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Effects of forestry on summertime low flows and physical fish habitat in snowmelt-dominant headwater catchments of the Pacific Northwest

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ABSTRACT

Periods of summertime low flows are often critical for fish. This study quantified the impacts of forest clearcutting on summertime low flows and fish habitat and how they evolved through time in two snowmelt-dominant headwater catchments in the southern interior of British Columbia, Canada. A paired-catchment analysis was applied to July-September water yield, the number of days each year with flow less than 10% of mean annual discharge, and daily streamflow for each calendar day. The post-harvest time series were divided into treatment periods of approximately 6-10 years, which were analyzed independently to evaluate how the effects of forestry changed through time. An instream flow assessment using a physical habitat simulation (PHABSIM)-style approach was used to relate streamflow to the availability of physical habitat for resident rainbow trout. About two decades after the onset of logging and as the extent of logging increased to approximately 50% of the catchments, reductions in daily summertime low flows became more significant for the July-September yield (43%) and for the analysis by calendar day (11-68%). Reductions in summertime low flows were most pronounced in the catchment with the longest post-harvest time series. Based on the temporal patterns of response, we hypothesize that the delayed reductions in late-summer flow represent the combined effects of a persistent advance in snowmelt timing in combination with at least a partial recovery of transpiration and interception loss from the regenerating forests. These results indicate that asymptotic hydrological recovery as time progresses follow logging is not suitable for understanding the impacts of forest harvesting on summertime low-flows. Additionally, these reductions in streamflow corresponded to persistent decreases in modelled fish habitat availability that typically ranged from 20-50% during the summer low-flow period in one of the catchments, suggesting that forest harvest may have substantial delayed effects on rearing salmonids in headwater streams.

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INTRODUCTION

Over the past few decades, there has been increasing concern about the ecological impacts of changes to streamflow regimes (Poff et al., 2010). Resource management decisions that consider changes to the flow regime 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). Assessments that consider EFN are typically concerned with low-flow periods when streamflow is an important determinant of the productive biological capacity of small streams. During summertime low flows, streamflow typically limits the seasonal food supply, and the subsequent growth and survival, of juvenile salmonids (Grant & Kramer, 1990). The limiting nature of summertime low flows on juvenile salmonids has been shown to have lasting effects that impact the productivity of adult salmonid populations following migration from riverine environments (Bradford, 1995). In addition to food supply, changes in summertime low flows also affect salmonid populations by influencing water quality, stream temperature, and physical habitat availability (Rosenfeld, 2017).

Although many previous studies have documented how forestry affects streamflow and water quality (e.g., Bladon, Segura, Cook, Bywater-Reyes & Reiter, 2018; Bywater-Reyes, Bladon & Segura, 2018; Gomi, Moore, & Hassan, 2005; Moore, Spittlehouse, & Story, 2005; Moore & Wondzell, 2005; Perry and Jones, 2017), EFN assessments have historically been concerned with consumptive use of water resources, such as power generation and agriculture, and have neglected to consider the impacts of forestry on the natural flow regime (e.g. Lewis, Hatfield, Chilibeck, and Roberts, 2004). In the context of EFN, the impacts of forest harvesting on summertime low flows are of particular interest. Forest harvesting could influence late-summer low flows through a variety of processes, with a potential range in the direction and magnitude of impacts.

In the Pacific Northwest, removal of trees typically increases snow accumulation by 30% to 40% by reducing interception loss (Dickerson-Lange et al., 2017; Winkler et al., 2010b). Within clearcuts, snowmelt tends to begin earlier in the spring and occur at a higher rate than under a forest canopy (Moore & Wondzell, 2005; Winkler, Spittlehouse, & Boon, 2017). Accelerated snowmelt in forest openings often results in an earlier disappearance of snow (Winkler et al., 2017), which could result in an earlier transition from snowmelt freshet to summer baseflow recession, ultimately resulting in lower late-summer baseflow. However, in

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some cases the increased accumulation in openings is associated with a delay in snow disappearance despite the accelerated melt (Dickerson-Lange et al., 2017; Hubbart, Link, & Gravelle, 2015). Such a situation would tend to promote higher late-summer flows due to a delayed shift from snowmelt freshet to baseflow recession.

Following the seasonal snowmelt period, interception loss reduces the amount of rainfall that reaches the soil, and transpiration further reduces subsurface storage by removing soil moisture. One study of Douglas-fir (*Pseudotsuga menziesii*) stands in coastal British Columbia found, using eddy covariance measurements, that both interception loss and transpiration were greatly reduced immediately following logging and tended to increase as the forest regenerated (Jassal, Black, Spittlehouse, Brümmer, & Nesic, 2009). A geographically diverse set of studies has used sap-flow measurements to show that young forests can transpire more than mature forests, suggesting that post-harvest recovery is not a simple asymptotic function of time. In a comparative study of transpiration in different-aged riparian forests in the Oregon Cascades, a 40-year-old forest comprised primarily of Douglas-fir had 21% more sapwood basal area and used 3.27 times more water at the stand level than a nearby, 450-year-old forest (Moore et al., 2004). In the interior of the Pacific Northwest, Ryan et al. (2000) found that 40-year-old Ponderosa pine (*Pinus ponderosa*) forests transpired approximately 45% more than 290-year old forests at the stand level over the June 1–August 31 period. A study in Australia linked increases in transpiration in relatively young, regenerating mountain ash (*Eucalyptus regnans*) forest stands to long-term decreases in water yield (Vertessy et al., 2001); and in France, maritime pine (*Pinus pinaster*) forest stands over 50 years old transpire less than 10-year-old forests (Delzon & Loustau, 2005).

In addition to the effects of canopy removal, the use of heavy machinery can compact soil, reducing its infiltrability (Moore & Wondzell, 2005), and roads, culverts, and ditches can intercept shallow subsurface flow, conveying it more quickly to the stream network (La Marche & Lettenmaier, 2001; Moore & Wondzell, 2005; Winkler et al., 2010b). Both these effects would tend to reduce the amount of subsurface recharge and thus could reduce late-summer baseflow. Because of the slow rate of recovery of soil permeability following compaction (Rab, 2004), effects of forestry operations such as road construction on hydrologic flow paths should recover more slowly than effects related to forest removal.

The impacts of forest harvesting on these different hydrological processes have distinct temporal patterns that are apparent at different times throughout the year. For example, while

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increased snow accumulation and melt can result in higher flows following freshet early in the summer, the effects of increased transpiration rates can produce streamflow deficits later in the summer, during the peak growing season. When combined, these effects may have minimal impact on annual or seasonal water yields, but the shorter-term temporal shift in late-summer flows on aquatic eco-systems may be significant.

Globally, the effects of forest plantations have been shown to cause reduced water yields in South Africa (Scott, Le Matire, & Fairbanks, 1998), Australia (Vertessy et al., 2001), and in Chile (Little, Lara, McPhee, & Urrutia, 2009). A number of paired-catchment studies have addressed forestry effects on summer low flows in snowmelt-dominant catchments in North America; however, the majority of these studies only examined responses for the first five to ten years following harvesting (Table 1) (Moore and Wondzell, 2005). These studies generally found minor effects or even slight increases in late-summer flow. A notable exception is a study focused on eight small rainfall-dominant and hybrid rain-on-snow catchments that ranged from 0.1 – 1.0 km^2 in the HJ Andrews Research Forest of the Oregon Cascades (Perry & Jones, 2017). The analysis showed that, 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%. 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, Beschta, & Harr, 1991; Moore et al., 2004). Considering the potentially important effect of snow dynamics on low flows, and the differences in forest tree species, it is unclear whether the results of Perry and Jones (2017) can be extrapolated to snow-dominant catchments within the region.

Although many paired-catchment studies have focused on the effects of forestry on streamflow, there has been little attention to the implications for fish habitat availability. One approach to linking streamflow changes and potential effects on fish habitat are 'eco-hydraulic' models such as the physical habitat simulation (PHABSIM) model (Bovee et al., 1998), the System for Environmental Flow Analysis (SEFA) (Jowett, Payne, & Milhous, 2016), and reach-averaged statistical approaches (e.g. McParland, Eaton, & Rosenfeld, 2016). These models quantify available fish habitat at different flow levels by computing the weighted usable area (*WUA*) for a specific species and life stage (Payne, 2003). While PHABSIM-style models have been considered to be a rigorous, scientifically defensible means to evaluate changes in fish habitat (Tharme, 2003), they have been criticized on the

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basis of issues with their statistical methods, the lack of studies that validate model output, and the challenges associated with evaluating the biological significance of *WUA* (Mathur, Bason, Purdy Jr., & Silver, 1985; Lancaster & Downes, 2010; Railsback, 2016). However, considerable efforts have been made to improve some of these shortcomings, and eco-hydraulic models continue to be a common and important component of water management decision processes (Reiser & Hilgert, 2018).

The objectives of this study were (1) to use a paired-catchment approach to quantify changes in summertime low flows following forest harvesting in two snowmelt-dominant catchments using data sets extending up to two or more decades following harvest, and (2) to quantify the resulting changes in physical fish habitat by using eco-hydraulic habitat modelling.

METHODS

Study Sites

This study was conducted within the Upper Penticton Creek Experimental Watershed, which includes three gauged sub-catchments located in the uplands east of Okanagan Lake (Figure 1). The Upper Penticton Creek catchment receives approximately 700 mm of mean annual precipitation, just under half of which falls as snow (Winkler et al., 2017). As is typical of snowmelt-dominant regions of the Pacific Northwest, the highest flows occur during the snowmelt freshet between April and June, followed by a general recession through summer and the following winter, with occasional increases in streamflow associated with autumn rainstorms. The soils in the Upper Penticton Creek catchments are generally less than 2 m thick and consist of sandy loam and loamy sand that overlie glacial till and granitic rocks (Winkler, Spittlehouse, Boon, & Zimonick, 2015). Rainbow trout (*Oncorhynchus mykiss*) have been observed throughout the Upper Penticton Creek watershed.

Daily streamflow records collected by the Water Survey of Canada are available since 1984 for 240 Creek and 241 Creek, and since 1985 for Dennis Creek. During the snowmelt freshet, streamflow data at all three sites are measured using broad-crested weirs, and once flows decline into the summer months, the broad-crested weirs are replaced with v-notch weirs that are better suited for the collection of low-flow data. Key biophysical characteristics of these catchments, including pre-harvest forest cover characteristics, are summarized in Table 2.

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. From 1984 to 1995, the catchments remained undeveloped with the exception of a small area

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of blowdown salvage in 1992 near the outlet of 241 Creek (Winkler et al., 2017). Road construction and conventional clearcut logging began in 1995 (Table 3). The extent of the planned harvest was intended to represent maximum levels of harvest likely to be considered in most catchments. All road construction, logging, and reforestation were completed according to the operational standards of the day. Logging was conducted during late fall and winter, on snow, using feller bunchers and ground-based skidding. Riparian management included the retention of 5- to 10-m-wide buffers on each side of streams greater than 1.5 m in bankfull width. Clearcuts were predominantly replanted with pine, or spruce on wetter sites, within four years of harvest with a target stocking density of 1,000-1,600 stems per hectare, with some ingress expected. Additional silviculture activities included mechanical site preparation, burning of slash piles where necessary, and the deactivation of all non-essential roads following logging and planting

Paired Catchment Analysis

The paired catchment approach is considered the most statistically rigorous way to detect the hydrologic effect of land-cover changes (Moore & Wondzell, 2005; Zégre, Skaugset, Som, McDonnell, & Ganio, 2010). The analysis proceeded in two ways: the first estimated the treatment effect on a year-by-year basis, and the second assessed the significance of the treatment effect for different sets of years in the post-harvest period.

This first approach involved fitting a statistical relation between streamflow metrics for two catchments for a pre-harvest calibration period:

$$y_t = b_o + b_1 x_t + e_t \quad (1)$$

where y_t and x_t are the streamflow metric in year t for the treatment and control catchments, respectively, b_0 and b_1 are regression coefficients, and e_t represents the residuals. This pre-harvest relation was applied in the post-treatment period to generate predicted values ($\hat{y}_t$) that provide estimates of what streamflow would have been had logging not occurred. The difference between observed and predicted streamflow in the post-harvest period provides an estimate of the effect of the treatment for a given year.

For the second approach, the post-harvest time series of streamflow data were split into separate treatment periods based on the major milestones in the logging history of these watersheds (Table 4). In general, the period lengths, which were each roughly six to ten

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years, were chosen as a trade-off between maximizing statistical power while maintaining the ability to detect temporal shifts in the magnitude and direction of the treatment effect. In 241 Creek, the final treatment period (2008-2017) encompasses the complete post-logging time series, and in Dennis Creek the post-logging time series (2001-2017) was split into two equal length treatment periods. The data were analysed using analysis of covariance (ANCOVA), which involved fitting two linear models between y_t and x_t for the entire period of record. The "reduced" model had the same form as equation 1 and represents the null hypothesis that the linear relation was unaffected by treatment. The "full" model included x_t as a predictor, a categorical variable γ_t that represented the treatment period ($\gamma_t = 1$ for the pre-harvest period, and 2, 3, etc. for post-harvest periods), and interaction terms. An *F* test was used to evaluate whether the difference between the full and reduced models was statistically significant (Moore and Scott, 2005).

Analyses focused on three streamflow metrics. The first was the seasonal low-flow water yield for the period from July 1 to September 30, which generally captures baseflow conditions following the snowmelt peak. Regression analyses and ANCOVA were performed as described above. Initial examination confirmed that the relation between seasonal low-flow yields for 241 and 240 creeks met the key assumptions underlying regression (linearity and homoscedasticity). In contrast, the pre-harvest relation for Dennis Creek was nonlinear, which was addressed by log-transforming the seasonal low-flow yields for both Dennis and 240 Creeks prior to conducting the regression analysis and ANCOVA.

The second metric analyzed was the frequency of low-flow days each year, where a low-flow day is defined as having flow less than 10% of the long-term mean annual discharge (*MAD*) during the July 1 – September 30 period. The threshold of 10% *MAD* was chosen to define low-flow days because it has historically been used to define the point at which further reductions in streamflow results in the severe degradation of aquatic habitat (Tennant, 1976).

The final metric that was analyzed was daily mean streamflow on a day-of-year basis for the period May through October. In this approach, daily streamflow values were extracted for each year on a given date, and then used in a paired catchment analysis. Because consecutive observations in this analysis were separated by a full 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). Further, because each calendar day was analyzed independently, it also allows for the seasonal variability in the treatment effect to be assessed. On the basis of initial exploratory analysis, daily streamflow data were

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log-transformed to reduce nonlinearity and heteroscedasticity prior to performing the regression analysis and ANCOVA. Because daily streamflow data were log-transformed, days of zero flow were removed from the analysis.

Back-transformed predictions based on a log-transformed regression will be biased. Therefore, the bias correction presented by Baskerville (1972) was applied where relevant. For both the day-of-year and seasonal water yield analyses, statistically significant differences were identified when $p < 0.05$.

Instream Flow Assessment

Field Methods

A field campaign was carried out over four field trips at different flows during the summer of 2017 (Table 5) to provide required input data to run the System for Environmental Flow Analysis (SEFA) for habitat modelling, which is a PHABSIM-style eco-hydraulic modelling method. The first component of the field program was to establish study reaches for each of the study streams, which were 94 m and 50 m long for 241 Creek and Dennis Creek, respectively. The lengths of the study reaches were chosen to be a minimum of 20 times the length of the average channel cross-section, per the guidance of Lewis et al. (2004), and were located near the upper threshold of fish habitat (Trotter, 2000; McCleary & Hassan, 2008). Study reaches were also located near the streamflow gauging stations so that the results from the instream flow assessment could be associated with the time series of measured discharge from each stream. Within each study reach, mesohabitats (e.g. pools, riffles, cascades) were mapped according to the criteria outlined by 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, with the number of samples in each mesohabitat roughly proportional to its total length within the reach (Table 6). 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). Mesohabitat classifications varied with streamflow in 241 Creek; therefore, the classifications that best represented low-flow conditions were ultimately used for the instream flow analysis.

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During each field visit, depth and velocity measurements were made at approximately 20 regularly spaced intervals along each transect for all surveys using a Marsh-McBirney current meter attached to a top-set rod. Channel substrate was characterized at low flows by visually estimating the relative percentages of different bed materials within a 10-cm radius of each depth/velocity measurement point (i.e. halfway between adjacent points) using the Wentworth classification scheme detailed by Lewis et al. (2004). Water surface elevations at two other flows were measured relative to the benchmarks along each transect during each field trip using a digital range-finder or a survey level and stadia rod. Streamflow during the field campaign declined as the summer progressed (Table 5).

Analytical Methods

Weighted usable area curves were simulated for flows from 0 to 100% *MAD* using the 'habitat mapping' method (Jowett et al., 2016). The first step was to use the water surface elevation (stage) measurements collected in the field for each transect, along with discharge reported by the streamflow gauges, to develop stage-discharge rating curves specific to each transect. For each transect, the stage was determined relative to a benchmark installed on the right bank. Discharge was calculated at each stream for each survey by averaging the reported discharge values at the streamflow gauging station 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 relevant stream gauge were used to develop the rating curves for all transects within each stream. Transect-specific rating curves developed within SEFA were 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 for each discharge. This process involved reproducing the velocities measured along each transect using Manning's equation and conveyance at each measurement point (for further information see Jowett et al., 2016).

Following the execution of the SEFA hydraulic model, habitat suitability curves specific to rainbow trout fry and parr rearing habitat in BC were used to determine the suitability of the study reaches based on depth, velocity, and substrate measurements (Figure 2). These suitability curves were provided by Ron Ptolemy (Instream Flow Specialist, BC Ministry of

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Environment) and are designed for use as relative indices of habitat quality and range from 0 (not suitable habitat) to 1 (preferred habitat).

The habitat suitability for each measurement point along a transect was represented by its composite suitability index (CSI), which was calculated as follows:

$$CSI = HSI_d \cdot HSI_v \cdot HSI_s \quad (2)$$

where HSI denotes the relevant habitat suitability indices for depth, velocity, and substrate characteristics, indicated respectively by the subscripts *d*, *v* and *s*. 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 that they represented, as determined by the mesohabitat classifications, to produce a reach-averaged weighted usable area (*WUA*) measurement specific to the modelled flow.

Confidence intervals were generated around the *WUA* curves using 2,000 bootstrapping simulations (Jowett et al., 2016). For each bootstrap replicate, transects were selected randomly with replacement for each mesohabitat. It is important to note that these confidence intervals only represent the uncertainty of the *WUA* curves due to the variation in the hydraulics (stage-discharge rating curves and velocity estimates) of each transect, given the habitat suitability curves that are applied. They do not represent uncertainty or error associated with the habitat suitability curves themselves (Williams, 2009).

Effects of Logging on Fish Habitat

Logging-induced changes in fish habitat availability were quantified by using the SEFA-derived *WUA* curves to compute habitat availability on a daily basis using both the observed and predicted streamflow from the day-of-year paired-catchment analysis. These values of *WUA* were plotted as time series, and differences were considered to be present on days when both of the following criteria were met: 1) the treatment effect for streamflow on that particular day was significant at $p < 0.05$ 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.

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RESULTS

Hydroclimatic overview of the study period

April 1 snowpack water equivalent measured at a snow course 5 km south of 241 Creek at an elevation of 1,550 m does not display any marked trends through the study period (Figure 3). Both above- and below-average snowpack water equivalent (-39% to +50%) occurred in each post-harvest period (Figure 3). In contrast, for the July 1 – September 30 period, total precipitation appeared to be below average and mean air temperature above average more frequently as the study period progressed (Figure 3). These observations are consistent with projected changes in summertime air temperature and precipitation in the southern interior of BC as a result of climate change (Werner, Schnorbus, Shrestha, & Eckstrand, 2013). Annual hydrographs for the study sites are included in the Supplemental Information.

Paired Catchment Analysis

Seasonal low-flow yield

The pre-harvest regressions for the seasonal low-flow yield had R^2 values of 0.90 and 0.88 for 241 Creek and Dennis Creek, respectively. The seasonal low-flow yield values varied around the pre-harvest regression during early treatment periods, and during later treatment periods they more regularly plotted near or below the lower prediction interval (Figure 4). The ANCOVA results confirmed that yields were significantly lower than predicted in the third treatment period for both 241 Creek and Dennis Creek (Table 7).

Number of low-flow days

The pre-harvest regressions for the number of low-flow days had R^2 values of 0.95 and 0.82 for 241 Creek and Dennis Creek, respectively. The frequencies of the annual number of summertime low-flow days (< 10% *MAD*) varied around the pre-harvest regression during early treatment periods. During later treatment periods the number of low-flow days more regularly plotted near or above the upper prediction interval (Figure 5). The ANCOVA results confirmed that low-flow days were significantly less frequent (on average, six fewer low-flow days each year) during the second treatment period in 241 Creek (Table 7). However, as time progressed following the onset of logging, the number of observed low-flow days increased, particularly in Dennis Creek (Figure 6). The ANCOVA results indicate

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that during the final treatment period at Dennis Creek, there were, on average, 19 additional low-flow days each summer (Table 7). There was also an increase in the number of low-flow days at 241 Creek in the final treatment period, although these results are not significant.

Days with zero flow were not frequent enough to analyse statistically. Zero-flow days only occurred in one year at 240 Creek, and both 241 and Dennis Creeks were more prone to having zero flow than 240 Creek (Figure 6). Both treatment catchments experienced prolonged periods of zero flow in 2017, despite the occurrence of continuous flow at 240 Creek.

Day-of-year analysis

For 241 Creek, the pre-harvest regressions explained from about 60% to 99% of the variance for log-transformed streamflow. The quality of the model fit was weakest in July (R^2 ~ 60%); however, the fit improved throughout the remainder of the summertime low-flow period (R^2 ~ 80%). For Dennis Creek, the pre-harvest regression explained from about 50% to 96% of the variance for log-transformed streamflow. Low R^2 values for Dennis Creek generally occurred during the seasonal peak flows, likely because of the difference in aspect compared to 240 Creek, resulting in differences in snowmelt timing. This point is further supported by improvements in the model fits for Dennis Creek following the spring snowmelt.

At 241 Creek, reductions in streamflow occurred throughout most of August during the first post-calibration period (1993-2000), when harvesting was ongoing, although these changes were not significant (Figure 7). During the second post-calibration period (2001-2007), which also included ongoing harvesting, the ratio of observed to predicted flow was consistently greater than unity in late April to early May (but not significantly), followed by a significant decrease in flow in late May to early June. Following completion of harvesting (2008-2017), there was a significant increase in flows from late April to mid-May, followed by significant reductions (14-40%) from late May through June.

Dennis Creek similarly exhibited no significant changes during the first post-calibration period (1995-2000), during which harvesting progressed (Figure 8). In the first period following completion of harvesting (2001-2008), there were significant increases in flow in April through mid-May, followed by reduced flows in June through August, with occasional days when significant reductions occurred. In the third period, the period of increased flows in April and May was followed by consistent and frequently significant flow declines from

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June through September. Flow declines in this period ranged from 11-68% of the predicted flow.

Instream Flow Assessment

For all three streams, the *WUA* curves indicate relatively more fry than parr habitat at a given discharge (Figure 9). Fry *WUA* is more sensitive to changes in discharge at the lowest flows, and gradually becomes less sensitive with increasing flow until peak *WUA* occurs anywhere from 50-80% *MAD*, while parr *WUA* changes more linearly with increasing flow throughout the range of discharge values. When averaged over the flows experienced during the summertime low-flow period, the widths of the confidence intervals in 241 Creek correspond to changes in *WUA* of 5% for fry and 9% for parr, and the widths of the confidence intervals in Dennis Creek correspond to changes in *WUA* of 12% for fry and 16% for parr. 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 both streams.

Logging Impacts on Fish Habitat

For 241 Creek, significant differences in *WUA* occurred on only two days during the 1993-2000 period for both fry (18-23% 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).

For 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).

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DISCUSSION

Effects on streamflow

The results of this study identified few significant changes in summertime low flows for the first decade or so after the beginning of harvest, which is consistent with many previous studies (Moore & Wondzell, 2005). However, in the treatment periods that began 15 years (241 Creek) and 17 years (Dennis Creek) after the onset of harvest, statistically significant decreases in summertime low flows became more prevalent. The least pronounced changes in summertime low flows occurred at 241 Creek, the most recently logged, and the most pronounced changes occurred at Dennis Creek, the earliest logged. However, interpreting the temporal evolution of these changes in flows is complicated by the ongoing nature of forest harvesting and regeneration that occurred over many years in these watersheds. Another complication is the shift toward warmer summers through the study period (Figure 3), which could potentially confound the effects of post-harvest changes in forest cover. For example, the warmer weather could have advanced snowmelt independently of the effects of forestry, leading to a more extreme streamflow response than would have been observed under the climatic conditions prevalent earlier in the study. Unfortunately, it is not possible to disentangle these influences in the absence of process-based measurements. The combination of process-based measurements with a physically based hydrological model may also provide a means of assessing how sensitive streamflow is to changes in climatic conditions and effects of forest harvesting on hydrological functioning.

Considering the temporal patterns of the treatment effect, both within and among years, the physical processes associated with the decreased low flows through time are hypothesized to be a combination of (1) the spring-ward shift in the hydrograph associated with an earlier onset of and accelerated rates of snowmelt caused by clearcutting, leading to an earlier onset of the post-freshet recession toward baseflow, (2) a reduction of snow accumulation associated with increasing interception loss by the regenerating forest, and (3) increasing transpiration rates and rainfall interception loss in regenerating forests as they reach ages of one to two decades (e.g., Jassal et al., 2009; Winkler et al., 2010b). According to this hypothesis, the post-harvest trajectory of streamflow response reflects the interacting effects of a range of processes, each of which has a distinct temporal signature, as discussed below.

In the paired-catchment results, the signature of a spring-ward shift in the hydrographs caused by changes in snow accumulation and melt is evident in significant increases in daily flows in

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April and May, followed by declines in June. Such a pattern is apparent in the last two treatment periods at both 241 Creek and Dennis Creek. However, an earlier freshet associated with forest harvesting does not necessarily lead to reduced late summer flows because the greater snow accumulation and reduced interception loss and transpiration that typically occur immediately following forest harvesting could lead to higher recharge of subsurface storage during and following the freshet, which in turn could maintain higher baseflow throughout the summer season. In addition, with reduced transpiration, higher soil moisture, and reduced interception loss, a greater fraction of summer rainfall could contribute to streamflow. Indeed, in the first two treatment periods at 241 Creek and in the first treatment period at Dennis Creek, there was either no significant decrease or a slight increase in flows from July to September. Based on these observations, it appears that an increase in one or both of transpiration and interception loss as the forest developed is required to explain the delayed onset of reduced late-summer flows a decade or more following the onset of harvesting.

In contrast to changes in snowmelt, which likely affected the earlier portion of the low-flow period, changes in transpiration rates and interception loss likely have had a greater influence on the latter portion of the low-flow period. Measurements of higher transpiration rates in relatively young, regenerating trees have been linked to decreases in streamflow in coastal Oregon (Moore et al., 2004; Perry & Jones, 2017) and Australia (Vertessy et al., 2001). In Dennis Creek, the reductions in streamflow observed in the later treatment period persisted into August and September, which corresponds with the peak growing season, when the highest transpiration rates occur (Winkler et al., 2010a). Winkler et al. (2010b) compared the water balances of regenerating lodgepole pine forests aged 0, 5, 10, and 25 years to that for a mature stand (>120 years old) in the Upper Penticton Creek catchment based on field measurements, including sapflow-based estimates of transpiration. These water balance estimates indicate that the amounts of water intercepted and transpired by the forest steadily increases in forests from 0 to at least 25 years old, but that the residual amounts of water available for subsurface drainage in the 25-year old forest and the mature forest were similar. While these estimates indicate that increased transpiration and interception loss could account for the gradual declines in summertime low flows found in this study, they do not provide a sufficient explanation for the overall reductions in summertime low flows that are evident in the final treatment periods for Dennis Creek. Considering all available evidence, the combination of an earlier freshet that has not yet returned to pre-harvest conditions, along

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with at least a partial recovery of transpiration and interception loss to pre-harvest conditions, provides a plausible explanation for the reduced summertime low flows.

In contrast to Dennis Creek, the significant decreases in streamflow seen in the final treatment period in 241 Creek were mainly in late May and June. These early-season reductions in streamflow are likely to be due to an advancement of the snowmelt timing following forest harvesting, which is supported by the findings of previous work in this catchment (Winkler et al., 2017). If higher transpiration rates and interception loss were affecting 241 Creek more strongly, they should have affected the entire summertime low-flow period. However, as the relatively young, replanted forests in 241 Creek mature into the future, it is plausible that changes to the streamflow regime may be increasingly controlled by changing rates of transpiration and interception loss.

Changes in groundwater recharge caused by decreased soil infiltration rates and the interception of shallow subsurface flow in roads and ditches do not appear to have had a major impact on the low-flow hydrology of these streams. Such effects would have resulted in an immediate decrease in summertime low-flows that should have been consistent through time. However, there was no immediate decrease in summertime low flows apparent in any of the three study catchments. Therefore, if soil disturbance, roads and ditches did have an effect, it appears to have been overwhelmed by the effects of forest cutting and re-growth.

The delayed reduction in low flows identified by this study, along with the work of Perry and Jones (2017), expand upon the understanding of how the impacts of logging on low flows in the Pacific Northwest change through time. Observations of these reductions in low flows in both coastal and interior watersheds, in combination with results that show replanted stands transpiring more as they mature in France and Australia (Delzon & Loustau, 2005; Vertessy et al., 2001), may indicate a general hydrological response to logging in which reductions in summertime low flows do not manifest until after a period (decades) of forest regrowth. The length of this period would conceivably vary with climate, elevation, and the physiology of the dominant tree species. In operational contexts, hydrologists commonly model hydrological recovery as an asymptotic function with decreasing treatment effect through time (e.g., Hudson, 2000). However, a more realistic model may be a reduction in growing season streamflow during the medium term, followed by a trend back toward conditions associated with mature forest – a trajectory modelled using the "Kuczera curve" in Australia (Vertessy et al., 2001). Further research is required at both the stand and catchment scales to clarify the time scales of and specific conditions under which this response occurs.

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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 km^2 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. In addition, following forest harvesting in these catchments, forest regeneration was able to occur with minimal effect of other forest disturbances. This situation is less likely to hold in larger catchments, where logging, insect infestation, and fires are likely to result in a more ongoing forest disturbance history, thereby complicating the associated hydrological response. Further analysis of existing paired catchment studies with an emphasis on how the effects of logging on summertime low flows change through time would allow for a more complete understanding of how generalizable these trends may be.

Delayed decreases in summer flows following harvest have now been documented for headwater streams in both rain-dominant and hybrid regimes (Perry and Jones, 2017) and snowmelt-dominant catchments (this study). However, studies that capture a broader range of climatic conditions and forest types would provide a more robust basis for generalizing these results. For example, Upper Penticton Creek experiences a relatively dry interior climate in which snow tends to disappear earlier in openings than under forest. In contrast, in addition to having different tree species, snow-dominated sites with maritime climates often experience longer snow persistence in openings than under forest (Dickerson-Lange et al., 2017), which could lead to higher late-summer flows, at least initially. In addition, a stronger understanding of the underlying processes is still required to define the broader applicability of these findings. In particular, further research is required to quantify changes in transpiration and interception loss as forests regenerate across a range of tree species and climatic contexts and to understand how transpiration rates vary spatially within watersheds.

Impacts of Logging on Fish Habitat

Many studies have explored the hydrological impacts of logging, but few have tried to quantify the effects of changes in streamflow on fish habitat availability. The current analysis is novel because it coupled paired-catchment analyses with eco-hydraulic instream flow assessments, allowing for the quantification of the effects of logging on fish habitat

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availability, as measured by *WUA*. Historically, paired-catchment analyses that examined forest harvesting have generally led to the understanding that logging increases streamflow (Moore and Wondzell, 2005), at least for the first five to ten years, which could be interpreted as a benign or even favourable effect of logging on fish habitat availability. The results from this study show otherwise: not only does logging have a delayed effect of reducing summertime streamflow, it reduces summertime streamflows at a time when further reductions to already low flows (<10% *MAD*) are likely to cause severe degradation in aquatic habitat (Tennant, 1976). The eco-hydraulic modelling also indicates that forest harvesting reduces *WUA* at a time when fish may also be subject to additional stresses associated with logging, such as increased stream temperatures (e.g. Gomi, Moore, & Dhakal., 2006). These reductions in streamflow and *WUA* were more prevalent in Dennis Creek, which has a longer post-harvest time series. Comparatively, few significant impacts to modelled fish habitat were observed in 241 Creek. Although this work was conducted in relatively small salmonid-bearing streams (McCleary & Hassan, 2008), the relative importance of these impacts to juvenile fry and parr should not be understated, as the productivity of rearing fry and parr in rivers is an important determinant of the health of adult populations (Bradford, 1995).

The assumptions and potential pitfalls involved with eco-hydraulic modelling need to be considered when interpreting the *WUA* results. Firstly, when applying the eco-hydraulic modelling results through time, as done here, it is assumed that channel hydraulics have remained consistent through time. While it is possible that the harvesting operations did modify channel dynamics, we believe that such changes would not invalidate the qualitative finding that summer low flows in these catchments lie in a range in which decreases in flow result in commensurate changes to physical habitat availability. One of the strengths of hydraulic simulation modelling is that it isolates the specific effects of altered flows while habitat structure is held constant; in that respect the reduction in available habitat at lower flows is unambiguous. Secondly, a key assumption inherent in eco-hydraulic modelling is the premise that physical habitat is responsible for limiting the size of fish populations (Mathur et al., 1985; Rosenfeld & Ptolemy, 2012). Because fish habitat tends to be most sensitive to reductions in streamflow in small streams (Bradford & Heinonen, 2008; Hatfield & Bruce, 2000), it is reasonable to assume that physical fish habitat availability is limiting fish populations in 241 Creek and Dennis Creek.

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There are also issues with the habitat suitability curves used in eco-hydraulic modelling approaches. For example, Lancaster and Downes (2010) and Railsback (2016) argued that habitat suitability curves are used inappropriately to infer habitat use by fish. In scenarios in which prey availability is likely a more important factor limiting populations than habitat availability, habitat suitability relationships generated using bioenergetic modelling approaches may be more appropriate (Rosenfeld & Ptolemy, 2012; Rosenfeld, Beecher, & Ptolemy, 2016). In addition, alternative modelling techniques that explicitly consider the effects of stream temperature on fish health and growth (e.g. Leach, Moore, Hinch, & Gomi., 2012; Penaluna et al., 2015) may be more relevant if stream temperature limits fish populations.

Habitat suitability modelling also assumes that the chosen habitat suitability curves are appropriate for the fish populations in question. The habitat suitability curves used in this study were generated from measurements taken from many streams and rivers throughout BC, most of which are likely much larger than the streams considered here, which are near the threshold size for streams that can support rainbow trout (Trotter, 2000; McCleary & Hassan, 2008). Considering that small streams typically constitute over 70% of the channel length within a stream network (Wohl, 2017), future work should also focus on improving methods for quantifying fish habitat in headward portions of the stream network.

The eco-hydraulic modelling conducted as a part of this study provides important information in regard to how changes in flow limit the availability of physical fish habitat. However, it is important to note that, in addition to physical fish habitat, there are many other factors that are potentially influenced by logging, such as stream temperature and/or prey availability, that could also simultaneously affect fish populations. To fully understand how fish populations are being affected in these watersheds, the eco-hydraulic modelling results presented herein should be interpreted along with additional work that evaluates the effects of stream temperature and prey availability on fish populations. This is consistent with contemporary instream flow assessments, which should use eco-hydraulic modelling results as one of a suite of tools the effects of changes in streamflow on fish populations (Reiser & Hilgert, 2018).

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CONCLUSION

This study evaluated how logging affected summertime low flows and fish habitat in two snowmelt-dominant headwater catchments of the Pacific Northwest. For the first one to two decades following the onset of forest harvesting, minimal changes in summertime low flows were observed. As time progresFsed following the onset of logging and the extent of logging increased to approximately 50% of the catchments, reductions in daily summertime low flows became more significant. In addition, the seasonal low-flow yields were significantly lower in both catchments during the final treatment period, when 47% of 241 Creek and 52% of Dennis Creek had been harvested. These results indicate that the concept of asymptotic hydrological recovery as time progresses following logging is not suitable for understanding the impacts of forest harvesting on summertime low flows. This study did not directly address the hydrological processes that could be causing the reduction in low flows observed in these three catchments. However, considering the temporal patterns of streamflow change, we hypothesize that the delayed reductions in summertime low flows observed in this study are likely caused by advanced snowmelt timing following logging in combination with a recovery of transpiration and interception loss in regenerating forests. Further work should explicitly test this hypothesis.

The instream flow assessments show that the summertime low-flow period is a time when reductions in streamflow correspond to relatively large reductions in physical fish habitat availability. When coupled with the paired catchment analyses, this analysis indicated reductions in modelled fish habitat of around 20-50% in one of the study creeks during the final treatment period. The interpretation of these results should acknowledge the uncertainty associated with applying generalized rainbow trout habitat suitability curves to small streams.

Data availability

The data that support the findings of this study are openly available at http://doi.org/10.5281/zenodo.3306253.

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Table 1: Summary of paired catchment studies that consider low flows in snowmelt-dominant watersheds of western North America.

Study Watershed	Author	Forest Type	Drainage Area (km^2)	Elevation Range (masl)	Harvest method	Forest cover removed (%)	Post Harvest Period Studied (yr)	Low-flow metric	Magnitude of change in low-flow metric
Camp Creek, BC	(Moore and Scott, 2005; Cheng, 1989)	Lodgepole and ponderosa pine	33.9	1070 - 1920	Clearcut	27	18	Monthly, 7-day low flows	No significant change
Upper Penticton Creek, BC	(Winkler et al., 2017)	Lodgepole pine, Engelman spruce	4.9	1614 - 2021	Clearcut	47	7	Monthly	July (-17% not significant), Aug (-20% not significant), Sep (none), Oct (+45%)
Mica Creek Experimental Watershed - C1, Idaho	(Hubbart et al., 2007)	Mixed coniferous	1.4	1205 - 1528	Clearcut	48	4 (after road construction), 4 (after harvest)	July - Oct yield	Road construction (none), Harvest: (+5% - considered negligible).
Mica Creek Experimental Watershed - C2, Idaho	(Hubbart et al., 2007)	Mixed coniferous	1.8	1201 - 1612	Partial-cut	24	4 (after road construction), 4 (after harvest)	July - Oct yield	Road construction (+28%), Harvest (none).
Marmot Creek Research Basin - Cabin Creek, AB	(Swanson et al., 1986; Harder et al., 2015;)	Mixed coniferous - alpine	2.4	1700 - 2824	Clearcut	23	33	Monthly, Frequency of flows < $Q10^{\dagger}$	Aug (+4% not significant), Sep (+11% not significant), no change in frequency <Q10
Marmot Creek Research Basin - Twin Creek, AB	(Harder et al., 2015)	Mixed coniferous - alpine	2.8	1700 - 2824	Honeycomb	17	42	Frequency of flows < Q10	No significant change
Wagon Wheel Gap, Colorado	(Van Haveren, 1988)	-	0.8	2818-3338	Clearcut	100	7	Annual 30 day low flow	No significant change

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Coon Creek, Wyoming	(Troendle et al., 2001)	Engleman spruce, subalpine fir, lodgepole pine	16.8	2682-3322	Clearcut	24	5	Monthly	Jul (-1% not significant), Aug (+14% not significant), Sep (+20% not significant)

† Denotes the 10% quantile flow (ie. 90% of flows exceed this value)

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Table 2: Key biophysical characteristics of the study catchments.

	240 Creek	**241 Creek**	**Dennis Creek**
Area (km^2)	4.94	4.50	3.73
Elevation range (m)	1,600 – 2,000	1,600 – 2,000	1,700 – 2,140
Dominant aspect	South	South	West
Pre-harvest forest species (% of catchment area)	Lodgepole pine (77%), Englemann spruce (6%), Sub-alpine fir (10%)	Lodgepole pine (88%), Englemann spruce (11%)	Lodgepole pine (18%), Englemann spruce (68%), Subalpine fir (3%)
Pre-harvest average stand age (years)	130	145	153
Average stand height (m)	22	20	23

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Table 3: Forest development history at Upper Penticton Creek.

Period	241 Creek			Dennis Creek		
	Roads	Clearcut	Total	Roads	Clearcut	Total
	(Cumulative Area Clearcut or Road Development [% of watershed area])					
F† 1992		4	4			
W 1995-1996	2		6	3	7	10
W 1998 -1999		12	18		11	21
1999-2000††				1	30	52
W 2002-2003		10	28			
W 2006-2007	1	18	47			

† F represents fall and FW represents fall through winter.
†† Logging occurred throughout this period.

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Table 4: Treatment periods for both study creeks.

Period	241 Creek			Dennis Creek		
	Period	**Activity**	**Cumulative Harvest (% Watershed Area)**	**Period**	**Activity**	**Cumulative Harvest (% Watershed Area)**
1	1984 - 1992	calibration	0	1984 - 1994	calibration	0
2	1993 - 2000	ongoing logging	18	1995 - 2000	ongoing logging	52
3	2001 - 2007	ongoing logging	47	2001 - 2008	post-logging	52
4	2008 - 2017	post-logging	47	2009 - 2017	post-logging	52

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Table 5: Mean discharge values recorded at the Water Survey of Canada gauges during the instream flow assessment field trips.

Location	Date	Q (L/s)	Q (% *MAD*)
241 Creek	25-26 Jun	22.6	39.6
	18-Jul	4.3	7.5
	21-22 Aug	0.0	0.0
	29-Sep	0.0	0.0
Dennis Creek	26-27-Jun	25.1	44.8
	19-Jul	1.6	2.9
	23-24-Aug	0.0	0.0
	29-Sep	0.0	0.0

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Table 6: Mesohabitat classifications used for the instream flow modelling.

Location	Mesohabitat Classification	No. Transects	Proportion of Reach (%)
241 Creek	Riffle	6	17
	Complex riffle	5	15
	Glide	1	16
	Complex glide	6	29
	Pool	8	23
Dennis Creek	Riffle	9	34
	Glide	6	47
	Pool	5	19

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Table 7: Analysis of covariance results for July 1 – September 30 low-flow yield and the frequency of low-flow days.

Site	Treatment Period	Jul 1 – Sep 30 low-flow yield ANCOVA		Number of low-flow days ANCOVA	
		p-value	Treatment effect† (% change in yield)	p-value	Treatment effect†† (Change in number of annual low-flow days)
241 Creek	1993 - 2000	0.51	7	0.027	-6
	2001 - 2007	0.26	-4	0.84	2
	2008 - 2017	0.047	-16	0.37	6
Dennis Creek	1995 - 2000	0.55	-11	0.44	0
	2001 - 2008	0.27	-24	0.62	4
	2009 - 2017	0.002	-43	0.004	19

† Represents the mean change over the treatment period relative to predicted yield.

†† Represents the mean annual change over the treatment period relative to predicted yield. Positive values indicate an increase in the number of days when mean daily streamflow was < 10% MAD.

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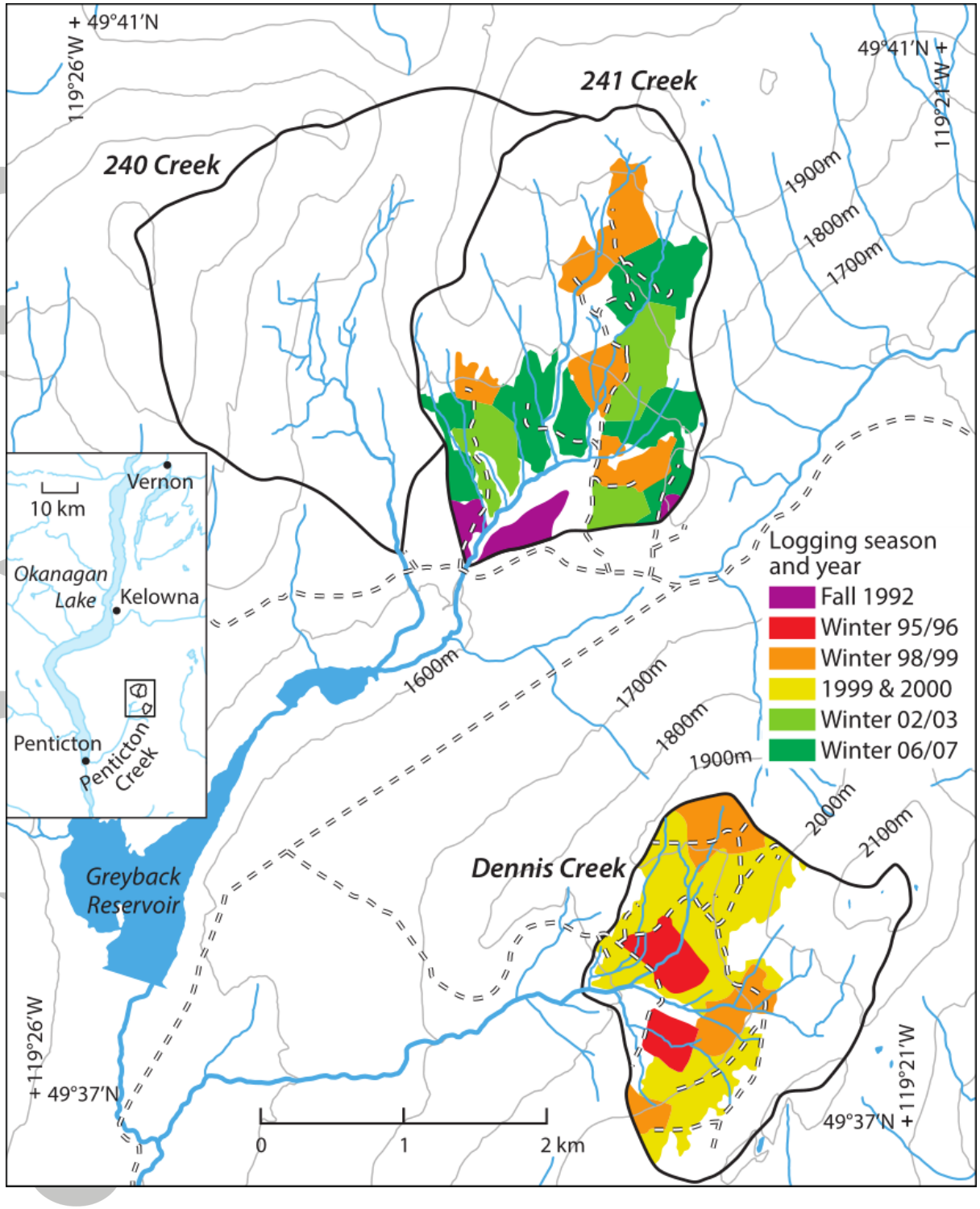

Figure 1. Catchment boundaries and clearcut extents at Upper Penticton Creek,

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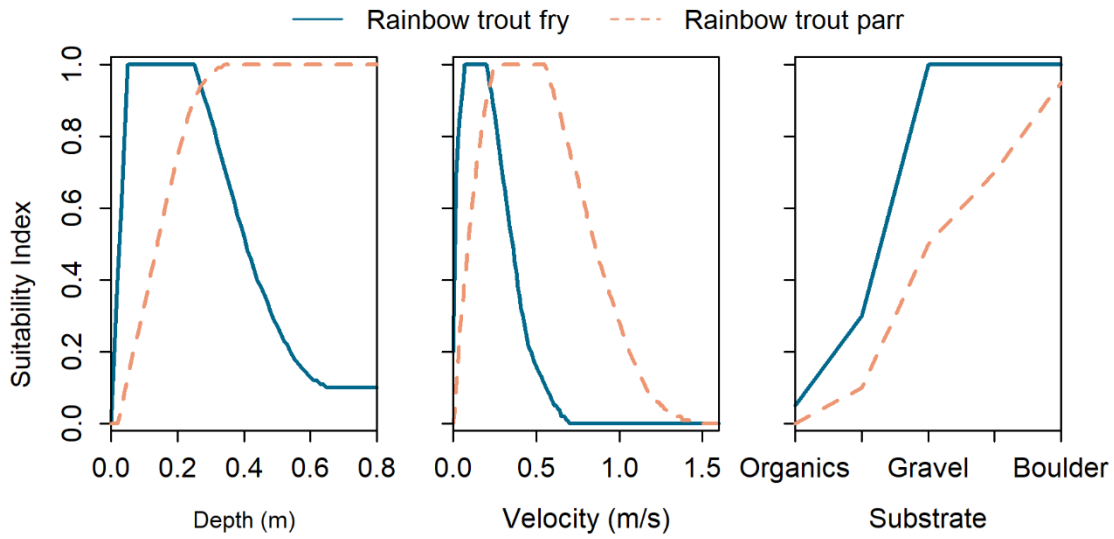

Figure 2. Habitat suitability curves for rainbow trout fry and parr. Source: Ron Ptolemy, Instream Flow Specialist, BC Ministry of Environment.

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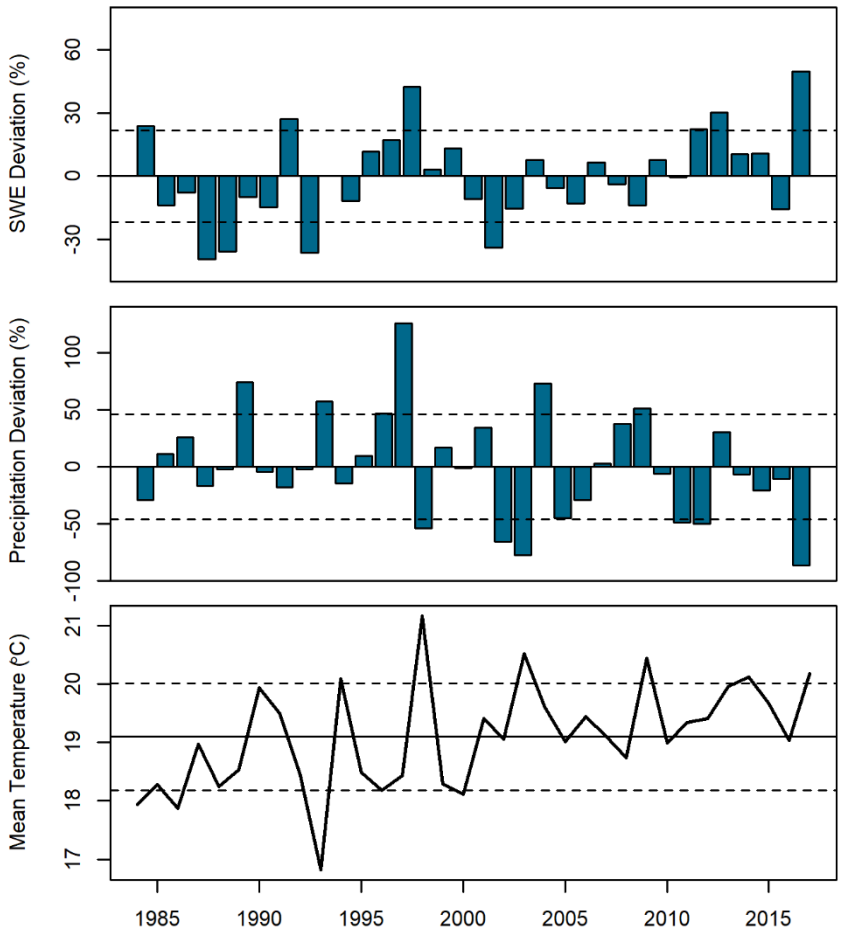

Figure 3. Deviation from mean April 1 snow water equivalent (SWE), total July 1 – September 30 precipitation, and mean July 1- September 30 daily temperature. Mean values calculated over the 1984-2017 study period. Dashed lines indicate ± one standard deviation. SWE data recorded from the Greyback Reservoir (2F08) snow course (BC Ministry of Environment). Precipitation and temperature data recorded at the Penticton Airport (WMO ID 71889) (Environment Canada).

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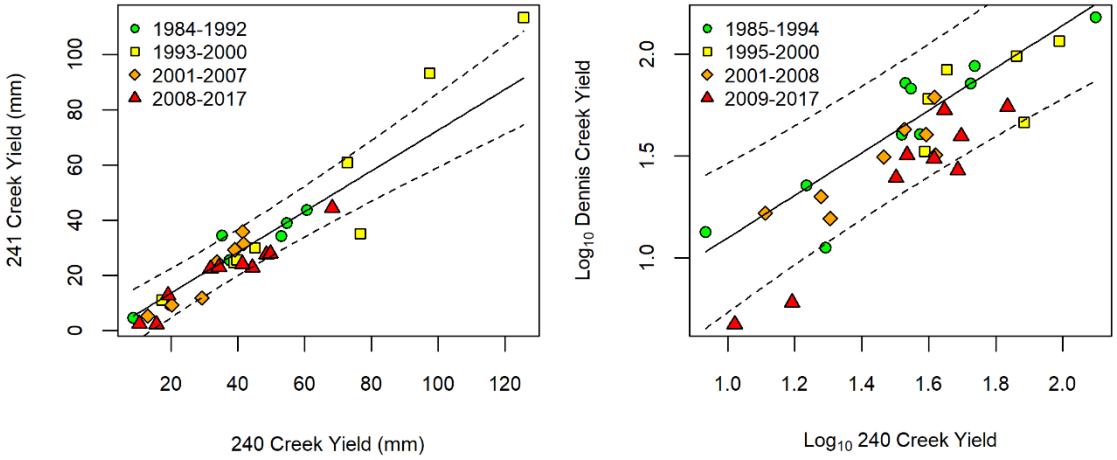

Figure 4. Scatterplots of July-September water yield. The solid black line represents the pre-harvest regression and the dashed black lines indicate 90% prediction intervals.

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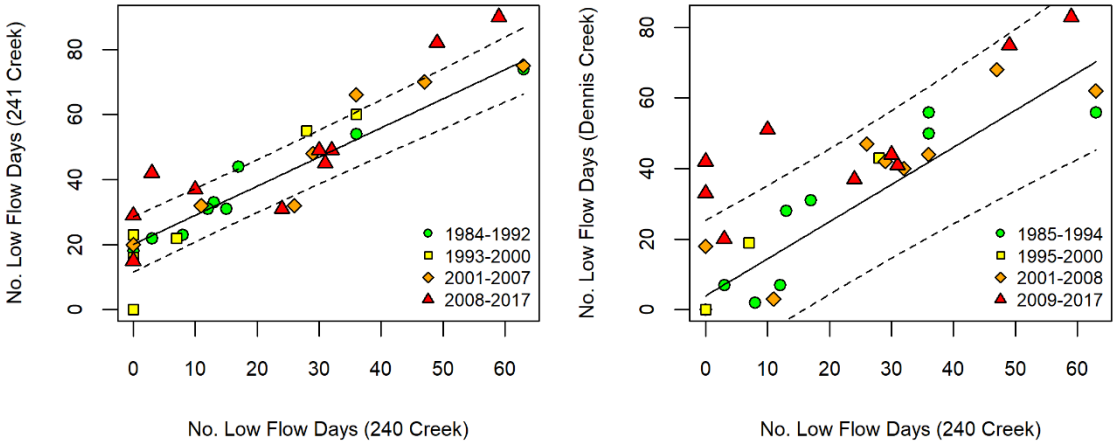

Figure 5. Scatterplots of the frequency of the occurrence of low-flow days. The solid black line represents the pre-harvest regression and the dashed black lines indicate 90% prediction intervals.

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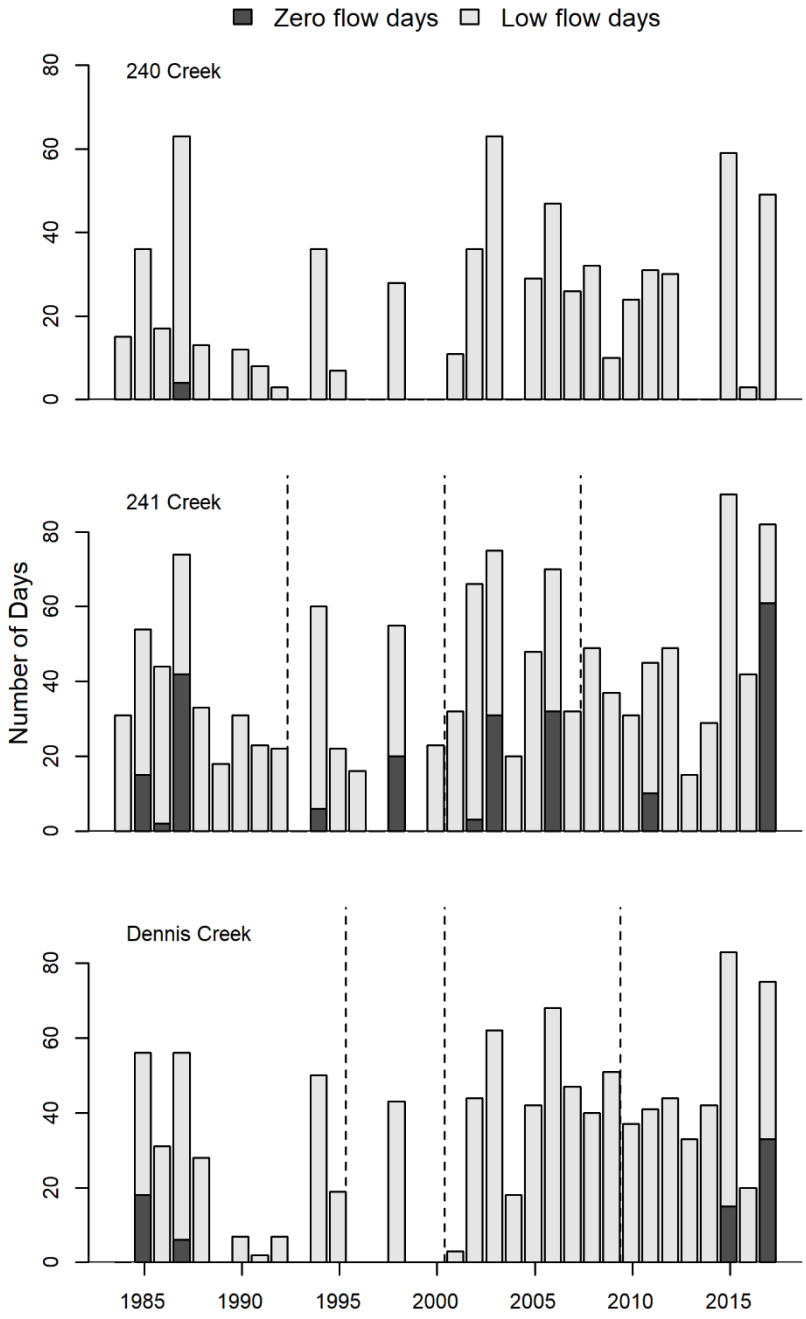

Figure 6. Annual occurrence of low flow days. Low flow days are defined when mean daily streamflow is < 10% MAD. Vertical dashed lines indicate different treatment periods in 241 Creek and Dennis Creek.

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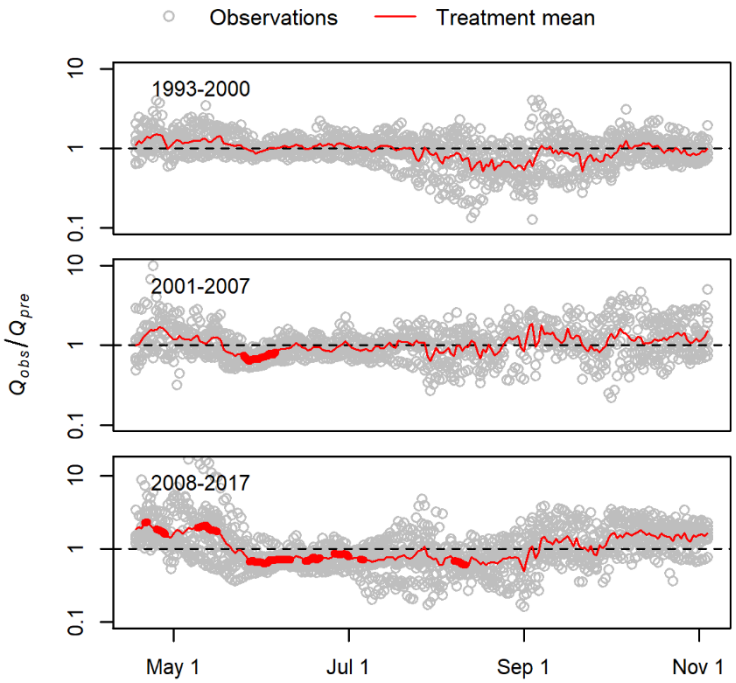

Figure 7. Day-of-year ANCOVA results for 241 Creek comparing each treatment period to the pre-harvest period with a logarithmic y-axis. Significant results at $p < 0.05$ for treatment means are denoted by the bold

lines.

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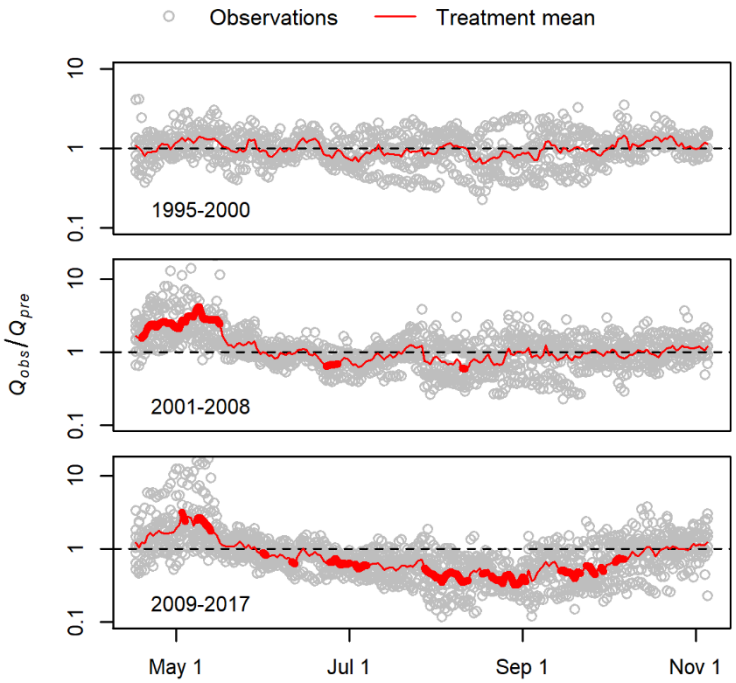

Figure 8. Day-of-year ANCOVA results for Dennis Creek comparing each treatment period to the pre-harvest period with a logarithmic y-axis. Significant results at $p < 0.05$ for treatment means are denoted by the bold lines.

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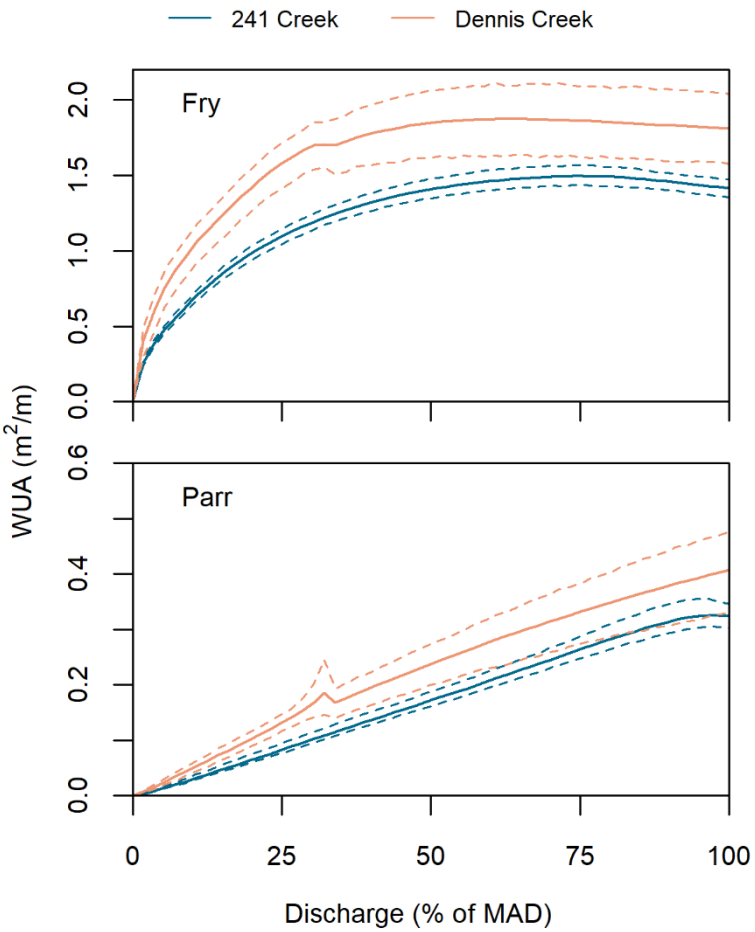

Figure 9. *WUA* – discharge relations. Solid line is predicted value and dashed lines indicate 95% confidence intervals based on bootstrapping.

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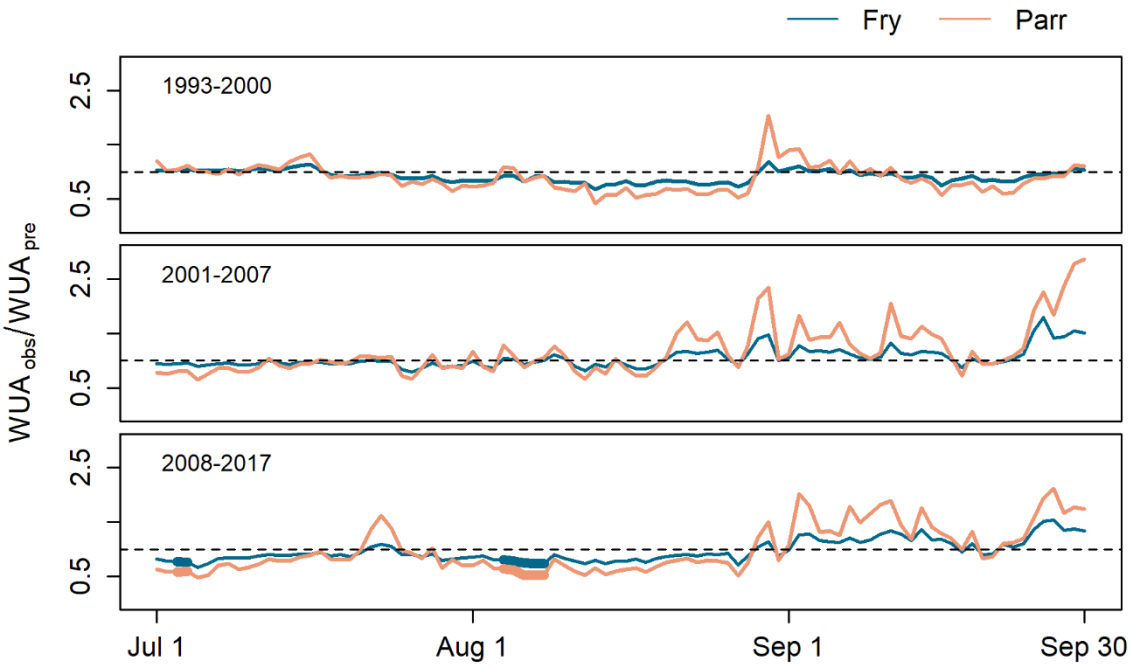

Figure 10. Time series of modelled fish habitat based on predicted and observed streamflow at 241 Creek. Bold lines indicate significant results.

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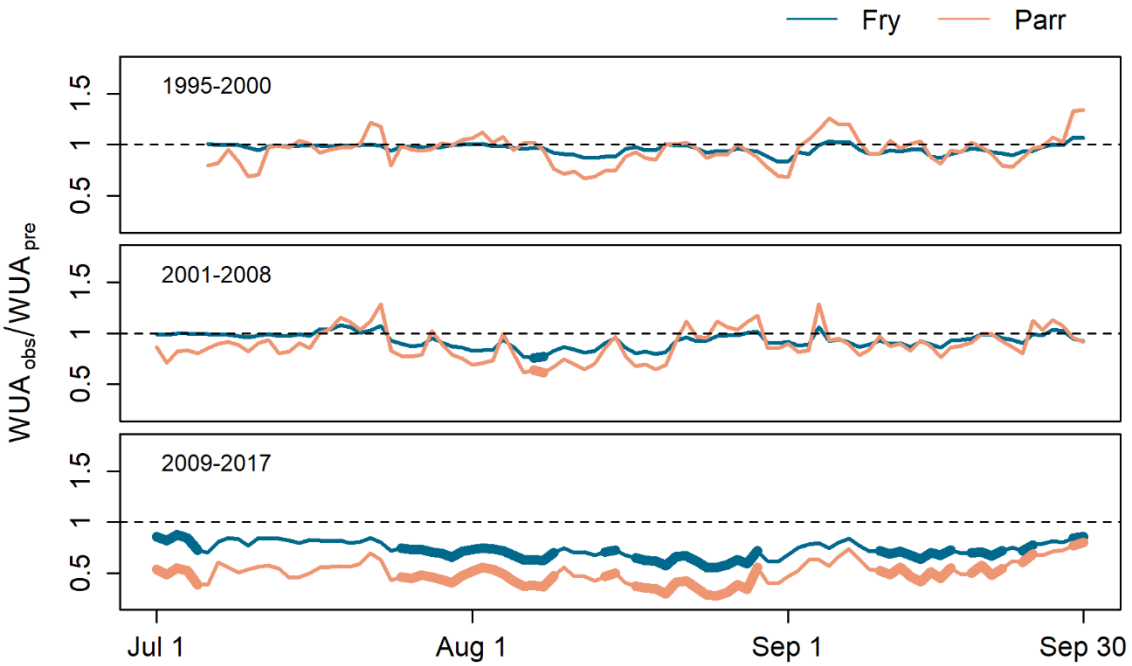

Figure 11. Time series of modelled fish habitat based on predicted and observed streamflow at Dennis Creek. Bold lines indicate significant results.

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