5.1 Abstract

Hydroelectric power plants managed in response to sub-daily changes of the electricity market undergo rapid variations of turbine discharge, entailing quickly fluctuating water levels downstream. This operation regime, called hydropeaking, causes numerous adverse impacts on river ecosystems. The hydrological alterations which affect hydropeaking rivers can be described by five parameters that change over space and time (magnitude, rate of change, frequency, duration, and timing), where each parameter may be correlated with distinct environmental impacts and therefore may be used to define flow thresholds and set targets for operational mitigation strategies. Thus, this study aims to present an extensive review on the so far established hydropeaking targets and thresholds regarding the outputs from the scientific community as well as from national regulations. We found that only few European countries (Switzerland and Austria) have legal regulations regarding hydropeaking flow thresholds. Other countries, such as Canada and the USA, present environmental legislation that can force hydropeaking mitigation measures. Most mitigation thresholds and management recommendations in literature deal with the effect of downramping on the stranding of salmonids, as well as with minimum flows between peak flows to avoid spawning ground desiccation. Regarding other fish species and parameters, information on mitigation targets or thresholds is scarcer or non-existent, as well as on hydropeaking mitigation case-studies, resulting in a lack of knowledge and guidelines for its implementation or regulation. Nevertheless, the available literature indicates that multiple aspects must be considered when assessing such values. Thus, to aid in that process, we propose that mitigation targets and thresholds must be based on key species, including particular features regarding season, life-stage and time of day, which must be combined with site-specific morphological characteristics. The presented approach may benefit impacted organism groups in hydropeaking reaches through the establishment of ecologically-based relevant mitigation thresholds and/or targets.
5.2 Introduction

Storage and pump-storage hydropower plants offer many advantages to present and future energy systems. Positive aspects include an excellent efficiency, the provision of stability to the energy grid by compensating fluctuations in power production caused by renewable energy sources (e.g., wind, solar), a rapid response to grid demand (flexibility), as well as the possibility to carry over electricity production from high flow to low flow seasons (Tonolla et al., 2017). Turbines are started up and shut down according to the demand of the electricity market, often on daily or sub-daily scales (Bejarano et al., 2017b). Especially this latter operation mode, called ‘hydropeaking’, leads to quick variations of river discharges which causes a rapid rise and fall of water levels downstream the tailrace (Jones, 2014; Moog, 1993). During non-peaking periods, water is stored in the reservoir, resulting in low river flows (base-flows or environmental flows). The unpredictability and intensity of flow variations are more permanent, frequent and severe than those caused by natural flow events, such as snow melt and intense precipitation (Greimel et al., 2016; Shuster et al., 2008).

Therefore, these anthropogenic induced rapid flow fluctuations may cause different ecological impacts, including periphyton biomass reduction (Bondar-Kunze et al., 2016), drift of macroinvertebrates (Schüting et al., 2016), and physical as well as physiological constraints for riparian vegetation (Bejarano et al., 2017a). Regarding fish biota, hydropeaking can reduce and alter spawning and rearing success (Becker et al., 1982; Casas-Mulet et al., 2014; McMichael et al., 2005), lead to downstream displacement and strandung (Auer et al., 2017; Boavida et al., 2017; Nagrodski et al., 2012), cause metabolic changes (Costa et al., 2018; Flodmark et al., 2002; Taylor et al., 2012) and influence fish growth (Kelly et al., 2017; Korman and Campana, 2009; Puffer et al., 2017). Furthermore, these flow and water level fluctuations can lead to variations in water quality and affect the riverbed morphology (Hauer et al., 2014).

In addition, the turbined discharges often use water from reservoir layers where the water temperature differs significantly from the one found in the river downstream. This phenomenon can thus create temperature peaks called ‘thermopeaking’, which may amplify the ecological impacts of hydropeaking concerning fish behavior (Zolezzi et al., 2011). Another hydropeaking-related impact is ‘saturopoeaking’, which can be described as an artificial, rapid, periodic and frequent fluctuation of gas saturation that follows the pattern of hydropeaking operations (Pulg et al., 2016). The median saturation of total dissolved gases in natural riverine environments is usually 99–101%. When it reaches values >110%, saturopoeaking will likely cause lethal effects on fish due to gas bubble disease, whereas at lower rates (103%–110%) fish may suffer indirect effects such as behavioral changes or increased susceptibility to infections (Pulg et al., 2016; Weitkamp, 2008). Furthermore, hydropeaking also leads to changes in aquatic soundscapes and sound pressure levels by temporal variations in the frequency composition (acoustic signature). This phenomenon has been named ‘soundpeaking’ and may affect fish physiology or behavior (Lumsdon et al., 2018).

Due to the extensive ecological consequences of hydropeaking on river ecosystems, it is stringent to develop suitable mitigation measures to reduce these adverse impacts. To reach this goal, a variety of measures have been proposed (e.g., Bruder et al., 2016; Moog, 1993; Person et al., 2014; Premstaller et al., 2017), which can be grouped into direct and indirect measures (Greimel et al., 2018a). Direct measures include operational as well as structural measures (e.g., the construction of retention basins...
or hydropoeaking diversion hydropower plants), whereby positive hydrological changes in the downstream river reaches are expected to occur (Premstaller et al., 2017). Indirect measures address river morphological aspects, aiming to compensate the negative impacts of hydropoeaking (e.g., through channel restructuring for habitat improvement).

A prerequisite for the establishment of efficient and cost-effective mitigation measures is the identification and establishment of mitigation targets and thresholds. Although hydropoeaking has been studied intensively in the last decades (Bejarano et al., 2017a), proposed thresholds for the different parameters, such as magnitude, rate of change, frequency, duration, and timing (cf. Harby and Noack, 2013), have not yet been consolidated, despite the fact that this has been pointed out to be a major further step for hydropoeaking research (Costa et al., 2017; Harby and Noack, 2013; Hauer et al., 2017; Young et al., 2011).

In this paper, we provide an overview on the current knowledge and present an extensive review on the so far established hydrological thresholds and targets for mitigating ecological impacts on fish. Based on the outputs from the scientific community as well as indicative values and targets from national regulations and guidelines, we intend to address the following questions: (1) Which are the proposed hydropoeaking mitigation thresholds in peer-reviewed literature? (a) Do these thresholds differ among distinct river reaches morphology? (b) Do these thresholds differ among species, their life-stage and time of day? (c) Are there any case studies regarding the successful implementation of operational measures? (2) Which are the established hydropoeaking mitigation thresholds and targets in national legislations, regulations and/or guidelines?

5.3 Methods

5.3.1 Literature search and analysis

We firstly obtained data on hydropoeaking mitigation thresholds by conducting a search for peer-reviewed literature. We used the Scopus database with the search string TITLE-ABS-KEY (“hydropoeaking” OR “hydropoeaking” OR “flow fluctuation” OR “pulsed flow” OR “peaking power” OR “flow ramping” OR “hydroelectric peaking” OR “hydro-electric peaking”) which was combined with TITLE-ABS-KEY (“threshold” OR “mitigat*” OR “ramping” OR “dewater*” OR “duration” OR “rate of change” OR “frequency”). We limited the search to the relevant subject areas, i.e., environmental science, agricultural and biological sciences as well as earth and planetary sciences. We did not set a lower date limit and included manuscripts published until September 2018. We initially found 237 peer-reviewed papers, for which we then screened the title, abstract and keywords to exclude articles that did not address the studied topic, reducing that number to 124 papers. Following, we removed papers that did not contain quantitative or qualitative recommendations on hydrological mitigation of peak-flow hydropower operation, reducing the number to 10 articles. We then added additional papers through snowball approaches and available grey literature was also integrated, leading to a final number of 22 publications.

5.3.2 Legislation and guidelines

We assumed that hydropoeaking is mostly present in countries which publish on this topic, and that the corresponding pressure extent in the country is related to the research conducted. We, therefore, identified relevant countries by conducting another Scopus literature search using the keywords “hydropoeaking” and “hydro peaking” in TITLE-ABS-KEY.
We retrieved 228 documents from 34 distinct countries, where 98 overlapped due to co-authorship, resulting in 326 single country documents (Figure 5.1). Based on this list, we assessed the status of national hydropeaking legislation or guidelines in the respective countries by contacting local experts or governmental authorities.

5.4 Results

5.4.1 Database

From the 22 papers which contained thresholds and targets for hydropeaking mitigation, the most commonly used parameters are downramping rate (vertical ramping velocity), baseflow and peak-flow magnitude, peak frequency and time between peaks (Table 5.1). The majority of the studies establishing quantitative thresholds assessed the impact of flow reduction on the stranding risk of early salmonid life-stages.

5.4.1.1 Downramping thresholds to mitigate stranding

From a fish ecological point of view, stranding caused by flow downramping can be considered the major pressure related to hydropower operation schemes (Nagrodski et al., 2012; Young et al., 2011). The effects of downramping can be quantified more easily than other ecological responses to hydropeaking through experiments in outdoor or indoor channels. Multiple studies reveal a clear reduction of stranding risk as downramping rates are lowered (Figures 5.2–5.4; Table 5.1). Figures 5.2–5.3 also show that as brown trout, *Salmo trutta*, and European grayling, *Thymallus thymallus*, grow from larvae into early juvenile life-stages, stranding risk is reduced, even if downramping velocity would remain the same, indicating that fish are less susceptible to stranding as they increase in size. Hence, Schmutz et al. (2013) conclude that lowering the downramping rate to <0.2 cm min$^{-1}$ and <0.4 cm min$^{-1}$ significantly reduces the stranding risk of grayling larvae and juvenile, respectively. Therefore, in stretches with hydropeaking, that
### Table 5.1 Mitigating adverse ecological impacts of hydropeaking through operational measures — literature recommendations and implemented case studies.

<table>
<thead>
<tr>
<th>Impact</th>
<th>Species, life-stage</th>
<th>Caused by</th>
<th>Description of operational mitigation measures and hydropeaking thresholds</th>
<th>Type of study</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stranding</td>
<td>Brown trout, <em>Salmo trutta</em>, fry and juvenile</td>
<td>Downramping</td>
<td>Decreasing downramping from 1 cm min⁻¹ to 0.3 cm min⁻¹ reduced the stranding of trout fry by &gt;50% in summer and fall, and almost eliminated stranding of 1+ trout. A further ramping rate reduction to &lt;0.16 cm min⁻¹ lead to even less stranding of trout fry.</td>
<td>Lab experiments</td>
<td>Halleraker et al. (2003)</td>
</tr>
<tr>
<td>Stranding</td>
<td>Brown trout, <em>S. trutta</em>, larval and juvenile (0+)</td>
<td>Downramping</td>
<td>A downramping threshold of &lt;0.1 cm min⁻¹ during the day and &lt;0.05 cm min⁻¹ during the night might reduce stranding of larvae, whereas for juvenile (65–70 mm) &lt;6.4 cm min⁻¹ and ≤3.2 cm min⁻¹ are recommended for day and night, respectively.</td>
<td>Outdoor flume experiments (Lunz, Austria)</td>
<td>Auer et al. (2014)</td>
</tr>
<tr>
<td>Stranding</td>
<td>Atlantic salmon, <em>Salmo salar</em>, juvenile (1+)</td>
<td>Downramping</td>
<td>Lowering the downramping rate from 0.9–1.0 cm min⁻¹ to 0.23–0.31 cm min⁻¹ (4–5 h dewatering time) almost eliminated stranding of wild juvenile salmon on natural substrate during spring daytime trials at low temperatures.</td>
<td>Field study (Nidelva River, Norway)</td>
<td>Saltveit et al. (2001)</td>
</tr>
<tr>
<td>Stranding</td>
<td>Atlantic salmon, <em>S. salar</em>, juvenile</td>
<td>Downramping</td>
<td>Avoiding ramping rates &gt;0.16–0.25 cm min⁻¹ can reduce stranding significantly. It is also advised to stabilize flow early in the growing season and restrict dewatering in darkness. Depending on discharge conditions (Q range), more stringent thresholds can be recommended to reduce juvenile stranding from late summer until spring.</td>
<td>Modelling (Surna River, Norway)</td>
<td>Halleraker et al. (2007)</td>
</tr>
<tr>
<td>Stranding</td>
<td>European grayling, <em>Thymallus thymallus</em>, larval and juvenile (0+)</td>
<td>Downramping</td>
<td>To reduce stranding losses in spring (May–July), maximum downramping rates per minute must be lower than 0.6 or 1 m³ s⁻¹ (equaling 7% or 11% of MQ).</td>
<td>Field study (Drava River, Austria)</td>
<td>Unfer et al. (2011)</td>
</tr>
<tr>
<td>Stranding</td>
<td>European grayling, <em>T. thymallus</em>, larval and juvenile (0+)</td>
<td>Downramping</td>
<td>Stranding risk of larvae is low if downramping rates are ≤0.2 cm min⁻¹ during the day, whereas for juvenile (∅ 35 mm and 53 mm TL) they can be ≤1.2 cm min⁻¹ and ≤3 cm min⁻¹, respectively.</td>
<td>Outdoor flume experiments (Lunz, Austria)</td>
<td>Auer et al. (2014)</td>
</tr>
<tr>
<td>Stranding</td>
<td>European grayling, <em>T. thymallus</em>, juvenile (0+)</td>
<td>Downramping</td>
<td>During the night, the daylight threshold of &lt;3 cm min⁻¹ is also recommended for larger juveniles (∅ &gt;53 mm TL) on homogeneous gravel bars, where the presence of depressions on heterogeneous gravel bars demands more stringent thresholds of ≤0.5 cm min⁻¹.</td>
<td>Outdoor flume experiments (Lunz, Austria)</td>
<td>Auer et al., (2014, 2017)</td>
</tr>
<tr>
<td>Stranding</td>
<td>European grayling, <em>T. thymallus</em>, larval and juvenile (0+)</td>
<td>Downramping</td>
<td>Lowering the downramping rate to &lt;0.2 and &lt;0.4 cm min⁻¹ significantly reduces the stranding risk of grayling larvae and juvenile, respectively.</td>
<td>Outdoor flume experiments (Lunz, Austria)</td>
<td>Schmutz et al. (2013)</td>
</tr>
<tr>
<td>Stranding</td>
<td>Coho salmon, <em>Oncorhynchus kisutch</em>, rainbow trout, <em>O. mykiss</em>, juvenile</td>
<td>Downramping</td>
<td>In winter (water temp. &lt;4 °C), fish losses due to stranding can be reduced if downramping is conducted during the night, as fish are active and do not hide in the substrate (diel shift). A slower downramping rate will furthermore reduce stranding.</td>
<td>Lab experiments</td>
<td>Bradford et al. (1995)</td>
</tr>
<tr>
<td>Stranding</td>
<td>Pacific salmon and steelhead rainbow trout, <em>Oncorhynchus</em> sp., larval and juvenile (0+)</td>
<td>Downramping</td>
<td>A summer, spring and winter downramping threshold of 0.05 cm min⁻¹ and 0.25 cm min⁻¹ is necessary to protect salmon and steelhead fry.</td>
<td>Field study (Sultan River, USA)</td>
<td>Olson (1990), in: Schmutz et al. (2015)</td>
</tr>
</tbody>
</table>

(continued on next page)
### Table 5.1 (continued)

<table>
<thead>
<tr>
<th>Impact</th>
<th>Species, life-stage</th>
<th>Caused by</th>
<th>Description of operational mitigation measures and hydropeaking thresholds</th>
<th>Type of study</th>
<th>Source</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pacific salmon and steelhead rainbow trout, <em>Oncorhynchus</em> sp.</td>
<td>Downramping</td>
<td>Interim ramping rate criteria, differentiated by three seasons and time of day for each season, are proposed: (1) mid-February–mid-June: no ramping during daylight, and 0.08 cm min⁻¹ during the night; (2) mid-June–October: 0.04 cm min⁻¹ (day and night); (3) November–mid-February: 0.08 cm min⁻¹ (day and night).</td>
<td></td>
<td>Hunter (1992)</td>
<td></td>
</tr>
<tr>
<td>Pink salmon, <em>Oncorhynchus gorbuscha</em>, Chum salmon, <em>O. keta</em>, and Chinook salmon, <em>O. tshawytscha</em>, juvenile (0+)</td>
<td>Downramping</td>
<td>Flow management measures at the Skagit Hydroelectric Project propose that fry stranding can be mitigated by releasing a sufficient minimum flow, by lowering the yearly number of downramping events and by reducing downramping amplitude to 113 m³ s⁻¹ (here: differences between the highest and lowest flow release during any 24 h period due to flow reduction). Also, downramping during the daytime is not allowed during the emergence and outmigration period, when fry are most vulnerable to stranding. The project set a general threshold for downramping flow rate of 85 m³ s⁻¹.</td>
<td>Field study (Skagit River, USA)</td>
<td>Connor and Pflug (2004)</td>
<td></td>
</tr>
<tr>
<td>Chinook salmon, <em>O. tshawytscha</em>, juvenile (0+)</td>
<td>Peak magnitude</td>
<td>During the swim-up phase (mid-March to mid-June), flows should not exceed 240 m³ s⁻¹ to avoid pool trapping, as fish are not able to reach higher bank areas with many depressions that will fall dry during downramping.</td>
<td>Field study (Dordogne River, France)</td>
<td>Cazeneuve et al. (2009)</td>
<td></td>
</tr>
<tr>
<td>Brown trout, <em>S. trutta</em>, Atlantic salmon, <em>S. salar</em>, juvenile</td>
<td>Upramping</td>
<td>Lowering the upramping rate from &gt;3 to 0.5 cm min⁻¹ can mitigate the risk of drifting for juveniles (&gt; 53 mm TL), especially during night experiments.</td>
<td>Outdoor flume experiments (Lunz, Austria)</td>
<td>Auer et al. (2017)</td>
<td></td>
</tr>
<tr>
<td>European grayling, <em>T. thymallus</em>, juvenile (0+)</td>
<td>Upramping</td>
<td>It is recommended to discourage fish from spawning in higher elevation areas of the river channel by reducing peak flows to prevent later redd desiccation or provide minimum flows during critical development periods.</td>
<td>Field survey (Columbia River, USA)</td>
<td>McMichael et al. (2005)</td>
<td></td>
</tr>
<tr>
<td>Pink, Chum, and Chinook salmon, <em>O. gorbuscha</em>, <em>O. keta</em>, and <em>O. tshawytscha</em>, egg and embryo</td>
<td>Peak magnitude and base-flow conditions</td>
<td>To protect eggs and embryos from redd dewatering, the Skagit Hydroelectric Project imposed constraints on maximum flows during spawning as well as prescribed higher minimum flows during incubation (70–140 m³ s⁻¹).</td>
<td>Field study (Skagit River, USA)</td>
<td>Connor and Pflug (2004)</td>
<td></td>
</tr>
<tr>
<td>Chinook salmon, <em>O. tshawytscha</em>, egg and alevin</td>
<td>Peak magnitude and base-flow conditions</td>
<td>Redd dewatering can be minimized by providing minimum incubation discharges. The effect is even greater, if these discharge magnitudes are similar to spawning discharges.</td>
<td>Field study and modelling (Columbia River, USA)</td>
<td>Hamish et al. (2014)</td>
<td></td>
</tr>
</tbody>
</table>

(continued on next page)
are suitable for fish spawning and recruitment (potential spawning grounds, habitat availability), a temporal “larval window” is suggested where such stringent thresholds shall be enforced (Greimel et al., 2017). Similarly, other authors proposed different seasonal thresholds to include length-specific distinctions regarding stranding risk (e.g., Auer et al., 2014; Hunter, 1992; Olson, 1990 in: Schmutz et al., 2015).

Aside from these recommendations related to fish length, we detected species-specific differences. Brown trout, for example, is more sensitive to downramping than grayling (FIGURES 5.2–5.3). While the critical rate for grayling larvae is 0.2 cm min\(^{-1}\) (Schmutz et al., 2013), stranding of brown trout larvae occurs already at rates >0.1 cm min\(^{-1}\) (Auer et al., 2014; Halleraker et al., 2003). For comparison, stranding of juvenile Atlantic salmon, *Salmo salar*, was almost eliminated at rates of 0.23–0.31 cm min\(^{-1}\) (Saltveit et al., 2001).

In summer, flow reduction during daytime can reduce stranding rates for European grayling and brown trout in comparison to nighttime downramping (Auer et al., 2014, Auer et al., 2017), whereas in winter the opposite could be detected for brown trout, Atlantic salmon, Coho salmon, *Oncorhynchus kisutch*, and rainbow trout, *O. mykiss*, (Bradford et al., 1995; Saltveit et al., 2001). A further parameter that determines stranding risk is riverbank morphology (Auer et al., 2017; Hauer et al., 2014), as well as the presence of structures and deep areas (Bradford et al., 1995). FIGURE 5.4 shows that the addition of cover and pools in laboratory flume experiments can both increase or decrease
stranding rates, depending on species assessed. Furthermore, as described above, Figure 5.4 depicts the increased stranding risk during daylight compared to the night in the winter.

Overall, Schmutz et al. (2015) related downramping rates to a multimetric fish index and showed that ramping velocity >0.5 cm min\(^{-1}\) is associated with a poor or bad fish ecological status, whereas a reduction to <0.25 cm min\(^{-1}\) increases the probability of attaining a higher ecological status in nature-like rivers. These recommendations agree with those from Halleraker et al. (2007), who state that stranding can be significantly reduced if ramping rates >0.17–0.25 cm min\(^{-1}\) are avoided.

5.4.1.2 Base- and peak-flow magnitude, peak frequency and time between peaks

Next to downramping velocity, base- and peak-flow magnitude, peak frequency and time between peaks are the most commonly reported parameters regarding hydropeaking, which mostly aim at mitigating the ecological effects related to spawning and intra-gravel life-stages (Table 5.1). In this category, however, the majority of papers only suggest qualitative targets. Common impacts include the dewatering of spawning grounds which can lead to mortality of eggs and larvae, whereas the sensitivity of these life-cycle stages can vary: In general, pre-hatch stages are more tolerant to desiccation than post-hatch stages (Becker and Neitzel, 1985), while pre-emergence alevins are especially sensitive and can die if the redd is dewatered for already less than one hour (Becker et al., 1982).

To protect eggs and larvae from redd dewatering during drawdown to base-flow between peaks, it is recommended to discourage fish from spawning during regular peak-flows, as they will spawn in higher elevation areas which can easily fall dry during base-flow. This can be achieved by, for example constraining maximum flows during spawning (Connor and Pflug, 2004). Furthermore, a sufficient base-flow should be provided during critical development periods to always cover spawning redds with water (Casas-Mulet et al., 2014; Connor and Pflug, 2004; Harnish et al., 2014; McMichael et al., 2005).
5.4.2 National legislation and guidelines

5.4.2.1 Europe

Based on a literature query, 34 countries that publish on hydropeaking were identified (FIGURE 5.1), where the top three were Norway, Switzerland, and Canada. Nineteen of these 34 countries belong to the European Union and are, therefore, obliged to comply with the goals of the EU Water Framework Directive (2000/60/EC; hereinafter ‘WFD’). However, the WFD does not specify methods, targets or thresholds for hydropeaking mitigation, but only refers to the achievement of the good ecological status or good ecological potential in all water bodies by 2027 (cf. Art. 4 WFD). Similar to the environmental flows (EC, 2015), the regulation and implementation of mitigation measures behooves the individual countries. While the assessment of the ecological status follows a standardized approach, the definition of good ecological potential depends on potential effects on use (cf. Art. 4, 3(a) WFD; EC, 2003). Hence, the definition of restoration targets for achieving the good ecological status may follow a more or less standardized approach, while mitigation targets for achieving the good ecological potential may vary depending on potential effects on use (cf. Art. 4 WFD).

A European survey (Halleraker et al., 2016) asked 30 European countries if mitigation of rapidly changing flows (incl. effects of hydropeaking) was included in the national list of mitigation measures. Twelve countries answered “yes”, of which we were able to get legal restrictions on hydropeaking for 8 of them (TABLE 5.2). Nine of them said the topic is not relevant, seven did not give a statement and two identified the impact but did not present any measure.

Austria is the only EU Member State that has already established hydropeaking thresholds. On a Federal level, the Autonomous Province of Bolzano, Italy, did it as well. Other countries or regions, like Spain and the German Province of Baden-Württemberg, have recommendations for the mitigation of hydropeaking considered in river basin management plans, while others still work on a case by case basis (e.g., Norway) (TABLE 5.2).

Austria set a base-flow to peak-flow threshold ratio of 1:3 and also demands a maximum

Figure 5.3 Stranding rates of different life-stages of European grayling, Thymallus thymallus, in relation to downramping velocity during spring and summer daytime experiments on homogeneous gravel bars. The large icons and the dark-colored trendline represent median values, whereas the small icons and the light grey trendline represent the 25th and 75th percentiles reported in the studies. Envelope curves are logarithmic. Data sources: Auer et al., 2014, Auer et al., 2017, Schmutz et al. (2013), Zeiringer et al. (2014).
change of 20% in wetted area for small and medium-sized rivers. In these cases, a ratio of >1:5 automatically leads to the failing of a good ecological status (QZVÖ, 2010). In large rivers, a case by case evaluation is required, as they are more sensitive to this pressure. There, a threshold ratio of 1:3 may already lead to the failing of the good ecological status (QZVÖ, 2010). Already existing hydropoeaking reaches are classified as heavily modified water body and, therefore, may not adhere to the above thresholds. Instead they must attain the good ecological potential. Recent R&D projects followed a case specific approach considering additional parameters such as ramping rates, peak frequency, timing, or river morphology. Finally, the ecological potential is defined within an integrative approach including ecological and economic analyses and scenario evaluation to avoid adverse effects on the use sensu WFD (Greimel et al., 2017). On a regional level, the government approved the water management framework plan for Western Tyrol to reach the targets of the WFD, as well as to increase the energy production along the Upper Inn River valley (Reindl et al., 2017). Through the construction of hydropoeaking diversion power plants and compensation basins, hydropoeaking thresholds of <15 and <12 cm h\(^{-1}\) for up- and downramping shall be attained in all affected river reaches. However, when determining thresholds, critical life-stages of fish shall receive special attention (Wasserwirtschaftlicher Rahmenplan Tiroler Oberland, 2014).

Similar to Austria, the Autonomous Province of Bolzano, Italy, set a threshold ratio between base-flow and peak-flow of 1:3 for new facilities.
### Table 5.2 Status of hydropeaking legislation thresholds and target values within the studied countries (only countries with information are displayed).

<table>
<thead>
<tr>
<th>Country</th>
<th>Legislation/guideline</th>
<th>Thresholds and target values (description)</th>
<th>Evaluation</th>
</tr>
</thead>
</table>
| Austria                  | Qualitätszielverordnung (QZVO), 2010; BMLFUW (2015)                                   | <1:3 and <20% change in wetted area (for small and medium-sized rivers)\(^b\)  
A ratio >1:5 leads to failing of good ecological status in small and medium-sized rivers | Case by case evaluation in large rivers (as they are more sensitive)\(^c\) |
| Province of Tyrol: Upper Inn River valley | Wasservirtschaftlicher Rahmenplan Troler Oberland, 2014 | After implementation of regional proposed hydropower projects, hydropeaking-induced flow changes should be <15 cm h\(^{-1}\) for upramping and <12 cm h\(^{-1}\) for downramping in all affected reaches. | Strategic planning instrument, detailed case by case analysis               |
| Canada                   | Fisheries Act from 1985 – last amended on April 5, 2016 (Canadian Ministry of Justice, 1985) | –                                                                                                           | Case by case                                                              |
| Finland                  | Water Act 2011 (Finnish Ministry of Environment, 2011)                                | –                                                                                                           | Case by case                                                              |
| Germany                  | –                                                                                      | –                                                                                                           | Case by case\(^d\)                                                        |
| Province of Baden-Württemberg | Wassergesetz für Baden-Württemberg (WG), 2013                                      | –                                                                                                           | –                                                                         |
| Italy                    | –                                                                                      | <1:3 at new facilities                                                                                     | Case by case evaluation of mitigation measures for impacted rivers       |
| Liechtenstein            | Gewässerschutzgesetz (GschG), 2003                                                    | –                                                                                                           | Structural or operational measures must prevent ecological impairment     |
| Norway                   | Water Regulation Act (”Vannforskrifter”) (Miljøverndepartementet, 2006)               | –                                                                                                           | Case by case                                                              |
| Spain                    | Instrucción de Planificación Hidrológica (ARM/2656/2008; 10 Sept. 2008); River Basin Management Plans (Confederaciones Hidrográficas de España, 2008) | Maximum rate of flow variation – a percentile <90–70% is recommended.                                      | River basin\(^e\)                                                        |
| Switzerland              | Gewässerschutzgesetz (GschG), 1991; Gewässerschutzverordnung (GschV), 1998; BAFU – Bundesamt für Umwelt, 2012, BAFU – Bundesamt für Umwelt, 2017 | Flow ratio <1:1.5 and abundance, composition, or diversity of local biota shall not be adversely changed\(^f\) | Each indicator category has its separate thresholds determining the ecological status classes (e.g., TABLE 5.3) |
| Sweden                   | Swedish Environmental Code 1999 (SEPA, 2017)                                          | –                                                                                                           | Case by case                                                              |
| United States of America | Clean Water Act (CWA), 2002 – Section 401: Water Quality Certification (WQC); Endangered Species Act (ESA), 1973; Federal Power Act (FPA), 1920 | –                                                                                                           | Case by case                                                              |

---

\(^a\) Mitigation of rapidly changing flows (incl. hydropeaking) is included in the national list of mitigation measures (according to Halleraker et al., 2016).

\(^b\) Threshold for attaining the “good ecological status” with a high probability. The “very good ecological status” can only be reached if anthropogenic river stage fluctuations (hydropeaking) do not occur.

\(^c\) In large rivers, any hydropeaking is considered as significant pressure.

\(^d\) Hydropeaking operations shall be avoided; the water authority remains the right to authorize exceptions (§ 23 (2) WG, 2013).

\(^e\) Each river basin authority is responsible for defining and calculating the maximum rate of change based on mean daily flow values.

\(^f\) Threshold for “non-significant pressure”. 

---

Table 5.2: Status of hydropeaking legislation thresholds and target values within the studied countries (only countries with information are displayed).
According to the regional Water Management Plan (WNP, 2017), it is not possible to derive general threshold criteria to mitigate the impact of existing hydropaking facilities. In these cases, the necessary measures will be defined and assessed individually within the framework of river protection plans.

In Finland, the Water Act 2011 (Finnish Ministry of Environment, 2011) defines general permit requirements for water resources management projects (ch. 3), but does not set general hydropaking thresholds. Hydropaking permits are set after a case-specific impact assessment. Projects with permits issued before 1 May 1991 may undergo an environmental investigation if considerable detrimental impacts on the aquatic environment are detected and the fisheries authority or municipality may apply for a review of the permit regulations or impose new regulations (ch. 19, sec. 7–8).

There are no legal thresholds for hydropaking in France. Rules are negotiated case by case. Nevertheless, for hydropower plants >4.5 MW, the procedure of concession includes specifications regarding water management issues such as minimum flow, turbine flow or hydropaking, which are defined in the Environmental Code (Code de l’Environnement, 2000) (L211-1 and L214-1 to L214-6).

All hydropaking operations in the province of Baden-Württemberg, Germany should be avoided (WG, 2013, §23 (2)), where the water authority is entitled to authorize exceptions. According to §126 (5), it is an administrative offence if non-authorized hydropaking occurs.

Liechtenstein legislation (GSchG, 2003, Art. 34a §1) states that the operators of hydropower facilities must prevent the impairment of native animals and plants through hydropaking operations by structural measures. At request of the hydropower plant owner, the government may also allow operational mitigation measures and can determine the type of measures and the deadlines to their implementation (§3). Compensation basins built for hydropaking mitigation may be used for pump-storage hydropower without the need of amendment to the license (§4).

In Spain, the River Basin Management Plans (Confederaciones Hidrográficas de España, 2008) recommends maximum rates of discharge variation for each river basin. These values must be estimated based on the analysis of mean annual flows series with, at least, 20 years. The annual rate of change should be calculated from the time series for both up- and downramping rates. The annual series of discharge variation rates, for up- and downramping, shall be computed. It is recommended that the mean rate of change shall not exceed the 90–70% percentile of those time series, for both up- and downramping values. In some particular cases, it may be necessary to consider a refined time scale, which may allow limiting the rate of change at an hourly level.

The Norwegian Water Regulation Act (Miljøverndepartementet, 2006) was adopted in 2006 to include the goals of the EU WFD. A report on setting environmental flows to implement the WFD in Norway (Bakken et al., 2012) devotes a chapter on hydropaking. However, general operational hydropaking mitigating measures have not yet been defined. From 2009 to 2016, a national hydropaking research project was carried out (‘EnviPEAK’, see Bakken et al., 2016a), where the outcomes were a set of guidelines in how to perform environmental adapted hydropaking operations in rivers. These guidelines include recommendations on maximum flow ratios, water level reductions, timing of the year/day and frequency, in the context of the considered rivers vulnerability exposed to hydropaking (Bakken et al., 2016b). Although some of these guidelines have been applied in few hydropaking rivers during the revision of hydropower licenses, those license requirements are still mostly issued on a case-by-case basis, as
each hydropower installation is unique, and historically there are few restrictions on hydropoeaking operations (L’Abée-Lund and Otero, 2018).

The Swiss legislation demands that major impairments caused by short-term pulsed flow shall be remedied by 2030, primarily through structural, but also by operational measures (Schweizer et al., 2016; Tonolla et al., 2017). A significant harm is present if the ratio between base-flow and peak-flow exceeds 1:1.5 and if the abundance, composition, or diversity of the local biota is adversely changed. To evaluate the biological aspects, the Federal Office for the Environment (BAFU – Bundesamt für Umwelt, 2012, 2017) developed a list of 15 indicators, divided into four categories (core indicators, hydropoeaking-sensitive indicators, broadband indicators, additional indicators) and five ecological status classes (TABLE 5.3). An adverse change is present if most of the core indicators show a moderate status, or if one core indicator shows an unsatisfactory or bad status (core indicators include: hydrological parameters, stranding of fish, spawning grounds of fish, habitat suitability for fish/macrozoobenthos, water temperature) (TABLE 5.3).

The Swedish Environmental Code was adopted in 1998 to combine 15 other acts, including the Water Act from 1918 (SEPA, 2017). A specific system which was established for the use of water resources, including a permit regime for water operations, and entered into force in 1999. Any hydropower plant or dam must have a permit which coheres with chapters 3–4 of the Code (river protection measures from hydropower exploitation). Regarding hydropoeaking, the permit will specify the highest and lowest water levels allowed in the reservoir, as well as the maximum and minimum discharge (and the corresponding rate of change) released from the dam and power station. Thus, hydropoeaking is generally allowed as long as the maximum and minimum water levels and discharge values set by the court are not exceeded.

### 5.4.2.2 North America

Hydropoeaking-specific regulations do not exist yet in Canada. However, the Canadian Fisheries Act (Canadian Ministry of Justice, 1985), the national legal instrument for water management and protection, can be used for peak-flow attenuation through, for example the prohibition of works that result in the harmful alteration, or disruption or destruction of fish habitat (Section 35). Furthermore, the governor in council may make regulations for, among others, the conservation and protection of fish, including their spawning grounds (Section 43(1)).

Although the United States of America do not have hydropoeaking-specific legislation as well, the Clean Water Act (CWA) (Federal Water Pollution Control Act, 2002), the Endangered Species Act (ESA, 1973) and the Federal Power Act (FPA, 1920) can be used in hydropoeaking-power permit negotiations. Any activity that may result in a discharge to U.S. waters must provide a Water Quality Certification (CWA – Section 401), in which the applicant declares that the discharge will comply with the applicable provisions of the act, including water quality standards. If there is sufficient justification and a supporting administrative record, this certification could include restrictions on hydropoeaking. If endangered or threatened species are present within the hydropoeaking reach, the Endangered Species Act may be used to stipulate conditions on a hydropower project to protect, restore or enhance certain species. If pulsed flow operation is likely to adversely affect a species listed under the Endangered Species Act, the U.S. Fish and Wildlife Service and the National Marine Fisheries Service may issue a biological opinion that contains conditions which require a modification to project operations. The Federal Power Act provides the groundwork for cooperation between the Federal Energy Regulatory
Commission (FERC) and other federal agencies in (re-)licensing hydropower projects. Section 10(j) allows Fish and Wildlife agencies to submit recommendations, for example regarding project operations that the FERC must consider when issuing a license.

5.5 Discussion

5.5.1 National legislation, regulations and recommendations

5.5.1.1 Europe

There is still a lack of quantitative hydrological thresholds for the mitigation of adverse ecological effects of hydropoaking. Unsurprisingly, only a few countries have adopted precise thresholds in national legislation and guidelines. Of these, the Swiss water laws contain the highest level of detail (e.g., TABLE 5.3; BAFU, 2017). By setting these thresholds, Switzerland has established various targets for hydropoaking mitigation until 2030 (Tonolla et al., 2017). Considering that many questions regarding the ecological effects of peak-flow attenuation still have to be more deeply addressed, it is questionable if setting thresholds for the next decades is suitable at this stage. Even now, some of the established thresholds do not necessarily reflect the current state of the art from hydropoaking research. For example, a downramping rate of <0.2 cm min\(^{-1}\) is enough to attain the very good ecological status during the larval life-stage of brown trout and grayling (cf. TABLE 5.3). Although this value will probably prevent stranding of grayling, a more stringent threshold of 0.1 cm min\(^{-1}\) might be necessary to halt stranding of brown trout larvae (Auer et al., 2014). Furthermore, if multiple events occur in one day, only the greatest and the lowest event are considered. Depending if this daily hydropoaking event is a distinct or a recurring event, the threshold targets of the various indicators must only be attained in 95% or 60% of the days (BAFU, 2017). Considering the high sensitivity of, for example post-hatched gravel life-stages (Becker et al., 1982), spawning ground dewatering can have detrimental effects on a fish population if occurring only 5% of the time.

Austria also adopted rather specific hydropoaking thresholds. Modeling discharge ratios of 1:3, Hauer et al. (2014) found that four out of ten channel bar sites featured a change in the wetted area >20%, which was caused by different river morphologies. Furthermore, Hauer et al. (2016) pointed out that base-flow conditions are entirely different between the seasons and, regarding river morphology, will lead to different extents of the ramping zone, even if the ratio remains the same. Therefore, the authors conclude that such ratios cannot universally be established as a general basis for mitigation thresholds if seasonal aspects of base-flow magnitude, as well as river morphology, are overlooked (Hauer et al., 2016). Additionally, these Austrian thresholds refer only to the good ecological status, whereas most existing hydropoaking rivers have the good ecological potential as a target condition. So far, the good ecological potential has not yet been defined, but feasibility studies have to be carried out by 2021 and then designed and implemented on a river-by-river basis by 2027. Therefore, the integrative assessment approach as developed by Greimel et al. (2017) is being applied in different case studies.

5.5.1.2 North America

In the USA, many hydroelectric dams are subject to relicensing by the Federal Energy Regulatory Commission (FERC) (Young et al., 2011). Although no hydropoaking-specific legislation exists, several laws affect hydropower relicensing and they require consideration or inclusion of conditions for the protection, mitigation, or enhancement of fish resources.
One example is the Skagit River Hydroelectric Project, Washington, where eggs and embryos of salmon and steelhead shall be protected from dewatering, and stranding of salmonid fry on gravel bars shall be minimized (Connor and Pflug, 2004). Therefore, the difference between spawning and incubation periods flows was reduced, which decreased the river area subjected to dewatering (see Table 5.1). To prevent stranding of fry, downramping was limited to night time hours, whereas also downramping rates and the amplitude of flow fluctuations were lowered. These measures boosted the fish population, which showed a steady yearly increase in spawner numbers of 5.2% (Connor and Pflug, 2004). Similarly, the Vernita Bar Settlement Agreement (Harnish et al., 2014), implemented on the Columbia River in 1984, includes discharge constraints to prevent Chinook salmon of spawning at higher water levels (see Table 5.1). During the fall spawning period, redd site selection (which was thought to occur mainly during daylight hours) should be limited to lower elevations by reversing the normal load-following pattern, providing low discharges during the day and higher discharges at night. In 1999, the Hanford Reach Fall Chinook Protection Program Agreement was enacted to protect other life-stages as well. Changes in dam operation led to a 217% increase in salmon productivity in comparison to the period before the Vernita Bar Settlement Agreement, which corresponded with constraints enacted to prevent redd dewatering. An additional increase of 130% coincided with enactment of constraints to limit stranding and entrapment of juveniles during the period of emergence and early rearing (Harnish et al., 2014).

### 5.5.2 Mitigating direct hydropeaking impacts through thresholds and targets: biological and hydromorphological variables

Hydropeaking events are defined by the magnitude of flows on one hand, and their timing on the other hand. Parameters such as the rapid decrease of flow and stage, daylight conditions and duration of wetted history are of ecological significance in terms of stranding risk (Halleraker et al., 2003; Irvine et al., 2009; Saltveit et al., 2001), as well as for dewatering of spawning grounds (Fisk et al., 2013; Casas-Mulet et al., 2016; McMichael et al., 2005) and rapid within-day flow increases are of major importance concerning downstream displacement of fish (Auer et al., 2017; Boavida et al., 2017; Flodmark et al., 2006; Jensen and Johnsen, 1999;
Scruton et al., 2003; Thompson et al., 2011; Zeiringer et al., 2014). Thus, the hydrological parameters (i.e., magnitude, duration, frequency, flow ratio and rate of flow change) which are related with distinct ecological responses may be used to define mitigation thresholds, where its design should consider key species and their ecological requirements (Bruder et al., 2016; Hauer et al., 2017). Furthermore, hydromorphological conditions must be included in the definition of mitigation measures since they are crucial for fish survival as well (Hauer et al., 2014, Hauer et al., 2017). Accordingly, FIGURE 5.5 presents a scheme with the sequence of the main aspects and the corresponding biological and hydromorphological variables that should be considered when designing thresholds and targets for hydropoeaking mitigation.

5.5.2.1 Species

Literature indicates that some species are more vulnerable to stranding than others. For example, brown trout are more sensitive than European grayling (FIGURES 5.2–5.3), and Coho salmon has a higher stranding risk than rainbow trout (FIGURE 5.4). Therefore, hydropoeaking mitigation designs shall select the species with the highest sensitivity to artificial flow fluctuations, assuming that all other species will be indirectly protected. Endangered species may also be considered of higher priority, although this does not necessarily assure the critical thresholds of the most sensitive species, such as in many Austrian rivers where brown trout and grayling cohabit. Although the grayling has a higher importance in terms of national protection status (Uiblein et al., 2001), brown trout are more sensitive to hydropoeaking (cf. FIGURES 5.2–5.3).

Sensitivity among species may also vary depending on life history strategies and behavioral patterns. Highly territorial species such as salmonids may be more vulnerable to stranding as they can be reluctant to abandon spawning territories during receding water levels (Boavida et al., 2017), while cyprinid species, typically of lower swimming performance compared to salmonids, may not have enough resistance to achieve a suitable habitat during downramping (Santos et al., 2014). Some studies also found that hydropoeaking may influence fish assemblages in general (e.g., Enders et al., 2017; Hedger et al., 2018; Sauterleute et
al., 2016; Scruton et al., 2008), while García et al. (2011) concluded that artificial flow fluctuations may provoke distinct impacts on native and non-native species.

Therefore, hydropeaking mitigation measures should consider, as a first step, the specific requirements (incl. sensitivity and life-history strategy) of the species present in the impacted river reach, as well as their conservation status. Targeting indicator or threatened species will indirectly improve the conditions of other species as well.

5.5.2.2 Life-stage

Literature shows that, in hydropeaking rivers, various life-stages can be influenced by different hydrological parameters. Salmonid eggs can survive dewatering for weeks in dewatered gravel if they are kept moist (at least 4% moisture by weight), do not freeze and are not subject to predation, or if temperatures do not exceed incubation tolerances (e.g., Becker et al., 1983; McMichael et al., 2005). Although salmon eggs are tolerant to dewatering, mortality increases once fish have hatched and larvae are dependent on gills for respiration. Thus, special attention should be given to newly hatched alevins, which are less tolerant and may die within a short time of dewatering (Becker et al., 1982; Fisk et al., 2013). Peak-flows may create temporarily suitable habitat for gravel-spawning fish, which will be subjected to periodic dewatering between the pulsed-flow releases (McMichael et al., 2005; Vocht and Baras, 2005). Therefore, peak flow reductions, combined with minimum flow releases, are a common mitigation recommendation to reduce early life-stages mortality (Table 5.1). The sooner and the longer minimum flow release is implemented during the spawning period, the higher is the probability of fish not spawning in high mortality risk areas (Casas-Mulet et al., 2016).

Juvenile fish are more susceptible to hydropeaking events than adults, as juvenile habitat is confined to the shallow banks, where their risk of stranding is enhanced, since they might not reach the central part of the channel during downramping event. In contrast, adults tolerate a wider range of stream conditions (Enders et al., 2017; Pragana et al., 2017; Saltveit et al., 2001). This is in line with our findings from literature, which show that fish are less likely to get stranded as they grown in size (Figures 5.2–5.3). Therefore, the establishment of hydropeaking thresholds should consider not only the species present, but also the respective life-stage and the associated season.

Furthermore, intra-annual flow differences have to be considered, especially when determining base-flow magnitudes, as life-cycle phases and their flow requirements are connected to certain periods of the year (Hayes et al., 2018). For example, fish movements are related to discharge alterations (Berland et al., 2004; Boavida et al., 2017; Jones and Petreman, 2015), which can vary according to seasons (Katzman et al., 2010; Scruton et al., 2005), where high flow fluctuations may affect spawning behavior. Under these conditions, different studies found out that both Chinook salmon, Oncorhynchus tshawytscha, and common barbel, Barbus barbus, repeatedly abandoned spawning redds before completion (Hamilton and Buell, 1976, in: Young et al., 2011; Vocht and Baras, 2005). In such situation, Chinook salmon may decide to move to less desirable and more crowded locations (Hunter, 1992).

5.5.2.3 Time of day

In hydropeaking rivers, seasonal flow thresholds which aim, for instance, at avoiding redd dewatering or stranding and drifting of larvae and juveniles, may attenuate negative effects on fish populations. However, diel variations have to be considered as well. In some cases, the discharge decrease should only be performed after dark to reduce the stranding risk of some
salmonid species, especially during winter when fish are less mobile and often hide in the sub-
strate during the daytime (Saltveit et al., 2001; Stickler et al., 2007), suggesting to limit dis-
charge-induced downramping to night time hours (Connor and Pflug, 2004). Similarly, after
modeling different operation scenarios in a Por-
tuguese river reach, Pragana et al. (2017) recom-
mand that, in winter, downramping should be
performed after 5 or 6 PM, and in the summer after 9 PM, to minimize impacts on juvenile
brown trout habitat. In contrast, other studies
concluded that, in summer, European grayling
(Auer et al., 2017) and brown trout (Auer et al.,
2014; Halleraker et al., 2003), as well as Austri-
an fish communities generally (Schmutz et al.,
2015) are less vulnerable during the day than
during the night. From the majority of studies,
it can be deduced that downramping thresholds
should be more stringent during nighttime in
summer as well as during daytime in winter, al-
though some recommendations (e.g., Connor
and Pflug, 2004; Pragana et al., 2017) do not
confirm this generalization. The literature is,
therefore, not completely consistent on the issue
whether is better to have a peak event during the
day or during the night since it may vary accord-
ing to species-specific characteristics and season.
It is clear, however, that the flow reduction rate
should be set to give fish sufficient time to leave
sheltered habitats near the substrate and to reach
the main channel, irrespective of time of day.

5.5.2.4 Hydromorphology

Multiple studies indicate that the impact of
hydropeaking is strongly dependent on river
reaches morphology (e.g., Boavida et al., 2015;
Bradford, 1997; Hauer et al., 2013, Hauer et
al., 2014; Parasiewicz et al., 1998; Tuhtan et
al., 2012; Vanzo et al., 2015). Person et al. (2014)
showed that braided reaches offer the best hab-
itat suitability in terms of quantity and stability
for different brown trout life-stages in compari-
son to other morphological types (e.g., groynes,
gravel bars, straight channel). Authors conclud-
ed that spawning and young-of-year life-stages
depict higher sensitivity to the discharge fluctu-
atations than adults for all morphologies. Due to
their wide riverbed, braided reaches are able to
retain the rapid fluctuations effects and to pro-
duce varying velocity conditions that may be
suitable for brow trout and other fish in differ-
ent life-stages (Person et al., 2014). Neverthe-
less, stranding risk was not considered in their
assessment. Vanzo et al. (2015) also concluded
that braided reaches are the most resilient to hy-
dropeaking, offering the highest habitat diver-
sity, and found out that alternate bars are ex-
tremely sensitive environments to drift but offer
safer regions from stranding.

Furthermore, several studies on salmonid
fish demonstrated that stranding risk is positive-
ly correlated to the presence of sheltering areas
or potholes (e.g., Auer et al., 2017; Saltveit et
al., 2001; Scruton et al., 2008). Fish may hide
in these spots during peak-flow events to escape
from high velocities, but when flow is reduced,
fish may get entrapped. Larger juveniles and
adults are more likely to inhabit deeper pools,
glides, overhanging banks, and mid-channel
habitats where they are less vulnerable to strand-
ing and entrapment (Hunter, 1992; Nagrodski
et al., 2012). In contrast, early juvenile life-stages
prefer shallow habitats along the river margins,
which is part of the ramping zone and might get
dewatered. In this regard, a river channel with
many side channels, potholes, and low gradi-
ent bars has a greater stranding potential than
a river with a single channel with steep banks
(Hunter, 1992). However, steep banks are less
favorable for juvenile fish. Controlling ramping
rate might be effective in reducing stranding
along the river margins but proved to be less ef-
fective for pothole and side channel entrapment
(Higgins and Bradford, 1996; Hunter, 1992). In
the latter cases, flows should be increased before
downramping to remove fish from potholes, combined with a low rate decrease that would allow their save return to the channel (Higgins and Bradford, 1996).

Coarse grain sizes on a smooth bank slope are another factor which can increase stranding risk (Boavida et al., 2015; Bradford, 1997; Hauer et al., 2014). Unsurprisingly, Hauer et al. (2014) stress the necessity to consider grain-size distribution of gravel bar surfaces when establishing peak operation thresholds and/or discharge variability in seasonal base-flow targets. In contrast to stranding, the presence of coarse substrate, acting as a velocity shelter, can help fish to avoid downstream displacement in a hydropeaking river (Heggenes, 1988). Multiple studies highlighted the importance of substrate as one of the main parameters structuring fish assemblages in hydropeaking rivers (e.g., Boavida et al., 2015; Chun et al., 2010; Scruton et al., 2008).

Due to river hydromorphology and related retention effects, hydropeaking parameters, such as downramping rate, vary along the course of the river, where the intensity of the impact is mostly directly below the tailrace and is reduced in downstream direction (Hauer et al., 2017; Halleraker et al., 2007). Therefore, the longitudinal variability in hydropeaking reaches must also be considered when defining flow mitigation thresholds.

5.5.3 Indirect impacts: macroinvertebrates

Pulsed flows may also have indirect impacts on fish through effects on food supplies such as benthic macroinvertebrates, which comprise the principal food source of fish populations (Cushman, 1985). As invertebrate populations are diminished, fish growth can be reduced (Bruno et al., 2010; Irvine, 1986; Moog, 1993). Hydropeaking negatively affects density, biomass and species diversity through the catastrophic drift occurring during peak-flow, particularly when combined with high content of suspended solids, and, for some taxa, through the behavioral drift in the base-flow conditions (Bruno et al., 2010; Moog, 1993). Also, the effects of thermopeaking on the drift of benthic invertebrates have been reported (Carolli et al., 2012; Schülting et al., 2016). In Europe, the assessment metrics and benthic habitats regarded in WFD may not reflect the effects of hydropeaking events (Leitner et al., 2017), which may require further research for the development of mitigation strategies regarding the benthic communities.

5.5.4 Economic impacts of mitigation thresholds

Hydrological mitigation thresholds can be achieved either through operational measures, as well as structural measures such as the construction of hydropeaking retention basins or hydropeaking diversion power plants (Greimel et al., 2018a). The latter requires suitable topographic conditions and a significant first-time investment but does not impact the ongoing hydropower operation. In contrast, operational measures entail ongoing restrictions in the power plant’s operation mode (Premstaller et al., 2017), reducing the capacity to produce flexible energy according to the current demand and leading to economic losses which are proportional to the intensity of the mitigation thresholds (Greimel et al., 2018b; Hauer et al., 2017). Additionally, some other possible technical constraints such as the start-stop operation and type and number of turbines may limit the application of those measures (Harby and Noack, 2013).

The importance of peak-flow operating hydropower in the energy grid and the adverse ecological impacts need to be balanced. Therefore, operational measures are being evaluated using a cost-benefit approach that assess the trade-offs involved (Niu and Insley, 2013). These include the costs imposed on hydropower operators in terms of lost profits, as well as
potential environmental impacts that result from the need to use alternative sources of
electricity (Niu and Insley, 2013; Pérez-Díaz and Wilhelmi, 2010).

5.5.5 Research needs

5.5.5.1 Units for defining hydropower mitigation thresholds

Stranding thresholds for the vertical ramping rate variation are reported in different velocity units, mainly cm h$^{-1}$ and, more recently, cm min$^{-1}$. When designing such flow constraints, it is important to consider not only how post-implementation and monitoring will be addressed. On the one hand, if discharge data is available only with hourly values, it might be more reliable to define thresholds in cm h$^{-1}$. On the other hand, if a finer scale of discharge is available (e.g., 15 min interval), it may be more feasible to monitor thresholds implementation in cm min$^{-1}$. From an ecological point of view, however, the units monitored also have to be in accordance with ecological processes to be investigated. Stranding, for example, is a behavioral response taking place within the time scale of minutes, so it might be more coherent to define thresholds in cm min$^{-1}$ instead of cm h$^{-1}$. However, no research has considered this topic yet, which may be a drawback when defining hydropeaking mitigation thresholds.

5.5.5.2 Lateral ramping velocity

The lateral gradient of river banks will, to a large extent, determine the extent of the ramping zone which can become dewatered. Studies found that stranding is lower on steeper river bars and was reduced when the bank slope was greater than 2% (Bradford et al., 1995; Monk, 1989, in: Schmutz et al., 2015), indicating that there is a trade-off between losing shallow water habitat and reducing stranding risk. Furthermore, it has been suggested that stranding susceptibility seems to be more related to the rate of stream margins dewatering (lateral ramping velocity), than to the vertical downramping rate (Hauer et al., 2017; Tuhtan et al., 2012). Hence, the lateral gradient of the river bar seems to play an important role in wetted history variation, which is a key parameter for stranding risk assessment and, therefore, for mitigation. Nevertheless, no thresholds were found in literature for lateral ramping velocity.

5.5.5.3 Non-salmonid species

Although most of the hydropeaking studies have been focusing on salmonid species (Nagrodski et al., 2012), some attention has been given to non-salmonid species such as cyprinids over the last decade (Alexandre et al., 2015, Alexandre et al., 2016; Boavida et al., 2015; Capra et al., 2017, Capra et al., 2018; Costa et al., 2018). However, in our literature search, we did not find thresholds or mitigation targets for cyprinids, which underlines the research need of this fish family, which is the largest in the world, and other non-salmonid species inhabiting hydropeaking rivers.

5.5.5.4 Thermopeaking, saturopeaking and soundpeaking

The release of hydropeaking discharges can also entail thermal alterations, where their duration is similar to that of the hydropeaks (Zolezzi et al., 2011). However, as most studies only deal with the effects of long-term temperature changes associated with hydropeaking (e.g., Céréghino et al., 2002), there is a lack of information on the short-term ecological effects of thermal alterations (Bruno and Siviglia, 2012; Zolezzi et al., 2011). Observations in fish migration found that the start of migration was linked to an increase in water temperature and a decrease in discharge (Benitez and Ovidio, 2017), which may be affected by (thermo)peaking events. Thus,
there is a need to assess the influence of thermopeaking on, for example, migration, spawning, larval growth rates, or on the behavioral drift of fish species (Zolezzi et al., 2011).

Similar to thermopeaking, also gas saturation can follow the pattern of hydropeaking operations (Pulg et al., 2016). Depending on fish species and life-stage, the levels at which supersaturation is harmful may begin at 103–100% of the total dissolved gases (TDG) saturation (Jensen et al., 1986). In natural environments, fish can compensate for supersaturation by moving into deeper water (e.g., 0.3–0.8 m) (Beeman and Maule, 2006), which is why the Canadian guidelines for supersaturation distinguish between deep (>1 m) and shallow water bodies, defining 110% and 103% TDG as the thresholds for deep and shallow rivers, respectively (Canadian Council of Ministers of the Environment, 1999). Nevertheless, so far there are no guidelines for supersaturation in European rivers, as possible ecological effects of saturopeaking in hydropeaking rivers still require more research (Pulg et al., 2016).

Soundscapes affected by hydropeaking are highly homogenized, when compared to unaffected ones, and sound pressure level variations are strongly correlated with turbine discharge, which results in rapid, multiple-fold spikes in low frequency amplitude levels (Lumsdon et al., 2018). As a consequence, fish or macroinvertebrates may be affected physiologically or behaviorally, but further research on this topic is needed to examine the response of biota to changes in soundscapes (Lumsdon et al., 2018).

### 5.5.5 Reporting and monitoring of implemented measures

Most hydropeaking studies report on adverse flow alteration-ecological response relationships and, based on these insights, propose mitigation measures. However, so far there are only a few papers reporting on the outcomes of the implemented measures, where most of these were implemented in the USA (e.g., Connor and Pflug, 2004; Fisk et al., 2013; Harnish et al., 2014) and Cazeneuve et al. (2009) present a French case study. Assessing the success of implemented measures is, therefore, an important step for future hydropeaking mitigation strategies and regulation development.

### 5.6 Conclusions

Hydropeaking causes severe changes in riverine environments, entailing adverse responses of organisms (e.g., Bejarano et al., 2017b). It is, therefore, stringent to develop ecologically-based criteria for hydropeaking mitigation. In-situ studies, laboratory experiments and numerical modeling are of vital importance to specify terms and conditions that minimize the effects of hydropeaking through the establishment of threshold standards and mitigation targets. These values should be achieved by adapting hydropower plants operation, or by constructing infrastructures to attenuate discharge fluctuations in the river (Charmasson and Zinke, 2011).

Reviewing the literature, we found that, so far, only few studies published quantitative hydropeaking thresholds for operational mitigation measures, most of them established for salmonid fish through stranding trials in experimental channels. Research showed that low downramping rates reduce the stranding risk, whereas exact thresholds are related to species, life-cycle stage, time of day, and river morphology. Other studies recommend management approaches to improve spawning and rearing success, such as restricting peak flows during spawning and raising minimum flows during incubation to prevent redd dewatering. Furthermore, literature indicates that the impact of hydropeaking is strongly
dependent on river reaches morphology, especially site-specific characteristics, such as lateral bar angle, grain size distribution, shelters or potholes, which have to be considered when to prescribe mitigation measures. Nevertheless, due to the above-described site-specific characteristics, the intensity of some hydraulic parameters, such as vertical ramping rate, will decrease longitudinally with distance from the turbine outlet, but this is not necessarily true for other parameters, such as lateral ramping velocity, which proved to be highly variable (Hauer et al., 2017).

Due to these factors which have to be considered in hydropeaking rivers, it is not surprising that, so far, only two countries (Austria and Switzerland) have established legal regulations regarding hydropeaking discharges. Other countries established constraints on a regional level (e.g., Germany, Italy). Few countries have recommendations for hydropeaking mitigation (e.g., Spain), while others have regulatory frameworks that may force a case-by-case analysis under specific legal requirements (e.g., Norway, USA). The lack of published literature reporting on the success of implemented measures might thus indicate that few measures have yet been implemented due to the shortage of legal regulations.

Although it might be hard to determine national thresholds due to case-specific effects of hydropeaking impacts, it is urgent to mitigate the ecological impacts caused by flow fluctuations, considering environmental objectives such as demanded by the WFD in Europe. Nevertheless, literature indicates that multiple aspects have to be considered when assessing mitigation targets. To assist in this process, we present a scheme regarding the main aspects and the corresponding biological and hydromorphological variables which should be considered for the design of hydropeaking mitigation measures with a focus on fish. We propose that mitigation targets and thresholds must be based on key species (e.g., hydropeaking-sensitive, protected or territorial species), including particular features regarding season, a parameter that determines life-stage phases (e.g., focusing on vulnerable life-stages, such as larvae) and diel variations, which must be combined with site-specific morphological characteristics (e.g., river geometry or bank gradient, grain size, habitat structures). Furthermore, the potential impacts on uses have to be considered when dealing with the ecological potential as target in river sections of heavily modified water bodies. We, therefore, conclude that the ecologically-based criteria for mitigation measures may benefit the impacted organisms in hydropeaking reaches. Nevertheless, further research is needed to establish thresholds and targets for more species and their life-stages throughout different habitat types and, complementary, the monitoring of hydropeaking mitigation implementation, which is not yet a widespread procedure.

5.7 Acknowledgments

MM and DSH were supported by Ph.D. scholarships funded by Fundação para a Ciência e a Tecnologia, I.P. (FCT), Portugal, under the Doctoral Programme FLUVIO – River Restoration and Management, with the respective grant numbers PD/BD/114336/2016 and PD/BD/114440/2016. CEF is a research unit funded by FCT (UID/AGR/00239/2013). IB was supported by the FCT grant SFRH/BPD/90832/2012. This project has received funding from the European Union’s Horizon 2020 research and innovation programme under grant agreement No 727830. The authors express their thankfulness to Christos Katopodis, Diego García de Jalón, Jo Halvard Halvaker, Teppo Vehanen, Laurence Tissot, Maria Cristina Bruno, Jeffrey Tuhtan, Egidijus Kasiulis, Marie Egerrup, Cíntia Veloso Gandini,
5.8 References


