Quantifying lost and inaccessible habitat for Pacific salmon in Canada's Lower Fraser River

Riley J. R. Finn, Lia Chalifour, Sarah E. Gergel, Scott G. Hinch, David C. Scott, & Tara G. Martin

July 2021

This article was originally published at:

https://doi.org/10.1002/ecs2.3646

esa

ECOSPHERE

FRESHWATER ECOLOGY

Quantifying lost and inaccessible habitat for Pacific salmon in Canada's Lower Fraser River

Riley J. R. Finn,[1,2,†] Lia Chalifour,[1,3] Sarah E. Gergel,[4] Scott G. Hinch,[5] David C. Scott,[2,5] and Tara G. Martin[1]

[1]*Department of Forest and Conservation Science, Faculty of Forestry, The Conservation Decisions Lab, University of British Columbia, 2424 Main Mall, Vancouver, British Columbia V6T 1Z4 Canada*
[2]*Raincoast Conservation Foundation, Sidney, British Columbia V8L 3Y3 Canada*
[3]*Department of Biology, University of Victoria, Victoria, British Columbia V8W 2Y2 Canada*
[4]*Department of Forest and Conservation Science, Faculty of Forestry, University of British Columbia, 2424 Main Mall, Vancouver, British Columbia V6T 1Z4 Canada*
[5]*Department of Forest and Conservation Science, Faculty of Forestry, Pacific Salmon Ecology and Conservation Laboratory, University of British Columbia, 2424 Main Mall, Vancouver, British Columbia V6T 1Z4 Canada*

Citation: Finn, R. J. R., L. Chalifour, S. E. Gergel, S. G. Hinch, D. C. Scott, and T. G. Martin. 2021. Quantifying lost and inaccessible habitat for Pacific salmon in Canada's Lower Fraser River. Ecosphere 12(7):e03646. 10.1002/ecs2.3646

Abstract. Loss of connectivity caused by anthropogenic barriers is a key threat for migratory freshwater species. The anadromous life history of salmonids means that barriers on streams can decrease the amount of habitat available for spawning and rearing. To set appropriate targets for restoration, it is important to know how different populations have been impacted in terms of the location and extent of historically available habitat that has been lost or has become inaccessible. Using mapped and predicted barriers to fish passage in streams and diking infrastructure, the amount of both floodplain and linear stream habitat that remains accessible today was estimated for 14 populations of salmon in the Lower Fraser River, British Columbia, Canada's most productive salmon river. To place these estimates within a historical context, the floodplain area was estimated using vegetation records from the 1850s, and lost streams were estimated using a digital elevation model-derived stream network. To bolster areas where little mapping has been done, current barrier data were used to predict locations likely to have barriers. Accessibility to floodplain was poor across the entire region with only 15% of the historical floodplain remaining accessible. Linear stream habitat ranged in accessibility from 28% to 99% across populations based on mapped barriers. Inclusion of predicted barriers revealed an additional 33 km of potentially inaccessible stream habitat and the modeled stream network located approximately 1700 km of stream length that has been completely lost. Comparing habitat accessibility and barrier density against the assessed status of populations revealed insights useful for understanding the impact of barriers on spawning and rearing and guiding the allocation of restoration effort. Applying methods for addressing missing data, such as lost streams and unmapped barriers, was essential for estimating the accessibility of habitat within a historical context. While much emphasis has been placed on the role of marine conditions in wild Pacific salmon recovery, the magnitude of habitat loss in the Fraser cannot be ignored and suggests it is a major driver of observed salmon declines.

Key words: anadromous species; barriers; connectivity; conservation planning; culverts; dams; flood control; fragmentation; habitat loss.

Received 6 July 2020; revised 21 January 2021; accepted 11 March 2021; final version received 10 May 2021. Corresponding Editor: Matthew J. Troia.

† **E-mail:** riley.finn@ubc.ca

Introduction

Loss of connectivity of freshwater habitat is widespread and recognized as one of the key threats to aquatic systems (Dudgeon et al. 2006, Fuller et al. 2015). Barriers that create connectivity issues are often anthropogenic structures that restrict the longitudinal movement of freshwater species throughout a stream network or lateral movement between stream and adjacent floodplain habitats (Coleman et al. 2018). While barriers in freshwater systems almost always have impacts on the quality of habitat itself, they can completely alienate stretches of otherwise usable habitat for anadromous species (Zhong and Power 1996, Gardner et al. 2012), which require connectivity between freshwater and marine environments. The impacts of habitat loss are broad and include both demographic and ecological impacts, from increased extinction risk (Seabloom et al. 2002), and decreased species richness (Helm et al. 2006), to altered evolutionary trajectories and resilience (McClure et al. 2008).

Pacific Salmon (*Oncorhynchus* spp.), like many anadromous species, are impacted by the alienation of habitat from barriers, with numerous studies demonstrating the impact of anthropogenic barriers (Gibson et al. 2005, Sheer and Steel 2006, van Puijenbroek et al. 2019). These barriers include dams, flood control structures, road culverts, and other structures. In addition to an increase in the number of barriers, the last century has seen other anthropogenic pressures including land use change (Bilby and Mollot 2008), climate change (McDaniels et al. 2010, Beechie et al. 2013), disease (Mordecai et al. 2019), and over-exploitation (Price et al. 2019) resulting in freshwater habitat degradation and declining marine conditions, which impact salmon at every stage of their lifecycle (Grant et al. 2019). As a result, many salmon populations are at record low numbers compared to historical levels (Peterman and Dorner 2012, Malick and Cox 2016, Grant et al. 2019, Price et al. 2019). Recently, much of the emphasis for regional declines has been focused on the change in productivity of the marine environment (Beamish et al. 2010, Peterman and Dorner 2012, Malick and Cox 2016). However, the relative resilience of pink (*Oncorhynchus gorbuscha*) and chum (*Oncorhynchus keta*) salmon has been noted as a possible indicator that the freshwater environment may be playing a bigger role in declines than previously thought, as these species are less dependent on freshwater habitats (Grant et al. 2019). Sockeye (*Oncorhynchus nerka*), Chinook (*Oncorhynchus tshawytscha*), and coho (*Oncorhynchus kisutch*) are all faring comparatively worse in the southern portions of their range where both increases in water temperature and the degree of anthropogenic disturbance on freshwater habitat have been most severe (Grant et al. 2019).

In addressing these threats to salmon, the removal of barriers and restoring access to freshwater habitat is among the most successful restoration strategies that can be undertaken due to its comparatively nominal cost, quick biological response, long-lasting effect, and relatively high likelihood of success (Roni et al. 2002). Understanding the extent of habitat connectivity loss and how it impacts different populations depending on their position in the landscape is important for informing management priorities and setting restoration baselines.

The establishment of appropriate baselines requires the consideration of the historical context of the habitat and is important for avoiding changing human perceptions of biological systems, also known as shifting baselines, which can lead to a managed decline of the ecosystem (Pauly 1995, Papworth et al. 2009). To do this, estimation of habitat that still exists, but has become inaccessible to salmonids, needs to be assessed in tandem with habitat that has been completely lost from the landscape. Particularly in urban locations, streams can be replaced with sub-surface infrastructure with no value as habitat (Napieralski and Carvalhaes 2016), which may contribute to significant habitat loss for species and populations that used them historically.

The Fraser River drains nearly a quarter of the province of British Columbia, Canada, and is one of the great salmon-producing rivers of the world (Northcote and Larkin 1989), yet it has pervasive anthropogenic impacts and many Fraser salmon runs are now at historical lows (Grant et al. 2019). The Lower Fraser River and its associated tributaries have a disproportionate importance for wild salmon as a migratory corridor and for the number of populations that rely on the area (Nesbitt and Moore 2016). Over the last century, resource extraction, urbanization, and land conversion to agriculture have eliminated, severely degraded, and alienated much of the freshwater stream systems in the area (Boyle 1997). Further, the tidal characteristics and low-lying nature of the Fraser Valley have precipitated the construction of floodgates along many of the tributaries (Thomson and Confluence Environmental Consulting 1999). The operation of floodgates in the Lower Fraser has been shown to create hotspots for invasive species, reduce native fish diversity, and cause hypoxic conditions up to 100 m upstream (Gordon et al. 2015, Scott et al. 2016, Seifert and Moore 2018). While improved management of the operation of these floodgates may mitigate some of these negative impacts (Seifert and Moore 2018), even when completely open, floodgates alter flow velocities and can become perched or otherwise inaccessible (Haro et al. 2004). In addition to floodgates, the ever-expanding road network has created additional barriers, with over 170,000 closed bottom culverts impeding fish passage across British Columbia (Fish Passage Technical Working Group 2014).

The purpose of this study was to quantify the amount of stream and floodplain salmon habitat that has been alienated by anthropogenic barriers in the Lower Fraser River. Specifically, the motivating questions were as follows: (1) What was the historical extent of salmon habitat in the Lower Fraser River? (2) How much of the historical habitat has been lost entirely? (3) How much of the historical habitat is now inaccessible as a result of anthropogenic barriers? (4) How does habitat accessibility and the presence of barriers compare against the assessed status of Pacific Salmon conservation units (CUs) that rely on the Lower Fraser River for breeding and rearing? By assessing salmon habitat availability within a historical context, a better understanding of the baseline conditions can be used to help guide restoration of habitat for these culturally, economically, and ecologically important species.

Methods

Study area

The Fraser River is the largest river in British Columbia (BC) with a total watershed area of 233,000 km^2, and it has an average annual discharge of about 3700 m^3/s and a bimodal hydrograph characterized by spring snow melt run-off and increased autumn precipitation (Northcote and Larkin 1989). The Lower Fraser region is generally delineated as the portion of river between the community of Hope, BC, where it begins flowing in a predominately western direction, and the Pacific Ocean. In this study, it has been delineated hydrographically using the watershed groups defined in the provincial Freshwater Atlas that contain at least a portion of this stretch of river. These watershed groups include Fraser Canyon, Chilliwack, Harrison, Lillooet, and Lower Fraser and drain an area of 20,203 km^2 (Fig. 1). The Fraser River as a whole is Canada's largest producer of wild salmon, yet despite representing a relatively small portion of the entire basin, the Lower Fraser has disproportionate importance for salmonids. The region supports a diversity of populations (Nesbitt and Moore 2016), as well as acting as a migration corridor for all other populations in the Fraser Basin.

Pacific Salmon in Canada are federally managed under the Wild Salmon Policy at the level of the conservation unit (CU). The CU is defined as a group of wild salmon that is sufficiently isolated from other groups that if it were to go extinct it would not recolonize within an acceptable time frame such as a human lifetime or specified number of salmon generations (Fisheries and Oceans Canada 2005). Spatial boundary polygons that delineate the freshwater habitats of all CUs were overlaid with the watershed group polygons to identify a total of 14 CUs whose habitats fell completely within the study area (Table 1; Appendix S1). Lake-type sockeye were excluded from this study as their CUs are delineated only by the lakes in which they rear, leaving no stream length for the assessment of accessibility—if the lake were not accessible, then the CU would not exist. For the purpose of status assessments, CUs act as the designatable units for the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). Information on the status of each CU as assessed by COSEWIC was collected from the species registry for comparison with habitat accessibility. Among the primary roles of COSEWIC are the identification of species for assessment and to carry out those assessments determining whether species are classified as extinct, extirpated, endangered,

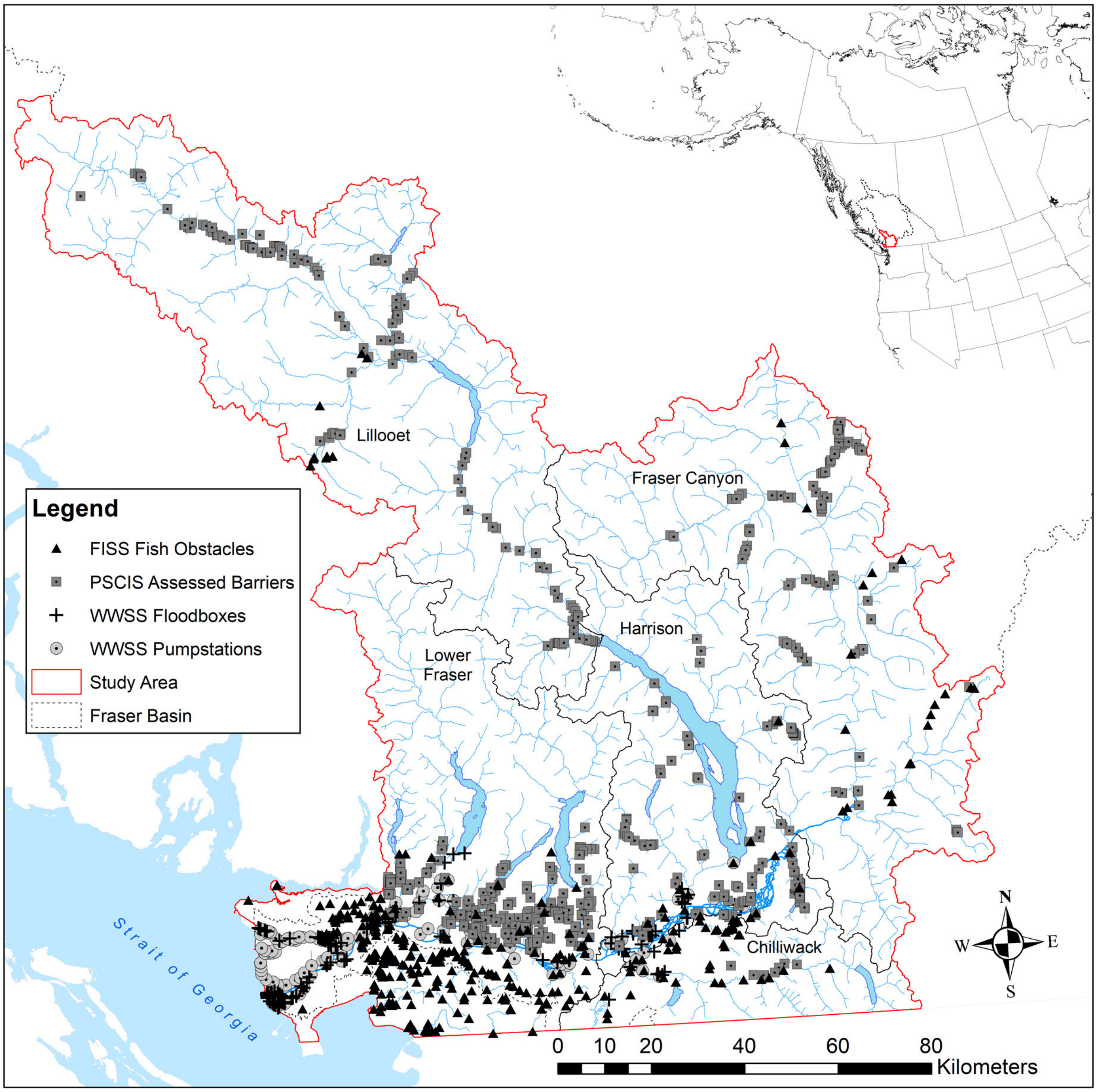

Fig. 1. Map of study area comprised of the Lillooet, Fraser Canyon, Chilliwack, Harrison, and Lower Fraser watershed groups showing location of mapped barriers. Sources of barrier data include the Fish Information Summary System (FISS), Provincial Stream Crossing Inventory System (PSCIS), and Watershed Watch Salmon Society (WWSS). Inset shows context of study in the broader Pacific Northwest Region.

threatened, of special concern, or not at risk (SARA 2002).

Quantifying extent of historical floodplain

Floodplain habitat downstream of each CU boundary was assumed to have been utilized or available to out-migrating juveniles or as part of their rearing habitat (Brown and Hartman 1988, Sommer et al. 2001). This was identified using a map of the historical vegetation of the Lower Fraser Floodplain created through the translation of surveyor's notebooks into spatial information on vegetation communities that existed between the years 1859 and 1890 (North and Teversham

Table 1. Conservation units (CUs) of salmon within the Lower Fraser River included in this study and the assessed status of the CU according to the Committee on the Status of Endangered Wildlife in Canada (COSEWIC).

Species	Conservation unit	COSEWIC status
Chinook (*Oncorhynchus tshawytscha*)	Fraser Canyon Spring 1.3	Endangered
	Lower Fraser River–Upper Pitt Summer 1.3	Endangered
	Lower Fraser River Fall 0.3	Threatened
	Lower Fraser River Summer 1.3	Threatened
	Lower Fraser River Spring 1.3	Special concern
	Boundary Bay Fall 0.3	Not assessed
	Maria Slough Summer 0.3	Not assessed
Chum (*Oncorhynchus keta*)	Lower Fraser	Not assessed
Coho (*Oncorhynchus kisutch*)	Boundary Bay	Not assessed
	Fraser Canyon	Not assessed
	Lillooet	Not assessed
	Lower Fraser	Not assessed
Sockeye (*Oncorhynchus nerka*)	Widgeon (river-type)	Threatened
	Harrison River (river-type)	Not at risk

Notes: Numbers in the Chinook CU names indicate the mean number of years spent in freshwater (before decimal) and marine environment (after decimal). Half of the CU's have not been assessed.

1984). From this dataset, 26 distinct vegetation communities were identified, and those with flood tolerant species compositions or were likely to have high hydrological connectivity to the Fraser mainstem or coast were assumed to be floodplain fish habitat with at least seasonal regularity. This assessment was informed by Kistritz et al. (1996), and habitats were generally characterized as marshes, grasslands, cottonwood, spruce, or other coniferous forest. A table of all communities outlined by North and Teversham (1984) and those selected as floodplain fish habitat is provided in supplemental material (Appendix S2). A large gap exists in the floodplain assessment due to the historical presence of Sumas Lake in the eastern portion of the Fraser Valley. Sumas Lake was drained in 1924 and is currently kept dry through a series of canals and pumps for farming in the eastern portion of the Lower Fraser Valley (Murton 2008). For this study, the lake was included as floodplain habitat as there are historical accounts of wide variation in lake levels throughout the year, ranging from a low of nine feet deep in winter to 36 feet deep during the spring freshet (Murton 2008). Due to these variations, it is likely that a precise measurement of the lake's area would not capture the dynamic nature of the system. The original mapping of the Lower Fraser vegetation communities done by North and Teversham (1984) omitted the lake. For this study, a polygon was created to fill this gap and act as an estimate for the size of the lake-floodplain habitat.

Accessibility to floodplain fish habitat was assessed using currently mapped dikes. Diking data were downloaded from the provincial data repository (Table 2, https://catalogue.data.gov.bc.ca/dataset?download_audience=Public). Floodplain isolated by diking infrastructure from either the coast or the Fraser River mainstem was marked as inaccessible. The area of lost and accessible floodplain was then calculated for each CU by selecting all floodplain polygons both within and downstream of each CU polygon.

Identifying naturally inaccessible stream habitat

To identify naturally inaccessible linear stream habitat, natural barriers were delineated and the accessible lengths of streams were measured by breaking the Freshwater Atlas stream network into 300 m segments and assessing their average gradient using a 0.75 arc-second digital elevation model (DEM) in ArcGIS (ESRI 2018). A reach length of 300 m was used as this corresponds to the maximum reach length used in a similar study looking at watershed accessibility by Sheer and Steel (2006). The gradient thresholds for each species were also determined following the methods of Sheer and Steel (2006), where a threshold of 16% was used for Chinook, coho, and sockeye, while a gradient of 5% was used for chum. These gradient thresholds are derived from recommendations by Washington State Fish and Wildlife Department (Washington Department of Fish and Wildlife 2019). The gradient barriers were combined with other known

Table 2. Description and source of all datasets used in the spatial analysis of salmonid habitat accessibility on the Lower Fraser River.

Layer name	Description	Source	Date downloaded
Freshwater Atlas Stream Network	Stream hydrography mapped at 1:20,000 scale	Province of BC	Jan. 10, 2019
Freshwater Atlas Lake Polygons	Lakes mapped at 1:20,000 scale	Province of BC	Feb. 1, 2019
Pacific Salmon Conservation Unit Boundaries	Polygons of the freshwater habitat range for each CU	Government of Canada	Mar. 20, 2019
BC Digital Roads Atlas	Roads in BC including resource roads	Province of BC	Jul. 29, 2019
Linear Diking Infrastructure	Linear flood infrastructure collated from multiple sources	Province of BC	Jun. 15, 2018
Provincial Stream Crossing Inventory System—Assessments	Assessments of culverts that follow the provincial protocol for Fish barrier assessment	Province of BC	Sep. 26, 2019
Fish Information Summary System—Fish Obstacles	Collation of various reported potential obstacles fish passage	Province of BC	Apr. 14, 2019
Floodgates and Pump stations	Point locations of floodgates and pumps that may represent barriers to fish movement	Watershed Watch Salmon Society	–
Digital Elevation Model	DEM at 0.75 arc-second resolution	Government of Canada	Jul. 20, 2019
Historical Vegetation of the Lower Fraser Floodplain	Vegetation communities of the Lower Fraser Floodplain estimated from surveyors' notebooks between 1859–1890	North and Teversham (1984)	–

natural barriers not identified by the slope analysis, most notably Stave Canyon, which blocks access to Stave Lake, and the upper Stave system, which acted as a historical fish barrier but is now a hydro dam (Stockner and Bos 2002). For the purposes of this study, all stream length below these natural barriers was assumed to be salmonid stream habitat and was measured linearly along the stream length.

Streams that have been classified as ditches in the Freshwater Atlas have been straightened or simplified into drainage channels. For the purposes of assessing habitat accessibility, these channelized streams often form loops in the network and were unreliably identified as accessible or inaccessible using the network topology. It is also likely that streams which have been channelized for drainage harbor only degraded habitat (Rosenvald et al. 2014), and for these reasons, the length of channelized streams was quantified separately to the naturally accessible habitat.

Mapped anthropogenic barriers to fish passage

To identify inaccessible habitat, information on barriers in the Lower Fraser was collated from three sources (Fig. 1; Table 2). Two of the sources were provided through the province of BC: the Provincial Stream Crossing Inventory System (PSCIS), which follows a standardized fish passage assessment (Fish Passage Technical Working Group 2014), and the Fish Information Summary System (FISS), which is a collation of potential obstacles to fish passage that have been observed to potentially cause fish habitat fragmentation but have not necessarily been assessed for their impact on connectivity. The third source is comprised of barriers specifically related to flood infrastructure along the Fraser River and was collected by a local non-governmental organization, Watershed Watch Salmon Society. Due to the potential unreliability of the FISS database, records from all three datasets were evaluated for redundancy and, where overlap of the FISS data was observed with either of the other two datasets, the FISS data were removed. All anthropogenic barriers were assumed to be a complete barrier to salmonids.

The combined barrier data were spatially joined and linear referenced to the Freshwater Atlas stream network using a 30 m snapping distance to account for spatial error in barrier coordinates. When a barrier was outside of this snapping distance, but assessment information identified the stream that the barrier was supposed to be located on, the point location was moved to the appropriate stream so as to be included in the analysis. Using a set of queries, the first barriers that cut off access upstream from either the mainstem of the Fraser River, the coast, or the southern border were identified. All streams upstream of these first barriers were assumed to be inaccessible, while streams downstream of these barriers were assumed to be accessible. Proportions of accessible and inaccessible habitat were calculated for the region as a whole using the estimated naturally accessible length and for each of the 14 CUs using the CU boundary as the population extent.

Identifying lost streams

The extent of streams that have been lost to urbanization or development was estimated by creating a digital elevation model (DEM)-derived stream network from the Canadian digital elevation model at 0.75 arc-second resolution and comparing it with the Freshwater Atlas stream hydrography. First, the DEM was filled, and the Freshwater Atlas stream hydrography was burned in using a rasterized version of the mapped hydrography. The burn-in ensured the DEM-derived network matched the Freshwater Atlas hydrography and to identify only large areas where streams are expected but have not been mapped within the Freshwater Atlas. Both the Freshwater Atlas stream network (1:20,000) and the DEM were at similar scales, which minimizes the negative impacts of burn-in on the final DEM stream network (Lindsay 2016). The burned in DEM was then converted to a flow direction raster and finally a flow accumulation raster using the D8 method (O'Callaghan and Mark 1984). To identify an appropriate threshold for channel initiation on the flow accumulation raster, points were created at the initiation of currently mapped streams not identified as ditches in the Fraser Valley below 100 m of elevation. The average flow accumulation at these points was approximately 0.25 km^2, which was used as the threshold value for channel initiation of the DEM-derived stream network. This estimate is a uniform critical support area for streams on the valley bottom, so it does not consider the variation in potential erosional forces such as slope and soil types (Montgomery and Foufoula-Georgiou 1993). Using streams initiated under 100 m of elevation is likely a conservative estimate for channel initiation and, subsequently, channel length. Streams within 50 m of currently mapped Freshwater Atlas streams were assumed to be the same as those already mapped and removed from the DEM-derived stream network, leaving only streams potentially lost due to filling and urbanization. The remaining DEM-derived streams were measured for their along-stream length, average slopes, and elevations. Natural barriers were delineated following the methodology already described, and lost streams were labeled as naturally inaccessible where appropriate. The polygon representing Sumas Lake derived from North and Teversham (1984) was used to remove any lost streams estimated in this area.

Predicting unmapped barriers

Whereas mapping of barriers has occurred in the Lower Fraser River, these maps are incomplete, particularly for road culverts. In order to account for unmapped potential barriers, all stream–road intersections were mapped, and a model was developed to predict whether these intersections represented potential barriers to fish passage in R (R Core Team 2019). The Freshwater Atlas was overlaid with the BC Roads Atlas Layer, and a point was created at each intersection except where the road was labeled as a bridge, ferry, or overpass or where streams were estimated to be naturally inaccessible to anadromous salmonids. Boosted regression trees (BRTs) from the gbm package (Greenwell et al. 2019) were used to develop a model that can predict the probability that each road crossing poses a barrier to fish passage. Boosted regression trees differ from traditional regression methods as they adaptively combine a large number of relatively simple models to optimize predictive performance, rather than producing a singular best model (Elith et al. 2008). Following the methodology of Januchowski-Hartley et al. (2014), the site, reach, and segment slope of the stream, as well as the catchment area of the intersection, were used as predictors. In addition to these stream characteristics, the number of lanes of the road and the surface type of the road were also used as predictors (Table 3). Fish passage assessments in the PSCIS that were labeled as either a barrier (n = 221) or passable (n = 346) were used to develop the predictive model. Code from Elith et al. (2008) was used to determine the optimal learning rate, tree complexity, and number of trees. Due to the smaller sample size, only tree complexities of 2 and 3 were tested across learning rates of 0.01, 0.005, and 0.001. The learning

Table 3. Predictors used to estimate the probability a given stream–road intersection would be assessed as a barrier to fish passage.

Predictor	Description	Relevance
Stream Segment Slope	Slope of confluence bound stream line (%)	Hydrological regime
Stream Reach Slope	Slope of 300 m stream reach (%)	Reach scale stream energy
Stream Site Slope	Slope of stream at site of barrier assessment (%)	Site scale topography
Catchment Area	Area contributing flow to the site of the potential barrier (m^2)	Stream size
Number of Lanes	Number of lanes on the road indicated by the BC roads Atlas	The number of lanes indicates the length of the culvert
Road Surface Type	The surface of the road as indicated in the BC roads Atlas categories include paved, loose, and rough	Difference in jurisdiction and landscape context between resource and municipal roads

rate determines the contribution of each tree to the growing model, while the tree complexity controls the fit of interactions or number of branches (Elith et al. 2008). The bag fraction was kept at 0.5, while multiple learning rates and tree complexities were tested. The bag fraction controls the level of stochasticity, as each iteration of the model uses the specified proportion of randomly selected data as the training data for the model. Final model parameters were selected based on the model with smallest deviance and at least 1000 trees using the gbm.step function (Elith et al. 2008). Model performance, as measured by the area under the receiver operator characteristic (AUC) curve, was estimated using 10-fold cross-validation along with standard errors. Variable importance was calculated based on the contribution to model fit attributable to a given predictor averaged across all trees, in order to understand the characteristics of stream–road intersections that are more likely to be barriers to fish passage.

The final model was then used to predict the probability of whether each road–stream intersection was a barrier. The optimal threshold for categorization of a barrier was determined using the optimal.thresholds function which identifies the threshold that correctly classifies the most sites in the PresenceAbsence R package (Freeman and Moisen 2008). These model predicted barriers were combined with mapped barriers and where overlaps were identified, predicted barriers were removed. This created a set of road–stream intersections that were not mapped as barriers but were estimated to have a high probability of being a barrier, combined with the mapped barriers. The assessment of accessibility was re-run with this combined barrier dataset to see how model predicted barriers influenced the amount of accessible habitat.

Results

Floodplain habitat

An estimated 659 km^2 of floodplain fish habitat existed historically in the Lower Fraser. Of this, approximately 102 km^2 remains accessible according to currently mapped dikes, representing only 15% of the historical habitat (Fig. 2). The amount of accessible historical floodplain habitat varies widely by CU depending on its context in the landscape and what lies downstream. An estimated 4–5% of the historical floodplain habitat for the Boundary Bay populations of both coho and Chinook remain accessible, whereas the Fraser Canyon Chinook and coho and the Lower Fraser coho are comparatively better off with an estimated 16–17% of their floodplain habitat remaining accessible (Table 4). A few remaining regions with accessible floodplain appear to be in the islands of the eastern portion of the Fraser River mainstem and in the Pitt River System.

Stream habitat

In total, 1264 barriers to fish passage have been previously mapped in the study area. Of these, 985 were within 30 m of a mapped stream and were included in the analysis of accessibility. These barriers are responsible for alienating approximately 2224 km of stream length, and 1727 km of stream length is estimated to be completely lost from the landscape, representing 64% of the estimated 6118 km of naturally accessible salmonid stream habitat in the Lower Fraser Region. Additionally, 516 km of stream length has been channelized, concentrated primarily in the developed valley bottom (Fig. 3). Marked differences in accessibility of stream habitat were observed among CUs (Table 4). As with

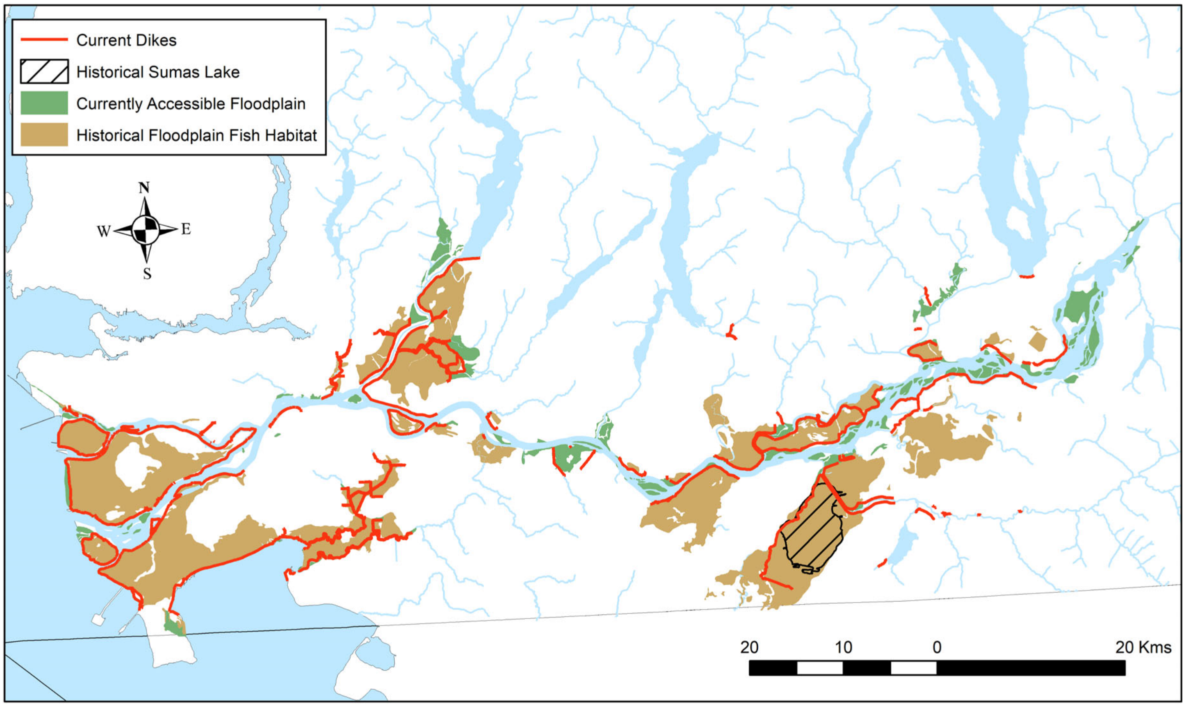

Fig. 2. Map showing the estimated historical extent of floodplain fish habitat (brown), current diking infrastructure and estimated accessible flood plain fish habitat (green). Historical habitat was estimated using North and Teversham (1984) historical vegetation maps. Area with diagonal lines represents historical extent of Sumas Lake that is no longer in existence, derived from North and Teversham (1984) data.

floodplain habitat, the Boundary Bay populations of Chinook and coho had the highest amount of alienated habitat, with mapped barriers, lost and channelized streams preventing fish access to approximately 70% of the naturally accessible stream length in those CUs. However, there are also some CUs with much smaller alienated habitat than the region as a whole (Table 4). For instance, mapped barriers did not appear to have much impact on the amount of habitat that is inaccessible to the Fraser Canyon CUs. According to currently mapped barriers, the more remote Fraser Canyon Chinook and coho, and the Lillooet coho have 87–99% of their linear stream habitat remaining accessible.

Model estimated barriers and updated accessibility estimates

The final model used 2550 trees, with a learning rate of 0.001 and a tree complexity of 2, and an estimated AUC of 0.742 (SE $\pm$ 0.028) based on 10-fold cross-validation. The most important predictors for whether a given stream–road intersection was a barrier were catchment area and segment slope (Appendix S4). The two road attributes, the number of lanes and the surface type, were the least important; however, all predictors had enough influence to remain in the model.

There were 819 stream–road intersections mapped, and the final model was used to estimate the probability that every stream–road intersection was a barrier. The optimal threshold used to identify potential unmapped barriers was determined to be 55%, and this value was chosen as it minimizes the distance between the receiver operator characteristic (ROC) curve and the upper left corner of the unit square (Freeman and Moisen 2008). If a stream–road intersection had a probability higher than 55% of being a barrier, it was added to the existing mapped barrier dataset, resulting in an additional 286 unmapped potential barriers in the study area. After rerunning the accessibility assessment with the

Table 4. Stream length and floodplain area accessibility estimates for 14 conservation units (CUs) of salmon in the Lower Fraser River.

	Stream									Floodplain			
	Mapped barriers			Mapped + predicted barriers									
CU	Access (km)	Inacc (km)	Access (%)	Access (km)	Inacc (km)	Access (%)	Chan (km)	Lost (km)	Total (km)	Access (km^2)	Inacc (km^2)	Total (km^2)	Access (%)
Chinook													
Boundary Bay	184.3	176.7	28	184.3	176.7	28	116.3	182.4	659.7	4.3	87.6	91.9	5
Fraser Canyon	167.4	1.1	99	164.3	4.2	97	0.0	0.0	168.5	73.4	396.6	470.0	16
Lower Fraser													
Fall	70.0	34.1	52	69.6	34.6	51	0.8	30.3	135.2	53.2	350.6	403.8	13
Spring	207.6	78.0	73	201.8	83.8	71	0.0	0.0	285.7	39.6	336.8	376.4	11
Summer	595.3	99.2	82	588.5	106.0	81	0.8	32.1	727.4	39.6	336.8	376.4	11
Upper Pitt	214.9	15.6	73	213.0	17.6	73	32.0	30.8	293.4	28.0	207.5	235.5	12
Maria Slough	21.2	21.3	38	21.0	21.5	38	0.0	12.7	55.2	71.4	396.0	467.4	15
Chum													
Lower Fraser	921.0	446.5	39	918.9	448.6	39	282.0	712.4	2361.9	97.2	469.8	567.0	17
Coho													
Boundary Bay	252.8	241.3	26	252.8	241.3	26	156.4	310.0	960.4	4.4	102.5	106.9	4
Fraser Canyon	503.2	4.7	92	494.2	13.6	90	1.1	38.9	547.8	73.4	396.6	470.0	16
Lower Fraser	1241.8	1315.2	32	1230.0	1327.0	32	382.1	904.9	3844.1	54.9	387.8	442.7	12
Lillooet	653.3	100.8	87	641.7	112.5	85	0.0	0.0	754.1	97.2	469.8	567.0	17
Sockeye													
Harrison River	30.0	1.9	70	30.0	1.9	70	0.8	10.3	43.0	53.2	350.6	403.8	13
Widgeon Creek	17.1	0.4	96	16.8	0.8	94	0.0	0.4	17.9	28	207.5	235.5	12

Note: Abbreviations are Access, accessible; Chan, channelized; Inacc, inaccessible; and % Access, percentage accessible.

combined mapped and predicted barriers, the amount of additional alienated habitat was generally small and varied by CU. Largely, those CUs with the greatest amount of remaining habitat were most impacted by predicted barriers. In particular, the estimated accessible habitat for the Fraser Canyon Chinook and coho CUs decreased by approximately 2% (Fig. 4) after accounting for modeled predicted barriers. In total, the predicted barriers highlighted up to 33 km of stream length that has a high probability of being inaccessible, but where no barriers have been mapped.

When predicted and mapped barriers were combined, the density of barriers per km of stream habitat increased slightly for most CUs except for Harrison River sockeye and the two Boundary Bay CU's (Fig. 5). When estimated proportion of accessible habitat was plotted against the mapped barrier density, the relationship was slightly negative and became somewhat more pronounced when predicted barrier density was also included (Fig. 6). After predicted barriers are considered, the only CU that was assessed as not at risk (Harrison River sockeye) had the lowest barrier density and access to most of its habitat (Fig. 6).

While most CUs are impacted by the loss of floodplain, some seem to be particularly worse off when it comes to the remaining accessible stream habitat. These CUs include the Chinook and coho of Boundary Bay and Lower Fraser Coho. Fig. 7 shows the proportions of both floodplain and stream habitat that remain

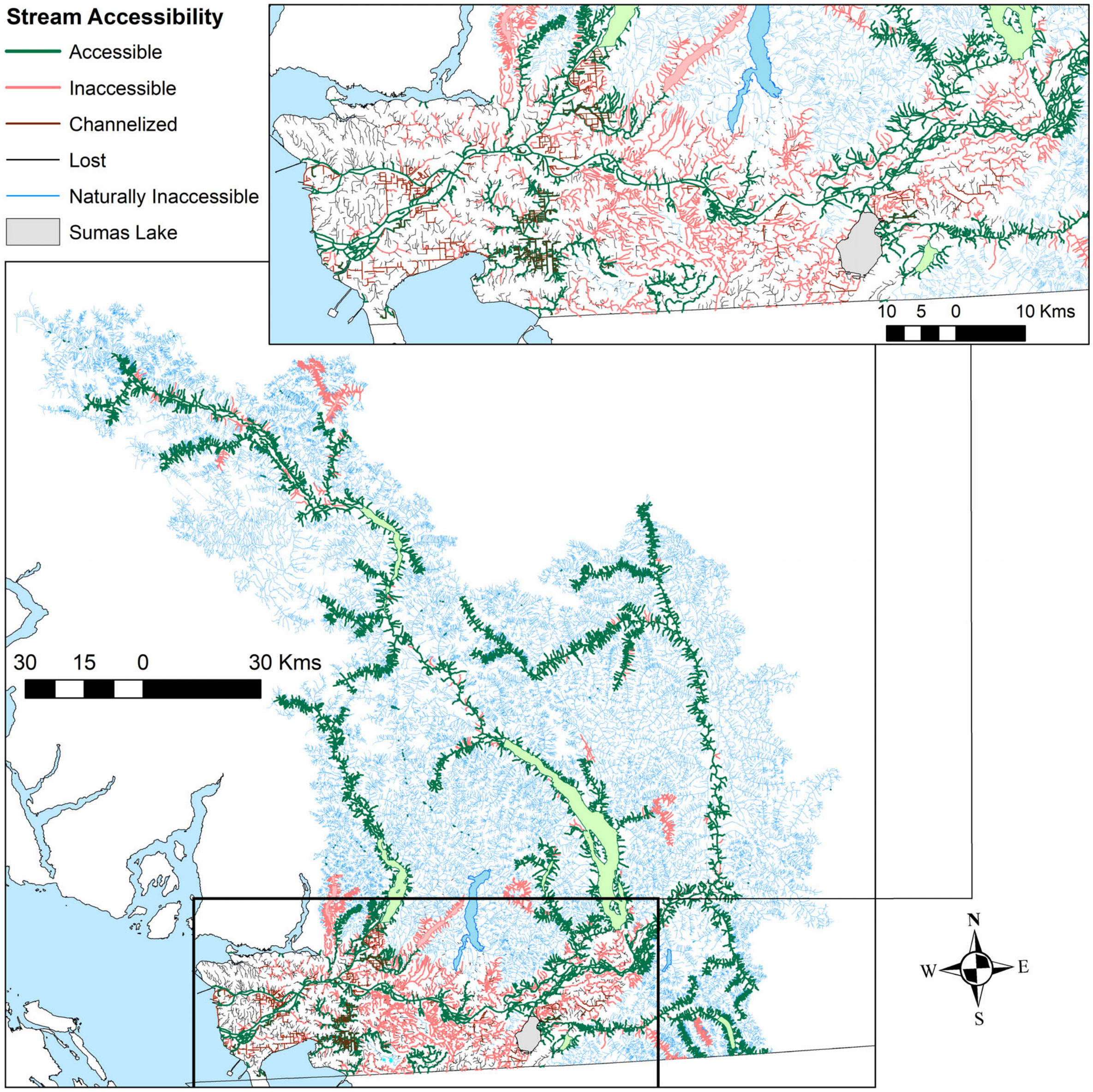

Fig. 3. Map of stream connectivity in the Lower Fraser according to currently mapped barriers to fish passage. Also included are lost streams identified from a digital elevation model-derived stream network, and a map of historic Sumas Lake derived from North and Teversham (1984). This map depicts the 16% gradient threshold for natural accessibility, and for the 5% threshold used to assess chum salmon habitat accessibility, see Appendix S3.

accessible and inaccessible as well as the proportions of streams that have been lost and converted to drainage channels. Many of the most highly impacted CUs remain unassessed by COSEWIC and are impacted by not only the alienation of habitat, but also the loss of streams from the landscape.

Discussion

The Lower Fraser River is an important location for salmon spawning, rearing, and migration. It is also a location with intense development and environmental degradation; restoring salmon habitat in this location requires

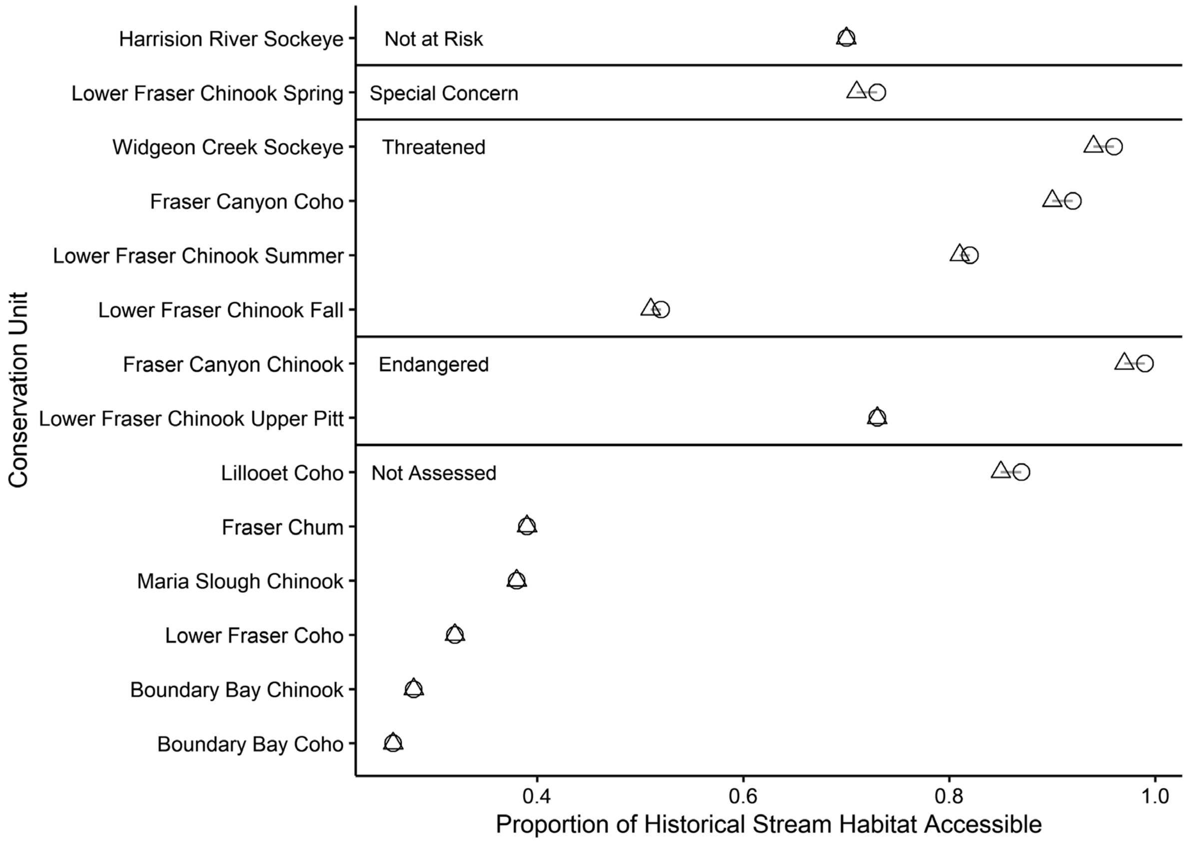

Fig. 4. Proportions of stream length accessible by conservation unit of salmon in the Lower Fraser Region ordered by the Committee on the Status of Endangered Wildlife in Canada assessed status. Circles indicate estimates based only on mapped barriers, while triangles are accessibility estimates with both mapped and model estimated barriers.

an understanding of the historical habitat of the area. In this study, we collated a dataset of over 1200 instream barriers and mapped lost streams which indicate the alienation of approximately 64% of otherwise accessible stream length in the region and use historical vegetation maps to demonstrate the disconnection of 85% of historically accessible floodplain by diking infrastructure. Importantly, we try to address key pieces of missing data throughout the region and include these in our assessments. By estimating lost streams using a DEM-derived stream network, we revealed 1727 km of habitat that may have been accessible historically, but no longer exist. Model predicted barriers highlighted an additional 33 km of habitat with a high probability of being inaccessible.

Pervasive loss of habitat

According to historical floodplain vegetation communities and current diking infrastructure, approximately 85% of the historical floodplain fish habitat has been alienated in the Lower Fraser River, and this number ranges from 96% to 83% depending on the salmon CU. A similar analysis that looked at all wetland fish habitat in the same region (Kistritz et al. 1996) found that 90% of wetlands that could have been used as fish habitat in the Lower Fraser were lost. Kistritz et al. (1996) used the same historical vegetation dataset used here, in combination with historical aerial photographs to identify wetlands outside of the floodplain and by comparing this to the Canadian Wildlife Service wetlands inventory to determine differences between the contemporary

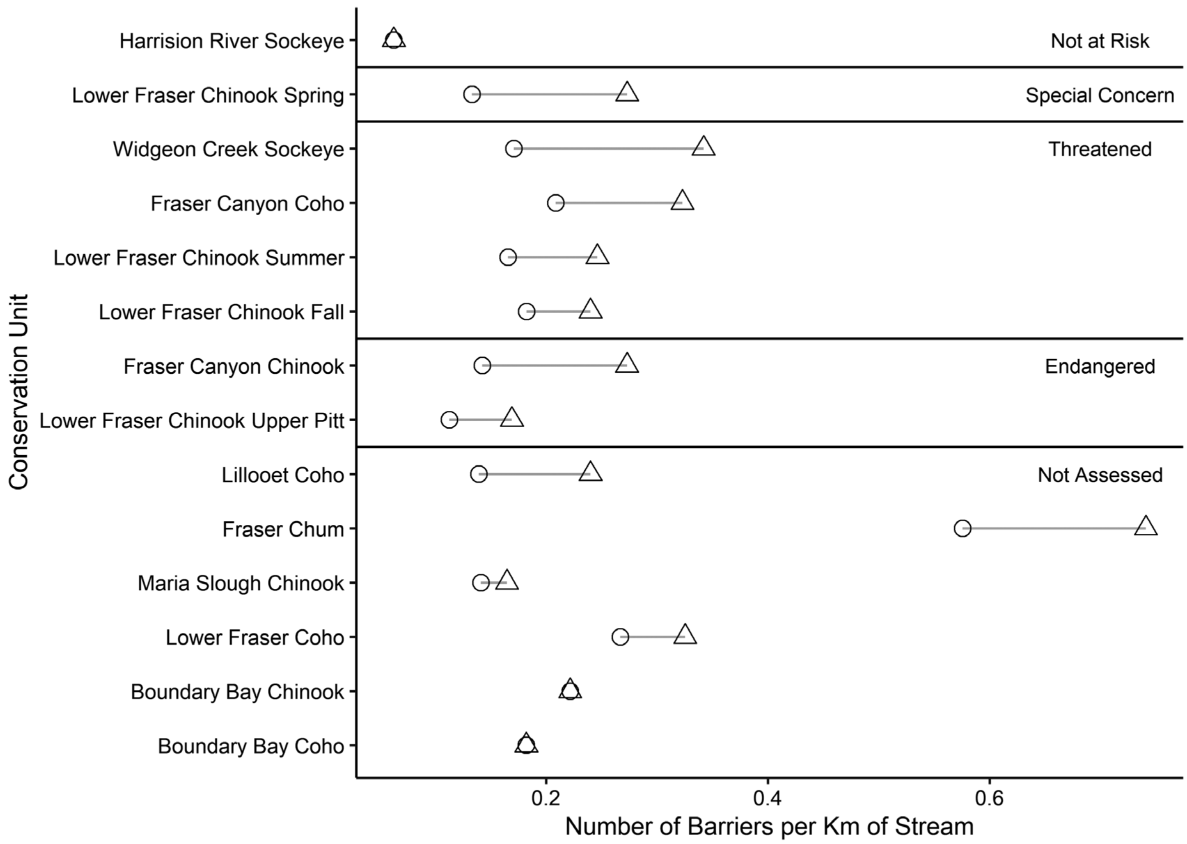

Fig. 5. Number of barriers per kilometer of stream habitat by conservation units of salmon in the Lower Fraser Region ordered by the Committee on the Status of Endangered Wildlife in Canada assessed status. Circles indicate mapped barrier densities; triangles indicate combined mapped and predicted barrier densities.

and historical wetlands. However, they did not have diking information and were unable to consider floodplain forests and Sumas lake within their estimates. The loss of floodplain habitat may have important implications for salmon productivity in the Lower Fraser system, particularly for coho and Chinook, which rely on these habitats for rearing and gain an advantage in the form of increased growth rates relative to other areas (Brown and Hartman 1988, Sommer et al. 2001, 2005). In California, Chinook salmon benefit from the use of seasonally flooded farmlands (Katz et al. 2017) and ephemeral floodplain habitats (Jeffres et al. 2008) in the form of increased growth rates and improved body condition. Given the importance of this habitat, the loss of access to 85% of historical floodplain habitat in the region likely has large impacts on the carrying capacity and condition for many of the Fraser Chinook and coho populations.

Our prediction of lost streams revealed an estimated 1727 km of linear stream habitat that may have been completely eliminated from the landscape. Descriptions from Indigenous Communities of the Lower Fraser indicate that these streams likely functioned as salmon habitat. For example, the ubiquity of salmon is captured by stories of the *Stó:lō*, who possess 147 words for the catching and processing of fish, fished salmon for most of the year, and describe salmon as the essence of *Stó:lō* identity and life (Smith 2001). Additionally, oral histories collected by Proctor (1978) from Vancouver pioneers describe nearly all of the streams found within Vancouver's boundary as having spawning habitat for salmon with many swamps and wetlands at the

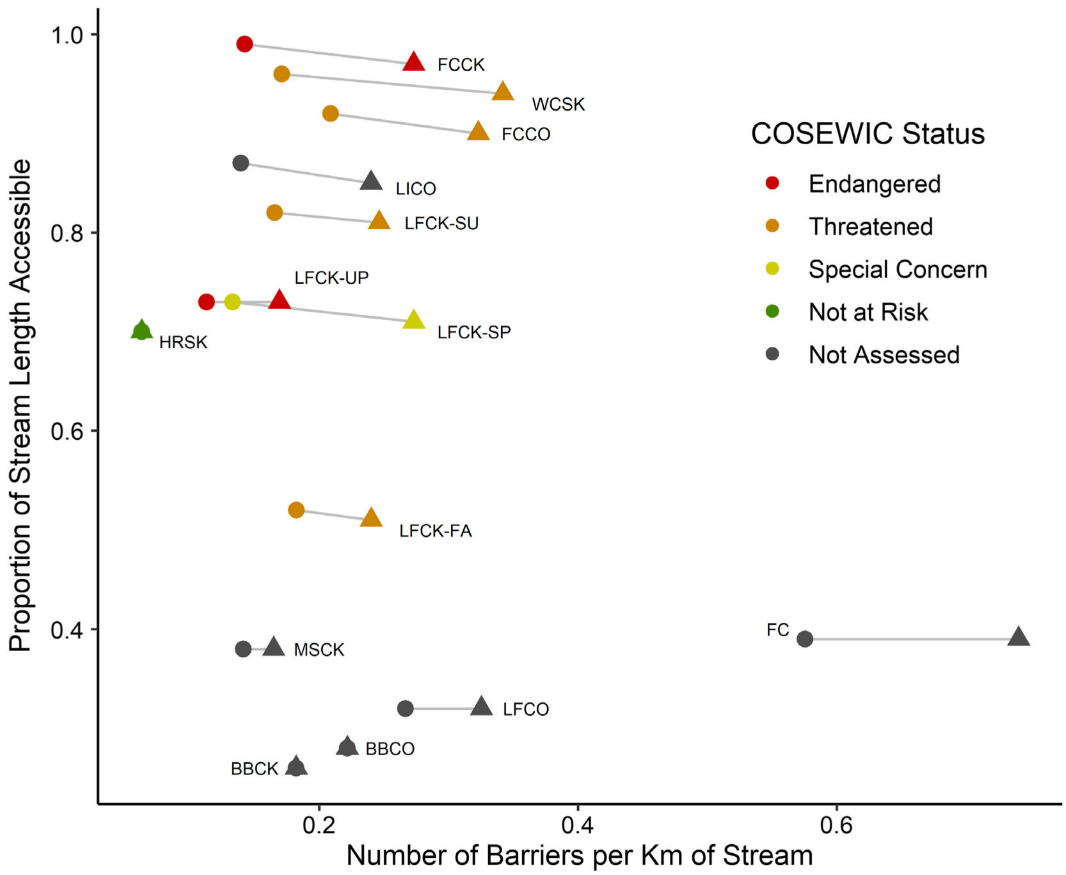

Fig. 6. Number of barriers for mapped (circles) and mapped and predicted (triangles) and the corresponding proportion of stream habitat that is accessible for 14 conservation units of salmon in the Lower Fraser River. Colors indicate the assessed status by Committee on the Status of Endangered Wildlife in Canada. Abbreviations are WCSK, Widgeon Creek Sockeye; LFCK-SP, Lower Fraser Chinook Spring; FCCK, Fraser Canyon Chinook; HRSK, Harrison River Sockeye; LICO, Lillooet Coho; LFCK-SU, Lower Fraser Chinook Summer; FCCO, Fraser Canyon Coho; LFCK-UP, Lower Fraser Chinook Upper Pitt; LFCK-FA, Lower Fraser Chinook Fall; FC, Fraser Chum; MSCK, Maria Slough Chinook; LFCO, Lower Fraser Coho; BBCO, Boundary Bay Coho; and BBCK, Boundary Bay Chinook.

headwaters acting as ideal rearing habitat. While the original assessment done by Proctor (1978) identified approximately 120 km of lost streams within Vancouver's boundary, Canada's Department of Fisheries and Oceans (DFO) built upon this work and raised this number from 120 to 157 km of lost streams in Vancouver City (Fraser River Action Plan (Canada) and Precision Identification Biological Consultants 1998). The estimates from the DEM-derived stream network are consistent with a total of 165 km of lost streams for the same area. Some of the differences between the DFO estimates and those of the DEM-derived stream network may be explained by the difference in shoreline estimation, where this study estimated stream length up to the current shoreline which has been altered due to reclamation over time, the DFO estimations use the historical shoreline. Although the estimation of extent is important for understanding the amount of habitat that may have existed historically, it is likely that the horizontal accuracy of the DEM-derived stream network is reduced in particularly flat areas such as the mouth of the Fraser River. However, the results from the DEM may be combined with or used to

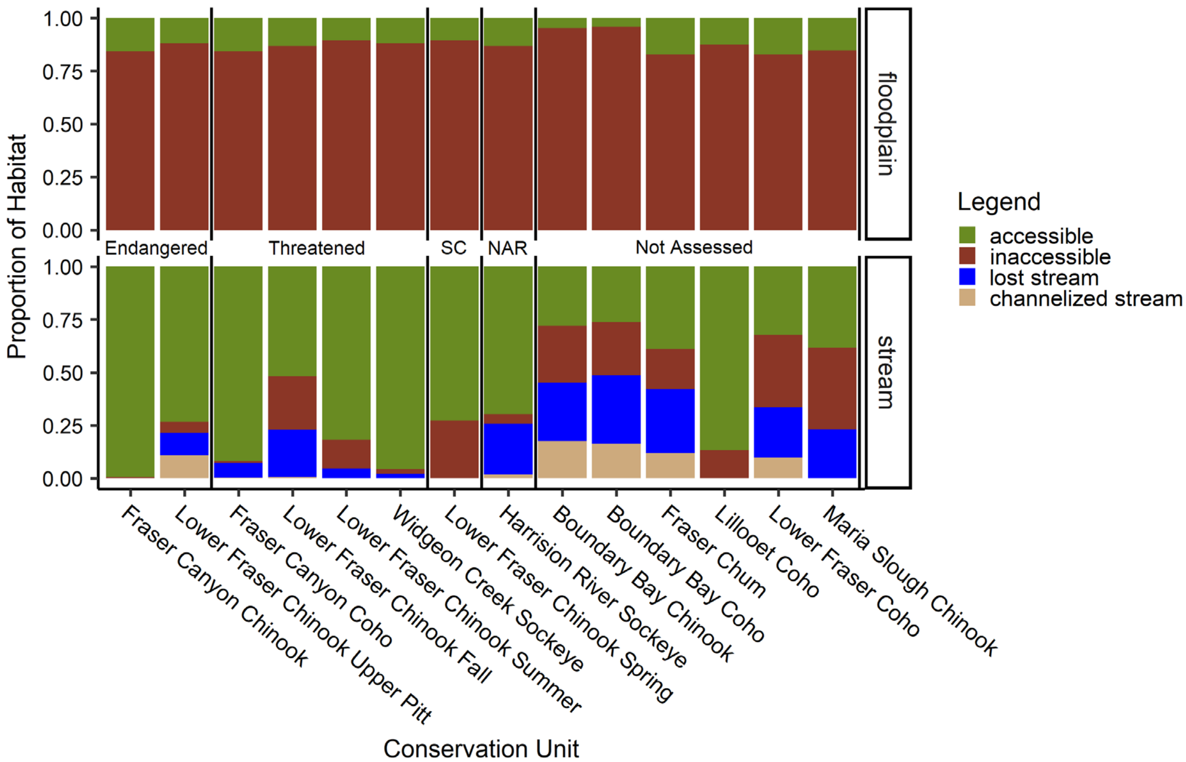

Fig. 7. Proportion of accessible and inaccessible stream and floodplain habitat for 14 conservation units of salmon in the Lower Fraser River. Proportion of stream habitat that has been completely lost and stream habitat that has been converted to drainage channels are also shown. Conservation units are organized by their assessed status according to the Committee on the Status of Endangered Wildlife in Canada, where SC indicates special concern and NAR indicates "not at risk" assessments.

narrow the scope of more exhaustive analyses using historical maps or sewer systems in the future (Broadhead et al. 2015). Moving forward, it will be important to develop multiple lines of evidence for confirming the locations and extents of lost streams and to identify cost-effective opportunities for the restoration of habitat connectivity.

Habitat accessibility varies by conservation unit

Some CUs appear to be much less impacted by instream barriers in terms of the amount of linear stream length that has been alienated. Even after estimating for potentially unmapped barriers, only 3% of the stream habitat was upstream of barriers in the watersheds relevant for Fraser Canyon Chinook. This may be explained by the steep topography and the nature of the road system in the Fraser Canyon. A majority of the road network in these areas is compromised of resource roads that run parallel to the larger rivers and intersect the smaller tributaries that begin in the steep mountains. It is likely that many of these intersections are potential barriers, but according to the gradient model for accessibility, even if a barrier does exist, it does not alienate a large length of habitat. In addition, the type of habitat being lost in these regions is unknown. Certain habitats are disproportionately important, such as spawning and overwintering areas; therefore, even modest losses in the estimated accessible stream length could lead to large implications for the population. For example, coho are known to rely on small streams and side channels for winter rearing (Beechie et al. 1994). For Fraser Canyon coho, 90% of their estimated habitat remains accessible but much of what is alienated may be habitat that is important for overwintering as small streams become cut off by the road network in the system. The alienation of small streams is also of broader concern beyond this region. When predicting the likelihood of a stream–road intersection, the size of the stream was an important determinant (Appendix S4), with smaller streams being more likely to have a barrier on them, consistent with the findings of Januchowski-Hartley et al. (2014). Generally, smaller streams are more likely to be inaccessible, yet they remain important for the production of salmonids (Brown and Hartman 1988).

While barriers in regions with high remaining habitat accessibility may not be disconnecting a substantial proportion of stream length, they may act as an indicator for more indirect impacts happening in the watershed. Some of the CUs with the most remaining habitat had higher barrier density after including model predicted barriers. This could reflect the steep topography and road network in these watersheds, where streams with steeper gradients generally exhibited stream–road intersections with higher probability of being a barrier to fish passage. Unpaved forest roads have been demonstrated to alter flow regimes of streams and provide a source of excess sediment that can impact salmon rearing and spawning habitat downstream (Al-Chokhachy et al. 2016). Development of road networks is ubiquitous with human developed landscapes and has the potential to influence flow regimes, connectivity, sediments, and geomorphology (Wellman et al. 2000) as well as facilitate the accelerated exploitation of previously undisturbed systems (Johnson et al. 2019).

Implications of barriers on status of conservation units

Based on our analysis of barrier density and habitat accessibility, we suggest loss and alienation of habitat is a major driver of observed salmon declines for some populations. Less than half (40%) of the Lower Fraser coho stream habitat and only 17% of the historical floodplain habitat remain accessible. Additionally, it is likely that a sizeable portion of what habitat remains accessible has experienced some form of degradation with impacts on productivity. While it is important to acknowledge the role of marine conditions in stock recovery, it is difficult to ignore a loss of habitat on this scale, especially when the condition of juvenile coho salmon leaving the freshwater environment has been observed to enhance survival when marine conditions are poor (Holtby et al. 1990).

The association between barrier density and habitat accessibility within the context of the population status is informative. When these values were plotted against each other after including predicted barriers, the Harrison river-type sockeye were the only CU identified to have both high habitat accessibility and relatively low barrier density. The Harrison river-type sockeye is also the only population to have been assessed as not at risk by COSEWIC. It is also important to note that many of the CUs that have not been assessed by COSEWIC (Table 1) appear to be the ones most impacted by loss of habitat connectivity and based on this, we suspect that when assessed by COSEWIC, they will be designated as threatened or endangered. These CUs include the Lower Fraser coho and both Boundary Bay Chinook and coho. As of 2019, Lower Fraser coho are considered a stock of concern within the integrated fisheries management plan for salmon in southern BC; however, the cause of concern is attributed purely to marine conditions (Fisheries and Oceans Canada 2019).

These measures of barrier density and habitat accessibility may also guide prioritization of watersheds for systematic assessment and implementation of connectivity restoration. The number of barriers mapped in a system may be used to identify candidates (e.g., Maitland et al. 2016); however, simply looking at the number of barriers can be misleading as many barriers may be alienating small amounts of habitat. By considering habitat accessibility and barrier density on a spatial scale that is relevant to multiple populations of concern, the identification of broad locations for restoring connectivity may be achieved. For example, the Maria Slough Chinook CU appears to have both low habitat accessibility and low barrier density (Fig. 6). The watersheds for this CU may represent a location where the removal of relatively few barriers could have large benefits in terms of the amount of habitat that would become accessible. However, the distribution of barriers in the watershed will determine the efficiency with which habitat will actually be gained from barrier removal. Indicators such as the Dendritic

Connectivity Index may be used to shed light on the orientation of barriers (Cote et al. 2009), and optimization can be used to understand the potential complementarity of multiple barrier removals (O'Hanley and Tomberlin 2005, McKay et al. 2020).

This study demonstrates a historical view of salmon habitat loss in the Lower Fraser and attempts to understand how different salmon CUs are impacted by loss of connectivity. It is important to note that salmon conservation units were only recently developed as part of the Wild Salmon Policy (Fisheries and Oceans Canada 2005) and COSEWIC's assessments of these CUs are based on recent estimates of abundance. In other words, the baselines on which we make these assessments have already shifted. For example, the loss of Sumas Lake occurred in 1924 and represents 7% of the entire floodplain habitat loss in the Lower Fraser, long before any written record keeping of salmon populations and their abundances. Another example of shifting baselines in the Lower Fraser is the Alouette and Coquitlam reservoirs, two lakes which historically had runs of sockeye salmon in them, but due to dam construction in the 1910s and late 1920s, these runs have been extirpated and are not considered to be modern CUs. Landlocked sockeye, called kokanee, currently populate the lake (Godbout et al. 2011). These examples illustrate the importance of considering historical landscape legacies and how the shifted baselines which guide current day management priorities hide historical declines in salmon productivity.

Broader implications

Our study attempts to identify a baseline for restoration and management for salmonid habitat in the Lower Fraser River through combining multiple measures of habitat loss and alienation. In order to be a useful tool for guiding restoration and management of freshwater systems, the assessment of habitat fragmentation needs to consider not only the current barriers and streams but also the historical conditions of the landscape. It is possible that the deceptive nature of the shifting baseline syndrome has skewed the perceptions of what appropriate targets for the restoration of freshwater habitat should look like (Humphries and Winemiller 2009). Further, the restoration of ecosystem function will require the consideration of more than just physical habitat but also important biotic components that may have also been lost (Byers et al. 2006).

Although we have demonstrated a high degree of habitat isolation in the Lower Fraser, we also provide examples of how our analysis can be used to identify cost-effective opportunities for restoration, such as restoring habitat connectivity through the remediation of a small number of barriers which prevent access to large areas of potentially intact habitat. It will also be important to pair species specific estimates of barrier density and habitat accessibility with methods for prioritizing specific barriers (McKay et al. 2017). A more detailed understanding of historical conditions provides a lens through which to understand the potential outcomes of barrier removal and where the daylighting of streams may be used to restore habitat (Wild et al. 2011).

In conclusion, the consideration of lost streams and the use of historical vegetation records were able to supplement an analysis of barriers on stream and floodplain habitat and reveals a high degree of lost and alienated habitat for multiple CUs of in the Lower Fraser River. We believe this demonstrates that freshwater habitat loss and connectivity is a factor which must not be ignored in the discussion of recovery actions for Pacific salmon.

Acknowledgments

We thank the Watershed Watch Salmon Society for sharing their data on flood infrastructure and the Canadian Wildlife Service for sharing spatial files of Lower Fraser floodplain vegetation originally created by Teversham and North (1983). This project was supported through an NSERC CGS and Mitacs Accelerate fellowship to RJRF in partnership with Raincoast Conservation Foundation as well as the Liber Ero Conservation Chair, generosity of Reid and Laura Carter, and an NSERC RGPIN-2019-04535 to TGM. Finally, we are grateful for the valuable discussions with members of the Conservation Decisions Lab and colleagues at Raincoast Conservation Foundation about this research. The authors have no conflict of interest to declare.

Literature Cited

Al-Chokhachy, R., T. A. Black, C. Thomas, C. H. Luce, B. Rieman, R. Cissel, A. Carlson, S. Hendrickson, E. K. Archer, and J. L. Kershner. 2016. Linkages between unpaved forest roads and streambed sediment: why context matters in directing road restoration: unpaved roads and streambed sediment linkages. Restoration Ecology 24:589–598.

Beamish, R. J., R. M. Sweeting, K. L. Lange, D. J. Noakes, D. Preikshot, and C. M. Neville. 2010. Early marine survival of Coho salmon in the strait of Georgia declines to very low levels. Marine and Coastal Fisheries 2:424–439.

Beechie, T., et al. 2013. Restoring salmon habitat for a changing climate. River Research and Applications 29:939–960.

Beechie, T., E. Beamer, and L. Wasserman. 1994. Estimating Coho salmon rearing habitat and Smolt production losses in a large river basin, and implications for habitat restoration. North American Journal of Fisheries Management 14:797–811.

Bilby, R. E., and L. A. Mollot. 2008. Effect of changing land use patterns on the distribution of coho salmon (*Oncorhynchus kisutch*) in the Puget Sound region. Canadian Journal of Fisheries and Aquatic Sciences 65:2138–2148.

Boyle, C. A. 1997. Changes in land cover and subsequent effects on lower Fraser basin ecosystems from 1827 to 1990. Environmental Management 21:185–196.

Broadhead, A. T., R. Horn, and D. N. Lerner. 2015. Finding lost streams and springs captured in combined sewers: a multiple lines of evidence approach: Finding captured streams and springs. Water and Environment Journal 29:288–297.

Brown, T. G., and G. F. Hartman. 1988. Contribution of seasonally flooded lands and minor tributaries to the production of Coho salmon in Carnation Creek, British Columbia. Transactions of the American Fisheries Society 117:546–551.

Byers, J. E., K. Cuddington, C. G. Jones, T. S. Talley, A. Hastings, J. G. Lambrinos, J. A. Crooks, and W. G. Wilson. 2006. Using ecosystem engineers to restore ecological systems. Trends in Ecology & Evolution 21:493–500.

Coleman, R. A., B. Gauffre, A. Pavlova, L. B. Beheregaray, J. Kearns, J. Lyon, M. Sasaki, R. Leblois, C. Sgro, and P. Sunnucks. 2018. Artificial barriers prevent genetic recovery of small isolated populations of a low-mobility freshwater fish. Heredity 120:515–532.

Cote, D., D. G. Kehler, C. Bourne, and Y. F. Wiersma. 2009. A new measure of longitudinal connectivity for stream networks. Landscape Ecology 24:101–113.

Dudgeon, D., et al. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews 81:163.

Elith, J., J. R. Leathwick, and T. Hastie. 2008. A working guide to boosted regression trees. Journal of Animal Ecology 77:802–813.

ESRI. 2018. ArcGIS Desktop. Environmental Systems Research Institute, Redlands, California, USA.

Fish Passage Technical Working Group. 2014. Fish passage strategic approach: protocol for prioritizing sites for fish passage remediation. British Columbia Ministry of Environment, British Columbia Ministry of Forests, Lands and Natural Resource Operations, British Columbia Timber Sales, British Columbia Ministry of Transportation and Infrastructure, British Columbia, Canada.

Fisheries and Oceans Canada. 2005. Canada’s policy for conservation of wild Pacific salmon. Fisheries and Oceans Canada, Vancouver, British Columbia, Canada.

Fisheries and Oceans Canada. 2019. Pacific region final integrated fisheries management plan. Management Plan. Department of Fisheries and Oceans, British Columbia, Canada.

Fraser River Action Plan (Canada) and Precision Identification Biological Consultants. 1998. Wild, threatened, endangered, and lost streams of the Lower Fraser Valley: summary report 1997. Fraser River Action Plan, Fisheries and Oceans Canada, Vancouver, British Columbia, Canada.

Freeman, E. A., and G. Moisen. 2008. PresenceAbsence: an R package for presence-absence model analysis. Journal of Statistical Software 23:1–31.

Fuller, M. R., M. W. Doyle, and D. L. Strayer. 2015. Causes and consequences of habitat fragmentation in river networks: river fragmentation. Annals of the New York Academy of Sciences 1355:31–51.

Gardner, C., S. M. Coghlan, and J. Zydlewski. 2012. Distribution and abundance of anadromous sea lamprey spawners in a fragmented stream: current status and potential range expansion following barrier removal. Northeastern Naturalist 19:99–110.

Gibson, R. J., R. L. Haedrich, and C. M. Wernerheim. 2005. Loss of fish habitat as a consequence of inappropriately constructed stream crossings. Fisheries 30:10–17.

Godbout, L., C. C. Wood, R. E. Withler, S. Latham, R. J. Nelson, L. Wetzel, R. Barnett-Johnson, M. J. Grove, A. K. Schmitt, and K. D. McKeegan. 2011. Sockeye salmon (*Oncorhynchus nerka*) return after an absence of nearly 90 years: a case of reversion to anadromy. Canadian Journal of Fisheries and Aquatic Sciences 68:1590–1602.

Gordon, J., M. Arbeider, D. Scott, S. M. Wilson, and J. W. Moore. 2015. When the tides don’t turn: floodgates and hypoxic zones in the Lower Fraser River, British Columbia, Canada. Estuaries and Coasts 38:2337–2344.

Grant, S. C. H., B. L. MacDonald, and M. L. Winston. 2019. State of Canadian Pacific salmon: responses to changing climate and habitats.

Greenwell, B., B. Boehmke, J. Cunningham, and GBM Developers. 2019. gbm: generalized Boosted Regression Model.

Haro, A., T. Castro-Santos, J. Noreika, and M. Odeh. 2004. Swimming performance of upstream migrant fishes in open-channel flow: a new approach to predicting passage through velocity barriers. Canadian Journal of Fisheries and Aquatic Sciences 61:1590–1601.

Helm, A., I. Hanski, and M. Pärtel. 2006. Slow response of plant species richness to habitat loss and fragmentation. Ecology Letters 9:72–77.

Holtby, L. B., B. C. Anderson, and R. K. Kadowaki. 1990. Importance of Smolt size and early ocean growth to interannual variability in marine survival of Coho salmon. Canadian Journal of Fisheries & Aquatic Sciences 47:2181–2194.

Humphries, P., and K. O. Winemiller. 2009. Historical impacts on river fauna, shifting baselines, and challenges for restoration. BioScience 59:673–684.

Januchowski-Hartley, S. R., M. Diebel, P. J. Doran, and P. B. McIntyre. 2014. Predicting road culvert passability for migratory fishes. Diversity and Distributions 20:1414–1424.

Jeffres, C. A., J. J. Opperman, and P. B. Moyle. 2008. Ephemeral floodplain habitats provide best growth conditions for juvenile Chinook salmon in a California river. Environmental Biology of Fishes 83:449–458.

Johnson, C. J., O. Venter, J. C. Ray, and J. E. M. Watson. 2019. Growth-inducing infrastructure represents transformative yet ignored keystone environmental decisions. Conservation Letters 13:e12696.

Katz, J. V. E., C. Jeffres, J. L. Conrad, T. R. Sommer, J. Martinez, S. Brumbaugh, N. Corline, and P. B. Moyle. 2017. Floodplain farm fields provide novel rearing habitat for Chinook salmon. PLOS ONE 12:e0177409.

Kistritz, R. U., K. J. Scott, and C. D. Levings. 1996. Changes in fish habitat in the Lower Fraser River analyzed by two wetland classification systems. Pages 19–40 *in* C. D. Levings, and D. J. H. Nishimura, editors. Created and restored sedge marshes in the Lower Fraser River and estuary: an evaluation of their functioning as fish habitat. Department of Fisheries and Oceans, West Vancouver, British Columbia, Canada.

Lindsay, J. B. 2016. The practice of DEM stream burning revisited. Earth Surface Processes and Landforms 41:658–668.

Maitland, B. M., M. Poesch, and A. E. Anderson. 2016. Prioritising culvert removals to restore habitat for at-risk salmonids in the boreal forest. Fisheries Management and Ecology 23:489–502.

Malick, M. J., and S. P. Cox. 2016. Regional-scale declines in productivity of pink and chum salmon stocks in Western North America. PLOS ONE 11: e0146009.

McClure, M. M., et al. 2008. Evolutionary consequences of habitat loss for Pacific anadromous salmonids. Evolutionary Applications 1:300–318.

McDaniels, T., S. Wilmot, M. Healey, and S. Hinch. 2010. Vulnerability of Fraser River sockeye salmon to climate change: a life cycle perspective using expert judgments. Journal of Environmental Management 91:2771–2780.

McKay, S. K., A. R. Cooper, M. W. Diebel, D. Elkins, G. Oldford, C. Roghair, and D. Wieferich. 2017. Informing watershed connectivity barrier prioritization decisions: a synthesis. River Research and Applications 33:847–862.

McKay, S. K., E. H. Martin, P. B. McIntyre, A. W. Milt, A. T. Moody, and T. M. Neeson. 2020. A comparison of approaches for prioritizing removal and repair of barriers to stream connectivity. River Research and Applications 36:1754–1761.

Montgomery, D. R., and E. Foufoula-Georgiou. 1993. Channel network source representation using digital elevation models. Water Resources Research 29:3925–3934.

Mordecai, G. J., et al. 2019. Endangered wild salmon infected by newly discovered viruses. eLife 8: e47615.

Murton, J. 2008. Creating order: the liberals, the landowners, and the draining of Sumas Lake, British Columbia. Environmental History 13:92–125.

Napieralski, J. A., and T. Carvalhaes. 2016. Urban stream deserts: mapping a legacy of urbanization in the United States. Applied Geography 67:129–139.

Nesbitt, H. K., and J. W. Moore. 2016. Species and population diversity in Pacific salmon fisheries underpin indigenous food security. Journal of Applied Ecology 53:1489–1499.

North, M. E. A., and J. M. Teversham. 1984. The vegetation of the floodplains of the Lower Fraser, Serpentine and Nicomekl Rivers, 1859 to 1890. Syesis 17:47–66.

Northcote, T. G., and P. A. Larkin. 1989. The Fraser River: a major salmonine production system. Pages 172–204 *in* Proceedings of the International Large River Symposium (LARS). Dept. of Fisheries and Oceans, Ottawa, Ontario, Canada.

O'Callaghan, J. F., and D. M. Mark. 1984. The extraction of drainage networks from digital elevation data. Computer Vision, Graphics, and Image Processing 28:323–344.

O'Hanley, J. R., and D. Tomberlin. 2005. Optimizing the removal of small fish passage barriers. Environmental Modeling & Assessment 10:85–98.

Papworth, S. K., J. Rist, L. Coad, and E. J. Milner-Gulland. 2009. Evidence for shifting baseline syndrome in conservation. Conservation Letters 2:93–100.

Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. Trends in Ecology & Evolution 10:430.

Peterman, R. M., and B. Dorner. 2012. A widespread decrease in productivity of sockeye salmon (*Oncorhynchus nerka*) populations in western North America. Canadian Journal of Fisheries and Aquatic Sciences 69:1255–1260.

Price, M. H. H., B. M. Connors, J. R. Candy, B. McIntosh, T. D. Beacham, J. W. Moore, and J. D. Reynolds. 2019. Genetics of century-old fish scales reveal population patterns of decline. Conservation Letters 12:e12669.

Proctor, S. 1978. Vancouver's old streams. Journal of the Vancouver Aquarium 3:1–32.

R Core Team. 2019. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Roni, P., T. J. Beechie, R. E. Bilby, F. E. Leonetti, M. M. Pollock, and G. R. Pess. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest Watersheds. North American Journal of Fisheries Management 22:1–20.

Rosenvald, R., R. Järvekülg, and A. Lõhmus. 2014. Fish assemblages in forest drainage ditches: Degraded small streams or novel habitats? Limnologica 46:37–44.

SARA. 2002. Species at Risk Act, SC 2002, c 29.

Scott, D. C., M. Arbeider, J. Gordon, and J. W. Moore. 2016. Flood control structures in tidal creeks associated with reduction in nursery potential for native fishes and creation of hotspots for invasive species. Canadian Journal of Fisheries and Aquatic Sciences 73:1138–1148.

Seabloom, E. W., A. P. Dobson, and D. M. Stoms. 2002. Extinction rates under nonrandom patterns of habitat loss. Proceedings of the National Academy of Sciences of the United States of America 99:11229–11234.

Seifert, R. E., and J. W. Moore. 2018. Floodgate operations and fish communities in tidal creeks of the Lower Fraser River (British Columbia, Canada). Estuaries and Coasts 41:1206–1221.

Sheer, M. B., and E. A. Steel. 2006. Lost watersheds: barriers, aquatic habitat connectivity, and salmon persistence in the Willamette and lower Columbia River Basins. Transactions of the American Fisheries Society 135:1654–1669.

Smith, D. A. 2001. Salmon populations and the *Sto:lo* fishery. Pages 120–121 *in* A Sto:lo coast Salish historical atlas. Stó:lō Heritage Trust, Chilliwack, British Columbia, USA.

Sommer, T. R., W. C. Harrell, and M. L. Nobriga. 2005. Habitat use and stranding risk of juvenile chinook salmon on a seasonal floodplain. North American Journal of Fisheries Management 25:1493–1504.

Sommer, T. R., M. L. Nobriga, W. C. Harrell, W. Batham, and W. J. Kimmerer. 2001. Floodplain rearing of juvenile chinook salmon: evidence of enhanced growth and survival. Canadian Journal of Fisheries and Aquatic Sciences 58:325–333.

Stockner, J., and D. Bos. 2002. History of Wahleach and Stave Reservoirs: a paleolimnological perspective. BC Hydro Environmental Services, Burnaby, British Columbia, Canada.

Thomson, A. R., and Confluence Environmental Consulting. 1999. Study of flood proofing barriers in Lower Mainland fish bearing streams. Department of Fisheries and Oceans, Whistler, British Columbia, Canada.

van Puijenbroek, P. J. T. M., A. D. Buijse, M. H. S. Kraak, and P. F. M. Verdonschot. 2019. Species and river specific effects of river fragmentation on European anadromous fish species. River Research and Applications 35:68–77.

Washington Department of Fish and Wildlife. 2019. Fish passage inventory, assessment, and prioritization manual. Page 284. Washington Department of Fish and Wildlife, Olympia, Washington, USA.

Wellman, J. C., D. L. Combs, and S. B. Cook. 2000. Long-term impacts of bridge and culvert construction or replacement on fish communities and sediment characteristics of streams. Journal of Freshwater Ecology 15:317–328.

Wild, T. C., J. F. Bernet, E. L. Westling, and D. N. Lerner. 2011. Deculverting: reviewing the evidence on the 'daylighting' and restoration of culverted rivers. Water and Environment Journal 25:412–421.

Zhong, Y., and G. Power. 1996. Environmental impacts of hydroelectric projects on fish resources in China. Regulated Rivers: Research & Management 12:81–98.

Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.3646/full