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## Hydropwer flushing events cause severe loss of macrozoobenthos in Alpine streams

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- Key points
- Flow abstraction at hydropower intakes in a glaciated stream improves downstream low flow habitat conditions compared to natural conditions
- Intake sediment flushing degrades habitat due to rapid discharge rise, increase in turbidity, reduction in euphotic depth, bed instability
- Sensitivity to flushing relates to discharge rise speed as suitable habitat moves across the floodplain faster than likely benthos migration

**18 Abstract**

19 Alpine hydroelectric power exploitation often aims to increase the volume of water stored  
20 behind impoundments which may be achieved through flow abstraction and lateral transfer  
21 to storage. Intakes are designed to separate water (for transfer) from sediment which  
22 accumulates in settling basins and may be flushed sometimes multi-daily in glaciated basins.  
23 In some countries (e.g. Switzerland) intakes drain a greater basin area than impoundments  
24 yet legislation designed to improve instream ecosystems impacted by hydropower has  
25 almost entirely ignored them. Some research suggests that such streams have exceptionally  
26 low abundance and diversity of macroinvertebrates for some kilometres downstream of the  
27 intake flushing at high frequency in summer, but that populations can recover rapidly as  
28 soon as flushing frequency decreases in early autumn. However, such patterns could also  
29 result from natural flow variability, sediment transport and morphological change in glacier-  
30 fed streams. Here, we combine field measurements with habitat modelling to assess the  
31 impacts of sediment flushing on macrozoobenthos as compared to what might be expected  
32 in a natural, morphologically dynamic Alpine stream. We show that water abstraction in  
33 itself could improve habitat conditions because it increases the importance of less turbid and  
34 groundwater/unregulated sources. However, intake flushing leads to short duration,  
35 sediment-laden flows that can destabilise substantial areas of the stream bed and cause rates  
36 of lateral displacement of habitat much greater than the possible response by  
37 macroinvertebrates. Our results challenge current emphasis on minimum flows in such  
38 streams and argue that much more emphasis needs to be placed on sediment management.

39

## 1 Introduction

The impact of changing streamflow regime on physical habitat has been the subject of numerous studies (e. g. Parasiewicz et al., 1998; Bunn & Arthington, 2002; Poff & Zimmerman, 2010). One of the current challenges is to optimize flow conditions to reduce the impact of flow abstraction and return on stream ecology whilst also meeting water demand. The science of the required environmental flow releases, also known as e-flows (The Brisbane Declaration, 2007), has developed to identify the management options needed to restore the ecological functions of a stream (Dyson et al., 2003; Petts, 2009; Arthington, 2012). Traditional focus was upon definition of a minimum flow to be released by hydroelectric power plants (Dyson et al., 2003). Wider research has shown that a focus on minimum flows is insufficient and that the required flow should mimic as far as possible the natural flow regime, notably through the incorporation of flow variability (Poff et al., 1997). Poff et al. (1997) argue that incorporating variability into flow requirements should include consideration of flow magnitude, frequency, duration, timing, and rate of change, and this is reflected in methods and tools that take now into consideration a broader range of flow conditions (e.g. Caissie & El-Jabi, 2003; Arthington et al., 2006).

Hydroelectric power (hydropower) represents a particular challenge for specifying e-flows because electricity generation potential generally scales linearly with the volume of water abstracted and production scales linearly with the volume of water released. Downstream of water intakes or dams, streams may lose some or all of their natural variability, even if they are subject to a minimum flow requirement. Downstream of electricity generation stations, water may be returned to the river, leading to multiple daily flow peaks (“hydropeaks”; Bratrich et al., 2004) that follow consumer demand for electricity rather than natural flow variability. The stress created by hydropower-induced low flows and hydropeaking has been shown to have major impacts upon stream ecosystems (e.g. Brooker & Hemsworth, 1978; Cushman, 1985; Moog, 1993; Lauters et al., 1996; Céréghino & Lavandier, 1998; Smokorowski et al., 2011; Schmutz et al., 2015; Schülting et al., 2016, 2019).

In Alpine hydropower systems, there is a third type of impact associated with flow intakes (Gabbud & Lane, 2016). Traditionally, hydropower is thought to lead to sediment starvation downstream of dams, and associated problems of erosion and bed armouring (Petts & Gurnell, 2005). However, intakes are different. At intakes, water is abstracted and stored at altitude either to maintain its hydraulic head or for transfer to adjacent valleys for storage behind dams, so increasing their power generation potential. Such intakes seek to separate water and coarser sediment (typically sand and coarser), the sediment accumulating in a settling basin until sediment storage capacity is reached and the basin must be flushed. In some Alpine environments, such as in Switzerland, the area of basins subject to water intake impacts is greater than dams (Margot et al., 1992). Depending on their capacity, in high altitude systems associated with glaciated basins, flushing may be more than daily in frequency during peak glacier melt (e.g. Gabbud et al., 2019). The result is “hydropeaking” but this is not driven by electricity demand, rather by sediment accumulation in intakes, and their capacity to store sediment.

Flushing has two fundamental differences compared with conventional hydropeaking: (a) the release involves a very rapid rise and fall in discharge as the peaks are designed to flush out sediment; the result is a short duration flow events (typically less than one hour in duration); and (2) the water that is flushed contains exceptionally high sediment loads (Gurnell, 1983; Gurnell & Warburton, 1990) that is transferred through the intakes that maintain sediment delivery downstream, unlike impoundments which interrupt sediment delivery. Both of these changes may impact negatively on the habitat available to

88 macrozoobenthos. It is well-established that different macrozoobenthos families, genus and  
89 species have different hydraulic preferences, commonly expressed as a shear stress (e.g.  
90 Möbes-Hansen & Waringer, 1998; Méricoux & Dolédec, 2004; Dolédec et al., 2007;  
91 Méricoux et al., 2009). As discharge rises, the spatial distribution of shear stress will change  
92 and the sites of suitable habitat will shift laterally. This relationship will depend on stream  
93 bed bathymetry; for example, in a braided river where changes in discharge are  
94 accommodated primarily by changes in width, suitable habitat (in terms of shear stress) in  
95 the primary channel at low flows will move into secondary channels at high flows.  
96 Implicitly, then, braided rivers should be resilient to flow changes provided  
97 macrozoobenthos can migrate laterally. If the discharge rise (or fall) is too great then  
98 required migration rates may be too high, and the implicit resilience cannot be realised.

99 High sediment loads, mobilized through shear stress increases during flushes, may also  
100 negatively impact available habitat. Data from hydropower impacted streams suggest almost  
101 continual reworking of the primary channel bed due to flushing (Lane et al., 2014; Bakker et  
102 al., 2019). Both erosion and deposition may lead to a reduction in the organic matter on the  
103 stream bed needed to sustain macrozoobenthos, due to either scour or burial (Orr et al.,  
104 2008; Müller et al., 2009; Fuller et al., 2011; Cullis et al., 2014). Scour can lead to direct  
105 wash out of organisms whilst burial can make emergence difficult (Wood et al., 2005;  
106 Conroy et al., 2018).

107 The sensitivity to these two sets of processes may be particularly great as water intakes are  
108 commonly at higher altitudes (typically at altitudes higher than dams to facilitate water  
109 transfer). Flushing tends to take place in systems that may be more environmentally  
110 marginal (e.g. colder, Milner & Petts, 1994; or more turbid, Milner et al., 2001) and hence  
111 more sensitive to perturbation. Recently, Gabbud et al. (2019) found a clear correlation  
112 between flushing event frequency and macroinvertebrate community abundance and  
113 diversity in an Alpine stream subject to flow abstraction and flushing. Whilst the stream had  
114 exceptionally low abundance and diversity of macroinvertebrates during the high frequency  
115 flushing period in summer, populations rapidly recovered as soon as the frequency  
116 decreased in early autumn. Rapid recovery was attributed to the ability of the main stream to  
117 recruit individuals from lateral streams. The result was more diversified and more abundant  
118 communities in winter and early spring, aided by the fact that the freezing of flow at the  
119 stream bed is prevented by krenal water supply. Diversity and population size both fell  
120 rapidly when the flushing of sediment accumulated in intakes recommenced in late spring  
121 and early summer.

122 An apparent sensitivity to flushing would follow from the two broad flushing impacts, rapid  
123 habitat displacement and rates of geomorphic change, identified above. However, this raises  
124 a challenge of attribution. A glacially-fed stream without hydropower impacts would have a  
125 natural variability in both discharge and sediment load at the sub-daily scale, variability that  
126 may also lead to habitat displacement and channel instability (Gurnell et al., 1999). Burgherr  
127 and Ward (2001) and Füreder et al. (2001) found that macroinvertebrate densities were  
128 higher in late spring and early autumn than they were in summer, and they attributed this to  
129 periods of less frequent stream bed perturbation, low rates of sediment transport, low  
130 turbidity, low bed shear stress, as well as the consequences of these for organic matter  
131 production and accumulation. Indeed, some species are well-adapted to winter conditions (e.  
132 g. Robinson et al., 2001; Brown et al., 2015). Thus, whilst contradicting conventional  
133 assumptions (OFEV, 2010), it is quite possible that the observation that winter has more  
134 diverse and abundant communities in glacially-fed streams is related to natural flow,  
135 sediment load and stream bed perturbation processes rather than hydropower impacts.

136 Given this question of attribution, the aims of this paper are two-fold. The first is to assess  
137 the extent to which the more diversified and abundant communities in winter and early  
138 spring, as compared with summer (Gabbud et al., 2019) can be attributed to what might be  
139 expected naturally in a morphologically dynamic Alpine stream as opposed to one impacted  
140 by flushing of sediment from intakes. Such evaluation is crucial as the percentage of basins  
141 impacted by sediment flushing in Alpine environments can be significant. The second is to  
142 use the results from this approach to reflect upon the changes in environmental flow  
143 frameworks needed in streams draining glaciated basins where the water that an  
144 environmental flow might deliver is typically glacier derived and so cold with high  
145 suspended sediment concentration. The design of optimal hydropower management systems  
146 to improve the ecosystems associated with such streams remains largely overlooked in  
147 management terms.

148 To address this aim, we revisit the Borgne d'Arolla (Gabbud et al., 2019) and focus upon a  
149 braided reach of Alpine stream downstream of glaciated basins with flow intakes,  
150 representative of many environments subject to an intake flushing regime. We combine field  
151 measurements with habitat modelling for macroinvertebrates to identify how intake flushing  
152 impacts instream habitat variation in space and time as compared to what would be expected  
153 under naturally varying flow and sediment transport conditions. The results are then used to  
154 test the following hypotheses: (1) that habitat suitable for macrozoobenthos potentially  
155 exists at all possible flows in the studied stream; (2) that whilst intake flushing produces  
156 increases in total habitat availability (as more floodplain is inundated) the sites of suitable  
157 habitat are displaced laterally more rapidly than in a natural stream; and (3) that this  
158 combines with significant geomorphic impacts (scour and burial) of suitable habitat.

## 160 2 Methods

### 161 2.1 Study site

162 This study was conducted in a 635 m long braided reach of the Borgne d'Arolla in the south-  
163 west Swiss Alps based on data acquired and modelled for the period 7<sup>th</sup> to the 15<sup>th</sup> August  
164 2016. This stream is defined by a glacial regime and is fed by a series of both kryal and nival  
165 tributaries. The Borgne d'Arolla originates from a number of glaciated tributaries which  
166 feed two major intakes, the Bertol Inférieur (or Lower Bertol) and the Tsijiore Nouve  
167 (Bakker et al., 2018), 5 km and 3.5 km upstream of the study reach; and one which is much  
168 smaller, the Pièce intake (Figure 1). Initial data collection (2014 and 2015) using a  
169 continuous record of stream temperature suggested that during the summer, daily water  
170 temperatures in the study reach were in the range of 5° to 10° C, and hence the chosen site is  
171 sufficiently far downstream from the glacier for water temperature not to be a limit on  
172 macroinvertebrate development (Milner & Petts, 1994; Milner et al., 2001).

173 This paper focuses on a braided reach next to the village of Satarma, at an altitude of 1'850  
174 m above sea level, which comprises an alluvial plain with a mixed gravel-sand, multi-thread  
175 stream (Figure 1). The upstream and downstream ends of the reach are bound by bridges that  
176 locally canalise the stream. The reach slope is relatively uniform at  $0.035 \pm 0.004$  and the  
177 mean grain size is  $0.05 \pm 0.02$  m (Bakker et al., 2018). The mean wetted width at summer  
178 baseflow is  $7.7 \pm 2.5$  m; during a flushing event, this rises to  $28.4 \pm 11.7$  m. The reach lies just  
179 below the tree-line and so at the Montane-Alpine interface. There are isolated trees within  
180 the floodplain, although many of these are dying as a result of sediment-related perturbation.

181 The reach has been aggrading and steepening since hydropower exploitation began in the  
182 1960s, and notably since the 1990s, with mean bed elevations rising over this period by 2 m  
183 at the upstream end of the reach and 1 m at the downstream end (Bakker et al., 2018). This  
184 bed level rise reflects the operation of hydropower intakes upstream. At each intake (circles  
185 on Figure 1), there is a small volume (100 to 200 m<sup>3</sup>) settling basin designed to trap gravel  
186 and coarser sediment. Then, water passes underground to a second settling basin beneath the  
187 intake which can trap up to 10 m<sup>3</sup> of sand (Bezinge et al., 1989). The water, which now only  
188 contains washload, passes through tunnels to an adjacent valley where it is stored behind a  
189 large dam (see Bakker et al., 2018 for more details). Currently, all water is abstracted and  
190 there is no minimum residual flow requirement. The gravel and sand settling basins have  
191 finite capacity and so have to be flushed when full. This is flushed by temporarily halting the  
192 transfer of water to the reservoir, and using the flow instead to wash the sediment out of  
193 either the gravel basin or the sand basin and into the main stream. This maintains sediment  
194 connection and the associated flushing of the intakes produces enough water to move the  
195 majority of flushed sediment downstream through the system (Bakker et al., 2018). As is  
196 common in the presence of aggradation, the stream bed does not armour.

197 Between the intakes and the reach there are a number of unglaciated and unregulated  
198 tributaries that join the stream and these, along with some groundwater (i.e. krenal) supply,  
199 are sufficient to maintain a baseflow of between 0.5 and 0.8 m<sup>3</sup>s<sup>-1</sup> at the study reach with  
200 negligible turbidity. When the intake is flushed, the discharge rises rapidly in the Satarma  
201 reach, between 40 minutes (typical of Tsijiore Nouve, Figure 1) and 55 minutes (typical of  
202 Bertol Inférieur) after flushing begins. There is relatively little attenuation between the  
203 intakes and Satarma.

## 204 **2.2 Turbidity and discharge measurements**

205 A pressure transducer with a built-in temperature sensor (Campbell Scientific CS451 (USA),  
206 water level resolution 0.0035% FS; temperature precision  $\pm 0.2^{\circ}\text{C}$ ) and infrared-based  
207 downward looking turbidity probes (Campbell OBS-3+ (USA); NTU precision 0.5 NTUs)  
208 were installed in the canalised section at the upstream and downstream end of the study  
209 reach (Figure 1). The turbidity probes were installed with a low and high signal range  
210 ranging up to 1000 and 4000 NTUs respectively. Data were logged to a river bank mounted  
211 Campbell Scientific data logger (CR200X, USA). The loggers were programmed to take 10  
212 measurements with 2 second intervals each minute. The mean and standard deviation were  
213 registered every 10 minutes under normal flow conditions (low concentration;  $< 50$  NTUs)  
214 and each minute during a flushing (high concentration;  $\geq 50$  NTUs) event.

215 We chose to use the downstream data as the data from upstream probes were sensitive to  
216 local scour and fill of the stream bed. Determining an accurate stage-discharge rating curve  
217 at this site is difficult because of frequent changes in the river bed cross section. However,  
218 all of the upstream intakes have built in broad-crested weirs. These measure discharge to a  
219 high precision (c. 0.1 m<sup>3</sup>s<sup>-1</sup>) for regulatory reasons. We use data from them to calibrate the  
220 stage record. During flushing, the flow intake discharge drops rapidly to zero, rising back to  
221 previous values as soon as flow abstraction recommences. As the natural rates of change of  
222 discharge in these systems are relatively slow, except during storm events, it is possible to  
223 interpolate the discharge that would have been abstracted if flushing had not occurred, and  
224 hence the total volume of water flushed downstream (e.g. Gabbud & Lane, 2016; Bakker et  
225 al., 2018).

226 As a first step, we identified those flushes, 11 in total during the study period, where  
227 discharge interpolation was straightforward (i.e. flushes within a period of steadily rising or

falling discharge) and where there was no impact on the events from the opening of intakes further upstream. For each such flush,  $j$ , we calculated the volume of water released ( $V_j$ ). We then calculated an index ( $I$ ) above baseflow stage ( $S_b$ ) from the stage data at our measurement sites up and downstream of the investigated reach:

$$I_j = \sum_{i=1}^n (S_i - S_b)^{1.67} \quad [1]$$

where :  $S_i$  = stage at time  $i$ ; and  $n$  = number of minutes where the stage was impacted by the flush. As this was for a canalised section, with near vertical side walls, and as all flows remained within bank, we would expect water level to increase at approximately the power 1.67. We then regressed  $V_j$  on  $I_j$  to produce the following relationship:

$$V_j = 6191.1I_j + 435.0 \quad [2]$$

with an  $R^2$  of 88.7%. We then applied this to each  $S_i$  record in the series by replacing  $I_j$  with  $S_j$ , to give an estimate of the volume of water passing through the reach per minute, from which we could calculate a discharge ( $\text{m}^3\text{s}^{-1}$ ). As this method explicitly uses the form of the hydrograph measured at the station to distribute the water due to the flush, it deals with attenuation between each intake and the station.

This method does not include the baseflow below the level of the gauge height  $S_b$  and which will be constant provided the gauge remains submerged, which was the case throughout the study period. As noted above, the bathymetric data collection focused on a period of lowest flow and this was coincident with the gauge being only just submerged. Thus, to estimate the low flow we used the inundated extent to calibrate the 2D hydraulic model described below, including both baseflow discharge and roughness as model calibration parameters. This gave us an estimated baseflow of  $0.55 \text{ m}^3\text{s}^{-1}$ , which we added to all flows estimated using [2].

Finally, as we wanted to compare the habitat associated with the natural flow regime and with the abstraction and flushing regime, we took the natural flows at the intakes and combined these with the  $0.55 \text{ m}^3\text{s}^{-1}$  baseflow, but delayed by the travel time that the difference between the flushing time and the stage record suggested. Thus, the natural flow regime is not corrected fully for attenuation effects between the intakes and the study reach. This is a limit to the comparison we make, but we note that: (1) the flow minima under natural conditions would have always been greater than  $1.4 \text{ m}^3\text{s}^{-1}$ , with the associated flow depth reducing attenuation, as compared with a flush release to the lower baseflow of  $0.55\text{m}^3\text{s}^{-1}$ ; and (2) the failure to represent attenuation is likely to make the natural flow regime more severe than it would otherwise have been (with additional attenuation effects).

### 2.3 Turbidity and organic matter availability

Temperature and turbidity themselves may have direct impacts upon the harshness of Alpine streams for macroinvertebrates. They may also be important for organic matter availability, particularly turbidity, and hence macroinvertebrate food sources. In practice, turbidity can have two different effects (Uehlinger et al., 1998). First, turbidity should reduce the amount of photosynthetically active radiation (PAR) that reaches the stream bed causing reduced primary production on the beds of glacially-fed streams during periods when kryal sources

are dominant over other sources (Uehlinger et al., 1998; Füreder, 1999; Burgherr & Ward, 2001; Füreder, 2007). Second, as the increase in turbidity in glacially fed streams is associated with increased suspended sediment concentrations, and these are commonly abrasive, turbidity could also lead to greater scour of benthic algae, such as periphyton, on the stream bed. We address this process below.

To calibrate the turbidity probes described above, we obtained thirteen water samples for the determination of suspended sediment concentration. The samples had concentrations ranging from 170 to 1200 mg l<sup>-1</sup>. The calibration relationship between turbidity and suspended sediment concentration was weakly non-linear (best represented by an exponential) in the range of concentrations considered here, reflected the expected response of this probe, and had an R<sup>2</sup> of 97.6%. The focus of the interpretation is on turbidity because of its control on PAR, but we report suspended sediment concentrations to help compare our results with others in the literature.

Relatively small increases in turbidity have been shown to result in quite major reductions in PAR in glacially-influenced water bodies (Lloyd et al., 1987; Uehlinger et al., 1998; Rose et al., 2014). To interpret the effects of hydropower management upon PAR we develop a simple index of possible light attenuation due to glacially-derived suspended sediment load. Research has shown (e.g. Modenutti et al., 2000; Rae et al., 2001; Rose et al., 2014) that glacially-derived suspended sediment load can have a significant influence on light attenuation but that this is commonly dependent upon catchment characteristics (e.g. geology) which influence the nature of the suspended sediment produced. In the absence of detailed data on turbidity impacts on light attenuation, we chose a relatively simple index to gauge how measured turbidity values ( $T$ ) might interact with flow variability to determine the spatial extent of the euphotic zone, where the PAR at the stream bed ( $I_d$ ) is at least 1% of the incoming PAR,  $I_o$  (Kirk, 1994). Following Lloyd et al. (1987) we apply

$$I_d = I_o e^{-Kd} \quad [3]$$

where  $d$  is depth. Given the lack of data needed for more physically based approaches (e.g. Gallegos, 2001) we use empirical relationships for the attenuation coefficient  $K$  obtained for glacially-fed lakes in Alaska, Canada and New Zealand:

$$\begin{aligned} K &= 0.064 T - 0.093 \text{ (Lloyd et al., 1987)} \\ K &= 0.041 T + 0.448 \text{ (Rose et al., 2014)} \\ K &= 0.037 T + 0.093 \text{ (Rose et al., 2014)} \end{aligned} \quad [4]$$

The relationships [3] and [4] allow us to convert the measured turbidity time-series into a corresponding euphotic depth ( $d_e$ , i.e. when  $I_d/I_o = 0.01$ ). By convolving this with the distribution of inundated hydraulically-modelled depths for the study reach (see below) and the time-series of discharge at Satarma, we could determine time-series of the inundated area of the stream and the percentage with water depths lower than the euphotic depth.

## 2.4 Stream and floodplain bathymetry and 2D hydrodynamic modelling

The 2D hydrodynamic modelling was performed with the BASEMENT v2.7 hydraulic model developed by the ETHZ (ETHZ, VAW, 2017). The model solves the depth-average form of the Navier-Stokes approximation of the system of differential equations for mass



317 and motion conservation on a finite element mesh (Vetsch et al., 2017). Reynolds  
318 decomposition with a zero order eddy viscosity turbulence model was used to represent the  
319 effects of turbulence. A Manning type friction relationship was used with a quadratic friction  
320 law to determine the bottom shear stresses. Model solution used an exact Riemann solver.  
321 Time steps were set automatically to satisfy the Courant-Friedrichs-Lewy condition. The  
322 downstream boundary used a depth-discharge relationship based upon the Manning  
323 equation. This boundary was set downstream of the zone of interest in this study such that  
324 the boundary condition had no impact upon the model results used in subsequent analyses.  
325 In all model runs a steady discharge was applied at the upstream section and the model run  
326 to steady state, with the mass balance error (the difference between mass inflow rate and  
327 mass outflow rate) less than 0.5%.

328 The model relies upon bathymetric data to establish the finite element mesh. A Digital  
329 Elevation Model (DEM) of the area was constructed by UAV imagery with the  
330 photogrammetric software Pix4Dmapper v.3.3<sup>1</sup> from images taken with a SenseFly ebee  
331 (plane) drone. Images were obtained on the 12<sup>th</sup> of August 2016 using two flying heights  
332 and at an angle of 7°, with a superposition of images of at least 70%, to reduce the effects of  
333 distortion (doming effects) (Carbonneau & Dietrich, 2017). The georeferenced DEM  
334 resolution was 0.03 m. A bathymetric correction was applied to the DEM to correct for the  
335 effects of refraction using Carbonneau & Dietrich (2017), developed for SfM following the  
336 work of Westaway et al. (2000, 2001). The DEM of the channel bathymetry was resampled  
337 at 0.5 m resolution in ArcGIS<sup>2</sup> for the hydraulic modelling, a resolution that reflected a  
338 compromise between representation of topographic detail and computational efficiency. This  
339 DEM was then converted into an unstructured triangular mesh through the BASEmesh  
340 module included in the open source geographic information system software Quantum GIS<sup>3</sup>.  
341 The computational mesh of triangular elements was the basic grid used for the simulations  
342 (Vetsch et al., 2017). Bathymetric data were also acquired on the 26<sup>th</sup> of July 2016 and this  
343 allowed us to have a wider sense of the extent to which the floodplain is reworked under the  
344 current flow regime.

345 Initial testing of model parameters suggested that turbulent viscosity and friction slope at the  
346 downstream end of the simulation were much less important influences upon model  
347 predictions than Manning's  $n$ . Thus, to calibrate the model, focus was placed upon  
348 Manning's  $n$  and discharge, treating baseflow discharge as unknown given the measured  
349 stage from the pressure transducer described above. We included  $n$  as a calibration  
350 parameter (constant value) as  $n$  is likely to be an effective parameter in this case (that is the  
351 values needed to optimise the model may be different to those measured from grain-size,  
352 because other processes are being represented in the model; see Lane, 2014). We applied  
353 uniform values of Manning's  $n$ , given evidence from Bakker et al. (2019), from 0.04 to 0.08.  
354 After some initial simulations, we estimated baseflow to be optimised (against inundation  
355 extent and depth, see below) between 0.50 and 0.60 m<sup>3</sup>s<sup>-1</sup> regardless of Manning's  $n$  used,  
356 and so we undertook preliminary simulations with 0.50, 0.55 and 0.60 m<sup>3</sup>s<sup>-1</sup>.

357 The bathymetry optimization process allows us to assess spatial patterns of inundation as  
358 one output of the correction is a map of inundation extent. As this is a multi-thread reach,  
359 small changes in water level, whether due to roughness or discharge, tend to lead to large  
360 changes in inundation extent so making inundation extent ideal for model calibration (see  
361 also Smith, 1997; Simeonov et al., 2013; Williams et al., 2013; Pan et al., 2016; Garambois

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<sup>1</sup> See <http://www.pix4d.com/> for further information

<sup>2</sup> See <http://www.arcgis.com/index.html> for further information

<sup>3</sup> See <http://www.qgis.org/en/site/> for further information

et al., 2017; Benjankar et al., 2018). Thus, model predictions of inundation extent were compared with the spatial patterns of inundation from the orthoimage using a *Kappa* ( $\kappa$ ) statistic (after Cohen, 1960) and a Reduced Major Axis (RMA) regression analysis. *Kappa* was defined by

$$\kappa = \frac{n \sum_{i=1}^2 n_{ii} - \sum_{i=1}^2 n_{i+} n_{+i}}{n^2 - \sum_{i=1}^2 n_{i+} n_{+i}} \quad [5]$$

where  $n$  is the total number of cells,  $i=1$  indicates wet,  $i=2$  indicates dry and  $+$  indicates summation along columns ( $+i$ ) or rows ( $i+$ ). This parameter allows us to objectively quantify the correspondence between inundation predictions and observations corrected for the effects of random agreement between them. The bathymetric correction also provided distributed information on water surface elevation and hence flow depth, allowing us to compare predicted and observed flow depths. RMA regression was chosen because of the possibility of error in both depth predictions and depth observations. Observed depths were in error because of the resolution of the model (0.50 m) as compared with grain size, assumptions made in the water surface interpolation (e.g. a locally horizontal water surface) used in the bathymetric correction and errors in identifying the water edge needed for the interpolation. We quantified both the slope of the regression ( $m$ ), as an indicator of either over- or under-prediction and also the scatter around the regression, using correlation ( $r$ ). All three parameters (*kappa*,  $m$  and  $r$ ) are optimised at values of 1. Table 1 summarises the main and best interpolation results. We adopted a baseflow discharge of  $0.55 \text{ m}^3\text{s}^{-1}$  and a Manning's  $n$  value of 0.06.

Once the model was optimized, discharges were simulated every  $0.05 \text{ m}^3\text{s}^{-1}$  from  $0.10 \text{ m}^3\text{s}^{-1}$  to  $1 \text{ m}^3\text{s}^{-1}$  and every  $0.25 \text{ m}^3\text{s}^{-1}$  to  $9 \text{ m}^3\text{s}^{-1}$ , leaving all other parameter values constant. This discharge range reflected the estimated discharge record for the study period. Each discharge was checked for mass conservation (loss less than 0.5%) and used to extract water depth and northing and easting velocity components per discharge.

## 2.5 Hydraulic determination of habitat availability

In order to quantify how available habitat by hydroelectric power related water management, a fuzzy habitat model was used (after Lane et al., 2006). The underlying philosophy of the modelling is the principle that stream hydraulics provides a critical linkage between human impacts on stream conditions and the availability of habitat which, if habitat is limiting, may translate into a biological response (Dunbar et al., 2011). In this paper, the focus is upon the microhabitat available at the reach scale and how this is impacted by hydroelectric power activities. Thus, it draws directly upon the notion of microhabitat habitat suitability models or indices (e.g. Bovee, 1986; Gore et al., 2001; Vadas & Orth, 2001; Lane et al., 2006; Dunbar et al., 2011), in which suitability is classed, based upon empirically determined organism preferences for certain combinations of hydraulic parameters. Such models have now proved to be a valuable tool for assessing environmental flows (e.g. Ahmadi-Nedushan et al., 2006, 2008; Garcia et al., 2011; Poff et al., 2017; Benjankar et al., 2018) especially when coupled with two-dimensional models (Benjankar et al., 2015), because they allow simulation of how the distribution of suitable habitat changes in space, as a function of discharge. This may be particularly important in Alpine streams with relatively complex bathymetry as suitable habitat may become available at higher (or lower) discharges in other zones of the stream-floodplain system.

405 In this paper we adopt an approach based upon fuzzy logic (e.g. Jorde et al., 2000;  
406 Adriaenssens et al., 2006; Van Broekhoven et al., 2006; Tonina & Jorde, 2013), which  
407 reflects the observation that the boundary between habitat classes will rarely be abrupt  
408 (Mouton et al., 2013) but gradual as a consequence of both producer and user uncertainty  
409 (Ahmadi-Nedushan et al., 2008); producer uncertainty in terms of classing hydraulic  
410 variables as habitat because the hydraulic variables may be in error; and user uncertainty  
411 because ecological boundaries (such as between classes) may be gradual and variable in  
412 time and space (Strayer et al., 2003), as well as compensatory and interacting, e.g. where  
413 less than optimal velocity is compensated for by more than optimal depth (Jorde et al.,  
414 2001). Although used less widely than for fish, fuzzy modelling has been used for estimated  
415 instream habitat availability for macroinvertebrates (e.g. Adriaenssens et al., 2006; Van  
416 Broekhoven et al., 2006, 2007; Mouton et al., 2009; Schneider et al., 2017; Theodoropoulos  
417 et al., 2018).

418 In our approach, we focus upon bed shear stress. Bed shear stress has proved to be valuable  
419 as an indicator of habitat availability because increases in shear stress can lead to density-  
420 independent loss of benthic macroinvertebrates (Lancaster and Hildrew, 1993) with a shear  
421 stress – loss relationship that depends on at least the family, sometimes the genus and even  
422 individual species and sub-species (e.g. Möbes-Hansen & Waringer, 1998; Méricoux &  
423 Dolédec, 2004; Dolédec et al., 2007; Méricoux et al., 2009). Relationships between  
424 invertebrate abundance and shear stress appears to be stronger than relationships between  
425 abundance and other hydraulic parameters (e.g. velocity, as used by Gore et al., 2001) and so  
426 we use shear stress in this paper.

427 Most studies of the preferences of macroinvertebrate families for particular shear stresses  
428 make no distinction between the effects of shear stress change on the ability of an organism  
429 to remain in a particular part of a stream, which we now label its habitability ( $Y$ ) and the  
430 effects of shear stress upon erosion and deposition processes. Erosion may lead to enforced  
431 drift for macroinvertebrates even if the hydraulic definitions defined by velocity and flow  
432 depth are acceptable. Scour may lead to loss of organic matter and hence food sources. At  
433 present, this is a subject of significant uncertainty. Studies have suggested net loss of  
434 primary production due to high discharge events in Alpine and polar ecosystems (e.g. Müller  
435 et al., 2009; Cullis et al., 2014). However, there appear to be few generalizable relationships  
436 (e.g. excess Shield stress values that might indicate organic matter loss) not least because:  
437 different communities appear to be more or less resistant to erosion (Biggs & Thomsen,  
438 1995; Müller et al., 2009); scour of organic matter depends on whether or not communities  
439 are actively growing (Biggs & Close, 1989); and local geomorphic conditions may  
440 determine sensitivity of organic matter scour to extreme flow events (Fuller et al., 2011). In  
441 addition, this process appears to be relatively understudied in Alpine environments (Gooseff  
442 et al., 2016).

443 For this reason, we take the basic assumption that organic matter loss is likely to correlate  
444 with the probability of bed movement, and hence we need to focus on the probability of  
445 scour. The question of burial is equally important. Evidence suggests that whilst the  
446 sensitivity of individual taxa to burial is variable (Wood et al., 2005), burial depth can lead  
447 to slower emergence, as can burial by finer sediment (Conroy et al., 2018). Burial may also  
448 reduce access at the stream bed surface to organic matter (Orr et al. 2008).

449 Given the above, we assume that both too much erosion (scour) or too much deposition  
450 (burial) could be detrimental to macroinvertebrates. It is also probable that some scour,  
451 notably of fines, may support macroinvertebrate life, especially given the very high silt and  
452 sand loads in this kind of stream. An organism that has a preference ( $Y$ ) for very low shear

stresses, for example, may be at risk of substantial sediment deposition at those shear stresses, such that what is optimal for the organism is shifted towards higher shear stresses. One that has a preference for high stresses may be at risk if those stresses also lead to scour (of sediment or potential food sources such as organic matter) in the particular stream setting under consideration. Thus, we introduce a second classification of shear stress which we call geomorphic ( $G$ ) and which reflects the exposure of sites to the risk of severe scour or deposition. Both  $Y$  and  $G$  are explained below, as is the way we combine them in a fuzzy habitat model.

## 2.6 Estimation of habitability preferences ( $Y$ )

Dolédec et al. (2007) compiled relationships between density and FliesswasserStammTisch (FST) hemisphere estimates of shear stress for small European streams, with data presented to family, genus and sometimes species level. We used data from Gabbud et al. (2019) to identify the dominant families present in the main stem of the study stream during the samples of May 2016 and May 2017. We chose May because at the altitudes of the streams used in this study, conditions should be suitable for macroinvertebrate life and flushing has not yet begun. Four families comprised 80% of the density of macroinvertebrates measured in the stream in 2016; Chironomidae (58%); Leuctridae (18%); Limnephilidae (0.4%); and Baetidae (4%); and 84% in 2017; Chironomidae (21%); Leuctridae (48%); Limnephilidae (8%); and Baetidae (7%). The difference between May 2016 and May 2017 can be attributed to a more clement winter in 2017 which meant that suitable conditions (i.e. temperature) for the Tricoptera Limnephilidae probably occurred slightly earlier in the year. We used Dolédec et al. (2007) to identify the normalised  $\ln(\text{abundance})$  of shear stress for each of these four families. Their normalisation was such that the  $\ln(\text{abundance})$  summed across all measured shear stresses was 1. Dolédec et al. (2007) do provide data to a finer taxonomic resolution but using finer data risked identifying habitat requirements that were too specific (i.e. locally adapted) to the sites used in their study. Each normalised  $\log(\text{abundance})$  was classified on the same scale into poor with  $\log$  abundance of 0.033 or less, medium as 0.033 to 0.067, and good as 0.067 to 1.0. The associated FST shear stress estimates were converted into shear stresses using data in Statzner & Müller (1989). It should be noted that there is an equivalence between the bottom shear stress measured using FST hemispheres and the bottom stresses modelled in a depth-averaged solution of the Navier-Stokes equations as used here: although such a model predicts the depth-averaged velocity, in the depth-averaged momentum equations, it is the bottom stress that is used and it is this that is predicted by the model. The bottom shear stress ( $\tau_b$ ) is calculated from the quadratic friction law used by the hydraulic model BASEMENT using

$$\tau_b = \rho g \frac{n^2}{d^{1/3}} v^2 \quad [6]$$

## 2.7 Geomorphic preferences ( $G$ )

The second element of the habitat availability we wished to quantify was a measure of the risk of burial/scour within the stream. In theory, this is a function of both local hydraulics and upstream sediment supply. We assume in this stream that the latter is not a limit given high rates of sediment transfer (Bakker et al., 2019). We quantify the critical Shields stress corrected for local bed slope:

$$\tau_{oc} = 0.06(\rho_s - \rho_w)gD_{50} \quad [7]$$

We assume that the bed has little armouring due to high sediment loads and frequent disturbance, which matches visual observations in the field. Erosion of the median grain sizes should occur when  $\tau_b > \tau_{oc}$ . We are using a single sediment size fraction. Thus as  $\tau_b$  rises towards  $\tau_{oc}$ , progressively coarser material should be transported. With continual increases of  $\tau_b$  above  $\tau_{oc}$  there is a growing likelihood of complete bed break up and scour to great depths. This provides a means of parameterising  $G$ .

In relation to burial, we assume that once  $\tau_{oc}$  falls below the threshold for transport of coarse sand, the risk of deposition increases and the location is likely to be poor. This broadly corresponds to what Conroy et al. (2018) observed as the threshold when time for body emergence from a deposit increase markedly, although this does depend on burial depth. For values of  $\tau_{oc}$  equivalent to the range coarse sand through to fine gravel, the shear stress is medium and for fine gravel through coarse gravel the shear stress is good. The coarse gravel threshold is important because coarser grain sizes are likely to act as anchoring refugia and if these sizes begin to move, then there is a growing probability of such refugia being lost. If cobbles (or greater) can move then we assume that the site is poor as the scour depths implied are greater than 0.064 m (OFEV, 2010), given the observed role of cobbles as refugia (Hose et al., 2007).

To aid interpretation, we calculate the percentage of the bed inundated at low flow ( $Q = 0.55 \text{ m}^3 \text{ s}^{-1}$ ) that becomes mobile through time according to [6] and [7]. With these relationships, 0.14% of the bed is mobile at baseflow, which also matches our observations of negligible sediment movement between flushes.

## 2.8 Fuzzy habitat model

Fuzzy subsets were defined for habitability ( $Y_i$ ) and geomorphology ( $G_i$ ) for each shear stress predicted by the hydraulic model simulations:

$$\begin{aligned} Y_p &= \left\{ \left[ \tau_o, \mu_{Y_p}(\tau_o) \right]; \tau_o \in Y, \mu_{Y_p}(\tau_o) \in [0,1] \right\} \\ Y_m &= \left\{ \left[ \tau_o, \mu_{Y_m}(\tau_o) \right]; \tau_o \in Y, \mu_{Y_m}(\tau_o) \in [0,1] \right\} \\ Y_g &= \left\{ \left[ \tau_o, \mu_{Y_g}(\tau_o) \right]; \tau_o \in Y, \mu_{Y_g}(\tau_o) \in [0,1] \right\} \\ G_p &= \left\{ \left[ \tau_o, \mu_{G_p}(\tau_o) \right]; \tau_o \in G, \mu_{G_p}(\tau_o) \in [0,1] \right\} \\ G_m &= \left\{ \left[ \tau_o, \mu_{G_m}(\tau_o) \right]; \tau_o \in G, \mu_{G_m}(\tau_o) \in [0,1] \right\} \\ G_g &= \left\{ \left[ \tau_o, \mu_{G_g}(\tau_o) \right]; \tau_o \in G, \mu_{G_g}(\tau_o) \in [0,1] \right\} \end{aligned}$$

[6]

where  $p$  = poor;  $m$  = medium and  $g$  = good;  $\mu_{L_i}(l)$  is the grade of membership of the predicted value  $l$  (i.e.  $\tau_o$ ) in  $L_i$  (i.e.  $Y_i$  or  $G_i$ ). If  $0 < \mu_{L_i}(l) < 1$ , membership of  $L_i$  is partial and the prediction (of  $\tau_o$ ) is a member of more than one  $Y_i$  or  $G_i$ . We then specify a fuzzy rule for Habitat ( $H_k$ ) where

$$\text{If } Y_i \otimes G_i \text{ then } H_k, \text{ for } K \text{ values of } k$$

[7]

where  $K$  is the number of habitat classes. We use a product operation rule (Wang, 1994) to define the degree of fulfilment of a particular  $H_k$ :

$$\mu_{Hk} = \mu_{Yi}(d) \mu_{Gi}(v) \quad [8]$$

As  $i=3$ , [7] gives 9 rules and hence  $K$ , the number of habitat classes, is 9. We reduce these 9 classes using a symmetrical habitat classification that weights  $Y_i$  and  $G_i$  equally (Table 2). This could be made more sophisticated by changing the weightings to reflect the relative importance of velocity and depth in contributing to a particular habitat class, notably through model calibration (e.g. Theodoropoulos et al., 2018). However, as this stream is largely void of life, and we do not know *a priori* which flow conditions (e.g. baseflows or flushing flows) limit the system, such calibration was not feasible. This is a weakness of our approach in this case. In a final stage, we defuzzify the results to provide a single crisp habitat suitability index (HSI) scaled between 0 and 1, with 1 corresponding to the situation when both  $Y$  and  $G$  are singularly classed within the categories  $Y_g$  and  $G_g$ .

The fuzzy model was applied for each of the four families identified above for each simulated discharge. This allowed visualisation of the spatial distribution of habitat. We produced two reach-scale habitat indices: (1) a measure of total habitat, the sum of all HSI values, which we label absolute habitat availability (AHA); and (2) a measure of habitat quality, equivalent to a weighted usable area (Lamouroux & Jowett, 2005), taken as the AHA divided by the area that was inundated, which we label weighted habitat availability (WHA) (Lane et al., 2006; Ahmadi-Nedushan et al., 2008). By looking spatially across all simulated HSI values for each simulated discharge, we were able to identify the maximum HSI simulated and the discharge responsible for that HSI, so allowing visualisation of how optimal habitat reorganised itself spatially as discharge changed. Finally, we used the discharge time-series for both the natural flow and the flow with abstraction and flushing to look up corresponding values of AHA and WHA, and so produce time-series of the two.

## 2.9 Validation of fuzzy habitat model predictions

In order to validate the model, we have compared fuzzy habitat model predictions with samples reported in Gabbud et al. (2019). This is not straightforward because the Borgne d'Arolla has negligible life during the flushing period (June through August) and so very little signal to explain. During the winter, it is snow covered. Logistical issues (time) also prevented macroinvertebrate sampling at the same moment as data collection for the hydraulic modelling. However, we attempted a partial validation by pooling the main channel sample results of Gabbud et al. (2019) (sites C1, D1 and D5) for three periods; March (characteristic of winter, when the river is snow-covered); May (characteristic of spring, when the river is snow free but before flushing starts) and June through August (characteristic of the main flushing period). We are able to attribute a characteristic discharge to each of these periods. For the May period, this is the baseflow given that water is being abstracted but because the abstraction is predominantly snow melt, there is no flushing. For June through August, we take the peak discharge characteristic of flushing as this is the flow that is likely to impact macroinvertebrate populations (when scour and burial are at a maximum). For March, we take an estimate of the flow as krenal sourced at about  $0.10 \text{ m}^3 \text{ s}^{-1}$ . For each of these flows, we calculate the mean and standard deviation of HSI values in the main channel zone characteristic of where the samples were measured. The results are encouraging (Figure 2). Only two points fall off a linear relationship between log abundance and mean habitat score. Including all data, the correlation is 0.510 (significant at  $p<0.05$ ); excluding the two outliers, the correlation is 0.941 (significant at  $p<0.05$ ). These two outliers are both for the family Baetidae for the March and May comparisons. Whilst there are only three data points (March, May and flushing) for each family, which is a limit of this validation approach, for Leuctridae, Limnephilidae and Chironomidae there is a

578 systematic increase in abundance with mean habitat score. The fact that we don't see this for  
579 Baetidae may imply that the hydraulic controls (i.e.  $Y$  and  $G$ ) are not sufficient to explain  
580 Baetidae populations in this kind of stream. Whilst this is only a partial validation, and has  
581 substantial limits, it confirms that the modelling approach is suitable for answering the  
582 questions being posed in this paper.

### 584 3 Results

585 Time series of the measured and modelled datasets are provided in Figure 3. Figure 3a  
586 shows how the regime with abstraction and flushing leads to relatively long duration periods  
587 of low flow, at or below  $0.55 \text{ m}^3\text{s}^{-1}$ , coupled with periodic high flows when flushes arrive.  
588 The baseflows are uniformly lower than would be expected under natural conditions (Figure  
589 3a). Despite attenuation, the flushing flow can be higher than expected under natural  
590 conditions because for some night-time flushes, an intake upstream of the Bertol Inférieur  
591 intake is opened and its water is added to that of the Bertol Inférieur water. Note that such  
592 data points are excluded from the calibration.

593 The temperature followed a clear diurnal cycle (Figure 3b) and was above  $6^\circ\text{C}$  and below  
594  $10^\circ\text{C}$  throughout the investigated period. The flushes typically lead to a drawdown in  
595 temperature but this was generally negligible. Aside from a short period in Day 1, the  
596 turbidity was extremely low between flushes (Figure 3c), around 50 NTUs. Flushes could  
597 lead to extremely rapid increases in turbidity, to as high as 2500 NTUs. From Day 5  
598 onwards, there is a general association between the size of the flush and the maximum  
599 turbidity reached. The latter may reflect the fact that the smaller flushes relate to the sand  
600 traps which, whilst producing significant sand supply, are of shorter duration and attenuate  
601 rapidly downstream (Figure 3a; see also Bakker et al., 2019), producing lower turbidity  
602 peaks.

603 Figure 3d shows how the euphotic depth evolves through time. First, flushes increase  
604 turbidity to reduce the euphotic depth to values typically less than 0.05 m. Second, the  
605 curves also show how sensitive the euphotic depth is to turbidity changes at low turbidity,  
606 such as between Days 5 and 6 after major flushing events during Day 4 and at the start of  
607 Day 5 (Figure 3c), when turbidity falls rapidly, but euphotic depth recovers more slowly.  
608 Third, between flushes, euphotic depth recovers over the timescale of hours to levels that are  
609 much greater than water depths typically found in this reach stream. This has the effect of  
610 making large areas of the stream bed shallower than the euphotic depth (Figure 3d) between  
611 purges, typically  $> 90\%$  for long periods of time. Given that the flow abstraction regime  
612 takes all kryal sources of water in this basin, what contributions remain are largely krenal  
613 (there are negligible nival sources in August) such that between flushes euphotic depths are  
614 much deeper than they might have been under natural conditions without flow abstraction.  
615 In terms of suspended sediment concentrations, whilst a purge is passing through, mean  
616 suspended sediment concentrations were very high with substantial variability ( $4,900 \pm 1,195$   
617  $\text{mg l}^{-1}$ ) (beyond the range of calibrated suspended sediment concentrations, hence the high  
618 uncertainty). During baseflow, mean concentrations were substantially lower and less  
619 variable ( $378 \pm 11 \text{ mg l}^{-1}$ ).

620 Figures 3e and 3f show two contrasting trends in terms of habitat availability. During  
621 flushing, total available habitat tends to increase rapidly (Figure 3e), to a degree that is  
622 variable between the four families. In rank order, conditions at all flows are best for  
623 Baetidae, followed by Chironomidae, Leuctridae and Limnephilidae. The weighted habitat

624 availability declines rapidly during flushing (Figure 3f), but this can reverse at the very  
 625 highest flushing flows (e.g. on days 8 and 9). This observation suggests the stream  
 626 morphology is conditioning habitat availability; initial rises in flow degrade available habitat  
 627 in existing inundated areas at a rate that is greater than the inundation of new areas creates it.  
 628 This is slowed and reversed at higher flows and we explore this below.

629 When compared with natural flows (Figures 3g and 3h), the abstraction regime reduces  
 630 absolute habitat availability across all families, but improves weighted habitat availability.  
 631 This is not surprising if the primary effect of higher flows (whether due to flushing or  
 632 natural stream flow due to glacier melt with no abstraction) is to increase absolute habitat  
 633 availability. However, comparing Figures 3g and 3h with Figures 3e and 3f reveals a second  
 634 and more critical issue. In a natural system, absolute and weighted habitat availability  
 635 change much more gradually than in a system subject to flushing meaning that very rapid  
 636 lateral displacement may be needed if an organism is to profit from improving habitat  
 637 elsewhere within the river-floodplain system. This spatial issue is further considered below.

638 Figure 3i shows percentage of the stream bed are likely to be active under both the  
 639 abstraction-flushing and natural regimes. Reflecting the discharge variation, this suggests  
 640 that abstraction leads to long periods of more stable bed conditions interspersed between  
 641 very rapid increases in bed instability during flushing events. Levels of bed activity in the  
 642 low flow inundated channel are much higher for the natural channel case, but as with  
 643 habitat, levels of bed instability rise and fall more slowly. Figures 3j and 3k show the  
 644 quantity and the percentage inundated area where the water depth is lower than the euphotic  
 645 depth and light can reach the stream bed.

646 The data shown in Figures 3e through 3h suggest that changing flow leads to spatial  
 647 reorganisation of the locations where suitable habitat is to be found. Figure 4 illustrates how  
 648 suitable habitat evolves in space for a range of discharges based upon those shown in Figure  
 649 3a, here for the case of Leuctridae. At a flow close to the summer baseflow ( $0.5 \text{ m}^3\text{s}^{-1}$ ) the  
 650 main channel is present as a meandering thread generally with HSI values between 0.1 and  
 651 0.15. However, the conditions in this main channel are marginally better at lower flows  
 652 (Figure 4,  $0.1 \text{ m}^3\text{s}^{-1}$ ). By  $1.0 \text{ m}^3\text{s}^{-1}$ , the white zones of no suitable habitat in the main channel  
 653 have widened, more so by  $1.5 \text{ m}^3\text{s}^{-1}$ . At this discharge there is the start of anabranch opening  
 654 and this continues to  $4.5 \text{ m}^3\text{s}^{-1}$ . At this discharge, the main channel evidence at  $0.5 \text{ m}^3\text{s}^{-1}$  is  
 655 entirely white, suggesting no suitable habitat is present. Thus, as discharge rises, zones of  
 656 more suitable habitat move from the main channel to anabranches and zones initially the  
 657 most suitable decline in suitability. This is reflected in the spatial displacement of where  
 658 optimum habitat suitability is to be found as a function of discharge (Figure 5). For all four  
 659 families (Figure 5), there is a range of discharges within the reach at which optimal habitat is  
 660 found, and reflecting the braided nature of the river-floodplain system (Figure 1), as  
 661 discharge rises, habitat suitability in some parts of the stream declines, but new zones  
 662 become inundated providing suitable habitat. Even at high flows there is potentially suitable  
 663 habitat available, but where it is to be found moves laterally.

664 Figure 6 provides plots of changes in absolute and weighted habitat availability as a function  
 665 of discharge for all four families. The absolute habitat availability rises for all families to  
 666 between  $0.75 \text{ m}^3\text{s}^{-1}$  (Limnephilidae) and  $0.90 \text{ m}^3\text{s}^{-1}$  (Baetidae) after which it declines very  
 667 slightly to between  $1.00 \text{ m}^3\text{s}^{-1}$  and  $1.25 \text{ m}^3\text{s}^{-1}$  for all families after which it rises to around  $5$   
 668  $\text{m}^3\text{s}^{-1}$  in all cases. This pattern reflects the stream bathymetry as  $0.80 \text{ m}^3\text{s}^{-1}$  is approximately  
 669 the channel capacity of the main channel and when this capacity is reached, there is  
 670 negligible increase in inundated width as discharge increases, water depth rises, roughness  
 671 falls, flow velocity rises, and so shear stress rises; and absolutes habitat availability falls.



This pattern is shown for Leuctridae in Figure 4 by comparing discharges  $0.5 \text{ m}^3\text{s}^{-1}$  and  $1.0 \text{ m}^3\text{s}^{-1}$ . By  $1.5 \text{ m}^3\text{s}^{-1}$ , the flow is sufficient to occupy anabranches and absolute habitat availability increases again.

The weighted habitat availability versus discharge curves are similar in shape for all families, increasing to a peak between  $0.15 \text{ m}^3\text{s}^{-1}$  (Chironimidae) and  $0.30 \text{ m}^3\text{s}^{-1}$  (Limnephilidae), then declining rapidly to  $1 \text{ m}^3\text{s}^{-1}$  and more slowly to  $2.25 \text{ m}^3\text{s}^{-1}$ . There is then a gentle rise to  $4 \text{ m}^3\text{s}^{-1}$  and after that a gentle decline. Again, these reflect the patterns in Figure 4 and the absolute habitat availability data shown in Figures 6a, 6c and 6e; increases in discharge translate into increases in absolute and weighted habitat availability as long as the discharge increase is accommodated by an increase in width, and the creation of new zones of lower shear stress.

Figure 7a provides a spatial visualisation of excess shear stress for the channel occupied at base flow with on the background the modelled inundation extent of the highest recorded discharge. This shows that almost all of the low flow stream bed has shear stresses that clearly exceed the critical shear stress, notwithstanding the uncertainty in the determination of the latter. Figure 7b lends credence to this analysis showing that over a slightly longer timescale than that considered here, the vast majority of the floodplain would be reworked under the current flow regime. It should be emphasised that Figure 7b shows a weakness in our approach: there may be evolution in the stream bathymetry during our study period, and hence in hydraulic conditions, and we do not take this into account during our analysis.

## 4 Discussion

The combination of field measurement with habitat modelling for macroinvertebrates allowed identification of how intake flushing may impact instream habitat variations in space and time as compared with what is likely under natural conditions.

In methodological terms, we adopted a mixed methods approach in which standard monitoring and hydraulic modelling methods were combined with a fuzzy habitat analysis. Before we discuss these results, we do wish to emphasise that despite a partial validation, the results of the habitat analysis should be treated with some uncertainty. On the one hand, the method did seem to reproduce a reasonable relationship between habitat availability and abundance for three of the four families studied here. On the other hand, the fact that Baetidae was over predicted in the winter and spring (no flushing) data is a reminder that the model only represents the effects of hydraulics and not other factors (e.g. stream temperature) on habitat availability. Fuzzy models have been found to be sensitive to how rules are set (Adriaenenssens et al., 2004) and are reliant upon expert knowledge of the problem at hand (Hernandez-Suarez & Nejadhashemi, 2018), as illustrated here with the use of the Shields criterion, observations of the lack of armouring, etc. In our application here, we did not include grain-size. There are also crucial issues of scale, especially in the kinds of gravel-bed rivers studied here where scales of just a few cm may need to be modelled in order to capture both local hydraulic variability and the availability of refugia. Finally, although the analyses are done here at the family scale, there will be within family variation in ecological traits that our application does not capture (in Baetidae, for example, Ussegli-Polatera & Tachet, 1994; Snook & Milner, 2002). As the Gabbud et al. (2019) data were to family level for the most part, this is the finest taxonomic resolution we could use. This is why the habitat model results in particular should be treated with caution.

717 That said, and taken together, the results allow us to evaluate the extent to which more  
718 diversified and abundant communities in winter and early spring, as compared with summer,  
719 can be attributed to what might be expected naturally in a morphologically dynamic Alpine  
720 stream as opposed to flushing of sediment from intakes. In so doing, we also draw some  
721 wider implications of these research results.

722 First, our evidence emphasises the need to consider minimum flows in terms of more than  
723 just hydrology (e.g. flow percentiles), the latter still being the case in some countries (e.g.  
724 the Swiss Water Law sets the minimum flow to be maintained at a flow abstraction site as  
725 the  $Q_{347}$ ). The weighted habitat availability (Figure 3f) tends to reduce for all families above  
726 discharges of between  $0.15 \text{ m}^3\text{s}^{-1}$  (Chironimidae) and  $0.30 \text{ m}^3\text{s}^{-1}$  (Limnephilidae). These  
727 discharges are lower than the summer baseflow delivered by krenal and unregulated sources  
728 for this reach of river ( $0.55 \text{ m}^3\text{s}^{-1}$ ) as all glacier-derived water is abstracted except during  
729 flushes. This stream is currently subject to a proposal to require a minimum non-abstracted  
730 flow of  $0.12 \text{ m}^3\text{s}^{-1}$  at one installation, which would increase the baseflow to between 0.65  
731 and  $0.75 \text{ m}^3\text{s}^{-1}$ . Assuming the results of the habitat modelling to be correct, this is exactly in  
732 the range of flows where absolute habitat availability is increasing marginally (Figures 6a,  
733 6c, 6e) but weighted habitat availability is declining rapidly (Figures 6b, 6d, 6f). It  
734 emphasises the need to evaluate minimum flow needs in terms of the ecology and not just  
735 the hydrology (e.g. a flow percentile) and also to take into account recharge from krenal and  
736 unregulated sources.

737 Second, the wider environmental data emphasise the need to look at not just discharge but  
738 also other environmental parameters as here the abstraction of water in itself may well  
739 reduce the harshness of the Alpine stream ecosystem. Temperature is commonly identified  
740 as a limiting parameter for certain macroinvertebrates in glacially-fed streams (Ward, 1994;  
741 Milner et al., 2001). Lower flows aid the recovery of temperature and, in this case, leads to  
742 relatively high stream temperatures. With the stream temperatures measured here (Figure  
743 3b) literature (e.g. Milner et al., 2001) suggests that the families studied are unlikely to be  
744 temperature limited. At the same time, as it is krenal water that is being abstracted in August,  
745 and as nival sources are largely exhausted at this time of year, krenal sources become  
746 dominant. The result is turbidity values typically between 20 and 30 NTUs substantially  
747 lower than those typical of natural glacier-fed streams in Switzerland (e.g. Malard et al.,  
748 2000; Robinson et al., 2001; Uehlinger et al., 2010). The same applies to mean suspended  
749 sediment concentrations ( $378 \pm 11 \text{ mg l}^{-1}$ ), about an order of magnitude lower than the peak  
750 suspended sediment concentrations reported for the unregulated streams of glaciers upstream  
751 of the study site during similar time periods (Clifford et al., 1995; Swift et al., 2005).

752 The reduction in turbidity due to the abstraction of suspended sediment rich glacier-derived  
753 water means that the euphotic depth of the residual discharges is increased (Figure 3d),  
754 except during flushes, and that the development of organic matter ought to be favoured. The  
755 greater relative importance of krenal sources might also increase nutrient supply relative to  
756 krenal dominated water, although its significance in terms of macroinvertebrate populations is  
757 less clear (Malard et al., 2003). The residual discharges also lead to relatively high levels of  
758 bed stability as compared with what might be expected under natural flow conditions  
759 (Figure 3i). Whilst the absolute habitat availability is lower (Figure 3e) as compared with  
760 natural flows, the weighted habitat availability (Figure 3g) is higher, as the residual  
761 discharges produce hydraulic conditions more suited to the families studied here. Thus, flow  
762 abstraction leads to less harsh and more stable conditions downstream of flow abstraction  
763 sites provided there are krenal and/or nival sources capable of providing a minimum flow.  
764 This is likely to reduce habitat suitability for macroinvertebrates better adapted to harsher

765 environmental conditions (Milner et al., 2001; Niedrist et al., 2018) but increase habitat  
766 suitability for generalists. A reduction in environmental harshness is likely to increase  
767 overall taxonomic richness (Jacobsen & Dangles, 2012) and if the target of stream  
768 management is measures of stream productivity, flow abstraction is beneficial.

769 Third, given this result, it is perhaps surprising that Gabbud et al. (2019) found that this  
770 stream is severely degraded in macroinvertebrate populations, even in the specialists that  
771 might be adapted to the flow variability and turbid water typical of kryal-dominated streams  
772 (Milner et al., 2001). Our analysis suggests rapid habitat deterioration during flushing events  
773 within the permanent channel (e.g. Figure 4), such that whilst these flushing events do  
774 increase absolute habitat availability (Figure 3e), this is through the inundation of new  
775 floodplain zones. The main channel itself becomes less suitable at higher discharges  
776 (Figures 4, 5) such that weighted habitat availability declines (Figure 3f).

777 In terms of other environmental parameters, there is only a slight drawdown in temperature  
778 directly after flushing. Temperature decline following hydropeaking has been reported (e.g.  
779 Bruno et al., 2012) but the decline here is lower than associated with hydropeaking (e.g.  
780 Toffolon et al., 2010). It is more the change in hydrograph shape (Figure 3a) and turbidity  
781 (Figure 3c) that is more striking. The marked increase in flow peakedness is accompanied by  
782 a very rapid increase in turbidity to values much more extreme than are normal for this kind  
783 of stream (e.g. Malard et al., 2000; Robinson et al., 2001; Uehlinger et al., 2010), and  
784 reflecting previous observations of this system (e.g. Gurnell, 1983; Petts & Bickertons,  
785 1994). These high turbidity values reflect the release of stored sediment during flushes and  
786 effectively counterbalance the reduction in turbidity during low flows by providing very  
787 high turbidity during flushes. Mean suspended sediment concentrations are higher than the  
788 maxima typical of glaciers upstream (Clifford et al., 1995; Swift et al., 2005) reflecting the  
789 combined effect of flushing by glacially derived melt with fine sediment stored in the  
790 sediment traps. Euphotic depths then reduce rapidly (Figure 3d). However, the duration of  
791 turbidity peak is short and whilst it leads to a marked reduction in the euphotic depth, there  
792 is rapid recovery to higher depths once the flush has passed (Figure 3d). The short duration  
793 is unlikely then to impact significantly upon the duration of time when organic matter  
794 growth is possible. However, the flushing leads to marked destabilisation of the stream bed  
795 (Figure 3i, Figure 7a) which is likely to remove organic matter accumulated between  
796 flushes. This destabilisation is likely to have a mixed scour and burial effect, as occurs in all  
797 braided streams during flushing events (e.g. see Bakker et al., 2019, for an illustration in this  
798 system), and it is this scour-burial process, as well as impacts on organic matter  
799 accumulation, that may be highly detrimental. Figure 8 illustrates for Baetidae the effects of  
800 including only habit preferences alone ( $Y$ ) as compared with scour-burial and habitat  
801 preferences ( $G$  and  $Y$ ) upon the optimum discharge for habitat at each point in the floodplain  
802 shown in Figure 5. Including  $G$  reduces the discharge that produces optimal habitat and this  
803 reduction is spatially variable, greater in the channel marginal zones where shear stresses  
804 increase most as discharge rises. Thus, geomorphic effects of erosion and deposition seem to  
805 render the stream more sensitive to discharge change than changes due to habitat preferences  
806 alone.

807 Fourth, the analyses suggested that both the abstraction-flushing regime and the natural flow  
808 regime tended to increase absolute habitat availability under higher flows (Figure 3e). This  
809 was a function of the braided nature of the stream, which resulted in the spatial  
810 reconfiguration of where habitat was most suitable as discharge changed (Figures 4, 5),  
811 observed elsewhere for Alpine braided rivers (Arscott et al., 2002; Sukhodolov et al., 2009;  
812 Vorste et al., 2016). The increase in absolute but decrease in weighted habitat availability  
813 occurs because the rate of creation of newly inundated zones is faster than the rate of

814 creation of more suitable habitat in these zones, whilst in the areas inundated at low flows,  
815 habitat availability declines rapidly as discharge rises. The low flow inundated zones are  
816 also the ones that are most likely to become unstable at high flows (Figure 7a). However,  
817 these patterns overlook a more notable consequence of flushing: the discharge (Figure 3a)  
818 passes from  $0.55 \text{ m}^3\text{s}^{-1}$  to  $5 \text{ m}^3\text{s}^{-1}$ , sometimes  $10 \text{ m}^3\text{s}^{-1}$ , very rapidly. Whilst there is  
819 progressive displacement in where habitat is optimum for all three orders as discharge rises  
820 (Figure 5) in practice there is a rapid switch between the low flow and the high flow state,  
821 much more rapid than would be the case under natural flow conditions (as also shown in  
822 Figures 3e through 3h). Macroinvertebrates can respond in two ways to such changes: burial  
823 in the gravel bed or drift (Gayraud et al., 2000). The hyporheic zone is recognised as an  
824 important refuge during disturbance (Williams & Hynes, 1974; Resh et al., 1988; Sedell et  
825 al., 1990; Stanford & Ward, 1993; Malard et al., 2001; Oldmeadow et al., 2010).  
826 Macroinvertebrate penetration into substrate is however only possible under two conditions.  
827 First, the bed sediment porosity must be sufficient (Maridet et al., 1996). We have no data to  
828 evaluate this. Second, the bed should remain relatively stable, with scour no greater than the  
829 depth of burial (Dole-Olivier et al., 1997), and not so much deposition that it becomes  
830 impossible to emerge after the flush has passed. The high level of bed instability in the low  
831 flow zones suggests there is likely to be substantial erosion and deposition during flushes  
832 (Figure 7).

833 Drift has been reported as a possible response to perturbation in glacial streams (Ilg et al.,  
834 2001; Saltveit et al., 2001), responsible for 82% of the stream bed macroinvertebrate  
835 colonisation in one case (Townsend & Hildrew, 1982) but still limiting the development of  
836 high levels of taxonomic richness (Jacobsen & Dangles, 2012). Baetidae, Leuctridae and  
837 Limnephilidae typically have their active drift maximum during the night (Brittain &  
838 Eikeland, 1988) while Chironimidae appear to be adapted to both daytime drift (Allan,  
839 1987) and nocturnal drift (Ali & Mulla, 1979; Stoneburner & Smock, 1979). Passive  
840 (catastrophic) drift results from sudden increases in discharge or substrate movement  
841 (Waters, 1961; Lancaster et al., 1996; Oldmeadow et al., 2010; Harby & Noack, 2013), such  
842 as might occur during a flushing event. Drift can occur when sediment is agitated even  
843 without transport (Gibbins et al., 2007). In the presence of disturbance, mean drift distances  
844 of 3-4 m have been observed in 10 to 15 minutes for Baetidae (Oldmeadow et al., 2010)  
845 while others suggest drift is only possible for shorter times, such as 5 to 10 minutes (Bruno  
846 et al., 2009). In the example studies here, drift would need to be both out of the low flow  
847 channel into anabranches and then back again to avoid stranding. This would need to happen  
848 often. The majority of flushes last 30 to 45 minutes (Gabbud et al., 2019), which is twice or  
849 even three times the duration that can be endured by macroinvertebrates, and the frequency  
850 of habitat condition disturbance can reach more than ten times a day in very extreme cases  
851 (Gabbud et al., 2019). Thus, whilst there may be suitable habitat available at flushing-  
852 induced high flows, it is not clear macroinvertebrates can migrate rapidly enough to get to  
853 such zones. There is evidence that at least some macroinvertebrates can survive the harsh  
854 conditions of glacial streams, and our study reach is well downstream of glaciers in zones  
855 where under natural flows the families investigated should be present (e.g. Milner et al.,  
856 2001). Thus, the fact that Gabbud et al. (2019) did not find macroinvertebrates in the low  
857 flow channel during August when flushes are of high frequency may be due to the  
858 combination of the bed instability caused by flushing flows coupled with the rapidity of flow  
859 change. It explains why macroinvertebrate recovery in this stream occurs as soon as the  
860 frequency of flushing events decreases (Gabbud et al., 2019).

861 To summarise the policy implications arising from this discussion, we argue that  
862 introduction of a minimum flow will probably not give the significant habitat improvement

needed. There are at least three reasons for this. First, given the morphology of the braided river here, increasing minimum flows causes negligible increase in absolute habitat availability and reductions in weighted habitat availability for each order considered (Figure 6). This is because for discharges up to  $1 \text{ m}^3\text{s}^{-1}$ , substantially greater than the minimum flows required by Swiss law (here the required  $Q_{347}$  is about  $0.12 \text{ m}^3\text{s}^{-1}$ ), the flow remains within its low flow bed, and the increases in discharge are not expressed in lateral flow expansion. The reason why a hydrological definition of low flow is insufficient in ecological terms is that stream bed morphology conditions the relationship between habitat availability and flow. Research is needed to better understand how braided river morphology conditions the response of macroinvertebrates to discharge change, ideally through the comparison of different Alpine braided rivers. Second, such minimum flows have to come from the sources that are being regulated, which are kryal. Water coming from kryal sources will be colder and turbid, the latter potentially leading to abrasion of accumulated organic matter (Church & Gilbert, 1975) and light penetration reduction (Robinson et al., 2001). Third, even with some optimisation of a minimum flow, the results emphasise that the real problem is the flushing of sediment. As long as flushing events continue to occur, habitats will be abruptly and frequently degraded and macroinvertebrates will not be able to recover. Flushing at present is needed for two reasons: (1) preventative, by emptying partly filled traps at night so that they don't need to be emptied during the following day, when there is a risk that tourists might be in the stream; and (2) because the sediment trap is full. Both of these issues might be addressed with upstream sediment management and, following Wohl et al. (2015) we argue that this is why this kind of stream needs sediment flows (or sed-flows) as well as e-flows.

## 5 Conclusion

The aim of this study was to explain why flow abstraction and associated intake flushing can lead to dramatic impacts on macroinvertebrate populations in high altitude Alpine streams. Our analyses suggest that whilst the water abstraction itself may reduce the harshness of such streams (in terms of temperature, turbidity, bed stability), flushing events lead to rapid degradation in the habitat conditions found in the channels inundated at low flow, in terms of rapid turbidity increase and increase in the probability of in-channel instability. Basic modelling of habitat preferences suggested that the kind of braided stream studied can have suitable habitat at all discharges (at least until those greater than the discharge which fully occupies the braidplain width). It is the rate of displacement of the sites of suitable habitat across the floodplain, driven by the speed of flushing flow rise and fall, that makes it unlikely that macroinvertebrates can profit from the theoretical resilience associated with the mosaic of braided river habitats. Burial and drift strategies are unlikely to be allow macroinvertebrates to profit from the rapid spatial reorganisation of suitable habitat. We argue that this explains why Gabbud et al. (2019) observed such a strong correlation between the absence of macroinvertebrates during intense flushing and the recovery of macroinvertebrate populations when flushing frequency fell even if the latter was in mid to late autumn.

In restoration terms, the work suggests that recommendations for increasing minimum flows in glaciated catchments need to be sensitive to the extent to which flow from unregulated parts of the basin and from groundwater, provide a minimum flow. If the regulated flows are kryal-sourced, using these to provide minimum flows may degrade low flow habitat through increasing turbidity and reducing stream temperature. Even then, a change in minimum flows should be evaluated with respect to where the current flow is in the relationship

912 between absolute/weighted habitat availability and discharge, as this is conditioned by the  
913 morphology of the stream bed as well as the size of the flow itself. This observation  
914 questions policies that set minimum flows in terms of flow distributions, without taking into  
915 account local geomorphological context. Even if it is concluded that a minimum flow is  
916 beneficial, it is unlikely that this kind of stream can be restored until there has also been  
917 some kind of sediment management, capable of reducing flushing rates. In addition to  
918 thinking of e-flows, it is also necessary to think of the kinds of management options that  
919 might better manage sediment delivery to intakes.

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## 1315 Captions

1316 **Figure 1:** The Arolla river basin in the Swiss Alps with the water intakes (left) and the focus study area at Satarma (right).

1317 **Figure 2:** Measured abundance versus mean habitat score with grey lines showing the standard deviation of the habitat  
 1318 score in the main channel zone; the data points are classed in two ways, in Figure 2a by time period and in Figure 2b by  
 1319 family.

1320 **Figure 2:** Time series of the variables measured and modelled for 9 days from the 6<sup>th</sup> of August 201: a) discharge under  
 1321 flushing (in red) and natural (in black) conditions [ $m^3s^{-1}$ ]; b) temperature [ $^{\circ}C$ ]; c) Turbidity [NTUs]; d) euphotic depth [m],  
 1322 the depth at which 95% of incoming light can penetrate to the stream bed; e) absolute habitat availability with a flushing  
 1323 regime for the four modelled families; f) weighted habitat availability with a flushing regime for the four modelled families;  
 1324 g) absolute habitat availability with a natural regime for the four modelled families; h) weighted habitat availability with a  
 1325 natural regime for the four modelled families; i) percentage of the bed that is inundated at base flow ( $0.55 m^3s^{-1}$ ) that is  
 1326 active given the discharge time-series in Figure 3a; j) the inundated area where the water depth is lower than the euphotic  
 1327 depth and light can reach the stream bed ; k) the percentage of the total inundated area where the water depth is lower than  
 1328 the euphotic depth and light can reach the stream bed.

1329 **Figure 3:** Spatial evolution of habitat suitability index according to discharge for the family Leuctridae using the fuzzy  
 1330 habitat modelling approach described in the text.

1331 **Figure 4:** The discharge ( $Q$ ) at each point in the floodplain that produces the optimum habitat suitability at that point.

1332 **Figure 5:** Absolute (left) and Weighted (right) Habitat Availability for: the four families studied here: (a) and (b), Baetidae;  
 1333 (c) and (d) Chironomidae; (e) and (f) Leuctridae; and (g) and (h) Limnephilidae.

1334 **Figure 7:** a) A map of modelled peak shear stress as multiples of the critical (Shields) shear stress for the highest discharge  
 1335 simulated, showing those zones inundated at low flow; b) DEM of difference between the 26<sup>th</sup> of July and the 12<sup>th</sup> of August  
 1336 2016 with in blue = positive change = deposition and in red = negative change = erosion, with LoD = Limit of Detection of  
 1337 change (= 0.13 m).

1338 **Figure 8:** Change in optimal discharge for Baetidae when the model is run with habitat preferences only (Y) as opposed to  
 1339 both habitat preferences and geomorphic suitability (Y and G, Figure 5a).

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1344 **Tables**1345 **Table 1:** Hydraulic model calibration with highest interpolation results;  $K$  = Kappa;  $m$  = Slope of the regression;  $r$  =  
1346 Correlation.

Discharge ( $\text{m}^3\text{s}^{-1}$ )	Manning's $n$	$\kappa$	$m$	$r$
0.5	0.04	0.837	2.64	0.731
0.5	0.05	0.845	1.989	0.729
0.5	0.06	0.868	1.253	0.726
0.55	0.04	0.881	1.122	0.713
0.55	0.05	0.879	1.068	0.722
<b>0.55</b>	<b>0.06</b>	<b>0.876</b>	<b>1.02</b>	<b>0.732</b>
0.55	0.07	0.87	0.968	0.736
0.55	0.08	0.868	1.125	0.754
0.6	0.04	0.878	1.254	0.762
0.6	0.05	0.876	1.026	0.727
0.6	0.06	0.753	0.218	0.701

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**Table 2:** The (symmetrical) definition of habitat classes in relation to the rule set defined in [7].

Symmetrical	<b>Y Poor</b> (presence rarely found)	<b>Y Medium</b>	<b>Y Good</b> (presence often found)
<b>G Poor</b> (risk of burial or scour)	Unsuitable Habitat 0	Very Poor Habitat 1	Poor Habitat 2
<b>G Medium</b>	Very Poor Habitat 1	Good Habitat 3	Very Good Habitat 4
<b>G Good</b> (fines washed out but no risk of catastrophic scour)	Poor Habitat 2	Very Good Habitat 4	Excellent Habitat 5

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**Figure1.**

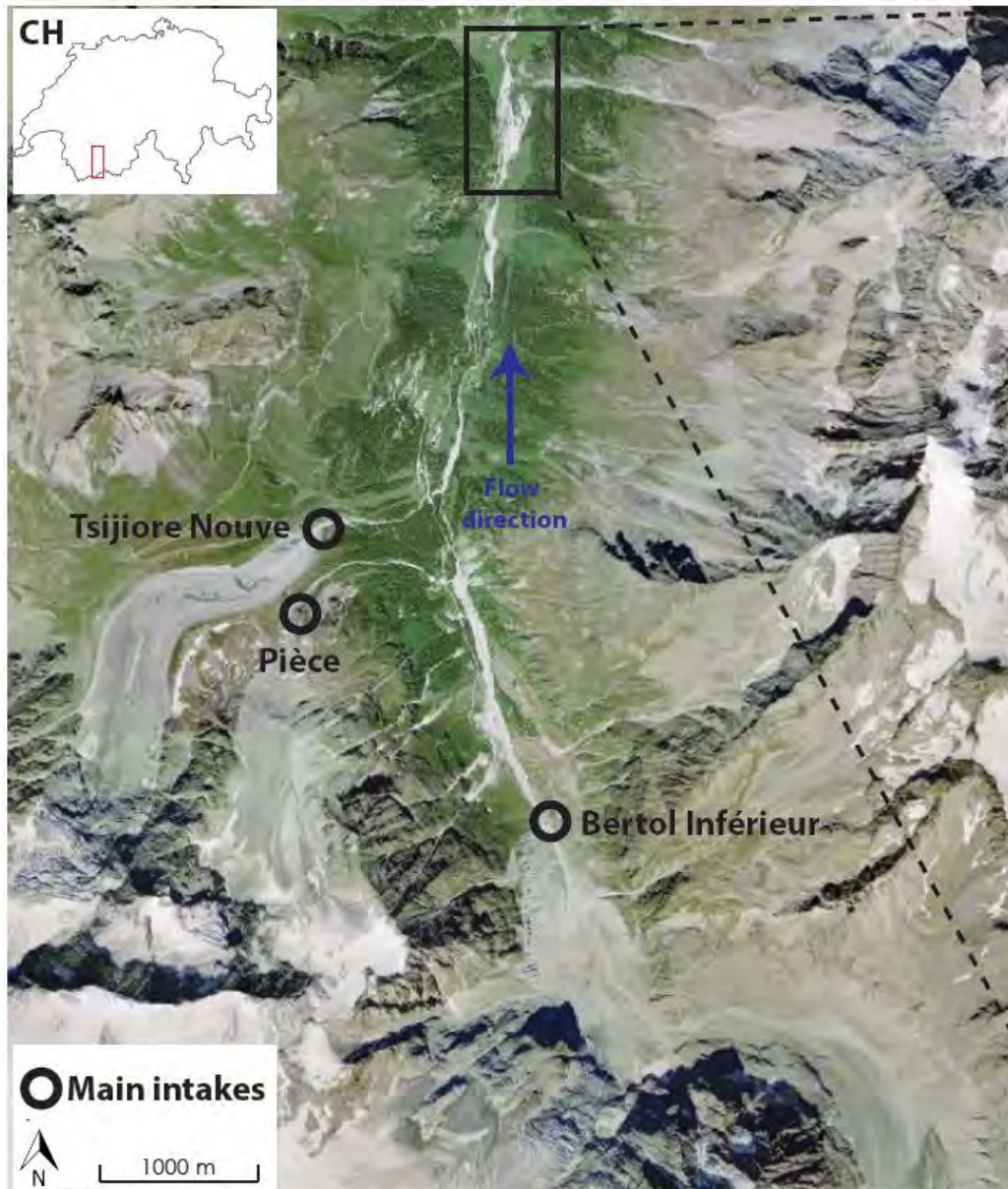


Figure2.

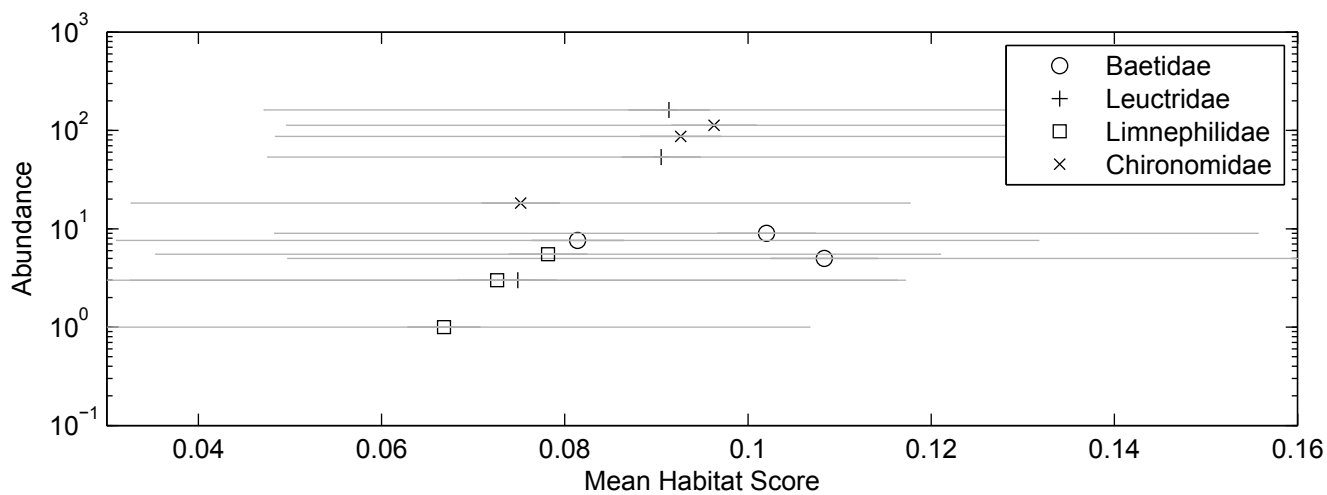
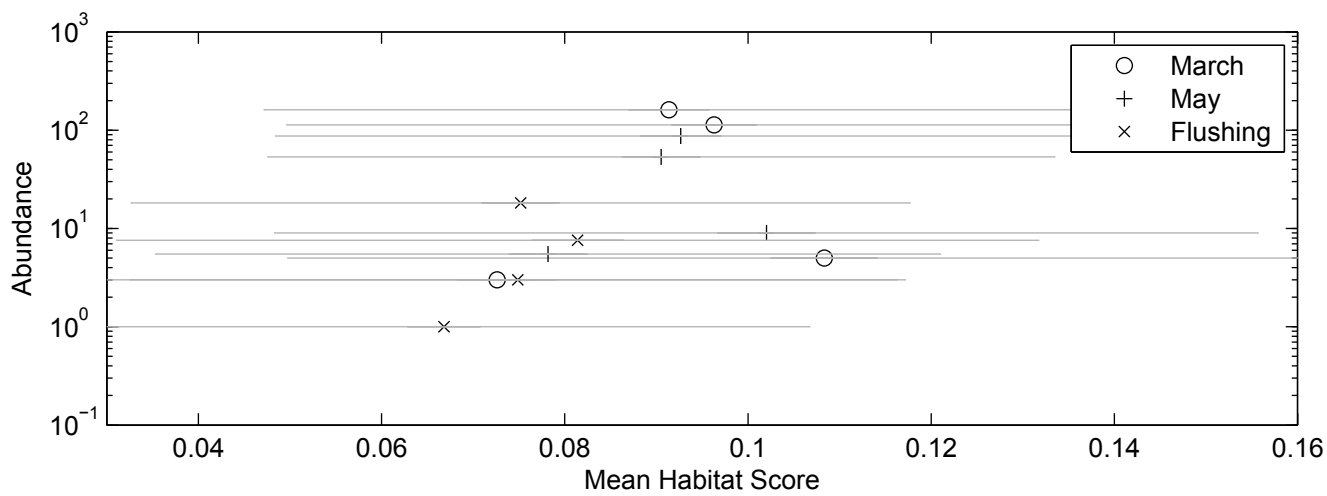


Figure 3ai.

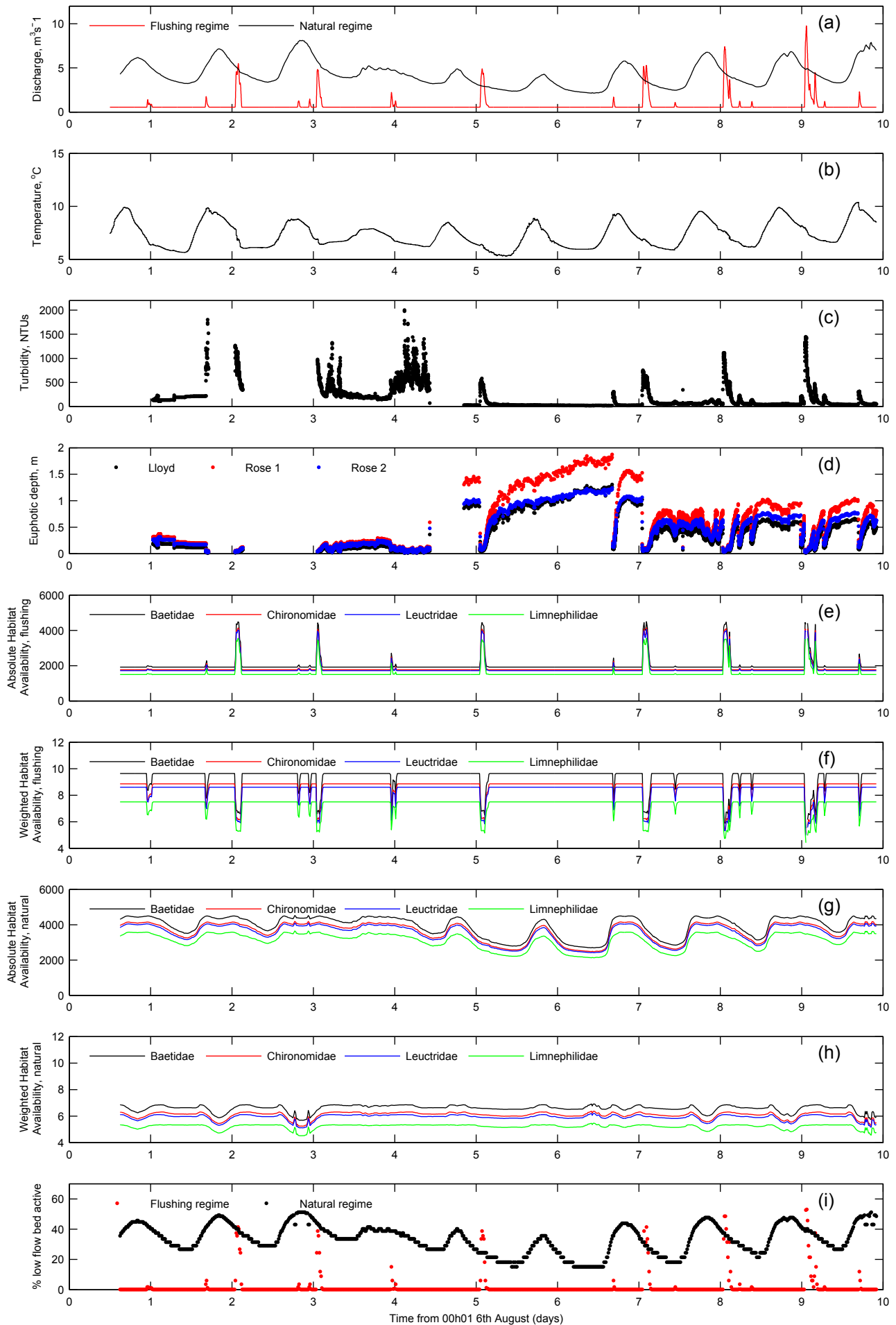




Figure3jk.

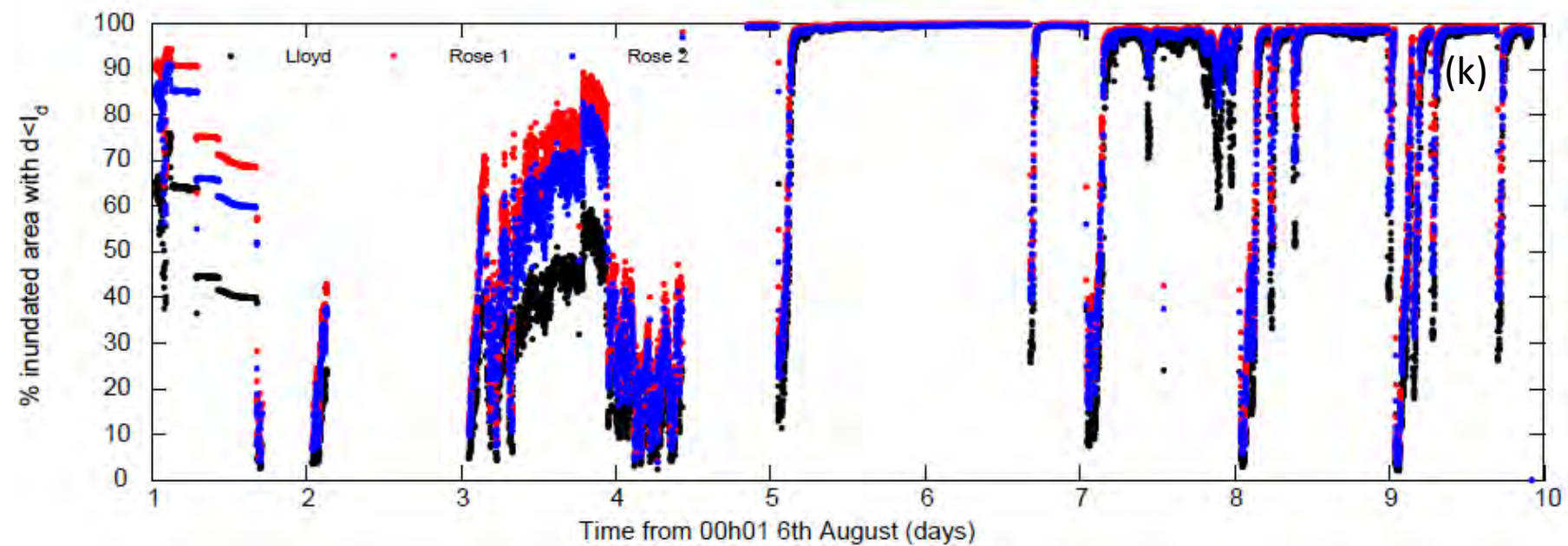
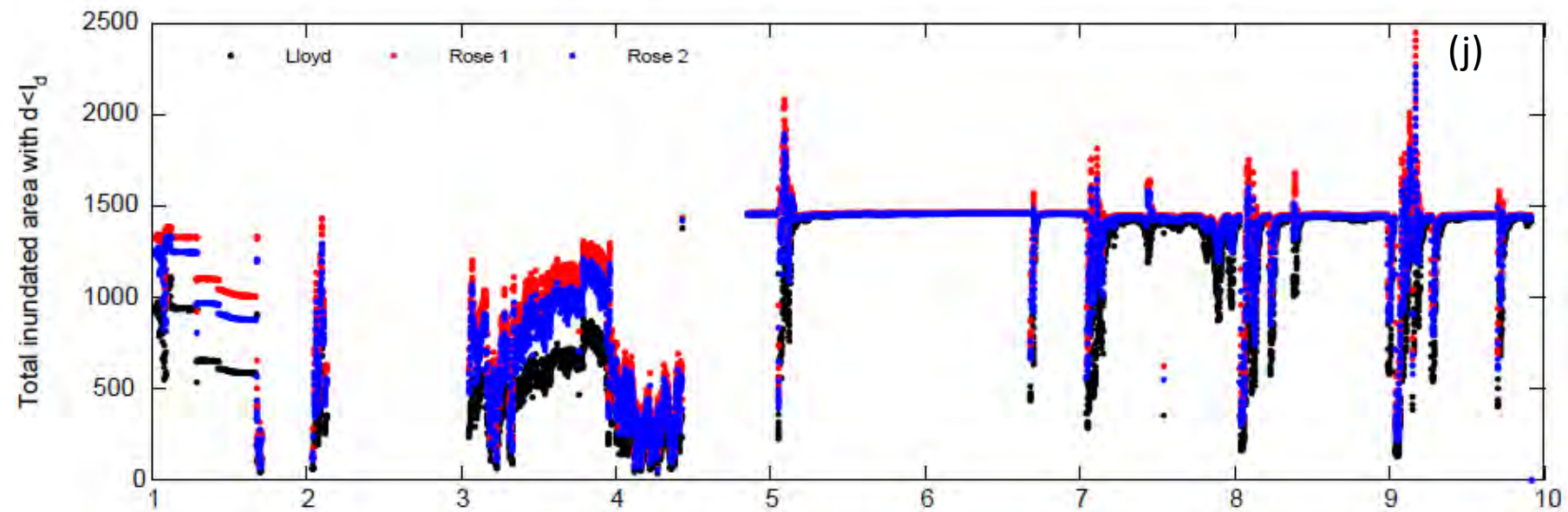


Figure4.

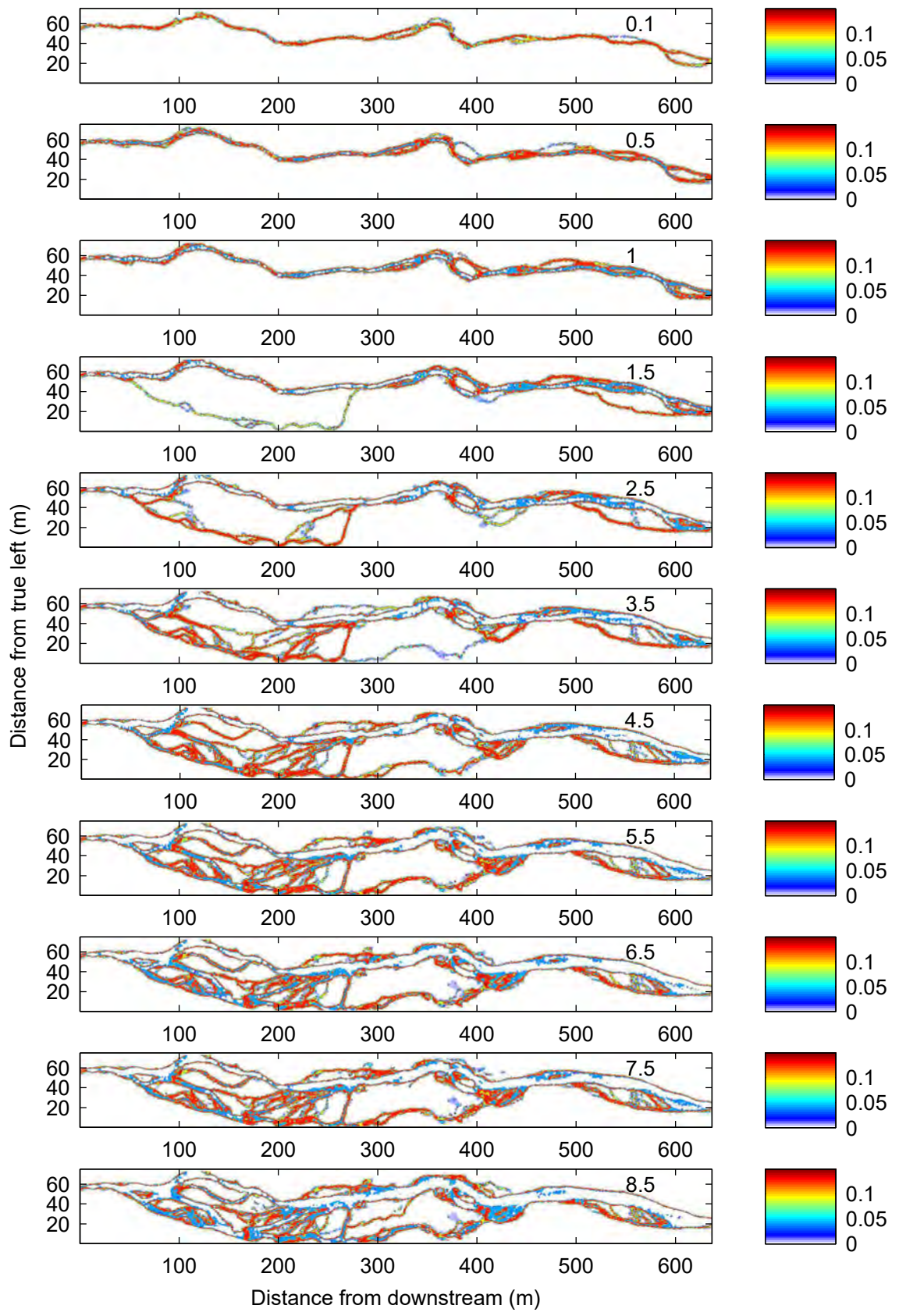


Figure 5.

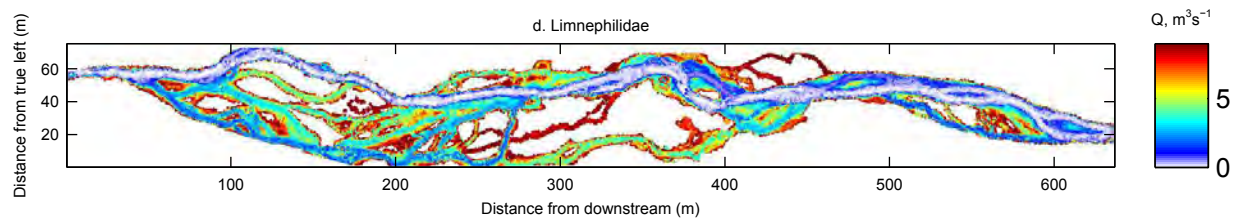
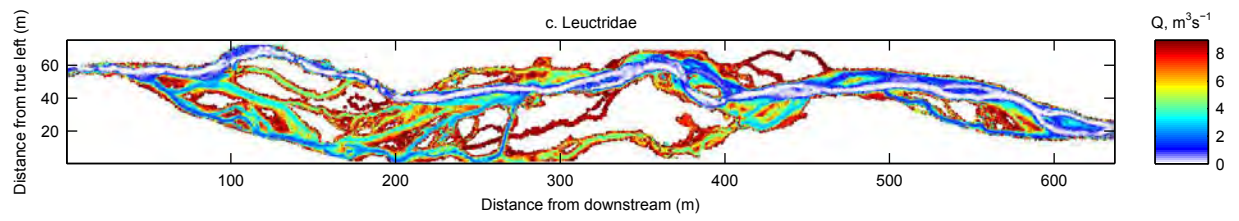
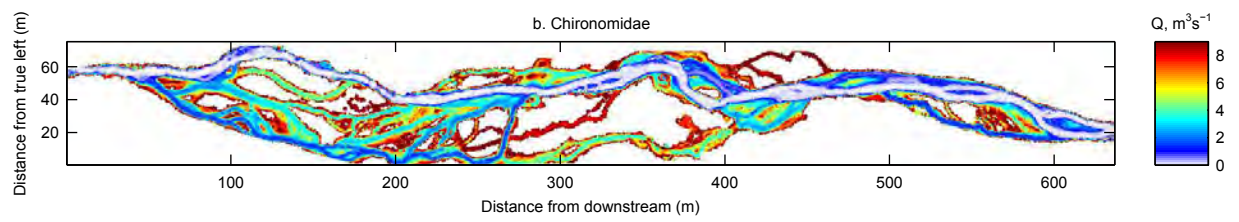
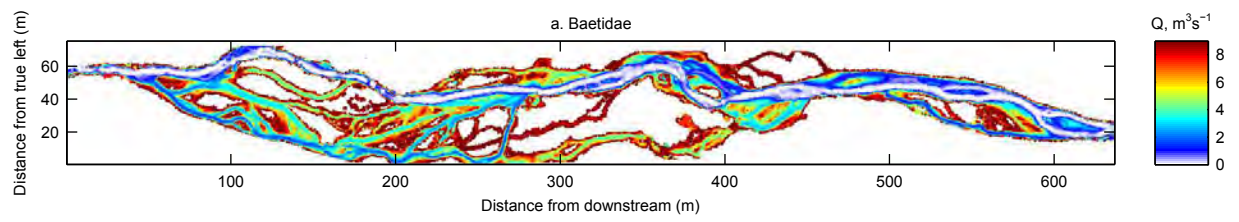
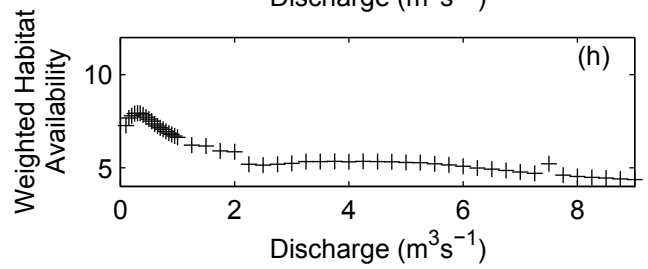
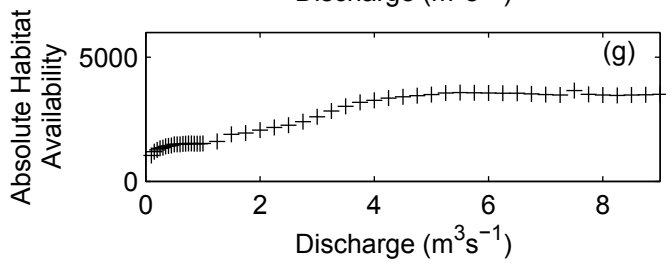
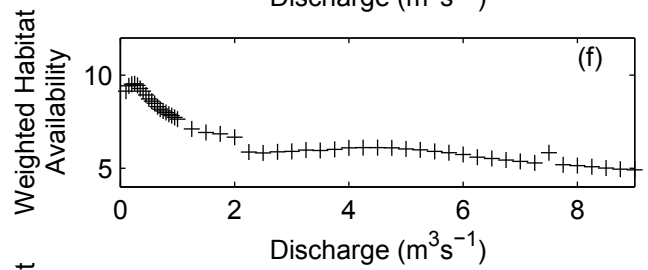
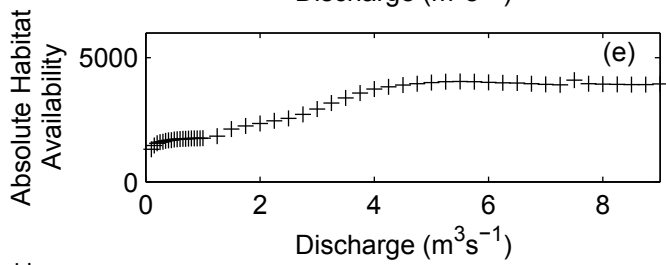
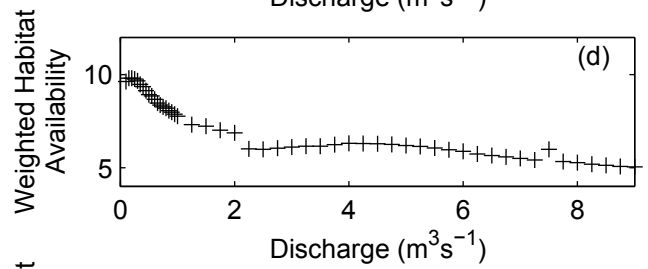
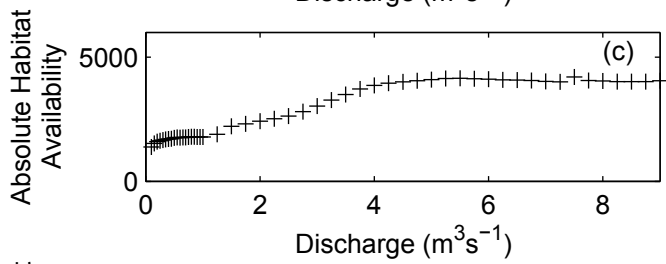
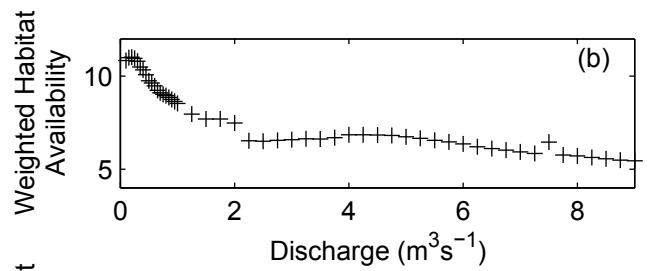
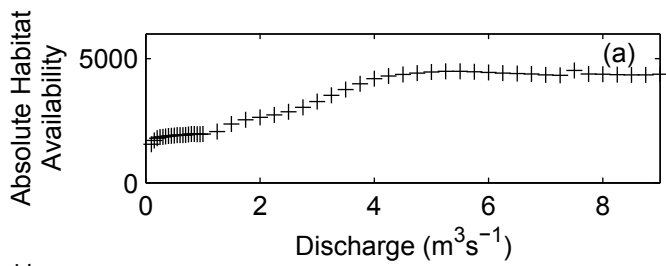


Figure6.





**Figure 7.**

Distance from true left (m)

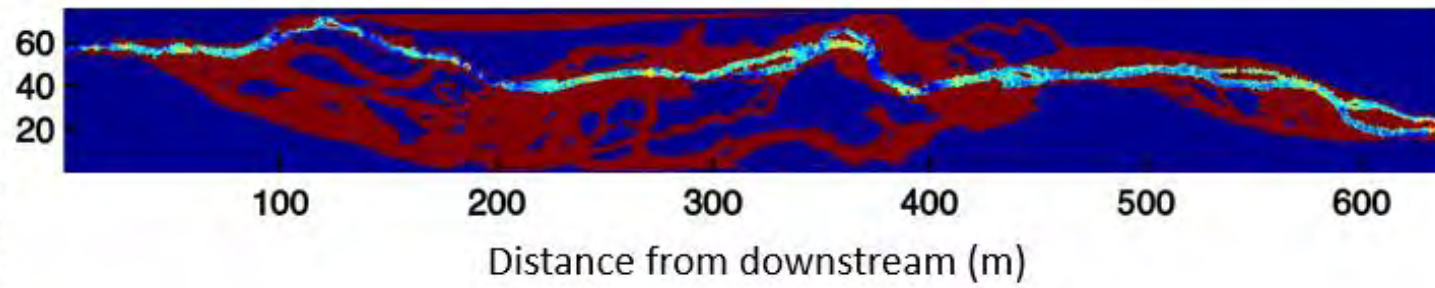


Figure 8.

Distance from true left (m)

