Subject Matter Expert Report: Mainstem dewatering. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout Population.

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EXECUTIVE SUMMARY

Abundances of both juvenile and adult life stages of Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*; WCT) in the upper Fording River (UFR) were substantively lower in 2019 than 2017, indicating a large decline during that two-year period (the Westslope Cutthroat Trout Population Decline Window, also referred to as the Decline Window). Teck Coal Limited (Teck Coal) initiated the "Evaluation of Cause" (EoC) to determine whether and to what extent various stressors and conditions played a role in the decline. One of several potential stressors that has been identified is dewatering of habitat in the mainstem UFR, which could cause stranding and potential mortality of fish. Mainstem dewatering occurs during low flow periods in the fall and winter in two main reaches (referred to here as the southern and northern drying reaches). Redd dewatering may also occur in the UFR in late spring or summer when flows during the incubation period are low enough to dewater areas used as spawning redds by fish. The potential impacts to fish from channel dewatering are stranding or isolation, which can lead to death of stranded individuals, and which can, in turn, lead to population decline if a large proportion of the population is affected.

Thus, the specific impact hypothesis evaluated was:

- Did dewatering of the UFR mainstem habitats cause or contribute to the observed WCT population decline?

This scope of work addresses the EoC assessment of dewatering in the UFR mainstem in relation to stranding risk to WCT. The timing and extent of dewatering in the UFR mainstem may also influence fish migration, including passage to preferred overwintering areas. Fish migration has been evaluated in the Fish Passage SME report for the EoC (Harwood *et al.* 2021). The approach for stranding risk to WCT included a literature review of dewatering and effects on fish, followed by a review of known stranding events in the UFR during the Decline Window, and an assessment of river segments or 'drying reaches' in the UFR known to seasonally dewater and the extent and timing of that dewatering. Flow records were evaluated to assess whether conditions in the UFR mainstem that can result in stream dewatering were different during the Decline Window relative to preceding years. For the purposes of this report, the term 'dewatering' is used equivalently to the term 'drying' and includes consideration of both natural and human-caused factors that can result in cessation of continuous surface flow in affected portions of a stream.

To bound the potential effect of dewatering to WCT mortality in the Decline Window, two potential effects of dewatering were evaluated: stranding and redd dewatering. First, seasonal dewatering in the fall and winter periods could cause WCT adult and juvenile mortality through stranding in the drying reaches during the summer rearing, fall migration, and overwintering periods. A combination of fish use data and data on the extent and timing of dewatering in the northern and southern drying reaches was used to estimate different scenarios of possible WCT mortality from stranding. Two scenarios were evaluated including one scenario that assumed that mortality from stranding would be equivalent to the proportion of the length of UFR that dried each year, and a second scenario that incorporated the historical relative fish use information during the rearing and overwintering periods in river

segments S7 and S9 that overlap with the southern and northern drying reaches, respectively. The estimates of mortality are considered upper bounds on potential effect since they do not include behavioural responses of fish to stage declines. We do not expect every fish in segments S7 and S9 would be stranded, but this estimation method provided a simple evaluation of whether stranding in the drying reaches may explain the WCT population decline. The purpose of these estimates was specific to the EoC and the identification of possible causes of the decline; the estimates should not be taken as true estimates of stranding for other assessment purposes.

The second WCT life stage that could be affected by dewatering is buried eggs and alevins during the spring/summer incubation period. The potential for redd dewatering was assessed with transect data from the upper Fording River instream flow study (Healey *et al*. 2020) to quantify the amount of the mainstem UFR that remained wetted through the incubation period. This allowed an estimate of annual "effective spawning habitat", which was compared across years, including in the Decline Window and historical periods to evaluate whether redd dewatering may have played a role in the decline of WCT juveniles.

Dewatering of the UFR mainstem during fall and winter months has been observed dating back to the 1970s in two main areas, the northern drying reach near station FR_FR1 and the southern drying reach near station FR_FR4 and downstream of FR_FRCP1SW (Lister and Kerr Wood Leidal 1980). Winter water quality surveys by Teck document dewatering at stations FR_FR4 and FR_FR1 dating back to the 1970s, which typically begins in November or December each year, with the northern section typically drying several weeks before the southern section. These incidents of channel dewatering are not unique to the Fording River and are also known to occur elsewhere in the region, such as within the upper reaches of the Kootenay River and the Elk River. The literature review also found that the dewatering of stream channels at predictable times and locations each year is common globally and comprises a large portion of many river networks. Stream salmonids are adapted to seasonal, periodic changes in stream drying, and exhibit behaviours that limit their exposure to harmful environmental conditions. In the UFR, WCT initiate movements in the fall to overwintering habitats as water temperature declines, with the majority of the middle and upper UFR mainstem population overwintering in deep water habitats in the S6 pools, the Clode Flats area (S8), and Henretta Pit Lake. However, extreme drying events that are outside of normal timing or extent have the potential to negatively impact individuals and populations.

Incidents of stranding in the UFR drying reaches causing WCT mortality have been observed within and prior to the Decline Window, indicating that stranding events during seasonal flow recession are common and likely place strong selective pressures on WCT to migrate to overwintering areas outside of these drying reaches. A documented stranding event of WCT in the mainstem UFR during the Decline Window occurred in 2018 in the Kilmarnock Phase 1 Discharge Channel and the lower portion of the Fording River Side Channel. This event occurred between August 30 and September 7, 2018, when flow changes caused isolation and stranding of WCT. Stranding was also recorded by consultants working in the area of the southern drying reaches during the Decline Window. Specifically, three observations were made of stranded or isolated fish in September and

October 2018. On September 10, 2018, staff from Minnow Environmental Ltd. were conducting field sampling as a part of the Fording River Local Environmental Effects Monitoring Program and noted dewatering of an approximately 800 m section of the UFR mainstem near FR_FRCP1SW. Within this section, a total of 15 WCT mortalities were observed with individuals ranging in size from 80 to 190 mm. Stage data from the southern drying reach was evaluated, which shows that there was a drop in stage from September 8 to 10, 2018 that is not seen in the stations further upstream (FRD-SD01, FRD-SD02, and FRD-SD03) over the same dates. This drop in stage overlaps with observations of stranding on September 10, 2018.

The timing and extent of dewatering was surveyed monthly by Lotic Environmental Ltd. (Lotic) during the Decline Window beginning in August 2017 in the southern drying reach and in August 2019 in the northern drying reach (shown in Appendix A: Zathey and Robinson 2021). The extent and timing of dewatering of the southern drying reach varied by year of survey between 2017-2019, with date of first dewatering at FR_FRCP1SW observed to be comparatively early in 2018/2019. First dates of drying were observed on December 14, 2017, September 10, 2018 and January 6, 2020 for the 2017/2018, 2018/2019, and 2019/2020 fall-winter seasons. The date of rewetting at FR_FRCP1SW occurred on March 24, 2018, between March 12 and 19, 2019, and on April 14, 2020. Dewatering was observed at the nearby station FR_FR4 in the 2018/2019 season only, which was first observed on January 29, 2019. FR_FR4 remained wetted throughout the 2017/2018 and 2019/2020 fall-winter seasons. The maximum extent of drying in the southern drying reach was 2,926 m during the 2018/2019 overwintering period. The northern drying reach downstream of station FR_FR1 was first surveyed by Lotic in 2019/2020 and was observed to first go dry on November 18, 2019. The maximum extent of dewatering observed in 2019/2020 in the northern drying reach was 970 m. Rewetting at FR_FR1 occurred on April 14, 2020 on the same day as FR_FRCP1SW in the southern section. Dewatering and subsequent rewetting of the UFR drying reaches at FR_FRCP1SW, FR_FR4 and FR_FR1 occurs when flows at FR_FRNTP are in the range of 0.3 to 1.0 m^3/s. This range of flows is common in the winter flow record in the historical period. However, flows at the time of drying at a single station vary among years, which indicates that there are factors other than upstream flow (e.g., groundwater inflows, temperature) that contribute to determining the timing and extent of drying.

Longer-term records of flow at the Fording River at the Mouth Water Survey of Canada (WSC) gauge suggest that flows during the Decline Window were lower than the recent historical period (since 2000) during the summer rearing, fall migration and overwintering periods, which indicates potential for stranding mortality from drying to contribute to the observed WCT decline. Average daily flows at the WSC gauge in September to mid-October 2018 (3.28 m^3/s) were lower than the average daily flows in September to mid-October since 1970 of 4.35 m^3/s, and lower than all preceding years since 2003 other than 2017. Fording River flow in summers of 2017 and 2018 in the months leading up to fall drying were also low relative to the historical period, while flows during overwintering in 2018 and 2019 were lower than the historical average and lower than most years since 2000. However, early fall low flow years were observed in the historical period prior to 2004, including in fall 1979 when dewatered sections were observed by Lister and Kerr Wood Leidal (1980). Fording River flows during the winters of the 1970s and 1980s were also commonly lower than the more recent period, including in the Decline Window, which suggests that overwintering conditions may have been improving in the last twenty to thirty years.

A combination of the maximum extent of observed dewatering in the drying reaches and relative fish use in S7 and S9 was used to evaluate whether stranding within the northern and southern drying reaches during the Decline Window could have affected a substantial portion of the UFR WCT population and partially or fully explain the decline. Two scenarios were developed to provide an upper-bound to potential effects of stranding from dewatering using simple assumptions of potential stranding mortality. Potential stranding mortality was estimated to be up to 7.0% of the WCT population during the overwintering period in 2018/2019 based on the maximum extent observed dewatered relative to the length of the UFR. In contrast, potential overwinter mortality was estimated to be up to 4.8% in 2017/2018 and 6.3% in 2019/2020 using this same method. Factoring in assumptions of fish distributions in segments S7 and S9 in the rearing and overwintering periods decreased the estimates of potential mortality during the overwintering period to a maximum of 2.5% of the WCT population in 2018/2019. Potential mortality from stranding during the rearing period in 2018/2019 was estimated to be up to a maximum of 2.1 to 2.3% of the WCT population (depending on scenario), with 0% mortality estimated in fall 2017 and fall 2019. The estimate for potential mortality from stranding during the rearing period was highest in 2018/2019 because drying was observed to occur beginning in September, which overlaps with the WCT rearing distribution prior to overwintering migration. These estimates are likely to be biased high because fish are expected to move in response to drying such that many fish will not be stranded.

Overall, the requisite conditions to cause, with respect to timing, duration, and location were met in that dewatering was observed early in the fall of 2018 and the extent of drying was greatest in winter 2018/2019 than in recent previous years with drying data. Flows in fall and winter 2017/2018 and 2018/2019 were also lower than most years since 2000. The dewatering in fall 2018 also coincided with observations of WCT stranding in the UFR mainstem and the Fording River Side Channel. However, the requisite condition related to spatial extent and intensity were not met for sole cause of the decline, particularly with respect to the proportion of fish affected, given a maximum of 2.3% of the UFR WCT population during the rearing period and 7.0% of the population when factoring in drying during overwintering. We conclude that dewatering in the UFR mainstem causing stranding mortality is unlikely to have been the primary cause of the WCT population decline. However, because dewatering occurred during the WCT summer rearing period in 2018, it is possible that stranding mortality from drying in the fall of 2018 and winter of 2018/2019 was greater than in other years and therefore could have contributed to the WCT decline for both adults and juveniles. However, due to uncertainty in the relationship between surface flow and drying it is unclear whether the timing and extent of drying may have occurred as early and widespread in the historical window as it did in 2018. Further, UFR mainstem drying events have been observed in the historical period back to the 1970s.

Therefore, our conclusion that mainstem dewatering may have contributed to the WCT population decline has some uncertainty.

Regarding the potential for redd dewatering to have been a sole or contributing cause for WCT population decline, there is no evidence to suggest that conditions during the Decline Window were worse than the historical period. Redd dewatering in the spring and summer is unlikely to have been a sole or contributing cause of the WCT population decline.

There is uncertainty in the assessments made here, mainly due to the limited amount of data on stranding, and the gaps in the data record for the timing and extent of drying; these limitations required use of coarse estimates for potential WCT mortality based on worse-case scenarios. There is also uncertainty in the environmental drivers of drying in the UFR mainstem, including the link between flow and drying timing and extent. Despite the data uncertainties, we believe the evaluation methods used are conservative, and therefore we consider the conclusions with respect to the requisite conditions to be reasonable.

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ACRONYMNS AND ABBREVIATIONS

DFO - Fisheries and Oceans Canada

EoC – Evaluation of Cause

FRO – Fording River Operations

SME – Subject Matter Expert

UFR – Upper Fording River

WCT – Westslope Cutthroat Trout

READER'S NOTE

What is the Evaluation of Cause and what is its purpose?

The Evaluation of Cause is the process used to investigate, evaluate and report on the reasons the Westslope Cutthroat Trout population declined in the upper Fording River between fall 2017 and fall 2019.

Background

The Elk Valley is located in the southeast corner of British Columbia (BC), Canada. It contains the main stem of the Elk River (220 km long) and many tributaries, including the Fording River (70 km long). This report focuses on the upper Fording River, which starts 20 km upstream from its confluence with the Elk River at Josephine Falls. The Ktunaxa First Nation has occupied lands in the region for more than 10,000 years. Rivers and streams of the region provide culturally important sources of fish and plants.

The upper Fording River watershed is at a high elevation and is occupied by only one fish species, a genetically pure population of Westslope Cutthroat Trout *(Oncorhynchus clarkii lewisi)* — an iconic fish species that is highly valued in the area. This population is physically isolated because Josephine Falls is a natural barrier to fish movement. The species is protected under the federal Fisheries Act and the Species at Risk Act. In BC, the Conservation Data Center categorized Westslope Cutthroat Trout as "*imperiled or of special concern, vulnerable to extirpation or extinction.*" Finally, it has been identified as a priority sport fish species by the Province of BC.

The upper Fording River watershed is influenced by various human-caused disturbances including roads, a railway, a natural gas pipeline, forest harvesting and coal mining. Teck Coal Limited (Teck Coal) operates the three surface coal mines within the upper Fording River watershed, upstream of Josephine Falls: Fording River Operations, Greenhills Operations and Line Creek Operations.

Evaluation of Cause

Following identification of the decline in the Westslope Cutthroat Trout population, Teck Coal initiated an Evaluation of Cause process. The overall results of this process are reported in a separate document (Evaluation of Cause Team, 2021) and are supported by a series of Subject Matter Expert reports.

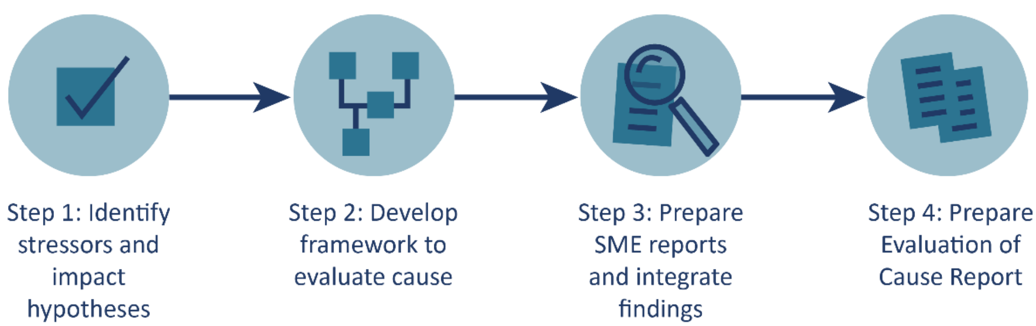
The report that follows this Reader's Note is one of those Subject Matter Expert Reports.

Monitoring conducted for Teck Coal in the fall of 2019 found that the abundance of Westslope Cutthroat Trout adults and sub-adults in the upper Fording River had declined significantly since previous sampling in fall 2017. In addition, there was evidence that juvenile fish density had decreased. Teck Coal initiated an *Evaluation of Cause* process. The overall results of this process are reported separately (Evaluation of Cause Team, 2021) and are supported by a series of Subject Matter Expert reports such as this one. The full list of SME reports follows at the end of this Reader's Note.

Building on and in addition to the Evaluation of Cause, there are ongoing efforts to support fish population recovery and implement environmental improvements in the upper Fording River.

How the Evaluation of Cause was approached

When the fish decline was identified, Teck Coal established an *Evaluation of Cause Team* (the Team), composed of *Subject Matter Experts* and coordinated by an Evaluation of Cause *Team Lead*. Further details about the Team are provided in the Evaluation of Cause report. The Team developed a systematic and objective approach (see figure below) that included developing a Framework for Subject Matter Experts to apply in their specific work. All work was subjected to rigorous peer review.



Conceptual approach to the Evaluation of Cause for the decline in the upper Fording River Westslope Cutthroat Trout population.

With input from representatives of various regulatory agencies and the Ktunaxa Nation Council, the Team initially identified potential stressors and impact hypotheses that might explain the cause(s) of the population decline. Two overarching hypotheses (essentially, questions for the Team to evaluate) were used:

- Overarching Hypothesis #1: The significant decline in the upper Fording River Westslope Cutthroat Trout population was a result of a single acute stressor[1] or a single chronic stressor[2].

[1] Implies September 2017 to September 2019.

[2] Implies a chronic, slow change in the stressor (using 2012–2019 timeframe, data dependent).

- Overarching Hypothesis #2: The significant decline in the upper Fording River Westslope Cutthroat Trout population was a result of a combination of acute and/or chronic stressors, which individually may not account for reduced fish numbers, but cumulatively caused the decline.

The Evaluation of Cause examined numerous stressors in the UFR to determine if and to what extent those stressors and various conditions played a role in the Westslope Cutthroat Trout's decline. Given that the purpose was to evaluate the cause of the decline in abundance from 2017 to 2019[3], it was important to identify stressors or conditions that changed or were different during that period. It was equally important to identify the potential stressors or conditions that did not change during the decline window but may, nevertheless, have been important constraints on the population with respect to their ability to respond to or recover from the stressors. Finally, interactions between stressors and conditions had to be considered in an integrated fashion. Where an *impact hypothesis* depended on or may have been exacerbated by interactions among stressors or conditions, the interaction mechanisms were also considered.

The Evaluation of Cause process produced two types of deliverables:

Individual Subject Matter Expert (SME) reports (such as the one that follows this Note): These reports mostly focus on impact hypotheses under Overarching Hypothesis #1 (see list, following). A Framework was used to align SME work for all the potential stressors, and, for consistency, most SME reports have the same overall format. The format covers: (1) rationale for impact hypotheses, (2) methods, (3) analysis and (4) findings, particularly whether the requisite conditions4 were met for the stressor(s) to be the sole cause of the fish population decline, or a contributor to it. In addition to the report, each SME provided a summary table of findings, generated according to the Framework. These summaries were used to integrate information for the Evaluation of Cause report. Note that some SME reports did not investigate specific stressors; instead, they evaluated other information considered potentially useful for supporting SME reports and the overall Evaluation of Cause, or added context (such as in the SME report that describes climate (Wright et al., 2021).

The Evaluation of Cause report (prepared by a subset of the Team, with input from SMEs): This overall report summarizes the findings of the SME reports and further considers interactions between stressors (Overarching Hypothesis #2). It describes the reasons that most likely account for the decline in the Westslope Cutthroat Trout population in the upper Fording River.

[3] Abundance estimates for adults/sub-adults are based on surveys in September of each year, while estimates for juveniles are based on surveys in August.

[4] These are the conditions that would need to have occurred for the impact hypothesis to have resulted in the observed decline of Westslope Cutthroat Trout population in the upper Fording River.

Participation, Engagement & Transparency

To support transparency, the Team engaged frequently throughout the Evaluation of Cause process. Participants in the Evaluation of Cause process, through various committees, included:

Ktunaxa Nation Council

BC Ministry of Forests, Lands, Natural Resource Operations and Rural Development

BC Ministry Environment & Climate Change Strategy

Ministry of Energy, Mines and Low Carbon Innovation

Environmental Assessment Office

Citation for the Evaluation of Cause Report

When citing the Evaluation of Cause Report use:

Evaluation of Cause Team, (2021). *Evaluation of Cause — Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited by Evaluation of Cause Team.

Citations for Subject Matter Expert Reports

Focus	Citation for Subject Matter Expert Reports
Climate, temperature, and streamflow	Wright, N., Greenacre, D., & Hatfield, T. (2021). *Subject Matter Expert Report: Climate, temperature, and streamflow trends. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by Ecofish Research Ltd.
Ice	Hatfield, T., & Whelan, C. (2021). *Subject Matter Expert Report: Ice. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Ltd. Report Prepared by Ecofish Research Ltd.
Habitat availability (instream flow)	Healey, K., Little, P., & Hatfield, T. (2021). *Subject Matter Expert Report: Habitat availability. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited by Ecofish Research Ltd.
Stranding – ramping	Faulkner, S., Carter, J., Sparling, M., Hatfield, T., & Nicholl, S. (2021). *Subject Matter Expert Report: Ramping and stranding. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited by Ecofish Research Ltd.
Stranding – channel dewatering	Hatfield, T., Ammerlaan, J., Regehr, H., Carter, J., & Faulkner, S. (2021). *Subject Matter Expert Report: Channel dewatering. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited by Ecofish Research Ltd.

Focus	Citation for Subject Matter Expert Reports
Stranding – mainstem dewatering	Hocking M., Ammerlaan, J., Healey, K., Akaoka, K., & Hatfield T. (2021). *Subject Matter Expert Report: Mainstem dewatering. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Ltd. by Ecofish Research Ltd. and Lotic Environmental Ltd. Zathey, N., & Robinson, M.D. (2021). *Summary of ephemeral conditions in the upper Fording River Watershed. In Hocking et al. (2021). Subject Matter Expert Report: Mainstem dewatering. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Ltd. by Ecofish Research Ltd. and Lotic Environmental Ltd.
Calcite	Hocking, M., Tamminga, A., Arnett, T., Robinson M., Larratt, H., & Hatfield, T. (2021). *Subject Matter Expert Report: Calcite. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Ltd. by Ecofish Research Ltd., Lotic Environmental Ltd., and Larratt Aquatic Consulting Ltd.
Total suspended solids	Durston, D., Greenacre, D., Ganshorn, K & Hatfield, T. (2021). *Subject Matter Expert Report: Total suspended solids. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by Ecofish Research Ltd.
Fish passage (habitat connectivity)	Harwood, A., Suzanne, C., Whelan, C., & Hatfield, T. (2021). *Subject Matter Expert Report: Fish passage. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Ltd. by Ecofish Research Ltd. Akaoka, K., & Hatfield, T. (2021). Telemetry Movement Analysis. In Harwood et al. (2021). *Subject Matter Expert Report: Fish passage. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Ltd. by Ecofish Research Ltd.
Cyanobacteria	Larratt, H., & Self, J. (2021). *Subject Matter Expert Report: Cyanobacteria, periphyton and aquatic macrophytes. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by Larratt Aquatic Consulting Ltd.
Algae / macrophytes	

Focus	**Citation for Subject Matter Expert Reports**
Water quality (all parameters except water temperature and TSS [Ecofish])	Costa, EJ., & de Bruyn, A. (2021). *Subject Matter Expert Report: Water quality. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by Golder Associates Ltd. Healey, K., & Hatfield, T. (2021). *Calculator to assess Potential for cryoconcentration in upper Fording River.* In Costa, EJ., & de Bruyn, A. (2021). *Subject Matter Expert Report: Water quality. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by Golder Associates Ltd.
Industrial chemicals, spills and unauthorized releases	Van Geest, J., Hart, V., Costa, EJ., & de Bruyn, A. (2021). *Subject Matter Expert Report: Industrial chemicals, spills and unauthorized releases. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by Golder Associates Ltd. Branton, M., & Power, B. (2021). *Stressor Evaluation – Sewage.* In Van Geest et al. (2021). *Subject Matter Expert Report: Industrial chemicals and unintended releases. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by Golder Associates Ltd.
Wildlife predators	Dean, D. (2021). *Subject Matter Expert Report: Wildlife predation. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by VAST Resource Solutions Inc.
Poaching	Dean, D. (2021). *Subject Matter Expert Report: Poaching. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by VAST Resource Solutions Inc.
Food availability	Orr, P., & Ings, J. (2021). *Subject Matter Expert Report: Food availability. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population.* Report prepared for Teck Coal Limited. Prepared by Minnow Environmental Inc.
Fish handling	Cope, S. (2020). *Subject Matter Expert Report: Fish handling. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat*

Focus	Citation for Subject Matter Expert Reports
	Trout population. Report prepared for Teck Coal Limited. Prepared by Westslope Fisheries Ltd.
	Korman, J. (2021). *Effects of capture and handling on Westslope Cutthroat Trout in the upper Fording River: A brief review of Cope (2020) and additional calculations*. Report prepared for Teck Coal Limited. Prepared by Ecometric Research.
Infectious disease	Bollinger, T. (2021). *Subject Matter Expert Report: Infectious disease. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population*. Report prepared for Teck Coal Limited. Prepared by TKB Ecosystem Health Services Ltd.
Pathophysiology	Bollinger, T. (2021). *Subject Matter Expert Report: Pathophysiology of stressors on fish. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population*. Report prepared for Teck Coal Limited. Prepared by TKB Ecosystem Health Services Ltd.
Coal dust and sediment quality	DiMauro, M., Branton, M., & Franz, E. (2021). *Subject Matter Expert Report: Coal dust and sediment quality. Evaluation of Cause – Decline in upper Fording River Westslope Cutthroat Trout population*. Report prepared for Teck Coal Limited. Prepared by Azimuth Consulting Group Inc.
Groundwater quality and quantity	Henry, C., & Humphries, S. (2021). *Subject Matter Expert Report: Hydrogeological stressors. Evaluation of Cause - Decline in upper Fording River Westslope Cutthroat Trout population*. Report Prepared for Teck Coal Limited. Prepared by SNC-Lavalin Inc.

1. INTRODUCTION

Abundances of adult and juvenile life stages of Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*) (WCT) in the upper Fording River (UFR) have been estimated since 2012 through high-effort snorkel and electrofishing surveys, supported by radio-telemetry and redd surveys (Cope *et al.* 2016). Annual snorkel and electrofishing surveys were conducted in the autumns of 2012-2014, 2017, and 2019. Abundances of both juvenile and adult life stages were substantively lower in 2019 than 2017, indicating a large decline during the two-year period between September 2017 to September 2019 (Westslope Cutthroat Trout Population Decline Window; hereafter referred to as Decline Window; Cope 2020). The magnitude of the decline as well as refinements in the timing of decline are reviewed in detail by Cope (2020) and Evaluation of Cause Team (2021).

Teck Coal Limited (Teck Coal) initiated the "Evaluation of Cause" (EoC) to assess factors responsible for the population decline. The EoC evaluates numerous impact hypotheses to determine whether and to what extent various stressors and conditions played a role in the decline of WCT. Given that the primary objective is to evaluate the cause of the sudden decline over a short time period (from 2017 to 2019), it is important to identify stressors or conditions that changed or were different from normal during the Decline Window. However, it is equally important to identify all potential stressors or conditions that did not change during the Decline Window but nevertheless may be important constraints on the population. Finally, interactions among stressors are also considered in the EoC. Where an impact hypothesis depends on interactions among stressors or conditions, or where the impact may be exacerbated by particular interactions, the mechanisms of interaction are considered as part of the evaluation of specific impact hypotheses.

A project team is evaluating the cause of WCT decline in abundance and is investigating two "over-arching" hypotheses:

- Over-arching Hypothesis #1: The significant decline in the UFR WCT population was a result of a single acute stressor[5] or a single chronic stressor[6].
- Over-arching Hypothesis #2: The significant decline in the UFR WCT population was a result of a combination of acute and/or chronic stressors, which individually may not account for reduced WCT numbers, but cumulatively caused the decline.

Ecofish Research Ltd. (Ecofish) was asked to provide support as Subject Matter Expert (SME) for an evaluation of stressors. This report investigates the potential for fish stranding in the mainstem UFR due to dewatering as a stressor that may have caused or contributed to the observed WCT decline.

[5] Implies the single acute stressor acted between September 2017 and September 2019.

[6] Implies a chronic slow change in the stressor (using 2011-2019 timeframe, data dependent).

1.1. Background

1.1.1. Overall Background

This document is one of a series of SME reports that supports the overall EoC of the UFR WCT population decline (EoC Team 2021). For general information, see the preceding Reader's Note.

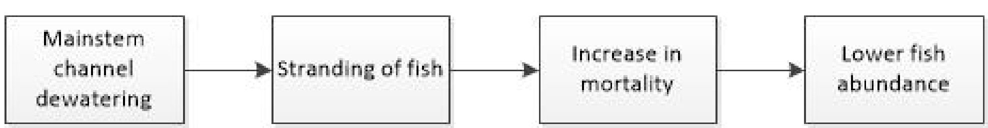
1.1.2. Report-specific Background

Dewatering of channels, or rapid changes in water level or flow (ramping), that result in stranding of fish can be caused by natural factors (e.g., changes to inflows, dry climatic conditions) or water withdrawal for mining or other water uses (e.g., hydropower, agriculture). When flows drop quickly or to a level where connectivity is lost, fish may become stranded in the interstices of exposed gravel or cobble substrate or isolated in pools (Irvine *et al.* 2009, Irvine *et al.* 2014). This can lead to mortality from suffocation, desiccation, freezing, or predation. The likelihood of fish stranding or isolation during changes in water level is dependent on fish life stage, species, wetted history of the habitat, rate of stage change (i.e., ramping rate), magnitude of stage change, substrate characteristics, bank slope, channel morphology, water temperature, time of day, and other biotic and abiotic factors (Nagrodski *et al.* 2012, Irvine *et al.* 2014). Fry are typically more sensitive to stranding than are older age classes due to the poor swimming ability of fry and their preference for shallow and low velocity habitat; however, complete dewatering may affect all age classes of fish.

In total, three SME reports assess risks to fish from stranding. This report addresses risks to fish from dewatering in the UFR mainstem and side channels that are not operationally influenced. The Channel Dewatering Report addresses risks to fish from dewatering of channels that are operationally influenced that discharge to the UFR, including a portion of one side channel of the UFR that is also operationally influenced (Faulkner *et al.* 2021a). The Ramping and Stranding Report addresses risks to fish from rapid changes in flow within the UFR mainstem (Faulkner *et al.* 2021b).

This report focuses on the risk to stranding in reaches of the UFR that can undergo seasonal drying in the fall and winter each year. The pathway of effect to WCT from dewatering in the mainstem UFR is shown in Figure 1. Note that dewatering in the mainstem UFR can also have potential effects on WCT by blocking migration to overwintering habitats. The connectivity effect pathway is addressed in the EoC Fish Passage Report (Harwood *et al.* 2021) and is not considered in this report.

Figure 1. Pathway of effect relevant to potential effects to fish from dewatering in the mainstem UFR.



1.1.3. Author Qualifications

Todd Hatfield, Ph.D., R.P.Bio.

This project is being led by Todd Hatfield, Ph.D., a registered Professional Biologist and Principal at Ecofish Research Ltd. Todd has been a practising biological consultant since 1996 and he has focused his professional career on three core areas: environmental impact assessment of aquatic resources, environmental assessment of flow regime changes in regulated rivers, and conservation biology of freshwater fishes. Since 2012, Todd has provided expertise to a wide array of projects for Teck Coal: third party review of reports and studies, instream flow studies, environmental flow needs assessments, aquatic technical input to structured decision making processes and other decision support, environmental impact assessments, water licensing support, fish community baseline studies, calcite effects studies, habitat offsetting review and prioritizations, aquatic habitat management plans, streamflow ramping assessments, development of effectiveness and biological response monitoring programs, population modelling, and environmental incident investigations.

Todd has facilitated technical committees as part of multi-stakeholder structured decision making processes for water allocation in the Lower Athabasca, Campbell, Quinsam, Salmon, Peace, Capilano, Seymour and Fording rivers; he has been involved in detailed studies and evaluation of environmental flows needs and effects of river regulation for Lois River, China Creek, Tamihi Creek, Fording River, Duck Creek, Chemainus River, Sooke River, Nicola valley streams, Okanagan valley streams, and Dry Creek. Todd was the lead author or co-author on guidelines related to water diversion and allocation for the BC provincial government and industry, particularly as related to the determination of instream flow for the protection of valued ecosystem components in BC. He has worked on numerous projects related to water management, fisheries conservation, and impact assessments, and developed management plans and guidelines for industry and government related to many different development types. Todd is currently in his third 4-year term with COSEWIC (Committee on the Status of Endangered Wildlife in Canada) on the Freshwater Fishes Subcommittee.

Morgan Hocking, Ph.D., R.P.Bio.

Morgan is Senior Environmental Scientist with Ecofish with over 20 years of experience conducting salmonid conservation and watershed resource management projects in British Columbia. For much of his career, he has studied how spawning Pacific salmon affect terrestrial biodiversity, and how this information can be used in ecosystem-based management. He uses a combination of field studies, experiments, watershed spatial data, quantitative modelling, and novel tools in ecology such as stable isotopes and environmental DNA to assess watershed status and the relationships between watershed developments and biodiversity, and has published 23 peer-reviewed articles on his work. Morgan has extensive experience in designing and implementing large-scale monitoring programs and has over 15 years of experience working with First Nations, primarily related to fisheries management in the Great Bear Rainforest.

With Ecofish, Morgan works on technical project management, community engagement, experimental design, data analysis, reporting and senior technical review on a diversity of projects such as the

Cumulative Effects Monitoring Program in the Skeena watershed (Environmental Stewardship Initiative), the Fish & Wildlife Compensation Program (FWCP) Action Plan Update (FWCP Coastal and FWCP Peace), the Site C Tributary Mitigation Program (BC Hydro) and the Ecofish environmental DNA program. Morgan is also the technical lead of the Calcite Biological Effects Program with Teck and the Teck Kilmarnock eDNA study. Morgan also holds a position as an Adjunct Professor in the School of Environmental Studies at the University of Victoria.

1.2. Objective

The objective of this report is to review the available information on effects of dewatering in the UFR mainstem. Mainstem dewatering occurs during low flow periods in the fall and winter in two main reaches (referred to here as the southern and northern drying reaches). Redd dewatering may also occur throughout the UFR in late spring or summer if flows during the incubation period become low enough to dewater areas used as spawning redds by fish. The potential impacts to fish from channel dewatering are stranding or isolation, which can lead to death of stranded individuals, and which can, in turn, lead to population decline if a large proportion of the population is affected.

Thus, the specific impact hypothesis evaluated was:

- Did dewatering of the UFR mainstem habitats cause or contribute to the observed WCT population decline?

1.3. Approach

This scope of work addresses the EoC assessment of dewatering in the UFR mainstem in relation to stranding risk to WCT. There are limited data on stranding events in the UFR mainstem during the Decline Window, and therefore, a more qualitative approach was taken to evaluate the requisite conditions for this pathway. The approach included a literature review of dewatering and effects on fish stranding and mortality (Section 2.1 and 3.1), followed by a review of documented stranding events in the UFR during the Decline Window, and an assessment of river segments or reaches in the UFR known to dewater in the Decline Window and the extent and timing of that dewatering (Section 2.2 and 3.2). Dewatering in the UFR mainstem was assessed for the southern and northern drying reaches based on field surveys by Lotic Environmental Inc. (Lotic) in 2017, 2018, and 2019 and reported in Appendix A (Zathey and Robinson 2021). Hydrologic conditions and data related to drying were also evaluated, including in the historical and Decline Window periods, to assess whether conditions in the UFR mainstem that can result in stream drying were different during the Decline Window relative to preceding years (Section 2.3 and Section 3.3). For example, if drying reach conditions were not markedly different during the Decline Window than in preceding years, seasonal dewatering of the drying reach is unlikely to be responsible for the recent WCT decline. The historical period is defined as the period of available data prior to the Decline Window and includes some information dating back to the 1970s.

Finally, to bound the potential effect of dewatering to WCT mortality in the Decline Window, two potential effects of drying were evaluated. First, seasonal drying in the fall and winter periods could cause WCT adult and juvenile mortality through stranding in the drying reaches during either the summer rearing or overwintering periods. A combination of fish use data from Teck (EoC Team 2021), and data on the extent and timing of drying in the northern and southern drying reaches (Appendix A: Zathey and Robinson 2021) was used to estimate different scenarios of possible WCT mortality from stranding in the Decline Window. This included a simple scenario where the combined extent of dewatering of northern and southern drying reaches each year was divided by the total length of the UFR (where UFR mainstem length = 55.5 km; Cope *et al.* 2016). A second scenario was also developed that incorporates the historical relative fish use information (EoC Team 2021) in river segments S7 and S9 that overlap with the southern and northern drying reaches, respectively. It was assumed that mortality from stranding would be equivalent to the proportion of fish use of each segment (S7 and S9) during the rearing or overwintering periods if drying was observed. The purpose of these estimates was specific to the EoC and the identification of possible causes of the decline; the estimates should not be taken as true estimates of stranding for other assessment purposes. The scenarios were used to place upper bounds on possible stranding within the UFR mainstem, and thus, whether stranding could have affected a substantial portion of the UFR WCT population across all age classes.

The second WCT life stage that could be affected by dewatering is during the spring/summer incubation period. The potential for redd dewatering (i.e., stranding of buried life stages) was assessed with transect data for fish habitat from the upper Fording River instream flow study (Healey *et al.* 2020) to quantify the amount of the mainstem UFR that remains wetted through the incubation period. This allowed an estimate of annual "effective spawning habitat", which was compared across years, including in the Decline Window and historical periods from 1997 to 2019. The comparison of effective spawning habitat among years allowed for an evaluation of whether redd dewatering may have played a role in the decline of WCT juveniles.

2. METHODS

2.1. Literature Review: Fish Stranding in Dewatered Reaches

A literature review was conducted to summarize physical features and major drivers of drying reaches and ephemeral side channels and their role in salmonid-supporting systems. The review summarized the literature with respect to the biological effects of channel dewatering, especially in relation to stranding of stream salmonid populations. For the systems identified, existing data were summarized to indicate whether, how frequently, and under what conditions, dewatering results in fish mortality, and if fish behavioural responses to such conditions have been documented. Similarities and differences were identified between systems described in the literature and the UFR, especially in relation to hydrology, habitat, timing and extent of dewatering, and fish stranding risk. The review also summarizes known population-level effects of fish stranding in drying reaches.

The literature review is used to provide background context related to the potential effects of the UFR drying reaches on WCT abundance in the UFR, including the environmental conditions described in the literature when dewatering is expected to result in effects to fish populations.

2.2. Dewatering and Stranding Observations in the UFR

Following the literature review, a more specific information review was conducted to summarize observations of dewatering and stranding in the UFR within the Decline Window and recent monitoring prior to the Decline Window (to 2015). Observations of fish mortality were compiled from the Teck fish incidents reporting and from other sources, including from historical periods (Lister and Kerr Wood Leidal 1980; Cope *et al.* 2016). Incidents of fish mortality reported by Teck from September 2015 to September 2019 were reviewed and classified by the likely cause of mortality including predation or natural causes, instream works (e.g., fish screen mortality), fish inventory (e.g., electrofishing mortality), stranding, and other causes. For each cause, the number of incidents, mortalities and average mortalities per incident were compiled. Note that the reporting of incidents of mortality are not standardized by the effort to document these events; effort has increased in recent years via increased frequency of on-the-ground surveys.

Two specific stranding events were reviewed in more detail, to the extent the available information would allow. The 2018 WCT stranding event in the Fording River Side Channel (Teck 2019a, 2019b) was reviewed, although this event is also presented in detail in the EoC Channel Dewatering Report (Faulkner *et al.* 2021a). Stranding information was summarized to assess the potential risk in this and other side channels during the Decline Window and in comparison to the preceding years. A second stranding event that occurred on September 10, 2018 associated with mainstem channel drying in the southern drying reach was also reviewed.

The timing and extent of channel dewatering of a 12.8 km section of the UFR mainstem was surveyed monthly by Lotic Environmental Ltd. (Lotic) beginning in August 2017 in the southern drying reach from Chauncey Creek to upstream to the south tailings pond. Starting in August 2019, drying surveys were expanded to a 6.1 km section of the northern drying reach from the Multiplate culvert to a point upstream of the confluence with Henretta Creek. The methods and results of this work by Lotic are presented in a memo attached as Appendix A (Zathey and Robinson 2021). Spatial data were recorded to document the extent of UFR mainstem that was dry each month during the fall and winter surveys. The start and end points of dewatered reaches were marked using GPS coordinates to estimate the continuous lengths of the UFR that was dry. Sections that were wetted in the middle of a dry reach were excluded from the estimates of the extent of drying. Records of stranding near the beginning of drying in September 2018 were reviewed in the context of drying patterns during and prior to the Decline Window to understand the timing and spatial extent of drying reaches by month and year from August 2017 to April 2020.

2.3. Decline Window Conditions Relative to the Historical Period

Additional information for the UFR drying reaches was reviewed to compare conditions in the drying reach during the Decline Window to the years prior to the Decline Window (historical period). Lotic reviewed water quality records collected by Teck at stations FR4 (in the southern drying reach) and FR1 (in the northern drying reach) that date back to the 1970s to extrapolate the frequency, timing and duration of drying (Appendix A: Zathey and Robinson 2021).

A comparison of hydrological conditions in the drying reach between time periods was conducted by comparing flows and stage data at hydrometric stations and temporary stage loggers deployed near to or within portions of the drying reaches (FR_FRNTP, FRD_SD01, FRD_SD02, FRD_SD03, FR_FRCP1SW). Flow and stage data from hydrometric gauges and spot sampling conducted by Teck Coal personnel and contractors were provided to Ecofish. Sampling methods are specified in Teck Coal's Flow Monitoring Protocol (KWL 2017). All finalized stage and flow data from hydrometric gauges were provided by Kerr Wood Leidal (KWL), Teck Coal's primary Qualified Professional for their hydrometric programs. Flow data were examined at their available frequency (5-minute, 15-minute, or 1-hour frequency, depending on gauge) during dates near stranding incidents and also for daily average when examining trends over longer periods. The analysis evaluated if conditions in the drying reach were different during the Decline Window relative to the preceding years. Stage data were reviewed at FR_FRCP1SW and other locations in the southern drying reach for periods when stranding was observed; this was done to confirm if significant drops in stage corresponded to fish stranding observations.

A more detailed comparison of climate and flow conditions in the historical versus Decline Window periods is presented in the Climate, Water Temperature, Streamflow and Water Use EoC Report (Wright *et al.* 2021). For example, Fording River flow has been monitored at the outlet at the convergence with the Elk River since the 1970s at the Water Survey of Canada (WSC) gauge 08NK018. Low flow conditions were examined at the Fording River WSC gauge to support evaluation of whether seasonal dewatering of the drying reaches is likely to have differed between the historical and Decline Window periods.

2.4. Fish Use of Stranding Sensitive Habitat

Fish stranding causing mortality can occur at different times of the year and affect different life stages. Drying during the fall and winter can overlap with the rearing and/or overwintering periods for WCT juveniles and adults. In addition, decreases in flow during WCT incubation compared to flow during spawning in the spring and summer can cause dewatering and subsequent mortality of WCT redds (eggs and alevins). Pathways were examined to estimate the potential magnitude of effect to buried life stages (i.e., redds) and free-swimming life stages.

2.4.1. Fording River Drying Reaches

Summaries of WCT monitoring data from telemetry and PIT tagging work were used to describe fish use of the UFR drying reaches. The data summaries are presented in Evaluation of Cause Team (2021) and were used to estimate fish use of the UFR drying reaches during the summer rearing and overwintering periods. Estimates of exposure to drying were generated based on the amount of habitat area estimated to be dry relative to total habitat area of the UFR, and based on the estimated fish use of the river segments S7 and S9 that dry. These estimates were used to provide an upper bound on the proportion of fish susceptible to stranding due to dewatering. As upper bounds, the estimates assume that fish do not move to avoid dewatering and all fish present in the dried sections of river segments S7 and S9 die when exposed to dewatering. This is likely an overly conservative assumption, but in the context of the EoC allows an exploration of whether this stressor may have contributed to the WCT decline.

Observations of the maximum extent of drying of the southern and northern reaches during the Decline Window were compiled from fall/winter surveys by Lotic (Appendix A: Zathey and Robinson 2021). The maximum extent of observed drying during the summer rearing and overwintering periods was used to estimate worst-case scenarios in which all fish presumed to be present within these habitats are assumed stranded. Two methods were employed. First, independent of the WCT distribution data, a simple estimate of the total length of the northern and southern reaches dried versus the length of the full UFR mainstem habitat was calculated (where UFR mainstem length = 55.5 km; Cope *et al.* 2016) for both the rearing and overwintering periods. Second, the southern and northern reaches overlap with river segments S7 and S9 respectively, for which there is historical relative fish use information (EoC Team 2021). It was assumed that mortality from stranding would be equivalent to the relative fish use for each segment (S7 and S9) during the rearing or overwintering periods that overlap with the dewatered areas. This calculation used the maximum observed dewatered extent during the rearing or overwintering periods and divided by the lengths of segments S7 and S9 (4,970 m and 3,820 m respectively) to estimate the proportion of S7 and S9 dried. The proportion of S7 and S9 dried was then multiplied by the estimate for fish use in S7 and S9 to estimate the number of fish potentially stranded within S7 and S9. These scenarios were used as worst-case to evaluate whether stranding within the UFR mainstem could have affected a substantial portion of the UFR WCT population. Fish use was assumed to be the same across age classes, and therefore a single proportion estimate was used to account for potential mortality from stranding for both WCT adults and juveniles.

2.4.2. Redd Dewatering

To assess the potential for redd dewatering during incubation, an effective spawning analysis was completed. The effective spawning analysis was based on transect data (collected in segments S6 to the Henretta Creek confluence in S9) from the upper Fording River instream flow study (Healey *et al.* 2020). The effective spawning analysis was used to estimate the amount of spawning habitat that remains sufficiently wetted through the incubation period. This allowed an estimate of annual "effective spawning habitat", which was compared across years within and prior to the Decline Window. This analysis supported a determination of whether redd dewatering may have been greater during the Decline Window than in prior years and hence played a role in the population decline for

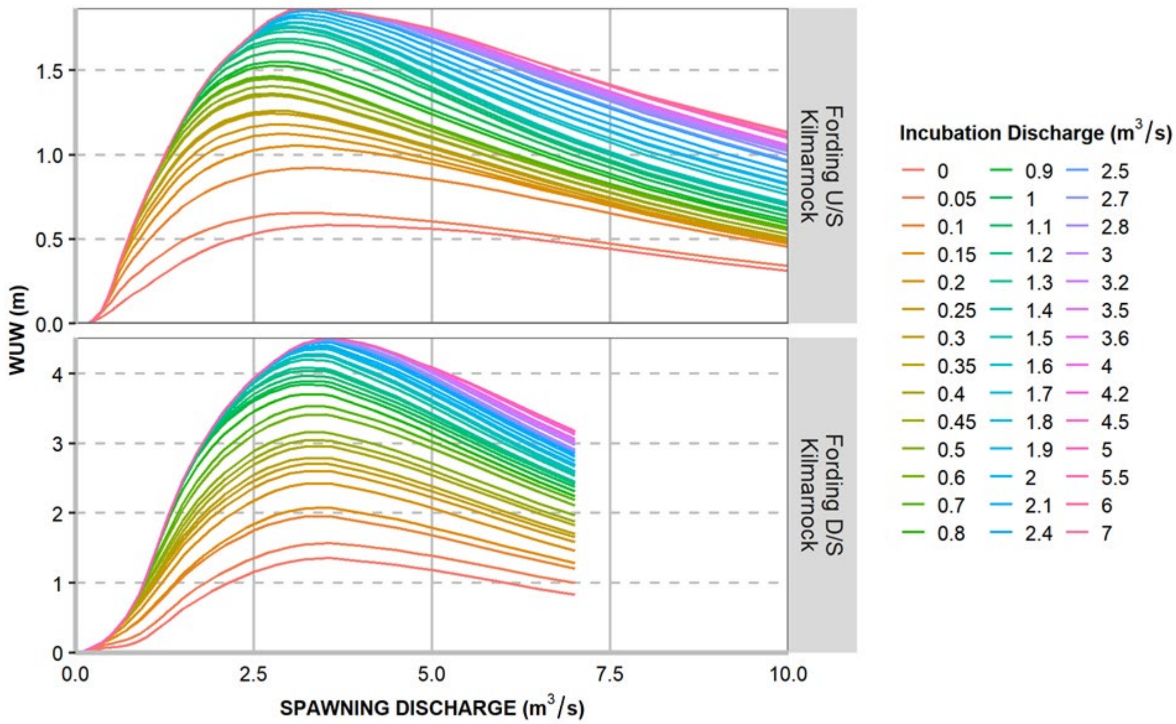
WCT juveniles. We note that this pathway would not explain a decline in adults within the Decline Window period.

For each flow condition modelled, a generic incubation water depth criterion of 10 cm was evaluated at each station across each transect to determine if the habitat had a suitable water depth for incubation. That is, if water depth declined to less than 10 cm at any time during the incubation period the habitat was deemed to be unsuitable; this assumption is somewhat arbitrary and is slightly less conservative than the 15 cm minimum spawning depth implied by the HSC, but is more conservative than a water depth of 0 cm. Habitat that was suitable for spawning (assuming provincially-recommended depth and velocity criteria for Steelhead spawning as a proxy for WCT) and that met the minimum water depth requirement for incubation was classified as "effective" spawning habitat. This analysis was run for flows ranging from 0 m^3/s to 10 m^3/s to determine if redds remained adequately wetted after spawning. The analysis was aggregated across two zones: Fording River from Henretta Creek to near Kilmarnock Creek, and near Kilmarnock Creek to Chauncey Creek (Figure 2).

Effective spawning was evaluated separately for each year from 1997 to 2019 using the methods described above and flow data for FR_FRNTP. This was completed by:

- Calculating the average flow during the spawning period for each year;
- Looking up the average WUW at this average flow rate;
- Determining the minimum flow present during the incubation period during the year; and
- Looking up the combination of average spawning flow and minimum incubation flow (i.e., from Figure 2) to determine the effective spawning habitat.

Figure 2. Habitat-flow relationships used to estimate effective spawning habitat in Fording River between Henretta Creek and Kilmarnock Creek (top), and between Kilmarnock Creek and Chauncey Creek (bottom). Effective spawning habitat is calculated with weighted usable width as a function of spawning discharge and discharge during incubation.



2.5. Evaluation of Requisite Conditions

Requisite conditions are defined as the circumstances that would need to be met for dewatering of the mainstem UFR habitat to cause or contribute to the WCT population decline. The methods described above were used to evaluate the potential for stranding of fish in the UFR mainstem and to determine whether requisite conditions were met. Requisite conditions (Table 1) were based on spatial (extent and location) and temporal (timing and duration) aspects of dewatering events and on the intensity (magnitude) of the stranding events. Overall, it should be noted that data limitations restricted our ability to quantitatively evaluate all requisite conditions. For example, while field crews were on the ground and documented stranding events during the fall 2018 period, field crews were present for only a portion of the period, and therefore stranding may not have been fully characterized. Therefore, a more indirect approach was needed by inferring effects in consideration of the available drying, hydrologic, and historical WCT distribution information.

Table 1. Requisite conditions for dewatering of the UFR mainstem to cause or contribute to the WCT population decline.

Spatial extent	The dewatering event affected a relatively large portion of accessible fish habitat relative to that available in the UFR (therefore assumed to affect a large portion of the population)
Duration	The dewatering event was of a duration great enough to cause fish mortality
Location	The dewatering event occurred in the mainstem UFR in a location where fish are present (accessible to fish and suitable for fish) and where habitat is sensitive to stranding
Timing	The dewatering event occurred during the Decline Window when fish were present
Intensity	The dewatering event led to stranding of a sufficient number of fish to cause or play a role in the decline.

A **requisite condition to cause** was identified when dewatering events during the Decline Window had the potential for stranding a large portion of the UFR fish population as inferred by stranding potential and spatial extent. A **requisite condition to contribute** was identified when dewatering events during the Decline Window had the potential for stranding a low to moderate portion of the UFR fish population as inferred by stranding potential and spatial extent. Since stranding is binary (almost all stranded fish die), the requisite conditions for spatial extent and intensity are essentially the same.

Stranding risk also occurred in the years prior to the Decline Window, so the difference in stranding risk between the two time periods (Decline Window and pre-Window period) affects the validity of stranding as an explanation for reduced fish abundance during the Decline Window. Therefore, the comparison of stranding risk between the two time periods was an important consideration. The greater the proportion of habitat and the higher the stranding risk during the Decline Window relative to the historical period, the more likely it is that dewatering events caused or contributed to the observed WCT population decline.

3. RESULTS

3.1. Literature Review: Fish Stranding in Dewatered Reaches

3.1.1. Major Drivers of Stream Drying

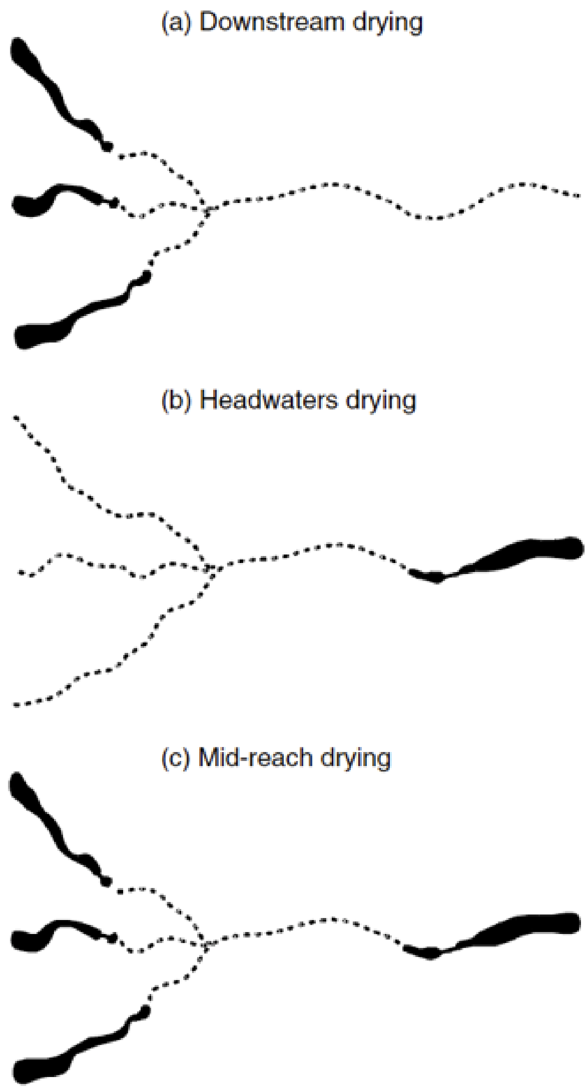
Intermittent river ecosystems cease surface flow for portions of the year (Hansen 2001) and are common in many parts of the world (Tolonen *et al.* 2019). Drying or intermittent segments can comprise a substantial length of a river network. Datry *et al.* (2014) estimated that intermittent rivers make up greater than 50% of all rivers globally, with this proportion likely to increase with increasing water abstraction and climate change (Döll and Schmied 2012). A variety of natural and anthropogenic factors contribute to stream drying including climate, watershed area, position in the river network (e.g., headwaters versus mainstem), channel gradient, abundance of instream wood, substrate composition and structure, groundwater and hyporheic flows, and water diversions and withdrawals (Lake 2003; Tolonen *et al.* 2019).

Stream drying can occur at different locations within a river network and can be classified as 'headwater drying', 'mid-reach drying', and downstream drying' (Figure 1, from Lake 2003) (Lake 2003). Many streams originate from permanent springs and if seasonal or longer-term drought occurs in such a stream type, the headwaters may contain water in pools, but downstream the channel may dry (Figure 1a). In other systems, headwaters may dry and as the stream channel increases in size in a downstream direction, surface water appears (Figure 1b). In other cases, the stream experiencing drought has water in headwater springs and at a considerable distance downstream it has water in continuous sections, but surface flow may be absent in between the wetted headwater and downstream wetted section (Figure 1c). This form of drying, which is like the mid-reach drying observed in the UFR, occurs where certain river sections may contain water while other river sections dry, although sub-surface hyporheic flow may persist. Particularly, in unconstrained channels, deep alluvial gravels can absorb large volumes of water (Stanley *et al.* 1997). Stream aspect can also influence stream drying, with south facing slopes receiving more solar radiation and thus exhibiting higher evaporation rates than north-facing aspects (Turner and Gardner 2015).

Timing and extent of drying is strongly influenced by hydrological factors. Stream drying can be predictable, resulting from seasonal dry periods, or can be a result of longer, unpredictable, aseasonal or supra-seasonal events marked by declining precipitation and water availability over several seasons (Lake 2003). Longer-term droughts have been linked with Pacific Ocean-atmospheric patterns including the El Niño–Southern Oscillation (ENSO) and the Pacific Decadal Oscillation (PDO) (Fleming and Whitfield 2010).

Regional climate strongly influences the probability of and extent of stream drying. For example, the extent and duration of stream drying are higher in years of below-average precipitation (Lake 2003), which may translate into more severe effects on instream biota (Woelfle-Erskine *et al.* 2017), especially if drought conditions persist over multiple years or increase in strength (Lake 2003). The nature, timing, and extent of stream drying also varies geographically largely because of prevailing climatic conditions and topography. For example, stream drying in summer and fall is a common occurrence in parts of coastal Western North America (e.g., parts of California and Oregon) with a Mediterranean climate; these coastal catchments are characterized by a pronounced winter rainy season followed by summer droughts (Ebersole *et al.* 2003; Woelfle-Erskine *et al.* 2017). In contrast, stream drying in continental climates with snow-dominated hydrographs, such as the Kootenay region, can also occur in winter when flows are at their lowest (Cope *et al.* 2016).

Figure 3. Three patterns of channel drying during a drought (from Lake 2003).



Local biophysical factors are also associated with stream drying. Substrate composition and structure (e.g., deep alluvium vs. bedrock) are key factors determining the probability and extent of stream drying. Stream reaches with deep, porous alluvial substrate are prone to drying at low flows because the water table falls below the channel elevation (Zathey and Robinson 2021). Furthermore, unconstrained, low gradient reaches with large width to depth ratios are likely more prone to dry as they are more exposed to evaporation than confined reaches (Stanley *et al.* 1997). The composition and age of riparian vegetation can potentially influence stream drying (Jones and Post 2004). Removal of riparian vegetation leads to a series of habitat changes that affect the propensity of a stream to dry. For example, logging of riparian trees reduces bank stability via losses of structural controls resulting in increased sediment flux to the streambed and high flood events, potentially altering the movement of water, nutrients, organic matter and organisms into substrate interstices. In alluvial channels, both the removal of bank vegetation and increased sediment supply can cause channels to become aggraded, wider and shallower with fewer pools and more riffles (Chamberlin *et al.* 1991). Shallow, riffle habitats are more prone to drying than deep pools due to increased exposure to solar radiation and higher bed elevation. Some studies suggest forest succession after logging influences stream discharge. For the first few years after logging, discharge may increase due to reductions in evapotranspiration. However, as riparian vegetation regrows (~6-30 years post-logging) flows during some seasons can decline due to increase water demands by growing plants (Jones and Post 2004). The presence of instream large woody debris (LWD) can also influence stream drying. The quantity, position, and orientation of the wood pieces within the river channel can influence flow complexity and water retention (Ehrman and Lamberti, 1992; Gurnell *et al.* 2002). For example, the transport of water was 1.5 to 1.7 times slower in reaches with accumulations of LWD than channelized reaches without LWD (Ehrman and Lamberti 1992).

Anthropogenic factors, including water withdrawal and diversions, and habitat alteration, also affect the likelihood of stream drying (Caldwell *et al.* 2012). For example, diverting tributary inflows can result in local stream drying in downstream mainstem reaches near tributary junctions. These anthropogenic drivers of drying can also affect stream fish. Historically, the Shasta River in northern California was one of the area's most productive salmon rivers because of the abundance of numerous cold-water springs. Today, surface water diversions, groundwater pumping, and dam construction have greatly decreased river flows and available fish habitat (Null *et al.* 2010).

3.1.2. Effects of Drying to Stream Salmonid Populations

Stream drying is associated with complex changes in abiotic and biotic conditions, resulting from reductions in habitat connectivity and habitat isolation that can affect fish and invertebrates. Stream drying disrupts the downstream movement of water, sediment, and organic matter and the bi-directional movement of stream invertebrates and fish. Stream drying therefore, has effects on river ecosystems, and the densities, and size- or age-structure of stream species including stream salmonids (Jeffres and Moyle 2012; Vorste *et al.* 2020) via a variety of pathways; however, the population consequences of stream drying are not well understood (Nagrodski *et al.* 2012).

In western North America, watersheds supporting species in the family Salmonidae are dynamic because of disturbance regimes that include landslides, fires, floods and droughts, and variations in marine and freshwater productivity, that operate on a variety of spatial and temporal scales (Waples *et al.* 2008). These natural disturbances create selective pressures for adaptive responses by stream salmonids. If anthropogenic changes or climate fluctuations produce disturbance regimes that are outside the historical range of conditions, salmonids inhabiting such systems may be poorly equipped to survive these challenges. For example, WCT living in high-elevation watersheds are adapted to seasonal changes in habitat conditions such as winter low flows and the occurrence of river ice. To avoid these stressful conditions, movement is common in the fall as fish seek suitable overwintering sites in response to declining water temperature (Cunjak 1996), which is the pattern also observed in the UFR (Cope *et al.* 2016). However, if anthropogenic changes in habitat (e.g., increased sedimentation, floods, loss of LWD) increase winter drying, overwinter habitats may be degraded.

Furthermore, if there are temporal changes in stream drying without a corresponding change in water temperature that cue fish to move, they may not be able to successfully access overwintering sites due to habitat isolation. Fish that fail to access overwintering habitats may experience higher rates of mortality due to stranding or predation.

Despite their abundance, the importance of intermittent channels to salmonid population productivity is poorly understood; however, a few recent studies, mostly in coastal ecosystems, quantified individual performance (i.e., growth, survival) of stream salmonids living in these habitats. It has been proposed that intermittent coastal streams in western North America play a vital but under-studied role in providing habitat for stream salmonids, especially juveniles (Woelfle-Erskine *et al*. 2017). Recent studies of intermittent Pacific coastal streams found that juvenile salmonids, typically Coho Salmon (*O. kisutch*) and steelhead trout (*O. mykiss*) occupy isolated pools in these channels during the summer dry season (e.g., Hwan and Carlson 2016). Some studies indicate that fish occupying these intermittent stream pools can grow larger than fish rearing in nearby perennial stream pools (Ebersole *et al*. 2003). One study showed that large, deep isolated pools were associated with higher survival of juvenile Coho in summer, while steelhead trout tended to survive better in pools with large surface area (Woelfle-Erskine *et al*. 2017). Survival rates in these isolated pools were also influenced by climatic variability, with higher survival rates for juvenile steelhead following wet winters compared with dry winters (Hwan *et al*. 2018). Most investigations suggest that how well salmonids perform in isolated habitats in intermittent stream channels was a result of the availability of deep pools with abundant cover and high rates of shallow, lateral hyporheic flow with high dissolved oxygen concentrations (Cunjak 1996; Brown *et al*. 2011; Woelfle-Erskine *et al*. 2017). Clearly, these advantages only accrue if the isolated pools become reconnected to the mainstem at some point to allow fish to move to other habitats as they grow.

Intermittent coastal stream reaches can also have negative consequences; they can serve as ecological traps where survival is reduced relative to perennial stream channels (Jeffres and Moyle 2012; Vorste *et al*. 2020). Adult Coho Salmon in the Shasta River appear to have equal preferences for spawning habitat that leads to reduced offspring survival and that of apparently similar quality where juvenile survival would be more likely (Jeffres and Moyle 2012). The number of pools in intermittent streams during summer where juvenile Coho Salmon survival was reduced or zero increased during drought years. However, even during drought years some pools, especially deep pools with lateral hyporheic exchange, served as refuges where survival was similar to non-drought years (Vorste *et al*. 2020).

While we are beginning to learn more about the importance of intermittent stream reaches for salmonid populations in rain-dominated Pacific coastal ecosystems, we have a poor understanding of these habitats in continental climates with a snow-dominated hydrograph where drying also occurs in winter. This knowledge gap is not surprising given the logistical challenges of conducting field work under extreme conditions. In higher elevation watersheds, such as those in the Kootenay River watershed, winter conditions are extreme with low flows and formation of river ice and thus access to overwinter habitat is hypothesized to limit stream salmonid populations. For example, the availability, quality, quantity, and distribution of winter-rearing habitat, such as deep pools, side-channels, beaver ponds and backwaters are frequently cited as a factor limiting WCT populations (Cleator *et al.* 2009, Cope *et al.* 2016). Incidences of channel dewatering are not unique to the Fording River and are also known to occur within the upper reaches of other upper Kootenay River tributaries such as the Wigwam River (Baxter and Hagen 2003, Prince and Cope 2001) and the Elk River (Prince and Morris 2003).

Extreme winter conditions also influence the survival of other stream salmonids. Estimated mortality of juvenile Atlantic salmon (*Salmon salar*) in Eastern Canada was 25- 57%, being highest in the stream where winter conditions were most severe and ice and low flow limited fish movement (Cunjak and Randall 1993, Cunjak 1996). Similarly, Cunjak (1996) observed that juvenile Atlantic salmon mortality in Catamaran Brook exceeded 65% during a particularly severe winter with low streamflow. Bustard (1986, cited in Cunjak 1996) estimated winter mortalities of 30–60% for juvenile salmonids in side-channels exposed to severe freezing and suggested that the reduced streamflow and ice formation of interior streams were the reasons many juvenile salmonids overwintered in mainstem habitat where suitable water depth was maintained.

Responses of biota to drought disturbance differ depending on whether the drought is a predictable seasonal event or supra-seasonal (Matthews 1998; Lake 2003). Because seasonal droughts are predictable, the biota can be expected to have evolved adaptations, such as life-history scheduling and the adaptive use of refugia to survive the disturbance periods. For example, stream salmonids commonly move to preferred overwintering habitats following a reduction in fall temperatures (e.g., Cunjak 1996). In general, it appears that the biotic response to seasonal droughts is characterized by a high capacity to withstand the event and to recover (Lake 2003). Supra-seasonal droughts, on the other hand, are unpredictable in timing and duration, and thus more difficult for organisms to adapt to. As a result, the biotic response to such events is characterized by low to moderate capacity to withstand the event and to recover after the event is over (Lake 2003).

Fish stranding is an event that restricts a fish to habitats as a consequence of physical separation from the main body of water as a stream dries (Nagrodoski *et al.* 2012). Stranding in riverine ecosystems can result from both natural and anthropogenic processes that follow a decline in water level, which can be rapid in systems modified by hydroelectric facilities but also as streams dry naturally. Most of the research on fish stranding has focused on reducing effects on stream salmonids in systems affected by hydropower: 82% of fish-stranding studies were represented by anthropogenic events compared to only 18% that were due to natural events (Nagrodoski *et al.* 2012).

While Nagrodoski *et al.* 2012 reviewed a number of studies describing some aspect of fish stranding, they note the consequences of stranding (via mortality) at the population-level has not been well studied. For example, Hvidsten (1985) noted stream drying downstream of a hydroelectric facility in Norway led to 'large' losses of 0+ Atlantic salmon and brown trout. The author implied recruitment of brown trout was reduced; however, no data were provided to support this effect. Some insights into the potential population-level effects of fish stranding are provided in Bradford *et al.* (1995). In a series of experiments conducted during winter (~3.5 to 4°C) with juvenile Coho Salmon and Rainbow Trout, stranding rates varied between day and night (higher at night) and species (higher standing rates for Coho Salmon). Stranding rates for juvenile Coho at night ranged from 25 to 85% compared to < 0.05 to 32 % for Rainbow Trout. Stranding rates during the day were < 10 %. Higher standing rates at night were thought to result from juvenile fish concealing themselves in the gravel where they did not respond to decreasing water levels.

The consequences of stranding due to stream drying range from sub-lethal effects to mortality. Stranding mortality occurs for a variety of reasons, including death from lack of oxygen or water. Death can also be a result from water temperature stress (e.g., cold shock; Donaldson *et al.* 2008). In cold climates in winter, fish can be trapped in ice also leading to mortality (Brown *et al.* 2001). In addition, fish stranded on dry riverbeds or isolated in pools with little protective cover are more susceptible to predation (Vorste *et al.* 2020). Harvest by humans may also result after dewatering as these fish are easily captured by humans (Nagrodski *et al.* 2012). If a fish survives isolation in a temporary pool, the sub-lethal impacts may affect overall fitness. As water level declines, fish can be exposed to stressful water quality conditions including low dissolved oxygen levels and high/low water temperatures depending on the season. Juvenile fish isolated in pools during stream dewatering may also reduce consumption rates leading to reduced growth rates and altered distributions and habitat use compared to juveniles in more stable environments. Fish stranding events also result in nest abandonment, home range reductions, and loss of habitat connectivity (Korman and Campana 2009).

A number of factors modify the effects of stream drying with fish behavior and availability of and access to refugia representing key components (e.g., Lake 2003). As mentioned earlier, WCT living in high-elevation watersheds are adapted to seasonal changes in habitat conditions and commonly migrate to suitable overwintering sites in response to declining water temperature (Jakober *et al.* 1998; Cunjak 1996; Brown *et al.* 2011). This behavior has been observed in other WCT populations living in similar environments. For example, radio-tagged bull trout (*Salmo confluentas*) and WCT occupying Montana headwater streams made extensive (> 1 km) downstream movements to overwintering habitats when temperatures declined in the fall (Jakober *et al.* 1998). Thus, fish living in intermittent streams with predictable, seasonal water level changes would be expected to have evolved adaptations, such as migration to refugia. Disconnection from these refugia, either as a result of natural or anthropogenic processes, can likely lead to increased rates of mortality, reductions in recruitment and population declines (Magoulick and Kobza 2003). Key habitats that provide refugia for stream salmonids from stream drying include low-velocity, deep-water pools, side-channels or beaver ponds, and backwaters that receive lateral inputs of well-oxygenated, warmer hyporheic flow (Cunjak 1996). In addition, fish likely benefit when these habitats have suitable cover (boulders, LWD, undercut banks) that protect them from predators or competitors (Cunjak 1996).

3.1.3. Summary

Intermittent stream channels are globally and regionally common and comprise a large portion of many river networks. A variety of natural and anthropogenic factors operating at different spatial and temporal scales determine the location in the river network, the frequency and the duration of stream drying. For example, high-elevation watersheds in the interior of British Columbia can dry and freeze in winter due to low stream flows and freezing air temperatures, while Pacific coastal ecosystems generally dry in summer and fall. Anthropogenic factors that influence stream drying including water diversions and withdrawals, forestry, and human-caused and flooding-influenced changes in streambed morphology, substrate composition, and LWD. Stream salmonids are adapted to seasonal, periodic changes in stream drying, exhibiting behaviours that limit their exposure to harmful environmental conditions. For example, stream salmonids often initiate movements in the fall to overwintering habitats as water temperature declines. However, extreme drying events that are outside of the natural range of conditions have the potential to negatively impact individuals and populations. Most of the research on the effects of stream drying on stream salmonids has been conducted in Pacific coastal rain-dominated ecosystems, focusing on the effects of summer drying on growth and survival. We have a poorer understanding of the biotic effects of stream drying on interior salmonid populations living in snowmelt-dominated systems that also dry in winter. Fish that are isolated in stream pools or become stranded on dry riverbeds can experience high rates of mortality via a variety of mechanisms. Factors that can mitigate the biotic effects of stream drying include access to overwintering habitats, especially deep, low velocity habitats with abundant cover and inputs of well-oxygenated water.

3.2. Dewatering and Stranding Observations in the UFR

Across all incidents of fish mortality in the Elk Valley reported by Teck between September 2015 and August 2019 (Table 2) three distinct stranding incidents were reported compared to 10 incidents of mortality from likely predation/natural sources, 7 incidents from instream works, 3 incidents related to fish inventories, and 11 from other/unknown causes. The stranding incidents both within and outside of UFR resulted in higher mortality than the other causes, averaging about 120 mortalities per stranding incident compared to an average of <10 mortalities for the other incident causes. While the confirmed stranding incidents had higher mortality than other causes, it is also important to note that both of the categories "predation/natural sources" and "other/unknown" could possibly include stranding of one or a handful of fish. In these cases, isolated dewatering could be the cause of some of these incidents, although the exact cause is often not known.

A documented stranding event of WCT in the mainstem UFR during the Decline Window occurred in 2018 in the Kilmarnock Phase 1 Discharge Channel and the lower portion of the Fording River Side Channel (Teck 2019a, Teck 2019b). This event occurred between August 30 and September 7, 2018, when flow changes in the Kilmarnock Phase 1 Discharge Channel and Fording River Side Channel caused isolation and stranding of WCT. Flow in the Fording River at FR_FRNTP declined rapidly from ~1.2 m^3/s on August 30 to ~0.7 m^3/s on September 3. To actively manage salvage efforts, discharge from Kilmarnock Phase 1 Discharge Channel was ceased on September 5. A total of 881 WCT were salvaged and 216 WCT died, ranging in size from approximately 76 mm to 147 mm (Teck 2019a, Teck 2019b). Overall, 743 (68%) of the 1095 total fish recorded (salvage plus mortalities) were in the Fording River Side Channel and 352 were in the Kilmarnock Phase 1

Discharge Channel. Further description of the event is provided in Faulkner *et al.* (2021a) and in Teck (2019a, 2019b).

Two additional stranding events were recorded in the Teck incident reporting between 2015 and 2019, but both occurred outside of the UFR in the Elk River Side Channel near Wolfram Pond. The first incident occurred in the winter on December 7, 2015 and killed approximately 100 Brook Trout and other unknown fish species. The second event occurred on September 10, 2018 and killed 30-60 Mountain Whitefish. This second event in the Elk River Side Channel occurred in the same 1–2-week period as the Fording River Side Channel stranding incident and the dewatering of the southern drying reach of the UFR mainstem (described further below).

Table 2. Incidents of fish mortality from Teck incident reporting (2015-2019).

Mortality Cause	Total # Incidents	Total # Mortalities	Average # Mortalities per Incident	Range in # Mortalities per Incident
Predation/Natural	10	15	1.5	1 to 4
Instream works	7	10	1.4	1 to 3
Inventory	3	30	10	3 to 15
Stranding	3	~361	~120.3	~45 to 216
Other/Unknown	11	19	1.7	1 to 6

Seasonal drying has been observed in the UFR mainstem in two main reaches of the river, the northern and southern reaches, near station FR_FR1 and stations FR_FR4 and FR_FRCP1SW, respectively (Map 1). The northern drying reach overlaps with river segment S9, and the southern reach overlaps with river segment S7, as defined by Cope *et al.* (2016).

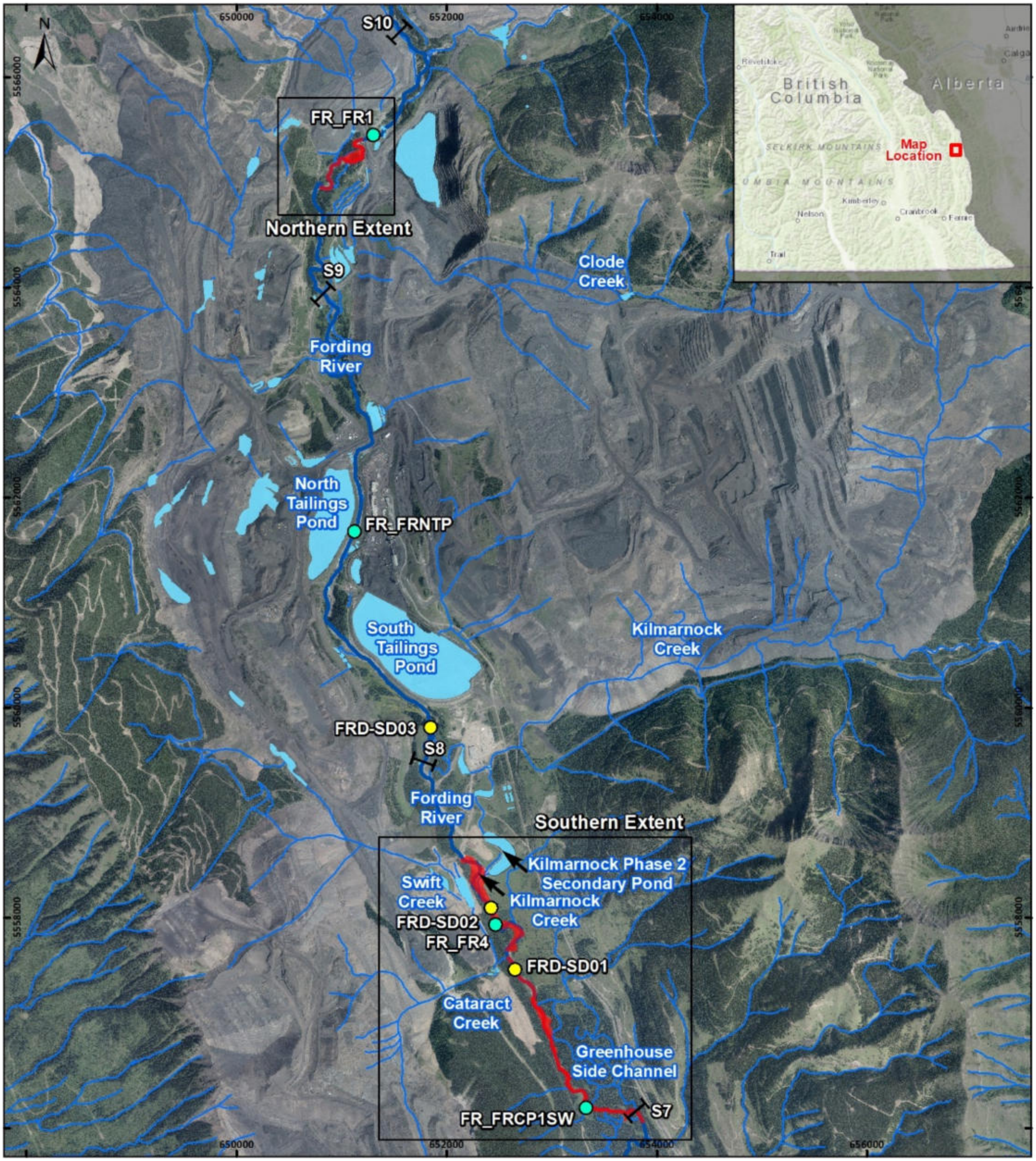
The extent of drying in these reaches was documented by Lotic from August 2017 to April 2020 during the Decline Window and is shown in Map 1. The hydrometric stations to monitor stage of FRD_SD01, FRD_SD02 and FRD_SD03 are also shown in Map 1. Dewatering extent and timing of the southern drying section varied by year of survey during the Decline Window, with date of first dewatering at FR_FRCP1SW observed on December 14, 2017, September 10, 2018 and January 6, 2020 for the 2017/2018, 2018/2019, and 2019/2020 fall-winter seasons, respectively (Table 3, Map 2, Map 3, Map 4). Date of rewetting at FR_FRCP1SW occurred on March 24, 2018, between March 12 and 19, 2019, and on April 14, 2020. Drying was observed at station FR_FR4 in the 2018/2019 season only, which was first observed on January 29, 2019. FR_FR4 remained wetted throughout the 2017/2018 and 2019/2020 fall-winter seasons.

The northern section near station FR_FR1 was first surveyed in 2019/2020 and was observed to first go dry on November 18, 2019 (Table 3, Map 5). Rewetting at FR_FR1 occurred on April 14, 2020 on the same day as FR_FRCP1SW in the southern section.

Stranding was recorded by field crews working in the area of the southern drying reaches during the Decline Window. Specifically, three observations were made of stranded or isolated fish in September and October 2018. On September 10, 2018, staff from Minnow Environmental Ltd. (Minnow) were conducting field sampling as a part of the Fording River Local Environmental Effects Monitoring Program and noted dewatering of an approximately 800m section of the UFR mainstem near FR_FRCP1SW. In a shallow riffle upstream of the dry section, a total of 15 WCT mortalities were observed with individuals ranging in size from 80 to 190 mm (Figure 4; Wilm pers. comm. 2020). On September 19, 2018 FRO was notified of an additional WCT mortality (95mm) in an isolated pool; the report was made by Ecofish staff working on critical riffle surveys near the area where the 15 previous mortalities were observed (Figure 5; Wilm pers. comm. 2020). Additionally, during the drying reach surveys in October 2018 multiple WCT were observed to be isolated in pools in the drying reaches (Figure 6); however, the fish were not disturbed or salvaged and there are no estimates of fish abundance within these pools.

Table 3. Dates of drying and corresponding flow at FR_FRNTP from the southern (FR_FRCP1SW, FR_FR4) and northern (FR_FR1) mainstem UFR reaches. Source: Fording River Drying Report (Appendix A: Zathey and Robinson 2021).

Station	Date of First Drying	Date of Drying End	Flow at FR_FRNTP (m^3/sec)	Comments
FR_FRCP1SW	14/Dec/2017		0.49	FR_FRCP1SW was only station to become dry in 2017/2018
		24/Mar/2018	0.38	
FR_FRCP1SW	10/Sep/2018		0.82	Date of re-watering unknown; between March 12 and 19, 2019
		12/Mar to 19/Mar/2019	0.31 to 0.72	
FR_FRCP1SW	06/Jan/2020		0.43	Drying not observed in fall 2019.
		14/Apr/2020	1.00	
FR_FR4	29/Jan/2019		0.40	Remained wetted for 2017-2018 and 2019-2020
		13/Mar/2019	0.31	
FR_FR1	18/Nov/2019		0.60	Only survey at FR_FR1 occurred in winter 2019-2020
		14/Apr/2020		



Legend

- Hydrometric Gauge
- Sensitive Stranding Monitoring Site
- Reach Break
- Water Network
- Water Management Polygons
- Mainstem Dewatering

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

0 500 1,000 1,500 m

Scale: 1:52,000

NO.	DATE	REVISION	BY
1	2021-06-09	1229_FRD_MainstemDewateringOverview_3966_20210301	ERL
2			
3			
4			
5			

Date Saved: 2021-06-09
Coordinate System: NAD 1983 UTM Zone 11N

ECOFISH RESEARCH

Map 1

Path: M:\Projects-Active\1229_EVWQP\MXD\Overview\1229_FRD_MainstemDewateringOverview_3966_20210301.mxd

Figure 4. **WCT mortalities observed on September 10, 2018 in the UFR mainstem near FR_FRCP1SW by Minnow Environmental personnel.**

Figure 5. Isolated pool in the UFR mainstem southern drying reach where a single WCT mortality was observed on September 19, 2018 by Ecofish staff.

Figure 6. WCT observed in isolated pools in the drying reaches of the UFR in October 2018 (photo credit: Kamila Baranowska).

3.3. Decline Window Conditions Relative to the Historical Period

Winter-season drying of the UFR drying reaches has been observed for many years including in the Decline Window and historical period. Winter water quality surveys by Teck document drying at stations FR_FR4 and FR_FR1 dating back to the 1970s (Appendix A: Zathey and Robinson 2021). For both the northern and southern sections, based on the spot measurements at FR_FR4 and FR_FR1, drying typically begins in November or December each year, with the northern section typically drying several weeks before the southern section. Some years are noted to have no seasonal drying observed at certain locations, such as at FR_FR4 in winter 2018 and 2020 (although drying did occur at FR_FRCP1SW in all years). Winter drying in the northern and southern reaches of the UFR has also been observed and reported in other studies. For example, Cope *et al.* (2016) observed drying in the winters of 2012/13, 2013/14, and 2014/15. December through February observations of dewatered sections were made during helicopter tracking surveys and verified by ground-truthing (Cope *et al.* 2016), although the full extent of dewatering was not evident due to intermittent flow, and snow and ice cover. Lister and Kerr Wood Leidal 1980 also noted that this section dewatered in winter in the past, including a dewatering event observed in October 1979. Overall, these results indicate that dewatering of at least two sections of the UFR is a phenomenon that is not unique to the Decline Window and has been occurring for many years, though the extent and duration may vary among years.

One way to evaluate whether conditions during the Decline Window were unusual compared to the historical period is to assess the streamflow conditions during these periods. Flow in the UFR mainstem has been monitored at several stations in the UFR since the late 1990s, including FR_FRNTP located in S8 between the northern and southern drying reaches (Map 1). There are gaps in the flow record at FR_FRNTP, particularly in the winter months due to ice effects. The flow series for the fall/winter periods from 2015/2016 to 2019/2020 is shown in Figure 7. Mean monthly flow (as m^3/s and %MAD) from September to April during the three years immediately prior to the Decline Window (2014/2015, 2015/2016, and 2016/2017) and the two years within the Decline Window (2017/2018 and 2018/2019) are provided in Table 4. For a complete presentation of these data and from other stations see Wright *et al.* (2021).

The lowest monthly streamflow at FR_FRNTP could not be confidently established due to data gaps in the continuous streamflow record during winter months; however, mean flows in February 2019 were the lowest of any month from September 2016 to September 2019. Manual streamflow measurements made in January, February and December 2018 were used to estimate mean flows in these months. These measurements indicate that streamflow during January and February 2018 may have been similar to flows measured in February 2019 (Wright *et al.* 2021) and do not appear to be anomalous. For example, flows were lower prior to the Decline Window in January and February 2016.

Dewatering and subsequent rewetting of the UFR drying segments at FR_FRCP1SW, FR_FR4 and FR_FR1 in the Decline Window occurs when flows at FR_FRNTP are in the range of 0.3 to 1.0 m^3/s

(Table 3). This range of flows appears to be common in the winter flow record in the historical period. However, flows at the time of drying at a single station vary across years (e.g., FR_FRCP1SW), which indicates that there are factors other than upstream flow (e.g., groundwater inflows, temperature) that contribute to determining the timing, extent, and duration of drying. For example, flow at FR_FRNTP on the date of first drying at FR_FRCP1SW was 0.49 m^3/s in 2017, which occurred on December 14, 2017. In comparison, drying occurred earlier in 2018 on September 10, 2018 at almost double the FR_FRNTP flow (0.82 m^3/s). Overall, flows in late summer and fall of 2018/2019 during the period that stranding was observed (September) were higher than flows in the same period of 2017/2018 (Figure 7, Figure 8, Table 5), again suggesting that environmental factors other than flow influence timing and extent of drying.

Fording River flow has also been monitored at the confluence with Elk River since the 1970s at the WSC gauge 08NK018. In general, 2018/2019 flows were lower than in many of the preceding years; for example, longer-term flow records show that 2018/2019 flows were lower in July 2018 to March 2019 than the historical median (Table 5, Appendix A: Zathey and Robinson 2021). Average daily flows at the WSC gauge in September to mid-October 2018 (3.28 m^3/s) during the overwintering migration period for WCT were lower than the average daily flows in September to mid-October since 1970 of 4.35 m^3/s, and lower than all preceding years since 2003 other than 2017. Some early fall low flow years were observed in the historical period, including in fall 1979 when dewatered sections were observed by Lister and Kerr Wood Leidal (1980) in October approximately 900 m upstream of FR_FRCP1SW. Fording River flow in summer months of 2017 and 2018 (leading up to fall drying) also were lower than the historical period. In contrast, while flows during overwintering in 2018 and 2019 were lower than the historical average and lower than most years since 2000, Fording River flows during the winters of the 1970s and 1980s were generally lower than the more recent period, including in the Decline Window (Table 5). Overall, the WSC flow data suggest that flows during the Decline Window were lower than the recent historical period (since 2000), which indicates potential for stranding mortality to contribute to the observed WCT decline.

Stage data from the southern drying reach was also evaluated, which shows that there was a drop in stage from September 8 to 10, 2018 that is not seen in the stations further upstream (FRD-SD01, FRD-SD02 and FRD-SD03) over the same dates (Figure 9). This drop in stage overlaps with observations of stranding of 15 WCT individuals on September 10, 2018. In comparison, a significant drop in stage was observed on September 27 and 28 at the station FRD_SD02, although no fish stranding was documented (Figure 9). The causes of these stage changes may have been due to instream works upstream of these locations related to construction of offsetting projects occurring at that time (Hemmera Envirochem Inc. 2018).

Figure 7. Daily average flow (m^3/s) in the UFR mainstem at FR_FRNTP from August 2014 to April 2019.

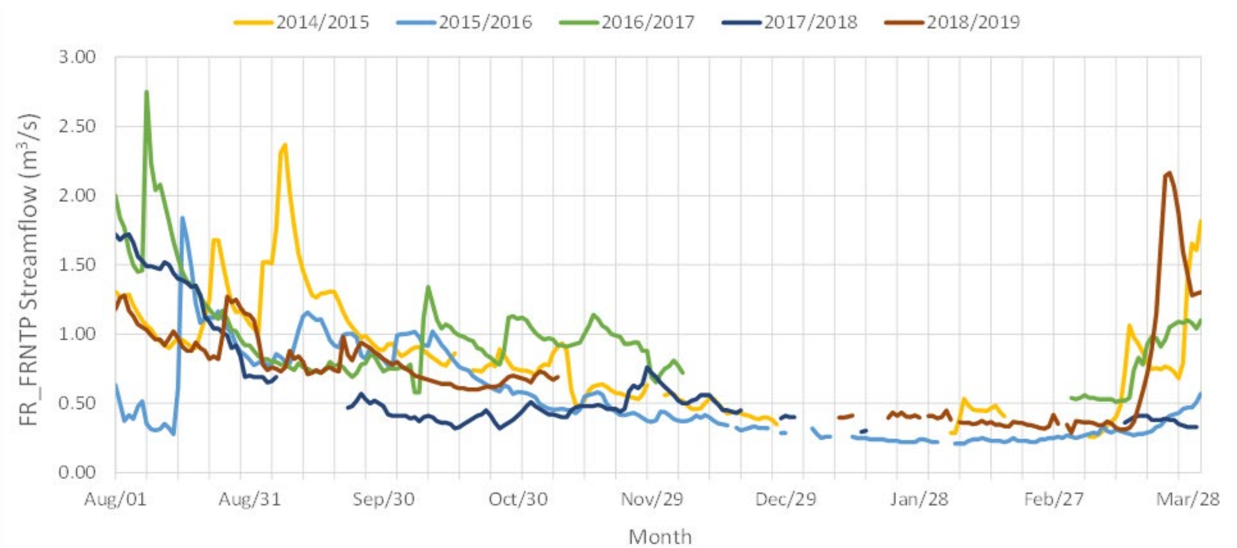


Table 4. Mean monthly streamflow at FR_FRNTP during the Decline Window and pre-Decline Window, expressed as percent mean annual discharge (MAD).

Station	Month	2014-2015		2015-2016		2016-2017		2017-2018		2018-2019	
		Mean Flow (m^3/s)	% MAD	Mean Flow (m^3/s)	% MAD	Mean Flow (m^3/s)	% MAD	Mean Flow (m^3/s)	% MAD	Mean Flow (m^3/s)	% MAD
FR_FRNTP	Sept	1.34	75	0.92	52	0.78	44	0.56	32	0.81	46
	Oct	0.83	46	0.79	44	0.96	54	0.39	22	0.66	37
	Nov	0.66	37	0.48	27	0.96	54	0.51	29	0.70	39
	Dec	0.47	27	0.37	21	0.76	43	0.50	28	0.42	23
	Jan	-	-	0.24	13	-	-	0.33	18	0.41	23
	Feb	0.44	25	0.23	13	-	-	0.36	20	0.36	20
	Mar	0.80	45	0.34	19	0.82	46	0.38	21	0.89	50
	Apr	2.23	125	4.47	251	1.87	105	1.48	83	1.21	68

Mean annual discharge (MAD; 1.778 m^3/s) based on average flows measured since January 2014. Earlier records were excluded due to being affected by data gaps.
"-" denotes months in which continuous data were unavailable due to ice effects and there were < 5 manual measurements
Values shaded in grey are computed from manual measurements made between 5-8 times in the month

Figure 8. Daily average flow (m^3/s) and stage (m) data from FR_FRNTP and FR_FRCP1SW, August 15 to October 15, 2017 to 2019. No stage data is available for FR_FRCP1SW in fall 2017.

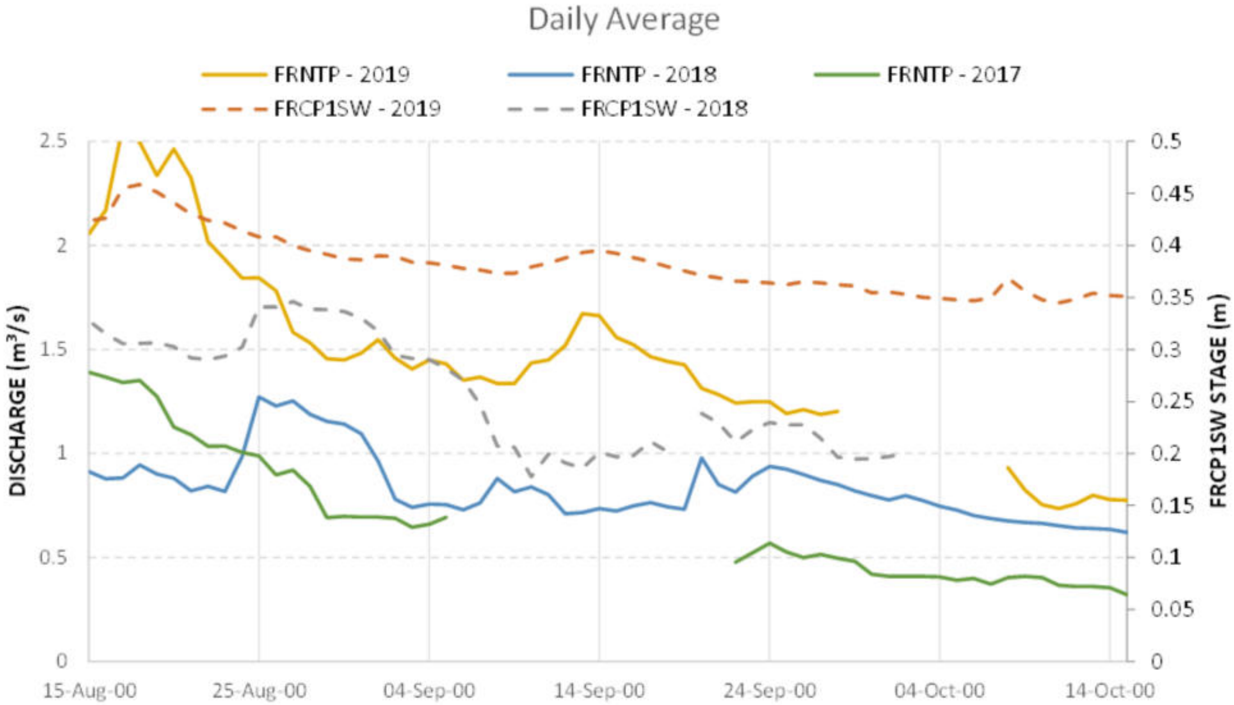
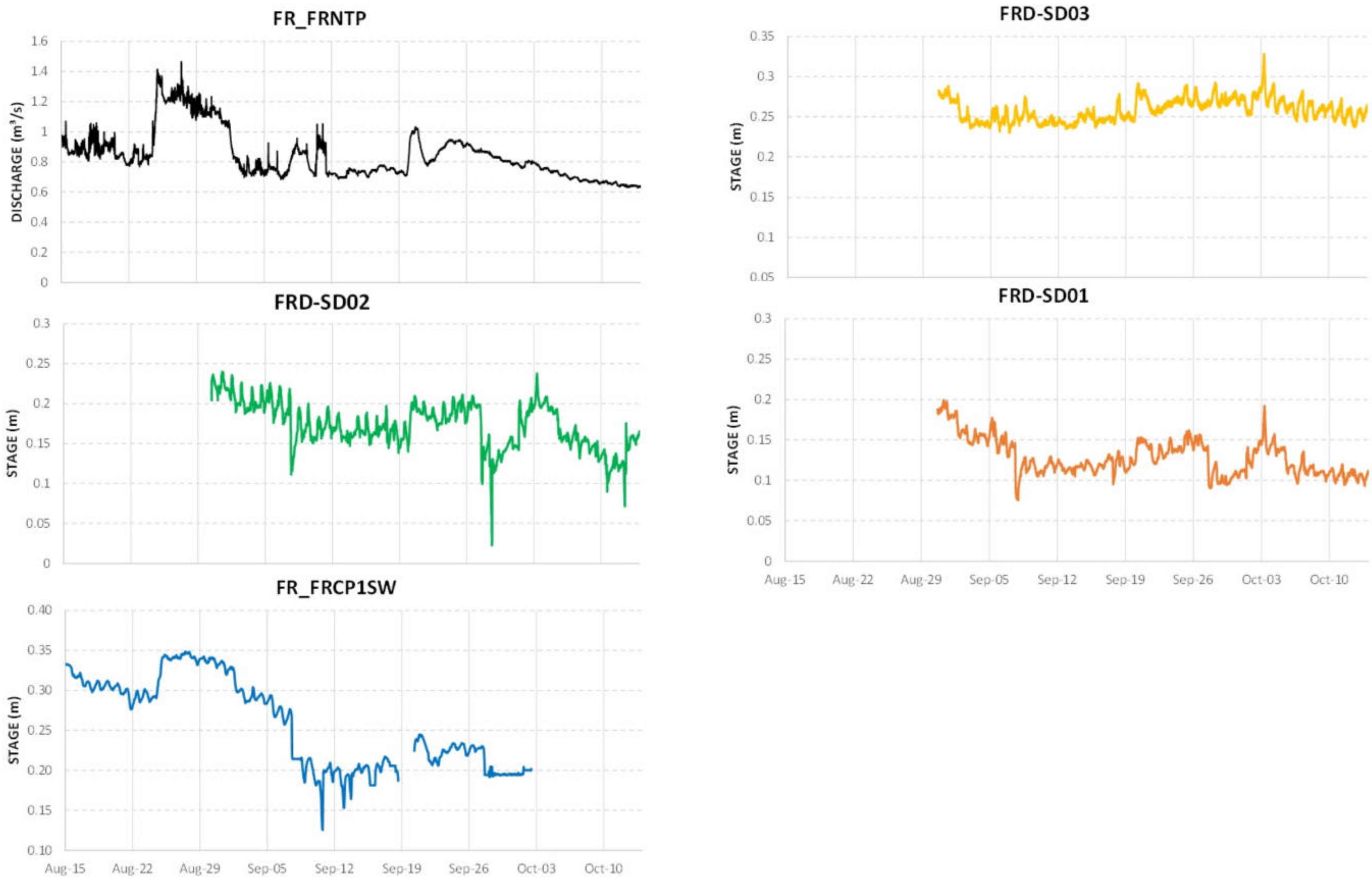


Table 5. Mean daily streamflow at WSC Fording River at the Mouth station (08NK018) during summer rearing, overwintering migration, and overwintering periods for WCT. Red shading highlights flows during the 2018-2019 season, when the drying extent was the largest observed in recent surveys (2017-2019). Blue shading highlights years with lower mean flows than 2018-2019.

Station	Year	Mean Daily Flow (m^3/s)		
		Summer Rearing	Overwintering Migration	Overwintering[1]
		July 15 to September 30	September 1 to October 15	October 15 to March 31
WSC 08NK018	1970	4.16	2.84	1.88
	1971	7.37	3.99	2.41
	1972	11.82	6.33	2.92
	1973	5.11	3.54	2.21
	1974	9.24	4.30	2.29
	1975	8.53	5.01	3.12
	1976	11.83	5.68	2.42
	1977	5.58	4.58	2.34
	1978	9.14	5.28	2.43
	1979	4.36	2.92	1.81
	1980	5.68	4.67	2.75
	1981	9.51	3.60	2.12
	1982	5.57	4.38	2.37
	1983	5.64	2.70	1.93
	1984	4.91	3.22	1.82
	1985	4.95	5.12	2.37
	1986	5.91	5.31	2.89
	1987	6.44	2.97	1.67
	1988	3.37	2.33	1.83
	1989	5.52	4.46	2.34
	1990	10.50	4.66	2.75
	1991	9.22	4.39	2.57
	1992	8.17	5.02	2.50
	1993	13.09	6.86	3.26
	1994	4.76	3.02	2.30
	1995	8.21	3.71	2.93
	1996	7.90	6.01	3.16
	1997	6.26	4.23	2.51
	1998	5.84	3.40	2.22
	1999	7.94	3.80	3.72
	2000	5.10	3.87	2.34
	2001	3.16	2.17	1.77
	2002	7.21	4.19	2.44
	2003	4.35	2.94	2.71
	2004	8.96	7.24	3.53
	2005	9.14	8.94	5.41
	2006	5.72	4.07	2.94
	2007	4.96	3.66	2.64
	2008	5.80	3.82	2.18
	2009	7.10	3.93	2.59
	2010	6.95	6.20	2.81
	2011	6.99	4.18	2.85
	2012	10.44	3.97	3.01
	2013	8.73	5.60	2.80
	2014	7.43	5.86	2.97
	2015	5.01	4.10	2.48
	2016	6.13	4.33	3.23
	2017	4.38	2.75	2.20
	2018	4.99	3.28	2.42
	2019	8.02	4.25	-
Average - all Years		6.94	4.35	2.59

[1] Overwintering period starts on October 15 of the previous year and goes to March 31 of current year.

Figure 9. Daily average flow (m^3/s) at FR_FRNTP and stage (m) data from gauges FRD-SD03, FRD-SD02, FRD-SD01 and FR_FRCP1SW between August 15, 2018 and October 15, 2018 from upstream to downstream in the UFR.



ECOFISH RESEARCH

3.4. Fish Use of Stranding Sensitive Habitat

3.4.1. Fording River Drying Reaches

Two scenarios based on the extent of drying in the northern and southern drying reaches and relative fish use in S7 and S9 were used to evaluate whether stranding within the UFR mainstem could have affected a substantial portion of the UFR WCT population: the relative habitat areas affected by drying and the relative fish use by river segment. These estimates are meant to approximate a conservative upper bound of possible stranding rather than as reliable estimates of stranding in the drying reaches.

The first estimate relies on relative habitat areas affected by drying. The maximum stream length of drying based on field surveys varied in the southern drying reach from 1,696 m in 2017/2018 to 2,926 m in 2018/2019 (Table 6). As a proportion of the total length of the UFR, 3.1% to 5.3% of the UFR length dried during fall and winter of these years, depending on year. Surveys were completed in the northern drying reach beginning in 2019/2020. The maximum stream length of drying was 970 m, which represents 1.7% of the total UFR length. The drying lengths noted above overlapped with the overwintering period for WCT and sum across the northern and southern reaches to 4.8% of the total length of the UFR in 2017/2018, 7.0% in 2018/2019, and 6.3% in 2019/2020 (Table 7). If WCT were evenly distributed during overwintering then these represent a maximum mortality if all fish are killed by stranding in the sections that dry. For the rearing period, no drying occurred prior to October 15 in the fall of 2017 or 2019. Intermittent drying and wetting was observed in fall of 2018 with a length of 316 m measured during surveys. In the one year of data when the northern section was surveyed it was observed to dry six weeks before station FR_FRCP1SW, which is the station that showed the earliest drying from 2017-2019 in the southern drying reach. In the absence of drying data for the northern reach, it is assumed that this reach also went dry during part of the rearing period in 2018. This sums to represent 2.3% of the available habitat in the UFR mainstem affected by drying in the northern drying reach (Table 7).

WCT are not evenly distributed throughout the UFR mainstem (Cope *et al.* 2016), so calculations were also completed based on proportional fish use as determined from the available telemetry data (EoC Team 2021). For example, the majority of WCT overwinter in deep pools in the S6 pools and in Henretta Pit Lake (Cope *et al.* 2016). River segment S7 (length = 4,970 m) overlaps with the southern drying reach, and S9 (length = 3,820 m) overlaps with the northern drying reach. The relative fish use of these two segments during the rearing period is estimated as 10.0% in S7 and 5.7% in S9 (Table 8). In the overwintering period, relative fish use declines to 2.7% and 3.8%, in S7 and S9. The relative fish use of S7 and S9 was multiplied by the proportion of each segment estimated to be dried based on the maximum extent of drying observed during the rearing and overwintering periods (Table 6). This assumes that all WCT in the drying reaches of S7 or S9 become stranded and do not move in response to drying. In the overwintering period, drying was observed in all years of the survey in the southern reach and is assumed to have occurred in all years in the northern reach. This yields a potential population effect of 1.9% to 2.5% of the WCT UFR mainstem population for the overwintering period (Table 7). No drying occurred during the rearing period in 2017 and 2019;

however, the relative fish use of this habitat indicates that when drying occurs earlier, such as observed in September 2018, then it is possible that a greater number of WCT may be exposed to stranding mortality because more individuals are presumed to be present at this time of year. However, only a relatively short section of the river dried during the rearing period yielding a maximum potential population effect in 2018/2019 of 2.1% of the mainstem UFR WCT population (Table 7). This assumes that fish do not move in response to declining water level. This also assumes that the northern drying reach went dry (970 m of length) during the rearing period in fall 2018, an assumption that cannot be verified.

Overall, the estimates of potential population effect (% loss) from drying cannot be summed between the rearing and overwintering periods as they represent the same fish potentially lost (Table 7). Therefore, to bound the effect across the three years of available data and the two scenarios, the maximum score is taken for each scenario by year. This resulted in an estimate of loss of 1.9% to 4.8% of the WCT population in 2017/2018, 2.5% to 7.0% of the WCT population in 2018/2019, and 2.3% to 6.3% of the WCT population in 2019/2020.

Table 6. Maximum length of drying relative to the total length of the UFR mainstem in the northern and southern drying reaches in 2017/2018, 2018/2019 and 2019/2020 during rearing and overwintering periods.

Drying Reach	Season	Periodicity[1]	Surveyed Maximum Length of Drying (m)	Proportion of total length of the UFR[2]
Southern	2017/2018	Rearing	0	0.0%
		Overwintering	1696	3.1%
	2018/2019	Rearing	316	0.6%
		Overwintering	2926	5.3%
	2019/2020	Rearing	0	0.0%
		Overwintering	2511	4.5%
Northern	2019/2020	Rearing	0	0.0%
		Overwintering	970	1.7%

[1] October 15 used as rearing and overwintering period breakpoint

[2] Total length of the UFR estimated as 55.5 km (Cope *et al.* 2016)

Table 7. Evaluation of potential WCT population effect in the UFR, based on drying reach length relative to the full UFR and relative fish use of S7 and S9.

Scenario	Season	Periodicity	Upper Bound of Population Effect (% Loss)
Drying Length Relative to Length of UFR	2017/2018	Rearing	0.0%
		Overwintering	4.8%
	2018/2019	Rearing	2.3%[1]
		Overwintering	7.0%
	2019/2020	Rearing	0.0%
		Overwintering	6.3%
Relative Fish Use of S7 and S9	2017/2018	Rearing	0.0%
		Overwintering	1.9%
	2018/2019	Rearing	2.1%[1]
		Overwintering	2.5%
	2019/2020	Rearing	0.0%
		Overwintering	2.3%

[1] Assumes that the northern drying reach went dry during the rearing period in fall 2018

Table 8. Relative WCT use of S7 and S9 during the rearing and overwintering periods (data from EoC Team 2021).

River Segment	Drying Reach	Periodicity	Relative fish use %	Length of River Segment (m)
S7	Southern	Rearing	10.0	4,970
		Overwintering	2.7	
S9	Northern	Rearing	5.7	3,820
		Overwintering	3.8	

3.4.2. Redd Dewatering

To assess the potential for redd dewatering (i.e., stranding of buried life stages), transect data from the upper Fording River instream flow study (Healey *et al.* 2020) were used to estimate annual effective spawning habitat, which accounts for the amount of habitat that may be dewatered due to decreases in flow in the incubation period versus the spawning period. Results of the effective spawning habitat analysis are summarized in Table 9; for each year, this table compares the theoretical spawning habitat under average spawning flow conditions to the habitat that remains sufficiently wetted during the incubation flow minimum. This table demonstrates that each year, some spawning habitat may not be effective, varying from 2% (in 2005) to 40% (in 2011) for spawning habitat potentially dewatered upstream of Kilmarnock Creek, to 3% (in 2005) to 41% (in 2015) downstream of Kilmarnock Creek.

Effective spawning habitat was compared between the Decline Window and the historical period (Table 10). Effective spawning habitat was similar from 2017 to 2019 compared to the historical period; upstream of Kilmarnock Creek, modelling predicts that on average, ~26% of spawning habitat was not effective during the Decline Window (compared to 28% during the historical period), and downstream of Kilmarnock Creek ~ 21% of spawning habitat was not effective during the Decline Window (compared to 24% during the historical period)dewatering of spawning during the Decline Window is estimated to be similar to that calculated for the historical record, which indicates that the potential for redd dewatering was not elevated in the Decline Window compared to the historical period.

Table 9. Effective spawning habitat for the UFR mainstem by year from 1997 to 2019. The change in habitat indicates the quantity of spawning habitat that experiences < 10 cm water depth during the incubation period.

Zone	Year	Average Spawning Flow (m^3/s)	Minimum Incubation Flow	Average Spawning Habitat (m)	Average Effective Spawning Habitat (m)	Change	
						m	%
Fording U/S of Kilmarnock	1997	4.24	0.615	1.82	1.29	-0.53	-29
	1998	6.68	0.618	1.55	0.94	-0.60	-39
	1999	6.82	1.28	1.55	1.12	-0.42	-27
	2000	5.34	0.612	1.75	1.16	-0.58	-33
	2001	3.41	0.439	1.87	1.33	-0.54	-29
	2002	8.46	0.955	1.48	0.95	-0.53	-36
	2003	8.37	0.922	1.48	0.95	-0.53	-36
	2004	4.34	0.787	1.82	1.30	-0.52	-29
	2005	6.87	3.87	1.55	1.52	-0.03	-2
	2006	5.11	0.415	1.75	1.10	-0.65	-37
	2008	7.11	1.56	1.48	1.09	-0.39	-26
	2009	3.01	1.34	1.86	1.73	-0.12	-7
	2010	4.74	1.61	1.80	1.60	-0.20	-11
	2011	7.92	0.781	1.48	0.89	-0.59	-40
	2012	9.11	1.05	1.48	0.99	-0.49	-33
	2014	6.47	0.899	1.62	1.08	-0.53	-33
	2015	3.35	0.280	1.87	1.17	-0.70	-37
	2016	2.97	0.920	1.82	1.55	-0.26	-15
	2017	5.80	0.690	1.68	1.09	-0.60	-35
	2018	4.98	0.820	1.77	1.27	-0.49	-28
	2019	4.70	1.45	1.80	1.55	-0.24	-14
Fording D/S of Kilmarnock	1997	4.24	0.615	4.39	3.23	-1.17	-27
	1998	6.68	0.618	3.40	2.31	-1.10	-32
	1999	6.82	1.28	3.40	2.67	-0.74	-22
	2000	5.34	0.612	4.09	2.93	-1.16	-28
	2001	3.41	0.439	4.43	2.96	-1.48	-33
	2002	8.46	0.955	3.18	2.31	-0.87	-27
	2003	8.37	0.922	3.18	2.31	-0.87	-27
	2004	4.34	0.787	4.39	3.33	-1.06	-24
	2005	6.87	3.87	3.40	3.30	-0.10	-3
	2006	5.11	0.415	4.09	2.54	-1.54	-38
	2008	7.11	1.56	3.18	2.58	-0.60	-19
	2009	3.01	1.34	4.34	4.06	-0.27	-6
	2010	4.74	1.61	4.29	3.88	-0.41	-10
	2011	7.92	0.781	3.18	2.19	-0.99	-31
	2012	9.11	1.05	3.18	2.38	-0.80	-25
	2014	6.47	0.899	3.64	2.67	-0.97	-27
	2015	3.35	0.280	4.43	2.60	-1.83	-41
	2016	2.97	0.920	4.19	3.76	-0.44	-10
	2017	5.80	0.690	3.87	2.72	-1.16	-30
	2018	4.98	0.820	4.16	3.22	-0.95	-23
	2019	4.70	1.45	4.29	3.78	-0.50	-12

Table 10. Average effective spawning habitat for the pre-Decline Window (1997-2016) and the Decline Window (2017-2018).

Zone	Year	Average Spawning Flow (m^3/s)	Minimum Incubation Flow (m^3/s)	Average Spawning Habitat (m)	Average Effective Spawning Habitat (m)	Change	
						m	%
Fording U/S of Kilmarnock	1997-2016	5.79	1.05	1.67	1.21	-0.46	-28
	2017-2019	5.16	0.99	1.75	1.30	-0.44	-26
Fording D/S of Kilmarnock	1997-2016	5.79	1.05	3.80	2.89	-0.91	-24
	2017-2019	5.16	0.99	4.11	3.24	-0.87	-21

4. DISCUSSION

4.1. Evaluation of Requisite Conditions

Requisite conditions are defined as the circumstances that would need to be met for dewatering of the mainstem UFR habitat to cause or contribute to the WCT population decline (Table 1). The drying of the UFR mainstem during fall and winter months has been observed dating back to the 1970s in two main areas, the northern drying reach near station FR_FR1 and the southern drying reach near station FR_FR4. Incidents of channel dewatering are not unique to the Fording River and are also known to occur elsewhere in the region, such as within the upper reaches of upper Kootenay River tributaries like the Wigwam River (Baxter and Hagen 2003, Prince and Cope 2001) and the Elk River (Prince and Morris 2003). Incidents of stranding in the UFR drying reaches causing WCT mortality have been observed within and prior to the Decline Window, indicating that these events are likely placing strong selective pressures on WCT to migrate to overwintering areas outside of these drying reaches. This is indeed what is observed in the WCT overwintering distribution, with the majority of WCT rearing in the middle and upper portions of the UFR mainstem and overwintering primarily in S6 pools and Henretta Pit Lake (Cope *et al.* 2016; EoC Team 2021). Additional overwintering locations near river segments S7 to S9 include: downstream of the confluence with Kilmarnock Creek in segment S7, the multi-plate plunge pool in segment S8, and Clode Flats in upper segment S8 and lower segment S9 (Cope *et al.* 2016). It is expected that some stranding mortality would nevertheless occur in most years in the UFR mainstem when drying occurs in side channels or drying reaches and this mortality may represent a consistent source of mortality for the UFR WCT population. However, because these events occur each year, most individuals in the population would be expected to avoid these areas when dewatering conditions are within a 'normal' range. In contrast, as noted in the literature (Section 3.1), drying events that are outside the natural range of conditions (timing, frequency or magnitude) have the potential to more negatively impact fish populations than during years with average conditions.

Relative fish use in S7 and S9, presumed based on historical data, were used to evaluate whether stranding within the northern and southern drying reaches during the Decline Window could have affected a substantial portion of the UFR WCT population. Drying was observed to occur in each year of observation at station FR_FRCP1SW in the southern reach but the dates of first drying and total extent of drying differed by year. In the three years of data in the Decline Window, drying was observed to be greatest in the fall and winter of 2018/2019, with first drying occurring on September 10, 2018, at FR_FRCP1SW. Using simple assumptions of potential stranding mortality from drying extent relative to UFR mainstem length provided an estimate for mortality from stranding in 2018/2019 of up to 7.0% of the population, including up to 2.3% during the early fall rearing period (Table 7). The estimates of mortality are considered upper bounds on potential effect since they do not include behavioural responses of fish to stage declines. However, the potential for mortality from stranding in 2018/2019 is supported by incidental observations of stranding in the UFR mainstem near FR_FRCP1SW in September and October 2018, including stranding of 15 WCT individuals on September 10, 2018. A stranding event was also observed in the upper Fording River side channel in the same time period between August 30 and September 7, 2018. Whether by coincidence or not, a stranding event was also observed in the Elk River Side Channel outside of the UFR on September 10, 2018.

Requisite conditions to cause, with respect to timing, duration, and location were met in that more extreme drying was observed in fall and winter 2018/2019 compared to other recent years of drying surveys, and this drying may have contributed to stranding mortality as the dewatered areas occur in habitats used by WCT in the UFR. In addition, flows at the Fording River WSC gauge were some of the lowest in the last 20 years in both 2017/2018 and 2018/2019 fall overwintering migration and overwintering periods (Table 5; Appendix A: Zathey and Robinson 2021). However, the requisite condition related to spatial extent and intensity were not met for sole cause of the decline, particularly with respect to the proportion of habitat affected as this represents a maximum of 7% of the UFR WCT population based on the relative length of the UFR that dried. We conclude that drying in the UFR mainstem causing stranding mortality is unlikely to have been the primary cause of the WCT population decline. However, because low flows occurred during the WCT summer rearing period, it is possible that stranding mortality from drying in the fall and winter of 2018/2019 was greater in 2018 and therefore a possible contributing cause for WCT decline for both adults and juveniles.

Regarding the potential for redd dewatering to have been a sole or contributing cause for WCT population decline, there is no evidence to suggest that conditions during the Decline Window were poorer than before the Window. Current evidence suggests that the potential for redd dewatering was similar in the Decline Window to average conditions in previous years. Therefore, redd dewatering in the spring and summer is not considered likely as a sole or contributing cause for WCT population declines.

4.2. Uncertainty

Key uncertainties in this assessment are listed here.

- There are limited empirical observations of WCT mortalities from dedicated stranding searches at the time of drying and therefore uncertainty on the intensity of stranding that may have occurred. The potential for stranding mortality was inferred from measurements of dewatered habitat and the historical fish use information.

- The relative fish use was estimated for S7 and S9 using telemetry data collected in 2012 to 2015 (EoC Team 2021). These data are an estimate of actual fish distribution in the UFR mainstem during the Decline Window and during key periods when WCT could have been stranded.

- The dewatering observations of the northern and southern drying reaches are limited to 2017 to 2020. This period overlaps with the Decline Window, but does not allow for an assessment of dewatering extent prior to the Window. Routine observations at FR_FR1 and FR_FR4 suggest that dewatering of the northern and southern drying reaches occurs each winter; however, there is a lack of data to evaluate changes in the timing and extent of drying between the Decline Window and the historical period. Long-term flow records at the Fording River WSC gauge suggest that 2017/2018 and 2018/2019 were dry years relative to flows since early 2000s. However, the uncertainty regarding historical drying timing and extent is further highlighted by the lack of a strong correlation between flow conditions in the UFR and drying conditions in the two drying reaches. A flow level model would help to better understand the relationships between drying timing and extent, the flow of the UFR, and other environmental drivers of drying, but may it be hard to achieve such a model with current data.

- The data related to extent of dewatering in the northern reach are the most limited, as detailed and specific data collection on extent began in that reach in fall 2019 after the Decline Window. For the assessment here, drying was assumed to occur in the northern drying reach at a similar time relative to timing of southern reach drying, and also assumed to occur during the rearing period in fall 2018. Current data suggest that the northern drying reach dewaters several weeks earlier than the southern drying reach.

- Potential effects of stranding to the UFR WCT population were estimated using worst-case scenarios based on the proportion of stream length and relative fish use in S7 and S9. These scenarios were used to place upper bounds on whether stranding within the UFR mainstem could have affected a substantial portion of the UFR WCT population across all age classes. These scenarios do not account for fish movement away from the drying reaches in response to decline water level.

- There are data gaps in the flow and stage data during and prior to the Decline Window due to icing and other effects. Data gaps and uncertainties associated with the flow and stage data record are summarized in Wright *et al.* (2021).

- The evaluation of effective spawning habitat does not account for annual differences in timing of spawning, nor does it account for fish selecting deeper habitats than indicated by the habitat suitability criteria.

5. CONCLUSION

This assessment evaluated the potential for UFR mainstem channel dewatering to have caused, or contributed to the observed decline of WCT. The drying of the UFR mainstem during fall and winter months has been recorded dating back to the 1970s. Incidents of channel dewatering are not unique to the Fording River and are also known to occur within the upper reaches of other tributaries in the region. Stream salmonids are adapted to seasonal, periodic changes in stream drying, and initiate movements in the fall to overwintering habitats as water temperature declines. However, extreme drying events that are outside the normal the range of timing or extent have the potential to negatively impact individuals and populations.

Requisite conditions for cause of the decline, with respect to timing, duration, and location were met in that drying was observed early in the fall of 2018 and the extent of drying was greater in winter 2018/2019 than in recent years with drying data. Flows in summer, fall, and winter 2017/2018 and 2018/2019 were also lower than most years since 2000. The drying in fall 2018 also coincided with observations of WCT stranding in the UFR mainstem and Fording River Side Channel. However, the requisite conditions of spatial extent and intensity were not met for sole cause of the decline, particularly with respect to the proportion of fish affected as this represents a maximum of 7.0% of the UFR WCT population. We further note that this value may be an unrealistically high estimate of WCT mortality and is provided solely for the purpose of the EoC and the identification of possible causes of the decline. We conclude that drying in the UFR mainstem causing stranding mortality is unlikely to have been the primary cause of the WCT population decline. Because dewatering occurred during the WCT summer rearing period, it is possible that stranding mortality from drying in the fall and winter of 2018/2019 was greater in 2018 than previous years and therefore could be a contributing cause for WCT decline for both adults and juveniles. However, due to uncertainty in the relationship between surface flow and drying it is unclear whether the timing and extent of drying may have occurred as early and widespread in the historical window as it did in 2018. Further, UFR mainstem drying events have been observed in the historical period back to the 1970s. Therefore, our conclusion that mainstem dewatering may have contributed to the WCT population decline has some uncertainty.

Regarding the potential for redd dewatering to have been a sole or contributing cause for WCT population decline, there is no evidence to suggest that conditions during the Decline Window were worse than before the Window. Redd dewatering in the spring and summer is unlikely to have been a sole or contributing cause of the WCT population decline.

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PROJECT MAPS

TECK COAL LTD.

Mainstem Dewatering Southern Extent Sep. 2017 - Apr. 2018

Legend

- Hydrometric Gauge
- Reach Break
- Water Network
- Water Management Polygons
- Mainstem Dewatering

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

0 100 200 400 600 800 1,000 m

Scale: 1:25,000

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3			
4			
5			

Date Saved: 2021-06-09
Coordinate System: NAD 1983 UTM Zone 11N

ECOFISH RESEARCH

Map 2

Path: M:\Projects-Active\1229_EVWQP\MXD\Fisheries\1229_FRDS_MainstemDewatering_20172018_3966_20210609.mxd

TECK COAL LTD.

Mainstem Dewatering Southern Extent Sep. 2018 - Apr. 2019

Legend

- Hydrometric Gauge
- Reach Break
- Water Network
- Water Management Polygons
- Mainstem Dewatering

MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

0 100 200 400 600 800 1,000 m

Scale: 1:25,000

NO.	DATE	REVISION	BY
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4			
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Date Saved: 2021-06-09
Coordinate System: NAD 1983 UTM Zone 11N

ECOFISH RESEARCH

Map 3

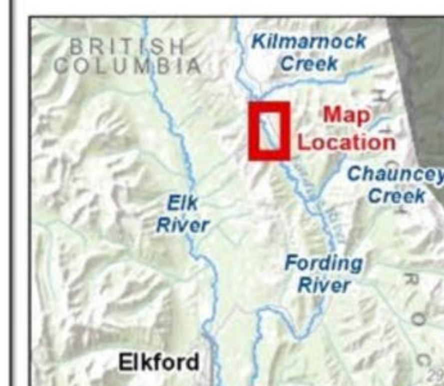
Path: M:\Projects-Active\1229_EVWQP\MXD\Fisheries\1229_FRDS_MainstemDewatering_20182019_3966_20201104.mxd

TECK COAL LTD.

Mainstem Dewatering Southern Extent Sep. 2019 - Apr. 2020

Legend

- Hydrometric Gauge
- Reach Break
- Water Network
- Water Management Polygons
- Mainstem Dewatering



MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

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4			
5			

Date Saved: 2020-11-12
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ECOFISH RESEARCH

Map 4

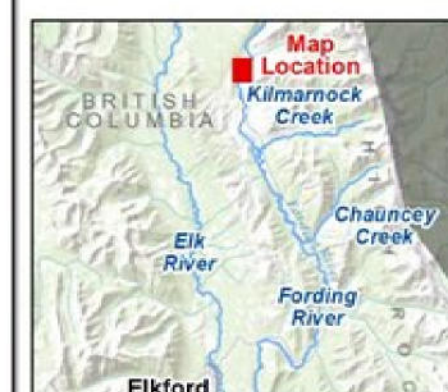
Path: M:\Projects-Active\1229_EVWQP\MXD\Fisheries\1229_FRDS_MainstemDewatering_20192020_3966_20201104.mxd

TECK COAL LTD.

Mainstem Dewatering Northern Extent Sep. 2019 - Apr. 2020

Legend

- Hydrometric Gauge
- Water Network
- Water Management Polygons
- Mainstem Dewatering



MAP SHOULD NOT BE USED FOR LEGAL OR NAVIGATIONAL PURPOSES

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Scale: 1:8,000

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4			
5			

Date Saved: 2021-03-04
Coordinate System: NAD 1983 UTM Zone 11N

ECOFISH RESEARCH

Map 5

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APPENDICES

Summary of ephemeral conditions in the upper Fording River watershed

Information Review

June 2021

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1 Introduction

Sections of streams that exhibit seasonal drying conditions (i.e. ephemeral) are not uncommon in mountain environments. While there are many factors that contribute to ephemeral sections, they can occur along the profile of a stream where excess substrate aggrades allowing the water table elevation to drop below the channel elevation under low flows. The upper Fording River has two sections that are known to dry seasonally. These ephemeral sections create seasonal fragmentation along the Fording River, and if not avoided by fish through timing of movements, can impede fish migration or create pools that strand fish.

This report characterizes the history of these drying sections to better describe where these sections are occurring, when drying occurs, and how long these sections stay dry. These accounts are limited by historical reports whose focus was not on the drying sections specifically, but noted them as part of overall understanding of flow and fish habitat. Teck began targeted studies of drying sections of the Fording River in 2017, however, these surveys have evolved overtime as the frequency of the surveys have changed and new areas have been included. Understanding the nature of these drying sections will provide context to the upper Fording River Evaluation of Cause.

2 Historical observations

As noted, there are two sections of the Fording River known to exhibit ephemeral conditions. One section, referred to as the southern drying section is located on the southern mine boundary of Fording River Operations, in the area of the Cataract Creek confluence with the Fording River. Drying in this section was documented as early as the 1970's, where low flow caused isolated pools (Lister and Kerr Wood Leidel 1980). The second is centered around what is now the Turnbull Bridge, downstream of the Fording River/Henretta Creek confluence to a point near the Post Pond Creek inlet. This is referred to as the northern section. This section was not historically documented in the 1970's due to the scope of the monitoring program (Lister and Kerr Wood Leidel 1980).

As noted, ephemeral conditions of the Fording River throughout the fall/winter is not a new phenomenon (Lister and Kerr Wood Leidel 1980). Water quality records at stations FR_FR4 and FR_FR1 have been used to provide long-term, historical accounts of whether or not surface water was present at these points in the southern and northern sections, respectively. These stations were chosen as they have long-standing data, as well as proximity to the current southern and northern drying regions. These stations were established as water quality data and not to describe ephemeral conditions. They are therefore limited to some extent. First, these are instantaneous accounts. The water quality records can provide results to a monthly resolution. Second, recent surveys have shown that dry conditions first appear at areas away from these stations within each section. Therefore, these records provide a general insight to surface water conditions and not the same spatial and temporal resolution as the current surveys provide. These records likely underestimate drying timing, duration and total extent. There are missing data within the historical record when samples were unable to be taken, Teck is unable to confirm that these sites were dry at those times, as other reasons may have precluded sampling.

With these caveats in mind, we can still extrapolate some useful information to form historical context of the southern and northern drying sections. Key findings are:

1. The FR_FR4 and FR_FR1 records provide evidence to support the statement that these ephemeral conditions are not new. Water quality sampling records at both stations date back to 1976 and both show dry conditions in every year.
2. Ephemeral periods are consistently observed during base flows and when ice would be expected. Both sections are routinely dry from December-February.
3. Lastly, is the extent of time when they are dry. Dry conditions typically occur in the southern section from December-February (Figure 1). Dry conditions in the northern section appear more persistent (relative to the southern section) as they more frequently occur in November and routinely extend into March (Figure 2). These last two points are consistent with current understanding. It is important to note that these data represent conditions at the water stations, and do not necessarily represent the whole drying stretches as we understand them today with focused drying surveys.

Note: Figure 1 and Figure 2 represent data from both monthly water sampling events at point locations and in more recent years, during monthly drying surveys over continuous reaches. A month denoted as drying does not mean that the site was dry all month, but that it was dry during the single sampling event for that month. When multiple drying surveys were conducted (2017 onwards), a month represents when conditions were dry for most of the month.

Southern Dry Section (FR4)												
Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1976												
1977												
1978												
1979												
1980												
1981			17-Mar									
1982												
1983												
1984				9-Apr								
1985			18-Mar									
1986			18-Mar									
1987				7-Apr								
1988				7-Apr								
1989				17-Apr								
1990												
1991				1-Apr								
1992			16-Mar									
1993				5-Apr								
1994				4-Apr								
1995			20-Mar									4-Dec
1996			18-Mar									3-Dec
1997				21-Apr								1-Dec
1998			17-Mar									7-Dec
1999				12-Apr								6-Dec
2000			27-Mar									5-Dec
2001			19-Mar									3-Dec
2002				1-Apr								3-Dec
2003			18-Mar									8-Dec
2004			22-Mar									6-Dec
2005		7-Feb									28-Nov	
2006												11-Dec
2007			19-Mar									3-Dec
2008			18-Mar									1-Dec
2009				6-Apr								7-Dec
2010			1-Mar								2-Nov	
2011				4-Apr								6-Dec
2012			20-Mar									
2013	8-Jan		4-Mar									
2014												2-Dec
2015			2-Mar									1-Dec
2016	5-Jan											12-Dec
2017			29-Mar							18-Oct		
2018		7-Feb	5-Mar	26-Apr								5-Dec
2019	29-Jan			9-Apr								
2020												

Figure 1. Summary of instantaneous surface flow conditions obtained from water quality sampling records and drying surveys at FR_FR4 in the southern dry section (red=dry; blue = wetted; slash=no data). Years not reported indicate years lacking data.

Northern Dry Section (FR1)												
Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1976												
1977												
1978												
1979												
1980												
1981												
1982												
1983												
1984												
1985												
1986												
1987												
1988												
1989				17-Apr								
1990												
1991				1-Apr							04-Nov	
1992			16-Mar									
1993												
1994												
1995				24-Apr								4-Dec
1996			18-Mar								19-Nov	
1997				21-Apr								1-Dec
1998				20-Apr								7-Dec
1999				12-Apr								6-Dec
2000				3-Apr							13-Nov	
2001				23-Apr								3-Dec
2002				9-Apr								3-Dec
2003				1-Apr								8-Dec
2004			30-Mar									6-Dec
2005		7-Feb									7-Nov	
2006	3-Jan		20-Mar									11-Dec
2007			19-Mar								5-Nov	
2008				7-Apr								1-Dec
2009					4-May							7-Dec
2010			8-Mar								2-Nov	
2011				12-Apr								6-Dec
2012				16-Apr								4-Dec
2013				1-Apr								17-Dec
2014				15-Apr								2-Dec
2015			16-Mar									8-Dec
2016				4-Apr								7-Dec
2017			27-Mar									4-Dec
2018			14-Mar								5-Nov	3-Dec
2019	10-Jan		19-Mar									
2020												

Figure 2. Summary of instantaneous surface flow conditions obtained from water quality sampling records and drying surveys at FR_FR1 in the northern dry section (red=dry; blue = wetted; slash=no data). Years not reported indicate years lacking data.

In 1979 it was noted that the lower reaches of Kilmarnock (from the Fording River upstream to the rail crossing) dried up by late July causing fish to move out to the exits of the railway and highway culverts for overwintering (Lister and Kerr Wood Leidel 1980). In 1982, ice cover was noted throughout the winter (November to March) in the Fording River and dry bed conditions were recorded at this time (Golder 1982). Lister and Kerr Wood Leidel (1980) accounts corroborate the historical context of drying in the southern section Fording River as far back as the late 1970's, with drying conditions being identified near the current FR_FRCP1 monitoring location in October 1979 (Figure 3).

The first extensive mapping of the drying sections came in 2011. March 8-24 2011, Lotic Environmental conducted overwintering fish surveys in the upper Fording River, as part of the Environmental Assessment process for the proposed Swift Creek Project (McPherson and Robinson 2011). A crew surveyed the Fording River in its entirety from the 1 km upstream of the Fording River/Henretta Creek confluence to approximately 2 km downstream of the FRO southern property. Both the northern and southern sections were noted as ephemeral. The northern section was dry for approximately ~2 km, between the confluence of Henretta Creek to just upstream of Fish Pond Creek. The southern section was dry for ~3 km, from station FR_FR4 to the Greenhouse side channel, with exception of a ~300 m long section wetted at the Cataract Creek confluence. Surface flow returned where the Greenhouse side channel entered the mainstem Fording River.

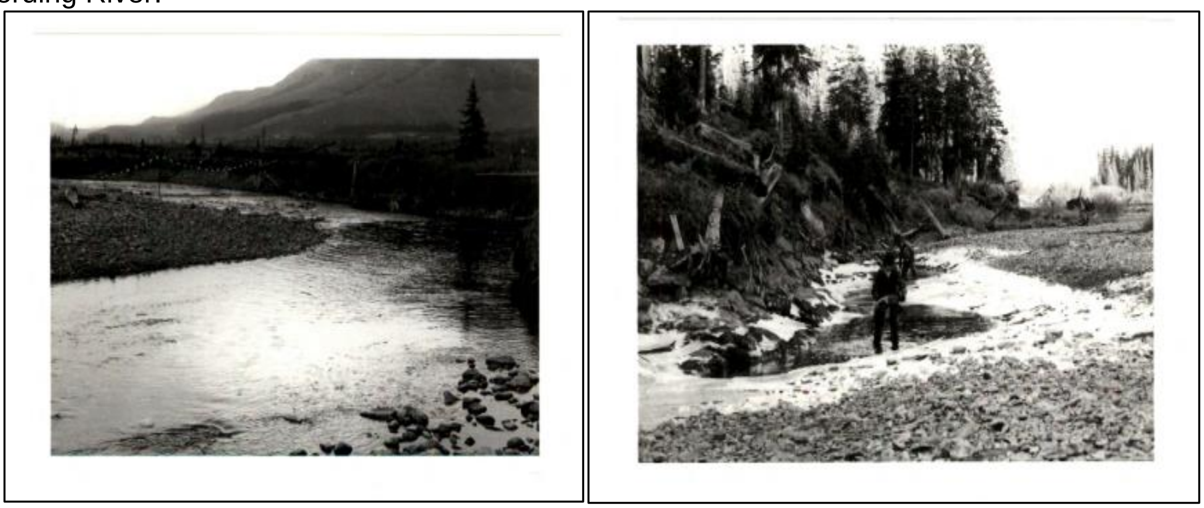

Figure 3. Photos of the Fording River flowing near FR_FR4 in July (left) and near FR_FRCP1 October (right) with isolated pools (from D. B. Listers & Kerr Wood Leidal 1980).

3 Drying surveys during decline window

Teck initiated more intensive surveys of the Fording River since October 2017 to describe the spatial and temporal characteristics of the southern drying section. These surveys were designed to support specific questions in the FRO Local Aquatic Effects Monitoring Program (LAEMP) to better understand how the drying sections would affect fish habitat and survival, and to inform water licencing discussions. Current surveys are conducted monthly under the FRO LAEMP from late summer (August-September) to approximately April (depending on rewatering of all sections) (Minnow and Lotic 2018). Surveys temporarily switched to biweekly during the drying period of 2019/20 to provide better resolution of drying conditions, but reverted back to monthly in winter 2020/21. Periodic high-flow measurements have been made in May/June.

During each monthly survey, a field crew walked the entire extent and recorded wet and dry areas to estimate lengths of dry sections. The resolution of these surveys was increased by installing continuous water level loggers in October 2017 at eight locations along the survey reach. The loggers provided a continuous record of wetted conditions that were confirmed by the ground-truthing surveys.

The southern drying surveys cover a 12.8 km long section of the Fording River from Chauncey Creek (FR_FRABCH) upstream to the South Tailings Pond (FR_FR2) (Table 1). The portion that dries can occur over a 3 km long section extending from FR_FRRD upstream to FR_FR4. Depending on the year, dry conditions can be limited to the lower 1.5 km (from FR_FRRD upstream to approximately FR_FRCP1), or over the entire 3 km section with a short 300 m wetted section in the middle that appears to be sustained by Cataract Creek (prior to cataract Creek diversion in August 2019, Figure 4. Fording River monitoring locations (Minnow and Lotic 2021).). This full extent (i.e. two dry sub-sections around Cataract Creek) was observed in 2018 and was also documented by McPherson and Robinson (2011), who also used an underwater camera to look for fish presence. They failed to observe any fish present during that survey.

Table 1. Fording River southern drying section site names and location descriptions

Site Name	Location
FR_FR2	Fording River upstream of the proposed AWTF discharge
GH_FR3	Fording River immediately downstream of the proposed AWTF discharge
FR_FR4	Fording River between Swift and Cataract
FR_FRCP1	Fording River Compliance Point
FR_FRCP1SW	Fording River ~1150 m downstream of the Compliance Point
FR_FRRD	Fording River upstream Porter Creek
GH_PC2	Fording River downstream of Porter
FR_FRABCH	Fording River upstream of Chauncey Creek
FR_UFR1	Fording River at Henretta Creek
FR_MULTIPLATE	Fording River Multiplate Culvert

<u>2017/2018</u>

The level logger installation and initial round of surveys were first completed in October 2017. Monthly surveys continued from October and first detected a dry section in January 2018. This ephemeral section extended 1.5 km between FR_FRCP1 downstream to FR_FRRD (Figure 4. Fording River monitoring locations (Minnow and Lotic 2021).). This section contained FR_FRCP1SW. The logger at this site showed that while the monthly surveys did not detect dry conditions until January 2018, the site had dried immediately after the December 2017 survey. The logger recorded dry conditions from December 14, 2017 to March 24, 2018 (Table 2. Drying times by site on Fording River from 2017-2019 based on level logger and drying survey data. Blue cells represented wetted conditions, white cells represent not sampled and slashed cells represent no data available.). April 2018 surveys confirmed water had returned to the dried section. No other dry section was reported in winter 2017/18. This year was a prime example that some drying can occur outside of the water quality stations used to describe historical context of drying in the upper Fording River (i.e. FR_FR1 and FR_FR4).

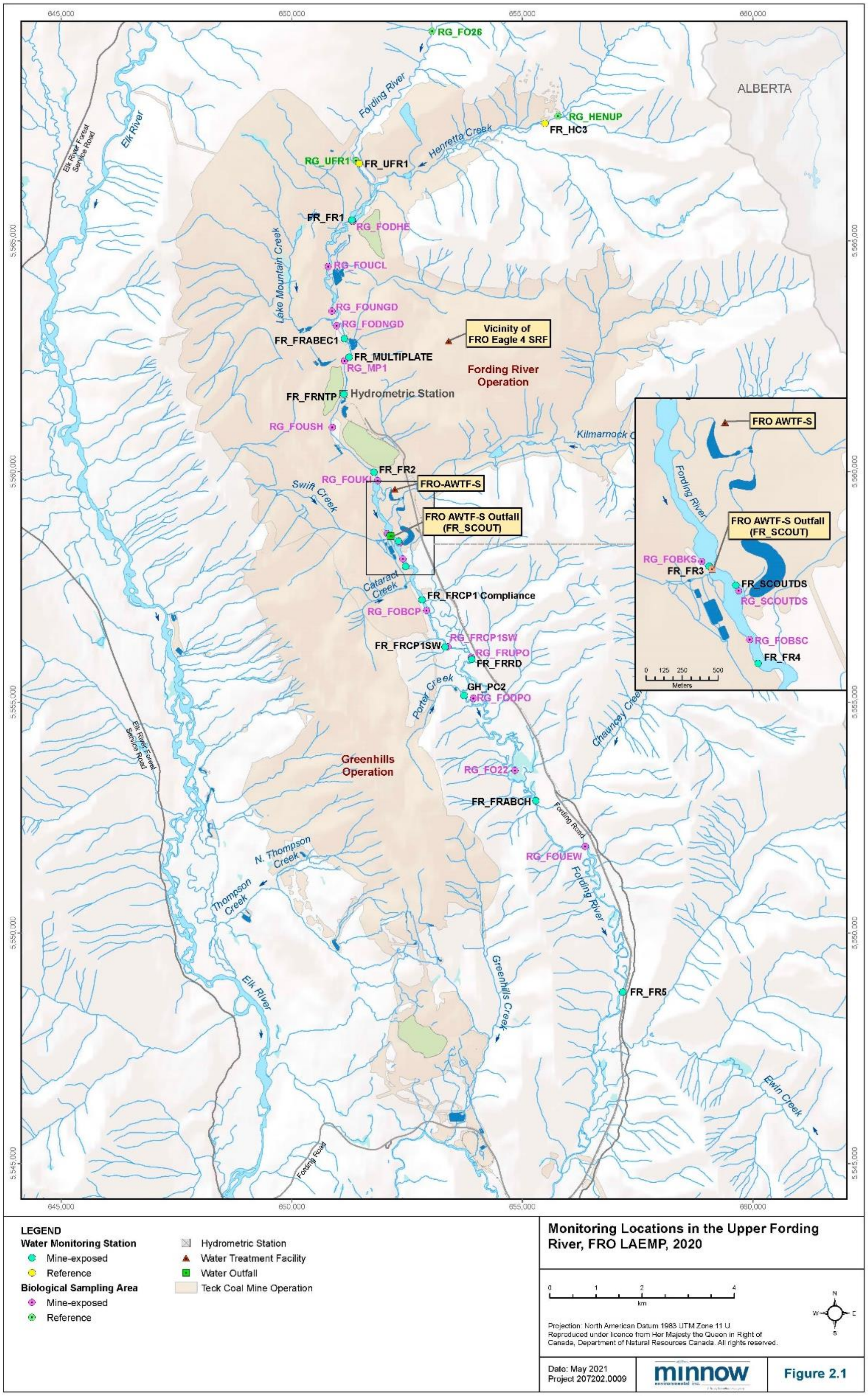

Figure 4. Fording River monitoring locations (Minnow and Lotic 2021).

2018/2019

Surveys were restarted in August 2018. The entire southern section was found wetted during August and September surveys. In October, there was a dried section of approximately 280 m between FR_FRRD and FR_FRCP1SW. Using water level records it was found that FR_FRCP1SW first went dry on approximately September 8, 2018 and exhibited a period of wetting/ drying corresponding with precipitation events for much of September/October (Table 2. Drying times by site on Fording River from 2017-2019 based on level logger and drying survey data. Blue cells represented wetted conditions, white cells represent not sampled and slashed cells represent no data available.). On September 18, 2018, 15 cutthroat mortalities were noted by Minnow Environmental Ltd. (Minnow) in a riffle upstream of the drying section (Hocking et al. 2020). An additional mortality was found by Ecofish on September 19, 2018 in an isolated pool. In October 2018, several cutthroat were found in isolated pools within the drying reach during drying surveys.

In November, the extent of the drying was 1,190 m. November surveys also identified a second dried section of approximately 170 m, located upstream of the Cataract Creek confluence. In December, the dried section at FR_FRCP1SW covered approximately 1,650 m and extended from upstream of FR_FRRD to just downstream of FR_FRCP1. The dried section near the Cataract Creek confluence lengthened to approximately 480 m, and a third dried section was identified upstream of FR_FR4 and downstream of Kilmarnock Creek pond, covering approximately 630 m. The logger at FR_FR4 remained wetted.

The dry section downstream of FR_FRCP1 remained fully dry and the section detected between FR_FRCP1 and FR_FR4 in November 2018 persisted in January, February and March 2019, and had increased in size to approximately 2 km in length and included the level logger at FR_FR4 (Table 2. Drying times by site on Fording River from 2017-2019 based on level logger and drying survey data. Blue cells represented wetted conditions, white cells represent not sampled and slashed cells represent no data available.). This section was observed to be fully wetted and flowing during the site visit on April 9, 2019. The water logger at FR_FRCP1SW was corrupt over the winter, so the re-watering date cannot be confirmed. The monthly survey identified FR_FRCP1SW as dry on March 12, 2019 and FRO staff found this section wet on March 19, 2019, therefore, the date of re-watering is sometime within this eight-day window.

2019/2020

Surveys continued monthly in 2019 with exception of July (Minnow and Lotic 2020). In December 2019, monthly surveys switched to biweekly frequency to monitor changes more closely in drying extent and timing. The southern section remained wetted throughout summer and fall, and the primary site of drying (FR_FRCP1SW) still had flow during a survey on December 18- 20, 2019.

The northern section was added to the monthly surveys in October 2019 to cover a 6.1 km section of the Fording River from the Multiplate culvert (FR_MULTIPLATE) to a point upstream of the confluence with Henretta Creek (at FR_UFR1). Drying surveys in the northern extent started on October 21, 2019 and continued the remainder of 2019. On November 19, 2019, there were two dry sections, separated by a short-isolated pool. The two sections were 173 m and 136 m long. This section expanded to connect as one long section approximately 300 m on December 10, 2019 and remained 300-350 m in length for the remainder of the year. Although this location is further downstream from FR_FR1 (the station used for historical context), the timing of drying is similar to previous years.

Table 2. Drying times by site on Fording River from 2017-2019 based on level logger and drying survey data. Blue cells represented wetted conditions, white cells represent not sampled and slashed cells represent no data available.

Site	2017			2018												2019												2020											
	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
South Section																																							
FR_FR2																																							
FR_AWTF-S																																							
GH_FR3																																							
FR_FR4																Jan. 29*-Mar. 13*																							
FR_FRCP1																																							
FR_FRCP1SW			Dec. 17 - Mar. 24									Sep. 9		Mar. 12*														Jan 6 - Apr 14											
FR_FRRD																																							
GH_PC2																																							
FR_FRABCH																																							
North Section																																							
FR_MULTIPLATE																																							
FR_FOUCL																																							
FR_FRUPP																																							
FR_FRDSCC1																																							
FR_FR1																										Nov 18-Apr 14													
FR_UFR1																																							
NTP flow (m^3/s) on dry date			0.45									0.99		0.67																									

Dry

Drying (periodic)

* approximate

4 Current relationship of dry conditions to Fording River discharge

Fording River operates a hydrometric station near the North Tailings Pond (FR_FRNTP) to provide a longer term, local hydrological dataset. This record has been found to be strongly correlated to the discharge record collected within the southern drying section at station FR_FRCP1SW. It is also relevant to review this record for historical context of both the period when the upper Fording River Population study was operating (2013-2019) and the period of decline for this population (2017-2019). As drying conditions are related to low flow, the FR_FRNTP discharge record was plotted along with other regional hydrometric station to show both the mean August discharge and mean September-October discharge for the entire record periods (Figure 5). Regional stations were required to bolster the relatively limited FR_FRNTP station. Based on the longer-term dataset, both low flow indicators do not suggest that flows were unusually low for over the first half of the monitoring period (2013-2017). However, flows in 2017 and 2018 were below the 25th percentile, suggesting that discharge may have been atypical in those two years relative to the long-term record.

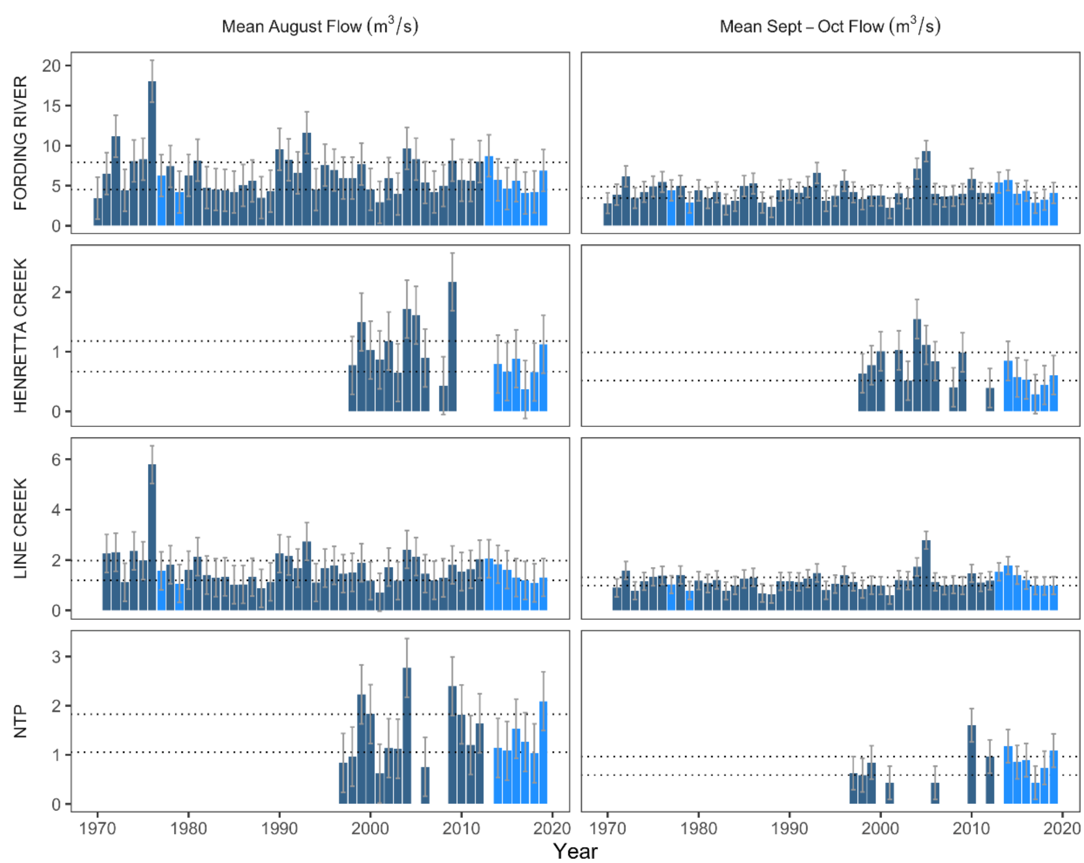

Figure 5. Two low-flow streamflow indicators for hydrometric stations within the Fording River watershed. Coloured bars indicate years since commencement of upper Fording River population monitoring. Dashed lines indicate the 25th and 75th percentiles.

Streamflow recorded at FR_FRNTP during periods when FR_FRCP1SW reported dry were fitted to a logistic regression model using streamflow at FR_FRNTP to predict the binary wet/dry variable at FR_FRCP1SW (Figure 6). The logistic regression has been trained on values between 0.2 and 14.1 m3/s (Dry discharges between 0.21 and 0.68 m3/s) and given the lack of data over lower discharges, model estimates of drying for discharges below 0.2m3/s are uncertain and should be treated with caution. Actual dry conditions were observed at a low probability (approximately 5%). Estimated probabilities were presented by calendar week with any probability > 0% highlighted in orange and probabilities >10% highlighted in red (Attachment 1). This table dating back to 1997 shows similar results to both monthly surveys and historical accounts in that dry periods center around baseflows in winter. This table does show that dry conditions were possible earlier in late summer/fall 2018 than in most other years. However, this is more an indication of lower flows in late summer than actual dry conditions. Monthly surveys provide more definitive accounts of dry conditions.

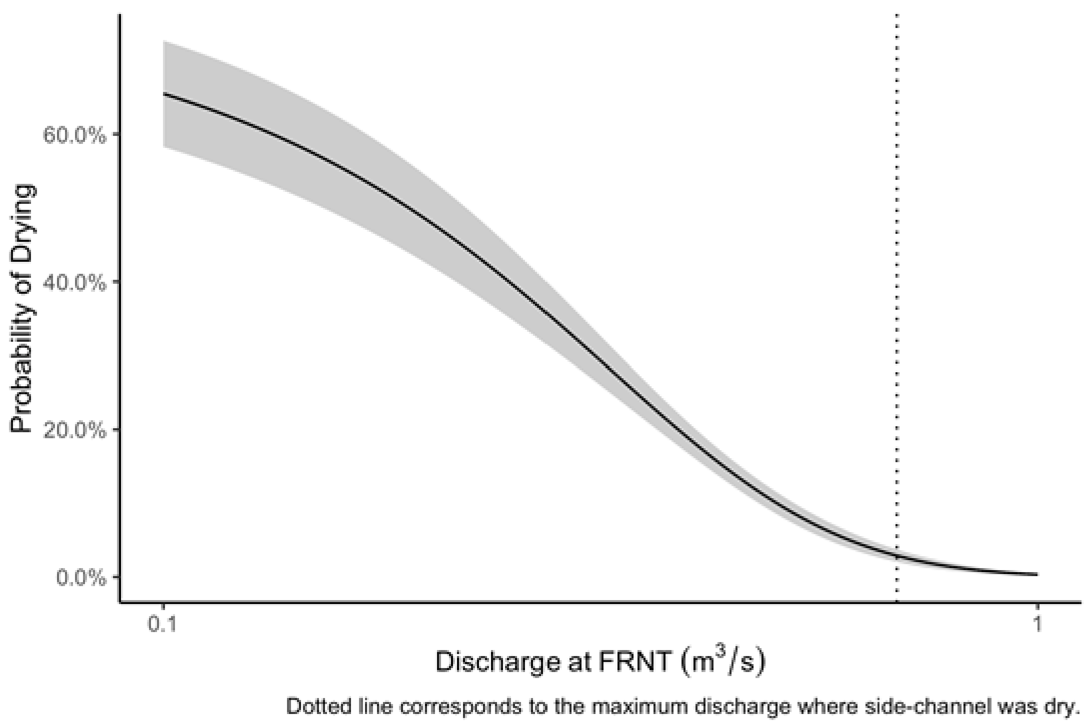

Figure 6. Logistic regression model using streamflow at FR_FRNTP to predict the binary wet/dry variable at FR_FRCP1SW.

5 Closure

Mainstem ephemeral conditions occurring on the Fording River near the southern and northern portions of Fording River property have been occurring for over 40 years as indicated through routine water quality monitoring and various studies (e.g., Lister 1980). All available information has been summarized to describe the spatial and temporal nature of these events. Until recently (2017 for the southern drying section and 2019 for the northern drying section), data are only available through monthly water quality sampling reports of a categorical "yes" or "no" for surface water present. These observations are typically limited to one observation per month and for one specific point location. Nonetheless, they show that drying typically occurs each year and in the same general locations.

Nonetheless, historical accounts demonstrate that the ephemeral events are strongly linked to the snow-melt dominated annual hydrograph in that the dry sections appear at low points in the year. While data limited, a predictive relationship between the longer-term FR_FRNTP hydrometric station and a station located within the southern dry section suggest flows were among the lower percentile (i.e. 25th percentile) in the August 2018 leading into winter of 2018/19, relative to this same period from 1997-2019 (i.e. when data are available from FR_FRNTP). The single year of concurrent monitoring in the northern and southern sections indicate drying occurring approximately six weeks earlier in the northern section compared to the southern section. This suggest that flow may have been limiting to fish movement in the northern section even early than September in the northern section in 2018. This assessment and conclusions are limited by data availability at this time and will require ongoing monitoring to better assess the relationships of flows along the Fording River profile.

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Attachment 1. Probability of FR_FRCP1SW dry conditions predicted from FR_FRNTP.

Year	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22	23	24	25	26
2019	NA	7.4%	NA	7.0%	7.4%	9.3%	9.7%	10.2%	9.1%	9.3%	11.4%	0.3%	0.0%	0.2%	0.4%	0.3%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2018											7.3%	8.4%	10.5%	11.1%	4.8%	2.7%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2017									3.4%	3.7%	2.0%	0.2%	0.1%	0.1%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2016	15.1%	15.9%	16.9%	17.9%	17.5%	17.4%	17.3%	17.4%	15.4%	13.4%	13.1%	9.7%	5.0%	0.2%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2015						6.6%	5.2%			11.7%	0.5%	1.0%	0.0%	0.1%	0.3%	0.0%	0.0%	0.0%	0.0%	0.0%						
2014	34.6%	31.9%	30.8%	30.4%	29.2%	NA	19.9%	17.7%	13.4%	9.2%	3.4%	3.3%	1.5%	0.2%	0.1%	0.0%	0.0%	0.0%							0.0%	0.0%
2012																				0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2011																				0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2010																			0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2009																		0.0%	0.0%	0.0%				0.0%	0.0%	0.0%
2008																						0.0%	0.0%			0.0%
2006																					0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2005																					0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2004																		0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2003																		0.0%	0.3%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2002																		0.1%	0.3%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2001																	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
2000																0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
1999																			0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
1998														10.8%	12.9%	9.9%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%
1997																			0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%

Year	27	28	29	30	31	32	33	34	35	36	37	38	39	40	41	42	43	44	45	46	47	48	49	50	51
2019	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	NA	0.9%	1.0%	1.4%	3.3%	2.7%	2.7%	3.4%	4.8%	3.1%		
2018	0.0%	0.0%	0.0%	0.0%	0.1%	0.3%	0.4%	0.4%	0.2%	0.9%	1.0%	0.7%	0.5%	1.1%	1.8%	2.3%	1.9%	1.5%	1.5%						
2017	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.2%	1.2%	1.8%		3.8%	4.4%	7.4%	8.3%	8.8%	8.6%	5.5%	6.6%	4.8%	4.1%	1.7%	3.8%	4.8%	6.3%
2016	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.2%	0.8%	1.0%	0.9%	0.6%	1.0%	0.5%	0.2%	0.6%	0.2%	0.3%	0.3%	0.3%	0.9%	0.9%		
2015							0.0%	0.1%	0.6%	0.7%	0.2%	0.3%	0.6%	0.4%	0.3%	1.1%	2.2%	3.1%	5.4%	4.0%	5.8%	8.8%	7.9%	7.6%	10.7%
2014	0.0%	0.0%	0.0%	0.0%	0.0%	0.2%	0.4%	0.1%	0.1%	0.0%	0.0%	0.1%	0.3%	0.5%	0.7%	0.7%									
2012	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.3%	0.5%	1.0%	1.1%	1.2%	1.1%	0.3%	0.1%	0.1%	0.0%	0.1%	0.1%	1.5%	2.2%		
2011	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.5%	0.7%																
2010	0.0%	0.0%						0.0%	0.0%	0.0%															
2009	0.0%	0.0%	0.0%			0.0%	0.0%	0.0%	0.0%																
2008	0.0%	0.0%	0.0%	0.0%																					
2006	0.0%	0.0%	0.0%	0.9%	0.5%	1.5%	0.2%	2.4%	6.4%																
2005	0.0%																								
2004	0.0%	0.0%			0.1%	0.2%	0.2%	0.0%	0.0%																
2003	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.2%	0.2%	0.3%																
2002	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.2%	0.2%	0.3%																
2001	0.0%	0.0%	0.3%	0.3%	0.7%	1.3%	2.4%	3.8%	5.5%	5.5%	7.3%														
2000	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	1.2%	0.1%																
1999	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.3%	0.9%	0.9%	1.6%	0.7%	1.1%	0.7%	1.1%	3.1%	0.1%	0.0%				
1998	0.0%	0.0%	0.0%	0.0%	0.0%	0.1%	0.6%	1.1%	2.0%	2.9%	3.4%	2.5%	1.2%	1.6%	2.8%	3.4%	4.8%	5.1%	5.8%	6.6%	7.1%	6.6%	7.3%	9.1%	
1997	0.0%	0.0%	0.1%	0.2%	0.2%	0.8%	0.7%	0.7%	1.9%	1.4%		1.2%	2.0%	2.2%	2.7%	2.6%	2.8%	3.4%	4.6%	5.9%	5.6%				