Science of the Total Environment xxx (2016) xxx–xxx

Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

A conceptual framework for hydropeaking mitigation

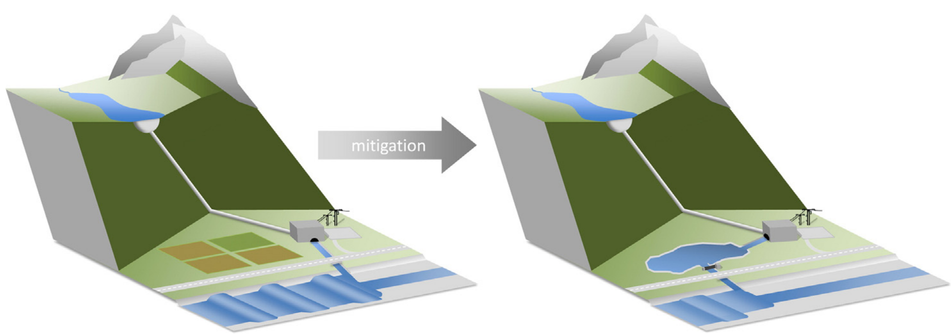
Andreas Bruder [a,b,c,*], Diego Tonolla [d], Steffen P. Schweizer [e], Stefan Vollenweider [c], Simone D. Langhans [f], Alfred Wüest [a,g]

[a] Eawag, Swiss Federal Institute of Aquatic Science, Surface Waters – Research and Management, Seestrasse 79, 6047 Kastanienbaum, Switzerland
[b] University of Applied Sciences and Arts of Southern Switzerland, Institute of Earth Sciences, Campus Trevano, 6952 Canobbio, Switzerland
[c] Wasser-Agenda 21, Überlandstrasse 133, 8600 Dübendorf, Switzerland
[d] Zurich University of Applied Sciences, Institute of Natural Resource Sciences, Grüental, 8820 Wädenswil, Switzerland
[e] Kraftwerke Oberhasli AG, 3862 Innertkirchen, Switzerland
[f] Leibniz-Institute of Freshwater Ecology and Inland Fisheries, Müggelseedamm 310, 12587 Berlin, Germany
[g] Physics of Aquatic Systems Laboratory – Margaretha Kamprad Chair, EPFL-ENAC-IIE-APHYS, 1015 Lausanne, Switzerland

HIGHLIGHTS

- High-head storage hydropower plants can cause hydropeaking
- Hydropeaking affects the biota and functioning of river ecosystems
- Hydropeaking mitigation can be achieved with structural and operational measures
- We developed a framework to predict the consequences of mitigation on river ecosystems
- Mitigation measures require interdisciplinary assessments in an integrative process

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 12 March 2016
Received in revised form 1 May 2016
Accepted 3 May 2016
Available online xxxx

Editor: D. Barcelo

Keywords:
Discharge fluctuations
Ecological indicators
Functional indicators
Hydroelectricity
Multiple stressors
Prediction tools
River restoration

ABSTRACT

Hydropower plants are an important source of renewable energy. In the near future, high-head storage hydropower plants will gain further importance as a key element of large-scale electricity production systems. However, these power plants can cause hydropeaking which is characterized by intense unnatural discharge fluctuations in downstream river reaches. Consequences on environmental conditions in these sections are diverse and include changes to the hydrology, hydraulics and sediment regime on very short time scales. These altered conditions affect river ecosystems and biota, for instance due to drift and stranding of fishes and invertebrates. Several structural and operational measures exist to mitigate hydropeaking and the adverse effects on ecosystems, but estimating and predicting their ecological benefit remains challenging. We developed a conceptual framework to support the ecological evaluation of hydropeaking mitigation measures based on current mitigation projects in Switzerland and the scientific literature. We refined this framework with an international panel of hydropeaking experts. The framework is based on a set of indicators, which covers all hydrological phases of hydropeaking and the most important affected abiotic and biotic processes. Effects of mitigation measures on these indicators can be predicted quantitatively using prediction tools such as discharge scenarios and numerical habitat models. Our framework allows a comparison of hydropeaking effects among alternative mitigation measures, to the pre-mitigation situation, and to reference river sections. We further identified key issues

* Corresponding author at: University of Applied Sciences and Arts of Southern Switzerland, Institute of Earth Sciences, Campus Trevano, 6952 Canobbio, Switzerland Eawag, Swiss Federal Institute of Aquatic Science, Surface Waters – Research and Management, Seestrasse 79, Kastanienbaum, 6047, Switzerland.
E-mail address: andreas.bruder@supsi.ch (A. Bruder).

http://dx.doi.org/10.1016/j.scitotenv.2016.05.032

that should be addressed to increase the efficiency of current and future projects. They include the spatial and temporal context of mitigation projects, the interactions of river morphology with hydropeaking effects, and the role of appropriate monitoring to evaluate the success of mitigation projects.

1. Introduction

High-head storage hydropower plants constitute a significant portion of the electricity industry in many mountainous regions of the world. These hydropower plants are an important source of renewable energy that can produce electricity more efficiently and with shorter response time than most other types of power plants. This practice, characterized by intermittent production, is crucial to respond to short-term changes in electricity demands, for instance as a consequence of stochastic electricity generation from solar and wind energy (Haas et al., 2015). Furthermore, these hydropower plants are able to store great volumes of water as potential energy – in particular for wintertime, when higher electricity demands coincide with lower rainfall. Besides these advantages, storage hydropower plants have detrimental impacts on downstream river ecosystems. In particular, intermittent electricity production can cause extreme downstream fluctuations in discharge and water levels, so-called hydropeaking (Meile et al., 2011; Moog, 1993; Zimmerman et al., 2010). The ecological consequences of hydropeaking add to other impacts of hydropower installations such as damming, sediment retention, and channelization (Merritt et al., 2010; Olden and Naiman, 2010; Poff et al., 2010).

Fluctuations of discharge and water level due to hydropeaking can greatly exceed those of the natural hydrological regimes repeatedly over the course of a day (Carolli et al., 2015; Jones, 2014). Hydropeaking regimes are characterized by distinct hydrological phases, which often last only for a few hours (Bevelhimer et al., 2015; Meile et al., 2011; Moog, 1993). They include: (i) low base discharge when no electricity is generated, (ii) rapid changes in discharge, when generation is increased or decreased, and (iii) high peak discharge during periods of maximal generation. Hydrological effects trigger changes in sediment dynamics and water temperature leading to multiple co-occurring abiotic stressors (Hering et al., 2015). Because in hydropeaking reaches these effects occur much faster and more frequently than those driven by natural events (Jones, 2014), they may have strong impacts on river ecosystems and their biota (De Jalon et al., 1994; Merritt et al., 2010; Moog, 1993; Smokorowski et al., 2011; Zimmerman et al., 2010).

Rapid increases in flow velocity can cause detachment and drift of benthic algae, fish, and invertebrates (Bruno et al., 2010; Miller and Judson, 2014; Zeiringer et al., 2014; Young et al., 2011). Fast flows during maximal discharge may mobilize sediment (Anselmetti et al., 2007), causing mechanical damage to exposed organisms (Holomuzki and Biggs, 2003; Jones et al., 2012), although beneficial effects might also ensue when high flows reduce clogging of the top layers of river beds (Young et al., 2011). Additionally, high turbidity during maximal discharge may affect foraging fish relying on visual cues and also reduce growth of benthic algae, with potential knock-on effects on higher trophic levels, i.e. invertebrates and fish (Jones et al., 2012). Rapid water level reductions can lead to stranding of fish and invertebrates and to dewatering of spawning grounds (Moog, 1993; Richards et al., 2013; Zeiringer et al., 2014; see review by Young et al., 2011). Abrupt changes in water temperature, as a consequence of contrasting water temperatures between reservoirs and downstream river reaches (Bonalumi et al., 2012; Zolezzi et al., 2011), can have flow-independent impacts on the biota including drift of invertebrates (Carolli et al., 2011). Low discharge and rapid discharge changes may enhance the deposition of suspended fine sediments in dewatered zones and increase clogging (Fette et al., 2007; Hauer et al., 2016a), which may in turn reduce suitable habitat size, oxygen exchange, and connection to the hyporheic zone.

Some animals may avoid hydropeaking effects by moving into refuges (Bruno et al., 2009; Bunt et al., 1999; Heggenes et al., 2013; Ribi et al., 2014). However, the availability of refuges is often reduced in hydropeaking reaches if morphological heterogeneity is also impaired. For instance, juvenile fish select microhabitats with low flow velocities during peak flows such as pools and side channels (Zeiringer et al., 2014; Scruton et al., 2008), but these structures are often lacking in channelized reaches. Furthermore, clogging reduces the interstitial space (Fette et al., 2007), which acts as a refuge for invertebrates and juvenile fish (Bruno et al., 2009; Heggenes et al., 2013). In combination, hydropeaking commonly results in a reduction of biomass at all trophic levels and in altered behaviour, biodiversity, and community composition of river biota (Lagarrigue et al., 2002; Lauters et al., 1996) which consequently affects river ecosystem functioning (Bond and Jones, 2015; Marty et al., 2009; Miller and Judson, 2014; Moog, 1993).

During the last decades, awareness of adverse hydropeaking effects on river ecosystems has increased in research (e.g. EnviPEAK, 2016; Hering et al., 2015; Moog, 1993; Person et al., 2014), management (e.g. Parasiewicz et al., 1998; Person et al., 2014), and legislation (e.g. in the Swiss Water Protection Act (WPA, 2011) and the European Union (WFD, 2000)). Hydropeaking can be mitigated by structural and operational measures (Table 1) (Brunner and Rey, 2014; Moog, 1993; Niu and Insley, 2013; Person et al., 2014). Structural measures commonly include a retention volume to retain the discharge peak and smoothen the release into the downstream river (e.g. basins or caverns at the outlet of the power plant; Bieri et al., 2014). They mainly reduce rates of discharge changes during hydropeaking, but can also achieve a pre-surge before the main discharge peak to allow for behavioural responses of river organisms, and - given a large enough volume - reduce discharge maxima (Meile et al., 2011; Parasiewicz et al., 1998; Person et al., 2014). In some situations, diverting the discharge peak to large downstream waterbodies (i.e. lakes or rivers with a large volume) or into parallel channels of lower ecological sensitivity may also be a realistic option (e.g. Brunner and Rey, 2014). Morphological restoration of hydropeaking reaches might also alleviate some hydropeaking effects (see Subsection 4.2) or can be used as ecological compensation measures in the catchment.

Operational measures on the other hand aim at reducing the hydrological effects of hydropeaking by adapting the production scheme (Person et al., 2014), to (i) reduce discharge extremes through the turbines, (ii) reduce rates of change in discharge, (iii) reduce the frequency of peaks, (iv) create a step-wise discharge increase (i.e. a pre-surge), and/or (v) by anticyclical production of sequential power plants

Table 1
Types of mitigation measures to reduce hydropeaking effects on river ecosystems.

Structural measures	Retention volumes (e.g. basins and caverns)
	Diversion of the discharge peak to a natural retention volume (lake or large river)
	Diversion of the discharge peak into a parallel channel
	Morphological restorations/modifications of the impacted river section
Operational measures	Reduction of the discharge extremes: increasing minimal discharge / decreasing maximal discharge
	Reduction of the rate of change in discharge and water level
	Reduction of the frequency of discharge peaks
	Generation of a pre-surge of low magnitude before the discharge peak
	Anticyclical production of sequential power plants

Please cite this article as: Bruder, A., et al., A conceptual framework for hydropeaking mitigation, Sci Total Environ (2016), http://dx.doi.org/10.1016/j.scitotenv.2016.05.032

(Table 1). Because operational measures affect the total amount and flexibility of electricity production, they may reduce the economic sustainability of the power plant and the stability of the electricity grid (Niu and Insley, 2013; Person et al., 2014). Nevertheless, in some socio-economic settings, operational measures or a combination of structural and operational measures can efficiently reduce hydropeaking effects, without affecting economic sustainability (Fette et al., 2007).

The ecological and socio-economic complexity of hydropeaking mitigation warrants a case-specific quantitative evaluation of mitigation measures based on a conceptual framework. A broad scientific discussion on the conceptual background of hydropeaking mitigation from an ecological standpoint is lacking but is crucial to maximize mitigation efficiency. Our study aims at providing a conceptual framework which combines hydropeaking impact analysis, evaluation of mitigation measures, and monitoring of mitigation success (Fig. 1). We further describe the methodological approaches required for the ecological evaluation of mitigation measures based on their consequences on the ecological conditions in river reaches impacted by hydropeaking. Our suggestions are based on various mitigation projects from Switzerland and on the scientific literature.

2. Material and methods

We scrutinized seven on-going hydropeaking mitigation projects that vary in size and structure of the hydropower scheme and in the approaches chosen to evaluate mitigation measures (Table 2). We interviewed the experts leading the ecological components of the respective projects following a standardized protocol (Supplementary Table 1) and examined the respective project reports. In particular, we assessed the mitigation measures considered and their advantages and limitations. We further assessed the ecological indicators used in the respective projects, the tools used to predict their change for different mitigation measures and the aggregation methods applied to combine results from different indicators. We also examined the socio-economic settings of each project because of its important role in defining the set of realistic mitigation measures. We then complemented the findings from these projects with the scientific literature on hydropeaking effects and on mitigation projects (e.g. in the Bregenzer Ach; Moog, 1993; Parasiewicz et al., 1998).

In a one-day workshop, we evaluated the ecological indicators identified in the assessed mitigation projects and the literature with an international panel of 17 experts on hydropeaking (see Supplementary material). Prior to the workshop, a list of candidate indicators (Supplementary Table 2) was sent to the participants for additions based on their experience. During the workshop, we first presented all indicators and clarified the ecosystem component assessed by each indicator and its metrics. Separate working groups then evaluated each indicator according to (i) its relevance for hydropeaking and (ii) the ability to estimate its change with mitigation measures using available prediction tools. Findings from this evaluation were then discussed with the entire panel and a final set of indicators covering the relevant ecosystem components and hydrological phases was defined (Table 3).

3. Results

3.1. Ecological indicators

Quantitative predictions of the effect of mitigation measures on river ecosystems can be based on abiotic and biotic indicators, which (i) assess river ecosystem characteristics and processes affected by hydropeaking and (ii) whose changes with mitigation can be predicted with existing tools. For a comprehensive evaluation, these indicators should be combined to cover the relevant abiotic and biotic ecosystem components as well as the typical hydrological phases of hydropeaking events (Table 3). Unlike for other steps of mitigation projects, the ability to predict how these indicators change with hydropeaking mitigation is fundamental. As a consequence, several indicators we propose for prediction (Fig. 1, Table 3) differ for methodological reasons from those commonly used for the assessment of current hydropeaking effects ("deficit analysis") and/or for post-mitigation monitoring programs. Predicted effects (and later the outcome of monitoring) can then be compared to mitigation goals that were defined based on a deficit analysis (Fig. 1). Mitigation goals should include key organism groups and ecosystem processes as well as threshold values of acceptable impact given socio-economic constraints (Hauer et al., 2016b; Palmer et al., 2005; Parasiewicz et al., 2013).

In particular, we suggest functional indicators, i.e. ecological processes, for the prediction of biotic ecosystem components. This is unlike the structural indicators more widely used for deficit analysis or for monitoring (i.e. indicator-set A in Fig. 1; Baumann et al., 2012). This is a consequence of many structural biotic indicators being difficult to predict, because they are simultaneously affected by various interacting abiotic and biotic processes. For instance, structural biotic indicators such as invertebrate species richness or fish biomass can be estimated relatively accurately for current conditions but their prediction entails large uncertainty. Thus, predicting key processes affecting these structural biotic indicators (e.g. drift, stranding, and movement of spawning grounds; Table 3) seems to provide a more reliable approach at present.

Hydrological indicators of hydropeaking directly affected by mitigation measures include minimal and maximal discharge, rates of discharge changes during increasing and decreasing discharge as well as the number of peaks per day and their duration (Table 3). These hydrological parameters in turn, affect ecological conditions both at the patch-scale (spatial resolution ≤1 m) and the reach-scale (spatial resolution 10 m to several km) of a hydropeaking reach (Fig. 2; see Gurnell et al., 2016 for an overview of the use of these spatial scales in river ecosystem management). Abiotic conditions control the patch-scale habitat suitability of river organisms through parameters such as water level, flow velocity, sediment grain size, and hydraulic shear stress as well as clogging intensity (Shields et al., 2003). However, at present quantitative predictions of clogging are limited due to insufficient mechanistic

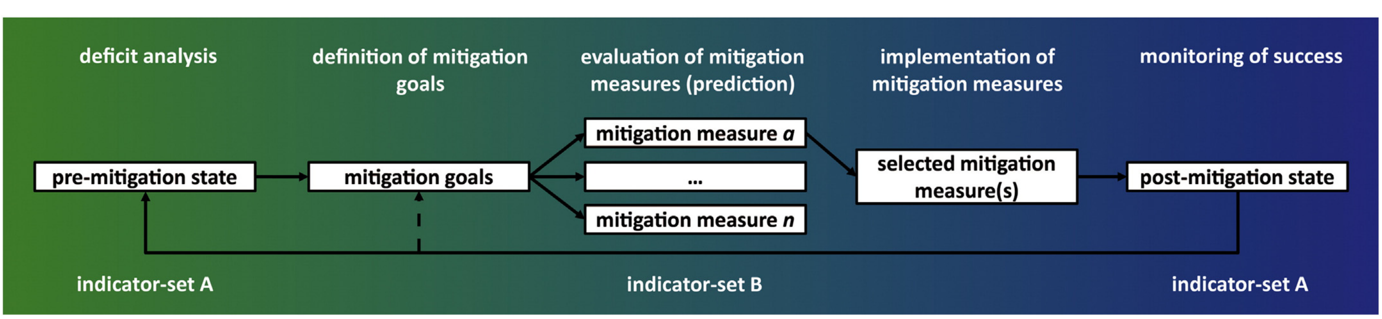


Fig. 1. The phases of planning and implementation of hydropeaking mitigation projects using mitigation measures *a* to *n* (measure *a* was chosen in this example). Prediction of the consequences of mitigation measures is based on a range of ecological indicators (indicator-set B), whereas the deficit analysis and monitoring of success is based on a different set of indicators (indicator-set A). Based on the outcome of the monitoring after implementation, adaptations to the mitigation measures and/or additional mitigation measures might need to be envisaged or the mitigation goals reassessed.

Table 2
Hydropeaking mitigation projects analysed in the present study and the most detrimental hydropeaking effects, the evaluated mitigation measures, and the applied prediction tools of each project.

River	Important hydropeaking effects[a]	Evaluated mitigation measures	Applied prediction tools
Hasliaare (upstream of Lake Brienz)	Macroinvertebrate drift, fish stranding	Retention volumes, morphological restorations, diversion of peak discharge to downstream Lake Brienz	Discharge scenarios[b], physical river model, hydrological models (1D), numerical habitat models (2D)
Linth (upstream of Lake Walen)	Macroinvertebrate drift, fish stranding, sediment transport, movement of spawning grounds	Retention volumes, operational reduction of maximal discharge, morphological restorations, diversion of peak discharge to downstream Lake Walen	Discharge scenarios, hydrological models (1D)
Reuss (upstream of Lake Lucerne)	Macroinvertebrate drift, fish stranding	Retention cavern, morphological restorations	Discharge scenarios, hydrological models (1D)
Ijentalerbach (tributary to River Thur)	Clogging, movement of spawning grounds	Operational improvement of base and peak discharge	Numerical habitat models (2D)
Poschiavino (upstream of Lago di Poschiavo)	Sediment transport, movement of spawning grounds	Diversion of peak discharge to a downstream lake, morphological restorations	Hydrological models for the lakes (3D)
Alpine Rhine (upstream of Lake Constance)	Fish stranding, clogging, movement of spawning grounds	Retention basins, operational measures morphological restorations	Hydrological models (1D), numerical habitat models (2D)
Ticino (upstream of Lago Maggiore)	Macroinvertebrate drift, movement of spawning grounds	Retention basins, operational measures, morphological restorations	Hydrological models (1D)

[a] Selection of indicators was based on expert opinion and thus varied among case studies.
[b] Discharge scenarios include targeted manipulations of discharge to simulate hydropeaking (see text).

knowledge linking clogging intensity to hydraulic conditions and to sediment transport (Hauer et al., 2016a).

Other abiotic indicators are relevant at the reach-scale and include processes of the sediment regime such as erosion, transport, and deposition of sediment (Hauer et al., 2008). Changes in water temperature and turbidity are mainly controlled by changes in the quantity of turbinated water, given that water in reservoirs differs from river water regarding these parameters (Bonalumi et al., 2012). Predictions of these reach-scale indicators can thus be based on estimates of the effects of mitigation measures on the ratio of water from reservoirs vs. upstream river sections. Moreover, assessments of changes in the wetted area at the reach-scale allow estimating the maximal potential habitat in the reach.

For focal organism groups, predictions of habitat availability and suitability should be complemented by prediction of key ecological processes (Table 3). For fish, the respective functional indicators include reach-scale indicators such as disturbance of spawning grounds, drift during high discharges as well as stranding of fishes and dewatering of redds during phases of decreasing and low discharge. For invertebrates, the risks of drift and stranding might be higher compared to fish due to their low mobility, however, some taxa are able to access refugia in the sediment (Bruno et al., 2009). At present, insufficient scientific knowledge on invertebrate stranding results in uncertainty associated with the prediction of this indicator (Table 3).

Table 3
Matrix of indicators to assess the effects of hydropeaking mitigation measures on river ecosystems (represented as indicator-set B in Fig. 1). Dark shading/orange background colour highlights indicators that can be quantitatively predicted with existing prediction tools with high accuracy, whereas prediction of indicators with light shading/yellow background colour is affected by greater uncertainty (see text and Supplementary material Section 2.2.). Q_b and Q_p = base and peak discharge, respectively.

Hydrological phases of hydropeaking

Ecosystem components	Minimal discharge	Increasing discharge	Maximal discharge	Decreasing discharge
Hydrology	Q_b, duration	Rate of increase in discharge and water level	Q_p, $\frac{Q_p}{Q_b}$, duration	Rate of decrease in discharge and water level
Sediment regime	No additional processes relevant	Sediment transport	Sediment transport	Sedimentation, Clogging
Turbidity	Background turbidity of the residual flow	No additional processes relevant	Turbidity with maximal contribution of water from reservoirs	No additional processes relevant
Water temperature	Background temperature of residual flow	Rate of temperature change with increasing contribution of water from reservoirs	Temperature with maximal contribution of water from reservoirs	Rate of temperature change with decreasing contribution of water from reservoirs
Habitat	Minimal wetted area, Habitat suitability[a)]	Habitat suitability[a)]	Maximal wetted area, Habitat suitability[a)]	Habitat suitability[a)]
Fish	Dewatering of redds	Drift	Movement of spawning grounds, drift	Stranding
Invertebrates	No additional processes relevant	Drift	Drift	Stranding

[a] This indicator can be predicted with numerical habitat models which take into account patch-scale habitat conditions including water level, flow velocity, sediment grain size, and hydraulic shear stress (see also Fig. 2).

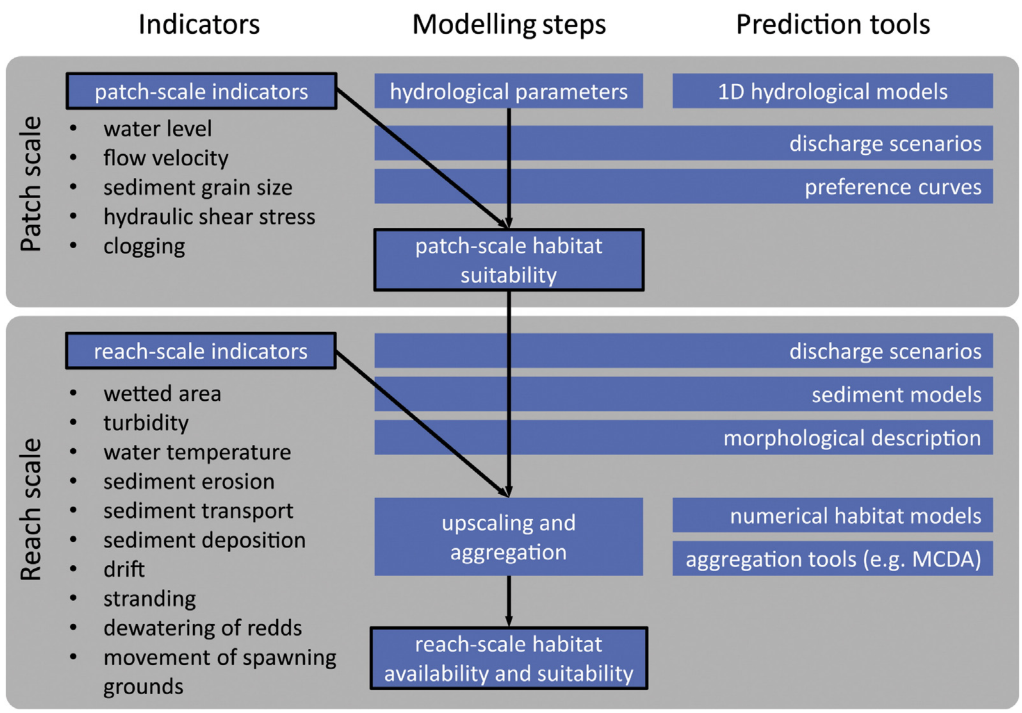


Fig. 2. Scheme of work flow to predict reach-scale habitat availability and suitability for a given organism group and/or life stage (representing the 3rd step in the conceptual framework shown in Fig. 1). The scheme describes the habitat features (i.e. ecological indicators from Table 3; left-hand column), the modelling steps (centre column) and the prediction tools required for these steps or to reach the next step (right-hand column). The hydrological parameters used as input for the work flow are defined by the considered mitigation options and described by respective representative discharge conditions (see text). The reach-scale habitat availability can subsequently be further aggregated for all relevant organism groups and with socio-economic consequences of mitigation measures. MCDA = Multi-Criteria Decision Analysis.

3.2. Tools to predict indicator values for alternative mitigation measures

We suggest a sequential prediction of the effects of mitigation measures on the ecology of hydropeaking reaches (Fig. 2; see also Parasiewicz, 2007 and Casas-Mulet et al., 2014). First, hydrological indicators are predicted for alternative mitigation measures, using for instance one-dimensional hydrological models that reflect the characteristics of mitigation measures, in particular the volume of retention basins or parameters of operational measures, etc. These models can take into account statistical descriptions of hydrological indicators (Table 3), including for instance their 95% percentiles (Bevelhimer et al., 2015). The result of this modelling step, which reflects the combined hydrological consequences of mitigation measures, is henceforth referred to as "representative discharge conditions". Second, representative discharge conditions are used as input parameters for a combination of prediction tools, including discharge scenarios and numerical habitat and sediment models to predict changes in other indicators (Table 3; Hauer et al., 2008; Person et al., 2014; Schweizer et al., 2010; Shields et al., 2003). Representative discharge conditions can also account for the effects of future changes to the production scheme and/or those due to climate change (Bieri et al., 2011; Gaudard et al., 2014).

Discharge scenarios are based on short-term adaptations of the production scheme of hydropower plants, and in turn of the hydrology in the hydropeaking reach to simulate representative discharge conditions (Schweizer et al., 2010). These scenarios allow in-situ measurements of patch-scale and reach-scale indicators (Fig. 2). Patch-scale indicators such as flow velocity, water depth, and grain size distribution can then be combined with information on the specific response of organisms and/or life stages to a range of values in these conditions, e.g. from discharge scenarios or the literature. Correlations between gradients in these abiotic indicators and preference of the biota can be described with statistical approaches including preference curves (Bovee, 1986) or random forest analyses (Vezza et al., 2015) commonly used in habitat models (described below). Some reach-scale indicators can also be predicted with discharge scenarios including changes in wetted area, turbidity, water temperature, the intensity of drift and stranding of invertebrates and fish, as well as occurrences of dewatered redds and interruptions of spawning.

During discharge scenarios, predictions of these indicators and the consequences of mitigation measures in general can be compared among impacted river sections with contrasting hydraulic and morphological conditions and habitat features (Hauer et al., 2016a). However, predictions based on discharge scenarios are limited to current river morphology and are, therefore, unable to account for effects of simultaneous morphological restorations - although some conclusions can be drawn by comparing morphologically distinct river reaches (Hauer et al., 2016b; IRKA, 2012). Another limitation of discharge scenarios includes their inability to simulate mid- and long-term effects of mitigation, such as the recovery of river communities or ecosystem functions. Moreover, because discharge scenarios affect the production scheme of power plants, they are usually limited to short periods and need to be planned and carried out in close collaboration between operators and ecologists.

Numerical habitat models based on correlations between abiotic indicators and preference of the biota can be used to integrate the consequences of patch-scale indicators for key taxa and life stages of the river biota. Several modelling approaches have been applied that account for the multivariate controls of habitat suitability defined by the various patch-scale indictors. They either directly compute multivariate metrics (e.g. random forest models; Vezza et al., 2015) or compute univariate metrics which are combined in subsequent steps (e.g. CASiMiR; García et al., 2011 or MesoHABSIM; Parasiewicz, 2007).

In combination with representative discharge conditions and morphological descriptions (i.e. hydromorphological metrics or maps; e.g. García et al., 2011; Lamouroux et al., 1998) of hydropeaking reaches, numerical habitat models can be used to upscale the respective patch-scale habitat suitability to the reach-scale (Freeman et al., 2001; García et al., 2011; Parasiewicz et al., 1998; Parasiewicz, 2007; Parasiewicz et al., 2013; Person et al., 2014; Tuhtan et al., 2012). Similarly, changes in reach-scale processes of the sediment regime including erosion, transport, and deposition of sediment can be predicted using sediment models (Hauer et al., 2008, Shields et al., 2003) and the results integrated into reach-scale habitat models. Validation of hydromorphological and sediment models can be based on measurements carried out during discharge scenarios. Differences in taxon-specific habitat availability and suitability at the reach-scale can then be compared among different mitigation measures and river reaches (Person et al., 2014; Tuhtan et al., 2012). Metrics for comparisons include for instance the cumulative area available to river organisms, with individual spatial grids represented in the models weighted by their suitability (e.g. weighted usable area; Bovee, 1986; Hauer et al., 2008).

The information content and precision of indicators and prediction tools depend on the accuracy of the input data, including the spatial resolution of the river morphology, the taxonomic resolution of the biotic community, and the preference of taxa to environmental conditions. The balance between measuring effort and model information content should therefore reflect the complexity of the river reach in these aspects. In any case, a great investment into prediction tools is warranted due to the importance of small spatial scales for habitat suitability (Parasiewicz et al., 2013). A detailed discussion of different numerical habitat and sediment models is beyond the scope of this study, but several studies provide useful insights on their approaches and how they can be applied in hydropeaking mitigation projects (Conallin et al., 2010; Dunbar et al., 2012; Shields et al., 2003; Tuhtan et al., 2012; Person et al., 2014).

3.3. Aggregation of indicator values

To select mitigation measures with highest ecological benefit, results from the reach-scale indicators and the respective habitat and sediment models should be aggregated. Similarly, the taxon-specific reach-scale habitat availability and suitability need to be aggregated for all key taxa and for all relevant river reaches. In most analysed mitigation projects, aggregation was based on expert opinion. However, tools exist for a quantitative and objective aggregation of indicator values. Although untested for hydropeaking mitigation, aggregation tools such as Multi-Criteria Decision Analysis (MCDA) frameworks for river management (Reichert et al., 2015; Vučijak et al., 2013) seem suitable for this aggregation step. In MCDA and similar approaches, the values of the individual indicators are standardized using value functions, which link the measured indicator values to a common scale or range of categories representing their ecological consequences. Scales might be defined as values ranging from 0 (worst condition) to 1 (optimal condition; Langhans et al., 2013) or categories ranging from poor to very good (WFD, 2000; Baumann et al., 2012). In a second step, indicator values are aggregated to an overall value for each mitigation measure, which can be used to rank the measures (Langhans et al., 2014; Vučijak et al., 2013).

Aggregation tools also allow comparing outcomes of mitigation measures to various other ecosystem states, including reference sites (Murchie et al., 2008; White and Walker, 1997). In addition to ecological indicators, aggregation should include socio-economic consequences of mitigation measures such as consequences on flood protection measures and energy policy goals (Langhans et al., 2014; Vučijak et al., 2013). Their consideration is crucial in hydropeaking mitigation projects given the size of these projects and the far-reaching consequences for electricity production, landscape protection, land-use, etc. (see Person et al., 2014). As with other aggregation tools, conclusions need to be developed with interdisciplinary expert knowledge of the local ecosystem and socio-economic situation including broad stakeholder participation (Reed, 2008; Reichert et al., 2015).

4. Discussion

4.1. The importance of the spatial and temporal context

The conceptual framework and the set of indicators we suggest to predict the ecological consequences of mitigation measures (Fig. 1; Table 3), is based on diverse mitigation projects and expertise and is thus widely applicable. However, the specific spatial and temporal context of each mitigation project defines the most important hydropeaking effects in the respective case as well as feasible mitigation measures. A great investment in assessing the case-specific context is thus warranted.

The local context includes specific hydrological effects of hydropeaking and of the natural flow and sediment regime (e.g. occurrence and magnitude of floods; Hauer et al., 2016b), the composition of the biota, the morphology of the impacted reach, its three-dimensional connectivity, as well as socio-economic constraints (discussed below). Furthermore, the intensity of hydropeaking varies seasonally as a consequence of temporal changes in electricity demand and natural discharge (Lauters et al., 1996). Similarly, effects of abrupt temperature changes due to differences in water temperature between reservoirs and downstream river reaches often varies seasonally (Zolezzi et al., 2011). Sensitivity of some river organisms may also differ in the course of the year, for instance due to habitat requirements of different life stages (Person et al., 2014; Zeiringer et al., 2014; Valentin et al., 1996). Additionally, sub-daily variation in sensitivity might also be important for some organisms and should be included into the ecological evaluation and development of mitigation measures. In particular, habitat use and in turn habitat suitability may undergo substantial diurnal variation (Davey et al., 2011). For instance, drift and stranding of fishes in response to hydropeaking can be substantially higher at night compared to daytime, due to different habitat use (Auer et al., 2016). Certain mitigation measures can - to some extent - adapt to this temporal variability. These include operational mitigation measures and the management of retention volumes, which allow more intense electricity production during times of low ecological sensitivity (Becker et al., 1982; Person et al., 2014).

Detailed, context-specific information is also crucial for the definition of realistic ecological goals of mitigation projects (Palmer et al., 2005; Trussart et al., 2002) and needs to be assessed during the deficit analysis (Fig. 1). For instance, the population of resident fish species might be most efficiently supported by enhancing processes relevant for their recruitment including small-scale migration, spawning, and survival of juveniles (Table 3; Murchie et al., 2008; Person et al., 2014).

4.2. The role of river morphology

River morphology plays a pivotal two-fold role for reaches impacted by hydropeaking. Morphological heterogeneity is crucial in providing diverse habitat and refuges for biotic communities, but it also interacts with hydrological and hydraulic hydropeaking effects (Casas-Mulet et al., 2014; Hauer et al., 2016a; IRKA, 2012; Vanzo et al., 2016). In many cases, hydropeaking reaches are channelized resulting in low morphological heterogeneity and habitat diversity (Ribi et al., 2014; Zeiringer et al., 2014). In channelized reaches, hydropeaking mitigation may only marginally improve the ecological condition, i.e. a likely outcome for hydropeaking mitigation of the Alpine Rhine (Hauer et al., 2016a) and the Hasliaare (Person et al., 2014; Supplementary Table 1).

Observations from rivers on the role of morphology are supported by experiments using artificial flumes. For instance, in studies by Zeiringer et al. (2014) and Auer et al. (2016), drift and stranding of grayling larvae and juveniles increased due to hydropeaking. However, if experimental channels were equipped with permanently wetted lateral groins, hydropeaking effects were strongly reduced (Zeiringer et al., 2014). On the other hand, some morphological elements might intensify hydropeaking effects. For instance, low-gradient banks or gravel bars can increase the risk of stranding, because the reduction of the wetted area during water-level declines occurs faster compared to high-gradient banks (Bell et al., 2008; Hauer et al., 2016a; Tuhtan et al., 2012) and because animals may become trapped in potholes (Zeiringer et al., 2014; Young et al., 2011). Further investigations are required to assess the effects of other morphological elements (e.g., logs, gravel banks, boulders, rip-rap structures) but also to elucidate their stability during floods with substantial sediment redistribution (Bremset and Berg, 1999; Hauer et al., 2008).

Morphological and hydraulic heterogeneity might alleviate some hydropeaking effects with increasing distance to the outlet of the powerhouses as a consequence of bed roughness, hydrological retention, the influence of tributaries, and in general by providing refuges (Ribi et al., 2014). Morphological restorations of hydropeaking reaches and tributaries might also be applied as ecological compensation measures, if direct hydropeaking mitigation is unfeasible (Table 1). Because of these diverse interactions, hydropeaking mitigation and morphological restorations should be planned in concert (Hauer et al., 2016a).

4.3. The socio-economic context

Socio-economic considerations have great influence on the mitigation goals and the set of realistic mitigation measures (Hauer et al., 2016b; Hering et al., 2015). They include technical and economic constraints of the power plant, other land and water uses in the catchment and legal regulations such as those dealing with protection of landscapes, flood protection, or energy policy (Person et al., 2014). Other land and water uses might add further stress on biotic communities in hydropeaking reaches, for instance via altered sediment regimes, (agro-)chemical effluents, lack of longitudinal, lateral and vertical connectivity, and fish stocking. On the other hand, some mitigation measures provide secondary uses (Brunner and Rey, 2014), e.g. retention volumes can be equipped with turbines for additional hydroelectric generation and/or used as storage volume for pumped-storage power plants (Pérez-Díaz et al., 2012; Schweizer et al., 2008). Retention basins can also provide aquatic habitat or opportunities for recreational activities (Heller et al., 2010). The importance of the socio-economic context requires its inclusion in prediction of mitigation measures (Section 3.3.), as well as a catchment perspective for the planning and coordination of multiple mitigation projects and restoration initiatives, e.g. in a framework of integrated water resources management (Hering et al., 2011; Palmer et al., 2005). Participation of relevant social and political stakeholders is thus paramount during all these stages of mitigation projects (Reed, 2008; Trussart et al., 2002).

4.4. Monitoring hydropeaking mitigation projects

Like other restoration projects, the outcome of hydropeaking mitigation measures should be monitored and compared to pre-defined goals (Fig. 1; Palmer et al., 2005; Parasiewicz et al., 2013; Trussart et al., 2002). Monitoring programs should already be included in the planning of mitigation projects and coordinated with ecological assessments of the pre-mitigation situation, using, if possible, the same indicators (Fig. 1; Parasiewicz et al., 1998). Long-term monitoring might be required for detecting ecological recovery, which is especially true for slow processes and taxa with complex life cycles such as migratory fish. If mitigation goals are not met, causes of failure have to be identified and mitigation measures – or mitigation goals if they prove unrealistic – need to be adapted (Fig. 1). However, because structural hydropeaking mitigation measures are extremely costly, iterative approaches are usually precluded unlike for other restoration types, e.g. morphological restoration projects (Palmer et al., 2005). Institutionalizing the assessment and publication of experiences made with mitigation projects is thus crucial to improve the effectiveness of future mitigation measures (Palmer et al., 2005) and their conceptual basis (this study).

4.5. Knowledge gaps affecting efficient hydropeaking mitigation

Our analysis of the scientific literature revealed some hydropeaking effects that are still poorly understood. These include stranding of invertebrates or substrate clogging (Table 3; Becker et al., 1982; Shen and Diplas, 2010). Also, knowledge of hydropeaking effects on benthic algae, on microorganisms in general, and on the riparian vegetation is scarce (but see Graf, 2006; Merritt et al., 2010; Miller and Judson, 2014). In addition to new targeted research projects, knowledge gaps could be addressed by analysing datasets from the great number of hydropeaking assessment projects, which have been carried out using standardized methods, such as projects initiated by the European Water Framework Directive (WFD, 2000) and the Swiss Water Protection Act (Baumann et al., 2012; Tonolla et al., 2016).

A better mechanistic understanding of hydropeaking effects would, for instance, benefit the use of habitat models and their application in hydropeaking situations. Many habitat models oversimplify the hydraulic conditions in reaches affected by hydropeaking (Murchie et al., 2008), e.g. they ignore dynamic flow periods (Heggenes, 1996; but see Tuhtan et al., 2012). Accounting for dynamic hydrologic effects is crucial to predict stranding of fish and invertebrates (Halleraker et al., 2003; Young et al., 2011). As an example, the habitat models applied by Person et al. (2014) and Parasiewicz et al. (1998) did not account for effects of the rate of change between minimal and maximal flows and were thus unable to assess fish stranding, although both studies estimated high stranding risk based on morphological features of the respective river sections. Likewise, the weighted usable area (Bovee, 1986) and similar metrics used in many numerical habitat models do not contain information on the relative locations of microhabitats and the distance between them. This limitation is particularly relevant in the context of hydropeaking due to the fast changes in habitat conditions (Bunt et al., 1999; Scruton et al., 2008) during which organisms need to reach refuges (Bond and Jones, 2015).

5. Conclusions

The indicator-set and conceptual framework we suggest in this study allows prediction and evaluation of mitigation measures and is based on experiences from mitigation projects and on the current scientific knowledge. Partly, these findings have been integrated in a new guideline document published by the Swiss Federal Office for the Environment (Tonolla et al., 2016). Predicting all indicators suggested in our study requires a great effort and a broad range of methods as well as the involvement of hydrologists, hydrogeologists, and ecologists as well as the support of the respective hydropower operator. Hence, a sub-set of indicators might be applied if options are limited (Supplementary Table 1). However, defining this sub-set requires careful analysis of the relevant hydropeaking effects affecting key ecosystem processes in the respective river and needs to be elaborated in a participative process. In any case, detailed ecological assessments, evaluations, and monitoring are the basis for the successful planning and implementation of mitigation measures. Because they are in most cases extremely costly constructions or operational adaptations, a great effort is justified also during early steps of the mitigation projects such as the prediction of the effects of mitigation measures on river ecosystems.

Acknowledgements

We greatly appreciate inputs from P. Baumann, A. Peter, M. Huber Gysi, J. Schmidli, T. Meile, and M. Kummer during numerous discussions and those of the workshop participants (see Supplementary material). We further acknowledge inputs and discussions from hydropower operators, environmental agencies and consultancies, and the Swiss Federal Office for the Environment for funding this project. Comments from M. Alp, C. Weber, K. Lange, and two anonymous reviewers greatly improved the manuscript. S. D. Langhans acknowledges funding from the Alexander von Humboldt-Foundation.

Appendix A. Supplementary material

Supplementary material to this article can be found online at http://dx.doi.org/10.1016/j.scitotenv.2016.05.032.

References

Anselmetti, F.S., Bühler, R., Finger, D., Girardclos, S., Lancini, A., Rellstab, C., Sturm, M., 2007. Effects of alpine hydropower dams on particle transport and lacustrine sedimentation. Aquat. Sci. 69, 179–198.

Auer, S., Zeiringer, B., Führer, S., Schmutz, S., Tonolla, D., 2016. Effects of river bank morphology and time of day on drift and stranding of European Grayling (*Thymallus thymallus* L.) caused by hydropeaking. Proceedings of the 11th ISE Conference in Melbourne, Australia.

Baumann, P., Kirchhofer, A., Schälchli, U., 2012. Sanierung Schwall/Sunk - Strategische Planung. Ein Modul der Vollzugshilfe Renaturierung der Gewässer. Umwelt-Vollzug Nr. 1203. Bundesamt für Umwelt, Bern.

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.

Bell, E., Kramer, S., Zajanc, D., Aspittle, J., 2008. Salmonid fry stranding mortality associated with daily water level fluctuations in Trail Bridge Reservoir, Oregon. N. Am. J. Fish. Manag. 28, 1515–1528.

Bevelhimer, M.S., McManamay, R.A., O'Connor, B., 2015. Characterizing sub-daily flow regimes: implications of hydrologic resolution on ecohydrology studies. River Res. Appl. 31, 867–879.

Bieri, M., Schleiss, A.J., Jordan, F., Fankhauser, A.U., Ursin, M.H., 2011. Flood retention in alpine catchments equipped with complex hydropower schemes — a case study of the upper Aare catchment in Switzerland. In: Schleiss, A.J., Boes, R. (Eds.), Dams and Reservoirs Under Changing Challenges. Taylor & Francis Group, London, pp. 387–394.

Bieri, M., Müller, M., Schweizer, S., Schleiss, A.J., 2014. Flow restoration in Alpine streams affected by hydropower operations — a case study for a compensation basin. In: Schleiss, A.J., Speerli, J., Pfammatter, R. (Eds.), Swiss Competences in River Engineering and Restoration. Taylor & Francis Group, London, pp. 181–190.

Bonalumi, M., Anselmetti, F.S., Wüest, A., Schmid, M., 2012. Modeling of temperature and turbidity in a natural lake and a reservoir connected by pumped-storage operations. Water Resour. Res. 48, W08508.

Bond, M.J., Jones, N.E., 2015. Spatial distribution of fishes in hydropeaking tributaries of Lake Superior. River Res. Appl. 31, 120–133.

Bovee, K.D., 1986. Development and evaluation of habitat suitability criteria for use in the instream flow incremental methodology. Biological Report 86. US Fish and Wildlife Service.

Bremset, G., Berg, O.K., 1999. Three-dimensional microhabitat use by young pool-dwelling Atlantic salmon and brown trout. Anim. Behav. 58, 1047–1059.

Brunner, D., Rey, B., 2014. Hydropeaking and fish migration — consequences and possible mitigation measures at the Schiffenen Dam. In: Schleiss, A.J., Speerli, J., Pfammatter, R. (Eds.), Swiss Competences in River Engineering and Restoration. Taylor & Francis Group, London, pp. 173–180.

Bruno, M.C., Maiolini, B., Carolli, M., Silveri, L., 2009. Impact of hydropeaking on hyporheic invertebrates in an Alpine stream (Trentino, Italy). Ann. Limnol. - Int. J. Limn. 45, 157–170.

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). Limnol. Ecol. Manag. Inl. Waters 40, 281–290.

Bunt, C.M., Cooke, S.J., Katopodis, C., McKinley, R.S., 1999. Movement and summer habitat of brown trout (*Salmo trutta*) below a pulsed discharge hydroelectric generating station. Regul. Rivers Res. Manag. 15, 395–403.

Carolli, M., Bruno, M.C., Siviglia, A., Maiolini, B., 2011. Responses of benthic invertebrates to abrupt changes of temperature in flume simulations. River Res. Appl. 28, 678–691.

Carolli, M., Vanzo, D., Siviglia, A., Zolezzi, G., Bruno, M.C., Alfredsen, K., 2015. A simple procedure for the assessment of hydropeaking flow alterations applied to several European streams. Aquat. Sci. 77, 639–653.

Casas-Mulet, R., Alfredsen, K., García-Escudero Uribe, A., 2014. A cost-effective approach to predict dynamic variation of mesohabitats at the river scale in Norwegian systems. Int. J. River Basin Manag. 12, 145–159.

Conallin, J., Boegh, E., Jensen, J.K., 2010. Instream physical habitat modelling types: an analysis as stream hydromorphological modelling tools for EU water resource managers. Int. J. River Basin Manag. 8, 93–107.

Davey, A.J.H., Booker, D.J., Kelly, D.J., 2011. Diel variation in stream fish habitat suitability criteria: implications for instream flow assessment. Aquat. Conserv. Mar. Freshwat. Ecosyst. 21, 132–145.

De Jalon, D.G., Sanchez, P., Camargo, J.A., 1994. Downstream effects of a new hydropower impoundment on macrophyte, macroinvertebrate and fish communities. Regul. Rivers Res. Manag. 9, 253–261.

Dunbar, M.J., Alfredsen, K., Harby, A., 2012. Hydraulic-habitat modelling for setting environmental river flow needs for salmonids. Fish. Manag. Ecol. 19, 500–517.

EnviPEAK, 2016. Effects of Rapid and Frequent Flow Changes. CEDREN: Norwegian Centre for Environmental Design of Renewable Energy. http://www.cedren.no/english/Projects/EnviPEAK last accessed: 23.04.

Fette, M., Weber, C., Peter, A., Wehrli, B., 2007. Hydropower production and river rehabilitation: a case study on an alpine river. Environ. Model. Assess. 12, 257–267.

Freeman, M.C., Bowen, Z.H., Bovee, K.D., Irwin, E.R., 2001. Flow and habitat effects on juvenile fish abundance in natural and altered flow regimes. Ecol. Appl. 11, 179–190.

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.

Gaudard, L., Romerio, F., Dalla Valle, F., Gorret, R., Maran, S., Ravazzani, G., Stoffel, M., Volonterio, M., 2014. Climate change impacts on hydropower in the Swiss and Italian Alps. Sci. Total Environ. 493, 1211–1221.

Graf, W.L., 2006. Downstream hydrologic and geomorphic effects of large dams on American rivers. Geomorphology 79, 336–360.

Gurnell, A.M., Rinaldi, M., Belletti, B., Bizzi, S., Blamauer, B., Braca, G., Buijse, A.D., Bussettini, M., Camenen, B., Comiti, F., Demarchi, L., García de Jalón, D., González del Tánago, M., Grabowski, R.C., Gunn, I.D.M., Habersack, H., Hendriks, D., Henshaw, A.J., Klösch, M., Lastoria, B., Latapie, A., Marcinkowski, P., Martínez-Fernández, V., Mosselman, E., Mountford, J.O., Nardi, L., Okruszko, T., O'Hare, M.T., Palma, M., Percopo, C., Surian, N., van de Bund, W., Weissteiner, C., Ziliani, L., 2016. A multi-scale hierarchical framework for developing understanding of river behaviour to support river management. Aquat. Sci. 78, 1–16.

Haas, J., Olivares, M.A., Palma-Behnke, R., 2015. Grid-wide subdaily hydrologic alteration under massive wind power penetration in Chile. J. Environ. Manag. 154, 183–189.

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.

Hauer, C., Unfer, G., Schmutz, S., Habersack, H., 2008. Morphodynamic effects on the habitat of juvenile cyprinids (*Chondrostoma nasus*) in a restored Austrian lowland river. Environ. Manag. 42, 279–296.

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

Hauer, C., Holzapfel, P., Habersack, H., Tonolla, D., 2016b. Hydrologische, morphologische und sedimentologische Analysen als Grundlage für die Konzipierung von Schwall-Sunk-Massnahmen - Fallbeispiel Alpenrhein. WasserWirtschaft 1, 16–22.

Heggenes, J., 1996. Habitat selection by brown trout (*Salmo trutta*) and young Atlantic salmon (*S. salar*) in streams: static and dynamic hydraulic modelling. Regul. Rivers Res. Manag. 12, 155–169.

Heggenes, J., Bremset, G., Brabrand, Å., 2013. Visiting the hyporheic zone: young Atlantic salmon move through the substratum. Freshw. Biol. 58, 1720–1728.

Heller, P., Bollaert, E.F.R., Schleiss, A.J., 2010. Comprehensive system analysis of a multi-purpose run-of-river power plant with holistic qualitative assessment. Int. J. River Basin Manag. 8, 295–304.

Hering, J.G., Hoehn, E., Klinke, A., Maurer, M., Peter, A., Reichert, P., Robinson, C., Schirmer, K., Schirmer, M., Stamm, C., Wehrli, B., 2011. Moving targets, long-lived infrastructure, and increasing needs for integration and adaptation in water management: an illustration from Switzerland. Environ. Sci. Technol. 46, 112–118.

Hering, D., Carvalho, L., Argillier, C., Beklioglu, M., Borja, A., Cardoso, A.C., Duel, H., Ferreira, T., Globevnik, L., Hanganu, J., Hellsten, S., Jeppesen, E., Kodeš, V., Solheim, A.L., Nõges, T., Ormerod, S., Panagopoulos, Y., Schmutz, S., Venohr, M., Birk, S., 2015. Managing aquatic ecosystems and water resources under multiple stress — an introduction to the MARS project. Sci. Total Environ. 503/504, 10–21.

Holomuzki, J.R., Biggs, B.J.F., 2003. Sediment texture mediates high-flow effects on lotic macroinvertebrates. J. N. Am. Benthol. Soc. 22, 542–553.

IRKA, 2012. Zukunft Alpenrhein. Quantitative Analyse von Schwall/Sunk-Ganglinien für unterschiedliche Anforderungsprofile. IRKA - Projektgruppe Gewässer- und Fischökologie.

Jones, N.E., 2014. The dual nature of hydropeaking rivers: is ecopeaking possible? River Res. Appl. 30, 521–526.

Jones, J.I., Murphy, J.F., Collins, A.L., Sear, D.A., Naden, P.S., Armitage, P.D., 2012. The impact of fine sediment on macro-invertebrates. River Res. Appl. 28, 1055–1071.

Lagarrigue, T., Céréghino, R., Lim, P., Reyes-Marchant, P., Chappaz, R., Lavandier, P., Belaud, A., 2002. Diel and seasonal variations in brown trout (*Salmo trutta*) feeding patterns and relationship with invertebrate drift under natural and hydropeaking conditions in a mountain stream. Aquat. Living Resour. 15, 129–137.

Lamouroux, N., Capra, H., Pouilly, M., 1998. Predicting habitat suitability for lotic fish: linking statistical hydraulic models with multivariate habitat use models. Regul. Rivers Res. Manag. 14, 1–11.

Langhans, S.D., Lienert, J., Schuwirth, N., Reichert, P., 2013. How to make river assessments comparable: a demonstration for hydromorphology. Ecol. Indic. 32, 264–275.

Langhans, S.D., Hermoso, V., Linke, S., Bunn, S.E., Possingham, H.P., 2014. Cost-effective river rehabilitation planning: optimizing for morphological benefits at large spatial scales. J. Environ. Manag. 132, 296–303.

Lauters, F., Lavandier, P., Lim, P., Sabaton, C., Belaud, A., 1996. Influence of hydropeaking on invertebrates and their relationship with fish feeding habits in a Pyrenean river. Regul. Rivers Res. Manag. 12, 563–573.

Marty, J., Smokorowski, K., Power, M., 2009. The influence of fluctuating ramping rates on the food web of boreal rivers. River Res. Appl. 25, 962–974.

Meile, T., Boillat, J.L., Schleiss, A., 2011. Hydropeaking indicators for characterization of the Upper-Rhone River in Switzerland. Aquat. Sci. 73, 171–182.

Merritt, D.M., Scott, M.L., LeRoy Poff, N., Auble, G.T., Lytle, D.A., 2010. Theory, methods and tools for determining environmental flows for riparian vegetation: riparian vegetation-flow response guilds. Freshw. Biol. 55, 206–225.

Miller, S.W., Judson, S., 2014. Responses of macroinvertebrate drift, benthic assemblages, and trout foraging to hydropeaking. Can. J. Fish. Aquat. Sci. 71, 675–687.

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.

Murchie, K.J., Hair, K.P.E., Pullen, C.E., Redpath, T.D., Stephens, H.R., Cooke, S.J., 2008. Fish response to modified flow regimes in regulated rivers: research methods, effects and opportunities. River Res. Appl. 24, 197–217.

Niu, S., Insley, M., 2013. On the economics of ramping rate restrictions at hydro power plants: balancing profitability and environmental costs. Energy Econ. 39, 39–52.

Olden, J.D., Naiman, R.J., 2010. Incorporating thermal regimes into environmental flows assessments: modifying dam operations to restore freshwater ecosystem integrity. Freshw. Biol. 55, 86–107.

Palmer, M.A., Bernhardt, E.S., Allan, J.D., Lake, P.S., Alexander, G., Brooks, S., Carr, J., Clayton, S., Dahm, C.N., Follstad Shah, J., Galat, D.L., Loss, S.G., Goodwin, P., Hart, D.D., Hassett, B., Jenkinson, R., Kondolf, G.M., Lave, R., Meyer, J.L., O'Donnell, T.K., Pagano, L., Sudduth, E., 2005. Standards for ecologically successful river restoration. J. Appl. Ecol. 42, 208–217.

Parasiewicz, P., 2007. The MesoHABSIM model revisited. River Res. Appl. 23, 893–903.

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

Parasiewicz, P., Ryan, K., Vezza, P., Comoglio, C., Ballestero, T., Rogers, J.N., 2013. Use of quantitative habitat models for establishing performance metrics in river restoration planning. Ecohydrology 6, 668–678.

Pérez-Díaz, J.I., Millán, R., García, D., Guisández, I., Wilhelmi, J.R., 2012. Contribution of re-regulation reservoirs considering pumping capability to environmentally friendly hydropower operation. Energy 48, 144–152.

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, 580–599.

Poff, N.L., Richter, B.D., Arthington, A.H., Bunn, S.E., Naiman, R.J., Kendy, E., Acreman, M., Apse, C., Bledsoe, B.P., Freeman, M.C., Henriksen, J., Jacobson, R.B., Kennen, J.G., Merritt, D.M., O'Keeffe, J.H., Olden, J.D., Rogers, K., Tharme, R.E., Warner, A., 2010. The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards. Freshw. Biol. 55, 147–170.

Reed, M.S., 2008. Stakeholder participation for environmental management: a literature review. Biol. Conserv. 141, 2417–2431.

Reichert, P., Langhans, S.D., Lienert, J., Schuwirth, N., 2015. The conceptual foundation of environmental decision support. Environ. Manag. 154, 316–332.

Ribi, J.M., Boillat, J.L., Peter, A., Schleiss, A.J., 2014. Attractiveness of a lateral shelter in a channel as a refuge for juvenile brown trout during hydropeaking. Aquat. Sci. 76, 527–541.

Richards, R.R., Gates, K.K., Kerans, B.L., 2013. Effects of simulated rapid water level fluctuations (hydropeaking) on survival of sensitive benthic species. River Res. Appl. 30, 954–963.

Schweizer, S., Neuner, J., Ursin, M., Tscholl, H., Meyer, M., 2008. Ein intelligent gesteuertes Beruhigungsbecken zur Reduktion von künstlichen Pegelschwankungen in der Hasliaare. Wasser Energ. Luft 100, 209–215.

Schweizer, S., Meyer, M., Heuberger, N., Brechbühl, S., Ursin, M., 2010. Zahlreiche gewässerökologische Untersuchungen im Oberhasli: wichtige Unterstützung des partizitiven Begleitprozesses von KWOplus. Wasser Energ. Luft 102, 289–300.

Scruton, D.A., Pennell, C., Ollerhead, L.M.N., Alfredsen, K., Stickler, M., Harby, A., Robertson, M., 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.

Shen, Y., Diplas, P., 2010. Modeling unsteady flow characteristics of hydropeaking operations and their implications on fish habitat. J. Hydraul. Eng. 136, 1053–1066.

Shields, F., Copeland, R., Klingeman, P., Doyle, M., Simon, A., 2003. Design for stream restoration. J. Hydraul. Eng. 129, 575–584.

Smokorowski, K.E., Metcalfe, R.A., Finucan, S.D., Jones, N., Marty, J., Power, M., Pyrce, R.S., Steele, R., 2011. Ecosystem level assessment of environmentally based flow restrictions for maintaining ecosystem integrity: a comparison of a modified peaking versus unaltered river. Ecohydrology 4, 791–806.

Tonolla, D., Chaix, O., Meile, T., Zurwerra, A., Büsser, P., Oppliger, S., Essyad, K., 2016. Schwall-Sunk - Massnahmen. Ein Modul der Vollzugshilfe Renaturierung der Gewässer. Swiss Federal Office for the Environment, Berne. Umwelt-Vollzug.

Trussart, S., Messier, D., Roquet, V., Aki, S., 2002. Hydropower projects: a review of most effective mitigation measures. Energy Policy 30, 1251–1259.

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.

Valentin, S., Lauters, F., Sabaton, C., Breil, P., Souchon, Y., 1996. Modelling temporal variations of physical habitat for brown trout (*Salmo trutta*) in hydropeaking conditions. Regul. Rivers Res. Manag. 12, 317–330.

Vanzo, D., Zolezzi, G., Siviglia, A., 2016. Eco-hydraulic modelling of the interactions between hydropeaking and river morphology. Ecohydrology 9, 421–437.

Vezza, P., Muñoz-Mas, R., Martinez-Capel, F., Mouton, A., 2015. Random forests to evaluate biotic interactions in fish distribution models. Environ. Model. Softw. 67, 173–183.

Vučijak, B., Kupusović, T., Midžić-Kurtagić, S., Ćerić, A., 2013. Applicability of multicriteria decision aid to sustainable hydropower. Appl. Energy 101, 261–267.

WFD, 2000. Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for community action in the field of water policy. Off. J. L. 327, 1–73.

White, P.S., Walker, J.L., 1997. Approximating nature's variation: selecting and using reference information in restoration ecology. Restor. Ecol. 5, 338–349.

WPA, 2011. 814.20 Federal Act of 24 January 1991 on the Protection of Waters (Swiss Water Protection Act, Version of 2011).

Young, P., Cech, J., Thompson, L., 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.

Zeiringer, B., Fohler, N., Auer, S., Greimel, F., Schmutz, S., June 23-27, 2014. Experiments on drifting and stranding of juvenile grayling during fluctuating flow in nature-like channels with different morphological structures. 10th International Symposium on Ecohydraulics 2014, Trondheim, Norway. In: Norwegian University of Science (Ed.), 10th International Symposium on Ecohydraulics 2014 E-proceedings.

Zimmerman, J.K.H., Letcher, B.H., Nislow, K.H., Lutz, K.A., Magilligan, F.J., 2010. Determining the effects of dams on subdaily variation in river flows at a whole-basin scale. River Res. Appl. 26, 1246–1260.

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