

1 **Automated analysis of lateral river connectivity and fish stranding risks– Part 1:**
2 **Review, theory and algorithm**

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19

20 Abstract

21 Riverine fish stranding is of significant concern due to its potentially devastating
22 impacts on fish populations already at risk. Because stranding is dependent on a
23 wide range of biotic and abiotic factors, it is difficult to accurately identify and
24 parameterize fish stranding risks for various river topographies, fish
25 species/lifestages, and flow ramping scenarios. This article presents a literature
26 review, new concepts, and a novel Python3 algorithm for post-processing two-
27 dimensional hydrodynamic numerical model results to identify spatially explicit
28 locations where fish stranding is likely, such as but not limited to downstream of
29 hydropeaking facilities. Compared to previous stranding algorithms, this one is
30 novel in its use of graph theory to find optimal fish escape routes and for its
31 embedding in the free, open-source river analysis software River Architect. Guided
32 by biological parameter selection and supplied with two-dimensional hydrodynamic
33 model rasters, River Architect's Stranding Risk module is suitable for
34 characterization of existing pool stranding risks, alternative flow regime and
35 topographic design evaluation, and post-project assessment of rivers during flow
36 recessions.

37 Keywords: fish stranding, hydraulic connectivity, hydrodynamic modeling,
38 ecohydraulics, fish modeling, hydropeaking

39

40 1. Introduction

41 In river science, connectivity is a term used to characterize the ability of water,
42 sediment, nutrients, energy and biota to move freely throughout a river in both space
43 and time (Ward, 1989; Grill et al., 2019). Lateral, longitudinal, and vertical river channel
44 connectivity plays a vital role in determining the ecologic success of river restoration,
45 habitat enhancement, and management efforts, yet the importance of lateral habitat
46 connectivity is often overlooked in the design of habitat enhancement projects (Casas-
47 Mulet et al. 2015). One reason for this oversight is an excessive focus on quantifying
48 metrics such as habitat abundance for a single species and lifestage without
49 considering how fish move between different locations within and between their
50 lifestages (Parasiewicz, 2007; Lancaster and Downes, 2010; Shenton et al., 2012).
51 However, if channel connectivity is not adequately addressed, fish can become
52 separated from the river mainstem due to a decrease in discharge; a phenomenon
53 referred to as stranding.

54 Stranding is a natural process as part of a river's disturbance regime. It often leads
55 to fish mortality caused by dewatering, hypoxia, temperature stress, or predation, while
56 sublethal effects can range from temporary metabolic stress to chronic hypoxia
57 (Cushman, 1985; Sabo et al., 1999; Quinn and Buck, 2001; Flodmark et al., 2002;
58 Evans, 2007). Single stranding events have been observed to affect thousands of fish
59 (Higgins and Bradford, 1996). Stranded fish serve as an important resource input to the
60 terrestrial ecosystem (Quinn et al., 2009).

61 In addition to naturally caused stranding, artificially caused fish stranding is

62 prevalent in regulated rivers, especially those influenced by hydropeaking operations
63 (Nagrodski et al., 2012). For rivers already degraded by cumulative anthropogenic
64 impacts, stranding mitigation can be financially costly to river managers and repeated
65 stranding events may have devastating effects on fish populations (Bauersfeld, 1978;
66 Saltveit et al., 2001; Sauterleute, et al., 2016). However, it is difficult to predict when,
67 where, and to what degree stranding risks may be present (Nagrodski et al., 2012;
68 Golder Associated Ltd., 2013; Harby and Noack, 2013). Stranding is not only an
69 important aspect of river degradation, but also a potential side effect of river restoration
70 projects seeking improved environmental conditions. As noted by both field (Hunter,
71 1992; Auer et al., 2017) and model investigations (Tuhtan et al., 2012; Hauer et al.,
72 2014; Vanzo et al., 2016b), the presence of heterogeneous river morphology provides
73 habitat benefit in degraded rivers, but the same areas are also subject to increased
74 stranding risks during hydropeaking. Consequently, successful river restoration efforts
75 should consider these competing effects when introducing heterogeneous structures in
76 rivers.

77 This article reviews the state of the science of fish stranding relevant for predictive
78 modeling and summarizes pre-existing ecohydraulic modeling of fish stranding risk
79 before presenting the theory and methods for a novel algorithm for fish stranding risk
80 assessment. This work offers new basic and applied science contributions to ecology
81 and ecohydraulics compared to pre-existing ones on several fronts, including the use of
82 graph theory to find optimal fish escape routes. Also, this algorithm has been made
83 highly accessible for widespread use through incorporation into the free, open-source
84 River Architect software (Schwindt et al., 2020; <https://riverarchitect.github.io/>). Detailed

85 explanation of the literature and new method sufficient for reader understanding
86 precludes inclusion of an application and vetting; those elements are supplied in the
87 subsequent companion article (Larrieu and Pasternack, submitted).

88 **2. Fish stranding science**

89 While fish stranding occurs in both natural and regulated flow regimes and in natural
90 and structurally degraded rivers, the majority of available literature on the topic focuses
91 on anthropogenically impacted systems. Most studies investigating the effects of
92 environmental variables on stranding are concerned with the occurrence of stranding in
93 river reaches downstream of hydropeaking power plants (Nagrodski et al., 2012). In
94 such locations, hydropower operations induce rapid and high-magnitude flow
95 fluctuations to meet electricity demands. Hydropeaking practices in such rivers can
96 regularly induce changes of water surface elevation (WSE) at rates in excess of 1 m/hr
97 (Vanzo et al., 2016a; Hauer et al., 2017). As a result, significant fish stranding can occur
98 during rapid flow decreases. Even modest fluctuations can yield stranding risks
99 (Bauersfeld, 1978; Higgins and Bradford, 1996).

100 2.1. Stranding terminology

101 Stranding is often categorized in the scientific literature as interstitial stranding (also
102 called bar stranding or beaching) or pool stranding (also called off-channel stranding,
103 isolation, or trapping). Interstitial stranding entails fish becoming stuck in substrate
104 interstices (typically gravel) either on the surface or below the substrate's surface layer.
105 Pool stranding entails a morphology in which a topographic saddle point separates a
106 wetted area from the river mainstem, trapping fish in a disconnected feature (Hunter,

107 1992). Pool stranding has been predominantly understood and explored at the
108 morphological unit scale (e.g., scour pools, side channels, swales, and flood runners).
109 Both types have been investigated separately in various studies (e.g. Bradford et al.,
110 1995; Saltveit et al., 2001; Halleraker et al., 2003; Irvine et al. 2014), though distinctions
111 between the two are not always clearly made, especially in field experiments.

112 As topographic mapping and hydrodynamic modeling progress, submeter-to-meter
113 topographic depressions on an otherwise smooth side slope can be resolved and
114 identified as causing isolation preventing fish from returning to the mainstem. Thus,
115 technological progress yields a grey area between the two stranding types. For any
116 given study, if the technology used can resolve disconnecting areas at a finer scale,
117 then the resulting stranding is termed pool stranding because it is still the result of an
118 isolation on the bed surface, not a result of interstitial dynamics. Consequently, the term
119 “pool stranding” encompasses stranding resulting from disconnecting areas of any
120 resolvable size, not just pools at the morphological unit scale. This study addresses
121 identification and assessment of pool stranding risks for isolating topographic
122 depressions at the resolution of the applied topographic data (typically at a meter-
123 scale), though factors relevant to both stranding types (e.g., ramping rate) are
124 considered.

125 2.2. Factors influencing fish stranding

126 *In situ* and flume studies have identified a broad range of physical factors that exert
127 significant impacts on fish behavior and stranding rates during flow reductions. Factors
128 relevant to fish stranding include topography, ramping rate (rate of water surface
129 elevation change), water temperature, time of day, and wetted history (length of time at

130 sustained high discharge before flow reduction occurs). Based on studies of these
131 factors, guidelines have been proposed for management of flow releases from dams to
132 mitigate stranding (Halleraker et al., 2003). While some general considerations have
133 also been suggested for morphological designs that mitigate stranding, more work is
134 needed to determine and guide best practices (Harby and Noack, 2013).

135 2.2.1. Topography

136 Topography is a critical factor controlling the presence of stranding risks. Local
137 topographic depressions enable formation of pools that disconnect from the main
138 channel, a prerequisite for the occurrence of pool stranding. Hunter (1992) noted from
139 field observations that long side channels with intermittent flows are notorious for
140 trapping fish, killing some or all. Artificially constructed side channels can be
141 morphodynamically unstable, leading to loss of side channel connectivity associated
142 with stranding risks (Riquier et al., 2017; van Denderen et al., 2019a,b). *In situ*
143 experiments by Irvine et al. (2014) also found that physically re-contouring topography
144 to remove pool-like depressions significantly lowered observed stranding rates. Flume
145 experiments with juvenile salmonids observed that lateral bank slopes influence
146 interstitial stranding rates, with generally less stranding occurring on steeper banks
147 (Bradford et al., 1995).

148 2.2.2. Ramping rate

149 In many experiments, a positive trend exists between ramping rate and stranding
150 rate for both interstitial and pool stranding (Bradford et al., 1995; Bradford, 1997;
151 Halleraker et al., 2003). For example, flume experiments conducted by Bradford et al.
152 (1995) investigated juvenile coho salmon and rainbow trout stranding on an artificial

153 gravel bar. In these experiments, higher ramping rates were associated with more
154 stranding. Similar experiments indicated a significant positive relationship between
155 ramping rates and side channel trapping of Chinook and coho salmon juveniles
156 (Bradford, 1997). Halleraker et al. (2003) also found that ramping rate had a significant
157 effect on interstitial stranding of brown trout in cold water (6-7 °C).

158 However, the degree to which ramping rates drive stranding is dependent upon
159 several other physical factors such as water temperature and time of day, and in some
160 cases ramping rate has negligible effect on stranding. While Bradford et al. (1995)
161 identified a positive trend between ramping rates and stranding rates (for both interstitial
162 and pool stranding), time of day had a much more significant effect than ramping rate.
163 In addition, for relatively steep lateral slopes (~6%) there was less stranding observed
164 at a ramping rate of 60 cm/hr than at 6 and 30 cm/hr. This may be caused by a stronger
165 escape response elicited by a combination of fast ramping and dramatic lateral
166 differences in water depth. Bradford (1997) also noted a statistically insignificant effect
167 of ramping rate on interstitial stranding of Chinook salmon fry. Field experiments
168 investigating stranding for a broader variety of fish species by Irvine et al. (2014) found
169 that ramping rate did not influence overall stranding rates either. Substantial stranding
170 has been observed even at the lowest achievable ramping rates, especially pool
171 stranding (Higgins and Bradford, 1996; Bradford, 1997). Nonetheless, it has been
172 proposed as a general guideline that ramping slower than 10-15 cm/hr may mitigate
173 stranding of salmonids in circumstances where it has a significant effect (e.g. in cold
174 water or during the day for juvenile salmonids) (Halleraker et al., 2003, 2007).

175 2.2.3. Water temperature

176 Water temperature plays a critical role in stranding. Both field and flume experiments
177 have shown that for a fixed topography, water temperature is the dominant factor
178 influencing stranding rates, especially for interstitial stranding of juvenile salmonids.
179 Temperatures below 10 °C are associated with higher stranding rates (Bradford, 1997;
180 Saltveit et al., 2001; Halleraker et al., 2003). For example, Bradford (1997) found that
181 for newly emerged Chinook salmon in an artificial channel, interstitial stranding rates
182 were six times greater at a temperature of 6 °C compared to 12 °C. Halleraker et al.
183 (2003) also identified temperature as the most important factor explaining variation in
184 interstitial stranding of juvenile brown trout, with significantly more stranding occurring
185 for 6–7 °C water temperature compared to 10–12 °C.

186 Increased stranding rates at low water temperatures have been linked to lower fish
187 activity, substrate-seeking behavior, and concealment exhibited by juvenile salmonids,
188 especially during daytime (Bradford et al., 1995; Saltveit et al., 2001; Halleraker et al.,
189 2003; Scruton et al., 2008). Concealment behavior leads to a higher degree of
190 interstitial stranding, potentially due to substrate isolating fish from cues to leave a
191 dewatering area.

192 2.2.4. Time of day

193 In addition to being less active in cold water, juvenile salmonids are also known to
194 be less active in daylight (Heggenes et al., 1993, Scruton et al., 2008). Bradford et al.
195 (1995) found time of day to be the most significant driver of juvenile salmonid stranding
196 and associated this effect with observed daytime concealment behavior. Moreover,
197 Saltveit et al. (2001) noted that among Atlantic salmon wild and hatchery fish stranded

198 less and more at night, respectively. This observation emphasizes that significant
199 differences in behavioral responses to environmental conditions may be present
200 between wild and hatchery fish, driving differences in stranding rates. However, Irvine et
201 al. (2009) found no influence of time of day on pool stranding.

202 2.2.5. Other considerations

203 Observations of fish behavior in response to flow reductions and environmental
204 conditions provide valuable insight regarding the process of stranding. In net pen
205 experiments on Atlantic salmon, Saltveit et al. (2001) observed that during flow
206 reductions fish would arrive late to the lower part of the enclosure just before it dried up.
207 Bradford et al. (1995) also observed that juvenile coho salmon and rainbow trout often
208 wait until their dorsal fin is dry before migrating to deeper water during a dewatering
209 event. Also, other studies have observed that fish concealed within substrate may wait
210 until their fins are no longer submerged before attempting to escape dewatering areas
211 (Bradford, 1997; Saltveit et al., 2001). In addition, Saltveit et al. (2001) noted that
212 snorkeling observations and unpublished laboratory experiments on brown trout by
213 Halleraker indicated that juveniles follow the water edge during flow fluctuations. This
214 behavior across a wide variety of salmonid species may partially explain their
215 susceptibility to stranding during rapid dewatering events, in large floodplain and side
216 channel features, and in cold water temperatures or daylight conditions when substrate
217 seeking behavior occurs.

218 Juvenile salmonids have been observed to exhibit fidelity to habitat areas once
219 occupancy is established. In relation to stranding, multiple studies have shown that
220 stranding rates are higher the longer high flows are maintained prior to a flow reduction,

221 suggesting that acquired fidelity to habitat areas leads to decreased movement during
222 flow reductions (Halleraker et al., 2003; Irvine et al., 2009; Irvine et al., 2014). Field
223 studies indicate that stranding rates increase with the acclimation time given to fish
224 upon entering an enclosure and prior to initiation of the flow reduction (Saltveit et al.,
225 2001). Site fidelity in salmonids may also be related to fish size and associated social
226 hierarchy, with more dominant fish exhibiting more fidelity to habitat sites while less
227 dominant fish move more readily during flow ramping (Scruton et al., 2008).

228 The size and age of juveniles largely determines their susceptibility to stranding as
229 well. Young, small juveniles are most prone to interstitial stranding. Risk is significantly
230 reduced once they exceed 50 mm in length (Hunter, 1992). Consequently, juveniles
231 larger than 50 mm are primarily at risk of pool stranding.

232 Most stranding studies address juvenile salmonids, though some have investigated
233 stranding of European grayling (Tuhtan et al., 2012; Auer et al., 2017) and a variety of
234 species (Irvine et al., 2009; Irvine et al., 2014). The influence of physical factors on
235 stranding may vary significantly between species and life stages, so further research
236 into stranding of other species would facilitate deeper understanding. Moreover,
237 because the vast majority of stranding studies are concerned with regulated
238 hydropeaking (mountain) rivers, the significance of stranding under natural flow regimes
239 and in lowland rivers is less understood (Nagrodski et al., 2012). Future research into
240 the role that stranding plays in unregulated rivers and how stranding affects population
241 dynamics and ecosystems could lead to new insights regarding the phenomenon.

242 2.3. Stranding consequences

243 Documented consequences of stranding for individual fish are numerous and wide-
244 ranging, from temporary stress response to mortality (Bauersfeld, 1978; Quinn and
245 Buck, 2001; Flodmark et al., 2002). In contrast, there have been no empirical studies
246 conducted investigating the long-term dynamics of fish populations in response to
247 repeated stranding events. However, existing evidence suggests that repeated
248 stranding induced by hydropeaking may lead to declines in fish populations, and single
249 stranding events can kill large numbers of fish (Bauersfeld, 1978; Higgins and Bradford,
250 1996). On a river in Washington State, Bauersfeld (1978) estimated that stranding
251 induced by a single flow reduction event resulted in the loss of 1.5% of the river's native
252 fall-run Chinook salmon population, corresponding to a potential loss of 59% of the
253 native Chinook fry population in a single rearing season. Saltveit et al. (2001) noted that
254 several Norwegian rivers experienced declines in juvenile Atlantic salmon density
255 following the initiation of hydropeaking practices. The most dramatic declines in fish
256 density were observed in the uppermost reaches (where ramping rates are typically
257 greatest and temperatures typically lowest), in conjunction with observations of
258 stranded fish following flow reductions. The decline of juvenile fish abundance in
259 regulated Norwegian rivers is primarily attributed to stranding induced by rapid flow
260 fluctuations (Saltveit et al., 2001; Ugedal et al., 2008). Sauterleute et al. (2016) applied
261 a population dynamics model to investigate population responses to repeated fish
262 stranding events, also suggesting that frequent and rapid flow fluctuations may lead to
263 significant reduction in fish populations.

264 **3. Stranding risk assessment methods**

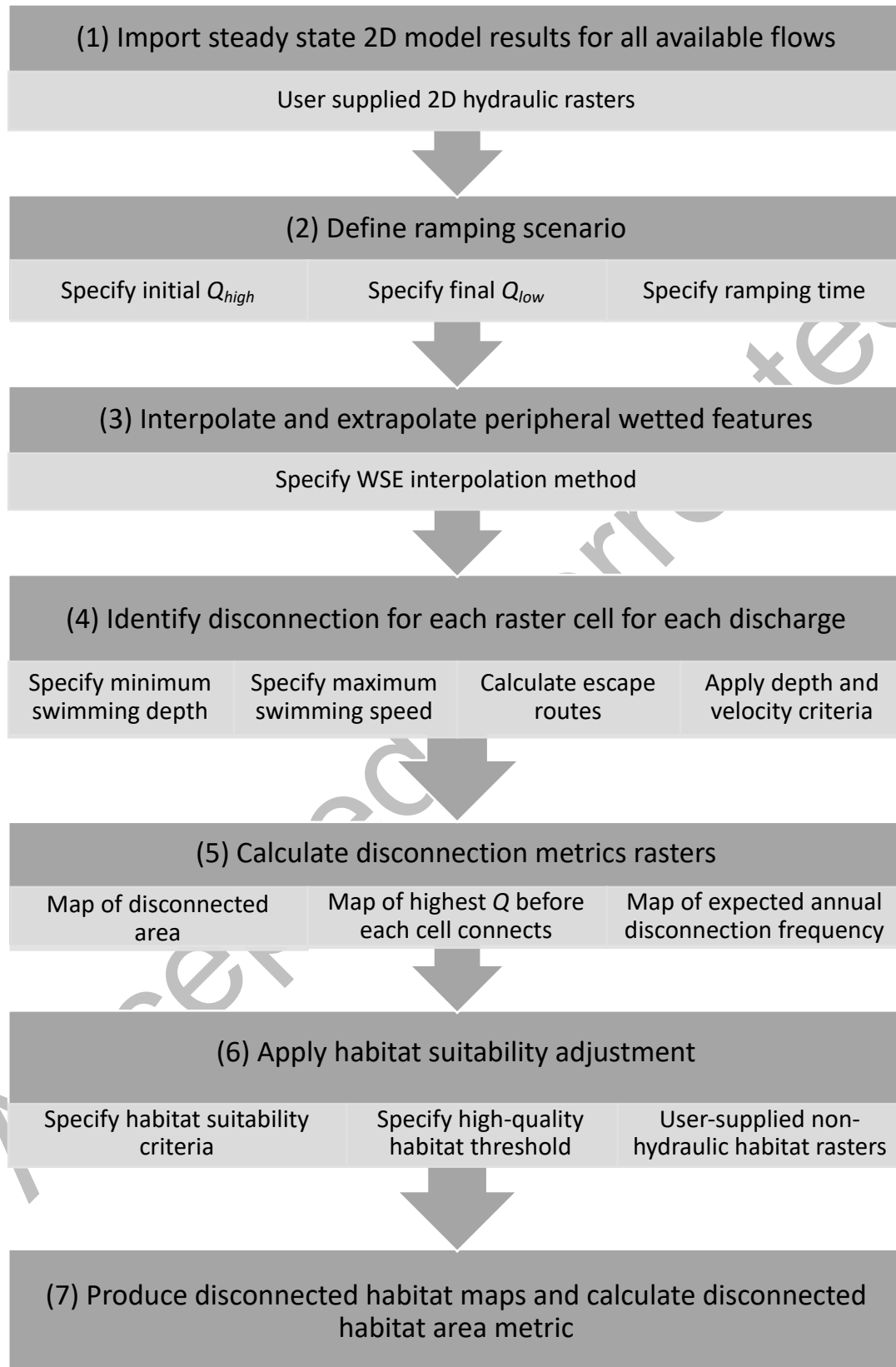
265 A variety of methods have been presented in the literature for quantifying stranding
266 risks with hydrodynamic models. Richmond and Perkins (2009) applied a one-
267 dimensional (1D) steady-state hydrodynamic model and interpolated output WSEs
268 across topography to map dewatered areas and pools that became disconnected. While
269 their method does not incorporate biotic factors or two-dimensional (2D) dynamics, its
270 relative simplicity enables efficient mapping for large domains where little data is
271 available. Another method simulates specific downramping scenarios with 2D unsteady
272 hydrodynamic models, yielding dewatered areas and ramping rates as functions of
273 space and time (Vanzo et al., 2016a; Juárez et al., 2019). Though this method has
274 significantly greater data needs and computational cost, directly simulating unsteady
275 hydraulics readily quantifies ramping rates between downramping scenarios. A third
276 approach by Noack and Schneider (2009) used the habitat-modeling software CASiMiR
277 to produce a habitat suitability index (HSI) as a function of depth, velocity, and substrate
278 with fuzzy sets (Noack et al., 2013). Fuzzy sets were also used to establish a Stranding
279 Risk Index (SRI) as a function of HSI, depth, and ramping rate for each timestep and
280 computational cell in the 2D model domain. SRI is greatest when the HSI is high, depth
281 is less than 0.2 m, timestep downramping rate is critically high (>0.2 m/hr), and/or the
282 area considered becomes hydraulically disconnected from the mainstem. Application of
283 this approach to two boulder-cobble reaches of a regulated alpine river with differing
284 morphologies (a steep, incised gorge and a wide, moderately sloped reach) found the
285 magnitude of stranding risk to be affected most by initial habitat suitability and channel
286 side slope, with steep banks posing lower risk than flatter, more heterogeneous
287 morphologies (Tuhtan et al., 2012). Hauer et al. (2014) applied similar methods,

288 combining 2D unsteady models with HSI and grain size distributions to form a
289 conceptual stranding risk model.

290 These methods are meaningful, but more work is needed. Existing methods either
291 do not account for important biotic variables that can drive stranding risks, or require
292 significant computational cost and technical development, encumbering reproducibility
293 and quick application. While further improvements could be made to the methods
294 presented herein, this study aimed to address some of the existing shortcomings by
295 developing a general framework for stranding risk assessment that leverages steady-
296 state model data often already available to restoration practitioners for other aspects of
297 habitat design and assessment.

298 **4. Novel method**

299 The new method presented herein characterizes fish stranding risks for a given
300 topography, target species/lifestage, and flow ramping scenario (Figure 1). The
301 concepts are explained in this section, while the source code is provided on GitHub,
302 implemented as part of a comprehensive, free, open-source software for river design
303 and ecohydraulic analysis called River Architect. Like other ecohydraulic analysis
304 software, such as CASiMiR (http://www.casimir-software.de/ENG/index_eng.html) or
305 HABBY (<https://github.com/Yannlrstea/habby>), River Architect automates post-
306 processing and analyses of 2D hydrodynamic model outputs, combining them with
307 hydrologic data, topographic data, and biological data to assess habitat abundance,
308 quality, and spatial pattern.



309

310 Figure 1. Conceptual diagram of the new model.

311 4.1. Steady-state 2D hydrodynamic modeling

312 The method applied herein assumes that a downramping scenario can be
313 approximated with a series of successively decreasing steady-state flows.
314 Consequently, we use series of steady-state 2D hydrodynamic model outputs to
315 approximate the hydraulic effects of a given flow reduction (limitations of this
316 approximation are discussed in the supplementary materials). It does not matter what
317 2D model is used, as long as the digital elevation model (DEM), WSE, depth, velocity
318 magnitude, and velocity angle outputs for each flow are available as georeferenced
319 raster data. Angle is clockwise degrees from north, spanning [-180, 180]. This method
320 does not preclude the use of unstructured model domains, though such model outputs
321 must be converted to raster data for post-processing in River Architect. The method
322 could be extended to work directly on vector data for unstructured domains due to the
323 generality of the graph theory-based approach, but the raster-based approach has been
324 implemented herein due to the simpler and more memory-efficient representation of
325 raster data.

326 In River Architect, a ramping scenario is defined by an initial and final steady-state
327 modeled discharge (Q_{high} and Q_{low} , respectively) and a time period over which linear
328 downramping occurs (used to estimate ramping rates). To facilitate use of already
329 generated data, stranding risk assessment is then conducted using all steady-state
330 model outputs within this range (i.e., at discharges $Q_{low}, Q_2, Q_3, \dots, Q_{high}$ where $Q_{low} \leq$
331 $Q_i \leq Q_{high}$ and Q_i is hereafter termed an “intermediate discharge”). To ensure
332 disconnection events are characterized as accurately as possible, it is suggested that
333 the increments between modeled discharges correspond to relatively modest changes

334 in stage. If stage changes are too large between subsequent modeled discharges, then
335 the stranding risk assessment may fail to identify shallower depressions. However,
336 stage changes smaller than the vertical resolution of the DEM are unlikely to identify
337 additional meaningful depressions. Therefore, it is suggested that stage changes
338 between subsequent discharges are on the order of 2-10 times the vertical resolution of
339 the DEM; e.g. for a DEM with centimeter-scale vertical resolution, stage change
340 increments should no greater than ~10 cm for optimal precision.

341 4.2. Interpolating peripheral wetted features

342 The analysis begins by iteratively analyzing the series of steady-state 2D model
343 output rasters (depth, velocity magnitude, velocity angle) from Q_{high} to Q_{low} . To identify
344 ponded areas left behind after a flow reduction, the WSE at each modeled discharge is
345 interpolated and extrapolated across the wetted area at Q_{high} . River Architect enables
346 user selection among four methods: nearest neighbor, inverse distance weighting,
347 ordinary kriging, and empirical Bayesian kriging. After interpolating WSE across the
348 wetted area of Q_{high} to make a new WSE raster for each Q_i , then a corresponding new
349 depth raster is made for each Q_i by differencing DEM and interpolated WSE rasters.
350 Finally, updated velocity rasters are created that assign values of zero velocity in ponds.

351 4.3. Identifying disconnected areas

352 A wetted area is considered disconnected from the mainstem of the river channel
353 during a flow reduction from Q_{high} to Q_{low} if it is not possible for fish of the
354 species/lifestage of interest to reach the main channel from that area at one or more
355 discharge(s) Q_i . This definition is applied to each raster cell, so the resolution of

356 disconnected areas is dependent upon the resolution of the hydrodynamic model and
357 underlying DEM. An area may be considered disconnected not only in the case of
358 physically separate wetted areas, but also if hydraulic barriers are present preventing
359 individuals from moving into the main channel. Depth barriers occur where water is too
360 shallow to swim through. Velocity barriers occur where current (considering both speed
361 and direction) is too strong to overcome. For the purpose of identifying disconnected
362 areas, the main channel is defined as the largest continuous wetted area deeper than
363 the minimum swimming depth (d_{min}) threshold at the final discharge Q_{low} . To define the
364 hydraulic barrier limitations on fish travel for a species of interest in a specific lifestage,
365 threshold values for d_{min} and maximum swimming speed (v_f) are user-defined inputs
366 into River Architect that can be found in the literature (Bell, 1991; Katopodis and
367 Gervais, 2016).

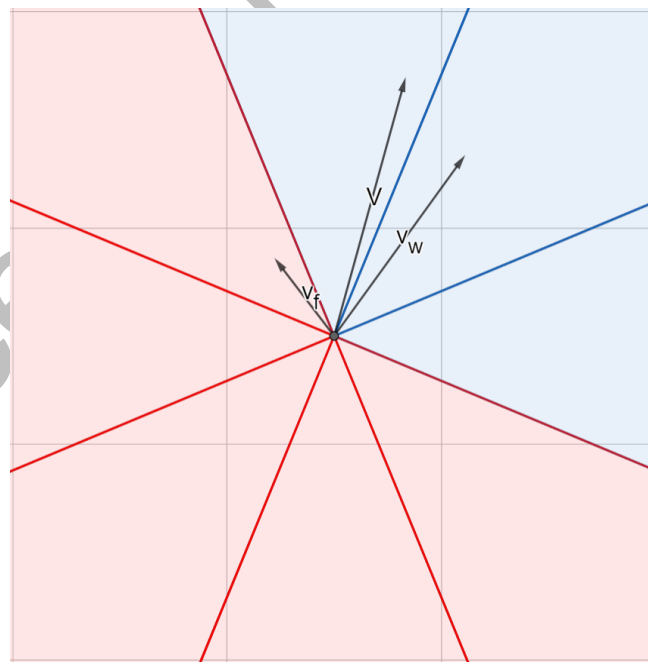
368 Depth and velocity thresholds parameterize the ability of the target fish to travel
369 throughout the river corridor. The following three criteria are assumed for the possibility
370 of travel from cell A to adjacent cell B :

- 371 1. *Domain criterion: both cells A and B are wetted.*
- 372 2. *Depth criterion: depth at cell B is $> d_{min}$.*
- 373 3. *Velocity criterion: the fish can overcome the current at cell A to reach cell B*
374 *traveling at v_f (Figure 2).*

375 Because the velocity criterion is applied to depth- and time-averaged values, it
376 overestimates barrier conditions. In deep enough water, fish could potentially find a
377 sufficiently low velocity path along the bed. Their passage may also be facilitated by

378 turbulent bursts. The use of a single v_f is also an approximation, because fish
379 swimming speeds can be distinguished as sustained or burst, and these can vary with
380 individual fish morphology, physiology, and abiotic factors such as water temperature.
381 Thus, careful selection and sensitivity testing of v_f is recommended to ensure it
382 sufficiently approximates local abilities of target fish species and lifestage.

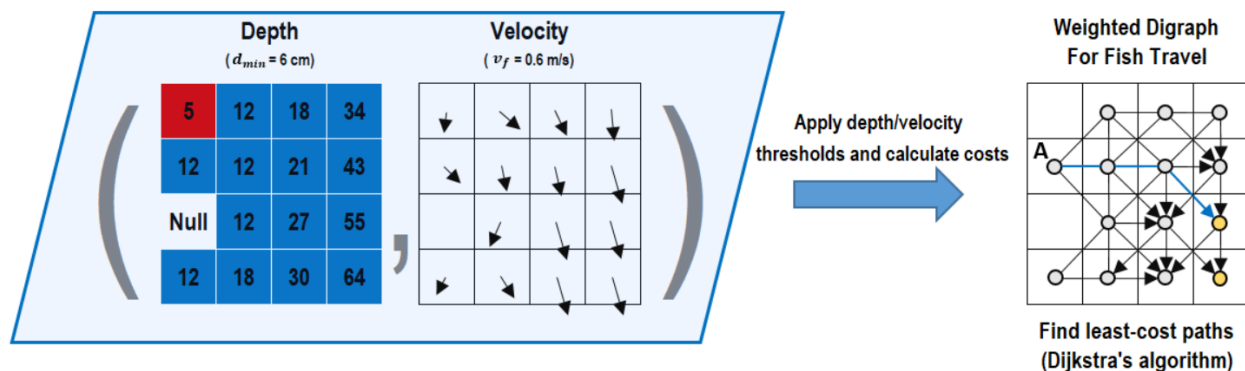
383 The ability for a fish to travel throughout a river corridor at a given discharge can be
384 represented by a (planar) weighted directed graph (Figure 3). Wetted raster cells are
385 represented as graph nodes, and a directed edge from node A to node B indicates that
386 travel is possible from A to B . Edges are created for all pairs of adjacent raster cells
387 satisfying the travel criteria. Weights can then be assigned to each edge based on user-
388 defined factors. For example, this method uses distance between corresponding



389

390 Figure 2: Illustration of the velocity criterion for fish travel. From a cell center, the
391 surrounding area is divided into eight octants corresponding to the eight neighboring
392 raster cells. If it is possible to add the water velocity vector v_w to a fish velocity
393 v_f to yield a vector V falling within a given octant, then the velocity criterion is satisfied

394 for travel to the corresponding adjacent cell. Octants are colored blue or red based on
 395 whether the criteria is or is not satisfied, respectively. Further details of the computation
 396 are provided in supplementary materials.



397

398 Figure 3: An example of the process to create a digraph representation of possible fish
 399 travel. Edges without arrows indicate possible travel in both directions. Gold nodes
 400 indicate cells in the main channel. The shortest path from cell A back to the mainstem is
 401 shown in blue.

402

403 raster cell centers for edge weighting. Another option given appropriate data could be
 404 estimated metabolic cost of traveling from A to B, but this is not implemented.

405 Graph representation of river navigability enables characterization of fish movement
 406 options via path-finding algorithms (McElroy et al., 2012; Etherington, 2016). Dijkstra's
 407 algorithm is a computationally efficient method for finding the shortest (or least-cost)
 408 path between any two nodes of a positively weighted directed graph (Dijkstra, 1959). In
 409 the context of this stranding risk assessment, Dijkstra's algorithm was used to
 410 determine whether a fish can move from any initial wetted cell to the main channel.
 411 Dijkstra's algorithm is applied by defining a starting "root" node, then iteratively
 412 spanning the rest of the graph from the root node. The algorithm yields a shortest-path
 413 tree, which identifies the shortest paths from the root node to all other reachable nodes.

414 After the original directed graph (digraph) is produced, edge contraction is
415 performed to merge all nodes in the main channel, resulting in a single node
416 representative of the main channel. The goal of this path-finding application is to find
417 paths from many possible starting nodes (starting locations of a fish escaping stranding
418 areas) to a single main-channel node. However, Dijkstra's algorithm works in a
419 converse fashion, finding paths from a single starting node to many possible target
420 nodes. Therefore, the novel algorithm first transposes the graph by reversing the
421 directions of its edges, such that Dijkstra's algorithm can then be applied to the main-
422 channel node. By transposing the graph, the main-channel node can be treated as the
423 root node for application of Dijkstra's algorithm, thus finding the shortest path to each
424 possible start node without the need for multiple applications of the algorithm. This
425 yields the shortest path from any possible start node back into the main channel and is
426 one of the core routines that makes the novel algorithm highly efficient. Nodes for which
427 no path exists back into the main channel are then considered to be disconnected at the
428 corresponding discharge.

429 4.4. Disconnecting discharge and frequency

430 After iteratively calculating and mapping the disconnected area at each Q_i between
431 Q_{high} and Q_{low} , each raster cell is queried to determine the highest Q_i at which it is
432 disconnected ($Q_{i,max}$, or $Q_{disconnect}$), yielding a $Q_{disconnect}$ raster. The discharge
433 precision of this raster hinges on how many and which Q_i are modeled. Next, given the
434 river's daily hydrologic record, the expected annual frequency of disconnection (f_d) is
435 computed for each cell by calculating the average number of times per year that flows
436 drop below the $Q_{disconnect}$ value. This analysis may be subset to a seasonal window to

437 align with ecological timing, so disconnection events are only counted for periods
438 relevant to the target species/lifestage. In summary, for the specified downramping
439 scenario the algorithm produces disconnection presence/absence rasters for each Q_i as
440 well as a disconnecting discharge ($Q_{disconnect}$) raster and disconnection frequency (f_d)
441 raster.

442 4.5. Habitat suitability adjustment

443 Disconnection discharge and frequency rasters are helpful in identifying areas with
444 potential stranding risks, but actual stranding also necessitates fish presence. While
445 large areas may disconnect frequently, associated stranding may not have a substantial
446 impact if habitat quality in disconnected areas is poor and has little to no fish presence.
447 Conversely, areas of high habitat quality with high fish density may pose greater
448 stranding risks, even if they are relatively small or rarely disconnect. Habitat suitability
449 modeling commonly implemented in riverine ecosystems (Pasternack, 2019b) serves as
450 a readily available proxy for fish presence likelihood and abundance in an area
451 preceding a disconnection event.

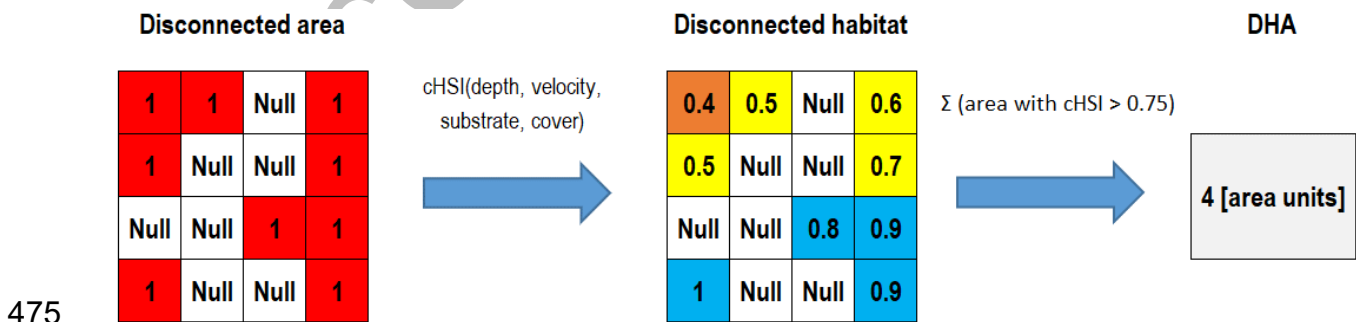
452 Habitat suitability criteria (HSCs) are functions commonly coupled with 2D
453 hydrodynamic model outputs and other data (e.g., substrate size, and cover) in
454 ecohydraulic analysis to create rasters indicating relative quality of each cell for the
455 ecological function the HSC addresses (Pasternack, 2019a). River Architect uses
456 univariate HSCs to produce univariate habitat suitability index (HSI) rasters with values
457 ranging from zero to one for each Q_i . It then combines all the univariate HSI rasters into
458 a single combined HSI (cHSI) raster for each Q_i using either a geometric mean or
459 product calculation (Schwindt et al., 2020) at the user's discretion.

460 To further assess fish stranding, the algorithm next weights disconnected area by
 461 cHSI at Q_{high} to produce a "disconnected habitat" raster. The habitat at Q_{high} is used
 462 because it is assumed to be the physical condition fish have acclimated to in the period
 463 prior to ramping. This raster then quantifies the suitability of habitat area (prior to the
 464 flow reduction) that becomes disconnected by the flow reduction and serves as a
 465 reasonable proxy for spatially explicit fish abundance (and thus stranding risk) in
 466 disconnected areas.

467 In addition to spatially explicit maps of disconnected habitat, a summary metric
 468 herein referred to as disconnected habitat area (DHA) was created to indicate the total
 469 amount of high-quality fish habitat disconnected by flow reduction (Figure 4). This
 470 metric is calculated for each downramping scenario by considering areas to be high-
 471 quality habitat if $cHSI > 0.75$ prior to the flow reduction. In other words:

472
$$DHA = \sum_i H(cHSI_i - 0.75) \cdot A_i \tag{1}$$

473 where the index i denotes a disconnected raster cell, H is the Heaviside step function,
 474 $cHSI_i$ is the combined habitat suitability index at the cell, and A_i is the area of the cell.



476 Figure 4: Illustration of the procedure to calculate disconnected habitat and DHA from
 477 disconnected area.

478 4.6. Ramping rates

479 Assuming that ramping occurs linearly (or in small steps that approximate linearity
480 over the full duration of ramping) over a user-specified time interval, the ramping rate $\frac{dh}{dt}$
481 can be approximated by differencing of depth rasters:

$$482 \quad \frac{dh}{dt} \approx \frac{\Delta h}{\Delta Q} \frac{dQ}{dt} = \frac{(d_{high} - d_{disconnect})}{(Q_{high} - Q_{disconnect})} \frac{dQ}{dt} \quad (2)$$

483 For each raster cell in a disconnected area, Δh and ΔQ are computed using the
484 depth and discharge values at the initial discharge Q_{high} and $Q_{disconnect}$. The rate of
485 change of discharge $\frac{dQ}{dt}$ is constant due to the assumption of linear downramping. The
486 result is the average ramping rate raster from the beginning of the downramping until
487 disconnection on a per-cell basis.

488 4.7. Limitations

489 Due to the approximation of unsteady flow downramping and resulting hysteresis
490 effects (i.e., water depth is higher in the falling limb of a hydrograph), this method is
491 expected to overestimate the average ramping rate. Thus, in locations where ramping
492 rates are expected to be the dominant driver of stranding risks, unsteady modeling may
493 be necessary to accurately characterize the impact of ramping rate on stranding risks.
494 Further considerations of the limitations of this approach are described in the
495 supplementary materials. Other factors observed to partially influence fish stranding that
496 are not accounted for by the applied method include water temperature and time of day.
497 Because water temperature is partially independent of hydraulics, more data collection
498 would be required to know what water temperatures are present during a downramping

499 scenario of interest.

500 For active monitoring of stranding risks, this method could be used in conjunction
501 with water temperature measurements and/or physical models to determine precisely
502 when water temperatures are low enough to contribute to stranding risks. While it is
503 assumed that fish have had sufficient time to acclimate to the hydraulic conditions of
504 Q_{high} prior to downramping, the exact duration of this wetted history is not explicitly
505 considered by this method. As a result, effective active management should exercise
506 more caution when downramping after long periods of high flows, when juveniles may
507 exhibit more fidelity to habitat areas (Saltveit et al., 2001; Halleraker et al., 2003; Irvine
508 et al., 2009; Irvine et al., 2014). Because River Architect is freely available and open-
509 source, existing code can be modified to incorporate these considerations and expand
510 the suite of freely available tools for river design, analysis and management.

511 5. Conclusions

512 The phenomenon of fish stranding is complex and not yet understood in terms of the
513 intertwining of biological and physical dynamics. In the absence of certain science,
514 certain prediction is impossible. However, humanity continues to tinker with rivers in
515 ways that often increase stranding risk. Other management needs may outweigh
516 stranding concerns, but at least decision-makers should have information to guide their
517 evaluations. While some methods already exist, this study offers a novel and highly
518 efficient algorithm to facilitate identification of disconnected areas in a regulated river
519 and quantify relevant parameters determining the severity of potential pool stranding
520 events. When used along with the other components of the River Architect software

521 suite, a thorough consideration of geomorphic sustainability, physical habitat, fish
522 stranding risk, and project financial cost may be possible in early design stages of river
523 projects in support of design revision to achieve a better ecological outcome. The
524 software is also useful as a hypothesis generation tool to facilitate and focus
525 observational campaigns.

526 **Authorship contribution statement**

527 K.G. Larrieu: literature review, conceptualization, method, software, original draft; G.
528 B. Pasternack: conceptualization, 2D modeling and ecohydraulics theory and practice,
529 articulation about types of stranding, original draft editing, producing draft into journal
530 manuscript, funding acquisition, supervision, project administration; S. Schwindt: Python
531 coding, River Architect coding, coding supervision, methodological development, journal
532 manuscript editing.

533 **Declaration of Competing Interest**

534 The authors declare that they have no known competing financial interests or
535 personal relationships that could have appeared to influence the work reported in this
536 paper.

537 **Supplementary materials**

538 Supplementary material associated with this article can be found on the journal's
539 website.

540 **Data Availability Statement**

541 This publication presents a literature review and algorithm development, so there is
542 no new observational data. Conforming to open-source standards, the algorithm is
543 available to the public at <https://riverarchitect.github.io/>.

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