

**Flood mitigation structures transform tidal  
creeks from nurseries for native fish to non-  
native hotspots**

**by**

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## **Abstract**

Habitat connectivity is important for ecosystem processes, yet globally is altered by anthropogenic structures. Anthropogenic barriers are common in coastal aquatic ecosystems, yet the effects of small-scale barriers such as floodgates have received relatively little study. The lower Fraser River in British Columbia, Canada, is a large river-floodplain ecosystem with numerous dikes and floodgates protecting valuable human infrastructure and developments. Here we assess fish communities in ten tributaries of the lower Fraser River, five with floodgates and at five reference sites, located primarily in agricultural areas. Floodgate presence was associated with reduced dissolved oxygen concentrations, increased abundance of three invasive fish species, and decreased abundances of five native fish species including two salmon species. These findings provide evidence that floodgates decrease suitable habitat for native fishes, becoming hotspots for non-native fishes. Given sea-level rise and aging infrastructure, there is an opportunity to incorporate biodiversity considerations into restoration of this infrastructure.

### **Keywords**

Flood mitigation; salmon; invasive species; aquatic barriers; tide gates; sea-level rise

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# 1. Introduction

Estuaries and coastal floodplains are ecologically important yet are some of the most threatened ecosystems on earth (Tockner and Stanford 2002). They are extremely productive and provide key ecosystem services such as functioning as nursery habitat for fishes of cultural and economic importance (Beck et al. 2001). However, multiple human actions are rapidly changing these systems (Lotze et al. 2006); for example, seagrass meadows, an important nursery habitat for juvenile marine and estuarine fish, have been increasingly in decline since 1990, now reaching loss rates of 7% per year globally (Waycott et al. 2009). In the future, coastal developments and ecosystems alike are predicted to be threatened by sea-level rise and increasing flood and coastal storm frequency due to climate change (Nicholls et al. 1999). While developed countries will likely offset flooding risk with engineered infrastructure such as dikes, these structures may have ecological consequences as they reduce connectivity between coastal rivers and their floodplains (Airoldi et al. 2005). Understanding the trade-offs associated with flood protection structures will be important for managers to protect aquatic ecosystem function in increasingly human-modified systems.

Connectivity within and between rivers, estuaries, and coastal floodplains creates heterogeneous habitats which are the foundation of aquatic ecosystems and the diverse species assemblages they support (Ward 1989; Ward et al. 1999; Tockner et al. 2010). Longitudinal connectivity within rivers allows downstream communities to utilize energy and organic matter transported from upstream inputs (Vannote et al. 1980), and facilitates the migration of species within the river channel and to the ocean, connecting areas for refuge, breeding and feeding (Welcomme et al. 2006). Connectivity also facilitates natural pulses of flow, resulting in water levels that rise and then fall, allowing rivers to expand and contract over floodplain habitat (Bayley 1995). This expansion connects aquatic and terrestrial ecosystems, promoting the movement of matter and organisms and creates temporary aquatic habitats (Junk et al. 1989). These temporary wetlands facilitate the mineralization and then mobilization of nutrients, the influx of detritus, and act as nursery habitat for young fishes and invertebrates (Junk et al. 1989). Floodplain inundation also triggers spawning in some riverine fish species (Górski et al. 2010) and drives successional patterns which are intricately linked to the typically high biodiversity of these systems (Ward et al. 1999). However, human-structures have

dramatically altered these dynamic processes, either by altering the flow regime or by altering the connectivity that enables this expansion and contraction.

Large river systems are highly modified by humans globally and there is growing appreciation of how some types of infrastructure alter aquatic connectivity and ecosystems. For example, 71% of large river systems in North America, Europe and Northern Asia have been dammed or otherwise modified with dikes and levees to regulate flows, create power, and divert water (Dynesius and Nilsson 1994). These large dams can not only block movements of materials and animals, they also often dampen flow regimes and reduce river floodplain connectivity (Arthington et al. 2010). Impassable dams result in the extirpation of upstream anadromous salmon (Gustafson et al. 2007), and reduce access to different habitats for feeding, spawning and refugia for fluvial migrants (Schlosser and Angermeier 1995). While some authors suggest that barriers can block the spread of invasive species (McLaughlin et al. 2013), dams may facilitate non-native species by providing novel (impounded) habitat (Moyle and Nichols 1974, Johnson et al. 2008) or altering flow regimes to which native fishes were previously adapted (Bernado et al. 2003, Propst and Gido 2004, Kiernan et al. 2010, Fausch et al. 2001). Although these effects of large dams are now recognized, there is arguably less understanding of the ecological effects of smaller-scale structures that also alter aquatic connectivity such as culverts (MacPherson et al. 2012; Favaro et al. 2014), weirs (Mueller et al. 2011; Rolls et al. 2014), dikes (Hood 2004), and floodgates (Pollard and Hannan 1994; Boys et al. 2012; Wright et al. 2014).

Small-scale barriers in aquatic ecosystems such as floodgates are commonly installed in dikes to prevent flooding while enabling the conversion of wetlands and floodplains into areas for agriculture and urbanization (Giannico and Souder 2005). Floodgates are typically installed in low gradient coastal areas to allow tidal tributaries to drain downstream through dikes into the mainstem while preventing backflows and resultant flooding (Pollard and Hannan 1994). Floodgates consist of culverts, often several side by side, with side or top mounted gates on the downstream side. Floodgates require tidal fluctuations on the downstream side to allow the tributary to drain gravimetrically, pushing open the gates and allowing the passage of water and organisms. This can only occur once the difference between tributary and mainstem water levels is sufficient which in semidiurnal tidal systems should occur twice daily

during low tides. Thus floodgates are dynamic temporal barriers which are dependent on fluctuating water levels to determine their operational regimes.

While floodgates are common in coastal areas around the world, their effects on ecosystems are relatively poorly understood. Also commonly referred to as tide gates, recent research has demonstrated effects of floodgates on aquatic ecosystems globally including in North America (Raposa and Roman 2001; Giannico and Souder 2005), Europe (Mouton et al. 2014; Wright et al. 2014), Australia (Pollard and Hannan 1994; Kroon and Ansell 2006) and New Zealand (Doehring et al. 2011). This research has found floodgates to be associated with reduced overhanging vegetation (Pollard and Hannan 1994), greater nutrient concentrations, increased abundance of aquatic weeds (Kroon and Ansell 2006), and reduced dissolved oxygen concentrations (Gordon et al. in review). In estuarine systems, floodgates can be associated with reduced abundance of commercially valuable species (Pollard and Hannan 1994), reduced fish passage (Doehring et al. 2011) including delayed downstream migration of salmonids (Wright et al. 2014), reduced diversity of estuarine fish (Boys et al. 2012), and reduced abundance, biomass, and diversity of juvenile fish (Kroon and Ansell 2006). Research has also demonstrated that removal of floodgates can improve passage of fish and crustaceans and restore fish biodiversity (Boys et al. 2012). This body of previous research has focused on floodgates in estuarine areas where they open daily with tides. However the potential effects of floodgates on snowmelt river systems, where the water might rise and as a result close floodgates for several months at a time, has yet to be extensively studied. As increased climate variability and impending sea-level rise leads to increased reliance on floodgates to offset flood risk in coastal floodplain systems, understanding their ecological consequences will be increasingly important.

In this study we measured the effect of small barriers on fish communities in freshwater tidal tributaries of a large river system. We focused on tidal creeks of the Fraser River (British Columbia, Canada), an enormous (220,000 km<sup>2</sup>) watershed that supports the largest salmon returns in Canada. Floodgates are ubiquitous on tidal creeks in the lower Fraser River, and these creeks are known to support juvenile salmon (Levy and Northcote 1982; Levings et al. 1995). In this system, the yearly spring freshet can result in river levels that rise by several meters for up to several months at a time before receding, likely preventing floodgates from opening (Thomson 1999; Thomson et al. 1999). We sampled tidal creeks with and without the presence of floodgates to

determine if fish communities upstream of floodgates are different than in reference creeks without in-stream barriers. We hypothesized that fish communities upstream of floodgates would be altered due to floodgates acting as physical barriers to fish passage or by the impoundment of water resulting in decreased habitat quality. We predicted that in gated tidal creeks: 1) salmon species would be absent or reduced in abundance as barriers are known to impede migration and possibility extirpate salmon; 2) other native species would also be reduced in abundance as they are known to be affected by habitat alteration; 3) non-native species may be found in increased abundance as they are known to thrive in altered and flow regulated habitats.

## 2. Methods

### 2.1. Study System

The lower Fraser River delta in British Columbia is an example of a highly settled floodplain in which floodgates have become abundant in a historically productive ecological system. The Lower Fraser region supports approximately 1 million people and \$13 billion in development, much of it on the floodplain of the lower Fraser watershed (Fraser Basin Council 2010). The Fraser River is tidal for 115 km upstream of the mouth, with a watershed exceeding 220,000 km<sup>2</sup>. Historically, the Fraser River delta was an intricate floodplain of tidally influenced freshwater and estuarine creeks (Levings et al. 1995). However since the early 20<sup>th</sup> century approximately 70% of the floodplain has become isolated by dikes (Healey and Richardson 1996) and floodgates have become ubiquitous with an estimated 500 installed to control flows (Thomson et al. 1999). The lower Fraser River is home to 42 fish species including at least six introduced species (Richardson et al. 2000). Among the native species are Pacific salmon (*Oncorhynchus* spp.) that move through the estuary during their migration—the Fraser River contains one of the world's largest salmon populations (Levy and Northcote 1982; Levings et al. 1995). In the lower Fraser, tidal freshwater tributaries provide critical rearing and overwintering habitats for juvenile salmon including Chinook (*O. tshawytscha*), coho (*O. kisutch*), and chum (*O. keta*) (Levings et al. 1995), and the use of these non-natal rearing areas is important to the survival of juvenile Chinook salmon migrating seawards from the middle and upper part of the system (Murray and Rosenau 1989). Floodgates likely remain closed for extended periods of time in the lower Fraser during the spring freshet, low flow periods, and high tide cycles, yet these effects on fish communities are poorly understood (Thomson 1999; Thomson et al. 1999; Thomson 2005).

### 2.2. Study Sites

We chose study sites from a larger pool of potential sites based on similarity in physical attributes and fish community potential. A pool of sites was initially generated from the Lower Fraser Valley Streams Strategic Review (Fraser River Action Plan 1999) and Government of British Columbia Ministry of Forests Lands and Natural Resource

Operations Lower Mainland Dike Inventory Maps, based on being in tidal floodplain areas, resemblance in watershed size, gradient, and land use (Table 1). We then conducted preliminary site evaluations to determine accessibility and feasibility of sampling before the final group of sites was selected. Reference sites were located similarly to floodgate sites geographically and in similar tidal, low gradient areas. Reference sites differed from floodgate sites in that flood protection was in the form of dikes running along the banks in the lower lengths in place of having floodgates at the confluence. All sites were located in areas which experience mixed semidiurnal daily tidal fluctuations with distance from the ocean ranging from 44 to 57km. Sites were generally located in agricultural and urban areas and all have been modified in the past through channelizing, diking and straightening. Floodgate sites were also chosen based on having associated pumping stations, the presence of which is typically related to a threshold in watershed drainage area. The presence of pumping stations does not confound our study as pumps only operate when floodgates are closed, thus the increase in turbulent flow in the surrounding area due to operation of the pump occurs when the gates are acting as physical barriers to fish passage.

We studied ten sites located throughout the lower Fraser River floodplain (Figure 1). Five of our sites had barriers in the form of floodgates and associated pumping stations and five of the sites were references, with no in-stream flood mitigation structures. The barrier sites included McLean Creek and Fenton Slough which drain directly to the Pitt River, Cranberry Slough which drains directly to the Alouette River, and Yorkson Creek and Nathan Slough which drain directly to the Fraser River. The pump station at Yorkson Creek contained “fish friendly” Archimedes screw pumps which are thought to impart a lower rate of mortality on out-migrating fish. Cranberry Slough had a single flap gate, however following our study it was determined to be operated solely as a pumping station. We included this site in our analysis to focus on the difference in use of these sites due to the presence of barriers as opposed to the direct mechanism by which they affect fish, therefore we will refer to all sites as floodgate sites. Reference sites included De Boville Slough and Smokwha Marsh which drain directly to the Pitt River, McKenny Creek which drains directly to the Alouette River, and West Creek and Nathan Creek which drain directly to the Fraser River (Table 1).



## 2.3. Sampling Methods

Each of the ten sites were sampled once per month from April through August during the summer of 2013. Sampling was conducted in ten consecutive days each month, except April in which Smokwha Marsh was sampled three days after completion of the other sites. Sampling generally alternated daily between reference and floodgate sites to reduce the potential effect of within month variation. Sampling was conducted from April 11th to 23rd, May 7th to 16th, June 10th to 19th, July 9th to 18th and August 14th to 23rd. Water levels at floodgate sites were consistent between different sampling occasions, presumably because of the pump operations and floodgates that buffered tidal and seasonal variation. At reference sites water levels significantly rose following the start of the spring freshet fluctuating by several meters between lows in April and August and a peak in late May. Water levels also fluctuated daily with tides therefore sampling was generally conducted at midday and low to mid tide height to maximize accessibility and increase sampling effectiveness.

We captured fish by seine hauls using a 15.24 m by 2.44 m net with 0.3175 cm mesh size on each sampling occasion. For each haul two crew members would walk out to the center of the creek, one crew member would hold the net while the other walked downstream fully extending the net. Both crew members would then circle towards the bank where two more crew members would assist in quickly pulling the net into a purse at the bank. We conducted three seine hauls at each sampling event; consecutive seine hauls were typically conducted immediately following completion of identification of fish from previous haul and were separated spatially and by habitat type if habitats were not homogenous. After identification fish were held in aerated buckets to prevent re-capture in consecutive hauls. We also set minnow traps with 0.3175 cm mesh size, approximately 25 m apart, set overnight for periods averaging 18 hours on each of our sampling occasions, baited with  $20.0 \pm 2.0$  g cured salmon eggs. Fish caught in traps were identified and measured prior to commencement of seine hauls and were typically held until seining was completed if seine hauls were conducted in the same area as traps were set. All fish were released following identification. Sampling was conducted under approval of the Simon Fraser University Animal Care Committee and permits were obtained from federal and provincial agencies. Water chemistry measurements of salinity, temperature, dissolved oxygen concentration, and conductivity were obtained

using a YSI (model 556 MPS, YSI Incorporated 2009). Water chemistry measurements were taken just below the water surface at noon or within thirty minutes, just upstream (~50m) of floodgates or the confluence at reference sites.

## **2.4. Watershed Land Use Analysis**

We determined the total area and analyzed proportions of lands uses for the watersheds we studied. This was done with the watershed tools in ArcGIS using a 25m resolution digital elevation model, land use spatial layers, and stream and river locations in British Columbia. As our sites are located in extremely low gradient areas the software had difficulty determining the correct dimensions for some of our sites. We therefore used a dataset outlining streams and rivers in B.C. created by the Ministry of Environment in 2005, along with Google Earth (Version 7.1.2.2041, Google Inc., Mountain View CA, USA) images and our knowledge of the watersheds, to draw polygons outlining our watersheds based on those initially delineated by ArcGIS, and then calculated total area. To determine land uses we obtained a land use dataset created by MetroVancouver in 2006 with 25 meter resolution at a 1:20,000 scale which indicates the dominant land use for each parcel. We then grouped watershed use into; 1) agriculture, 2) urban which represented all forms of residential land use along with commercial and institutional, 3) other human use which represented industrial, transportation, recreation and parks, and 4) undeveloped and protected areas. Our land use data set did not cover all of the watershed areas for Nathan Creek and Nathan Slough with data coverage for 44 and 34 percent of each watershed respectively. Based on visual inspection of Google Earth images of the remaining portions of each watershed the land use appeared similar therefore we used proportions based on the available data. All analysis was conducted using ArcGIS version 10.2 (ESRI 2014).

## **2.5. Statistical Analysis**

We analyzed fish data at the community and species levels. For both sets of analyses, we summed our catch data from our traps and seine hauls for each sampling

occasion at each site, as they represented an equal sampling effort for each sampling date. Our aggregated catch data thus represents a metric of the fish community at each site. We used non-metric multidimensional scaling (NMDS) (Prentice 1977) to explore the relationship between floodgate presence and community composition at our sites. NMDS analysis was used to visualize community dissimilarity across sites and across time and to visualize which species were influencing community composition. Species abundances were fourth root transformed to satisfy normality for multivariate analysis. Unidentified juvenile minnows were grouped with peamouth chub and northern pikeminnow under the category minnow. We also combined fish identified as pumpkinseed and black crappie with our un-identified juvenile sunfish under the category sunfish, and as only one individual black crappie was captured during our entire sampling, it is likely sunfish primarily represent pumpkinseed. Then a community dissimilarity matrix was generated based on the species composition. For our NMDS we used two dimensions ( $k=2$ ) and our stress score was 0.174. We ran a permutational multivariate analysis of variance test (PERMANOVA; Anderson 2001) to test the significance of floodgate presence and date on our community composition. Our model included floodgate presence, date and an interaction term between floodgate presence and date. These analyses were done in the program R (version 3.1.1; R Development Core Team 2014), using the vegan package (Oksanen et al. 2011).

We examined the relationship between floodgate presence and abundance for each species with adequate data using generalized additive models (GAM). For each species, we used a GAM to testing the effect of floodgate presence on our abundance data for each species while accounting for the effect of date with a smoothing function. Based on preliminary data exploration and our NMDS analysis it appears that our data was significantly affected by sampling occasion due to seasonal variation in fish abundances. For instance, highest salmon abundance was obtained in April and May, as was expected based on knowledge of salmon life histories (Murray and Rosenau 1989). We therefore decided to use generalized additive models (GAM; Hastie and Tibshirani 1987), which function as an extension of generalised linear models that can incorporate a non-linear smoothing function for an independent variable. GAM's allowed us to use an additive smoothing function for the effect of date, which we knew to be non-linear, thereby providing better model fit. For non-salmon species we ran our GAM with a negative binomial error distribution. We normalized our data by dividing our abundances

for each sampling occasion by the total standard deviation for each species prior to analysis to allow better comparison between species. We ran models for each species of minnow separately and for the category of unidentified juvenile minnow which likely represented a combination of northern pikeminnow and peamouth chub. We excluded species caught at very low abundances ( $n \leq 10$ ) and frequency, including rainbow trout (*Oncorhynchus mykiss*), redbreasted sunfish (*Lepomis gibbosus*), redbreasted sunfish (*Richardsonius balteatus*) and largescale sucker (*Catostomus macrocheilus*) as sample size for these species did not allow meeting conditions of normality. We again combined fish identified as pumpkinseed and black crappie with our un-identified juvenile sunfish for analysis. As our salmon data was highly skewed, particularly for Chinook and chum, to satisfy normality we used a  $\log_{10}(x+1)$  transformation prior to analysis, divided by the standard deviation to allow comparison, then ran our GAM using a quasipoisson error distribution. As Chinook and chum salmon were only captured in the first two and three sampling periods respectively we only used those data for our GAM's. Dates used for analysis were cumulative days passed since the first day of sampling. GAM's were run using the mgcv package in R (Wood 2001; R Development Core Team 2013). We used an alpha level of 0.05 to determine statistically significant results.

### 3. Results

Reference and floodgate sites were similar in watershed area and dominant land uses (Table 1). Study watersheds were typically small, floodgate watersheds averaged 7.00 km<sup>2</sup>, ranging from Fenton Slough at 3.33 km<sup>2</sup> to Yorkson Creek at 17.12 km<sup>2</sup>, while reference watersheds averaged 8.92 km<sup>2</sup> and ranged from Smokwha Marsh at 4.74 km<sup>2</sup> to West Creek at 15.29 km<sup>2</sup>. Land use was predominantly agriculture and urban, being the highest percentage of use of four of our five reference sites and all of our five floodgate sites. The one exception was the reference site Smokwha Marsh which is mostly situated in what is now a protected area, but was historically used for agriculture and as such is channelized, diked and does not experience a natural hydrological cycle, and is therefore arguably similar to our other sites. Floodgate and reference sites were similarly distributed through the region (Figure 1).

Variation in measured water quality parameters was associated both with sampling date and floodgate presence. Temperatures increased throughout the summer in all sites with no trends related to floodgate presence (Table 2). Salinity was measured at very low concentrations in both floodgate and reference sites throughout the study period (Table 2). More notably, floodgates were associated with decreased dissolved oxygen levels (Figure 2). Dissolved oxygen concentrations were initially similar among all sites, however by later sampling periods concentrations decreased in floodgate sites compared to reference sites. During our August sampling period, dissolved oxygen concentrations at floodgate sites fell to levels below BC Ministry of Environment safe minimum standards (5 mg/l) for the protection of aquatic life (GBCME 1997) (Figure 2).

We captured a total of 30,759 fish of 21 different species throughout our sampling. We captured 674 juvenile salmon of five different species, 29,051 fish from 10 different non-salmon native species (hereafter referred to as other native species), and 734 fish of six different non-native species. The majority of juvenile salmon captured were fry while a few (< 20) smolts were also captured. The majority of juvenile salmon species captured were chum, Chinook, and coho respectively, while a few pink (*O. gorbuscha*) and sockeye (*O. nerka*) were also captured at one site. Native three-spine stickleback (*Gasterosteus aculeatus*) dominated catches, with 27,791 individuals captured. Other native species captured in abundance included the northern pikeminnow (*Ptychocheilus oregonensis*), prickly sculpin (*Cottus asper*), and peamouth chub

(*Mylocheilus caurinus*). Non-native species captured included pumpkinseed (*Lepomis gibbosus*), largemouth bass (*Micropterus salmoides*), common carp (*Cyprinus carpio*), brown bullhead (*Ameiurus nebulosus*), black crappie (*Pomoxis nigromaculatus*) and weather loach (*Misgurnus angullicaudatus*).

Community-level analyses indicated fish community composition to be significantly different between floodgate and reference sites. Community composition varied both by site type and by sampling occasion (Figure 3). Visualization of fish communities with NMDS indicated that the community composition shifted throughout the summer in reference sites from fish communities with salmon to communities more dominated by minnow (*Cyprinidae*) and prickly sculpin. Fish communities at floodgate sites shifted from communities dominated by stickleback towards communities dominated by sunfish (*Centrarchidae*) and brown bullhead. We found significant differences in fish communities associated with floodgate presence ( $F = 12.46$ ;  $P = 0.001$ ), date ( $F = 11.58$ ;  $P = 0.001$ ), and an interaction between floodgate presence and date ( $F = 2.09$ ;  $P = 0.015$ ).

Floodgates were negatively associated with abundances of salmon. Juvenile salmon were captured at all five reference sites but at only two floodgate sites. Total juvenile salmon abundance was 2.5 times greater in reference sites relative to floodgate sites, and on average consistently greater for each sampling period and for each species (Figure 4). Total abundance was 11.7 times greater for coho, 1.5 times greater for chum and 2.2 times greater for Chinook salmon, in reference sites relative to floodgate sites. There was also a strong seasonal trend in abundance as would be expected with the majority of individuals captured in April and May (Figure 4). These differences in total abundance in floodgate sites relative to reference sites were statistically significant for coho (GAM:  $\beta = -1.700$ ,  $SE = 0.381$ ,  $t = -4.466$ ,  $P = 0.0001$ ), and chum ( $\beta = -1.319$ ,  $SE = 0.492$ ,  $t = -2.683$ ,  $P = 0.013$ ) but not for Chinook salmon ( $\beta = -0.808$ ,  $SE = 0.444$ ,  $t = -1.819$ ,  $P = 0.087$ ) (Figure 7).

Floodgates were also negatively associated with the majority of other native species. Three-spine stickleback comprised 95.6% of our catch of other native fish species, and were similar in abundance between floodgate and reference sites throughout the summer (Figure 5a,b). Prickly sculpin and minnow (*Cyprinidae*) species were 37.2 and 11.7 times more abundant respectively at reference sites relative to

floodgate sites throughout our sampling periods (Figure 5c,d). Using GAMs, we found these differences to be statistically significant for prickly sculpin (GAM:  $\beta = -3.607$ , SE = 0.796,  $t = -2.62$ ,  $P = 0.0001$ ; Figure 7), northern pikeminnow (GAM:  $\beta = -2.094$ , SE = 0.592,  $t = -3.540$ ,  $P = 0.001$ ; Figure 7), peamouth chub (GAM:  $\beta = -1.350$ , SE = 0.395,  $t = -3.423$ ,  $P = 0.0015$ ; Figure 7), and unidentified juvenile minnows (GAM:  $\beta = -1.883$ , SE = 0.628,  $t = -3.00$ ,  $P = 0.005$ ; Figure 7).

Floodgates were positively associated with the majority of non-native fish species. In total, non-native species were 3.1 times greater in abundance at floodgate sites relative to reference sites (Figure 6a). Sunfish were 4.3 times more abundant at floodgate sites (Figure 6b), and this difference was statistically significant (GAM:  $\beta = 1.477$ , SE = 0.577,  $t = 2.560$ ,  $P = 0.0137$ ; Figure 7). We found a similar statistically significant positive effect of floodgate presence on brown bullhead (GAM:  $\beta = 2.733$ , SE = 0.969,  $t = 2.819$ ,  $P = 0.007$ ; Figure 7) and common carp abundance (GAM:  $\beta = 2.037$ , SE = 0.843,  $t = 2.417$ ,  $P = 0.020$ ; Figure 7). Largemouth bass were the only non-native species that were not statistically higher in floodgate sites (GAM:  $\beta = -0.276$ , SE = 0.537,  $t = -0.515$ ,  $P = 0.61$ ; Figure 7). Overall we found statistically significant effects of floodgate presence on the abundance of eight of the fish species we studied, including negative effects on two salmon species and three other native species, and positive effects on three non-native species.

## 4. Discussion

Our results demonstrate significant negative effects of floodgates on native fish communities and their habitats in tidal creeks. We found floodgate presence to be associated with decreased abundance of salmon and other native fish species (excluding stickleback), which were found in 2.5 times and 14.3 times greater abundance at reference sites respectively. We also found floodgates to be associated with 3.1 times greater abundance of non-native fishes and depressed dissolved oxygen concentrations. Although all of our sites are in areas impacted by human land uses, it appears the presence of floodgates is a key driver of fish community change. While large dams are known to profoundly impact freshwater aquatic systems, our results demonstrate that small scale barriers also impair native fish and facilitate non-native fishes. As floodgates are ubiquitous in many coastal aquatic systems, such as in the lower Fraser River, the collective impact of these small structures may be an important yet relatively unconsidered driver of undesirable change.

Floodgates were strongly associated with decreased dissolved oxygen concentrations, a key attribute of habitat quality. Dissolved oxygen concentrations were lower in floodgate sites than reference sites, particularly in August during which time they fell below the local British Columbia Provincial Criteria for the Protection of Aquatic Life of 5 mg/L, while reference sites remained near saturation levels. Similarly Santucci et al. (2005) studied a river fragmented by low head dams and found that in impounded reaches dissolved oxygen concentrations regularly fell below local protection criteria, while in free flowing reaches they remained at safe levels. Concurrently, we also investigated the spatial extent of floodgate related hypoxia in a in our study system and found that reduced oxygen concentrations extended at least 100m upstream of floodgates yet downstream of floodgates concentrations remained at safe levels (Gordon et al. *in review*). Thus floodgates may result in upstream “dead zones”, creating areas which are no longer suitable habitat for oxygen-sensitive fishes (Gordon et al. *in review*), leading to hypoxic fish kills (Breitburg 2002). While there is widespread appreciation for large-scale hypoxia in coastal oceans, there is less appreciation for the potential cumulative impacts of small-scale hypoxia (Gordon et al. *in review*; Pressy and Middleton 1982). Floodgate-related hypoxia may be an important implication of tidal restriction for managers to consider in developed coastal floodplains.



Floodgates were negatively associated with salmon. We found 2.5 times greater abundance of juvenile salmon at reference sites relative to floodgate sites. Juvenile salmon were also completely absent at three of our five floodgate sites, a pattern which was unrelated to the river to which each tributary directly connected. Large barriers are known to extirpate salmon (Sheer and Steele 2006), and our results demonstrate that small scale barriers, which are much more abundant, also exclude salmon. Floodgates could negatively affect salmon abundance by preventing adults from reaching spawning grounds (Wright et al. 2014), preventing the re-distribution of juveniles or by reducing water quality thereby making areas uninhabitable. Floodgates had the greatest negative effects on juvenile coho and chum salmon. We also found reduced abundances of Chinook salmon, however these results were not statistically significant likely due to small sample size as this species was only captured in the first two sampling periods. Tributary habitats are known to be important for winter growth and survival of juvenile coho, therefore lost connectivity due to in-stream barriers may have significant population level effects (Ebersole et al. 2006). Coho productivity has shown to be impacted by habitat alteration (Bradford and Irvine 2000), diking and other small scale modifications (Beechie et al. 1994), and coho distributions have been shown to be altered by culverts (Davis and Davis 2011). Chum salmon typically spend little time in freshwater before migrating towards the ocean, therefore reduced abundance of juveniles is likely related to differences in spawner abundance or distribution. However we lack data regarding spawners specifically for our sites which limits our ability to fully understand how floodgates affect chum salmon. Interestingly, we documented juvenile Chinook salmon presence in two of our floodgate sites. As Chinook are not known to spawn in any of our study areas their presence suggests successful passage of juveniles, indicating floodgates at these sites were only acting as partial barriers to juvenile movement during our study period. Conversely the absence of juvenile Chinook salmon at three of our sites, may indicate that floodgates impede Chinook salmon usage of gated tidal creeks. Given that there are approximately 500 floodgates in the lower Fraser area, these structures likely have cumulatively eroded nursery capacity for Fraser River salmon.

We found floodgate presence to be associated with statistically significant reductions in abundance of three common other native fish species in our system. With the exception of three-spine stickleback, which were abundant at all of our sampling

locations, the total abundance of other native species was over 14 times greater at reference sites relative to floodgate sites. Floodgate presence had the greatest negative effect on the abundance of prickly sculpin, which were 37 times more abundant at reference sites. Prickly sculpin are typically present in coastal streams of the Pacific Northwest but have been shown to be limited by small barriers including culverts (Favaro et al. 2014), and fish ladders passable by salmon and trout (LeMoine and Bodensteiner 2014). Prickly sculpin adults spawn in the rivers and streams, and larvae drift downstream to a lake, estuary, or other lentic habitat to rear before moving back up as 1+ year olds (Krejsa 1967); floodgates may prevent this upstream migration. Furthermore, prickly sculpin have been shown to be more abundant in areas with natural hydrological cycles and greater floodplain connectedness (Crain et al. 2003).

Minnow (*Cyprinid*) species were also negatively affected by floodgate presence including statistically significant reductions in northern pikeminnow and peamouth chub abundance. The effects of hydrological modifications on minnow species typically has received little attention as they are not economically important, however minnow biodiversity is in decline across North America (Whittier et al. 1997). Winston (1991) described the upstream extirpation of four minnow species related to construction of a mainstem dam and Porto (1999) found reduced abundances of seven species of stream fishes upstream of low-head dams relative to reference sites. Our results further demonstrate that small scale barriers can also influence native stream fish communities. How floodgates effect the minnow species we studied may be related to reproductive strategy, for example Platania and Altenbach (1998) found that interactions between dam-related flow modifications and downstream transport of eggs and larvae led to declines in seven minnow species they studied. In the Columbia River system, northern pikeminnow are known to inhabit the mainstem as adults, move upstream into tributaries to spawn, then rear as juveniles in shallow low velocity areas (Gadomski et al. 2001). In our system, floodgates may interfere with the use of different habitats across life stages for these native fishes.

We found floodgate sites to be a hot-spot for non-native fish species including pumpkinseed, brown bullhead and common carp, all of which are considered to be invasive. In total, non-native species were 200% more abundant in floodgate sites relative to reference sites. Our results are consistent with a recently growing body of literature associating invasive species abundance with river impoundments (Johnston et

al 2008; Clavero et al. 2014; Gois et al 2014). Chu et al. (2014) found increased numbers of low head dams to be associated with increased non-native abundances, and our data demonstrate floodgates have similar effects. Pumpkinseed, the most common invader in our study sites, has been found in high abundances downstream of dams, indicating they may gain an advantage in highly altered flow regimes (Clavero et al. 2014). While these invasive species were introduced to this system long ago (Dextrase and Mandrak 2006), floodgates may support source populations of these invasive species, facilitating their spread into nearby areas, enabled by dispersal through the periodic barrier that floodgates represent.

While our results demonstrate that floodgates are associated with altered fish communities, we acknowledge that other differences between our sites may have contributed to these patterns. Floodgate presence is likely non-random and associated with particular land use decisions. However our reference sites were similar in size and gradient, the main difference being they were typically isolated from their floodplain by parallel dikes. Another challenge is that floodgate sites unavoidably differ in the number and construction of flap gates, as well as the height at which they are installed, inevitably leading to differences in the timing, duration and magnitude of flap gate opening versus closure. However, the differences in fish communities we found are supported by previous findings from Australia which found reductions in eight commercially valuable species when comparing sites with floodgates to un-gated reference channels (Kroon and Ansell 2006). Kroon and Ansell (2006) also found that floodgates were acting as physical barriers to migratory fish and invertebrate species and resulting in environmental conditions which favoured exotic species. Therefore, our data and previous research indicates that floodgates are the main driver of the observed shifts in fish communities.

While our study design prevented isolation of the precise mechanisms by which floodgates are affecting fish communities, mechanisms likely are associated to changes in hydrologic connectivity and habitat quality. Floodgates may directly prevent fish passage, reducing access to habitats important for survival, growth, or reproduction. In snowmelt-driven systems such as the Fraser River, high mainstem levels during spring freshet may prevent gates from opening during tidal cycles (Thomson 1999; Thomson et al. 1999). Floodgates have been shown to delay migration of salmonids (Wright et al. 2014) and floodgate opening during low tide cycles depends on upstream hydraulic head

differential, which may create water velocity barriers rendering them impassable, especially for less mobile species such as sculpin. Floodgates may also impact fish communities indirectly, by altering habitat through impounding water (Johnston et al. 2005) leading to oxygen depletion (Gordon et al. *in review*). This hypoxia alters habitat quality for fishes, can drive fish kills (Richardson 1981), and may act as a chemical barrier to fish passage. Respiration rates necessary to deplete oxygen concentrations are likely influenced by high nutrient concentrations from agricultural runoff, as fertilizer and manure applications in our study areas typically exceed soil needs (Hall and Schreier 1996). Non-native species may benefit from reduced competition due to reduced abundance of native species in floodgate sites, or from higher levels of disturbance and habitat alteration (Moyle and Light 1996). Although we did not determine the mechanisms by which floodgates impacted the fish species we studied, it seems likely they affect different species in different ways related to individual species traits. Further research to determine the mechanisms by which different species are affected by floodgates will be important to guide floodgate remediation efforts.

## 5. Management Implications

Our results indicate that the effects of floodgates, combined with their ubiquity, may represent an important yet relatively unconsidered contributor to habitat loss in the lower Fraser River, an important nursery area for juvenile salmon. Floodgates are common in the lower Fraser area, with an estimated total of over 500 separate structures (Thomson et al. 1999). Fraser River salmon are an important economic resource, yet many populations have long been declining (Northcote and Atagi 1997). Juvenile coho and Chinook salmon are abundant in the lower Fraser River (Richardson et al. 2000), yet the vast majority of tributary streams and sloughs are considered lost, threatened or endangered by human activity (Precision Identification Biological Consultants 1997). This has serious management implications, as lower Fraser tributaries are known to be important areas for rearing and feeding during the seaward migration of juveniles from throughout the system (Murray and Rosenau 1989). As floodgates are present on a substantial proportion of lower Fraser tributaries, there cumulative effects may be of significant management concern for salmon, limiting their ability to access critical nursery habitats during their seaward migration.

Freshwater fish are in decline and one of the most threatened groups of vertebrates globally, with the number one and two main threats viewed as habitat alteration and non-native species (Fausch et al. 2002). The spread of invasive species is of significant management concern, with economic costs due to lost services and control expenditures estimated at 120 billion dollars per year in the United States alone (Pimentel et al. 2005). Our results indicate that small scale flood mitigation structures are associated with both increased abundance of invasive species and decreased abundance of native species. In particular we found floodgate presence associated with increases in pumpkinseed and brown bullhead. These particular invasive species are the two most commonly listed as threats to freshwater fish species at risk in Canada (Dextrase and Mandrak 2006). As floodgates are likely only partial barriers, increased abundances of invasive species means floodgates may be creating hot spots of invasive species which can then spread to nearby locations. Habitat restoration, such as floodgate remediation, has potential to prevent the further spread of invasive species while restoring native fish populations (Scoppettone et al. 2005). Remediation efforts which aim to restore natural hydrologic connectivity may offer the greatest potential for

protecting native species. For example flow regimes manipulated to resemble natural processes are associated with relative increases in native fishes and decreases in non-native species (Propst and Gido 2004). While some studies suggest that restoring connectivity could promote the spread of invasive species (McLaughlin et al. 2013), our results suggest the opposite.

While floodgates may be responsible for undesirable effects on salmon and other native fish in the lower Fraser, they also represent a potential opportunity for restoration. Restoring habitat for salmon is potentially economically valuable (Knowler et al 2003), and the use of floodgates has likely isolated significant amounts of habitat once used for rearing and spawning. Removing or remediating floodgates, like other small barriers such as culverts, may represent a cost-effective alternative to habitat restoration to restore productivity in some areas (Beechie et al. 1994; Ebersole et al. 2006). For salmon, ensuring floodgate remediation options allow both upstream movement of spawners and downstream movement of juveniles will be crucial to success (Calles and Greenberg 2009). Furthermore, ensuring that dissolved oxygen concentrations are adequate throughout the year is important for species which inhabit these freshwater streams year-round, such as juvenile coho salmon (Beechie et al. 1994). While conservation efforts often invest in creation of new habitats, restoring passage of water and fish through floodgates may be particularly economically efficient as it has the potential to reconnect and restore large areas of existing habitat previously utilized by salmon and other native species.

Floodgate removal and remediation may be a valuable restoration tool yet little research has been done on the various management options available. Floodgate removal has been shown to result in desirable changes to water chemistry and plant communities (Wetzel and Kitchens 2007), and floodgate remediation has been shown to restore fish communities (Boys et al. 2012). Floodgate remediation can range from technical solutions, such as modifying existing gates or installing self-regulating tide gates, or management options, including chaining gates open when flood risk is low. Self-regulating gates have attractive potential as they allow inflow of tidal water up to a pre-set depth threshold before closing to prevent flooding, but their effectiveness has yet to be studied, especially in rivers that have a strong seasonal flood-pulse. Boys et al. (2012) studied floodgates remediated through installation of smaller flap gates into larger gates and manually winching gates open, both of which resulted in recolonization by

species common at un-gated sites. They found that following floodgate remediation fish communities soon resembled reference sites, and that this response was sustained at two of three sites, leading to increased fish biodiversity (Boys et al. 2012). Conversely Wright et al. (2014) found that installing orifices into floodgates did not significantly improve salmonid passage. As floodgates can typically be modified to improve connectivity (Williams and Watford 1997), research into the effectiveness of different options will guide future restoration efforts. Although removing floodgates is likely infeasible in many situations due to the need for flood control, other floodgate remediation options which improve passage and water quality may represent an effective and efficient restoration measure.

Flood risk is predicted to increase as a result of climate change (Arnell and Gosling 2014). Coupled with sea level rise, there will undoubtedly be an increase in the use of flood protection structures in coastal aquatic systems. Sea level rise will also impact the function of existing structures, many of which are near the end of their predicted lifespan, requiring they be modified or replaced to continue to protect against flooding (Walsh and Miskewitz 2013). This creates an opportunity with future flood mitigation structures to prevent or reverse changes to the tidal ecosystems in which they are installed. This will be increasingly important as climate change, along with increased human demand for water resources, will further modify flow regimes (Vörösmarty et al. 2000), likely creating conditions which put native freshwater species at risk while aiding the spread of invasive species. Therefore as dam operations are modified to mimic natural flow regimes (Olden and Naiman 2010) with positive effects for native freshwater species (Kiernan et al. 2012), a similar approach should guide the remediation of small scale barriers in coastal systems. Restoring tidal action may be an achievable goal, and has been shown to set coastal marshes on a trajectory towards full recovery over time (Warren et al. 2002). Overall the future of flood mitigation structures represents both a challenge and opportunity for managers in coastal ecosystems. Given climate change, sea-level rise, and aging infrastructure, there is an opportunity to incorporate biodiversity considerations into further development or restoration of this infrastructure. Therefore to protect valuable ecosystem services and freshwater biodiversity future flood mitigation structures must be designed and managed to allow fish passage and natural hydrological regimes.

## Tables

**Table 1. Site information, watershed area and proportions of different land uses in the watersheds of our study sites. Watershed area determination and land use analysis completed using ArcGIS, land use calculations based on MetroVancouver land use dataset created in 2006.**

Sites	# Flap gates (year installed)	Distance from ocean (km)	Total Area (km <sup>2</sup> )	Agriculture (%)	Urban (%)	Other Human Use (%)	Undeveloped/Protected (%)
<i>Reference</i>							
De Boville	-	42.1	8.63	4.17%	48.15%	1.30%	46.39%
McKenny	-	46.6	5.42	24.89%	51.71%	23.06%	0.35%
Smokwha	-	50.7	4.74	10.87%	0.00%	0.00%	89.13%
West	-	52.5	15.29	77.84%	0.79%	13.33%	8.04%
Nathan C.	-	55.2	10.54	89.59%	0.21%	6.87%	3.34%
<i>Floodgate</i>							
McLean	4(1984)	42.3	4.06	73.27%	0.00%	17.00%	9.73%
Cranberry	*(1984)	44.7	5.27	90.84%	0.00%	9.10%	0.06%
Fenton	2(1984)	45.7	3.33	86.80%	8.36%	4.84%	0.00%
Yorkson	2(1994)	43.3	17.12	34.34%	46.34%	12.68%	6.65%
Nathan S.	2(1950)	57.4	5.20	95.91%	0.00%	4.09%	0.00%

\*Following our sampling it was determined that the structure at Cranberry Slough functions solely as a pumping station.

**Table 2. Average water chemistry measurements with standard deviations for each site type from each sampling month. Measurements were taken just upstream of the floodgates or at equivalent locations at reference sites, just below water surface at 12:00 pm plus or minus 30 minutes on each sampling occasion.**

Month	Type	Temperature (°C)	Dissolved Oxygen (mg/L)	Salinity (ppt)	Conductivity (mS/cm)
April	<i>Reference</i>	10.3 (±2.0)	11.69 (±0.79)	0.042 (±0.026)	0.084 (±0.053)
	<i>Floodgate</i>	9.8 (±1.0)	8.48 (±1.76)	0.064 (±0.026)	0.135 (±0.056)
May	<i>Reference</i>	14.4 (±0.6)	8.34 (±2.63)	0.046 (±0.029)	0.100 (±0.061)
	<i>Floodgate</i>	16.7 (±3.3)	9.25 (±5.10)	0.092 (±0.047)	0.194 (±0.098)
June	<i>Reference</i>	15.2 (±0.9)	7.95 (±1.36)	0.054 (±0.033)	0.113 (±0.066)
	<i>Floodgate</i>	17.0 (±0.9)	7.70 (±2.94)	0.122 (±0.053)	0.263 (±0.115)
July	<i>Reference</i>	17.6 (±2.1)	7.73 (±1.08)	0.056 (±0.038)	0.123 (±0.081)
	<i>Floodgate</i>	18.7 (±1.8)	3.91 (±1.82)	0.122 (±0.051)	0.257 (±0.109)
August	<i>Reference</i>	18.2 (±4.4)	7.88 (±1.38)	0.062 (±0.033)	0.133 (±0.067)
	<i>Floodgate</i>	18.8 (±2.1)	1.68 (±1.33)	0.120 (±0.060)	0.251 (±0.124)



# Figures

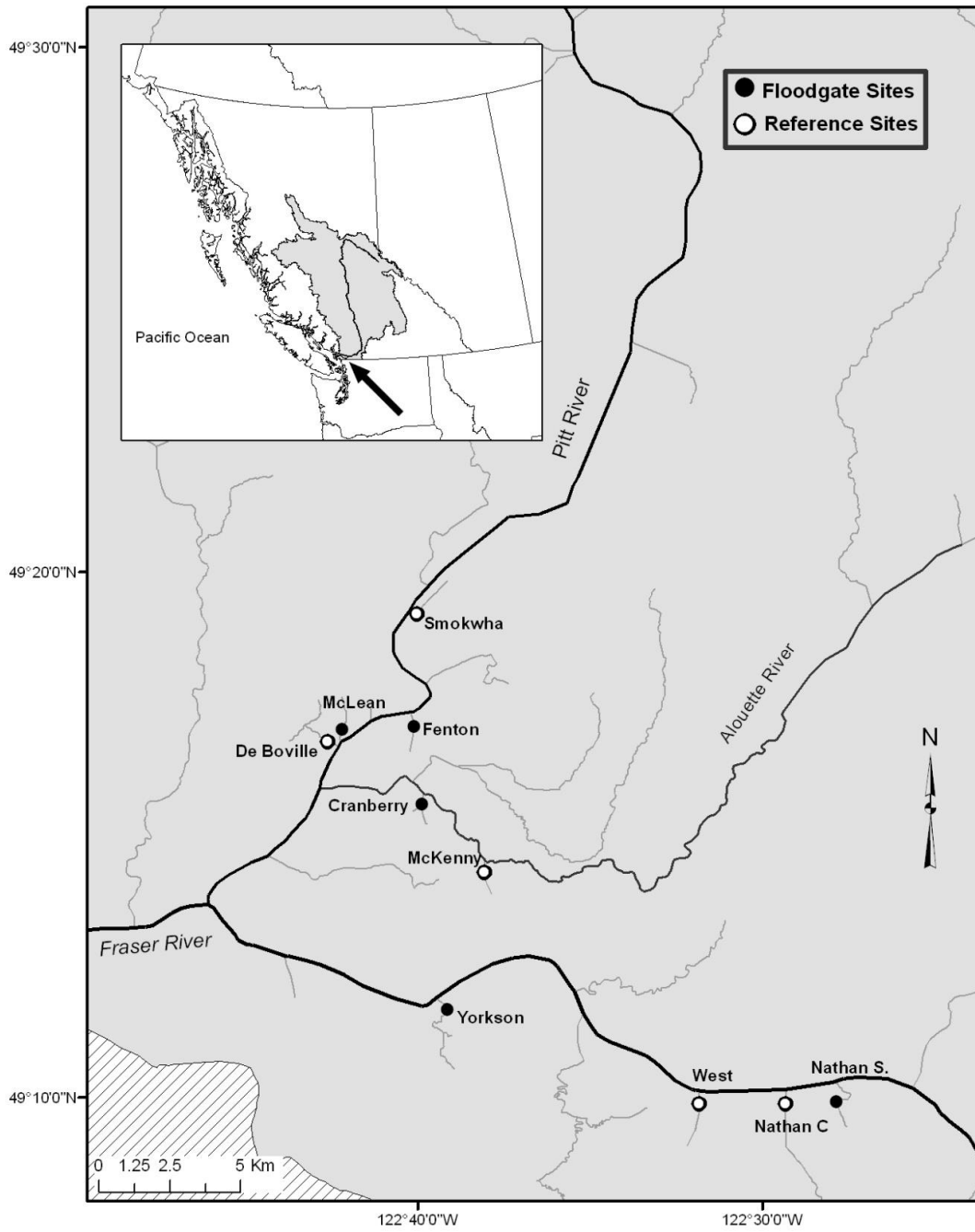
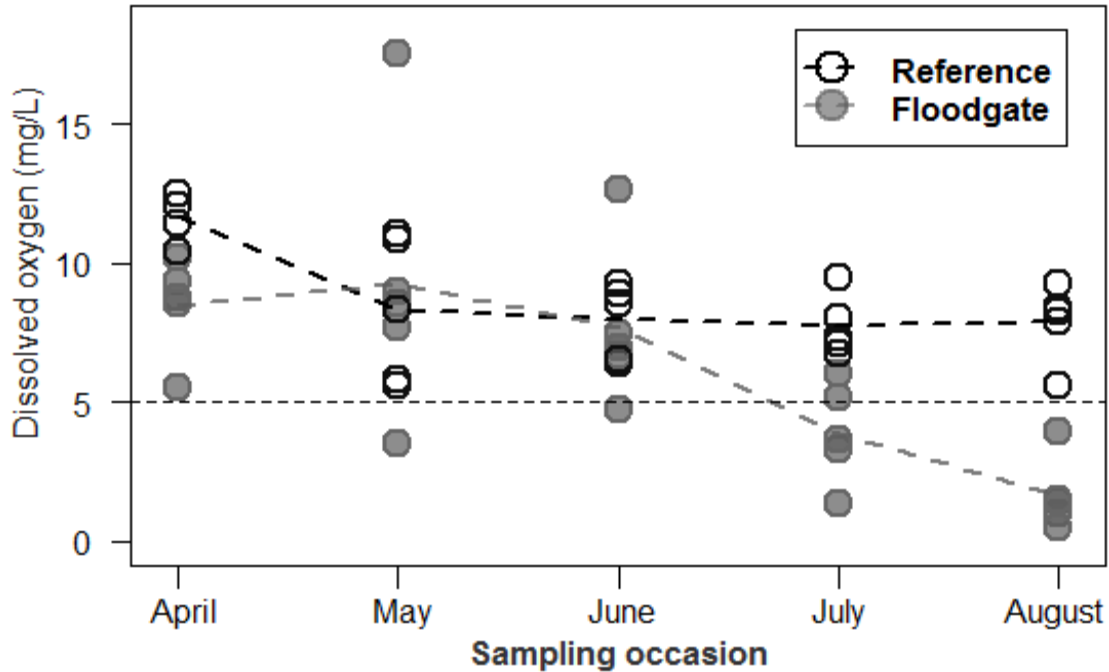
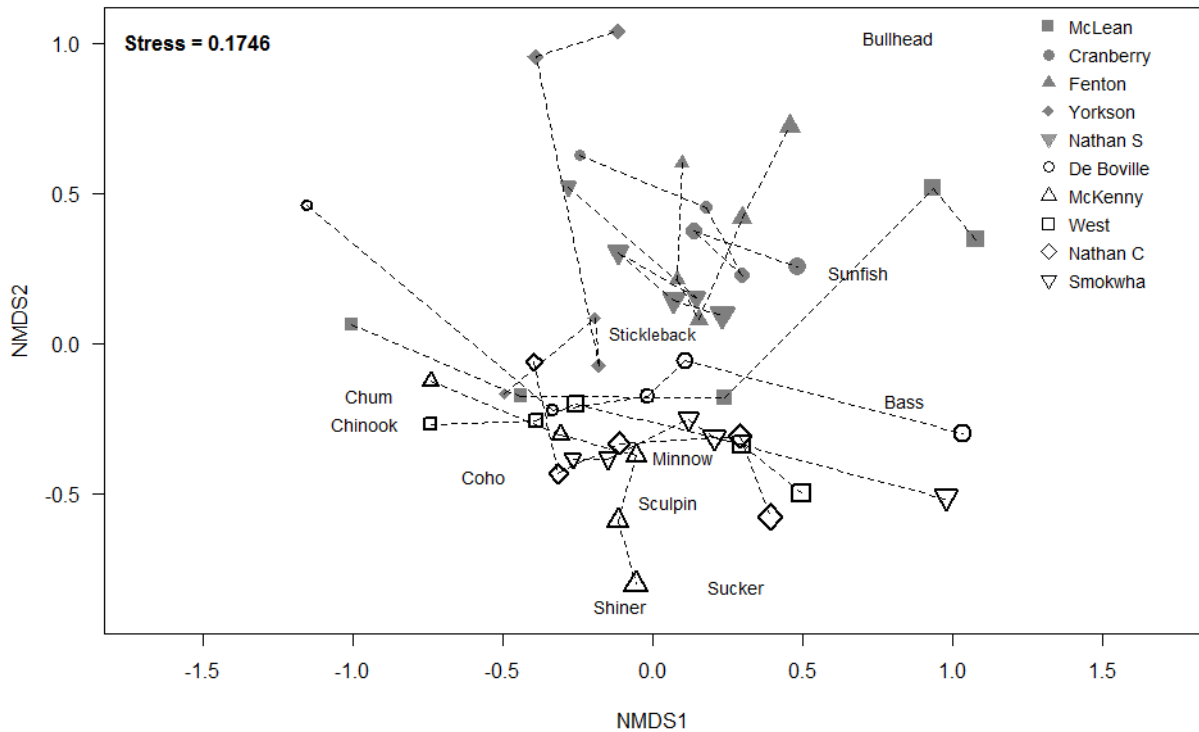


Figure 1. Location of reference and floodgate sites denoted by white and black circles respectively, within the lower Fraser River watershed which is outlined in

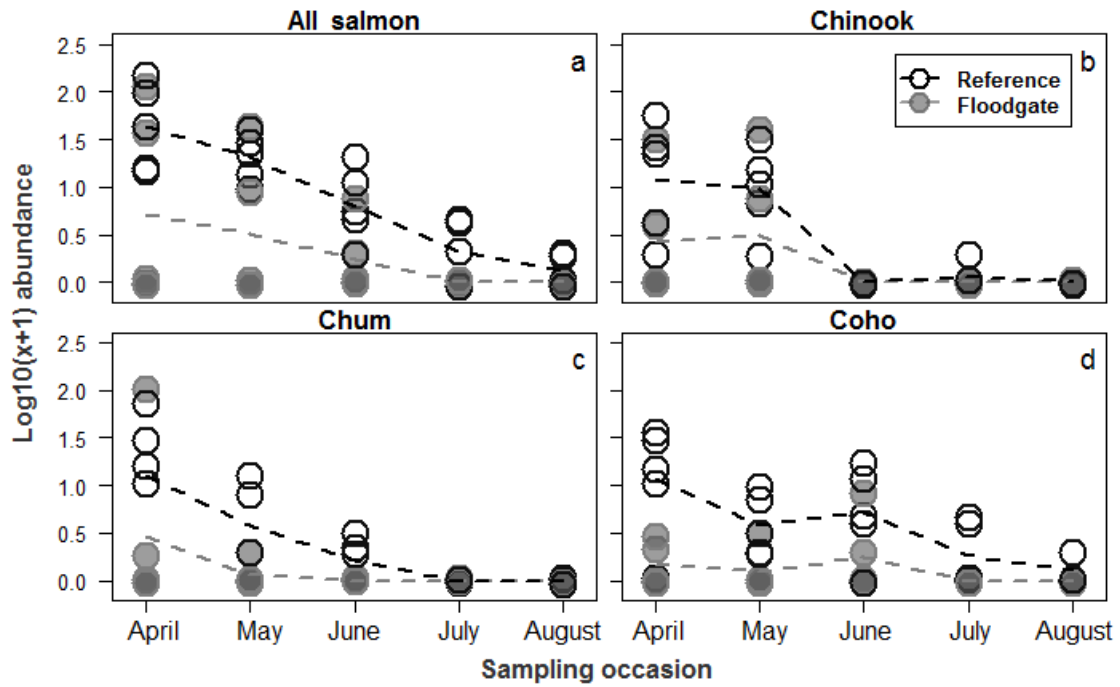


grey. Inset displays location of Fraser River watershed in western North America

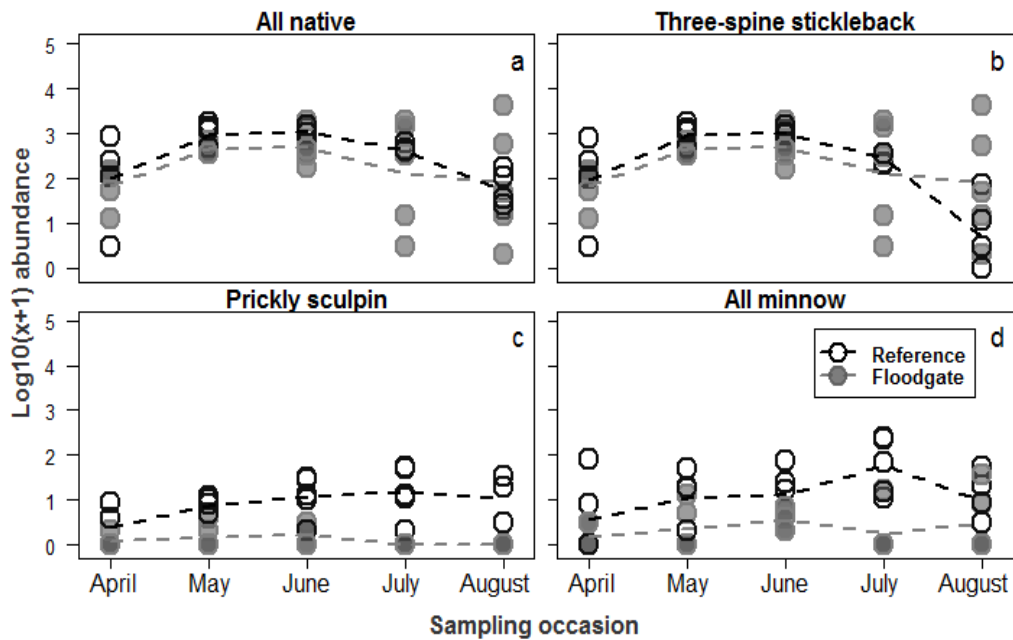
Figure 2. Monthly individual surface measurements of dissolved oxygen concentrations taken at each site on each sampling occasion. Measurements were taken just below the surface at noon or within thirty minutes, just upstream of floodgates or the confluence in reference sites. The horizontal dotted line at 5mg/L represents the instantaneous minimum dissolved oxygen concentration outlined by the Government of British Columbia's recommended criterion for the protection of aquatic life.



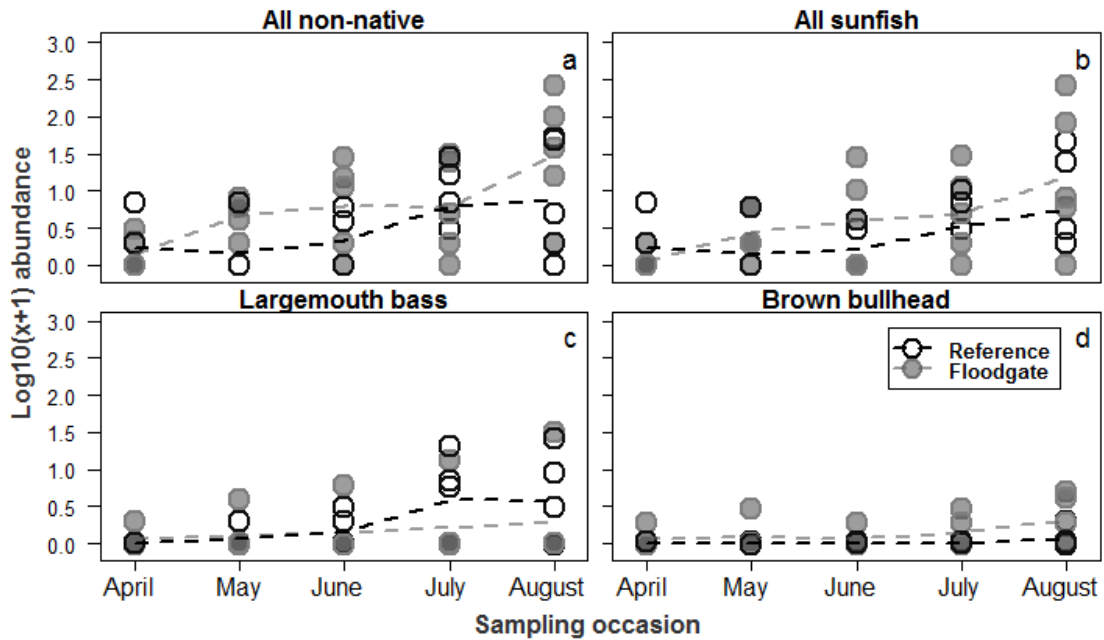
**Figure 3. Non-metric multidimensional scaling plot created using data for all fish species captured throughout our sampling. Unidentified juvenile minnows are grouped with peamouth chub and northern pikeminnow under the category minnow. Unidentified juvenile sunfish are grouped with pumpkinseed and black crappie under the category sunfish. Each point represents one sampling occasion for one site, grey and black colouring indicates reference and floodgate sites respectively, and size of points scales from beginning to end of sampling period going from smallest to largest. Position of points is relative to Bray-Curtis dissimilarity matrix generated from our catch data, position of species names represent weighted average scores of species for ordination configuration. The stress score indicates the degree to which the ordination explains the dissimilarity matrix in two dimensions.**



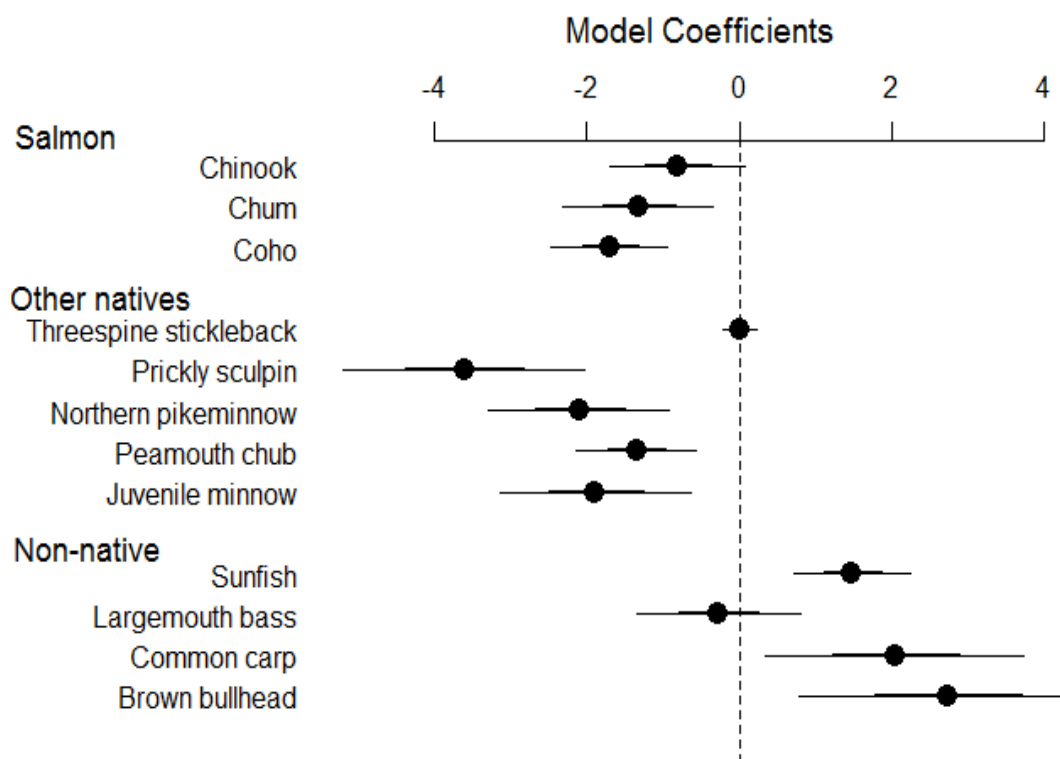
**Figure 4. Abundance data after  $\log_{10}(x + 1)$  transformation of a) all juvenile salmon species combined, b) juvenile Chinook, c) juvenile chum and, and d) juvenile coho salmon. Points represent the sum of three seine hauls and six minnow traps for an individual site for each sampling occasion with black open circles representing reference sites and grey full circles representing floodgate sites. Dotted lines connect means across sites for floodgate and reference sites on each sampling occasion.**



**Figure 5. Abundance data after  $\log_{10}(x + 1)$  transformation of a) all native species combined excluding salmon species, b) threespine stickleback, c) prickly sculpin, and d) all minnow species (northern pikeminnow, peamouth chub, reidside shiner and un-identified juvenile minnows combined). Points represent the sum of three seine hauls and six minnow traps for an individual site for each sampling occasion with black open circles representing reference sites and grey full circles representing floodgate sites. Dotted lines connect means across sites for floodgate and reference sites on each sampling occasion.**



**Figure 6. Abundance data after  $\log_{10}(x + 1)$  transformation of a) all non-native species combined, b) all sunfish (pumpkinseed, black crappie and un-identified juvenile sunfish combined), c) largemouth bass, and d) brown bullhead. Points represent the sum of three seine hauls and six minnow traps for an individual site for each sampling occasion with black open circles representing reference sites and grey full circles representing floodgate sites. Dotted lines connect means across sites for floodgate and reference sites on each sampling occasion.**



**Figure 7. Points representing model coefficients for the effect of floodgate presence on abundance of each fish species. More positive values indicate larger positive impacts of floodgates on fish abundance, more negative values indicate more negative impacts of floodgates on fish abundance. Data were normalized by division by the standard deviation for each species prior to analysis; the model coefficients thus indicate the impact of floodgate relative to observed variation of that species. Data coefficients are derived from generalized additive models for the effect of floodgates on abundance data with a smoothing function for the effect of date. Error distributions used for salmon and non-salmon species data were quasipoisson and negative binomial respectively out of necessity to satisfy normality. The thick and thin lines represent 1 and 2 standard errors for these estimates respectively.**

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