by the BC Conservation Data Centre (2018).

Figure 2.3: Camp Creek study site.
Between 1976 and 1982, 27% of Camp Creek was logged following a mountain pine beetle infestation (Moore and Scott, 2005). Analysis of satellite imagery between 1984-2011 indicates that further harvesting occurred through the 1980s and early 1990s, bringing the cumulative harvested area up to 39% by 1991. Since the early 1990s, minor intermittent logging has occurred within the catchment, bringing the cumulative harvest up to 45% by the 2011 (Figure 2.4). In the Greata Creek catchment, previous work indicates that the logging occurred over about 9% of the catchment area as of 2004 (Moore and Scott, 2005). However, a more detailed analysis of normalized difference vegetation index images and satellite imagery indicates that as of 2011, approximately 7% of the Greata Creek catchment had been logged. After 2011, the intensity of logging increased significantly, with over 10% of the catchment being logged by 2016. Of note, because logging occurred within Greata Creek over a period of approximately 40-years some hydrological recovery has likely occurred as replanted trees have matured. Further analysis of the logging history of Greata Creek is detailed in Appendix A. In addition to impacts from logging, analysis of aerial imagery indicates that 4.5% of the Greata Creek catchment was killed due to a mountain pine beetle infestation during the summer of 2007. Due to the mountain pine beetle infestation in 2007 onwards, Greata Creek can only be used as a suitable control catchment up until this time.

Further to the logging history of the Greata Creek catchment, historical water storage management in the Glen Lake reservoir, which is in the upper part of the catchment, may have affected the hydrology of Greata Creek. Historically, the district of Peachland maintained a 2-3 tall earthen dam on Glen Lake to support the storage of approximately 300,000 m$^3$ of water for domestic use; however, this earthen dam was decommissioned in 1996 (Golder, 2010). The change in water storage in the upper part of this catchment following the removal of this dam may have altered the streamflow regime of this catchment.
2.2 Paired Catchment Analysis

The paired catchment approach is a quasi-experimental design method that is widely used for detecting changes in streamflow following landscape disturbances (Bosch and Hewlett, 1982; Brown et al., 2005). This method involves fitting a statistical relationship between streamflow indices for two different catchments during an initial calibration period in which both catchments remain undisturbed. Then, one catchment undergoes a treatment (e.g. logging), while the second catchment remains unaltered and is used as an experimental control. Using the statistical model generated from the calibration period, a time series of predicted streamflow is generated that represents what streamflow would have been had logging not occurred. This simulated time series is compared to the observed streamflow time series following logging. The treatment effect represents the magnitude of the change in streamflow that can be attributed to logging. The significance of the treatment effect is typically assessed using analysis of covariance (ANCOVA). This approach is considered the most statistically rigorous.
way to detecting changes in hydrology, although its success is largely contingent on how similar the control and treatment catchments are in terms of geology, soils, topography, and vegetation (Moore and Wondzell, 2005; Zégre et al., 2010). This approach may also be limited in scenarios when it is used to assess changes in hydrology that occur during periods when there are novel climatic or hydrological conditions that were not captured by the calibration period.

The temporal evolution of changes in streamflow are of particular interest for this study. Therefore, the post-harvest time series of streamflow data was split into separate treatment periods (Table 2.1), and each treatment period was analyzed using a separate ANCOVA. Grouping data into appropriate treatment periods was done by considering the major milestones in the logging history of these watersheds alongside the selection of periods long enough to have adequate statistical power to detect changes in streamflow.

Table 2.1: Summary of treatment periods used in ANCOVA.

<table>
<thead>
<tr>
<th>Period</th>
<th>241 Creek</th>
<th>Dennis Creek</th>
<th>Camp Creek</th>
</tr>
</thead>
<tbody>
<tr>
<td>5</td>
<td>-</td>
<td>-</td>
<td>1998 - 2007 post-logging¹</td>
</tr>
</tbody>
</table>

1. Some minor intermittent logging (~5% watershed area) occurred in Camp Creek between 1990 and 2011 (see Figure 2.4).

For this study, the paired catchment method was conducted using two separate sets of streamflow metrics: firstly, by each day-of-year, and secondly for the seasonal water yield from the July 1–September 30 low-flow period. These analyses are described below.
2.2.1 Day-of-year analysis

To generate an understanding of the effect of logging on the seasonal timing and volume of streamflow, the paired catchment analysis was conducted separately for each day-of-year between April 1 and October 31. This approach differs from many previous studies that have commonly assessed changes in streamflow at a more coarse, monthly time step (e.g. Moore and Scott, 2005; Winkler et al., 2017, among others). To execute this analysis, the daily streamflow data for the control and treatment catchments were extracted from the complete dataset for all years for a given date (i.e. all mean daily streamflow records for a particular date were subset and analyzed independent of other dates), and a paired-catchment analysis was performed. For example, mean daily streamflow records for July 1st were analyzed independently from July 2nd, and so on. Change detection analysis was conducted for each period using linear regression and ANCOVA. Because each observation is separated by an entire calendar year, this approach avoids issues with temporal auto-correlation that often complicate paired catchment analyses conducted using a sub-annual time step (Zégre et al., 2010).

The ANCOVA was conducted following the procedure outlined by Kunter et al. (2005) and Moore and Scott (2005) using the `lm()` function in the R programming language (R Core Team, 2018). Prior to analysis, the streamflow data were log-transformed to reduce nonlinearity and heteroscedasticity. Data from the calibration period and each treatment period were extracted from the complete time series, and two different regression models were fit to these data. These regression models included a categorical ‘dummy’ variable ($\gamma_t$) to represent the effect of the treatment period, coded as follows:

$$\gamma(t) = \begin{cases} 0, & \text{calibration period} \\ 1, & \text{treatment period} \end{cases}$$

The full form of the regression models fit to the data were expressed as follows:

$$\hat{y}_t = b_0 + b_1 x_t + b_2 \gamma_t + b_3 x_t \gamma_t$$

(1)
where \( y_t \) and \( x_t \) represent the predicted streamflow for the treatment catchment and the observed streamflow from the control catchments, respectively, in year \( t \), and \( b_o, b_1, b_2, \) and \( b_3 \) are the fitted regression coefficients. During the calibration period, the \( y_t \) term equals 0, and equation (1) was reduced to the following form:

\[
\hat{y}_t = b_o + b_1 x_t
\]  

(2)

The ANCOVA tests the null hypothesis that there is no effect of logging on streamflow. This hypothesis can be expressed as follows:

\[
H_0: \beta_2 = \beta_3 = 0
\]

The alternative hypothesis is thus

\[
H_1: \beta_2 \text{ and/or } \beta_3 \neq 0
\]

These hypotheses were evaluated by calculating a partial F statistic (\( F^* \)):

\[
F^* = \frac{((SSE_1 - SSE_2)/(df_1 - df_2))/MSE_2}{MSE_2}
\]  

(3)

where \( SSE_1 \) and \( SSE_2 \) are the sum of squared errors for models 1 and 2, \( df_1 \) and \( df_2 \) are the degrees of freedom for models 1 and 2, and \( MSE_2 \) represents the mean squared error of model 2. Considering the small sample sizes and the exploratory nature of the study, a \( p \)-value of 0.10 was used to identify potential significance of \( F^* \).

Because streamflow data were log-transformed, regression relations and ANCOVA results were based on log-transformed data. When predictions based on log-transformed values are transformed back to the original units, they will contain a negative bias and will tend to be lower than they should. To account for this, the bias correction presented by Baskerville (1972) was applied:

\[
\hat{y} = 10^{\left(\hat{\mu} + \frac{\hat{\sigma}^2}{2}\right)}
\]  

(4)

where \( \hat{\sigma}^2 \) is the estimated variance and \( \hat{\mu} \) is the value predicted from the regression model in log-transformed units.
2.2.2 Low-flow yield

The low-flow yield was calculated by integrating the total volume of streamflow over the period from July 1 to September 30, and dividing by the catchment area to generate the water yield in mm. The July 1 – September 30 period has been used in other studies which have evaluated summertime low flows in the Pacific Northwest (Perry and Jones, 2017). Changes in low-flow yield were assessed separately for each treatment period following the ANCOVA approach outlined in Section 2.2.1, with the exception that the data did not require log-transformation.

2.3 Instream Flow Assessment

The instream flow assessments involved the execution of an eco-hydraulic model to develop a relation between streamflow and fish habitat, represented by WUA curves. The steps taken to develop these relations are detailed below.

2.3.1 Field Methods

The field campaign for this study was carried out over the summer of 2017. Four separate field trips were conducted at each stream between the middle of June and the end of September; the activities conducted during these field trips are summarized in Table 2.2.
The first component of the field program was to establish study reaches for each of the three study streams (Figure 2.5). The lengths of the study reaches were 94 m, 50 m, and 146 m, for 241 Creek, Dennis Creek, and Camp Creek, respectively. The locations of the study reaches were chosen near the Water Survey of Canada streamflow gauging stations so that the results from the instream flow assessment could be coupled with the time series of measured discharge from each stream for further analysis. After choosing each study reach, the length of mesohabitats (e.g. pools, riffles, cascades) contained within each reach were mapped according to the criteria shown in Table 2.3 (Johnston and Slaney, 1996). Some deviations from these classification schemes occurred for reaches with complex morphologies (e.g. large amounts of in-stream wood or multi-thread
channels). Then, a series of transects were established using a stratified random approach within the different mesohabitats. Transects were marked by placing a benchmark (a labelled wooden survey stake) above the bankfull elevation on both sides of the stream so that the transect was perpendicular to the thalweg. Finally, mesohabitat classifications were repeated for each field trip in case mesohabitats changed with streamflow (e.g. a glide at higher flows may become a riffle as flow declines). Where mesohabitat classifications varied with streamflow, the classifications that best represented low flow conditions were ultimately used for the instream flow analysis.

<table>
<thead>
<tr>
<th>Mesohabitat</th>
<th>Characteristics</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pool</td>
<td>Relatively slow velocities, deeper water with concave bed profile, low gradient, often impounded with a back-water effect</td>
</tr>
<tr>
<td>Glide</td>
<td>Relatively high velocity, non-turbulent flow, flat bed profile</td>
</tr>
<tr>
<td>Riffle</td>
<td>Relatively high velocity, turbulent flow, shallow, moderate gradient, gravel and cobble substrates. Often bed materials project above the water surface.</td>
</tr>
<tr>
<td>Cascade</td>
<td>Steep, stepped riffles that occur around bedrock or cobbles. Gradients are typically &gt; 4%.</td>
</tr>
</tbody>
</table>
Figure 2.5 Instream flow assessment site plans
As a part of the instream flow assessment, depth and velocity measurements were required from one ‘survey flow,’ which is preferably when discharge is greater than 40% MAD (Lewis et al., 2004). Depth and velocity measurements were made at regularly spaced intervals along each transect for all surveys using a Marsh-McBirney current meter attached to a top-set rod. Measurements were taken at either 20 cm increments across the transect, or at a spacing that allowed for approximately 20 different vertical measurements across the wetted width (Figure 2.6).

![Figure 2.6: Depth and velocity measurements at Camp Creek, June 18.](image)

Channel substrate was characterized during August field trips for each stream. The relative percentages of different bed materials were visually estimated within a 10 cm radius of each depth/velocity measurement point (i.e. halfway between measurement points) using the Wentworth classification scheme detailed by Lewis et al. (2004).

Water surface elevations at two other ‘calibration flows’ were measured relative to the benchmarks along each transect during subsequent field trips. For the first
two field trips, water surface elevations were measured using a digital range-finder that quantifies vertical distances using measurements of slope-distance and the angle of inclination to calculate vertical distances. Water surface elevations for the third and fourth field trip, as well as the topographic survey conducted during the third field trip, were measured using a survey level and stadia rod.

2.3.2 Analytical Methods

The WUA curves were simulated for flows from 0 to 100% mean annual discharge (MAD). Streamflows above 100% MAD were not modelled in this study because the maximum discharges recorded during transect measurements were about half that value and extrapolating to higher flows would introduce increasing uncertainty into the analyses. As previously indicated, deriving the WUA of a stream involves combining both a hydraulic model and a biological model, both of which are discussed separately, below.

Hydraulic Modelling Component

For this study, the WUA analyses were conducted using the ‘habitat mapping’ method within the System for Environmental Flow Analysis (SEFA) software package (Jowett et al., 2016). The first step in this modelling approach was to use the water surface elevation (stage) measurements collected in the field for each transect along with discharge reported by the Water Survey of Canada to develop stage-discharge rating curves specific to each transect. For each transect, the stage was determined relative to the benchmark installed on the right bank, each of which was assigned an arbitrary elevation of 10.000 m. The stage for each survey was calculated as the difference in elevation between the water surface elevation and the right bank benchmark, as measured in the field. Discharge was calculated at each stream for each survey by averaging the reported Water Survey
of Canada discharge values that coincided with the period when the survey was conducted. Whereas the stage measurements were specific to each transect, the same discharge measurements from the Water Survey of Canada were used to develop the rating curves for all transects within each stream. In addition to the stage/discharge pairs collected during the different surveys, the lowest elevation of each transect (i.e. the stage of zero flow, \([SZF]\)) was also used to create the rating curves.

Within SEFA, power-law rating curves were computed by fitting a linear regression to log-transformed stage and discharge \((Q)\) data (Jowett et al., 2016). The fitted regression equations exhibited the following form:

\[
\log(Q) = \log(a) + b \log(\text{Stage} - \text{SZF})
\]

(5)

where \(a\) and \(b\) are regression coefficients. By taking the antilog of both sides of equation 5, the rating curves were back-transformed to the original units, and were expressed as a power-law relationship with the following form:

\[
Q = a (\text{Stage} - \text{SZF})^b
\]

(6)

Using these rating curves, SEFA was then used to simulate depths for each measurement point along each transect for discharges in increments of 1.0 L/s.

Following the prediction of depth profiles using these rating curves, the default hydraulic settings within SEFA were used to model velocities (see Jowett et al., 2016). The first step in this process was to reproduce the velocities measured during the ‘survey flow’ (the Water Survey of Canada discharge that coincided with the field trip where depth/velocity measurements were taken) for every measurement points along each transect. This was done using Manning’s equation and conveyance at each measurement point. Manning’s equation is an empirically derived equation that estimates velocity \((v)\) as a function of hydraulic radius \((R)\), channel gradient \((S)\), and Manning’s \(n\) – a metric of channel roughness.
(Henderson, 1966). When conducting measurements using the metric system, Manning’s equation can be expressed as follows:

\[ v = \frac{\frac{2}{R^3} \frac{1}{S^2}}{n} \]  \hspace{1cm} (7)

Meanwhile, channel conveyance \( (K) \) can be represented as follows:

\[ K = \frac{Q}{\frac{1}{T} \cdot S^2} \]  \hspace{1cm} (8)

Stream discharge can be represented using the continuity equation:

\[ Q = v \cdot A \]  \hspace{1cm} (9)

where \( A \) represents the cross-sectional area of the transect (m\(^2\)). By substituting equations 7 and 8 into equation 9, the conveyance of a cross-section can be represented using the following formula, so that it is independent of channel gradient:

\[ K = \frac{A \cdot \frac{2}{R^3}}{n} \]  \hspace{1cm} (10)

Henderson (1966) showed that, when stream cross-sections are subdivided into parts (denoted by subscript \( i \)), the ratio of discharge of a cell to the conveyance of the same cell is equivalent to the ratio of total cross section discharge to cross-sectional conveyance:

\[ \frac{Q_i}{K_i} = \frac{Q}{K} \]  \hspace{1cm} (11)

By substituting equations 8 and 11 into equation 7, it is possible to calculate the velocity at any given measurement point using the following relationship:

\[ v_i = \frac{\frac{2}{R_i^3} \cdot Q}{n_i \cdot \frac{1}{K}} \]  \hspace{1cm} (12)
For each measurement point \( i \), Manning’s \( n \) values were automatically adjusted as an empirical ‘best-fit’ so the calculated velocity \( (v_i) \) matched the velocity measurements collected in the field. Of note, the default hydraulic model used within SEFA maintains constant Manning’s \( n \) values at different flows. Then, by substituting equation 10 into equation 12, velocity can be calculated as follows:

\[
v_i = \frac{2}{n_i} \cdot \frac{Q \cdot n}{K \cdot R_i^3} \quad (13)
\]

Based on equation 13, and assuming that Manning’s \( n \) is uniform across the entire transect, the velocity at each point varies with \( R_i^2 \), which is related to depth. Considering this observation, and assuming \( n_i = n \), the predicted velocity at point \( i \) \( (v_i) \) can be calculated using a simplified version of equation 14:

\[
v_i = \frac{2}{R_i^2} \cdot \frac{Q}{K \cdot R_i^3} \quad (14)
\]

In reality, the roughness across a transect is not likely to be uniform, so \( n_i \neq n \), and the measured velocity \( (v) \), will differ from that of \( v_i \). To compensate for this, SEFA calculates a velocity distribution factor \( (VDF) \) as the ratio of measured to predicted velocities, specific to the survey flow.

\[
VDF = \frac{v_i}{v} \quad (15)
\]

Using the \( VDF \) values for each measurement point, the velocities were predicted for the range of modeled discharges using the following equation.

\[
v_i = VDF_i \cdot \frac{2}{R_i^2} \cdot \frac{Q}{A \cdot R_i^3} \quad (16)
\]

The transects used for the instream flow assessment were selected to sample the hydraulics of the stream, not to calculate discharge. Therefore, the hydraulic characteristics of transects were often comprised of turbulent flow and highly
variable bed profiles: conditions that likely introduce substantial error when measurements from these transects are used to calculate discharge. Thus, estimates of discharge derived from these field measurements are expected to vary considerably when compared to the survey flow. However, the velocities estimated by equation 16 were modelled using the survey flow, and were, therefore, likely not accurate. To address these potential errors SEFA applies a velocity adjustment factor (VAF) that is specific to each transect. The VAF values are calculated by taking the ratio of calculated discharge using the measured velocities ($Q_m$) for each transect to the reach-averaged survey flow ($Q$):

$$VAF = \frac{Q_m}{Q}$$

(17)

Using these VAFs that are specific for each transect, SEFA then adjusted the velocities that were modelled by equation 16.

**Biological Modelling Component**

Following the execution of the SEFA hydraulic model, the habitat suitability curves (Figure 2.7) were applied to simulated depths and velocities. Habitat suitability curves derived by Ron Ptolemy (BC Ministry of Environment), specific to rainbow trout fry and parr rearing habitat in BC, were used to determine the suitability of depth, velocity, and substrate measurements for all three sites. These suitability curves are designed for use as relative indices and range from 0 (not suitable habitat) to 1 (preferred habitat).
Adult rainbow trout approximately 20 cm in length were visually observed in both 241 Creek and Dennis Creek during the second field trip (Figure 2.8). However, suitability curves for adult trout were not applied for this study because the depths observed during this study were particularly low, and when coupled with the adult rainbow trout curves, they would produce negligible habitat values. See Section 4.2 for further discussion about issues using adult-specific rainbow trout suitability curves to model WUA in these streams.
The habitat suitability for each measurement point along a transect was represented by its composite suitability index ($CSI$) which was calculated as follows:

$$CSI = HSI_d \cdot HSI_v \cdot HSI_s$$  \hspace{1cm} (18)

where $HSI$ denotes the habitat suitability indices for depth, velocity, and substrate characteristics, respectively. The average $CSI$ for each transect was then multiplied by the modelled width of that transect for that flow to determine its weighted usable width. Finally, the weighted usable widths for all transects were aggregated using the relative proportions of habitat which they represent, determined by the mesohabitat classifications, to produce a reach-averaged $WUA$ measurement. By conducting this process for a series of different flows, a $WUA$ curve was created for each stream, for both rainbow trout fry and rainbow trout parr.

Using the stratified random sampling approach outlined above, confidence intervals were generated around the $WUA$ curves using 2,000 bootstrapping simulations, executed within the SEFA software package (Jowett et al., 2016). For each bootstrap replicate, transects were selected randomly with replacement for each mesohabitat, and confidence intervals were created using the percentile approach (e.g. for 2,000 boot-strapping runs, the $50^{th}$ and $1950^{th}$ ranked values were used to generate the 95% confidence limits).

It is important to note that the confidence intervals generated through this bootstrapping routine do not represent the true uncertainty inherent in the instream flow assessment. Instead, these confidence intervals only represent the uncertainty of the $WUA$ curves due to the variation in the hydraulics of each transect, given the habitat suitability curves that are applied. These confidence intervals do not represent uncertainty or error associated with the habitat suitability curves (Williams, 2009).
2.4 Effects of logging on fish habitat

Logging-induced changes in fish habitat availability were quantified by using the WUA curves derived using SEFA to compute habitat availability on a daily basis using both the observed and predicted streamflow from the daily paired-catchment analysis. These values of WUA were plotted as time series, and differences were considered to be present on days when the following two criteria were met: 1) the treatment effect for streamflow on that particular day was significant and 2) there was a difference in WUA that exceeded the average width of the confidence intervals around the relevant WUA curve for the range of flows experienced at that location.
3 Results

3.1 Day-of-Year Paired Catchment Analysis

3.1.1 241 Creek

The pre-harvest regression for 241 Creek explained from about 60% to over 90% of the variance for log-transformed streamflow (Figure 3.1). High $R^2$ values in May and June indicate that this model was a good fit through the seasonal peak flows, likely because the catchments for 240 Creek (the control catchment) and 241 Creek have similar aspects and elevation ranges, and thus similar snowmelt timing. The quality of the model fit was weakest in July ($R^2 \sim 60\%$); however, the fit improved throughout the remainder of the summertime low flow period ($R^2 \sim 80\%$).

![Figure 3.1: Daily coefficient of determination of the pre-harvest regression for 241 Creek.](image)

Figure 3.2 displays the results of the ANCOVA analysis. From 1993–2000 there were only a few days for which treatment effects were statistically significant. In contrast, from 2001–2007 there was an approximately two-week period from the end of May to the beginning of June when statistically significant reductions in streamflow were evident. Detectable changes in streamflow were most apparent in the final period, from 2008–2017. During much of April and May there were...
50–100% increases in streamflow which were significant, and from the start of June to the beginning of July, 20-30% reductions in streamflow were also significant.

Figure 3.2: Day-of-year ANCOVA results for 241 Creek comparing each treatment period (2, 3, 4) to the pre-harvest period (1): the panels alternate between the p-value (blue line) and the ratio of observed to predicted streamflow (grey circles and red line). Y-axis is logarithmic.
Hydrographs of predicted and observed streamflow for each treatment period are shown in Figure 3.3. During the periods from 1993–2000 and 2001-2007, the predicted and observed hydrographs appear similar from April through the middle of July, followed by increased variability between the predicted and observed flows through the end of September. In 2008-2017, the observed streamflow peak and the receding limb of the hydrograph shifted spring-ward relative to the predicted hydrograph. This spring-ward shift persisted until the beginning of September.

Figure 3.3: Predicted and observed mean hydrographs by treatment period at 241 Creek with a logarithmic y-axis.

3.1.2 Dennis Creek

The pre-harvest regression for Dennis Creek explained from about 50% to over 90% of the variance for log-transformed streamflow, except for a brief period at the beginning of June when the regression explained less than 50% of the variance (Figure 3.4). Unlike the case for 241 Creek, low $R^2$ values in May and June indicate that this model fit poorly during the seasonal peak flows, likely because Dennis Creek and 240 Creek have different aspects and slightly different elevation ranges, resulting in differences in snowmelt timing. During the low flow period, the $R^2$ ranged from approximately 50 – 80%.
Figure 3.4: Daily coefficient of determination of the pre-harvest regression for Dennis Creek.

Figure 3.5 displays the results of the ANCOVA analysis. During the first treatment period from 1995–2000, the ratio of predicted to observed streamflows varied around the value of 1. Except for an isolated occurrence in October, the predicted and observed streamflows from this period were not statistically different. More pronounced trends in streamflow began to appear in the second treatment period, from 2001–2008. During that period, there was a statistically significant increase (>100%) in flows throughout May. There were also a few, sporadic incidents of statistically significant changes in streamflow throughout the remainder of the year, but these did not last for more than a few consecutive days. The most pronounced changes in summertime low flows occurred in the final treatment period, from 2009–2017. From the beginning of July to the middle of October, there were extended periods in which reductions in streamflow (30–60%) were statistically significant. In addition, the increase in May flows seen in previous periods was more subdued in magnitude and significant for a shorter period.
Figure 3.5: Day-of-year ANCOVA results for Dennis Creek comparing each treatment period (2, 3, 4) to the pre-harvest period (1): the panels alternate between the p-value (blue line) and the ratio of observed to predicted streamflow (grey circles and red line). Y-axis is logarithmic.

Hydrographs of predicted and observed streamflow are plotted for each treatment period in Figure 3.6. During the 1995-2000 period, the predicted and observed streamflows were similar. In contrast, from 2001–2008 there was a spring-ward shift in both the springtime peak-flows and the receding limb of the hydrograph
that persisted until around the end of August. The most evident differences between the predicted and observed hydrographs occurred from 2009–2017. During that period, the spring-ward shift in the peak flow and the receding limb of the hydrograph were comparable to the previous period, from 2001-2008. However, unlike the response in the previous period, the difference between observed and predicted streamflow intensified from the beginning of July through until the middle of October.

**Figure 3.6: Predicted and observed mean hydrographs by treatment period at Dennis Creek with a logarithmic y-axis.**

### 3.1.3 Camp Creek

The pre-harvest regression for Camp Creek explained between 25% and 90% of the variance of log-transformed streamflow from April until the end of May (Figure 3.7). The relatively poor fit during the spring likely occurred because the Camp and Greata Creek catchments have different aspects, resulting in differences in snowmelt timing. The regression fits were better from the middle of June to the middle of August, often explaining well over 90% of the variance in streamflow. After the middle of August, the regression model typically explained over 80% of the variance, except for certain days when the fit was poor, and the $R^2$ dropped dramatically. These drops in the $R^2$ likely occurred because only six years of pre-harvest data were available to fit the regression equation, so
single days with atypical streamflow data (e.g. due to rain storms) could have had a strong influence on the model fit.

![Graph showing R^2 values over time](image)

**Figure 3.7: Daily coefficient of determination of the pre-harvest regression for Camp Creek.**

Figure 3.8 displays the results of the ANCOVA analysis. During the first treatment period from 1977–1982, streamflow was significantly increased by 50% to over 100% for a 10-day period at the start of May and by approximately 50% for a two-week period near the end of August. During the second period, from 1983–1990, streamflow increased anywhere from around 50 to over 100% for approximately two weeks at the end of April. This increase in April/May streamflow was also apparent in the subsequent treatment periods; however, as time progressed, the magnitude of the increase and the length of time they were statistically significant declined. During the 1983-1990 and 1991-1997 periods, the ratio of predicted to observed streamflows varied around the value of 1 with few instances where there was a significant difference from the middle of May onward. In contrast, statistically significant reductions in streamflow during the low flow portion of the hydrograph were more apparent in the last two treatment periods. During the 1998–2007 period flows were significantly reduced by up to 46% for most days from the end of June until the end of September.
Figure 3.8: Day-of-year ANCOVA results for Camp Creek comparing each treatment period (2, 3, 4) to the pre-harvest period (1): the panels alternate between the p-value (blue line) and the ratio of observed to predicted streamflow (grey circles and red line). Y-axis is logarithmic.
Hydrographs of predicted and observed streamflow are plotted for each treatment period in Figure 3.9. During the first three treatment periods, the rising limb of the hydrograph shifted spring-ward, but the timing of peak flows and the post-peak flows were similar. In the final period, from 1998–2007, the observed streamflow was consistently below the predicted streamflow from the beginning of June until the end of September.

![Figure 3.9: Predicted and observed hydrographs by treatment period at Camp Creek with a logarithmic y-axis.](image)

3.2 Low-Flow Yield Paired Catchment Analysis

3.2.1 241 Creek

The pre-harvest regression for the low-flow yield had an $R^2$ of 0.90. The yields during the 1993-2000 treatment period were highly variable, with two points plotting above and one point plotting below the prediction intervals (Figure 3.10). During the 2008-2017 treatment period, the points generally fell on or below the best-fit line, with three points plotting near the bottom 90% prediction interval (Figure 3.10).
The ANCOVA results show insignificant changes in yield during the 1993-2000 and 2001-2007 periods (Table 3.1). In contrast, there was a significant decrease in yield during the 2008–2017 period (-16%) (Table 3.1).

### Table 3.1: ANCOVA results for low-flow yield at 241 Creek.

<table>
<thead>
<tr>
<th>Treatment Period</th>
<th>ANCOVA p-value</th>
<th>Treatment effect(^1) (% change in yield)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1993 - 2000</td>
<td>0.51</td>
<td>7</td>
</tr>
<tr>
<td>2001 - 2007</td>
<td>0.26</td>
<td>-4</td>
</tr>
<tr>
<td>2008 - 2017</td>
<td>0.047</td>
<td>-16</td>
</tr>
</tbody>
</table>

1. Represents the mean change over the treatment period

### 3.2.2 Dennis Creek

The pre-harvest regression for the low-flow yield had an \( R^2 \) of 0.88. The yields during the 1995-2000 and 2001-2008 periods varied around the best-fit line, with one point from the 1995-2000 period plotting below the lower prediction interval (Figure 3.11). From 2009-2017, all the data points plotted below the best-fit line,
with one plotting below the lower prediction interval (Figure 3.11). The ANCOVA results show insignificant changes in yield during the 1995-2000 and 2001-2008 periods (Table 3.2). In contrast, there was a significant decrease in yield (−43%) during the 2009-2017 period (Table 3.2). For that period, all the data points plotted below the best-fit line, with two below the lower prediction limit (Figure 3.11).

![Figure 3.11: Scatterplot of low-flow yields, Dennis Creek against 240 Creek.](image)

Table 3.2: ANCOVA results for low-flow yield at Dennis Creek.

<table>
<thead>
<tr>
<th>Treatment Period</th>
<th>ANCOVA p-value</th>
<th>Treatment effect(^1) (% change in yield)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1995 - 2000</td>
<td>0.55</td>
<td>-11</td>
</tr>
<tr>
<td>2001 - 2008</td>
<td>0.27</td>
<td>-24</td>
</tr>
<tr>
<td>2009 - 2017</td>
<td>0.003</td>
<td>-43</td>
</tr>
</tbody>
</table>

1. Represents the mean change over the treatment period

3.2.3 Camp Creek

The pre-harvest regression had an \( R^2 \) of 0.93. During the 1976-1989 period the low flow yields in Camp Creek were highly variable, with three points plotting
above and one point plotting below the 90\% prediction intervals (Figure 3.12). During the 1977-1982 period all but one point plotted above the best-fit line and two points plotted above the upper prediction interval. The results from the ANCOVA confirm that there was a significant 36\% increase in yield during this period (Table 3.3). Data points from the 1983-1990 and 1991-1997 periods vary around the best-fit line, although there is more variability in the latter period, with one point plotting above and another plotting below the prediction intervals (Figure 3.12). There were insignificant changes to the low flow yields for both these periods (Table 3.3). From 1998-2004 and 2005-2011, the data plot on or below the best-fit line, with one point from the 1998-2004 period plotting on the lower prediction interval and one point from the 2005-2011 period plotting below the lower prediction interval (Figure 3.12). The results for the ANCOVA confirm that significant reductions in yield occurred for the final two treatment periods (1998-2004 and 2005-2011), when yields were reduced by 18\% and 20\% of the average predicted values, respectively (Table 3.3).

Figure 3.12: Scatterplot of low-flow yields for Camp Creek against Greata Creek.
### Table 3.3: ANCOVA results for low-flow yield at Camp Creek.

<table>
<thead>
<tr>
<th>Treatment Period</th>
<th>ANCOVA p-value</th>
<th>Treatment effect (Δ%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1977 - 1982</td>
<td>0.04</td>
<td>36</td>
</tr>
<tr>
<td>1983 - 1990</td>
<td>0.80</td>
<td>8</td>
</tr>
<tr>
<td>1991 - 1997</td>
<td>0.83</td>
<td>3</td>
</tr>
<tr>
<td>1998 - 2007</td>
<td>0.008</td>
<td>-18</td>
</tr>
</tbody>
</table>

#### 3.3 Instream Flow Assessments

Discharge in 241 Creek and Dennis Creek were highest in the spring and both declined to zero-flow conditions during August; meanwhile, discharge in Camp Creek was more sustained throughout all field trips (Table 3.4).

### Table 3.4: Mean discharges as recorded by the Water Survey of Canada during the instream flow assessment field trips.

<table>
<thead>
<tr>
<th>Location</th>
<th>Date</th>
<th>Q (L/s)</th>
<th>Q (% MAD)</th>
</tr>
</thead>
<tbody>
<tr>
<td>241 Creek</td>
<td>25-26 Jun</td>
<td>22</td>
<td>40</td>
</tr>
<tr>
<td></td>
<td>18-Jul</td>
<td>5</td>
<td>8</td>
</tr>
<tr>
<td></td>
<td>21-22 Aug</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>29-Sep</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Dennis Creek</td>
<td>26-27 Jun</td>
<td>25</td>
<td>45</td>
</tr>
<tr>
<td></td>
<td>19-Jul</td>
<td>2</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>23-24-Aug</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td></td>
<td>29-Sep</td>
<td>0.2</td>
<td>0</td>
</tr>
<tr>
<td>Camp Creek</td>
<td>25-26-Aug</td>
<td>44</td>
<td>30</td>
</tr>
<tr>
<td></td>
<td>29-30-Sep</td>
<td>39</td>
<td>27</td>
</tr>
</tbody>
</table>

The different mesohabitats identified within the study reach and the number of transects that were used to characterize each mesohabitat are summarized in Table 3.5 and displayed on Figure 2.5.
Table 3.5: Mesohabitat classifications used in the instream flow assessments.

<table>
<thead>
<tr>
<th>Location</th>
<th>Mesohabitat Classification</th>
<th>No. Transects</th>
<th>Proportion of Reach (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>241 Creek</td>
<td>Riffle</td>
<td>6</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>Complex riffle</td>
<td>5</td>
<td>15</td>
</tr>
<tr>
<td></td>
<td>Glide</td>
<td>1</td>
<td>16</td>
</tr>
<tr>
<td></td>
<td>Complex glide</td>
<td>6</td>
<td>29</td>
</tr>
<tr>
<td></td>
<td>Pool</td>
<td>8</td>
<td>23</td>
</tr>
<tr>
<td>Dennis Creek</td>
<td>Riffle</td>
<td>9</td>
<td>34</td>
</tr>
<tr>
<td></td>
<td>Glide</td>
<td>6</td>
<td>47</td>
</tr>
<tr>
<td></td>
<td>Pool</td>
<td>5</td>
<td>19</td>
</tr>
<tr>
<td>Camp Creek</td>
<td>Glide</td>
<td>9</td>
<td>40</td>
</tr>
<tr>
<td></td>
<td>Complex Glide</td>
<td>1</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>Cascade</td>
<td>8</td>
<td>53</td>
</tr>
<tr>
<td></td>
<td>Complex Cascade</td>
<td>2</td>
<td>2</td>
</tr>
</tbody>
</table>

For all three streams, there is relatively more fry habitat at a given discharge than there is parr habitat (Figure 3.13). Changes in fry habitat are typically most sensitive at the lowest flows, and gradually become less sensitive with increasing flow until peak habitats occur anywhere from 50-80\% MAD. At 241 Creek and Dennis Creek, the shape of the WUA curves for fry have a logarithmic profile until the peak WUA value is reached. The WUA curve for fry at Camp Creek deviates from this logarithmic shape, as it shows relatively consistent changes in habitat between streamflows of 10-50\% MAD before levelling off to a peak around 70\% MAD. In contrast to the fry curves, the WUA curves for parr steadily increase over the range of flows that was modelled for all three streams. The confidence limits generated around the WUA curves indicate that, given the habitat suitability curves used, there is relatively high certainty in the hydraulic component of the WUA curves for both fry and parr in all three streams.
Figure 3.13: WUA curves for all three study creeks.

3.4 Effects of Logging on Fish Habitat

Figure 3.17 – 3.19 displays the results generated by coupling the instream flow assessment with the day-of-year ANCOVA analysis for July 1–September 30. At all
three sites changes in modelled parr habitat is comparatively more sensitive to changes in streamflow than is changes in modelled fry habitat.

3.4.1 241 Creek

Significant differences in WUA occurred on only two days during the 1993-2000 period for both fry (18-33% reductions) and parr (40-43% reductions) and no significant changes in WUA were observed during the 2001-2007 period (Figure 10). Seven days of significant changes occurred during the final period from 2008-2017 period for both fry (20-26% reductions) and parr (36-48% reductions).

![Figure 3.14: Time series of fish habitat based on predicted and observed streamflow at 241 Creek. Bold lines indicate significant results.](image)

3.4.2 Dennis Creek

No significant changes in WUA were observed for fry or parr from 1995-2000 (Figure 11). During the next treatment period, from 2001-2008, significant
changes in WUA occurred on two days for fry (24-25% reductions) and on three days for parr (29-38% reductions). During the final treatment period from 2009-2017, significant changes in WUA occurred on 54 days throughout the summertime low flow period for both fry (12-45% reductions) and parr (20-72% reductions).

![Figure 3.15: Time series of fish habitat based on predicted and observed streamflow at Dennis Creek. Bold lines indicate significant results.](image)

**3.4.3 Camp Creek**

During the 1977-1982 period, significant increases in WUA occurred on one day for fry (19%) and on eight days for parr (18-43%), generally around the end of July. From 1983-1990, there were no days with a significant change in fry WUA and only two days with significant increases in parr WUA. From 1991-1997, statistically significant increases in WUA occurred on four days for both fry (30-42%) and parr (48-91%), during the beginning of July. In the final treatment period from 1998-2007, significant changes in WUA were observed on 19 days for fry (reductions of 19-36%) and 36 days for parr (51% reduction to 14% increase.
- although there was only one day with an increase), primarily from the beginning of August until the end of September.

Figure 3.16: Time series of daily predicted and observed fish habitat at Camp Creek. Bold lines indicate significant results.
4 Discussion

4.1 Paired Catchment Analyses

The results of this study identified few significant changes in summertime low flows for the first decade after harvest. These findings are consistent with previous literature, which also identified minimal changes or small increases to summertime low flows following logging (Pike and Scherer, 2003; Moore and Wondzell, 2005). However, statistically significant decreases in summertime low flows were more prevalent in the treatment periods that began 15 years (241 Creek), 17 years (Dennis Creek), and 22 years (Camp Creek) following the onset of harvest, compared to the earlier periods. The least pronounced changes in summertime low flows occurred at 241 Creek, the most recently logged, and the most pronounced changes occurred at Camp Creek, the earliest logged, further supporting the inference that the effects amplify with time following harvest. However, interpreting the temporal evolution of these changes in flows is complicated by the ongoing nature of forest harvesting that occurred over many years in these watersheds.

The delayed reduction in low flows identified by this study, along with the work of Perry and Jones (2017), expand our understanding of the impacts of logging on low flows in the Pacific Northwest beyond the immediate 5-10 years following harvesting. These longer-term reductions in summertime low flows have now been observed in both coastal and interior watersheds, indicating the possibility of a more general hydrological response to logging. However, further analysis of other Pacific Northwest watersheds should be conducted before these results can be generalized beyond these study catchments to allow for broader understanding of how summertime low flows evolve through time because of logging.
The physical processes associated with the decreased low flows through time are likely to be a combination of 1) a spring-ward shift in the hydrograph associated with an earlier onset of and accelerated rates of snowmelt, leading to an earlier onset of the post-freshet recession toward baseflow, and 2) changing transpiration rates and interception loss in regenerating forests. The remainder of this section interprets these changes in summertime low flows within the context of these affected hydrological processes.

In the paired-catchment results, the signature of a spring-ward shift in the hydrographs is evident through significant increases in daily flows in May, followed by declines in June. Such a pattern is apparent in the last two treatment periods at 241 and Dennis Creeks. However, the pattern is less apparent at Camp Creek, possibly because the pre-harvest regressions were not strong during the freshet due to differences in snowmelt timing between Camp and Greata catchments. Of note, an earlier freshet does not necessarily lead to reduced late summer flows, because greater snow accumulation following harvest could lead to increased recharge of subsurface storage, which in turn could lead to higher, sustained discharge to the stream later in the season. In addition, with reduced transpiration, higher soil moisture, and reduced interception loss, more summer rainfall could also contribute to streamflow. Indeed, in the first two treatment periods at both 241 Creek and Camp Creek, there is either no significant decrease or a slight increase in flows from July to September. Similar observations hold for the first treatment period at Dennis Creek. Based on these observations, it appears that an increase in transpiration (or the sum of transpiration and interception loss) is required to explain the reduced late-summer flows.

Measurements of higher transpiration rates in relatively young trees have been linked to decreases in streamflow in coastal Oregon (Perry and Jones, 2017) and Australia (Vertessy et al., 2001). In both Dennis Creek and Camp Creek, the reductions in streamflow persisted throughout most of the low flow period, which
corresponds with the peak growing season, when the highest transpiration rates occur (Winkler et al., 2010a). However, the forests of the interior watersheds considered in this study are mainly composed of lodgepole pine, Engelmann spruce, and in the case of Dennis Creek, subalpine fir. These different tree species have different growth characteristics, reach different ages, and have distinct physiological behaviour (Klinka et al., 1999) – all factors that could affect how transpiration rates vary in regenerating forests and limit the geographic transferability of results.

Limited research has addressed the water balance of interior forest stands, particularly regarding transpiration. Figure 7.3 of Winkler et al. (2010b) compares the water balance of regenerating lodgepole pine forests aged 0, 5, 10, and 25 years to that for a mature stand (no age given) in the Upper Penticton Creek catchment. These water balance estimates indicate that the amount of water intercepted and the amount of water transpired by the forest steadily increases in an asymptotic fashion in forests from 0 to 25 years old. These estimates also indicate that increased transpiration and interception loss could account for the gradual declines in summertime low flows found in this study. However, the residual water available for drainage in the 25-year old forest and the mature forest appear similar (Winkler et al., 2010b). Therefore, the water balance estimates do not provide a sufficient explanation of the overall reductions in summertime low flows that are evident in the final treatment periods for Dennis Creek and Camp Creek. The combination of an earlier freshet with the recovery of transpiration and interception provides a plausible explanation.

In contrast to Dennis Creek and Camp Creek, the significant decreases in streamflow seen in the final treatment period in 241 Creek were mainly at the beginning of the summertime low flow period. The final treatment period in 241 Creek was directly preceded by a period of active logging; therefore, the regenerating vegetation was likely relatively immature during this time. These
early-season reductions in streamflow, along with the younger trees present in the 241 Creek catchment, suggest that increasing transpiration rates were not affecting changes to streamflow as strongly as in the other two catchments. If higher transpiration rates were affecting 241 Creek more strongly, they should affect the entire summertime low-flow period. Instead, these earlier reductions in summertime low flows are more likely to be due to an advancement of the snowmelt timing following forest harvesting, which was evident in the final treatment period for this study. This assertion is also supported by previous work, which has identified an increase in May streamflow and a decrease in June streamflow in 241 Creek following harvesting (Winkler et al., 2017).

Unlike the processes discussed above, the impacts to groundwater recharge caused by decreased soil infiltration rates and the interception of shallow subsurface flow in roads and ditches are expected to be consistent through time. Because there is no immediate decrease in summertime low flows apparent in any of the three study catchments, it is unlikely that these processes are strongly influencing summertime low flows in the catchments considered by this study.

An important implication of this study is the light it sheds on the concept of hydrological recovery in the context of summertime low flows. In operational contexts, hydrologists commonly model hydrological recovery as an asymptotic function with decreasing treatment effect through time. However, this study and others (Vertessy et al., 2001; Perry and Jones, 2017) have shown that summertime low flows may “over-recover” before eventually trending back toward conditions associated with mature forest – a trajectory sometimes called the “Kuczera curve” (Vertessy et al., 2001).

Of note, reductions in streamflow observed in Camp Creek could alternatively be due to changes in the streamflow regime related to the decommissioning of a storage reservoir in the upper part of the Greata Creek catchment in 1996 (Golder, 2010). The presence of this storage reservoir may have augmented summertime
low flows by impounding snowmelt runoff, and then slowly releasing that water through the earthen dam via seepage over the remainder of the summer months. Therefore, the removal of this dam is another possible mechanism by which summertime low flows were reduced in the final treatment period.

The transferability of these results may be a function of catchment size. The catchments considered here, along with those studied by Perry and Jones (2017), have areas of around 5 km² or less. Larger catchments typically contain longer and deeper groundwater flow paths capable of attenuating and buffering variations in recharge. Therefore, a post-harvest increase in recharge during snowmelt could conceivably generate higher late-summer flows in a large catchment even if flows were reduced in its headwater tributaries.

Although delayed decreases in summer flows following harvest have now been documented for both rain dominant (Perry and Jones, 2017) and snowmelt dominant catchments (this study), a stronger understanding of the underlying processes is required to define the broader generality of these findings. In particular, further research is required to quantify the changes in transpiration and interception loss as forest regenerates across a range of tree species and climatic contexts.

### 4.2 Impacts of Logging on Fish Habitat

Many studies have explored the hydrological impacts of logging (e.g. reviews by Bosch and Hewlett, 1982; Pike and Scherer, 2003; Moore and Wondzell, 2005; Winkler et al., 2010b), but few have tried to quantify the impacts of logging induced changes in flow on fish habitat. Leach et al. (2012) explored the effects of logging-related stream temperature changes on fish growth by coupling a paired catchment study with a bioenergetic fish growth model in a headwater stream in coastal BC. Another study in coastal Oregon simulated the impacts of forestry and
climate change on streamflow and stream temperature and then used these data to evaluate the associated response of fish populations using a fish population dynamics model (Penaluna et al., 2015).

The current analysis is novel because it coupled paired catchment analyses with eco-hydraulic instream flow assessments, allowing for the direct quantification of the effects of logging on fish habitat. Historically, paired catchment analyses that examine forest harvesting have generally led to the understanding that logging increases streamflow, and in some circumstances have led to the suggestion that logging be used as a way to increase water yield (Troendle et al., 2001). The findings of such studies have obvious implications when management decisions are made that consider how logging operations might impact the aquatic ecology of their respective watersheds. Under the assumption that increased streamflow would tend to have positive impacts on stream ecology (e.g. Hatfield and Bruce, 2000), the effect of logging on aquatic ecology would be beneficial. However, the results from this study clearly show otherwise: not only does logging have a longer-term effect of reducing summertime streamflow, instream flow modelling indicates that it also reduces fish habitat availability.

Interpreting the WUA results should consider the assumptions and potential pitfalls involved with eco-hydraulic modelling. Firstly, by adopting this approach it is assumed that physical habitat is actually limiting fish populations (Rosenfeld and Ptolemy, 2012). Because fish habitat is relatively sensitive to reductions in streamflow in small streams (Hatfield and Bruce, 2000; Bradford and Heinonen, 2008), it is reasonable to assume that, at least in part, physical habitat availability is limiting the fish populations of the small streams considered by this study. However, it is possible that many other factors, such as stream temperature and/or prey availability, could also simultaneously be limiting fish populations.

The application of the hydraulic modelling routine of the instream flow assessment for this study was likely complicated by the steep gradients, coarse
bed materials, and high spatial variability of depths and velocities common in small, headwater streams (Powell, 2014). Due to the difficulties of modelling these kinds of environments, the distributions of hydraulic data generated through these instream flow assessments may not be accurate. To account for this, the confidence intervals that were generated around the WUA curves represent uncertainty with the hydraulic component of WUA (Figure 3.16; Williams, 1996, 2009). If highly unusual or unrealistic hydraulic conditions were present for certain transects, when transects were selected through the bootstrapping routine they would have resulted in wide confidence limits - but only if the habitat suitability criteria used were sensitive to this variability. For example, velocities may have been highly variable between transects, but if the variability is outside of the bounds of what is considered suitable habitat (e.g. if velocities only vary above 0.5·ms\(^{-1}\) for fry, Figure 2.7), this variability would not be reflected by the WUA curves themselves, as the corresponding suitability (0) would not change. Therefore, despite the challenges involved in modelling the hydraulics of these small streams, the confidence intervals generated through this study indicate relatively high confidence in the hydraulic component of the WUA curves.

In addition to potential complications with the hydraulic modelling component of eco-hydraulic instream flow assessments, there are also many issues with the habitat suitability curves used in these approaches (Tharme, 2003; Lamouroux et al., 2010; Lancaster and Downes, 2010; Railsback, 2016). Lancaster and Downes (2010) have argued that habitat suitability curves are developed using spurious statistical methods and that they are inappropriately used to demonstrate habitat choices by fish. Instead, a fish’s choice to occupy certain habitats could be due to there being no other, alternative habitats to occupy, or fish may choose to occupy certain parts of streams for entirely different reasons (e.g. prey availability) that may be correlated to depths or velocities (Lancaster and Downes, 2010). In scenarios where prey availability is likely a more important factor in controlling
populations, habitat suitability relationships generated using bioenergetic modelling approaches would be more appropriate (Rosenfeld and Ptolemy, 2012; Rosenfeld et al., 2016). In addition, alternative modelling techniques that explicitly consider the effects of stream temperature on fish production have also been used in previous studies (Leach et al., 2012; Penaluna et al., 2015).

The application of the habitat suitability curves also assumes that they are appropriate for the fish populations of the relevant streams. In the case of this study, these habitat suitability curves were generated from measurements taken from many streams and rivers throughout BC, most of which are likely much larger than the streams considered here. Particularly for 241 Creek and Dennis Creek, these watersheds are relatively small for fish bearing streams, and are near the threshold of stream size that can support rainbow trout (Trotter, 2000; McCleary and Hassan, 2008). Relatively little is known about the rainbow trout that occupy these small, headwater streams and it is possible that the fish in these streams may be better suited for survival in the limiting habitat conditions of these streams (e.g. shallower depths), making them respond to changes in streamflow in a different way than fish resident to larger streams and rivers. This is especially true for interior rainbow trout populations, which easily adapt to different environmental conditions (Behnke, 2002).

Evidently, the instream flow assessment conducted as a part of this study, as well as instream flow assessments conducted by practitioners, are underpinned by many assumptions that can impact the validity of their results. Regardless, these types of studies remain an important fixture within a regulatory context, and their empirical nature often makes them the preferred approach for setting EFN when fish habitat is at risk. For this study, they provide a useful means for modelling how logging affects fish habitat, although the potential issues and assumptions inherent in this approach should be considered when interpreting these results.
5 Conclusion

This section summarizes the key findings in relation to the research questions presented in Section 1.5, followed by a discussion of potential future research to build on this study.

5.1 Key Findings

This study evaluated how logging impacted summertime low flows and fish habitat in three small, snowmelt-dominant catchments of the Pacific Northwest. For the first one to two decades following the onset of forest harvesting, few significant changes in summertime low flows were observed. These findings are largely consistent with previous literature, which has found that forestry has minimal impact of summertime low flows immediately following harvest.

As time progressed following the onset of logging, reductions in the summertime low flows of these watersheds became commonplace. In 241 Creek, reductions in streamflow were detectable for approximately two weeks and the low flow yield was 16% lower in the 15-24-year period following the onset of harvest. In Dennis Creek, reductions in streamflow of up to 50% persisted for many days and the low flow yield was 43% lower in the 14-22-year period following the onset of harvest. In Camp Creek, reductions of streamflow that were typically around 20-40% were statistically significant for most days in the 21-34-year period following the onset of harvest. As well, in Camp Creek the low flow yields were reduced by 18% for the period 22-28 years following harvest and by 20% for the period 29-35 years following harvest. Results in Camp Creek may be confounded by changes in the streamflow regime that may have occurred in Greata Creek (the control catchment for the paired catchment analyses) following the
decommissioning of a water storage reservoir in 1996. The results from all these creeks indicate that the concept of hydrological recovery may not suitable for understanding the impacts of forest harvesting on summertime low flows and masks complex interactions among key watershed processes.

The instream flow assessments indicate that summertime low flow periods are likely a critical time for the resident rainbow trout that live in the study streams. The WUA curves show that reductions in streamflow at low flows correspond to relatively large reductions in physical fish habitat availability. When coupled with the paired catchment analyses, this analysis indicated large reductions in fish habitat of 6-52% in 241 Creek and 12-72% in Dennis Creek during the final treatment period. This analysis also generally showed a reduction in habitat in Camp Creek, although changes varied from a 14% increase to a 49% decrease during the final treatment period. There is reasonably high confidence in the hydraulic component of the WUA analysis, but there is considerable uncertainty associated with the habitat suitability curves that were used. This study is one of few that has directly quantified how logging induced changes to streamflow effect fish.

5.2 Future Research

This study did not directly address the hydrological processes that could be causing the reduction in low flows observed in these three catchments. However, the temporal patterns of streamflow change, along with results from one stand-level water balance study, suggest that the decreased low flows were caused by the combination of an earlier freshet and a recovery of interception loss and transpiration by the regenerating forest. Studies from other forest types suggest that increased transpiration and interception loss in regenerating forests causes a reduction in low flows in the medium to long term follow harvest, but given the
variability among different forest types, it would be an overgeneralization to directly apply these findings to this study. Further work should explicitly test how transpiration and interception loss rates change through time in the forests typical of snowmelt-dominant catchments.

The eco-hydraulic instream flow assessments provided a means to understand the ecological impacts of changes to streamflow. The use of this technique to quantify habitat as a function of discharge is used frequently by practitioners, but its suitability for small, headwater streams is not often evaluated. Future work should explicitly evaluate how well these instream flow assessments reproduce the hydraulics of small streams. Secondly, there is also limited information about whether the fish populations resident to these small streams have adapted to environments where there is limited hydraulic habitat. Future work should also evaluate whether habitat suitability curves generated from larger streams can be suitably applied to the fish populations resident to smaller watersheds. Finally, although stream temperature is recognized as an important control on fish health it is rarely explicitly evaluated as a part of EFN. Further work should also focus on the development of methods to assess how changes in stream temperatures influence the environmental flow needs of streams.
References


Dickerson-Lange SE, Gersonde RF, Hubbart JA, Link TE, Nolin AW, Perry GH, Roth TR, Wayand NE, Lundquist JD. 2017. Snow disappearance timing is dominated by forest effects on snow accumulation in warm winter climates of the Pacific Northwest, United States. *Hydrological Processes* 31 (10): 1846–1862


Payne TR. 2003. The Concept of weighted usable area as relative suitability index. *IFIM Users Workshop 1-5 June 2003, Fort Collins, CO*


Railsback SF. 2016. Why it is time to put PHABSIM out to pasture. *Fisheries* 41 (12): 720–725


Williams JG. 2009. Lost in space, the sequel: spatial sampling issues with 1-D PHABSIM. *River Research and Applications* 26 (3): 341–354


Zhang M, Wei X. 2012. The effects of cumulative forest disturbance on streamflow in a large watershed in the central interior of British Columbia, Canada. *Hydrology and Earth System Science* 16
Appendix A:

NDVI Analysis of Clearcuts in Camp Creek and Greata Creek

Introduction

At the Water Survey of Canada gauging site, Greata Creek drains a 43.5 km$^2$ snowmelt-dominant watershed in the southern interior of British Columbia. Previous studies have used Greata Creek as an experimental control in paired catchment analyses to investigate the hydrologic effects of logging in the neighbouring Camp Creek catchment (Cheng, 1989; Moore and Scott, 2005). Although Greata Creek has been used as an experimental control, as of 2005, logging reportedly affected approximately 9% of the Greata Creek catchment (Moore and Scott, 2005). Anecdotal accounts indicate that logging activities have increased within the Greata creek catchment since the early 2000s; however, to the author's knowledge, there is no detailed chronology of logging that has occurred within the Greata Creek watershed.

Ideally, a control watershed within a paired catchment analysis should be completely unlogged; however, this is not always practical. In catchments that have experienced some logging, like Greata Creek, a chronology of historical logging events is useful for determining when it is suitable for use as an experimental control. Thus, the purpose of this investigation is to develop a more complete, quantitatively rigorous chronology of logging that has occurred within the Greata Creek watershed. This is done two ways: 1) by analyzing changes in vegetation cover using normalized difference vegetation index (NDVI) images, and 2) by visually identifying clearcuts using satellite imagery.
NDVI Analysis Methods

NDVI images, which are commonly used to study the abundance, spatial extent, and temporal patterns of terrestrial vegetation, have been collected at a fine scale resolution by Landsat satellites since 1984 (Pettorelli et al., 2005). The NDVI is a measurement of the relative reflectance of different wavelengths of light by the ground surface. It is calculated using red and near infrared (NIR) light that is reflected by vegetation as follows (Pettorelli et al., 2005):

\[
\text{NDVI} = \frac{\text{NIR} - \text{Red}}{\text{NIR} + \text{Red}} \quad (A1)
\]

Land surfaces that are heavily vegetated typically absorb red light and scatter NIR, so relatively denser vegetation cover tends to correspond to high NDVI values, while sparse and non-vegetated surfaces have lower values. NDVI values range from -1 (e.g. deep water) to 1 (e.g. dense vegetation).

Analysis of the annual clearcut area of Greata Creek was conducted through analysis of NDVI composite images on a yearly basis from 1985 to 2016. Using the Google Earth Engine code editing tool (Gorelick et al., 2017), 8-day and 32-day NDVI composites were obtained without having to manually calculate the vegetation index. Every year, the most cloud-free image from the 8-day and 32-day Landsat collections were selected from the June 1 - September 30 period to capture seasonal vegetation during the peak growth period. Landsat 5 composites were used between 1985 and 2011 and Landsat 8 composites were used for 2013 to 2016. Landsat 7 composites were not used because most images in the collection have large bands of missing data over the Greata Creek catchment. In addition, there were no usable images from 2010 or 2012.

After selecting satellite images, an analysis of yearly changes in NDVI was performed using R (R Core Team, 2018). NDVI composites from each year were converted to rasters and were sequentially overlaid to examine changes in the NDVI values for each pixel from year-to-year. Then, the NDVI pixel values from
each year were subtracted from that of the year prior, generating an NDVI difference value. A negative NDVI difference between two years corresponds to a reduction in vegetation cover – likely due to logging; however, negative NDVI differences could also be from natural causes such as pine beetle infestations, or windthrow.

Further analysis was conducted to isolate the effects of logging from other natural causes. Firstly, an NDVI difference threshold value (-0.15) was selected to identify changes in vegetation cover caused by logging from other means. The validity of the -0.15 threshold was assessed two ways. Firstly, the extent of pixels that were classified as ‘logged’ (i.e. < -0.15) were visually compared to obvious clearcuts on corresponding satellite images and were found to be a close match. Secondly, the distribution of NDVI differences from areas believed to be unlogged was assessed by removing pixels that had undergone obvious logging from the dataset, and then generating a histogram of the NDVI differences from the remaining pixels. The histogram (Figure A1) shows that the distribution of these NDVI differences are predominantly greater than the threshold of -0.15, indicating that this is an acceptable threshold value.

![Figure A1: Distribution of year-to-year NDVI differences for non-logged pixels.](image-url)
Following the classification of pixels using the NDVI difference threshold, further analysis was conducted to identify small clusters of pixels that had NDVI differences below the difference threshold, but otherwise represented too small of an area to realistically be a clearcut. This analysis involved passing a 3-by-3-pixel window (approximately 0.8 hectare) over the NDVI difference raster to reclassify the pixel at the centre of the window to the most frequently observed value of all those contained within window.

Analysis using this NDVI difference approach has its limitations. For example, natural events such as wind storms and insect infestation have the potential to kill entire stands of trees that would likely be incorrectly classified as having been logged. Conversely, small logging events, such as the selective logging of mature individuals or the logging of isolated trees infested with pine beetle, may also have been missed.

**Satellite Imagery Analysis Methods**

Landsat satellite imagery available on Google Earth© was analyzed every year from 1985-2016. For each new year, new clearcuts from the Greata Creek watershed were visually identified and traced to quantify the logging history on an annual time step. The areas of these clearcut polygons were then calculated for each year. When available, aerial photographs were also used to corroborate the timing of clearcuts.

**Results**

Clearcuts identified through both the NDVI difference analysis and the satellite imagery analysis are displayed in Figure A2. These two methods identify clearcuts in similar areas during similar years; however, the satellite imagery analysis identified comparatively more area than the NDVI difference analysis.
Figure A2: Clearcuts identified within the Gareta Creek catchment, 1985 - 2016.

The cumulative harvest over time in Gareta Creek calculated using both the NDVI difference and satellite imagery analysis is shown in Figure A3. These results show that logging in the Gareta Creek watershed was relatively minimal until the last 3-8 years, at which point the intensity of logging increased. Additionally, these results show similar trends in the cumulative harvest identified by both the NDVI difference analysis and the satellite imagery analysis; however, the cumulative harvest identified in the satellite images is consistently about 3% to 7% higher.
Discussion

Identifying when a watershed has experienced too much logging to be an effective control catchment requires knowledge of a watershed’s logging history. Previous studies show that logging relatively small portions of watershed (< 15%) does not tend to produce statistically significant changes to streamflow (Troendle and King, 1987; Stednick, 1996). However, just because statistical tests are insignificant does not necessarily mean the streamflow regime goes unchanged. In such cases, it is possible that small changes to streamflow do occur, but that the statistical tests used do not have the power to identify them (Moore and Scott, 2005). Although the total area logged in an important consideration, using only an area-based threshold could mask meaningful site-specific complexities, such as the temporal nature of logging because differences in when logging occurs could produce quite different hydrological effects. For example, in watersheds that experience small amounts of incremental logging over a period of decades, trees would regrow over time - resulting in some degree of hydrological recovery.
that could negate some of the cumulative hydrological impacts (Winkler et al., 2010). In contrast, if the same watershed was subject to the same total harvest area, but it occurred over one season, that watershed would likely experience more pronounced hydrological changes.

Moore and Scott (2005) indicated that up to 9% of the Greata Creek watershed was harvested by the early 1990s, which is about 2-4% greater than the cumulative harvest identified by the satellite imagery analysis by this time. The cumulative harvest identified by the NDVI analysis is around 5% lower, but it does not capture for harvesting that occurred before 1984.

Considering the findings of Moore and Scott (2005) along with this analysis, it is likely that anywhere from 7-11% of the Greata Creek catchment was logged by 2011. From 2011 onward, the intensity of logging began to increase, making it less suitable as a control catchment from this point forward. Considering the uncertainty around the total harvest area, the pattern of this recent increased logging intensity is particularly important for determining the suitability of Greata Creek as a control catchment.

**Conclusion**

This study examined the suitability of Greata Creek as an experimental control catchment over time by determining when logging occurred through the analysis of NDVI images. The analysis indicates that the intensity of logging in Greata Creek was relatively minimal until 2011, when it increased. Based on this analysis, Greata Creek can be used as a suitable control catchment until this time.
References


