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EVALUATING UNCERTAINTY IN PHYSICAL HABITAT MODELLING IN A HIGH-GRADIENT MOUNTAIN STREAM

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ABSTRACT

Predictions of habitat-based assessment methods that are used to determine instream flow requirements for aquatic biota are uncertain, but instream flow practitioners and managers often ignore those uncertainties. Two commonly recognized uncertainties arise from (i) estimating the way in which physical habitat within a river changes with discharge and (ii) the suitability of certain types of physical habitat for organisms. We explored how these sources of uncertainty affect confidence in the results of the British Columbia Instream Flow Methodology (BCIFM), which is a commonly used transect-based habitat assessment tool for small-scale water diversions. We calculated the chance of different magnitudes of habitat loss resulting from water diversion using a high-gradient reach of the North Alouette River, BC, as a case study. We found that uncertainty in habitat suitability indices for juvenile rainbow trout generally dominated uncertainty in the results of the BCIFM when large (>15) numbers of transects were used. In contrast, with small numbers of transects, variation in physical habitat among sampled transects was the major source of uncertainty in the results of the BCIFM. Presentations of results of the BCIFM in terms of probabilities of different amounts of habitat loss for a given flow can help managers prescribe instream flow requirements based on their risk tolerance for fish habitat loss. Copyright © 2015 John Wiley & Sons, Ltd.

KEY WORDS: instream flow; low-flow period; fish habitat; run-of-river hydroelectric projects; habitat suitability indices; physical habitat simulation; rainbow trout

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INTRODUCTION

The increasing demand for water resources has resulted in changes in the discharge of streams around the world, and impacts of such alterations on river biota have been documented (e.g. Richter *et al.*, 1997; Poff and Zimmerman, 2010). Of particular importance is the human demand for water during naturally occurring low discharge periods, which is often recognized as a critical period for aquatic ecosystems (Bradford and Heinonen, 2008). During such periods, most stream-habitat types experience a reduction in habitat area, invertebrate production and water quality, which can be stressful for fish and other biota (Poff and Zimmerman, 2010). As a result, resource managers frequently face the difficult task of setting instream flow requirements (IFRs) for the low-flow period that balance the needs of industry, agriculture, other human activities and environmental objectives.

Instream flow requirements are often required for run-of-the river (RoR) hydroelectric project developments. A typical RoR project has a low-head weir that diverts a portion of the river's discharge into a penstock and to a powerhouse, where it is subsequently returned to the channel, thereby restoring natural stream discharge downstream of the project. The diversion reach, which can extend for several kilometres, experiences reduced discharge during power generation. During the permitting stages prior to construction of RoR facilities, resource managers must make decisions regarding the IFR in the diversion reach that will meet their objectives. The assessment of the impacts of reduced flow in the diversion reach and the setting of an IFR are often informed by an evaluation of how fish habitat conditions change as discharge changes.

In British Columbia (BC), Canada, RoR hydropower has emerged as a key renewable energy source. There are currently 56 RoR projects that have been built, 25 others that have awarded contracts and many hundreds of water licence applications (CEBC 2015). All of these projects must undergo an assessment to determine the potential impact of low flow in the diversion reach. The methodology used for this assessment is to predict the suitable habitat for a species of interest from measured (e.g. Lewis *et al.*, 2004) or numerically modelled (e.g. Bovee *et al.*, 1998) water depth and velocity, bed-material grain size and sometimes

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type and abundance of cover, all collected at transects on the study stream. Transect data, either measured or extracted from a hydraulic model, are weighted by biological models, or habitat suitability indices (HSIs), which describe the suitability, between 0 and 1, of the physical habitat variables for the organism of interest (Williams, 1996). Estimates of available habitat are either measured or predicted at various discharges, and those changes in habitat features with flow are used for the negotiation between a regulatory agency and a project's proponent.

In British Columbia, the BC Instream Flow Methodology (BCIFM; Lewis *et al*., 2004) has been developed for RoR hydropower projects. The BCIFM encourages the use of a stratified random design for the selection of transect locations, which enables investigators to calculate the uncertainty in final estimates (see Williams (2010b) for a discussion of statistical versus deliberate sampling designs). The BCIFM also requires transects to be sampled at three to five different discharges to allow the development of an empirical relation between discharge and habitat values. However, little analysis has been carried out on the nature and magnitude of uncertainties in the BCIFM procedure, or a related procedure, PHABSIM, that relies on output from a numerical hydraulic model. It is now recognized that uncertainties in such habitat-based instream flow studies can be large (Williams, 1996, 2010a; Ayllón *et al*., 2012). Those uncertainties result from measurement error, variation in physical habitat variables among transects and across different discharge levels, uncertainties in HSI curves of a given species and inaccuracies in hydraulic models (Williams, 1996). For example, Williams (1996, 2010a, 2013) explored the relation between number of transects and precision under the assumption that transects are from a random or stratified random sample of river habitats. Consistent with expectations, he found that precision increased with the number of transects but noted that specific recommendations regarding transect number depend on the river and sampling design. Gard (2005) and Payne *et al*. (2004) described similar results, although their methods have been criticized (Williams, 2010a).

Uncertainties associated with the biological inputs to transect-based assessments have also long been noted as potentially significant but have not often been investigated. Specifically, the form of the HSI curves chosen for fish habitat use may be a significant source of uncertainty (Williams, 2010b). Ayllón *et al*. (2012) showed how uncertainty in site-specific HSI curves causes uncertainty in the habitat–flow relation as a result of natural variation among individual fish in their habitat use. The best-case scenario for IFR studies would include the creation of river-specific HSI curves (Waite and Barnhart, 1992), but this can be a large undertaking because collection of sufficient data for all life stages and species of interest is often beyond the means of individual projects.

In the absence of site-specific information, investigators can apply empirically derived HSI curves from streams thought to be similar to the one being analysed. In some cases, resource management agencies produce standard curves that may be a composite of regional data and expert opinion and ask that all assessments be carried out with the same set of HSI curves if site-specific information is not available (e.g. WDFW, 2004). Even though various researchers have commented on the potential for the choice of HSI curves to influence the habitat–discharge analysis (Williams *et al*., 1999; Ayllón et *al*., 2012), actual analyses appear few (Waite and Barnhart, 1992). There has also not been a direct comparison of the relative importance of transect-based versus HSI-based uncertainty on habitat–flow relations.

We implemented the BCIFM in a typical RoR setting to (i) estimate the uncertainty in the habitat–flow relation generated by HSI curves as well as by variability in samples among transects, (ii) evaluate the use of aggregate HSI curves for situations where site-specific data are not available and (iii) show how uncertainty in the habitat–flow relation translates into various chances of habitat loss under different scenarios of flow alteration. These results can be used to explore, and communicate to managers, the uncertainty in the habitat–flow relation produced by the BCIFM.

METHODS

Field site

The North Alouette River flows out of the Golden Ears mountain range and drains into the Pitt River near the town of Maple Ridge, BC. The watershed is located within the coastal temperate rainforest region, which is characterized by dry summers with low discharge and wet winters with heavy rainfall events that cause sporadic high discharge (Wade *et al*., 2001). Our study site was located directly above a waterfall complex (49°15′54.24″N, 122°34′03.13″ W), within the University of British Columbia Malcolm Knapp Research Forest approximately 15 km upstream from the confluence with the Pitt River.

The North Alouette River is gauged 3.1 km downstream of the study site (Water Survey of Canada Station No. 08MH006; WSC 2011). The river has a mean annual discharge of $2.8\,m^3\,s^{-1}$ and a drainage area of $37.3\,km^2$. The channel of the study reach has a gradient of 2.0–3.1%, with an average channel width of 18.6 m. The study reach is dominated by boulder, cobble and gravel bed material and is exclusively composed of plane bed alluvial channel type (Montgomery and Buffington, 1997) with riffle-run mesohabitat type (Maddock, 1999). Both rainbow trout (*Oncorhynchus mykiss*) and cutthroat trout (*Oncorhynchus clarkii*) have been captured in the reach (Mathes and Hinch, 2009).

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Physical habitat data

We collected physical habitat data from the North Alouette River on five dates during the summer and fall of 2010. We established two groups of 10 transects ($n=20$) within the study reach. The first group of transects was systematically spaced 10 m apart from an arbitrarily chosen starting point. The second group of transects was similar but was located approximately 250 m upstream of the first group, past a reach with multiple active channels that was difficult to sample. Physical characteristics of the stream were similar between both groups of transects. The reach-averaged low-flow channel width and slope differed by less than 10%, and the range of flow depths and velocities were the same. Transects were placed perpendicular to the stream flow. Measurements of river physical habitat (depth, velocity, width and bed-material grain size) were collected at 0.5-m points along each transect, following the BCIFM method (Lewis *et al.*, 2004). Depth and velocity measurements were collected using a wading rod and Marsh-McBirney Flo-Mate™ flow metre. A local estimate of river discharge was calculated each day that physical data were collected; the estimated discharge ranged from 0.13 to 1.79 $m^3 s^{-1}$.

Habitat suitability data

We chose *O. mykiss* fry for our analysis because its life stage and species have been observed in our study area (Mathes and Hinch, 2009) and is the most common species encountered in streams that have run-of-river hydropower developments in BC. Our choice of a single life stage and species is deliberate, because it allows us to focus on how to assess uncertainty. We compiled five sets of HSIs for *O. mykiss* fry from a variety of locations in North America (Figure 1). Four of the five sets were the result of expert opinion of individuals or groups, and one (Higgins *et al.*, 1999) was based on an empirical study in a stream in BC similar to our study site. To simplify the analysis, uncertainty in habitat suitability of bed-material grain size was not considered; instead, we used a single bed-material HSI that is commonly used in BC for *O. mykiss* fry. It is often argued in the literature that HSI curves should be based on site-specific data (Waite and Barnhart, 1992; Williams *et al.*, 1999; Ayllón et *al.*, 2012), but in practice, this is rarely implemented for small-scale water withdrawals, so our inclusion of HSIs based on expert opinion is consistent with practice.

Habitat–flow relation

Physical habitat data and HSI information were combined to produce a metric of availability of habitat for *O. mykiss* fry, weighted usable width (*WUW*; Lewis *et al.*, 2004), for each transect at each discharge level as follows:

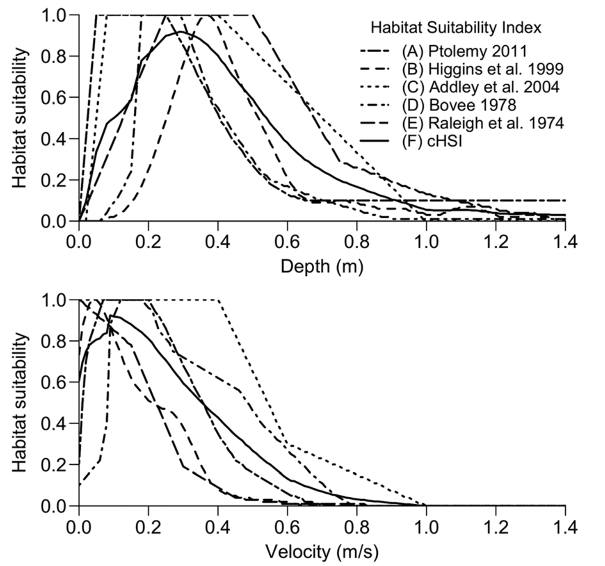

Figure 1. Functions for habitat suitability of *O. mykiss* fry for depth (top panel) and velocity (bottom panel). Data were drawn from five studies as indicated in the legend, except (F) combined habitat suitability index (cHSI), which is the median of the bootstrapped mean of the five habitat suitability curves (A–E). Curve A was provided by R. Ptolemy, Rivers Biologist, Fisheries Science Section, Ecosystems Branch, Ministry of Environment, Victoria, BC, 7 June 2011

$$WUW = \sum_{i=1}^{n} (w_i \times dHSI_i \times vHSI_i \times sHSI_i) \qquad (1)$$

where the *WUW* of each transect equals the sum of the weighted width of all *n* cells along that transect. The weighted width of each cell along the transect, *i*, is calculated as width (w_i) of the cell multiplied by its suitability of depth ($dHSI_i$), velocity ($vHSI_i$) and substrate size ($sHSI_i$). This approach assumes that suitabilities for each habitat measure are independent of each other; field evidence supports this assumption for age 0 salmonids (Lambert and Hanson 1989; Ayllón *et al.* 2009). The BCIFM approach treats each transect as a sample of aquatic habitat within the study reach.

The relation between average weighted usable width across all transects (WUW_{avg}) and discharge (i.e. the habitat–flow relation) was estimated for the study reach by fitting a log-normal function with a multiplicative scalar to WUW estimates for each transect. The log-normal form is flexible and can fit typical habitat–flow relations (Lewis *et al.*, 2004). However, using the log-normal function to fit the habitat–flow relations assumes a smooth relation between the two variables, which may introduce additional uncertainty into the analysis if the data do not follow this form. A habitat–flow relation for the study reach was thus estimated as follows:

$$WUW_{avg} = A \cdot \frac{1}{Q\sqrt{2\pi\sigma^2}} \cdot e^{-\frac{(\ln Q - \mu)^2}{2\sigma^2}} \qquad (2)$$

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where the WUW_{avg} is a function of discharge (Q), a scalar (A) and a location (μ) and scale parameter (σ). The log-normal function was fit to WUW–discharge data using a least-squares optimizing function, 'optim', in R (R Development Core Team, 2008).

We derived three management parameters from the habitat–flow relation. These were (i) the maximum WUW_{avg}, which was the amount of habitat available at the peak of the habitat–flow relation; (ii) the optimal discharge, that is, the discharge at which the maximum WUW_{avg} occurs; and (iii) the discharges at which different percentages of habitat loss (relative to the maximum WUW_{avg}) occur on the ascending limb of the habitat–flow relation. All management parameters were calculated numerically using a maximum optimizing function, 'optimize', in R (R Development Core Team, 2008).

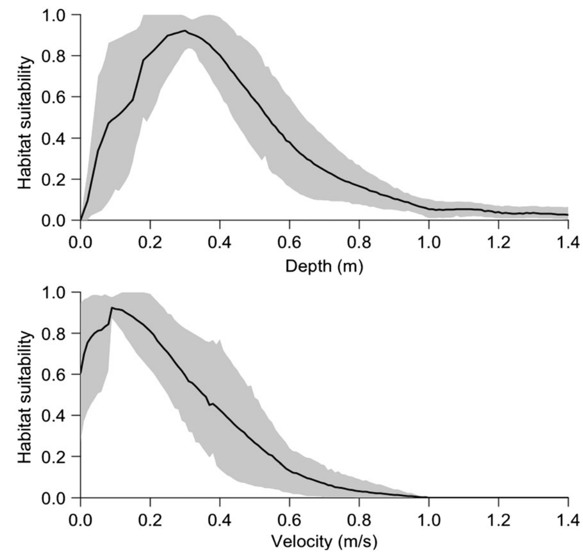

Figure 2. Combined habitat suitability indices for *O. mykiss* fry for depth (top panel) and velocity (bottom panel). The solid line is the median, and the grey band is the empirical 2.5% and 97.5% confidence interval from bootstrapping the mean of five habitat suitability indices in Figure 1. Bootstrapped means were generated at 0.01 intervals along the *x*-axis

HSI uncertainty

We first evaluated the effect of the choice of HSI curves on the resulting habitat–flow relation from the BCIFM and the corresponding management parameters. Habitat–flow relations were generated, and management parameters were calculated using each of the five different sets of HSI curves for depth and velocity from Figure 1. Physical habitat data from all 20 transects were used.

In the next analysis, we assumed that there are general HSIs for the region, and that the five sets of curves we obtained were equivalent to independent samples drawn from the regional relation. We estimated the regional relation by combining five HSI curves into a single aggregate curve using bootstrap analysis (Efron and Tibshirani, 1993). We first randomly drew (with replacement) a sample of five pairs of depth and velocity curves from our collection of HSI curves. The mean suitability value was calculated from the sample at intervals of $0.01\,\mathrm{m\,s^{-1}}$ for velocity and 0.01 m for depth to create a pair of curves based on the mean values estimated from the bootstrap sample. This process was repeated 1000 times. Pairings between velocity and depth were maintained in the bootstrapping process, so selection of a velocity and depth from different curve sets was not possible. Combined curves for depth and velocity (now called cHSI curves) were computed as the median of the bootstrap samples, and uncertainty was expressed as the 2.5% and 97.5% quantiles of the bootstrap samples for each interval (Figure 2). We computed a sixth habitat–flow relation using these cHSI curves.

To evaluate the uncertainty in the habitat–flow relation resulting from uncertainty in the cHSI curves, we generated a habitat–flow relation from each bootstrap sample of the cHSI curves for depth and velocity. This step generated 1000 habitat–flow relations and corresponding management parameters for which the median and empirical 95% confidence interval (CI) were computed. We also calculated the coefficient of variation (CV) of each management parameter as the standard deviation of those 1000 parameter estimates divided by their mean.

Physical habitat uncertainty

We used bootstrap analysis to estimate the contribution of variability in physical habitat data among transects to variation in the habitat–flow relation and management parameters (Figure 3). For this analysis, we assumed no uncertainty in HSIs and used the fixed, deterministic cHSI curves for depth and velocity. We assumed that each transect could be treated as an independent sample of stream habitat. For each bootstrap iteration, 20 transects were randomly sampled with replacement. A habitat–flow relation was estimated for each of the bootstrap sample, and management parameters were calculated. This process was repeated 1000 times. Again, the median, empirical 95% CI and CV of the resulting management parameters were calculated.

We also evaluated the effect of the number of transects on uncertainty in the habitat–flow relation. The number of randomly drawn transects was reduced incrementally from 20 to 3 in separate analyses. A habitat–flow relation was estimated for each sample, and management parameters were calculated. This process was repeated 1000 times for each increment in transect sample size, and management parameters were summarized as described earlier.

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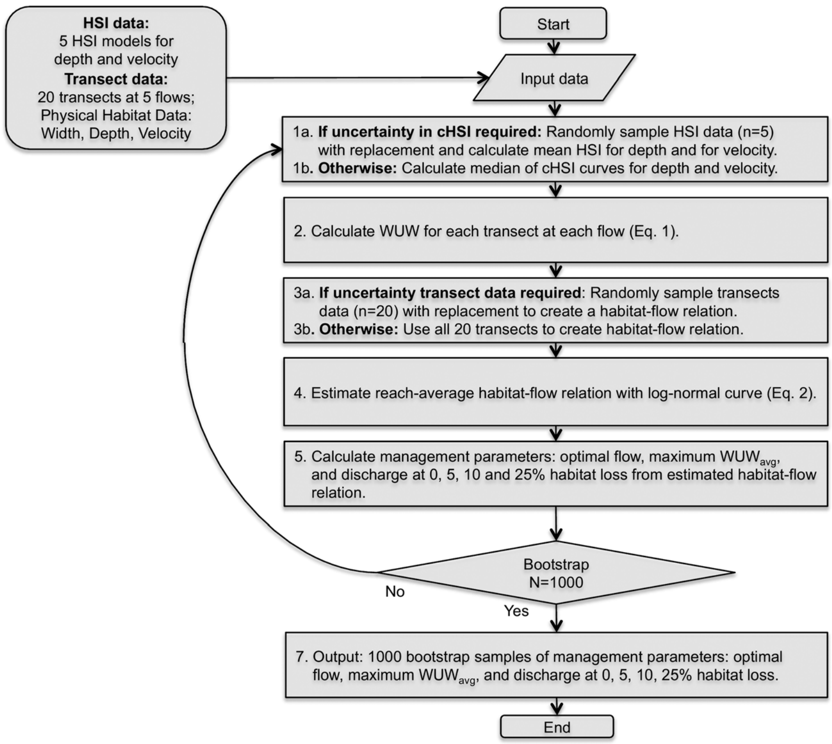

Figure 3. Flow diagram of the method used to incorporate the uncertainties of the habitat suitability indices (HSIs) and transect data into the calculation of management parameters

Combined uncertainty

We used another bootstrap analysis to develop uncertainty bounds on the habitat–flow relation resulting from the combination of uncertainties in the cHSI curve and transect variability (Figure 3). For each bootstrap sample, 20 transects were randomly sampled with replacement. For each transect in the sample, WUW was calculated at each discharge level using a set of HSI curves randomly sampled from the cHSI curves. Finally, a habitat–flow relation was estimated for the bootstrap sample, and management parameters were calculated. This entire process was repeated 1000 times, and the median, empirical 95% CI and CVs of the resulting management parameters were calculated.

Habitat loss

Using the 1000 habitat–flow relations from the analysis that incorporated both uncertainty in the estimate of the cHSI curve and variability in physical habitat data among transects, we calculated the probability of a particular magnitude (0%, 5%, 10% and 25%) of habitat loss occurring as a function of discharge. We defined habitat loss as the percent decrease in WUW_{avg} relative to the maximum WUW_{avg}. Habitat loss was only considered on the ascending limb of the habitat–flow relation because habitat losses occurring from high discharges are of little concern when considering minimum discharge requirements for a stream. For each habitat–flow relation, the discharge at which a particular magnitude of habitat loss occurred was solved using the function 'uniroot' in R, which resulted in distributions of discharge values for each magnitude of habitat loss. The complement of the cumulative probability distribution of discharge values for each magnitude of habitat loss was plotted, resulting in the probability of each particular magnitude of habitat loss occurring as a function of discharge.

RESULTS

HSI, physical habitat and combined uncertainty

Different sets of HSI curves for depth and velocity for *O. mykiss* fry produced substantially different habitat–flow relations and management parameters (Figure 4; Table 1). For example, optimal flow varied from 0.4 to 1.1 $m^3 s^{-1}$, corresponding to 14% to 39% of the river's mean annual discharge, respectively. Uncertainty in estimates of cHSI curves (Figure 2) generated uncertainty in the habitat–flow relation (Figure 5A) and management parameters (first line, Table 2).

Variation among sampled transects generated less uncertainty around the habitat–flow relation (Figure 5B) than uncertainty in cHSI curves (Figure 5A). Uncertainty was generally greater for the optimal discharge than the maximum WUW_{avg}

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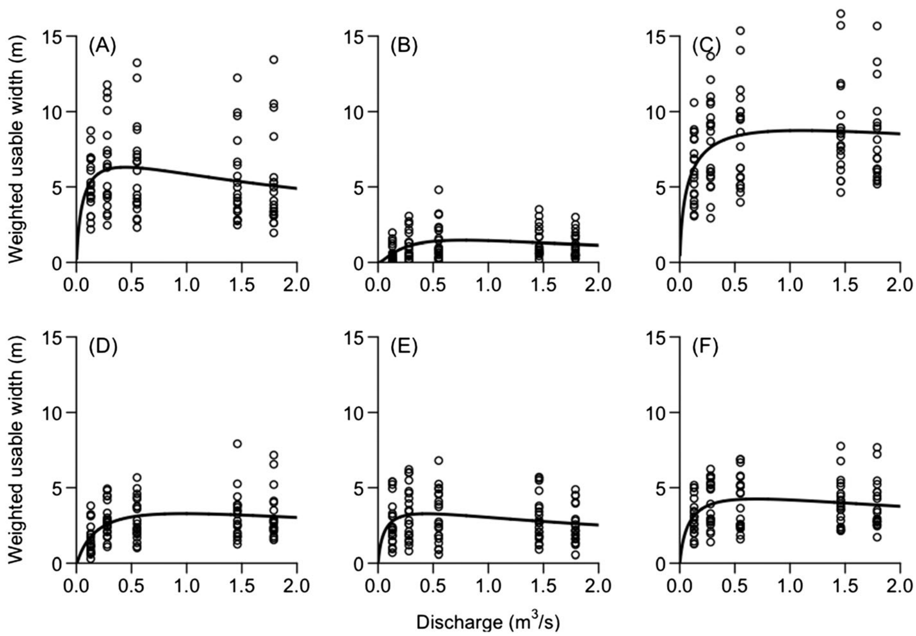

Figure 4. Habitat–flow relations for *O. mykiss* fry in the North Alouette River for each set of habitat suitability curves for depth and velocity. Letters correspond to habitat suitability curves A–F in Figure 1. Open circles are weighted-usable-width calculations for each of the 20 transects at five discharge levels. The solid line is the fit of the log-normal function (Equation (2))

(Table 2) because of the relatively flat functions in Figure 5. As the number of transects used in the analysis was reduced from 20 down to 3, the magnitude of uncertainty about both parameters increased at an accelerating rate (Table 3). Uncertainty in the estimate of optimal discharge was particularly sensitive to the number of transects.

When both variability in physical habitat among sampled transects and uncertainty in the estimated cHSI curves were combined, uncertainty in the habitat–flow relation increased (Table 2).

Habitat loss

The probability of a given magnitude of habitat loss decreased nonlinearly with increasing discharge values (Figure 6). The 0% habitat loss isopleth (solid line) is equivalent to the optimal discharge from the habitat–flow relation; therefore, values to the left of this curve in Figure 6 can be interpreted as the probability that *any* habitat loss will occur. The 5%, 10% and 25% habitat loss isopleths can be used to interpret the probability of those different magnitudes of habitat losses associated with different river discharge values. For example, there is a 10% chance of 10% or smaller habitat loss at a discharge of approximately 0.37 $m^3 s^{-1}$, whereas there is a 50% chance of 10% habitat loss or smaller at a discharge of approximately 0.26 $m^3 s^{-1}$. That same discharge of 0.26 $m^3 s^{-1}$ would be associated with an 88% chance of loss of 5% of the habitat.

Table I. Estimated maximum weighted usable width (WUW_{avg}) and optimal discharge from the habitat–flow relation produced by each of the five habitat suitability indices (HSIs) (A–E from Figure 1) and the combined cHSI (F)

HSI curve set	Maximum WUW_{avg} (m)	Optimal discharge ($m^3 s^{-1}$)
A	6.3	0.4
B	1.5	0.8
C	8.7	1.1
D	3.3	1.0
E	3.3	0.4
F	4.3	0.7

Habitat–flow relations are shown in Figure 4.

DISCUSSION

We found that both variability among transects sampled and uncertainty in HSI curves were important when generating a habitat–flow relation. We demonstrated how the quantification of uncertainty could be used by decision makers to prescribe IFRs based on a specified risk tolerance for different magnitudes of habitat loss. The method requires no special computing facilities or training beyond statistics and statistical programming in R. Therefore, the method can be implemented by any biologist, hydrologist or statistician quickly and efficiently as long as suitable HSI curves can be identified and transect data are available.

We evaluated a scenario in which instream flow practitioners may be forced to choose among multiple pre-existing sets of HSI curves because budget or time limitations do not

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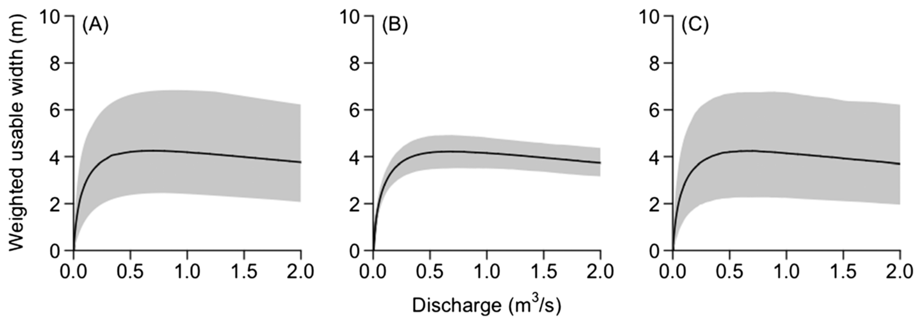

Figure 5. Estimated habitat–flow relations produced by the BCIFM when integrating the uncertainty from (A) the combined habitat suitability indices (cHSIs; Figure 4), (B) the variability among transects while using a constant cHSI for depth and velocity and (C) both sources combined. Solid line is the median weighted usable width; grey band is the empirical 2.5% and 97.5% confidence interval from a bootstrap analysis

allow them to develop location-specific curves. We found that the choice of HSI curve has a large impact on the habitat–flow relation and management parameters. From inspection of the curves, it appeared that differences in velocity suitability had the greatest impact on the results; the effect of depth differences was less apparent, which are results similar to those of others (Waite and Barnhart, 1992; Ayllón *et al*., 2012). Variation among sets of HSI curves for *O. mykiss* fry may be attributed to several factors. Most of the curve sets were derived from expert-opinion processes, centred in different regions of western North America, and differences among expert-opinion processes can be significant (Czembor *et al*., 2011). The diversity of curves may reflect real biological differences among fish populations and their habitats in the different regions, as habitat use can vary as a result of variation in discharge, stream size, productivity, competition and predation risk (Shirvell, 1990; Heggenes *et al*., 1996), although consistency in habitat use among sites has been observed (e.g. Beecher *et al*., 1995).

As an alternative to choosing a single set of HSI curves, we developed a method for combining HSI curves for depth and velocity. Our procedure was based on the assumption that each curve was equally likely and represented an attempt by experts or data collection to represent the true state of nature. Thus, variation among curves was considered to be an estimate of sampling error, which tends to cancel out when the curves are averaged. Because all curve sets were developed in western North America, the cHSI curves could be considered a regional average for the species and life stage. Other weighting schemes could be used if there were reasons to choose or prefer one or more HSI curves to others.

When all 20 transects were used in the analysis, variability in physical habitat among transects contributed roughly equally to the uncertainty in the optimal discharge as that arising from the uncertainty in the estimated cHSI curves. However, the study reach of the North Alouette River contained relatively homogenous river morphology. Thus, the variability among transects was relatively small. Rivers with more variable morphology (e.g. cascade-pool sequences) will likely generate more variability among transects. This increased

Table II. Median, empirical 95% confidence interval (CI) and coefficient of variation (CV) of the estimated maximum weighted usable width (WUW_{avg}) and optimal discharge from the habitat–flow relations produced when incorporating (i) uncertainty from the combined habitat suitability indices (cHSIs) with the transects fixed, (ii) variability among transects but using a constant cHSI and (iii) both sources of uncertainty

	Maximum WUW_{avg} (m)			Optimal discharge ($m^3\,s^{-1}$)		
Source of uncertainty	Median	95% CI	CV (%)	Median	95% CI	CV (%)
cHSI	4.3	2.5–6.8	25	0.7	0.5–1.0	21
Transect	4.2	3.5–4.9	8	0.7	0.5–1.1	21
cHSI and transect	4.3	2.3–6.8	27	0.7	0.4–1.3	34

Table III. Median, empirical 95% confidence interval (CI) and coefficient of variation (CV) of the estimated maximum weighted usable width (WUW_{avg}) and optimal discharge from the habitat–flow relations produced when bootstrapping the variability among transects for a range of numbers of transects sampled

	Maximum WUW_{avg} (m)			Optimal discharge ($m^3\,s^{-1}$)		
Number of transects	Median	95% CI	CV (%)	Median	95% CI	CV (%)
20	4.2	3.5–4.9	8	0.7	0.5–1.1	21
15	4.3	3.4–5.1	10	0.7	0.5–1.1	23
10	4.3	3.3–5.4	13	0.7	0.5–1.3	31
5	4.2	2.9–5.7	17	0.7	0.4–2.1	52
3	4.4	2.7–6.2	22	0.7	0.3–2.7	75

Habitat–flow relations were generated with constant combined habitat suitability indices (cHSIs).

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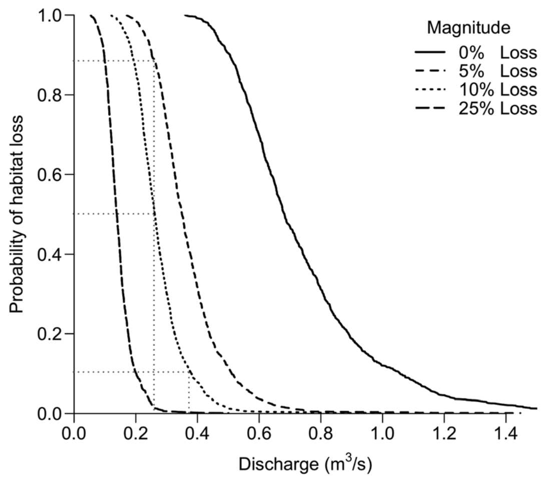

Figure 6. Estimated probability of habitat loss for *O. mykiss* fry in the North Alouette River as a function of discharge for the ascending limb of the habitat–flow relation. Habitat loss is defined as the percentage loss in WUW relative to the maximum value at the optimal flow. Both uncertainty in cHSI and transect data are included. Grey straight dashed lines are for the examples used to guide interpretation of the figure (see text)

variability may result in more uncertainty in the habitat–flow relations and in estimates of the optimal discharge.

Variability in management parameters from the habitat–flow relation increased nonlinearly as the number of transects used in the analysis decreased (Table 3). Both variability in the maximum WUW_{avg} and the optimal discharge did not increase dramatically when the number of transects was reduced from 20 to 15. However, for fewer transects than 15, variability increased substantially and was a dominant source of uncertainty in the analysis, particularly in the estimates of optimal discharge. This result suggests that, for streams with similar characteristics to the North Alouette River, a minimum of 15 transects should be used to minimize variability in transect data when conducting a BCIFM analysis. Because of the lack of heterogeneity in the river morphology of the North Alouette River, this number is on the lower end of the 15–20 transects recommended in the literature to capture the full variability of rivers and produce a meaningful habitat–flow relation (Williams, 1996; Thomas *et al.*, 2004).

Habitat loss

Probability-of-loss curves such as those in Figure 6 are a tool for managers to use to account for risks associated with uncertainties in the BCIFM process. If managers express their decision criteria in the form of being able to accept an *X%* chance of a *Y%* or smaller loss of habitat, the corresponding IFR can be read directly from Figure 6. For example, if a manager wants to ensure that the IFR will result in no more than a 10% chance of a 10% loss in habitat, the corresponding flow should be at least $0.37\,m^3\,s^{-1}$. In contrast, if a manager is willing to accept a 50% chance of a loss of 10% or less of the habitat, then the median IFR would be calculated as a far lower value, $0.26\,m^3\,s^{-1}$. In this situation, the difference in flows ($0.11\,m^3\,s^{-1}$) is the 'risk premium' or 'insurance' against loss of habitat caused by the uncertainty in the assessment process, which in this example is 42% of the median IFR.

For the North Alouette River, our analysis suggests that it is unlikely that increasing the number of transects beyond 20 could significantly reduce the risk premium. A detailed site-specific habitat suitability study may reduce the risk premium somewhat if precise HSI curves can be developed for that location. However, uncertainty in field-collected HSI data can be significant (Williams *et al.*, 1999; Ayllón *et al.*, 2012).

Alternatively, proponents of run-of-river projects may view reducing the uncertainty within the instream flow assessment as an opportunity to increase their allowable water withdrawals. If managers are risk adverse to habitat loss due to uncertainty, then a project proponent may benefit from increasing data collection, which will steepen the curves of Figure 6 and reduce the risk premium that resource managers may have.

Physical habitat has proven to be an effective heuristic for making decisions about water management. Although the relation between changes in physical habitat and fish populations is often weak because of other factors that affect fish survival and abundance (Mathur *et al.*, 1985; Sabaton *et al.*, 2008), minimizing losses to habitat should reduce some of the risks associated with an altered flow regime. Uncertainties that are ignored in an assessment create risk for decision makers. The failure to account for uncertainties also creates doubts about the quality of the assessment, which can exacerbate the difficulty of negotiations between project proponents, regulatory agencies and other stakeholders.

We incorporated both natural and measurement error in hydraulic conditions and uncertainty about habitat suitability for fish into predictions of habitat change with changes in flow, following the call of Williams (2010a) for a more statistical approach to instream flow assessments. In the case of rainbow trout fry in the North Alouette River, our results suggest that even with a reasonable number of transects and observations across a range of flows to capture the optimum flow, uncertainty remains in the predictions of habitat loss. If a manager is very risk averse about physical habitat loss, then the risk premium could be more than 40% of the median IFR. A less risk-averse approach may be warranted if the biological resources potentially affected by low water have low value to society, or if there is other information to suggest that the biological resources may not be strongly affected by changes in flow. Our analysis was conducted on a relatively homogenous section of stream and for one

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species and life stage; additional studies in different rivers with other species are needed to determine how generally applicable our results are. We urge researchers and practitioners to calculate and report uncertainty in flow–habitat analyses to permit managers to consider the risks when making decisions about IFRs.

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