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1 **Automated analysis of lateral river connectivity and fish stranding risks– Part 1:**

2 **Review, theory and algorithm**

3

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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

6

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.

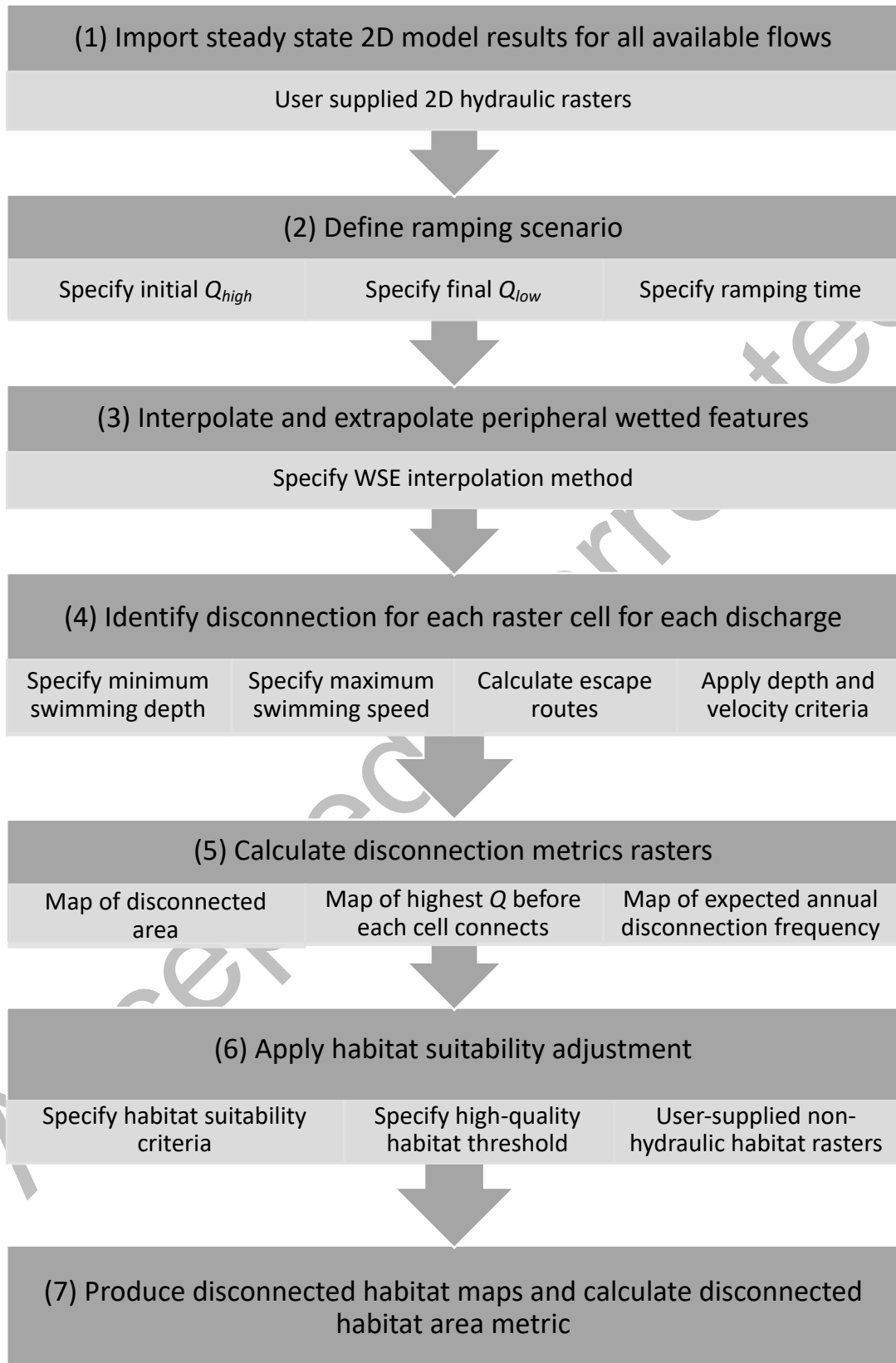264   **3.   Stranding risk assessment methods**

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/YannIrstea/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.

(1) Import steady state 2D model results for all available flows
User supplied 2D hydraulic rasters

(2) Define ramping scenario		
Specify initial Q_{high}	Specify final Q_{low}	Specify ramping time

(3) Interpolate and extrapolate peripheral wetted features
Specify WSE interpolation method

(4) Identify disconnection for each raster cell for each discharge			
Specify minimum swimming depth	Specify maximum swimming speed	Calculate escape routes	Apply depth and velocity criteria

(5) Calculate disconnection metrics rasters		
Map of disconnected area	Map of highest Q before each cell connects	Map of expected annual disconnection frequency

(6) Apply habitat suitability adjustment		
Specify habitat suitability criteria	Specify high-quality habitat threshold	User-supplied non-hydraulic habitat rasters

(7) Produce disconnected habitat maps and calculate disconnected habitat area metric

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 $Q_{low}, Q_2, Q_3, \ldots Q_{high}$ 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

16

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
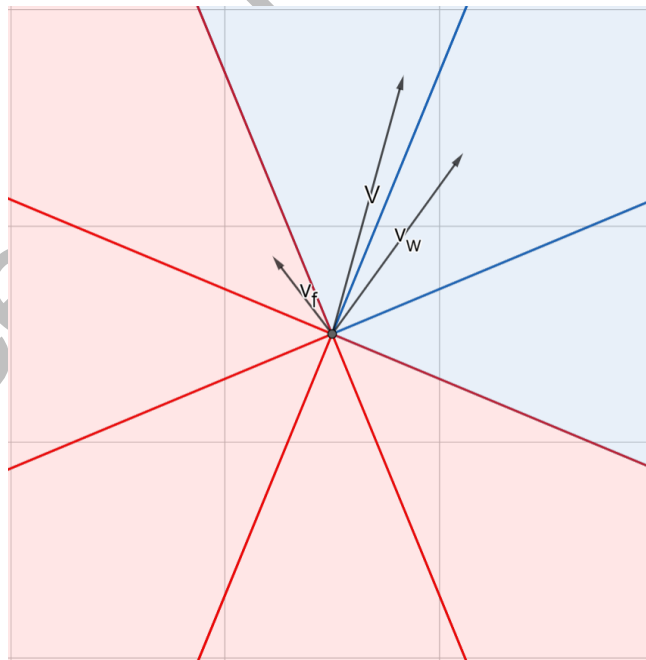
389

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 vector
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.
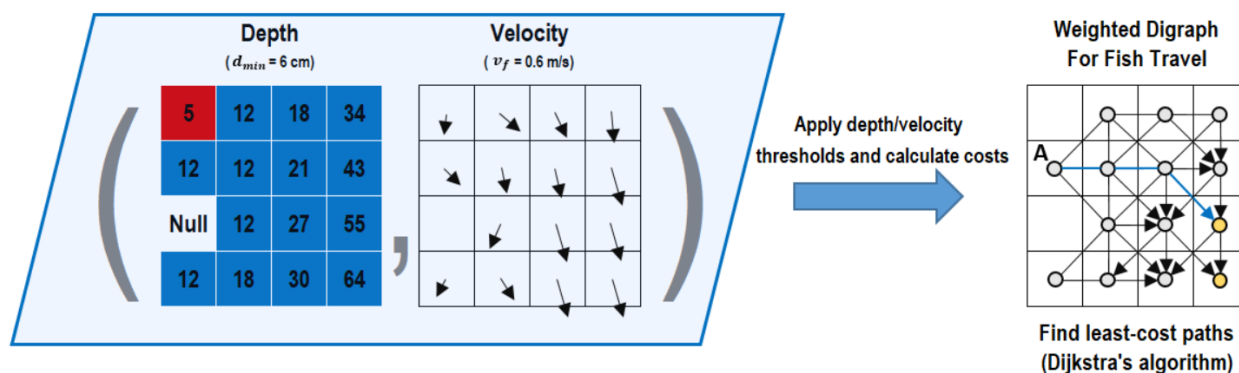
397

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
$$DHA = \sum_i H(cHSI_i - 0.75) \cdot A_i \qquad (1)$$

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.

475
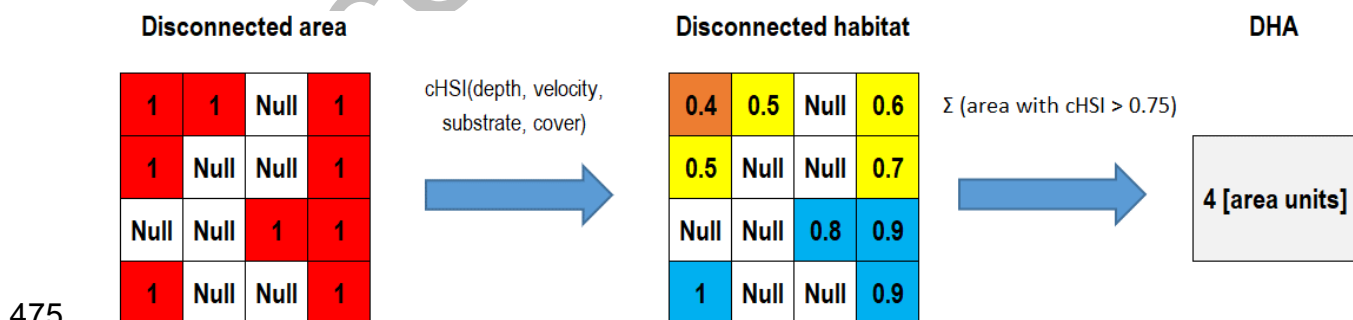
476   Figure 4: Illustration of the procedure to calculate disconnected habitat and DHA from
477   disconnected area.

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

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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