Impacts and Risks of Hydropower

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4.1 Introduction

The detrimental effects hydropower plants have on aquatic ecosystems and biodiversity are manifold and comprehensively reviewed (e.g., Gasparatos et al. 2017, Hecht et al. 2019, Jungwirth et al. 2003. Lees et al. 2016, Reid et al. 2019, Schmutz and Sendzimir 2018, Stendera et al. 2012, Ziv et al. 2012). In the following section, however, we review, categorize and outline hydropower-related impacts on freshwater fishes only. This is due

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to various reasons: For one, fishes are of great socio-economic interest. Their unquestionable cultural and societal value has caused managing efforts to support self-sustained, exploitable fish stocks for several thousand years, and today they are a priority target for many restoration and conservation programs. Furthermore, fish are most affected by the operation of hydropower (Larinier 2001) and the high level of hydromorphological degradation and resulting habitat loss associated with hydropower has been identified as one of the bottlenecks in reaching the Water Framework Directive targets (Freyhof et al. 2019).

Therefore, this chapter draws a comprehensive conceptual model depicting what kinds of impacts on fish potentially happen beginning from habitat loss/modification upstream due to the impoundment, migration delays, indirect mortality due to increased predation, the hydropower plant (HPP) itself, with potential spillway, bypass, trash racks and also turbine effects (blade strike, shear forces, barotrauma) and down to tailwater effects, such as increased predation, residual flows, habitat and flow modifications (Fig. 4.1).

The resilience of fish species and populations as well as species most at risks will be addressed based on narratives derived as risk factors and the empirical evidence provided by the literature review.

4.2 Barrier Effects

The central, most prominent element of every hydropower scheme is undoubtedly a dam or a weir. Although these types of barriers are not exclusive to hydropower plants, they always have the same principal effects on fishes. Because barriers become impassable

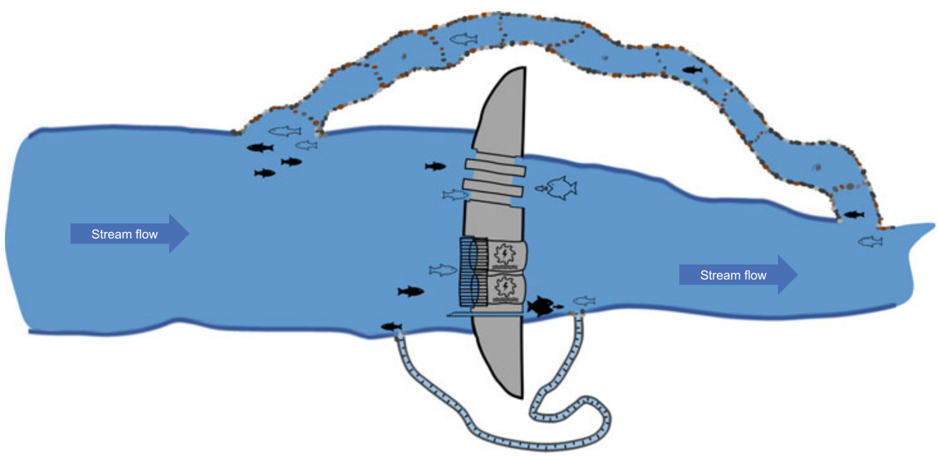

Fig. 4.1 Conceptual sketch of elements of a hydropower scheme potentially impacting fish, like the barrier itself, upstream and downstream migration routes, turbines, trash racks and fish protection facilities

obstacles for fishes once they exceed certain dimensions, they segregate resident populations into isolated upstream and downstream components. Barriers disrupt the original river continuum (Allan and Castillo 2007; Mueller et al. 2011; Vannote et al. 1980) and the natural migration corridors for fishes (Jonsson et al. 1999). Dams and weirs act as migration barriers for migratory species that then face substantial migration delay (Buysse et al. 2015; Ovidio et al. 2017; Stich et al. 2015, Winter et al. 2006), and they render critical habitats inaccessible to fishes (Larinier 2001; Pelicice et al. 2015). However, unhindered upstream migration is particularly critical for diadromous migratory species like salmonids, lampreys, some clupeids or sturgeons that only spawn in the upper regions of rivers where hydraulic and geomorphic conditions support egg development and provide larval habitats (Katano et al. 2006; Lucas et al. 2009; Penczak et al. 1998). But also, migrations of potamodromous species are impaired by barriers (Britton and Pegg 2011; Lucas and Frear 1997). This can result in reduced natural recruitment (McCarthy et al. 2008), differences in population structure and species assemblages up- and downstream of the dam (Franssen and Tobler 2013; Morita and Yamamoto 2002; Mueller et al. 2011) and even result in the extinction of entire fish stocks (Larinier and Travade 1992), unless habitat heterogeneity and availability in the system remains high enough to support the native assemblage (Santos et al. 2006). Furthermore, because dams act as bi-directional nutrient traps that can cause a reduction of far-downstream fish biomass (Jackson and Marmulla 2001) and a lack of nutrients (i.e., due to a lower number of spawners remaining in the headwaters of streams), which directly affects the dietary composition of a range of different fish species (Piorkowski 1995). The mechanisms described in this paragraph primarily impact population endpoints that ultimately, cause a decline in recruitment, whereas individual mortality of affected fishes is only of secondary concern.

The negative ecological impacts of barriers can be partly mitigated by maintaining certain flow velocity through the impounded area that resembles the ecological functioning of the former stream. These flow patterns are cues for up- and downstream migrating species and ensure sediment transport and aeration.

4.3 Upstream Flow Alterations

Dams cause substantial alterations of the stream's original discharge regime (Egré and Milewski 2002; Schiemer et al. 2001). Reservoirs and impoundments considerably slow down the stream's flow velocity causing higher sedimentation rates of finer particles, stratification, increased temperature, and potential oxygen depletion in the hypolimnion due to an imbalance in aerobic production and consumption (Thornton et al. 1990). In principle, impoundments transform lotic habitats into ones with more lentic characteristics (Sá-Oliveira et al. 2015) that are unsuited for most riverine, lithophilic species that require well aerated, fast flowing coarse gravel beds as spawning habitats (Wood and Armitage 1997). These conditions result in habitat loss for a range of rheophilic species (Agostinho et al. 2002; Birnie-Gauvin et al. 2017; Larinier 2001; Tiffan et al. 2016), changes in water quality (Fantin-Cruz et al. 2016), shifts in biomass and ultimately, changes of species abundance and diversity relative to non-impounded reaches downstream (Sá-Oliveira et al. 2015). These conditions also affect species-specific length-frequency distributions, species richness (Gehrke et al. 2002) and species composition (Tundisi and Straškraba 1999). Manipulated abiotic conditions in impoundments were further associated with temperature-related changes of growth patterns (i.e., younger age of maturity and smaller individual sizes) (Reed et al. 1992). For example, another study by Yang et al. (2020) showed reduced energy transfer efficiency in impoundments, suggesting potential energetic bottlenecks of fish at higher trophic levels. In impoundments altered hydromorphological conditions have caused increased predation, most likely because of the novel environment, lack of navigation cues for diadromous species (Agostinho et al. 2002; Jepsen et al. 2000; Tiffan et al. 2016) and the resulting migration delay (Larinier 2001; Larinier and Travade 2002) and reinforce negative impacts of introduced predators (Pelicice and Agostinho 2009). This can lead to local extinction of native and proliferation of non-native species (Martinez et al. 1994).

4.4 Downstream Flow Alterations

Different types of HPP have to be distinguished. There are run-of-river HPP of both instream or diversion-type schemes and storage HPP as well as pump-storage plants (Egré and Milewski 2002; Matt et al. 2019). Particularly storage, but to some extent also run-of-river hydropower plants dampen high natural discharge amplitudes by cutting flow peaks and increasing very low discharges. As such, they completely alienate the natural discharge regime of a stream, with flow fluctuations downstream being most problematic at all plants that do not release approximately as much water through the dam (i.e., through the turbines, spill gates or sluices) as would normally be discharged in the stream.

In diversion plants, the main purpose of the dam is to divert water away from the main stream towards the (potentially very remote) powerhouse where the water is turbinated and returned to the original river bed further downstream (Egré and Milewski 2002). The residual old river bed usually suffers from water scarcity, and methodological frameworks for defining sufficient environmental flow in the affected stretch are summarized by the CIS Guidance 31 "Ecological Flows in the Implementation of the Water Framework Directive" that can be consulted to mitigate the negative effects. At HPPs in which only a fraction of the original discharge remains in the residual river stretch severe consequences regarding water depths, flow velocities, and temperature extremes were observed. These do not support some fish populations anymore, cause species shifts and population declines (Anderson et al. 2006; Benejam et al. 2016; Habit et al. 2007) and sometimes even render whole river stretches uninhabitable. At some HPPs with state-of-the-art environmental flows of at least 10% mean annual stream flow (Huckstorf et al. 2008) these impacts are less pronounced. However, maintaining the comparably high environmental flow usually comes at the expense of hydroelectricity generation and loss of revenues.

Hydropeaking plants typically store larger amounts of water and release it for electricity generation in times of peak demand, mostly in the morning and evening (Moreira et al. 2019; Schmutz et al. 2015; Schmutz and Sendzimir 2018). Many species cannot cope with manipulated flow alterations induced by turbine operation which can lead to reduced food availability (De Jalon et al. 1994; Gandini et al. 2014; Young et al. 2011), erosion and habitat loss due to periodical dewatering (Almodóvar and Nicola 1999; Boavida et al. 2015, 2013; Choi et al. 2017; Person 2013; Shen and Diplas 2010; Young et al. 2011) and impaired egg development (Casas-Mulet et al. 2015a, b; Person 2013; Young et al. 2011), all of which commonly resulting either in reduced recruitment or increased direct mortality e.g., by stranding (Hedger et al. 2018, Schmutz et al. 2015, Tuthan et al. 2012, Young et al. 2011) in particular of smaller species or younger specimen with weaker swimming performance (Hayes et al. 2019; Person 2013).

If water shortage or pulse flows are not evident, manipulated flows can still exert major pressures on fishes e.g., because new habitat types immediately emerge beneath the dam that support accumulation of fishes (Jackson 1985) that attract unnaturally high abundances of predators able to deplete already impaired stocks (Larinier 2001; Stansell et al. 2010). In addition, hydropeaking can lead to altered sediment dynamics in rivers with severe consequences for lithophilic fish species (Casas-Mulet et al. 2015a, b).

4.5 Upstream Passage

Upstream migration needs of fishes have received much more attention relative to downstream migration needs, and respective efforts to increase passage rates date back longer, too (Katopodis and Williams 2012). The decline of the highly valued anadromous salmonids and the respective fisheries in response to damming became obvious very early on and had resulted in first legal acts that obliged e.g., mill owners to care about fish migration. In this context, attempts to facilitate upstream movement of fish that actively search for passage corridors have been more successful compared to attempts to guide fish following the main current in downstream direction (Geist 2021). Correspondingly, comprehensive guidelines exist to facilitate operational upstream migration facilities under varying environmental, technical and biotic conditions e.g., the DWA guidance M 509 (DWA 2014). However, upstream migration facilities show highly varying passage rates between 0 and 100% (Bunt et al. 2012; Gowans et al. 2003; Hershey 2021; Kemp et al. 2011), mostly due to the unique and highly complex interaction between the species' internal state and motivation to migrate, their anatomy and swimming ability, ambient hydraulic conditions and type and design of the passage facility (Banks 1969, Castro-Santos et al., 2009, Crisp 2000, USFWS (U.S. Fish and Wildlife Service) 2019). In Europe the implementation of the WFD stipulated the re-establishment of the longitudinal connectivity (Schletterer et al. 2016) and various technical as well as natural fishways were developed or species-specifically improved (Clay 2017; Hershey 2021; Jungwirth et al. 1998; Katopodis 1992; Santos et al. 2014).

Factors determining passage success of an upstream fishpass include attraction efficiency mediated by position of entrance and attraction flow and passability mediated by slope, flow velocity in the migration corridor, height differences and physical dimensions (Banks 1969; Bunt et al. 2012; DWA 2014; Hershey 2021, USFWS 2019). Failing upstream passage success of fish result in excessive energy expenditure and migration delays (Noonan et al. 2012; Silva et al. 2019; Thorstad et al. 2008) and thus, delayed arrival at spawning events (Silva et al. 2019), and increased predation (Agostinho et al. 2012). When HPPs are aligned in cascades their cumulative barrier effects must be considered (Geist 2021) as it aggravates already significant delays, migration failures and mortalities threatening the persistence of fish populations (Caudill et al. 2007; Gowans et al. 2003; Muir et al. 2001; Roscoe et al. 2011; Williams et al. 2001).

4.6 Downstream Passage

Downstream passage attained attention only much more recently, but is of similar relevance especially for iteroparous species spawning more than once in a lifetime. Beside the target species (diadromous or potamodromous) and the biocoenotic region (upper vs. lower course and associated species guilds) also HPP constellation (size, turbine type, etc.) and operational mode need to be considered (Schmidt et al. 2018; Travade and Larinier 2002). Particularly, juveniles of anadromous and adults of catadromous guilds but also potamodromous species require unobstructed downstream migration corridors. Therefore, HPPs must be equipped with fish guiding structures that facilitate downstream fish migration. Generally, all routes downstream over barriers and through HPPs are inherently dangerous for fishes and may result in migration delay or elevated mortality.

Spillways, mostly used to release excess water in times of higher discharge, can serve as effective and comparably fish-friendly downstream paths through a hydropower plant with bypass efficiencies of >90% (Muir et al. 2001). However, water released through spillways, particularly from bigger heights, tends to supersaturate with nitrogen and oxygen and, together with shear forces, pressure changes and blunt trauma or abrasions, can cause substantial damages and high mortality rates: up to 2% at a height of <3 m, up to 40% at 10 m and up to 100% at 50 m (Algera et al. 2020; Heisey et al. 1996; Schilt 2007; Wolter et al. 2020), with larger fish being significantly more susceptible to drop-induced injuries than smaller ones (Ruggles and Murray 1983).

Sluice gates installed at hydropower plants are mostly opened to spill debris or discharge excess inflow and may constitute temporarily available pathways for downstream migrating fish, too. Because the hydraulic conditions around an open (esp. undershot) gate act as a strong cue for migrating species sluices have proven efficient in conveying e.g., European eels downstream (Egg et al. 2017). However, undershot pathways may expose passing fish to rapid pressure changes that by far exceed those at overshot routes (Pflugrath et al. 2019), causing up to 95% mortality rates, especially for juveniles, small species and those with pressure-sensitive swim bladders (Algera et al. 2020; Baumgartner et al. 2006; Marttin and De Graaf 2002), while passage efficiency varies between <20% (Kemp et al. 2011) and >90% (Gardner et al. 2016).

Bypasses are dedicated downstream migration routes for fishes and most often used in combination with deflection screens or behavioural guidance facilities (Ebel et al. 2015). Their set-up is usually relatively simple, comprising concrete or metal chutes, slides or pipes that flush entering fishes downstream. Operational and efficient bypasses must be easily accessible, sufficiently dimensioned and supplied with enough water (commonly measured as a proportion of the turbine flow rate), and the entering water should have a slightly higher flow velocity than the recommended approaching flow of deflection screens (Ebel et al. 2015; Larinier and Travade 2002). Studies quantifying bypass mortalities are comparably scarce (Algera et al. 2020), but documented bypass-related damages and mortalities are mainly caused by sheer forces, rapid pressure changes, collisions, disorientation and subsequent predation in the tailrace (Williams et al. 2001); however, mortalities remained generally lower compared to other downstream routes (Algera et al. 2020). Bypass passage rates of fish showed significant variation between 0 and 95% (Gosset et al. 2005; Nyqvist et al. 2018; Ovidio et al. 2017).

Trash racks are installed in front of turbine intakes to protect them from large debris like wood. Normally, they feature vertical bars that—depending on design requirements—may be slightly inclined. The bar spacing is usually very wide to minimize head loss and constitute a substantial risk for larger fish that may get impinged and damaged when the approaching flow velocity is too high, during trash cleaner operations or when debris accumulates in the forebay (Weibel 1991). Studies investigating mortality rates of fishes due to trash racks are methodologically very challenging and thus, scarce.

Deflection screens with much smaller bar spacing installed at HPP behind or instead of trash racks are mechanical and behavioural barriers that prevent fishes from entering the turbines. Fish deflection screens come in a wide variety of designs e.g., vertically inclined with vertical bars and horizontally angled screens with horizontal bars that mostly deflect fishes mechanically, or horizontally angled screens with vertical bars inducing an additional behavioural change that increases the deflection performance up to 95% (Albayrak et al. 2020; Amaral 2003; Beck 2019; Calles et al. 2013; Ebel 2013a; Ebel et al. 2015; Nyqvist et al. 2018). The purely mechanical deflection rate can be approximated using empirical length-width-regressions by (Ebel 2013b): for example, 18 mm bar spacing would deflect fusiform fish of approximately ≥16 cm and eel of approximately ≥55 cm length; 15 mm bar spacing would lower these values to 13.6 and 48 cm. In contrast, a common trash rack with 80–100 mm bar spacing is consequently passable for almost all native species. When the approaching flow exceeds the recommended value of approximately 0.5 m/s (Calles et al. 2013; DWA 2014; Ebel et al. 2015; Larinier and Travade 2002), fish may be impinged in the screen and get damaged (Calles et al. 2013; Larinier 2001). Typically, physical/behavioural deflection screens and downstream bypasses form a functional unit (Ebel et al. 2015; Gosset et al. 2005; Larinier and Travade 2002; Nyqvist et al. 2018; Økland et al. 2019) and are not considered operational in absence of each other.

Turbine passage is probably the best-studied, most dangerous downstream route for fishes (Algera et al. 2020, Eicher et al. 1987). Depending on type and size of the turbine, fishes can get damaged or killed usually by either one or a combination of i) abrupt pressure changes (barotrauma), ii) turbulent flow, iii) shear forces, and iv) turbine blade strikes (USFWS 2019). Generally, the consequences of direct and delayed mortality as well as external (Mueller et al. 2017) and internal (Mueller et al. 2020a, b, c, d, e, f, g, h, i) injuries following turbine passage must be distinguished. Reported mortalities were highly variable across and within turbine types e.g., 1–7.7% in "Very Low Head" (VLH) turbines (Hogan et al. 2014; Reuter and Kohout 2014), 2% in Alden turbines (Hogan et al., 2014), 2–2.4% for the "Minimum Gap Runner" (MGR) (Čada et al. 1997; Hogan et al. 2014), 0.1–2.5% in water wheels (Pulg and Schnell 2008; Quaranta and Wolter 2021; Reuter and Kohout 2014), 0–32.7% in Archimedes screws (Buysse et al. 2015; Hogan et al. 2014; Piper et al. 2018; Pulg and Schnell 2008; Reuter and Kohout 2014), 0.3–100% in Kaplan turbines (Anon et al. 1987, Čada et al. 1997, 2006; Čada 2001; Cramer and Oligher 1964; Reuter and Kohout 2014; Richmond et al. 2014), although the risk of lethal blade strike in large Kaplan turbines can be substantially reduced compared to that of smaller ones (Bell and Kynard 1985), 15 to >70% in Ossberger turbines (Gloss and Wahl 1983), 4–100% in Francis turbines (Anon et al. 1987, Cramer and Oligher 1964; Pulg and Schnell 2008; Reuter and Kohout 2014) and 100% in Pelton wheels (Reuter and Kohout 2014). Fish mortality increases with increasing rotational speed (Anon et al. 1987, Buysse et al. 2015; Cramer and Oligher 1964; Odeh 1999; Turnpenny et al. 2000) usually inversely correlates with turbine size and positively correlates with fish size (Čada 1990; Colotelo et al. 2012; Pracheil et al. 2016) and hydraulic head (Anon et al. 1987, Larinier 2001) i.e., with rapid decompression and lack of acclimation time (Brown et al. 2009, 2012; Colotelo et al. 2012; Cramer and Oligher 1964; Odeh 1999; Pracheil et al. 2016; Richmond et al. 2014; Stephenson et al. 2010; Turnpenny et al. 2000). Further, mortality decreases with increasing turbine load (Čada et al. 1997; Cramer and Oligher 1964) and depends on fish behaviour and species (Amaral et al. 2015; Calles et al. 2010; Coutant and Whitney 2000; Ebel 2013a; Havn et al. 2017). Even if direct mortality rates are not evident, fishes may die from their injuries later (Ferguson et al. 2006; Mueller et al. 2020c, 2020f, 2020a, 2020e, 2020d, 2020b, 2020g; Muir et al. 2006; Taylor and Kynard 1985). This delayed mortality can be substantial and not accounting for it might severely underestimate damage rates during field studies and therefore, must be considered in the experimental design.

Turbine entrainment can cause damages and mortalities, and thus, be a significant population impact factor not only for juveniles with weaker swimming abilities or migratory species (i.e., salmonid smolts) (Mathur et al. 2000; Thorne and Johnson 1993) but also for potamodromous (Harrison et al. 2019) and even resident adult fishes, mainly in fall and winter (Martins et al. 2013). However, survival for smaller (i.e., juvenile) fishes at turbine passage is often higher than for adults, and turbine entrainment may therefore contribute to the persistence of downstream populations, albeit at the expense of populations upstream (Amaral et al. 2018; Harrison et al. 2019). Entrainment and mortality of drifting fish larvae are severely understudied and have not been quantified so far.

4.7 Risk and Impact Assessment

Measuring, describing, and predicting the actual impact of a HPP or specific, hydropower-related stressors on fish populations is challenging and almost impossible, regardless of the knowledge about single, site- or constellation-specific factors. This is due to several reasons.

First, the lack of information on the reference state, that is the undisturbed condition of the system (Nijboer et al. 2004). The fundamental elements of many HPP (i.e., dams or weirs) are fairly old, and (at least in Europe) new, and particularly small hydropower plants are commonly built on top of existing infrastructure. This imposes serious constraints on typical means of impact investigations like BA (before-after) or BACI (before-after-control-impact) designs (Conner et al. 2015b; Eberhardt 1976; Green 1979; Smith 2014), unless the scientific objective is to assess the additional impact or mortality factor of the hydropower plant compared to that of the already existing dam. If construction work on the HPP or dam has not yet started studies applying BACI designs could be used to investigate hydropower-related impacts before and after completion (e.g., Almodóvar and Nicola 1999), but if a particular stressor is already in place meaningful conclusions about its impact are more difficult to obtain. Pressure-release studies, for example in the context of dam removals or restoration (Catalano et al. 2007; Conner et al. 2015a), could identify improvements from the prevalent condition without knowledge about the reference condition. However, such studies merely describe the “opportunistic” response of the ecosystem and not its resilience i.e., its proximity to the pre-disturbance state. Further, most river systems are facing multiple stressors (Mueller et al. 2020a, b, c, d, e, f, g, h, i) and the single impacts of HPPs are hard to disentangle.

Second, investigations of impacts from hydropower on fish populations are biased towards migratory (i.e., diadromous) species that express clearly distinguished, life stage-critical habitat shifts (Geist 2021). Species with a pronounced migration tendency like anadromous salmonids and lampreys will by default always attempt to pass the hydropower plant if their spawning or rearing grounds are located upstream of the plant. In contrast, it becomes much more difficult to detect impacts at the population level of resident, non-migratory or potamodromous species that do not express long-distance migratory behaviour, migrate within the river system or even stay in the impoundment.

Furthermore, the complexity of different hydropower-related stressors, their interactions, cumulative effects on river system scale (Geist 2021) and summed impact on resident or migratory fishes raise difficulties in predicting their impact in isolation, especially in relation to varying susceptibility of fish assemblages across sites. Conclusions drawn from observations at one site are not necessarily valid at another. While the constellation of a few hydropower components (e.g., turbine type and hydraulic head or turbine size, rotational speed and flow rate) will remain relatively constant across sites and applications, others are much more subject to either the operator's intentions (e.g., operation modes), geo- and hydro-morphologically imposed structural design decisions (e.g., plant type, stream and discharge, mode of operation), spatial limitations (e.g., upstream migration facilities), composition and diversity of the ambient fish community, and fish protection facilities installed (e.g., dimensions of fish deflection screens and design or location of bypass systems). These elements can not only be combined in many different ways, they also interact uniquely with fish species and their life stages. Last but not least, site-specific environmental and conservation concerns do not only constrain the implementation details of a HPP, they also frame the environmental impact assessment. In conservation priority areas, even low impacts from hydropower might not be tolerable, while in heavily modified rivers HPPs of moderate impact might be acceptable.

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