CHAPTER 5

Ecologically-based criteria for hydropeaking mitigation: a review

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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 (Schülting 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 stranding (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 'saturopeaking', 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%, saturopeaking 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 hydropeaking 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 hydropeaking (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 hydropeaking 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 hydropeaking 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 hydropeaking 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 hydropeaking 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 hydropeaking mitigation thresholds by conducting a search for peer-reviewed literature. We used the Scopus database with the search string TITLE-ABS-KEY (“hydropeaking” OR “hydro peaking” 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 hydropeaking 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 “hydropeaking” and “hydro peaking” in TITLE-ABS-KEY.

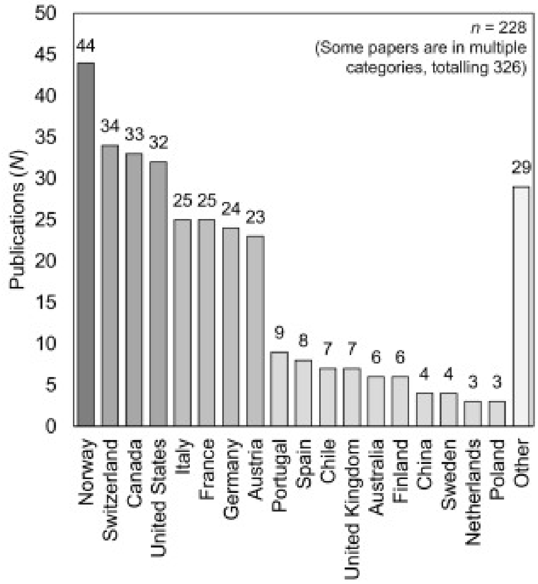

Figure 5.1 Number of papers found in the Scopus database using the keywords "hydropeaking" and "hydro peaking", sorted by country/territory of author affiliation (the literature search includes results until September 2018 based on Title-Abstract-Keywords). "Other" includes: Belgium, Brazil, New Zealand, South Korea, Cyprus, Czech Republic, Denmark, Estonia, Greece, Hong Kong, Iceland, Japan, Croatia, Malaysia, Slovenia, Taiwan, and undefined.

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.

Impact	Species, life-stage	Caused by	Description of operational mitigation measures and hydropeaking thresholds	Type of study	Source
Stranding	Brown trout, *Salmo trutta*, fry and juvenile	Downramping	Decreasing downramping from 1 cm min^{-1} to 0.3 cm min^{-1} reduced the stranding of trout fry by >50% in summer and fall, and almost eliminated stranding of 1+ trout. A further ramping rate reduction to <0.16 cm min^{-1} lead to even less stranding of trout fry.	Lab experiments	Halleraker et al. (2003)
	Brown trout, *S. trutta*, larvae and juvenile (0+)	Downramping	A downramping threshold of ≤0.1 cm min^{-1} during the day and ≤0.05 cm min^{-1} during the night might reduce stranding of larvae, whereas for juvenile (65–70 mm) ≤6.4 cm min^{-1} and ≤3.2 cm min^{-1} are recommended for day and night, respectively.	Outdoor flume experiments (Lunz, Austria)	Auer et al. (2014)
	Atlantic salmon, *Salmo salar*, juvenile (1+)	Downramping	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.	Field study (Nidelva River, Norway)	Saltveit et al. (2001)
	Atlantic salmon, *S. salar*, juvenile	Downramping	Avoiding ramping rates >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.	Modelling (Surna River, Norway)	Halleraker et al. (2007)
	European grayling, *Thymallus thymallus*, larvae and juvenile (0+)	Downramping	To reduce stranding losses in spring (May–July), maximum downramping rates per minute must be lower than 0.6 or 1 m^3 s^{-1} (equaling 7% or 11% of MQ).	Field study (Drava River, Austria)	Unfer et al. (2011)
	European grayling, *T. thymallus*, larvae and juvenile (0+)	Downramping	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.	Outdoor flume experiments (Lunz, Austria)	Auer et al. (2014)
	European grayling, *T. thymallus*, juvenile (0+)	Downramping	During the night, the daylight threshold of <3 cm min^{-1} is also recommended for larger juveniles (∅ 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}.	Outdoor flume experiments (Lunz, Austria)	Auer et al., (2014, 2017)
	European grayling, *T. thymallus*, larvae and juvenile (0+)	Downramping	Lowering the downramping rate to <0.2 and <0.4 cm min^{-1} significantly reduces the stranding risk of grayling larvae and juvenile, respectively.	Outdoor flume experiments (Lunz, Austria)	Schmutz et al. (2013)
	Coho salmon, *Oncorhynchus kisutch*, rainbow trout, *O. mykiss*, juvenile	Downramping	In winter (water temp. <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.	Lab experiments	Bradford et al. (1995)
	Pacific salmon and steelhead rainbow trout, *Oncorhynchus* sp., larvae and juvenile (0+)	Downramping	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.	Field study (Sultan River, USA)	Olson (1990), in: Schmutz et al. (2015)

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Table 5.1 (continued)

Impact	Species, life-stage	Caused by	Description of operational mitigation measures and hydropeaking thresholds	Type of study	Source
	Pacific salmon and steelhead rainbow trout, *Oncorhynchus* sp.	Downramping	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).		Hunter (1992)
	Pink salmon, *Oncorhynchus gorbuscha*, Chum salmon, *O. keta*, and Chinook salmon, *O. tshawytscha*, juvenile (0+)	Downramping	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 $m^3 s^{-1}$ (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 $m^3 s^{-1} h^{-1}$.	Field study (Skagit River, USA)	Connor and Pflug (2004)
Entrapment in side channels and potholes	Multiple, mostly juvenile	Downramping	Flow reduction after a forced reservoir spill lead to side channel and pothole entrapment. Downramping rates of >0.08–0.16 cm min^{-1} trapped many fish, in some cases even at receding flows ranging from 0.04–0.08 cm min^{-1}. An alternative strategy to gradual downramping (for cases where stranding risk in substrate depressions and entrapment in off-channel areas is high), is an increase in flow to remove fish from potholes, coupled with a sudden decrease that would allow little time for their return.	Field study (Bridge River, Canada)	Higgins and Bradford (1996)
	Brown trout, *S. trutta*, Atlantic salmon, *S. salar*, juvenile	Peak magnitude	During the swim-up phase (mid-March to mid-June), flows should not exceed 240 $m^3 s^{-1}$ to avoid pool trapping, as fish are not able to reach higher bank areas with many depressions that will fall dry during downramping.	Field study (Dordogne River, France)	Cazeneuve et al. (2009)
Drift	European grayling, *T. thymallus*, juvenile (0+)	Upramping	Lowering the upramping rate from >3 to 0.5 cm min^{-1} can mitigate the risk of drifting for juveniles (∅ 53 mm TL), especially during night experiments.	Outdoor flume experiments (Lunz, Austria)	Auer et al. (2017)
Spawning redd dewatering	Chinook salmon, *O. tshawytscha*, spawning	Peak magnitude and base-flow conditions	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.	Field survey (Columbia River, USA)	McMichael et al. (2005)
Spawning redd dewatering, mortality of intra-gravel life-stages	Pink, Chum and Chinook salmon, *Oncorhynchus gorbuscha*, *O. keta*, and *O. tshawytscha*, egg and embryo	Peak magnitude and base-flow conditions	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 $m^3 s^{-1}$).	Field study (Skagit River, USA)	Connor and Pflug (2004)
	Chinook salmon, *O. tshawytscha*, egg and alevin	Peak magnitude and base-flow conditions	Redd dewatering can be minimized by providing minimum incubation discharges. The effect is even greater, if these discharge magnitudes are similar to spawning discharges.	Field study and modelling (Columbia River, USA)	Harnish et al. (2014)

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Table 5.1 (continued)

Impact	Species, life-stage	Caused by	Description of operational mitigation measures and hydropeaking thresholds	Type of study	Source
	Chinook salmon, *O. tshawytscha*, egg and alevin	Drawdown to base-flow between peaks	Prevent dewatering of spawning redds after post-hatch life-stages, especially pre-emergence alevins which are very sensitive to redd desiccation (mortality at <1 h dewatering).	Lab experiments	Becker et al. (1982)
	Brown trout, *S. trutta*, Atlantic salmon, *S. salar*, egg and alevin	Drawdown to base-flow between peaks	To prevent dewatering of spawning grounds, minimum flows shall be increased from 10 $m^3 s^{-1}$ to 30 $m^3 s^{-1}$ from mid-November to mid-May, assuring that 90% of the spawning grounds will stay underwater.	Field study (Dordogne River, France)	Cazeneuve et al. (2009)
	Atlantic salmon, *S. salar*, egg	Drawdown to base-flow between peaks	To improve egg survival during cold air temperatures, it is recommended to minimize the duration of hydropower production stops and/or to increase minimum instream flow to prevent exposure of eggs to dry and freezing conditions.	Field study (Lundesokna River, Norway)	Casas-Mulet et al. (2014)
	Robust redhorse, *Moxostoma robustum*, egg and larvae	Drawdown to base-flow between peaks	To reduce dewatering mortality, minimum flows have to maintain the inundation of spawning areas, especially for post-hatched larvae.	Lab experiments	Fisk et al. (2013)
Ecological status	Multimetric index: Fish Index Austria	Ramping rates	Ramping rates of >0.5 cm min^{-1} are 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 river channels.	Modelling	Schmutz et al. (2015)

TL = total length of fish (mm); MQ = average yearly discharge ($m^3 s^{-1}$).

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

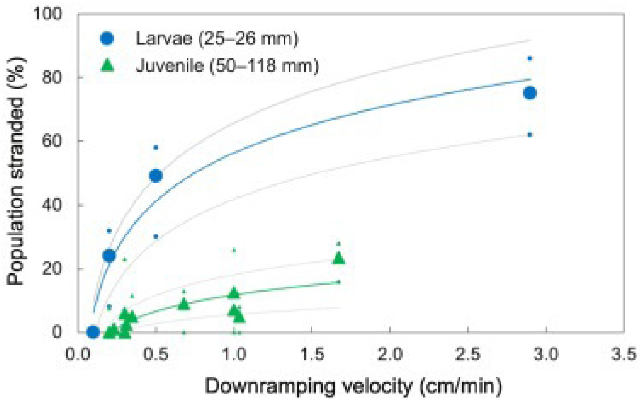

Figure 5.2 Stranding rates of different life-stages of brown trout, Salmo trutta, in relation to downramping velocity during spring and summer daytime experiments on homogeneous gravel bars. The large icons and the dark-colored trendline represent median values, whereas the small icons and the light grey trendline represent the 25th and 75th percentiles reported in the studies. Envelope curves are logarithmic. Four values of juveniles also contain S. salar because Hessevik (2002) did not distinguish between S. trutta and S. salar but grouped them. Data sources: Auer et al. (2014), Halleraker et al. (2003), Hessevik (2002), Saltveit et al. (2001).

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).

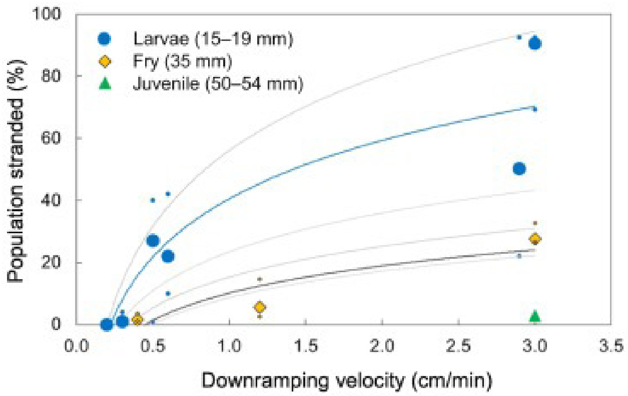

Figure 5.3 Stranding rates of different life-stages of European grayling, Thymallus thymallus, in relation to downramping velocity during spring and summer daytime experiments on homogeneous gravel bars. The large icons and the dark-colored trendline represent median values, whereas the small icons and 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).

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

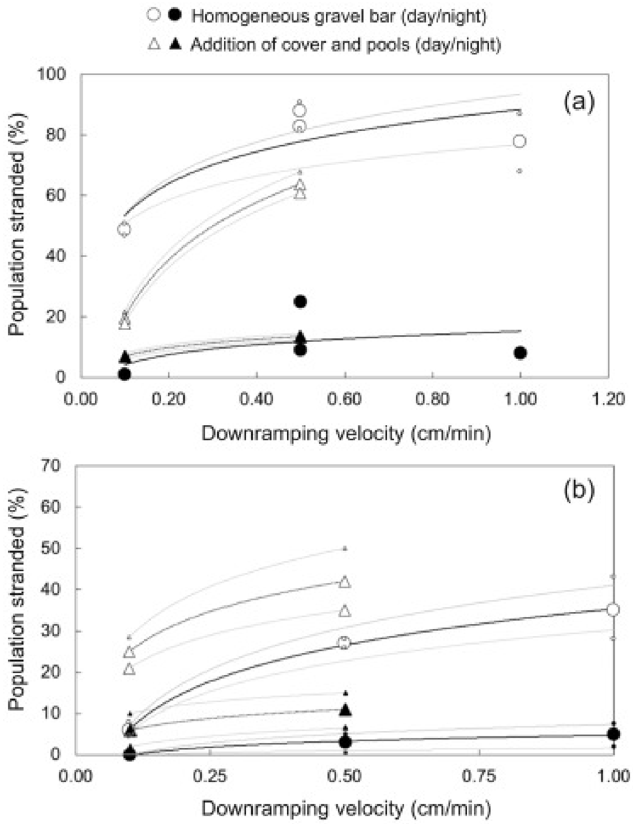

Figure 5.4 Stranding rates of (a) juvenile (53–98 mm) Coho salmon, Oncorhynchus kisutch, and (b) juvenile (79–101 mm) rainbow trout, Oncorhynchus mykiss, in relation to downramping velocity during winter day (white icons) and night (black icons) experiments on homogeneous gravel bars (circles) and with the addition of cover and pools (triangle). The large icons and the black trendline represent median values, whereas the small icons and the light grey trendline represent the standard error reported in the study. Envelope curves are logarithmic. Data source: Bradford et al. (1995).

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 hydropeaking 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 hydropeaking diversion power plants and compensation basins, hydropeaking 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).

Country	Legislation/guideline	Thresholds and target values (description)	Evaluation
Austria[a]	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	Wasserwirtschaftlicher Rahmenplan Tiroler 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[a]	Water Act 2011 (Finnish Ministry of Environment, 2011)	–	Case by case
France[a]	Article L214 – 18 from the Environmental Code (Code de l'Environnement, 2000)	–	Case by case
Germany[a]	–	–	–
Province of Baden-Württemberg	Wassergesetz für Baden-Württemberg (WG), 2013	–	Case by case[d]
Italy[a]	–	–	Case by case
Province of Bolzano	Durchführungsverordnung (6/2008) zum Landesgesetz Nr. 8/2002; Wassernutzungsplan, 2017	<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[a]	Water Regulation Act ('Vannforskriften') (Miljøverndepartementet, 2006)	–	Case by case
Spain[a]	Instrucción de Planificación Hidrológica (ARM/2656/2008; 10 Sept. 2008); River Basin Management Plans (Confederaciónes 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[a]	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".

According to the regional Water Management Plan (WNP, 2017), it is not possible to derive general threshold criteria to mitigate the impact of existing hydropeaking 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 hydropeaking thresholds. Hydropeaking 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 hydropeaking 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 hydropeaking, which are defined in the Environmental Code (Code de l'Environnement, 2000) (L211-1 and L214-1 to L214-6).

All hydropeaking 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 hydropeaking 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 hydropeaking 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 hydropeaking mitigation may be used for pump-storage hydropower without the need of amendment to the license (§4).

In Spain, the River Basin Management Plans (Confederaciónes 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 hydropeaking. However, general operational hydropeaking mitigating measures have not yet been defined. From 2009 to 2016, a national hydropeaking 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 hydropeaking 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 hydropeaking (Bakken et al., 2016b). Although some of these guidelines have been applied in few hydropeaking 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 hydropeaking 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, hydropeaking-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 shows 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 hydropeaking, 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, hydropeaking 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

Hydropeaking-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 hydropeaking-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 hydropeaking-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 hydropeaking. If endangered or threatened species are present within the hydropeaking 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 hydropeaking. 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 hydropeaking 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 hydropeaking 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 hydropeaking 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 hydropeaking 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 hydropeaking 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 hydropeaking-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.

Table 5.3 Swiss legislation core indicator "stranding thresholds" (BAFU, 2017).

Ecological status	Dry falling surface area in relation to wetted area (%)	Downramping rate for larval grayling/ brown trout at daylight ($cm\ min^{-1}$)	Downramping rate for juvenile grayling at daylight ($cm\ min^{-1}$)	Downramping rate for juvenile brown trout at daylight ($cm\ min^{-1}$)
Very good	<10	<0.2	<1	<1.5
Good	10–30	0.2–0.3	1–1.2	1.5–3
Moderate	30–40	0.3–0.4	1.2–2	3–4.5
Unsatisfactory	40–50	0.4–0.5	2–3	4.5–6
Bad	>50	>0.5	>3	>6

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;

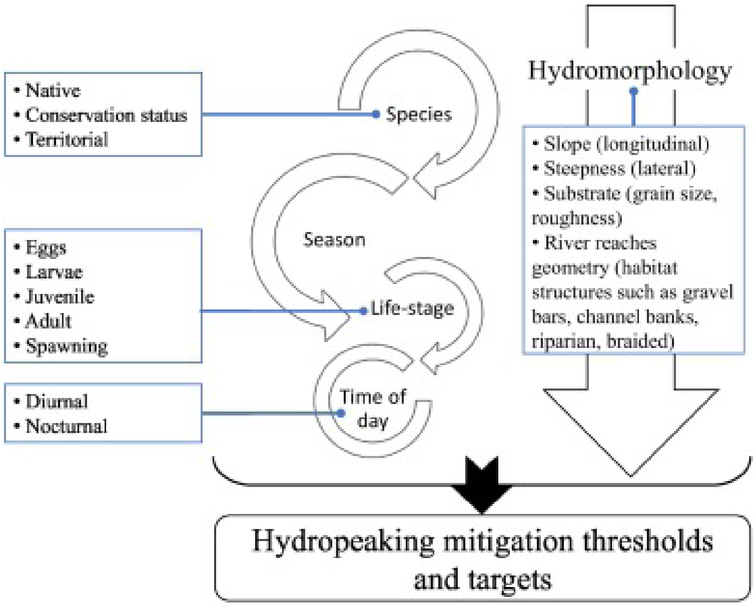

Figure 5.5 Main aspects and corresponding biological and hydromorphological variables for defining hydropeaking mitigation thresholds and targets for fish.

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 hydropeaking 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, hydropeaking 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 hydropeaking (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 hydropeaking 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 substrate during the daytime (Saltveit et al., 2001; Stickler et al., 2007), suggesting to limit discharge-induced downramping to night time hours (Connor and Pflug, 2004). Similarly, after modeling different operation scenarios in a Portuguese river reach, Pragana et al. (2017) recommend 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 Austrian 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, although 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 according 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 habitat suitability in terms of quantity and stability for different brown trout life-stages in comparison to other morphological types (e.g., groynes, gravel bars, straight channel). Authors concluded that spawning and young-of-year life-stages depict higher sensitivity to the discharge fluctuations than adults for all morphologies. Due to their wide riverbed, braided reaches are able to retain the rapid fluctuations effects and to produce varying velocity conditions that may be suitable for brow trout and other fish in different life-stages (Person et al., 2014). Nevertheless, stranding risk was not considered is their assessment. Vanzo et al. (2015) also concluded that braided reaches are the most resilient to hydropeaking, offering the highest habitat diversity, and found out that alternate bars are extremely sensitive environments to drift but offer safer regions from stranding.

Furthermore, several studies on salmonid fish demonstrated that stranding risk is positively 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 stranding 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 gradient 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 effective 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).

5.5.5.5 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.

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5.8 References

Alexandre, C.M., Sales, S., Ferreira, M.T., Almeida, P.R., 2015. Food resources and cyprinid diet in permanent and temporary Mediterranean rivers with natural and regulated flow. *Ecol. Freshw. Fish* 24, 629–645. https://doi.org/10.1111/eff.12176.

Alexandre, C.M., Almeida, P.R., Neves, T., Mateus, C.S., Costa, J.L., Quintella, B.R., 2016. Effects of flow regulation on the movement patterns and habitat use of a potamodromous cyprinid species. *Ecohydrology* 9, 326–340.

Auer, S., Fohler, N., Zeiringer, B., Führer, S., 2014. *Experimentelle Untersuchungen zur Schwallproblematik. Drift und Stranden von Äschen und Bachforellen während der ersten Lebensstadien.*

Auer, S., Zeiringer, B., Fuhrer, S., Tonolla, D., Schmutz, S., 2017. Effects of river bank heterogeneity and time of day on drift and stranding of juvenile European grayling (*Thymallus thymallus* L.) caused by hydropeaking. *Sci. Total Environ.* 575, 1515–1521. https://doi.org/10.1016/j.scitotenv.2016.10.029.

BAFU – Bundesamt für Umwelt, 2012. *Sanierung Schwall-Sunk, Strategische Planung.* Bundesamt für Umwelt, Bern (128 pp).

BAFU – Bundesamt für Umwelt, 2017. *Schwall-Sunk – Maßnahmen.* Bundesamt für Umwelt, Bern (133 pp).

Bakken, T.H., Zinke, P., Melcher, A., Sundt, H., Vehanen, T., Jorde, K., Acreman, M., 2012. *Setting environmental flows in regulated rivers. Implementing the EU Water Framework Directive (EU WFD) in Norway.* SINTEF Energy Research – TR A7246. Available at. https://www.hydroreform.org/bibliographic-reference/setting-environmental-flows-regulated-rivers.

Bakken, T.H., Forseth, T., Harby, A., 2016a. Simulation of river water temperatures during various hydropeaking regimes. *J. Appl. Water Eng. Res.* 4. https://doi.org/10.1080/23249676.2016.1181578.

Bakken, T.H., Forseth, T., Harby, A., 2016b. *Miljøvirkninger av Effektkjøring: Kunnskapsstatus og råd til Forvaltning og Industri. 62.* NINA Temahefte.

Becker, C.D., Neitzel, D.A., 1985. Assessment of intergravel conditions influencing egg and alevin survival during salmonid redd dewatering. *Environ. Biol. Fish* 12, 33–46.

Becker, C.D., Neitzel, D.A., Fickeisen, D.H., 1982. Effects of dewatering on Chinook salmon redds: tolerance of four developmental phases to daily dewaterings. *Trans. Am. Fish. Soc.* 111, 624–637.

Becker, C.D., Neitzel, D.A., Abernathy, C.S., 1983. Effects of dewatering on Chinook salmon redds: tolerance of four developmental phases to one-time dewatering. *North Am. J. Fish. Manag.* 3, 373–382.

Beeman, J.W., Maule, A.G., 2006. Migration depths of juvenile Chinook salmon and steelhead relative to total dissolved gas supersaturation in a Columbia River reservoir. *Trans. Am. Fish. Soc.* 135, 584–594. https://doi.org/10.1577/T05-193.1.

Bejarano, M.D., Jansson, R., Nilsson, C., 2017a. The effects of hydropeaking on riverine plants: a review. *Biol. Rev.* 93, 658–673. https://doi.org/10.1111/brv.12362.

Bejarano, M.D., Sordo-Ward, Á., Alonso, C., Nilsson, C., 2017b. Characterizing effects of hydropower plants on sub-daily flow regimes. *J. Hydrol.* 550, 186–200. https://doi.org/10.1016/j.jhydrol.2017.04.023.

Benitez, J., Ovidio, M., 2017. The influence of environmental factors on the upstream movements of rheophilic cyprinids according to their position in a river basin. *Ecol. Freshw. Fish* 0, 1–12. https://doi.org/10.1111/eff.12382.

Berland, G., Nickelsen, T., Heggenes, J., Kland, F., Thorstad, E.B., Halleraker, J.H., 2004. Movements of wild Atlantic salmon parr in relation to peaking flows below a hydropower station. *River Res. Appl.* 20, 957–966. https://doi.org/10.1002/rra.802.

BMLFUW, 2015. *Leitfaden zur hydromorphologischen Zustandserhebung von Fließgewässern.* Bundesministerium für Land- und Forstwirtschaft. Umwelt und Wasserwirtschaft, Vienna.

Boavida, I., Santos, J.M., Ferreira, T., Pinheiro, A., 2015. Barbel habitat alterations due to hydropeaking. *J. Hydro Environ. Res.*, 237–247 https://doi.org/10.1016/j.jher.2014.07.009.

Boavida, I., Harby, A., Clarke, K.D., Heggenes, J., 2017.Move or stay: habitat use and movements by Atlantic salmon parr (*Salmo salar*) during induced rapid flow variations. *Hydrobiologia* 785, 261–275. https://doi.org/10.1007/s10750-016-2931-3.

Bondar-Kunze, E., Maier, S., Schönauer, D., Bahl, N., Hein, T., 2016. Antagonistic and synergistic effects on a stream periphyton community under the influence of pulsed flow velocity increase and nutrient enrichment. *Sci. Total Environ.* 573, 594–602. https://doi.org/10.1016/j.scitotenv.2016.08.158.

Bradford, M.J., 1997. An experimental study of stranding of juvenile salmonids on gravel bars and in sidechannels during rapid flow decreases. *Regul. Rivers Res. Manag.* 13, 395–401.

Bradford, M.J., Taylor, G.C., Allan, J.A., Higgins, P.S., 1995. An experimental study of the stranding of juvenile coho salmon and rainbow trout during rapid flow decreases under winter conditions. *N. Am. J. Fish Manag.* 15 (2), 473–479.

Bruder, A., Tonolla, D., Schweizer, S.P., Vollenweider, S., Langhans, S.D.,Wüest, A., 2016. A conceptual framework for hydropeaking mitigation. *Sci. Total Environ.* 568, 1204–1212. https://doi.org/10.1016/j.scitotenv.2016.05.032.

Bruno, M.C., Siviglia, A., 2012. Assessing impacts of dam operations – interdisciplinary approaches for sustainable regulated river management. *River Res. Appl.*, 675–677 https://doi.org/10.1002/rra.

Bruno, M.C., Maiolini, B., Carolli, M., Silveri, L., 2010. Short time-scale impacts of hydropeaking on benthic invertebrates in an Alpine stream (Trentino, Italy). *Limnologica* 40, 281–290. https://doi.org/10.1016/j.limno.2009.11.012.

Canadian Ministry of Justice, 1985. *Fisheries Act.* https://laws-lois.justice.gc.ca/eng/acts/f-14/.

Canadian Council of Ministers of the Environment, 1999. *Canadian water quality guidelines for the protection of aquatic life: dissolved gas supersaturation. Canadian Environmental Quality Guidelines.* Canadian Council of Ministers of the Environment, Winnipeg (Publication No. 1299).

Capra, H., Plichard, L., Berge, J., Pella, H., Ovidio, M., McNeil, E., Lamouroux, N., 2017. Fish habitat selection in a large hydropeaking river: strong individual and temporal variations revealed by telemetry. *Sci. Total Environ.* 578, 109–120. https://doi.org/10.1016/j.scitotenv.2016.10.155.

Capra, H., Pella, H., Ovidio, M., 2018. Individual movements, home ranges and habitat use by native rheophilic cyprinids and non-native catfish in a large regulated river. *Fish. Manag. Ecol.* 25 (2), 136–149. https://doi.org/10.1111/fme.12272.

Carolli, M., Bruno, M.C., Siviglia, A., Maiolini, B., 2012. Responses of benthic invertebrates to abrupt changes of temperature in flume simulations. *River Res. Appl.*, 678–691 https://doi.org/10.1002/rra.1520.

Casas-Mulet, R., Saltveit, S.J., Alfredsen, K., 2014. The survival of Atlantic salmon (*Salmo salar*) eggs during dewatering in a river subjected to hydropeaking. *River Res. Appl.* 31, 433–446. https://doi.org/10.1002/rra.2827.

Casas-Mulet, R., Saltveit, S.J., Alfredsen, K.T., 2016. Effects of hydropeaking on salmonid gravel stages: a modeling approach for implementing mitigation strategies. *Sci. Total Environ.* 573, 1660–1672. https://doi.org/10.1016/j.scitotenv.2016.09.208.

Cazeneuve, L., Lagarrigue, T., Lascaux, J.M., 2009. *Etude de L'impact Écologique des Éclusées sur la Rivière Dordogne.* ECOGEA (Rapport final de la phase 2, 50 pp).

Céréghino, R., Cugny, P., Lavandier, P., 2002. Influence of intermittent hydropeaking on the longitudinal zonation patterns of benthic invertebrates in a mountain stream. *Int. Rev. Hydrobiol.* 87, 47–60.

Charmasson, J., Zinke, P., 2011. *Mitigation Measures Against Hydropeaking Effects.* 7192. SINTEF (Report TR A).

Chun, S.N., Cocherell, S.a., Cocherell, D.E., Miranda, J.B., Jones, G.J., Graham, J., Klimley, P., Thompson, L.C., Cech, J.J., 2010. Displacement, velocity preference, and substrate use of three native California stream fishes in simulated pulsed flows. *Environ. Biol. Fish* 90, 43–52. https://doi.org/10.1007/s10641-010-9716-8.

Code de l'Environnement, 2000. Available at: https://www.legifrance.gouv.fr/Droitfrancais/Codification/Tables-de-concordance/Code-de-l-environnement.

Confederaciónes Hidrográficas de España, 2008. *Instrucción de Planificación Hidrológica (ARM/2656/2008; 10 Sept. 2008).* Spanish River Basin Management Plans 2015–2021.

Connor, E.J., Pflug, D.E., 2004. Changes in the distribution and density of pink, chum, and Chinook salmon spawning in the upper Skagit River in response to flow management measures. *N. Am. J. Fish Manag.* 24, 835–852.

Costa,M.J., Lennox, R.J., Katopodis, C., Cooke, S.J., 2017. Is there evidence for flow variability as an organism-level stressor in fluvial fish? *J. Ecohydraul.* 2 (1), 68–83. https://doi.org/10.1080/24705357.2017.1287531.

Costa, M., Boavida, I., Almeida, V., Cooke, S., Pinheiro, A., 2018. Do artificial velocity refuges mitigate the physiological and behavioural consequences of hydropeaking on a freshwater Iberian cyprinid? *Ecohydrology* https://doi.org/10.1002/eco.1983.

Cushman, R.M., 1985. Review of ecological effects of rapidly varying flows downstream from hydroelectric facilities. *North Am. J. Fish. Manag.* 5, 330–339.

EC (European Commission), 2003. *Common Implementation Strategy for the Water Framework Directive (2000/60/EC). Guidance Document No. 4, Identification and Designation of Heavily Modified and Artificial Water Bodies.* European Union 92-894-5124-6.

EC (European Commission), 2015. *Ecological Flows in the Implementation of the Water Framework Directive. Guidance Document No. 31.* European Union 978-92-79-45758-6.

Enders, E.C., Watkinson, D.A., Ghamry, H., Mills, K.H., Franzin, W.G., 2017. Fish age and size distributions and species composition in a large, hydropeaking Prairie River. *River Res. Appl.* 33, 1246–1256. https://doi.org/10.1002/rra.3173.

ESA – Endangered Species Act of 1973. 1973. Department of the Interior, U.S. Fish and Wildlife Service. Washington, D.C. 20240. Available at: http://www.nmfs.noaa.gov/pr/pdfs/laws/esa.pdf

Federal Power Act – FPA, 1920. *U.S. code, federal regulation and development of power. Chapter 12; Subchapter I, Regulation of the Development of Water Power and Resources.* Available at: https://www.fws.gov/laws/lawsdigest/FEDPOWR.HTML.

Federal Water Pollution Control Act, 2002. *United States Environmental Protection Agency.* Available at: https://www.epa.gov/sites/production/files/2017-08/documents/federal-water-pollution-control-act-508full.pdf.

Finnish Ministry of Environment, 2011. *The Water Act.* Available at: https://www.finlex.fi/en/laki/kaannokset/2011/20110587.

Fisk, J.M., Kwak, T.J., Heise, R.J., Sessions, F.W., 2013. Redd dewatering effects on hatching and larval survival of the robust redhorse. *River Res. Appl.* 29, 574–581. https://doi.org/10.1002/rra.2561.

Flodmark, L.E.W., Urke, H.A., Halleraker, J.H., Arnekleiv, J.V., Vollestad, L.A., Poléo, A.B.S., 2002. Cortisol and glucose responses in juvenile brown trout subjected to a fluctuating flow regime in an artificial stream. *J. Fish Biol.* 60 (1), 238–248. https://doi.org/10.1006/jfbi.2001.1845.

Flodmark, L.E.W., Forseth, T., L'Abee-Lund, J.H., Vollestad, L.A., 2006. Behaviour and growth of juvenile brown trout exposed to fluctuating flow. *Ecol. Freshw. Fish* 15, 57–65. https://doi.org/10.1111/j.1600-0633.2006.00127.x.

García, A., Jorde, K., Habit, E., Caamaño, D., Parra, O., 2011. Downstream environmental effects of dam operations: changes in habitat quality for native fish species. *River Res. Appl.* 27, 312–327. https://doi.org/10.1002/rra.1358.

Greimel, F., Zeiringer, B., Höller, N., Grün, B., Godina, R., Schmutz, S., 2016. A method to detect and characterize sub-daily flow fluctuations. *Hydrol. Process.* 30 (13), 2063–2078. https://doi.org/10.1002/hyp.10773.

Greimel, F., Neubarth, J., Fuhrmann, M., Führer, S., Habersack, H., Haslauer, M., Zeiringer, B., 2017. *SuREmMa – Sustainable River Management – Energiewirtschaftliche und umweltrelevante Bewertung möglicher schwalldämpfender Maßnahmen.* IHG BOKU, Wien.

Greimel, F., Schülting, L., Wolfram, G., Bondar-Kunze, E., Auer, S., Zeiringer, B., Hauer, C., 2018a. Hydropeaking impacts and mitigation. In: Schmutz, S., Sendzimir, J. (Eds.), *Riverine Ecosystem Management.* Aquatic Ecology Series. vol. 8. Springer, Cham.

Greimel, F., Neubarth, J., Zeiringer, B., Hayes, D.S., Haslauer, M., Führer, S., Auer, S., Höller, N., Hauer, C., Holzapfel, P., Fuhrmann, M., Pfleger, M., Matt, P., Koller-Kreimel, V., Schmutz, S., 2018b. Sustainable river management in Austria. *Proceedings of the 12th International Symposium on Ecohydraulics*, Tokyo, Japan.

GSchG, 1991. *Gewässerschutzgesetz Schweiz From 24 January 1991*, Status as of 1 January 2017.

GSchG, 2003. *Gewässerschutzgesetz Liechtenstein From 15 May 2003*, Status as of 1 Oct. 2014.

GSchV, 1998. *Gewässerschutzverordnung Schweiz From 28 October 1998*, Status as of 1 May 2017.

Halleraker, J.H., Saltveit, S.J., Harby, A., Arnekleiv, J.V., Fjeldstad, H.-P., Kohler, B., 2003. Factors influencing stranding of wild juvenile brown trout (*Salmo trutta*) during rapid and frequent flow decreases in an artificial stream. *River Res. Appl.* 19, 589–603. https://doi.org/10.1002/rra.752.

Halleraker, J.H., Sundt, H., Alfredsen, K.T., Dangelmaier, G., 2007. Application of multiscale environmental flow methodologies as tools for optimized management of a Norwegian regulates salmon watercourse. *River Res. Appl.* 23, 493–510.

Halleraker, J.H., van de Bund, W., Bussettini, M., Gosling, R., Döbbelt-Grüne, S., Hensman, J., Kling, J., Koller-Kreimel, V., Pollard, P., 2016. *Working Group ECOSTAT Report on Common Understanding of Using Mitigation Measures for Reaching Good Ecological Potential for Heavily Modified Water Bodies – Part 1: Impacted by Water Storage.* https://doi.org/10.2760/649695.

Hamilton, R., Buell, J., 1976. *Effects of Modified Hydrology on Campbell River Salmonids.* Tech Rep, Pac/T-76-20, Vancouver.

Harby, A., Noack, M., 2013. Rapid flow fluctuations and impacts on fish and the aquatic ecosystem. In: Maddock, I., Harby, A., Kemp, P., Wood, P. (Eds.), *Ecohydraulics*. Wiley Blackwell, p. 446.

Harnish, R.A., Sharma, R., McMichael, G.A., Langshaw, R.B., Pearsons, T.N., Hilborn, R., 2014. Effect of hydroelectric dam operations on the freshwater productivity of a Columbia River fall Chinook salmon population. *Can. J. Fish. Aquat. Sci.* 71 (4), 602–615. https://doi.org/10.1139/cjfas-2013-0276.

Hauer, C., Schober, B., Habersack, H., 2013. Impact analysis of river morphology and roughness variability on hydropeaking based on numerical modeling. *Hydrol. Process.* 27, 2209–2224. https://doi.org/10.1002/hyp.9519.

Hauer, C., Unfer, G., Holzapfel, P., Haimann,M., Habersack, H., 2014. Impact of channel bar form and grain size variability on estimated stranding risk of juvenile brown trout during hydropeaking. *Earth Surf. Process. Landf.* 39, 1622–1641.

Hauer, C., Holzapfel, P., Tonolla, D., Habersack, H., 2016. Diskussion hydrologischer, morphologischer und sedimentologischer Kriterien für die implementierung möglicher Schwall-Sunk-Maßnahmen. *WasserWirtschaft* 1, 23–28.

Hauer, C., Holzapfel, P., Leitner, P., Graf, W., 2017. Longitudinal assessment of hydropeaking impacts on various scales for an improved process understanding and the design of mitigation measures. *Sci. Total Environ.* 575, 1503–1514. https://doi.org/10.1016/j.scitotenv.2016.10.031.

Hayes, D.S., Brändle, J.M., Seliger, C., Zeiringer, B., Ferreira, T., Schmutz, S., 2018. Advancing towards functional environmental flows for temperate floodplain rivers. *Sci. Total Environ.* 633, 1089–1104. https://doi.org/10.1016/j.scitotenv.2018.03.221.

Hedger, R.D., Sauterleute, J., Sundt-Hansen, L.E., Forseth, T., Ugedal, O., Diserud, O.H., Bakken, T.H., 2018. Modeling the effect of hydropeaking-induced stranding mortality on Atlantic salmon population abundance. *Ecohydrology* https://doi.org/10.1002/eco.1960.

Heggenes, J., 1988. Effects of short-term flow fluctuations on displacement of, and habitat use by, brown trout in a small stream. *Trans. Am. Fish. Soc.* 117, 336–344.

Hessevik, T., 2002. *Stranding and Behaviour of Wild Juvenile Brown Trout (Salmo trutta) and Atlantic Salmon (Salmo salar) During Rapid Flow Decreases in an Artificial Stream Channel.* Master thesis, Department of Biology and Nature Conservation. The Agricultural University of Norway.

Higgins, P.S., Bradford, M.J., 1996. Evaluation of a large-scale fish salvage to reduce the impacts of controlled flow reduction in a regulated river. *North Am. J. Fish. Manag.* 16, 666–673. https://doi.org/10.1577/1548-8675(1996)016b0666:EOALSFN2.3.CO;2.

Hunter, M., 1992. *Hydropower Flow Fluctuations and Salmonids: A Review of the Biological Effects, Mechanical Causes and Options for Mitigation. Report no. 119.* Department of Fisheries, State of Washington.

Irvine, J.R., 1986. Effects of varying discharge on the downstream movement of salmon fry, Oncorhynchus tshawytscha Walbaum. *J. Fish Biol.* 28, 17–28.

Irvine, R.L., Oussoren, T., Baxter, J.S., Schmidt, D.C., 2009. The effects of flow reduction rates on fish stranding in British Columbia, Canada. *River Res. Appl.* 25, 405–415. https://doi.org/10.1002/rra.

Jensen, A.J., Johnsen, B.O., 1999. The functional relationship between peak spring floods and survival and growth of juvenile Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*). *Funct. Ecol.* 13, 778–785.

Jensen, J.O.T., Schnute, J., Alderdice, D.F., 1986. Assessing juvenile salmonid response to gas supersaturation using a general multivariate dose-response model. *Can. J. Fish. Aquat. Sci.* 43, 1694–1709. https://doi.org/10.1139/f86-213.

Jones, N.E., 2014. The dual nature of hydropeaking rivers: is ecopeaking possible? *River Res. Appl.* 30, 521–526. https://doi.org/ 10.1002/rra.2653.

Jones, N.E., Petreman, I.C., 2015. Environmental influences on fish migration in a hydropeaking River. *River Res. Appl.* 31, 1109–1118. https://doi.org/10.1002/rra.2810.

Katzman, S.M., Greathouse, J., Roessig, J.M., Graham, J., Cocherell, D.E., Cech, J.J., 2010. Water velocity preferences of Coho Salmon during the parr-smolt transformation. *Environ. Biol. Fish* 88, 79–84. https://doi.org/10.1007/s10641-010-9619-8.

Kelly, B., Smokorowski, K.E., Power,M., 2017. Impact of river regulation and hydropeaking on the growth, condition and field metabolism of brook trout (*Salvelinus fontinalis*). *Ecol. Freshw. Fish* https://doi.org/10.1111/eff.12310.

Korman, J., Campana, S.E., 2009. Effects of hydropeaking on nearshore habitat use and growth of age-0 rainbow trout in a large regulated river. *Trans. Am. Fish. Soc.* 138, 76–87.

L'Abée-Lund, J.H., Otero, J., 2018. Hydropeaking in small hydropower in Norway—compliance with license conditions? *River Res. Appl.* 2018, 1–10. https://doi.org/10.1002/rra.3258.

Leitner, P., Hauer, C., Graf, W., 2017. Habitat use and tolerance levels of macroinvertebrates concerning hydraulic stress in hydropeaking rivers – a case study at the Ziller River in Austria. *Sci. Total Environ.* 575, 112–118. https://doi.org/10.1016/j.scitotenv.2016.10.011.

Lumsdon, A.E., Artamonov, I., Bruno, M.C., Righetti, M., Tockner, K., Tonolla, D., Zarfl, C., 2018. Soundpeaking – hydropeaking induced changes in river soundscapes. *River Res. Appl.* 34, 3–12. https://doi.org/10.1002/rra.3229.

McMichael, G.A., McKinstry, C.A., Vucelick, J.A., Lukas, J.A., 2005. Fall Chinook salmon spawning activity versus daylight and flow in the tailrace of a large hydroelectric dam. *N. Am. J. Fish Manag.* 25, 573–580.

Miljøverndepartementet, 2006. *Forskrift om rammer for vannforvaltningen: FOR 2006-12-15 nr. 1446.* Miljøverndepartementet, Oslo.

Monk, C., 1989. *Factors that influence stranding of juvenile Chinook salmon and steelhead trout.* Master thesis, University of Washington, Seattle.

Moog, O., 1993. Quantification of daily peak hydropower effects on aquatic fauna and management to minimize environmental impacts. *Regul. Rivers Res. Manag.* 8, 5–14.

Nagrodski, A., Raby, G.D., Hasler, C.T., Taylor, M.K., Cooke, S.J., 2012. Fish stranding in freshwater systems: sources, consequences, and mitigation. *J. Environ. Manag.* 103, 133–141. https://doi.org/10.1016/j.jenvman.2012.03.007.

Niu, S., Insley, M., 2013. On the economics of ramping rate restrictions at hydro power plants: balancing profitability and environmental costs. *Energy Econ.* 39–52 https://doi.org/10.1016/j.eneco.2013.04.002.

Olson, F., 1990. *Downramping Regime for Power Operations to Minimize Stranding of Salmon Fry in Sultan River.* Report by CH2M Hill for Snohomish County PUD 1, Bellevue.

Parasiewicz, P., Schmutz, S., Moog, O., 1998. The effects of managed hydropower peaking on the physical habitat, benthos and fish fauna in the river Bregenzerach in Austria. *Fish. Manag. Ecol.* 5, 403–417.

Pérez-Díaz, J.I.,Wilhelmi, J.R., 2010. Assessment of the economic impact of environmental constraints on short-term-hydropower plant operation. *Energy Policy* 38, 7960–7970. https://doi.org/10.1016/j.enpol.2010.09.020.

Person, E., Bieri, M., Peter, A., Schleiss, A.J., 2014. Mitigation measures for fish habitat improvement in Alpine rivers affected by hydropower operations. *Ecohydrology* 7 (2), 580–599. https://doi.org/10.1002/eco.1380.

Pragana, I., Boavida, I., Cortes, R., Pinheiro, A., 2017. Hydropower plant operation scenarios to improve Brown trout habitat. *River Res. Appl.* 33, 364–376. https://doi.org/10.1002/rra.3102.

Premstaller, G., Cavedon, V., Pisaturo, G.R., Schweizer, S., Adami, V., Righetti, M., 2017. Hydropeaking mitigation project on a multi-purpose hydro-scheme on Valsura River in South Tyrol/Italy. *Sci. Total Environ.* 574, 642–653. https://doi.org/10.1016/j.scitotenv.2016.09.088.

Puffer, M., Berg, O.K., Einum, S., Saltveit, S.J., Forseth, T., 2017. Energetic consequences of stranding of juvenile Atlantic salmon (*Salmo salar* L.). *J. Water Resour. Prot.* 9, 163–182. https://doi.org/10.4236/jwarp.2017.92012.

Pulg, U., Vollset, K.W., Velle, G., Stranzl, S., 2016. First observations of saturopeaking: characteristics and implications. *Sci. Total Environ.* 573, 1615–1621. https://doi.org/10.1016/j.scitotenv.2016.09.143.

QZVÖ, 2010. *Qualitätszielverordnung Ökologie.* BGBl. II Nr. 99/2010, Vienna, Austria.

Reindl, R., Egger, K., Fitzka, G., 2017. Der Wasserwirtschaftliche Rahmenplan Tiroler Oberland. *WasserWirtschaft* (7–8), 69–74.

Saltveit, S.J., Halleraker, J.H., Arnekleiv, J.V., Harby, A., 2001. Field experiments on stranding in juvenile Atlantic salmon (*Salmo salar*) and brown trout (*Salmo trutta*) during rapid flow decreases caused by hydropeaking. *Regul. Rivers Res. Manag.*,609–622 https://doi.org/10.1002/rrr.652.

Santos, J.M., Branco, P., Katopodis, C., Ferreira, T., Pinheiro, A., 2014. Retrofitting pool-and weir fishways to improve passage performance of benthic fishes: effect of boulder density and fishway discharge. *Ecol. Eng.* 73, 335–344.

Sauterleute, J., Hedger, R.D., Hauer, C., Pulg, U., Skoglund, H., Sundt-Hansen, L.E., Bakken, T.H., Ugedal, O., 2016. Modeling the effects of stranding on the Atlantic salmon population in the Dale River, Norway. *Sci. Total Environ.*, 574–584 https://doi.org/10.1016/j.scitotenv.2016.08.080.

Schmutz, S., Fohler, N., Friedrich, T., Fuhrmann, M., Graf, W., Greimel, F., Höller, N., Jungwirth, M., Leitner, P., Moog, O., Melcher, A., Müllner, K., Ochsenhofer, G., Salcher, G., Steidl, C., Unfer, G., Zeiringer, B., 2013. *Schwallproblematik an Österreichs Fließgewässern – Ökologische Folgen und Sanierungsmöglichkeiten.* BMFLUW,Wien.

Schmutz, S., Bakken, T.H., Friedrich, T., Greimel, F., Harby, A., Jungwirth, M., Melcher, A., Unfer, G., Zeiringer, B., 2015. Response of fish communities to hydrological and morphological alterations in hydropeaking Rivers of Austria. *River Res. Appl.* 31, 919–930. https://doi.org/10.1002/rra.2795.

Schülting, L., Feld, C.K., Graf, W., 2016. Effects of hydro- and thermopeaking on benthic macroinvertebrate drift. *Sci. Total Environ.* 573, 1472–1480. https://doi.org/10.1016/j.scitotenv.2016.08.022.

Schweizer, S., Schmidlin, S., Bieri, M., Büsser, P., Meyer, M., Money, J., Schläppi, S., Schneider, M., Tonolla, D., Tuhtan, J., Wächter, K., 2016. Die erste Schwall-Sanierung der Schweiz: die Hasliaare als Fallbeispiel. *Wasser-Wirtschaft* 1, 10–15.

Scruton, D.A., Ollerhead, L.M.N., Clarke, K.D., Pennell, C.J., Alfredsen, K., Harby, A., Kelley, D., 2003. The behavioural response of juvenile Atlantic salmon (*Salmo salar*) and brook trout (*Salvelinus fontinalis*) to experimental hydropeaking on a Newfoundland (Canada) river. *River Res. Appl.* 19, 577–587. https://doi.org/10.1002/rra.733.

Scruton, D.A., Pennell, C.J., Robertson, M.J., Ollerhead, L.M.N., Clarke, K.D., Alfredsen, K., Harby, A., McKinley, R.S., 2005. Seasonal response of juvenile Atlantic salmon to experimental hydropeaking power generation in Newfoundland, Canada. *North Am. J. Fish. Manag.* 25, 964–974. https://doi.org/10.1577/M04-133.1.

Scruton, D.A., Pennell, C.J., Ollerhead, L.M.N., Alfredsen, K., Stickler, M., Harby, A., Robertson, M.J., Clarke, K.D., LeDrew, L.J., 2008. A synopsis of "hydropeaking" studies on the response of juvenile Atlantic salmon to experimental flow alteration. *Hydrobiologia* 609, 263–275. https://doi.org/10.1007/s10750-008-9409-x.

SEPA – Swedish Environmental Protection Agency, 2017. *Swedish Environmental Law –An Introduction to the Swedish Legal System for Environmental Protection (Report 6790).* (38 pp). Naturvårdsverket, Stockholm, Sweden.

Shuster, W.D., Zhang, Y., Roy, A.H., Daniel, F.B., Troyer, M., 2008. Characterizing storm hydrograph rise and fall dynamics with stream stage data. *J. Am. Water Resour. Assoc.* 44, 1431–1440. https://doi.org/10.1111/j.1752-1688.2008.00249.x.

Stickler,M., Alfredsen, K., Scruton, D.A., Pennell, C., Harby, A., Økland, F., 2007. Mid-winter activity and movement of Atlantic Salmon Parr during ice formation events in a Norwegian regulated river. *Hydrobiologia* 582, 81–89.

Taylor, M.K., Cook, K.V., Hasler, C.T., Schmidt, D.C., Cooke, S.J., 2012. Behaviour and physiology of mountain whitefish (*Prosopium williamsoni*) relative to short-term changes in river flow. *Ecol. Freshw. Fish* 21, 609–616.

Thompson, L.C., Cocherell, S.A., Chun, S.N., Cech, J.J., Klimley, A.P., 2011. Longitudinal movement of fish in response to a single-day flow pulse. *Environ. Biol. Fish* 90, 253–261. https://doi.org/10.1007/s10641-010-9738-2.

Tonolla, D., Bruder, A., Schweizer, S., 2017. Evaluation of mitigation measures to reduce hydropeaking impacts on river ecosystems – a case study from the Swiss alps. *Sci. Total Environ.* 574, 594–604. https://doi.org/10.1016/j.scitotenv.2016.09.101.

Tuhtan, J.A., Noack, M., Wieprecht, S., 2012. Estimating stranding risk due to hydropeaking for juvenile European grayling considering river morphology. *KSCE J. Civ. Eng.* 16, 197–206.

Uiblein, F., Jagsch, A., Honsig-Erlenburg, W., Weiss, S., 2001. Status, habitat use, and vulnerability of the European grayling in Austrian waters. *J. Fish Biol.* 59, 223–247.

Unfer, G., Leitner, P., Graf, W., Auer, S., 2011. *Der Einfluss von Schwallbetrieb auf den Fischbestand der Oberen Drau.* VERBUND – Austrian Hydro Power AG; Bundesministerium für Land- und Forstwirtschaft. Umwelt und Wasserwirtschaft, Vienna.

Vanzo, D., Zolezzi, G., Siviglia, A., 2015. Eco-hydraulic modeling of the interactions between hydropeaking and river morphology. *Ecohydrology* 9 (3), 421–437. https://doi.org/10.1002/eco.1647.

Vocht, A.D., Baras, E., 2005. Effect of hydropeaking on migrations and home range of adult Barbel (Barbus barbus) in the river Meuse. *Proc. Fifth Conf. Fish Telem.* 9–13.

Wasserwirtschaftlicher Rahmenplan Tiroler Oberland, 2014. *BGBl. II Nr. 274/2014.*

Weitkamp, D.E., 2008. *Total Dissolved Gas Supersaturation Biological Effects, Review of Literature 1980–2007.* Parametrix, Bellevue, WA.

WG – Wassergesetz für Baden-Württemberg, 2013. *Wassergesetz für Baden-Württemberg. 2013.* GBl, p. 389.

WNP – Wassernutzungsplan, 2017. *Gesamtplan für die Nutzung der öffentlichen Gewässer.* Paritätischer Ausschuss (DPR Nr. 381/74, Art. 8), endgültig beschlossen am 01. März 2017. Autonome Provinz Bozen, Südtirol.

Young, P.S., Cech, J.J., Thompson, L.C., 2011. Hydropower-related pulsed-flow impacts on stream fishes: a brief review, conceptual model, knowledge gaps, and research needs. *Rev. Fish Biol. Fish.* 21, 713–731. https://doi.org/10.1007/s11160-011-9211-0.

Zeiringer, B., Fohler, N., Auer, S., Greimel, F., Schmutz, S., 2014. Experiments on drifting and stranding of juvenile grayling during fluctuating flow in nature like channels with different morphological structures. *Proceedings of the 10th International Symposium on Ecohydraulics*, Trondheim, Norway, June 23–27.

Zolezzi, G., Siviglia, A., Toffolon, M., Maiolini, B., 2011. Thermopeaking in Alpine streams: event characterization and time scales. *Ecohydrology* 4, 564–576.