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THE EFFECTS OF FLOW REDUCTION RATES ON FISH STRANDING IN BRITISH COLUMBIA, CANADA

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ABSTRACT

Juvenile fish can strand in pools or in interstitial spaces when the water elevation drops in regulated rivers due to flow reductions. Three years of summer and winter experiments on the Columbia and Kootenay Rivers (Canada) assessed the effect of the rate of change in water level (ramping rate) on the probability of pool and interstitial stranding for juvenile (<100 mm) fish. The factors of wetted history of the site, time of day, natural fish density and the occurrence of a conditioning reduction prior to the operational reduction were also examined for their effect on stranding. Experimental net pens were constructed to test these factors *in situ* in the varial zones of the two rivers. Linear models with plausible additive combinations of the potential explanatory factors and a null model were fitted to the logistically transformed data and ranked using the second-order Akaike Information Criterion (AIC_c). The null model was the top ranked model for the interstitial stranding analyses, highlighting that none of the factors tested were significant variables in predicting the probability of stranding. Natural fish density, wetted history of the site, ramping rate and the presence of a conditioning reduction were variables included in the top three ranking models for the pool stranding analyses. Probability of pool stranding in summer was reduced by the occurrence of a conditioning reduction prior to the operational reduction. Higher natural fish density, longer periods of wetted history and higher ramping rates all led to higher probabilities of pool stranding. Copyright © 2008 John Wiley & Sons, Ltd.

KEY WORDS: hydro-peaking; fish stranding; hydro-electric dam; Columbia River; Kootenay River; AIC_c; conditioning reduction

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INTRODUCTION

Although rivers and streams are naturally highly heterogeneous environments, the biota below dams are often poorly adapted to the short-term, recurring disturbances that can result from hydro-electric dam operation (Berland *et al*., 2004). The stranding of juvenile fish due to rapid reductions in river height is one of several ecological problems associated with dam operations (Cushman, 1985; Bragg *et al*., 2005).

Fish stranding can occur in either the pools that remain when the water level drops or in the interstices on cobble or gravel banks or bars (Bradford, 1997). Factors shown to influence the rate of stranding include ramping rate (the rate of stage change in the river) (Bradford *et al*., 1995), time of day (Bradford *et al*., 1995), water temperature and season (Saltveit *et al*., 2001). Numerous previous studies have explored the impact of these various factors on the stranding of juvenile fish, but have generally only addressed the effects on salmonid species (Woodin, 1984; Hunter, 1992; Bradford *et al*., 1995; Bradford, 1997; Harby *et al*., 1999; Halleraker *et al*., 1999; Saltveit *et al*., 2001). Exceptions include studies on the Hanford Reach of the Columbia River and a study on the Mississippi River. Hoffarth *et al*. (1997) assessed stranding rates of all species in the Hanford reach, but primarily focussed on the response of Chinook salmon (*Oncorhynchus tshawytscha*). Experimental studies in the Mississippi River system addressed warm water species such as shovelnose sturgeon (*Scaphirhynchus platorynchus*), blue catfish (*Ictalurus furcatus*) and largemouth bass (*Micropterus salmoides*) (Adams *et al*., 1999). These authors found large variation in stranding rates among species but no significant effect of ramping rate. However, the Mississippi River study

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measured the impact of boat wakes, so the ramping (drawdown) rates were very rapid (0.21–0.76 cm s^{-1}) and not directly comparable to those associated with dam operations.

Studies have been conducted in the field environment to achieve more reality and within artificial flume channels to attain more control over the varied environmental factors that may affect stranding rate (e.g. Halleraker *et al*., 2003; Scruton *et al*., 2003). The goal of much of the research in this field has been to define the operational guidelines for hydro-electric facilities to minimize the risk of stranding (Bradford, 1997).

The experiments reported here were conducted on resident fish species of the Columbia and Kootenay Rivers in British Columbia, Canada. The experiments were conducted in winter and summer seasons over 3 years (2004–2006) below hydro-electric dams. Each of the phases of the study programme focussed on small-bodied fish (<100 mm) of various species including, juvenile rainbow trout (*Oncorhynchus mykiss*), longnose dace (*Rhinichthys cataractae*), northern pikeminnow (*Ptychocheilus oregonensis*), sculpin (*Cottus* spp.), sucker (*Catostomus* spp.) and Umatilla dace (*Rhinichthys umatilla*).

The objective of this series of studies was to assess the effect of the various factors of flow reduction on the probability of stranding for all species of juvenile fish combined on the Columbia and Kootenay Rivers. The rate of stranding in pool and interstitial habitat was assessed in relation to the factors of: ramping rate, time of day, wetted history, overall cover, natural fish density and conditioning flow reduction. Conditioning flows are a new procedure in flow regulation that was an attempt to minimize stranding through different operational protocols for hydro-electric facilities. They consist of a rapidly decreasing flow followed by a rapid increase to the original river stage after approximately 1 h within 24 h of a planned major flow reduction. Fish stranded during a conditioning flow experience low mortality rates due to a shorter dewatered time span (Hessevik, 2002). Stranding rates during flow reductions with or without a preceding conditioning flow were compared.

METHODS

Study area

The Columbia River is 2044 km long and has a mean discharge of approximately 7500 $m^3 s^{-1}$. It is the largest hydro-electric producing river in North America with 14 dams on its mainstem and several dams on its tributaries. The Kootenay River (spelled Kootenai in the U.S.) is the largest tributary of the Columbia and is 731 km long with a mean discharge of 838 $m^3 s^{-1}$. Dams on the Columbia and Kootenay Rivers regulate the flows in the lower Kootenay and Columbia Rivers near the Canada–U.S. border which is the site of the current study (Figure 1). Hugh L. Keenleyside Dam (HLK) is located on the Columbia River 10 km upstream of Castlegar, British Columbia (Figure 1) while the lower Kootenay River system has a series of dams regulating the flow with Brilliant Dam (BRD) as the final dam before the confluence with the Columbia River (Figure 1).

The combined flow regulation from HLK and the Kootenay River facilities impacts downstream fish populations. HLK Dam primarily responds to flow requests by downstream stakeholders in the U.S. who aim to maximize power generation and control flooding. Due to the size of the system, flow changes from HLK typically occur on a bi-weekly basis. The flows in the Kootenay River are primarily regulated to meet peak power demands for the region. Consequently, the flow on the lower Kootenay changes on a daily basis. This primarily results from load shaping operations at the Kootenay Canal facility, but is also due to limited hydro-peaking operations at BRD. BRD will generate significantly more peak power following the completion of an approved expansion project, scheduled for completion in fall 2007.

The effect of hydro-electric dam operation on fish stranding rates was studied at multiple sites located on the Columbia River between HLK Dam and Kinnaird Rapids (11 km), and on the Kootenay River downstream of BRD to the confluence (Figure 1). The primary sampling location for the interstitial stranding experiments was the left upstream bank (LUB) of the Kootenay River though sampling was also done at: Kootenay River (LUB), Kootenay River (RUB), and Kinnaird Rapids (Figure 1). Norn’s Creek Fan was the primary sampling location for the pool stranding experiments though three other locations were used for pool stranding experiments: downstream of Robson Bridge (RUB), CPR Island, Tin Cup Rapids (RUB) and Kinnaird Rapids.

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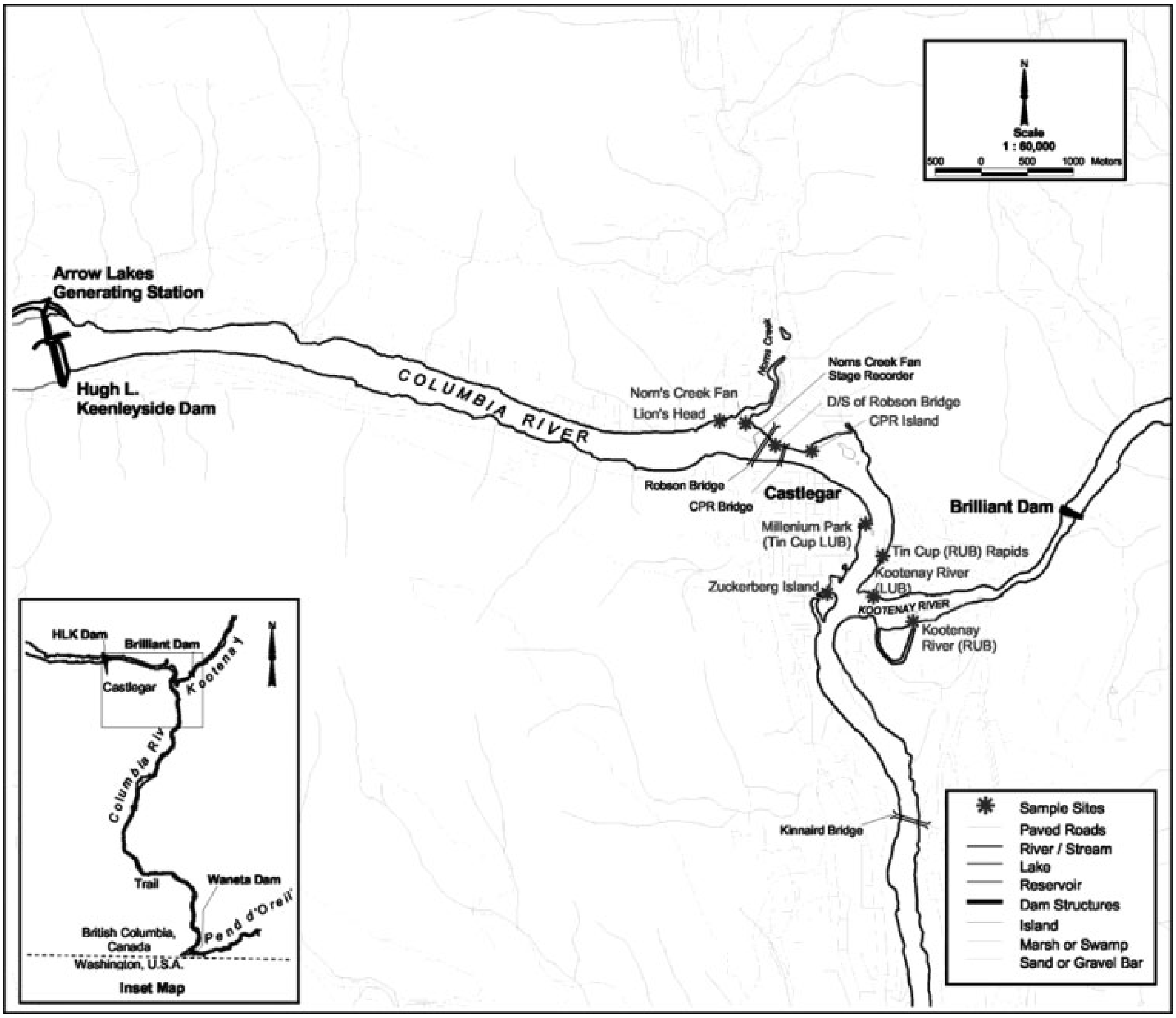

Figure 1. Study sites on the lower Kootenay and Columbia Rivers for ramping rate reduction experiments conducted from 2004 to 2006

Experimental reductions and ramping rates

The magnitude of each reduction was planned to provide enough dewatered area in the varial zone to be sampled for the interstitial experiments or to isolate the stranding pool from the escape pool in the pool stranding experiments. A stranding pool is a pool that becomes completely separated from the mainstem as a result of the flow reduction so that fish are unable to escape from it unless water levels are raised. An escape pool is a pool into which they can swim from the stranding pool and which is still connected to the main river channel. The reduction for most of the experiments was 12.5% of the combined flow at Birchbank (28.0 km) on the Columbia River, where the flow at Birchbank is the discharge from HLK plus BRD spill and discharge.

For the pool stranding experiments in which the conditioning reduction concept was tested, it was found that the 2-h window afforded by the 12.5% reduction was insufficient for complete isolation of the sample pool. As a result, the magnitude was increased to $\sim$20% for all remaining pool experiments to effectively isolate the sample pools.

River stage data during the experiments were recorded every 15 min by BC Hydro's stage recorder at Norn's Fan (7.5 km, Figure 1). Data were also recorded onsite at 30 s intervals by a Solinist^{TM} Levelogger 3001 and a Solinist^{TM} Barologger 3001 (used for atmospheric correction of the stage data).

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Ramping rates were calculated as the average stage reduction rate (cm h^{-1}) over the first 80% of the entire reduction. A moving 15 min average was applied to the Solinist^{TM} Levelogger 3001 data in order to smooth fluctuations caused by recording noise. All calculations were programmed in R (V 2.4.1, R 2006). Ramping rates ranged from 3.9 to 35.3 cm h^{-1} over the 3 years of experiments.

Field methods

Pool experiments were predominantly done in summer due to available habitat. Typical winter water levels do not provide pool habitats suitable for sampling. Twenty-eight pool net pen experiments were done in the summer seasons of 2005 and 2006. Ten interstitial net pen experiments were done in the summers of 2004 and 2005. Only winter interstitial experiments were conducted after 2005 since there was virtually no interstitial stranding in the summer (Golder, 2005; Golder, 2006). Twenty-four interstitial net pens were sampled throughout the three winter seasons (2004–2006) of the study.

All net pen experiments used an overall enclosure study design based on the methodologies of Saltveit *et al*. (2001). For the interstitial experiments, a triangular shaped net pen enclosure was constructed along the shoreline. The base of the triangular enclosure was aligned along the upper wetted edge of the river channel with the peak of the triangle positioned towards the downstream (or escape) end and in the deeper portion of the river channel. For each of the interstitial experiments, the determination of how many fish stranded and escaped began once the habitat within the net pen was sufficiently dewatered. Fish that were either in the fyke net (attached at the downstream end) or adjacent to the net pen walls were considered to have escaped while fish in the substrate were considered stranded. Measurements of net pen area, substrate types and parameters, channel slope and velocity measurements were also recorded.

For the pool experiments completed in 2005, a net pen enclosed the stranded pool area and the adjacent escape pool area, with the escape pool area positioned at the downstream end of the net pen. Due to concerns about affecting fish movements, experiments in 2006 were modified so that the escape area was not enclosed and fish were counted upon exit or entry to the stranding pool prior to the isolation of the stranding pool by two observers and one data recorder. Each observer enumerated unmarked fish and marked fish. When the upstream pool isolated, field crews would stop the behavioural monitoring and beach seine and electrofish the remaining fish from the pool. Measurements were made of wetted area, pool depth and size, and per cent cover. This methodology was deemed superior for not affecting behaviour, but precluded the identification of fish to the species level for the pool stranding experiments.

For both interstitial and pool stranding experiments, fish numbers, species, length, marked or unmarked, status of fish (escaped or stranded) and the overall health of the fish were recorded.

The explanatory variables tested varied through time as an adaptive experimental procedure was adopted over the 3 years of study (see Table I), but overall included: ramping rate, time of day, wetted history, natural fish density and conditioning reductions. Wetted history was defined as the period of time that the varial zone habitat was inundated with water prior to experimental dewatering ignoring conditioning reductions and was a continuous variable. Time of day was categorical and was either day or night. Natural fish density was continuous and was calculated as the number of fish m^{-2} of net pen. Conditioning reductions were categorically either present or absent and if present

Table I. Summary of variables assessed and methodologies used on the Columbia River and lower Kootenay River flow reduction and fish stranding studies, 2004–2006

Year	Season	Variables assessed	Sample methodology
2004	Winter	Ramping rate, time of day, wetted history	Interstitial
	Summer	Ramping rate, time of day, wetted history	Interstitial
2005	Winter	Ramping rate, time of day, wetted history	Interstitial
	Summer	Ramping rate, time of day, wetted history	Interstitial and pool
2006	Winter	Ramping rate, time of day, wetted history, natural fish density	Interstitial and pool
	Summer	Ramping rate, time of day, wetted history, natural fish density, conditioning reductions	Pool

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were a short, rapid reduction 24 h prior to the experimental reduction as defined above. Ramping rate is defined in the above experimental reduction and ramping rate methods section. Other factors that were considered to potentially affect the risk of stranding such as substrate type, fish habituation time after capture, shoreline gradient, substrate size, embeddedness and compaction were controlled as much as possible by site selection and experimental design.

Fish

Only wild fish were used in the experiments since hatchery and wild fish can exhibit qualitatively different responses to flow reduction (Saltveit *et al*., 2001). Within each experiment, two classes of fish were used: marked and unmarked. The unmarked fish were those naturally occurring in the varial zone or pool areas that were enclosed and entrapped by the placement of the net pen. The marked fish were captured using backpack electroshocking in an area far enough away from the net pen area that the shocking would not affect the naturally occurring fish in the net pen. Once captured, the fish were dyed with Bismarck brown Y (Ewing *et al*., 1990). After a recovery period of approximately 30 min, the dyed individuals were counted and released into the upstream end of the net pen or the pool that would become isolated. The marked fish were included in the experimental protocol to allow the capture efficiencies of the unmarked fish to be estimated.

After the stage reduction, marked and unmarked fish were recovered from the escaped and stranded zones in the net pens using electrofishing, beach seining and cobble searches. Since the number of marked fish in each net pen at the start of the stage reduction was known, the number of marked fish that were not recovered could be calculated. These unrecovered fish had either (1) escaped from the net pen undetected or (2) were still stranded in the net pen. The capture efficiencies for unmarked fish were then estimated assuming that either all of the unrecovered marked fish had escaped undetected from the net pen or all had remained stranded. For each of these extreme scenarios, the number of unmarked fish that had escaped or stranded was computed and the probability of stranding was then calculated.

Statistical analysis

For all analyses, the response variable was the number of unmarked fish stranded divided by the total number of fish available for stranding and all species were pooled together to obtain sufficient statistical power and due to the abovementioned methodological constraints on the pool stranding experiments. The species or species groups assessed within the pooled data were dominated by: longnose dace, rainbow trout, sculpin, sucker and Umatilla dace. The data were transformed using the logistic transformation and were tested to ensure that they were not statistically significantly different from normal before fitting linear models to the data. A suite of plausible candidate models were fitted to the data (*sensu* Burnham and Anderson, 2002) and ranked using the second-order Akaike Information Criterion (AIC_c). The model set included all one factor models and all additive models that made biological sense. No interactions were tested due to the low number of experiments. The data to which the models were fitted was weighted by the square root of the total number of fish in each experiment so that experiments in which very few fish were assessed for their behaviour did not influence the analysis to the same extent as experiments where fish were more numerous.

The interstitial stranding rate of fish in the lower Columbia and Kootenay Rivers from the 2004–2006 winter seasons was analysed to test the effect of the four factors of ramping rate, time of day, natural fish density and wetted history. A null model was also fitted to the data. The pool stranding rate of fish was assessed using data from the summers of 2005 and 2006. The factors of wetted history, conditioning status, ramping rate and fish density were tested for their effect on the stranding rate. A null model was also fitted to the pool stranding data. For each of the analyses completed, only those experiments with no missing data were able to be included in the analytic set given the constraints of the methods. Percentage of cover within the stranding pool was a variable that was planned to be included in the analysis of the pool stranding experiments, however, so many experiments had data missing for this variable, that it was not able to be included.

As described above, a correction was calculated where fish were assumed to have either: (1) stranded in the substrate in the net pen but remained undiscovered or (2) had escaped the study area. In the first case, the numerator was the number of marked fish in the net pen plus the number of missing fish and the denominator was the total

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number of marked seed fish. In the second case, the numerator was the number of marked fish in the net pen and the denominator was the number of marked fish in the net pen plus the number of marked fish in the escape area. There were no *a priori* reasons for selecting between the two assumptions regarding the fish that were unaccounted for in the interstitial stranding experiments, so the analyses were done and are presented for both assumptions. Due to the spatial structure of the pool stranding experiments, the most likely assumption was that the fish that were unaccounted for were stranded, therefore, only the analyses using this assumption is presented for pool stranding.

Relative importance was calculated *sensu* Burnham and Anderson (2002) for each of the variables included in the candidate model set. Relative importance is calculated as the sum of the Akaike weights for each of the models in which the variable is included (Burnham and Anderson, 2002) and is quite intuitive to interpret. The larger the relative importance, the more support there is for models including that variable. The maximum relative importance is a value of 1; if a variable has a relative importance of 1, the models without that variable included lack any support within the model set. However, if there is a relative importance of 0.50, it is important to note that the variable is only responsible for explaining half of the possible Akaike weight.

All analyses were completed using R version 2.4.1 (R, 2006).

RESULTS

The null model was the winning model for the interstitial stranding experiments with either the assumption that the unaccounted for fish had stranded or that they had escaped. For the model set making the assumption that the unaccounted for fish had stranded, the one factor model with wetted history as the explanatory variable was the second ranked model and the one factor model with ramping rate as the independent variable was the third ranked model (Table IIa). For the model set making the assumption that the unaccounted for fish had escaped, the one factor model with natural fish density as the explanatory variable was ranked second and the one factor model with ramping rate as the independent variable was ranked third (Table IIb). The R^2 values for all models fitted to the interstitial stranding data with either assumption were extremely low with a mean value of 0.142 and a range from 0 to 0.36 (Tables IIa,b).

Table IIa. The candidate model set and AIC_c rankings for the interstitial stranding experiments from the winters of 2004, 2005 and 2006. In addition, R^2 values to assess model fit, the number of experiments in analytic data set, the number of parameters in a particular model, the log likelihood value, the delta AIC_c (the arithmetic difference in the AIC_c values of the top model and all other models) and the AIC_c weight of each model are all reported. This candidate model set was run on the data with the assumption that the fish that were unaccounted for had all stranded

Rank	Model	AIC_c	R^2	n	p	Log likelihood	Delta AIC_c	AIC_c weight
1	Null	58.21	0	14	2	−26.56	0	0.306
2	Wetted history	59.36	0.14	14	3	−25.48	1.15	0.172
3	Ramping rate	59.94	0.11	14	3	−25.77	1.73	0.129
4	Fish density	60.34	0.08	14	3	−25.97	2.13	0.106
5	Time of day	61.52	0	14	3	−26.56	3.31	0.059
6	Ramping rate + fish density	61.64	0.24	14	4	−24.60	3.42	0.055
7	Wetted history + ramping rate	61.86	0.23	14	4	−24.71	3.65	0.049
8	Wetted history + fish density	62.10	0.22	14	4	−24.83	3.89	0.044
9	Wetted history + time of day	63.41	0.14	14	4	−25.48	5.19	0.023
10	Time of day + ramping rate	63.96	0.11	14	4	−25.76	5.75	0.017
11	Ramping rate + wetted history + fish density	64.37	0.36	14	5	−23.44	6.16	0.014
12	Time of day + fish density	64.37	0.08	14	4	−25.97	6.17	0.014
13	Ramping rate + time of day + fish density	66.69	0.24	14	5	−24.60	8.48	0.004
14	Ramping rate + wetted history + time of day	66.92	0.23	14	5	−24.71	8.71	0.004
15	Time of day + wetted history + fish density	67.13	0.22	14	5	−24.82	8.92	0.004
16	Ramping rate + wetted history + time of day + fish density	70.87	0.36	14	6	−23.44	12.66	0.001

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Table IIb. The candidate model set and AIC_c rankings for the interstitial stranding experiments from the winters of 2004, 2005 and 2006. In addition, R^2 values to assess model fit, the number of experiments in analytic data set, the number of parameters in a particular model, the log likelihood value, the delta AIC_c (the arithmetic difference in the AIC_c values of the top model and all other models) and the AIC_c weight of each model are all reported. This candidate model set was run on the data with the assumption that the fish that were unaccounted for had all escaped

Rank	Model	AIC_c	R^2	*n*	*p*	Log likelihood	Delta AIC_c	AIC_c weight
1	Null	72.80	0	19	2	−34.03	0	0.236
2	Fish density	72.85	0.14	19	3	−32.62	0.05	0.230
3	Ramping rate	74.26	0.07	19	3	−33.34	1.50	0.112
4	Time of day	74.97	0.04	19	3	−33.68	2.17	0.080
5	Time of day + fish density	75.22	0.18	19	4	−32.18	2.42	0.070
6	Ramping rate + fish density	75.55	0.16	19	4	−32.34	2.75	0.060
7	Wetted history	75.64	0.01	19	3	−34.02	2.84	0.057
8	Wetted history + fish density	76.00	0.14	19	4	−32.57	3.20	0.048
9	Time of day + ramping rate	76.95	0.10	19	4	−33.05	4.15	0.030
10	Wetted history + ramping rate	77.55	0.07	19	4	−33.34	4.75	0.022
11	Wetted history + time of day	78.23	0.04	19	4	−33.68	5.43	0.016
12	Ramping rate + time of day + fish density	78.50	0.20	19	5	−31.94	5.70	0.014
13	Time of day + wetted history + fish density	78.92	0.18	19	5	−32.15	6.12	0.011
14	Ramping rate + wetted history + fish density	79.23	0.17	19	5	−32.31	6.43	0.009
15	Ramping rate + wetted history + time of day	80.71	0.10	19	5	−33.05	7.91	0.005
16	Ramping rate + wetted history + time of day + fish density	82.86	0.20	19	6	−31.93	10.05	0.002

Given the structure of the pool stranding net pens and the vegetative and depth cover available in some of the experimental isolated pools, the most reasonable assumption *a priori* was that the fish that were unaccounted for in the experiments were stranded, so only the results reflecting that assumption are presented. The top ranked model was the two factor additive model that included the variables of wetted history and natural fish density. The second and third ranked models were within 4 AIC_c values of the top model so cannot be discounted (Burnham and Anderson, 2002) and included the factors of conditioning reduction and ramping rate (Table III). Probability of pool stranding in summer was reduced by the occurrence of a conditioning reduction prior to the operational reduction (Figure 2a). Higher natural fish density, longer periods of wetted history and higher ramping rates all led to higher probabilities of pool stranding (Figure 2b–d).

The pool stranding variables of natural fish density and wetted history showed high relative importance (>0.95), and the variables of conditioning reduction and ramping rate had relative importance values of 0.18 and 0.16, respectively (Table IV). For the interstitial experiments, the relative importance is calculated for each of the two assumptions. In the case where the assumption is that all unaccounted for fish have actually escaped, natural fish density has the highest relative importance (0.44), ramping rate is second (0.252) and the null model ranks third with a value of 0.236. In the case where the assumption is that all unaccounted for fish have stranded, wetted history has the highest relative importance (0.31), the null model ranks second (0.306) and ramping rate ranks third (0.273) (Table IV).

DISCUSSION

It is clear from the very low R^2 values obtained for the interstitial stranding models and the fact that the null model was the top-ranked model for all interstitial experiments that the model set and the explanatory variables tested do not adequately capture the structure in the data from the interstitial stranding experiments. Neither the assumption that the unaccounted for fish had stranded nor the assumption that they had escaped altered the result of a top ranking for the null model. Interstitial stranding is not explained by any of the variables tested.

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Table III. The candidate model set and AIC_c rankings for the pool stranding experiments from the winters of 2004, 2005 and 2006. In addition, R^2 values to assess model fit, the number of experiments in analytic data set, the number of parameters in a particular model, the log likelihood value, the delta AIC_c (the arithmetic difference in the AIC_c values of the top model and all other models) and the AIC_c weight of each model are all reported. This candidate model set was run on the data with the assumption that the fish that were unaccounted for had all stranded

Rank	Model	AIC_c	R^2	n	p	Log likelihood	Delta AIC_c	AIC_c weight
1	Fish density + wetted history	105.80	0.59	23	4	−47.79	0	0.663
2	Fish density + conditioning reduction + wetted history	108.86	0.59	23	5	−47.66	3.05	0.143
3	Fish density + ramping rate + wetted history	109.02	0.59	23	5	−47.75	3.25	0.131
4	Fish density + ramping rate + conditioning reduction + wetted history	112.54	0.59	23	6	−47.66	6.77	0.022
5	Fish density	112.76	0.36	23	3	−52.75	6.99	0.020
6	Fish density + conditioning reduction	113.84	0.41	23	4	−51.81	8.06	0.012
7	Fish density + ramping rate	115.66	0.37	23	4	−52.72	9.88	0.005
8	Fish density + ramping rate + conditioning reduction	117.08	0.42	23	5	−51.79	11.31	0.002
9	Null	120.52	0	23	2	−57.96	14.75	0
10	Wetted history	120.85	0.10	23	3	−56.79	15.08	0
11	Ramping rate	121.64	0.06	23	3	−57.19	15.87	0
12	Ramping rate + wetted history	121.76	0.17	23	4	−55.77	15.98	0
13	Conditioning reduction	122.7455	0.02	23	3	−57.74	16.97	0
14	Conditioning reduction + wetted history	123.792	0.10	23	4	−56.79	18.02	0
15	Ramping rate + conditioning reduction	124.4816	0.07	23	4	−57.13	18.71	0
16	Ramping rate + conditioning reduction + wetted history	124.945	0.18	23	5	−55.71	19.17	0

The independent variables selected for inclusion in the model set were based on previous work on hydro-peaking in Norway (e.g. Saltveit *et al*., 2001; Halleraker *et al*., 2003) and in North America (e.g. Bradford, 1997; Hoffarth *et al*., 1997), but they did not explain much of the variability in the interstitial stranding on the lower Columbia and Kootenay Rivers. There have been incidents of high levels of interstitial juvenile fish stranding on the lower Columbia and Kootenay Rivers. However, only the initial interstitial net pen experiment in the winter of 2004 has shown an equivalently high rate of stranding as the major events recorded for the area (Golder, 2005). The modelling output for the interstitial stranding demonstrates that there is either a key factor that is not in the model that is very important or that the variability in the numbers of fish that strand is high and so random as to be essentially unpredictable. It is also important to note that the experimental issue of finding each fish that is interstitially stranded is quite difficult and is likely a contributing factor to the variability.

In contrast, for the pool stranding experiments, natural fish density, wetted history of the site, ramping rate and the presence of a conditioning reduction were included in the top three ranking models for the pool stranding analyses and the R^2 values for the models were substantially higher than for the interstitial experiments. Although the R^2 values are just an indication of model fit, it does indicate that the explanatory variables used in this model set were much more able to capture the variability in the data than in the interstitial experiments. The trends for the probability of pool stranding in summer were for a reduction due to the occurrence of a conditioning reduction and an increase in stranding due to high fish density, longer periods of wetted history and higher ramping rates.

The ramping rates tested in this series of studies were generally lower than those tested in other studies and ranged from 3.9 to 13.3 $\mathrm{cm\,h^{-1}}$ for the interstitial stranding experiments and from 7.4 to 35.3 $\mathrm{cm\,h^{-1}}$ for the pool stranding experiments. This lower range of rates was due to the maximum rate at which flows could operationally change from HLK Dam and to the hydraulic conditions upstream of the study sites which tended to dampen the observed rate of water changes at the sites where fish were stranded. This is in comparison to rates as high as 105 $\mathrm{cm\,h^{-1}}$ tested in other studies (Hessevik, 2002). In general, past studies have demonstrated a decrease in the rate of stranding with slower ramping rates and particularly with ramping rates less than 10 $\mathrm{cm\,h^{-1}}$ (Hessevik, 2002; Halleraker *et al*., 2003). This trend is borne out by our results for pool stranding (Figure 2d) though it is

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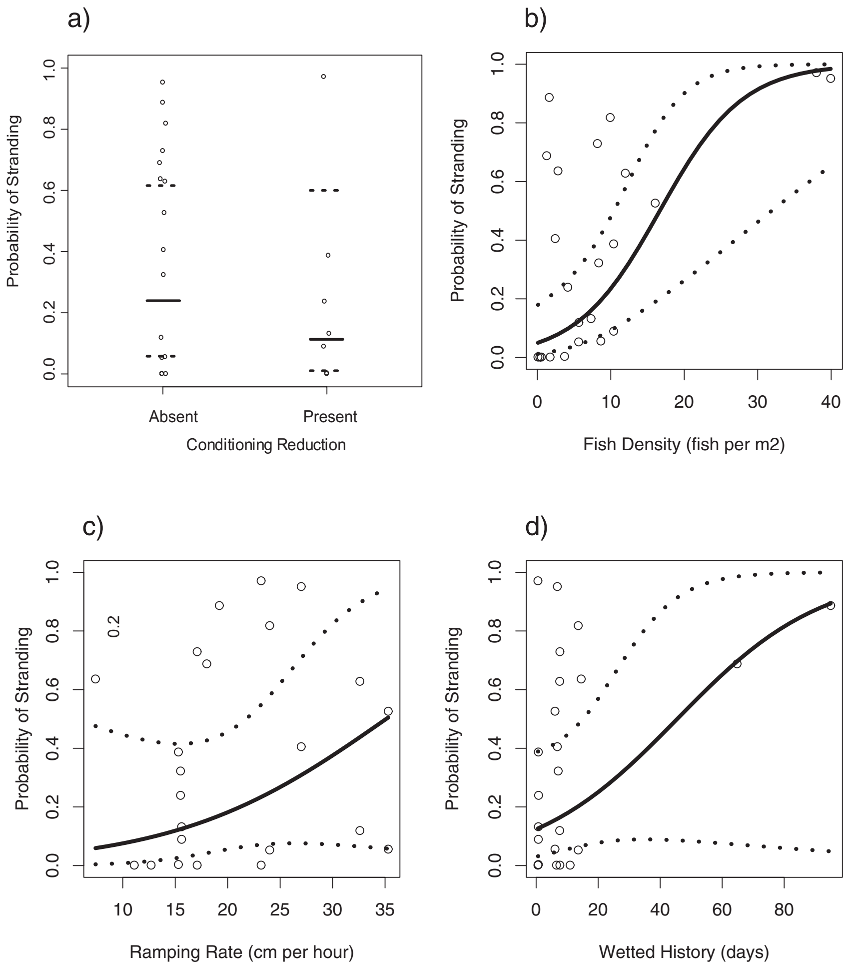

Figure 2. Probability of pool stranding in the summers of 2005 and 2006 (all species combined) as a function of (a) conditioning reduction (present or absent), (b) natural fish density (fish m^{-2}), (c) wetted history (days) and (d) ramping rate (cm h^{-1}). Mean predicted values from the one factor model for each variable are shown in the solid line and upper and lower 95% confidence bounds for each model are represented by the dashed lines

important to note that ramping rate was never included in the top model for either interstitial or pool stranding and had a relative importance less than 0.27 for all situations (Table IV). It has been suggested by Olson (1990) that a ramping rate of less than 2.5 cm h^{-1} would prevent stranding more universally, but as increasing information about species differences in behavioural responses emerges (Scruton *et al*., 2003; Flodmark *et al*., 2006) and as more is learned about the site specific characteristics that influence ramping (e.g. Bradford, 1997), it seems likely that experimentally determined ramping rates will be only locally applicable (Flodmark, 2004).

Previous studies have found contradictory results for the effect of time of day on stranding. Some studies have found higher stranding during the day when concealment behaviour in the substrate is prevalent amongst juvenile fish (Bradford *et al*., 1995), yet other studies have found higher stranding during the night (Hamilton and Buell, 1976; Bradford, 1997). Our analyses showed no effect of time of day for interstitial stranding experiments though this could be either a genuine lack of effect in the Columbia and Kootenay systems, or artifactual as there were few night-time experiments completed and the data are, therefore, heavily unbalanced in favour of daytime experiments. For the pool stranding experiments, the wetted history variable was missing for the two experiments that were conducted at night which precluded the testing of time of day on stranding rates.

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Table IV. The relative importance for each variable for pool and interstitial stranding experiments. Assumptions are explicitly stated and the relative importance is calculated as the sum of the weights of each model in which the variable has been listed

Stranding type	Assumption	Variable	Relative importance
Pool	Fish unaccounted for were considered stranded	Natural fish density	0.99
		Wetted history	0.96
		Conditioning reduction	0.18
		Ramping rate	0.16
		Null model	0
Interstitial	Fish unaccounted for were considered escaped	Natural fish density	0.44
		Ramping rate	0.25
		Null model	0.24
		Time of day	0.23
		Wetted history	0.17
	Fish unaccounted for were considered stranded	Wetted history	0.31
		Null model	0.31
		Ramping rate	0.27
		Natural fish density	0.24
		Time of day	0.13

Fish density did not emerge in the top model for predicting the degree of interstitial stranding risk, but the variable's relative importance was high. The top model for predicting the probability of pool stranding included fish density and wetted history and these two variables had relative importance values close to the maximum of one as a result. The effect of fish density in previous experiments has been found to be minimal though there have been hypotheses developed rationalizing both the negative and the positive impact of density on stranding (Hessevik, 2002). For example, less fish in an area could lead to lower stranding as more of the fish would have adequate locations in which to stay when the water dropped. Although, there was a positive relationship between the probability of stranding and the natural fish density for pool stranding (Figure 2b), the strong trend is mainly driven by only two experiments and should be considered preliminary. This variable was assessed out of interest, but there is little that can be managed with regards to fish density unless another variable can control the number of fish in the nearshore area.

Operationally, a conditioning reduction is a tenable strategy for hydro-electric dams and the trend for decreased risk of stranding with a conditioning reduction is encouraging and should be further investigated. These preliminary conditioning reduction results suggest that short-term exposure to flow fluctuations could possibly reduce the stranding of juvenile fish in the Columbia River system. These experiments need to be repeated during different seasons and habitats to determine if these results are consistent, particularly since they are in opposition to the results obtained previously by Hessevik (2002) where individually tagged fish in an experimental stream channel did not demonstrate a learned escape response to dewatering after multiple experiments.

Wetted history for interstitial experiments had a relative importance of 0.31 when the assumption that all unaccounted for fish had stranded, and was only 0.17 when the opposing assumption was made. It was extremely important for the pool stranding experiments with a value of 0.96 and inclusion in the top model. The trend for both interstitial and pool stranding experiments was for increased probability of stranding with longer periods of wetted history. It has previously been suggested that after longer periods with stable flows, a gentle and slow drop in discharge be done to reduce stress and possible sub-lethal effects on juvenile fish (Halleraker *et al*., 2003). An alternate approach may be to drop the flows rapidly, but for a short duration to minimize mortality of juvenile fish in order to condition them to move out the areas of highest risk for stranding. This may not be an effective strategy since the time to death for juvenile fish can be quite variable and can occur in less than 30 min (Hessevik, 2002), but depending on temperature and the type of stranding, it may be a potential management tool. Only further experimentation may determine if this procedure is effective in multi-faceted conditions at reducing the probability of stranding.

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