AMBER

www.amber.international

This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No. 689682.

D2.2 Conceptual model of ecological impacts of barriers in EU considering fish habitat selection criteria for running waters

This is version 2.0 of D2.2 "Conceptual model of ecological impacts of barriers in EU considering habitat selection criteria for running waters". This document is a deliverable of the AMBER project that has received funding from the European Union's Horizon 2020 Programme under Grant Agreement (GA) # 689682.

History of changes

Version	Date	Changes	Pages
1.0	18 June 2018	Original Document	
2.0	11 April 2019	Changes have been made to the Executive Summary Changes have been made to the Authors Changes have been made to 1.1 Changes have been made to 1.3.1 Section 2.3 updated and added as Section 3.4 New Section 2.3 has been added Changes have been made Section 2.4 Changes have been made to Section 3 Changes have been made to Section 4 Changes have been made to Section 5	2 3 5-9 12-15 55 23-39 40-41 41-62 62-67 67

DISCLAIMER

Executive Summary

This document is version 2.0 of D2.2 the *Conceptual model of ecological impacts of barriers in EU considering habitat selection criteria for running waters*. This report is a deliverable of the AMBER project. This project has received funding from the European Union's Horizon 2020 research and innovation programme under grant agreement No 689682.

The core element of this deliverable is the development of a conceptual model of barrier impacts on rivers using the European Fish Community Macrohabitat River types (FCMacHT) presented in D2.1**.** Since new data has become available, the FCMacHT classification has been modified and this improvement is presented here. We based the typology on a number of physio-geographic factors associated with Expected Fish Communities (EFC), in order to assess the response of fish assemblages to riverine habitat changes caused by barrier impacts on a continental scale, macro habitat use and tolerance guilds were adopted as the most useful indicators.

We summarized 21 physical river attributes that can be modified by barriers, and developed macro-habitat suitability criteria to estimate their influence on the occurrence of fish guilds. We then estimated barrier impacts as a proportional alteration of habitat area for each attribute. For each FCMacHT type we calculated the amount of remaining suitable habitat area, weighted by the proportions of each guild in the EFC for 6 identified barrier types, assuming that all possible mitigation measures have been taken and the water body has achieved good ecological potential (GEP). The weighted Remaining Habitat proportion (*wRHp*) was arranged in five impact categories. Finally, for barriers where the ecological potential is affected by shortcomings in construction and operation or lack of mitigation measures, a penalty of 25% of *wRHp* was assumed, changing the impact class for each barrier type in each FCMacHT. Dams in mountainous areas have the highest impact on fish habitat. The model presented here provides a framework for regional assessment of barrier impacts on riverine habitats, as well as a foundation for examining barrier impacts under various scenarios of landscape management and climate change. One key variable is flow alteration, which will be incorporated via more detailed analysis in Tasks 2.2 and 2.3.

Author(s)

Piotr Parasiewicz (SSIFI), Maciej Zalewski (ERCE), Małgorzata Łapińska (ERCE), Paweł Prus (SSIFI), Mikołaj Adamczyk (SSIFI), Kamila Belka (ECRC), Piotr Frankiewicz (ERCE), Zbigniew Kaczkowski (ERCE), Kinga Krauze (ERCE), Katarzyna Suska (SSIFI), Janusz Ligięza (SSIFI), Josh Jones (SU), Luca Börger (SU), Devid Krull (innogy SE), Claus Till Schneider (innogy SE), Carlos Garcia de Leaniz (SU) for the AMBER consortium.

Abbreviations

These abbreviations are used throughout the text:

EFI+	new European Fish Index (method to assess ecological status of fish in European Rivers)
NDS	"non disturbed sites" **-** fish sampling sites classified as representing low anthropogenic pressure in the European Intercalibration Database (Fish)
FCMacHT	Fish Community Macro Habitat Type (physically defined river types based on habitats considered to be associated with fish guilds)
EFC	Expected Fish Communities (fish guilds proportion expected in undisturbed, reference rivers of a given type)
GP	Fish guild proportion in EFC
wRHp	Weighted remaining habitat proportion
GSI	Guild specific suitability of habitat attribute
Bi	Barrier impact
WFD	Water Framework Directive (European Freshwater Legislation)
CIS	Common Implementation Strategy of the WFD
WB	Water Body (concerning requirements in WFD)
HMWB	Heavily Modified Water Body (concerning requirements in WFD)
GEP	Good Ecological Potential of the Heavily Modified Water Bodies
GES	Good Ecological Status of the Natural Water Bodies
D1.1	Guidance on Stream Barrier Surveying and Reporting. Part A: Locating, Surveying and Prioritising Mitigation Actions for Stream Barriers. November, 2016.
D2.1	Classification map of running waters considering fish community structure and barrier impacts (AMBER)

Contents

1 Introduction 5

1.1 Background for the concept 5

1.2 General overview and catchment perspective 10

1.3 The impact of barriers on fish 11
1.3.1 Dams and weirs 12
1.3.2 Sluices 15
1.3.3 Culverts 15
1.3.4 Ramps 16
1.3.5 Fords 16

1.4 Barrier impact on biota other than fish 17
1.4.1 Dams 17
1.4.2 Weirs 17
1.4.3 Sluices 18
1.4.4 Ramp-bed sills 18

1.5 Cumulative environmental effects of barrier impacts 18

2 Materials and Methods 20

2.1 Assumptions for the Conceptual Model 20

2.2 Description of habitat attributes 22

2.3 Creation of the Fish Macrohabitat Types for European rivers 23
2.3.1 Data source 23
2.3.2 Fish habitat use guilds 35
2.3.3 Statistical analysis 39
2.3.3.1 Cluster Analysis 39

2.4 Quantification of barrier impact on fish habitat 40

3 Results 41

3.1 Cluster Analysis 41
3.1.1 CART 49

3.2 Map 51

3.3 Expected Fish Communities 52

3.4 Guild specific habitat suitability criteria 55

3.5 Quantification of barrier impact on fish habitat 59

4 Discussion and Conclusions 62

5 References 67

6 Appendix A: Review of Barrier Impact on Biota 74

7 References in Appendix A 104

1 INTRODUCTION

Habitat fragmentation has been recognized as one of the main causes of reduced biodiversity in rivers. It is estimated that over a million physical barriers are dividing and altering riverine ecosystems in Europe, and our knowledge of the consequences is not well structured or organized. It is therefore essential to develop a conceptual model of how barriers affect aquatic habitat suitability, and consequently alter the composition of aquatic communities. Fragmented and impounded rivers are frequently considered Heavily Modified Water Bodies (HMWB). Member States can designate HMWB where the changes to the hydromorphological characteristics necessary for achieving Good Ecological Status (GES) would have significant adverse effects on certain societal uses such as drinking water supply, irrigation, power generation or navigation, and those beneficial uses cannot be provided by different means. The key goal defined in the Water Framework Directive (WFD) on such waterbodies is to reach or respectively maintain Good Ecological Potential (GEP), through the application of mitigation measures.

1.1 Creation of the Fish Community Macrohabitat Types for European Rivers

The Fish Community Macrohabitat Types developed and presented in D2.1 underwent thorough internal review, which revealed considerable shortcomings of the model with regard to expected spatial distribution of identified types. For example, the model did not distinguish the Scottish Highlands, indicating that is insensitive to such changes. Therefore, the model has been revised according to the description below. Specifically, we included new classification of Environemental Zones to better take into account the climatic variability of Europe. The result is presented below.

The WFD specifies that for maximum ecological potential, "the values of the relevant biological quality elements [should] reflect, as far as possible, those associated with the closest comparable surface water body type, given the physical conditions which result from the artificial or heavily modified characteristics of the water body". For the good ecological potential (GEP), "there are slight changes in the values of the relevant biological quality elements as compared to the values found at maximum ecological potential." (WFD Annex V 1.2.5). According WFD Annex V, continuity, quantity and dynamics of flow, as well as morphology, are hydromorphological elements and as such, support the biological quality elements for the classifcication of water body status or potential.

In 2016 the EC Joint Research Centre (JRC) published a Technical Report on common understanding of using mitigation measures for reaching good ecological potential (Halleraker et al., 2016) providing an overview of the key measures used to mitigate hydromorphological alterations caused by water storage, and is linked to the WFD ecological impacts and mitigation measures outlined in the CIS reporting guidance (2016) (**Table 1**). The ECOSTAT report (2016) states that:

1. 'The free passage of migratory fish is a key requirement of the WFD, and may be used as an indicator for assessing whether water bodies are meeting Good Ecological Potential or Status. River continuum is explicitly mentioned in Annex V of the WFD, and even covers more than fish migration, but also other water related biota. WFD art 4 and Annex V on ecological potential have a special emphasis on ensuring ecological continuity. This is also a key conclusion from several CIS workshops on HMWBs.'

2. 'Quantity and dynamics of flow are crucial elements for the achievement of the WFD environmental objectives. The recognition that the hydrological regime plays a primary role in determining physical habitats, which in turn determines the biotic composition and support production and sustainability of aquatic ecosystems, is well documented. As the structure and functioning of aquatic ecosystems is largely depending on the flow regime, significant changes in flow characteristics with regard to magnitude, seasonality, duration, frequency, rate of change, and in intra-annual and inter-annual variability of the flow regime are likely to cause significant impacts on the ecology of water bodies. The natural flow regime of a river influences aquatic biodiversity via several interrelated mechanisms that operate over different spatial and temporal scales. Bunn & Arthington (2002) have proposed four principles for understanding how the flow regime influences the life cycle of aquatic populations' (**Figure 1).**

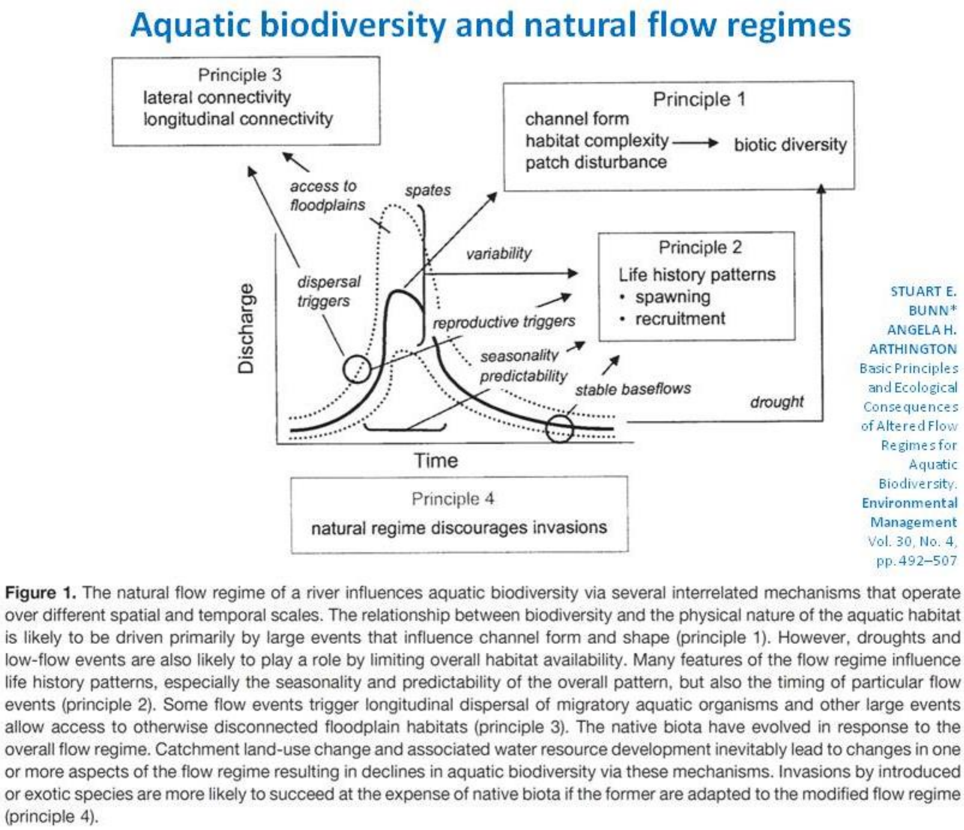


Figure 1. Four principles for understanding how the flow regime influences river biodiversity (from Bunn and Arthington, 2002).

3. ‘A natural river changes its morphology fundamentally when installing a dam or other barrier across a river. Permeability is still given for water and mostly for aquatic life by discharge elements and, if applicable, fish ladders. The extended cross-section upstream of a barrier leads to low current velocities and therefore to an increase in sedimentation, while erosion is minimized. Therefore, barriers often prevent sediment transport. Many reservoirs are affected by massive sedimentation and consequently a loss of storage volume (REFORM wiki).’

4. ‘Any transverse barrier to the flow in a river impounds water upstream. Barriers less than 10 m high may be called termed small dams. Barriers that are taller than 15 m are all termed dams. All of these barriers are used for retaining water for many purposes and the river is transformed into an impoundment upstream. Adjacent from reservoirs, the ponded rivers described here are mostly caused by smaller dams; in many cases rivers are altered not only by one, but by several impounded reaches. Natural flow velocity is reduced in these impoundments due to the presence of the dam, resulting in the deposition of transported sediments. In between the impounded reaches there are often free flowing sections in the water bodies with transitional zones between free flowing and ponded.’

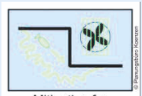
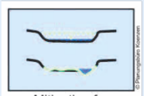
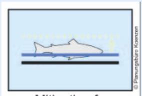
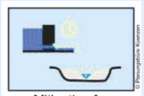
5. ‘Large dams with reservoirs might be built for several single use or multiple water uses such as hydropower, water supply (e.g. drinking water), flood protection, water regulation (e.g. low water elevation). The fluctuation of the reservoir level has an influence on the hydromorphological quality of the reservoir, and especially on the habitat quality for biota. A rapid draw down of the reservoir level can affect young fish and macroinvertebrates or cause

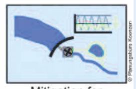
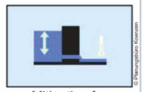
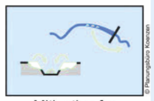
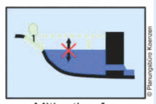
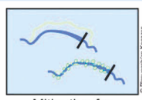
a total dry out of aquatic plants. Flat shore zones are in general the most important habitats affected by alteration in reservoir water levels.'

6. 'Water storage and river regulation may alter physical and/or chemical conditions downstream the water storage, with changes such as water temperature, supersaturation of nitrogen and altered patterns of ice formation in winter (mainly in alpine areas or in Northern Europe). Typically for deep reservoirs (not ponded rivers) the water temperature in a river stretch below the dam often increases in winter and decreases in summer due to deep-water intake in reservoirs, with decreased water-air contact of water.

   Ecological impacts: Altered composition or growth of macro invertebrate communities, fish and aquatic flora or increased fish mortality due to e.g. increased smolt age, due to slowed down migration to the sea or diving disease due to oversaturation while passing the barrage. Altered water temperature may lead to changed energetics /metabolism due to increased winter temperature (increased metabolism after regulation) and decreased metabolism and growth during summer. The time for hatching (fish & invertebrates emerge from eggs) may be significantly altered, increased larvae stage and suboptimal feeding conditions are also registered. Reduced ice cover, altered water quality, temperature and oversaturation with oxygen may also lead to behavioural impacts for fish, and reduced ice cover leads to changed light penetration with impact on aquatic vegetation.'

Table 1. Overview of the most widespread key measures used to mitigate hydromorphological alteration caused by water storage, related to main WFD and ecological impacts and mitigation measures in the CIS reporting guidance 2016 (from ECOSTAT, 2016).

Hydromorphological alteration	Main ecological impact*	Mitigation for	Mitigation measures options	Mitigation measures in WFD reporting guidance 2016	Pictogram
River continuity for upstream fish migration reduced/disoriented or interrupted	Fish: Populations of migratory fish absent or abundance reduced	**Upstream continuity for fish**	Ramp Fish pass By-pass channel Catch, transport & release *(Fish stocking from hatchery)*	Fish ladder* Bypass channels* Removal of structures	 Mitigation for upstream continuity for fish
River continuity for downstream fish migration reduced or interrupted	Fish: Populations of migratory fish absent or abundance reduced	**Downstream continuity for fish**	Fish-friendly turbines Fish screens By-pass channel Trap, transport & release Fish pass		 Mitigation for downstream continuity for fish
Artificially extreme low flows or extended low flows	Reduced abundance of plant & animal species. Alterations to composition of plant & animal species	**Low flow**	Provide additional flow River morphology changes	Setting of Ecological flows	 Mitigation for low flow
Loss of, or reduction in, flows sufficient to trigger & sustain fish migrations	Migratory fish absent or abundance reduced	**Fish flow**	Provide fish flow		 Mitigation for fish flow
Loss, reduction or absence of variable flows sufficient for flushing	Alteration/reduced abundance of fish & invertebrate species	**Variable flow**	Passive flow variability Active flow variability		 Mitigation for variable flow

Rapidly changing flows (including hydro peaking)	Reduction in animal & plant species abundance due to stranding & wash out	**Rapidly changing flows**	Balancing reservoir(s) (internal) Relocate tailrace Reduce rate Modify river morphology Balancing reservoir(s) (external) *(Fish stocking)*	Operational modification for hydropeaking* (only partly the same) Retention basins	 Mitigation for rapidly changing flows
Alteration of general physico-chemical conditions downstream (e.g. temperature, super saturation etc.)	River: Altered composition or growth of macro invertebrate communities and fish or fish mortality Lake: Impact on organic matter, primary production	**Physico-chemical alteration**	Flexible intake Multiple intakes Manage reservoir level		 Mitigation for physico-chemical alteration
River continuity for sediment disrupted or reduced leading to changes in substrate composition, disruption of morphodynamics in the ponded reaches (artificially stable river banks, disruption of lateral erosion processes)	Reduction in fish & invertebrate abundance & alterations in species composition Thermal changes Alteration or reduction in hyporheic species Alteration of self-purifying properties	**Sediment alteration**	Mechanical break-up of bed armouring Removal of sediment Re-introduce sediment (intake structures) Re-introduce sediment (reservoirs) Restore lateral erosion processes Introduce mobilising flows *(Fish stocking)*	Sediment management Removal of structures Restoration of bank structure Ecological flows Dredging minimisation Restoration of modified bed structure	 Mitigation for sediment alteration
Artificially extreme changes in lake level, reductions in quality and extent of shallow water & shore zone habitat	Reduction in abundance of plant & animal species. Alteration to species composition Alteration of spawning grounds and nursery areas Hydrological disconnection of wetlands	**Lake level alteration**	Reduce abstraction Increased inflows Create embayment(s) Manage shore/shallow habitats (renaturalisation) Connectivity to tributaries Artificial floating islands *(Fish stocking)*	Restoration of bank structure	 Mitigation for lake level alteration
Dewatered shore line and reduced river flow – ponded river	Alterations to plant & animal species composition (e.g. favouring disturbance-intolerant species/still water species) Barrier effect/disoriented fish migration	**Ponded rivers (impoundments)**	Bypass channel Reduce storage level In-channel habitat improvements Lateral reconnection	Bypass channels Habitat restoration Reconnection of side arms	 Mitigation for ponded rivers (impoundments)

* For certain types of mitigation, there is emphasis on fish as biological quality element according to GEP, especially according to continuity. Nonetheless, all relevant BQEs have to be taken into account for assessment of ecological potential and evaluation of measure effects.

In defining water body status or potential, we take into consideration biological quality elements (BQEs) that respond to hydromorphological processes (Southwood, 1977; Vannote *et al.,* 1980; de Jalón *et al.,* 2013 - REFORM project https://reformrivers.eu/). Agreement was reached at the CIS 2009 Workshop on HMWB that ecological continuum is a relevant consideration in defining GEP and that "There must be fish" (in particular, migratory species, since they are a good indicator of an ecological continuum). Providing a continuum for fish migration and preserving and restoring fish habitat is therefore a key consideration in achieving a GEP (ECOSTAT, 2016). For these reasons, fish formed the primary consideration in the design of the conceptual model. However, ECOSTAT (2016) also states that fish are only one biological quality element (BQE) in assessing the good ecological potential (GEP) of the HMWB, even if, when considering river continuity and therefore fish migration, they are the most evident element. Therefore, in the conceptual model developed within this study, fish are treated as an "umbrella" group for all other BQE, as they are most sensitive to river habitat and continuity alterations.

1.2 General overview and catchment perspective

A catchment perspective is essential for developing a conceptual model of the ecological impacts of barriers at a European scale. Many natural and anthropogenic interactions occur within a catchment area. Therefore, when predicting the impact of barriers at the catchment level, we need to consider many variables and interactions between processes across different scales, including water, biodiversity, ecosystem resilience, ecosystem services and cultural heritage (WBSRC - **W**ater, **B**iodiversity, Ecosystem **S**ervices, **R**esilience, **C**ultural Heritage - Zalewski, 2014). This results in a bi-directional consideration of dependencies and interactions: BOTTOM-UP and TOP-DOWN (fish on the top) in given geographical situation (**Figure 2**).

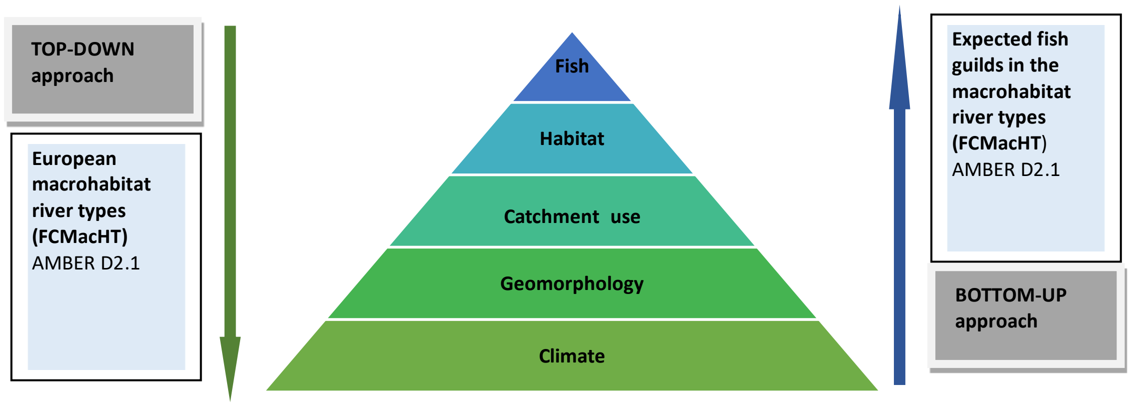


Figure 2. A general scheme for the Conceptual model of ecological impacts of barriers in EU considering habitat selection criteria for running waters (Zalewski *et al.*, in prep.).

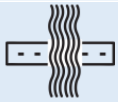
The conceptual model presented here uses this framework to identify the relationship between barriers and fish habitat availability at the regional scale. Each approach has its own starting point.

1. The BOTTOM-UP approach starts with the typology of fish community macrohabitat types (FCMacHT) of European rivers presented in D2.1. and modified in D2.2. **(Figure 24).** This typology is based on a combination of physio-geographic factors associated with Expected Fish Communities (EFC), which create a solid foundation for describing broad landscape scale influences on biota. Interactions are presented for each FCMacHT all the way to the fish response.

2. In the TOP-DOWN approach, fish assemblages are considered good indicators of the environmental state of the river as well as of riverine habitat suitability and availability (Pont *et al.,* 2006, Schmutz *et al.,* 2007, EFI+ Consortium, 2009). There are several advantages of using fish as indicator organisms (Fausch *et al.,* 1990; Harrison and Whitfield, 2004) and they can be regarded as an "umbrella group" for other aquatic organisms. The habitat requirements of fish in determining fish habitat guilds as presented in D2.1 and modified in D2.2 **(Figure 25)** aid in identifying suitable habitat and therefore, in predicting changes in fish community structure specific to FCMacHT. By scaling the process up and taking into account the influence of catchment use, geomorphology and climate change; we can extrapolate from ecological group level to the regional scale (Parasiewicz 2003).

The classification of river barrier types (**Table 2**) forms a further building block of the conceptual model. It is expected that each barrier type will have a particular impact on aquatic fish communities. This classification is adopted as a basic approach in the entire AMBER project – see Deliverable 1.2.

Table 2. Main types of river barriers identified in the AMBER project.

	BARIER DESCRIPTION	PICTOGRAM
1	**Dam** - a barrier that blocks or constrains the flow of water and raises the water level with permanent ponding.	
2	**Weir** - a barrier aimed at regulating flow conditions and water levels.	
3	**Sluice** - a movable barrier aimed at controlling water levels and flow rates in rivers and streams.	
4	**Culvert** - a structure aimed at carrying a stream or river under an obstruction.	
5	**Ford** - a structure in a river or stream which creates a shallow place for crossing the river or stream by wading or in a vehicle.	
6	**Ramp** - a ramp or a bed sill is a structure aimed at stabilizing the channel bed and reducing erosion; it is recognizable by its stairway-like shape.	

1.3 The impact of barriers on fish

The fragmentation of habitats by barriers is one of the five main factors of biodiversity loss (Baudoin *et al*., 2014). Currently available methods and technical solutions do not encompass the full range of aspects of "ecosystem functioning", i.e. availability of suitable habitat for particular species and functional guilds. Novel assessment protocols and solutions for improving riverine continuity are therefore needed in order to ensure river ecosystem functioning and maintenance of fish populations. The concept presented in the deliverable goes beyond that in the sense that it considers the effect of barriers (with mitigation measures taken) on habitat availability for functional fish guilds. This approach can be incorporated into guidelines for river-basin management plans (RBMP) (SNIFFER, 2010ab; Baudoin *et al*., 2014; AMBER D1.1).

1.3.1 Dams and weirs

Dams (**Figure 3**) and large weirs (**Figure 4**), can impact river continuity, habitat and biota to a greater extent than any other barrier type.

Figure 3. Examples of dams (photos: I.Wagner, ERCE; K. Suska, SSIFI, Poland).

Figure 4. Examples of overflowing weirs.

The impacts of barriers and damming on fish and river ecology is well documented, especially for large dams (height >15 m), and more recently for weirs (Petts, 1984; Welcomme, 1985, 2001; WCD 2000). The main impacts include:

1. Interruptions in connectivity

Channel blocking

Extirpation of diadromous fish (anadromous, catadromous) and reduction in abundance of potamodromous fish. Connectivity may improve with the use of fish passes (**Figure 5**).

Figure 5. Fishpass on the Włocławek dam – the attraction flow is visible at the entrance of the fishpass (photo: K. Suska, SSIFI, Poland).

Ponding

- Changes in community structure with shifts from rheophilic to limnophilic species due to the conversion of the lotic environment into lentic one (reservoir).
- Changes in community structure due to an increase in planktivorous fish caused by greater plankton abundance.
- Changes in community structure due to the introduction of pond and exotic species.
- Food base alteration due to changes in the plankton and macroinvertebrate communities.

2. Changes in morphology

Flood prevention by dams and levees

- Fish spawning and growth reduction due to loss of floodplain area and associated habitats (oxbow lakes etc.).
- Reduction in fish diversity due to loss of habitat diversity.
- Changes in species composition due to loss of obligatory floodplain spawners.
- Decrease in productivity of whole river system.

River channelisation below dam

- Changes in abundance and loss of species.

3. Changes in the flow pattern

Temporal changes

- Spawning pattern, spawning and breeding success disruption – changes in community structure from seasonal spawners to more flexible ones due to unnatural flow patterns.
- Community productivity and biodiversity reduction due to a shift from pulse regulated to stable river system dynamics.
- Fish mortality increase due to changes in flushing rates resulting in the accumulation of toxic contaminants in the water below the dam.

Changes in velocity

- Shifts in species composition in the tail water with an accumulation of rheophilic predators due to an increase in flow rate in the channelised stretch just downstream from the dam. Loss of appropriate sites for colonization of young drifting fish.
- Shifts in species composition from rheopilic to lentic communities due to a decrease of current velocity in the reservoir upstream and in controlled reaches downstream of the dam.
- Reproductive success disruption (decline of lithophils and psammophils) due to the flooding of nesting sites, stranding nests and eggs, drowning of developing vegetation and a reduction in the development of food base organisms caused by a rapid filling or drawdown of the reservoir.

4. Water quality changes

Siltation pattern changes

- Spawning success disruption of lithophils and psammophils due to silting of spawning substrates. Changes in community structure usually toward phytophils due to the development of vegetation in the reservoir.
- Changes in community structure towards an increase of illiophages due to an increase in siltation in the reservoir.
- Changes in community structure with a reduction of non-visual predators and omnivores due to a decrease in siltation below the dam.
- Changes in fish community structure with a reduction of illiophages and an increase in benthic limnivores due to a lack of siltation downstream of the dam.

Water temperature changes

- Shift in spawning time and success for both cold and warm water spawners due to an increase in temperature variation.
- Difficulties of passage for migrating species due to stratification in the reservoir.
- Elimination of fish in deoxygenated hypolimnion of the reservoir.
- Fish mortality downstream of the dam due to the flushing of anoxic waters and hydrogen sulphide, when bottom discharge from a thermally stratified reservoir takes place.

Water quality changes

- Increased fish mortality, reduced growth, suppression of breeding and disappearance of food organisms due to water pollution and toxic algal blooming (**Figure 6**), especially in high retention time reservoirs.

Figure 6. Toxic cyanobacterial bloom in high retention time (average 30 days) Sulejow Reservoir, Poland (photo: M. Tarczyńska, ERCE Poland)

5. Operational effects

Water uptake

Damage to fish by turbines and pumps (**Figure 7**), loss of fish at downstream migration, especially adult potamodromous fish and eel, but also juvenile stages of anadromous salmonid species (smolts).

Figure 7. Vimba (*Vimba vimba*), an anadromous fish damaged by a turbine while crossing the Włocławek Dam, Poland – AMBER case study (photo: Z. Kaczkowski, ERCE Poland)

1.3.2 Sluices

Sluice gates (**Figure 8**) are used to change water levels and can block rivers temporarily; they have similar impacts as dams and weirs, but with adequate management their impact can be reduced. For example, they can be used to assist fish production in floodplain river systems through periodic inundation (Junk *et al.,* 1989; Tockner *et al.,* 2000). Without adequate management, they can cause several types of impacts. For instance, rapid flow fluctuation may lead to the drying up of fish habitat or washing out of sensitive life stages or species downstream. Especially common and damaging are flow releases for shipping or reservoir clean-up during low flow periods.

Figure 8. Examples of sluice gates (photos: Z. Kaczkowski (ERCE; Polish Waters Warsaw).

1.3.3 Culverts

Culverts (**Figure 9**) are the most economically feasible road crossings, but are potentially damaging to river morphology, hydraulics and biota. Culverts can have negative effects on migratory and resident fish populations, and may hinder fish passage due to high water velocities, low water depth, lack of shelter, high outflows and debris jams (Roni et al., 2002; Kemp and Williams, 2008). This results in higher fish energy expenditure during upstream migration and increased predation, angling mortality and disease in deeper downstream areas below barriers. Individuals that are unable to overcome barriers may be forced to spawn in less suitable habitats downstream (e.g. under higher risk of siltation or predation of eggs and larvae), limiting population size (Garcia de Leaniz, 2008). Particularly problematic are purged culverts, where the downstream outflow is located above the stream bottom.

Figure 9. Examples of culverts (photos: Z. Kaczkowski (photos: AMBER).

Culverts and other barriers can also degrade fish habitat by altering or limiting the downstream movement of sediment, woody debris, and organic materials. Some culverts and bridges can be modified to provide adequate adult fish passage at road crossings, but not all can provide passage for juvenile fishes, or maintain sediment and wood transport, and many affect channel morphology. Bridges often allow the passage of other materials and the formation of a natural stream channel but are costly. Open-bottom culverts or embedded (e.g., countersunk) pipe-arch culverts allow a natural substrate to form within the channel and are effective at passing both juvenile and adult salmonids (Furniss *et al.,* 1991; Clay 1995; Roni *et al.,* 2002).

1.3.4 Ramps

Ramps can be designed to provide suitable conditions for fish passage in low-flow channels. Rock ramps **(Figure 10**) are an alternative to weirs, and use rip-rap to make a steep, immobile reach that creates sufficient water depth, maintains fish passage, controls discharge capacity and improves sediment continuity (Kitchen *et al.,* 2016; CIRIA C763, 2016). However, a poorly designed block ramp cannot provide longitudinal connectivity and migration opportunities for all the fish that might occur in the stream. Some crevices in the side zones of the ramp could form a part of the migration corridor, but only for small and medium-sized fish (Plesiński *et al.,* 2018).

Figure 10. Examples of ramps (photos: Z. Kaczkowski (photos: SSIFI; AMBER).

1.3.5 Fords

The impact of fords (**Figure 11**) on biota, especially on migratory fish, has only recently started to be investigated. Fords may impede the migration of native fish. In some cases, the removal of unnecessary fords or improved ford design can help remedy the problem (Williams *et al.,* 2005).

Figure 11. Examples of fords (photos: Z. Kaczkowski (photos: SSIFI; AMBER).

More information concerning the impacts of other barriers on fish can be found in Appendix A.

1.4 Barrier impact on biota other than fish

1.4.1 Dams

In all the cases reviewed in the APEM project (http://www.apemltd.co.uk) invertebrates respond to the combination of water quality, hydraulics and changes in morphology below impoundments. Due to thermal and chemical alterations, deep, eutrophic reservoirs present the greatest risk to downstream invertebrates. In non-eutrophic reservoirs, the greatest risk to downstream invertebrates is from cooler summer temperatures that disrupt development and life cycles. Increased nutrients released from the bottom of large impoundments of eutrophic reservoirs increase the coverage and taxon richness of submerged macrophytes, and raise the coverage and taxon richness of algae. Algae derived from nutrient-rich reservoirs can also show the phytobenthos characteristics of a eutrophic river (Petts, 1986).

Low head and run-of-the-river dams can have small positive and negative effects on macroinvertebrate abundance, and negative effects on macroinvertebrate richness downstream, respectively (Tiemann *et al.*, 2004; Martínez *et al.*, 2013; Wang *et al.*, 2013). These patterns arise as tolerant species such as dipterans are favoured over more sensitive plecopteran species (Camargo, Alonso and De la Puente, 2005). The increase in the richness of tolerant species can mask decreases in overall richness following the loss of more sensitive species (Dean *et al.*, 2002). These changes are often brought about through changes in flow velocity and subsequent changes to sediment habitat structure (Tiemann *et al.*, 2004). Additional ephemeral changes in flow velocity such as hydropeaking negatively impact macroinvertebrate communities. Both single and consecutive hydropeaking events caused by dam releases reduce abundance downstream and increase the drift of macroinvertebrates, causing community shifts (Lauters *et al.*, 1996; Harby and Halleraker, 2001; Céréghino *et al.*, 2004; Gillespie *et al.*, 2015).

Downstream of dams, a reduction in macrophyte abundance and richness occurs due to a combination of scouring during hydropeaking and increases in water temperature and pH taking place in dam reservoirs (Bernez *et al.*, 2004; Belmar *et al.*, 2013). Whilst impoundments upstream of dams create hydrological conditions which increase local coverage of riverbank and aquatic macrophytes (Moura *et al.*, 2013; Ceschin *et al.*, 2015).

Impoundments may also affect aquatic birds, promoting, for example, large cormorant colonies which have a big effect on fish populations. Flow fluctuations may lead to the destruction of nests of birds breeding on sand and gravel banks and islands.

1.4.2 Weirs

Weirs can have both positive and negative effects on various groups of macroinvertebrates upstream and downstream. Lotic conditions immediately downstream favour the abundance and richness of shredders, while lentic conditions upstream favour filter-feeding collectors. However, the same conditions upstream reduce the abundance of benthic and habitat structure-specialists by clogging interstitial spaces in stream beds (Lenat *et al.,* 1981; Mueller *et al.,* 2011). The same studies also noted that there was no change in macrophyte abundance or richness due to weirs, whilst others observed poorer habitat structure and a reduction in the presence of macrophytes in impacted reaches (Benejam *et al.*, 2016).

The aquatic stages (nymphs and larvae) of some stream insects only move short distances upstream, with just a few capable of covering longer stretches (1-2 km). The ability of most adult aquatic insects to fly upstream can compensate for their restricted movement at the aquatic stages, and reduce the impact of barriers (Dillon, 1988; Vaughan, 2002).

Unlike insects, other aquatic macroinvertebrates (molluscs, crustaceans, worms, mites, and other macrozoobenthos) have no aerial life stage; barriers can therefore have a bigger impact on these taxa. Some freshwater mussels disperse within a river system as larvae (glochidia) attached to the gills of suitable hosts, usually fish. If fish hosts cannot move upstream, freshwater mussels cannot colonize or recolonize new, suitable habitat (Dillon, 1988; Vaughan, 2002).

1.4.3 Sluices

The operation of barriers controlling water levels have implications for taxa reliant on being submerged (Capers, 2003) and on the complicated dynamics of submerged macrophytes found in tidal environments (Madsen *et al.*, 2001). Periodic reductions in water levels, controlled by tidal sluices, can have catastrophic effects for macrophytes, macroinvertebrates, and molluscs inhabiting the littoral zone (Richardson *et al.,* 2002).

1.4.4 Ramp-bed sills

In the majority of cases, barriers constructed using materials similar to natural river features, such as wood, boulders and gravel, do not influence macroinvertebrate abundance or richness when placed in the stream channel (Tikkanen *et al.*, 1994; Hilderbrand *et al.*, 1997; Laasonen *et al.,* 1998; Larson *et al.,* 2001; Brooks *et al.*, 2002; Roni *et al.*, 2006). However, rock ramps similar in size to low head weirs support lower macroinvertebrate and macrophyte densities than reference reaches (Scrimgeour *et al.*, 2013), highlighting the importance of barrier head height in contributing to impacts on macroinvertebrates through their control on sedimentation and flow velocity.

Additional negative or positive impacts of sluices, ramp-bed sills, fords and culverts are likely to be similar to the impacts of low head dams and weirs with similar sized impoundments and head drops. However, studies on macroinvertebrates and macrophytes in systems impacted by these structures are lacking.

Barrier impacts on phytobenthos and plankton have been reported for dams, but little is known for smaller barriers (Growns and Growns, 2001; Wu *at al.,* 2010).

As stated in the ECOSTAT report (2016), fish cannot be considered as the only biological quality element (BQE) when assessing a WFD good ecological potential (GDP) of heavily modified water bodies (HMWB), even if the barrier impact on fish is highest. Therefore, we also need to consider further research of other WFD BQEs such as macroinvertebrates, macrophytes and phytobenthos (ECOSTAT, 2016). Barrier impacts on macroinvertebrates, macrophytes and other taxa important for WFD implementation and water quality assessment biota is included in Appendix A.

1.5 Cumulative environmental effects of barrier impacts

River barriers, especially when there is damming, interfere with the natural hydrological cycle and produce an unpredictable chain of effects that might be detectable at local, regional and even global scales. Dam construction and associated water diversion, river channelization and interbasin water transfer have environmental effects at the global-scale (WCD, 2000).

The cumulative effects of hydrological alterations associated with dam and reservoir development can be summarized as follows (Rosenberg *et al.*, 2000):

- Creation of new storage capacity within the catchment hydrological cycle (Petts, 1984);
- Alteration of natural flows of water and sediments and seasonal patterns of river discharge (Varosmarty and Sahagian, 2000);
- Changes in ecosystem level-processes: nutrient cycling and primary productivity (Pringle, 1997; Rosenberg et al., 1997), biogeochemistry of downstream, offshore areas (Ittekkot et al., 2000);
- Fragmentation of river habitat (Dynesius and Nilsson, 1994) and associated fauna (Dudgeon, 2000; Pringle et al., 2000);
- Deterioration and loss of floodplains and riparian zones downstream of the dam (Nilsson and Berggren, 2000);
- Deterioration and loss of river deltas and ocean estuaries (Rosenberg et al., 1997) and reduction in sea-level (Chao, 1995);
- Deterioration of irrigated terrestrial environments and associated surface waters (McCully, 1996);
- Dewatering, eutrophication, pollution and contamination issues (Postel, 1998; Zalewski, 2000, 2002);
- Methylmercury contamination of food webs, due to changed microbial activity on flooded areas (Kelley et al., 1997);
- Cyanotoxic contamination of reservoir and river water and trophic levels (Zalewski, 2000);
- Genetic isolation as a result of habitat fragmentation (Pringle, 1997; Neraas and Spruell, 2001);
- Hybridization and speciation effects on fish (Balon, 1992);
- Impact on biodiversity (Rosenberg et al., 1977; Master et al., 1998);
- Destruction of fish habitat and populations, complemented by fisheries decline (Petts, 1984);
- Acceleration of greenhouse gases (CO_2, CH_4) emission from reservoirs (Kelley et al., 1997; Rosenberg et al., 1997) contributing to global climate change (St.Louis et al., 2000);
- Increase of epidemics (tropical dams) (Jobin, 1999);
- Change in the earth's rotation speed, magnetic field, earthquakes (Chao, 1995).

2 MATERIALS AND METHODS

2.1 Assumptions for the Conceptual Model

As shown above, barriers can have multiple effects on riverine habitats, including habitat fragmentation, flow fluctuations, changes in hydrodynamics and hydro-morphology features, substrate composition, physio-chemical water parameters, aquatic vegetation, and cover. These may have different consequences for habitat availability and suitability. Our first step in developing a conceptual model of barrier impacts on aquatic habitats was to establish several analytical background assumptions:

1) Fish assemblages are considered good indicators of the environmental state of rivers and of riverine habitat suitability and availability (Pont *et al.,* 2006; Schmutz *et al.*, 2007; EFI+ Consortium, 2009). Fish present several advantages as indicator organisms (Fausch *et al.*, 1990; Harrison and Whitfield, 2004), and are regarded as an "umbrella group "for other aquatic organisms. Relatively long-lived and present in almost all lotic ecosystems, fish reflect the cumulative effects of long-term anthropogenic stressors. They represent the higher levels of the trophic pyramid and respond to changes in other groups such as benthic invertebrates

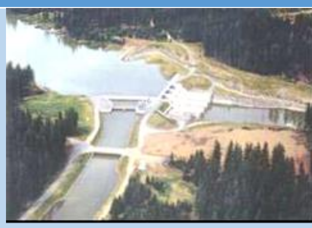
and zooplankton, a food resource for non-predatory species. Some fish species also feed on algae and higher aquatic plants, but macrophytes are mostly important as habitat structural elements and spawning substrate. Due to their high mobility, fish use various habitats within

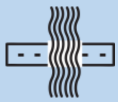
river ecosystems, so they are particularly sensitive to disturbances in river morphology (Schinegger *et al.*, 2011). As the only riverine organisms that actively migrate over long distances, fish are also strongly affected by river continuum disturbances.

2) The conceptual model of barrier impacts on rivers is built on the European macrohabitat river types (FCMacHT; D2.1, modified in D2.2 **Figure 24**), which was based on a number of physio-geographic factors associated with Expected Fish Communities (EFC). Fish species numbers in European inland waters is limited (250 – 300 species), and in most cases they are easy to identify taxonomically to species level. However, the species composition varies greatly between biogeographic regions. Therefore, a more useful approach is to assess fish assemblage responses to riverine habitat changes caused by barrier impacts on a continental scale using macro habitat guilds.

3) The barriers assessed are assumed to be state of the art and represent good ecological potential (GEP) (**Table 3**) i.e. ramps are constructed out of rocks, culverts are not purged, dams have fish passage facilities and mitigation measures have been implemented.

Table 3. Description barriers used in Conceptual Model where HMWB is able to achieve a GEP.

BARIER DESCRIPTION		PHOTO	PICTOGRAM
1	**Dam** - a barrier that blocks or constrains the flow of water and raises the water level. **Typical Requirements for GEP:** **Fish passage:** from the well designed nature-based way - with a design based on simulating natural stream characteristics, using natural materials and providing suitable passage conditions, as well as additional habitat over a range of flows for a wide variety of fish species and other aquatic organisms (Wildman *et al.,* 2003), to a properly functioning technical pass with adequate size, slope, water velocity prameters etc, allowing for passage of all fish species characteristic for the river community.	  	

	BARIER DESCRIPTION	PHOTO	PICTOGRAM
2	**Weir** - a barrier aimed at regulating flow conditions and water levels. **Typical Requirements for GEP:** **Fish passage** – from a complex, nature-based bypass channel to a properly functioning technical fish ladder (see abiove).	 	
3	**Sluice** - a movable barrier aimed at controlling water levels and flow rates in rivers and streams. **Typical Requirements for GEP:** Frequently open, does not block upstream migration when open if the sluice is not built on an additional weir.	  	
4	**Culvert** - a structure aimed at carrying a stream or river under an obstruction. **Typical Requirements for GEP:** Connected to river bed and substrate, with the water depth and velocity passable all year.		
5	**Ford** - a structure in a river or stream that creates a shallow crossing place. **Typical Requirements for GEP:** Water depth must guarantee fish passage all year.		

	BARIER DESCRIPTION	PHOTO	PICTOGRAM
6	**Ramp** - a ramp or a bed sill is a structure aimed at stabilizing the channel bed and reducing erosion, it is recognizable by its stair-like shape. **Typical Requirements for GEP:** Space between stones must guarantee fish passage all year.		

Photos of barriers (from top): DAM: 1- EZB cited in Wildman *et al., 2003* (Fishing, Austria, River Mur, A step-pool bypass channel); 2 - J. Ligięza (SSIFI); 3 - K. Suska (SSIFI); WEIR: 4-5 – Z. Kaczkowski (ERCE); SLUICE: 6- AMBER, 7- Z. Kaczkowski (ERCE), 8 – Polish Waters, Warsaw; CULVERT: 9 – AMBER; FORD: 10 - Z. Kaczkowski (ERCE); RAMP: 11 - AMBER.

2.2 Description of habitat attributes

To assess changes in fish habitat availability caused by the impact of barriers we identified a list of 21 habitat attributes that can be modified by various barrier types:

1. **high velocity** – decrease of water velocity and therefore share of lotic habitats such as rapids, riffles and fast runs, loss of suitable areas for rheophilic species.
2. **low velocity** – enlargement of slack water areas and share of lentic habitats, such as pools and backwaters, increase in areas suitable for limnophilic species.
3. **deep areas** – increase of water depth caused by impoundment, increase of areas suitable for pelagic species.
4. **shallows** – decrease of shallow water areas like margins of hydro-morphological units or shallow backwaters, loss of nursery habitats for fry and habitats for small benthic species.
5. **interstitial space** – loss of coarse substrate structure caused by siltation, decline of spawning areas for lithophilic and speleophilic fish, loss of habitat for small benthic species and fry.
6. **sand** – increase of sand share in sediments, improved habitat conditions for psammophilic species.
7. **mud** – increase in share muddy sediments, improved habitat conditions for stagnophilic fish, generalists and benthic species feeding in soft bottom sediments.
8. **gravel** – decrease of gravel share in bottom substrate, loss of spawning areas for lithophilic and speleophilic fish, decline of habitat for benthic species preferring gravel bottom, change in macroinvertebrates assemblages causing decrease of fish feeding base.
9. **boulder** – loss of structural habitat elements and cover for fish in impounded areas, decrease of habitat diversity and diminished suitability for rheophilic and benthic fish species, especially for large specimens.
10. **woody debris** – loss of structural habitat elements and cover for fish in impounded areas, decrease of habitat diversity and diminished suitability for rheophilic, benthic and water column fish species, especially for large specimens.
11. **oxygenated water** – decrease of water aeration in impounded areas, threat for highly rheophilic and intolerant species.
12. **cold water** – increase of water temperature causing less suitable conditions for stenothermic fish species.

13. **low trophic level** – increase of nutrient load and accumulation in impoundment causing changes towards eutrophy – threat for intolerant and indicator species;
14. **rheophilic macrophytes, mosses** – loss of valuable macrophyte taxons as structural habitat elements, cover and feeding grounds for rheophilic fish in impounded areas.
15. **macrophytes** – increase of submerged, nymphaeid and emerged, stagnophilic macrophytes – habitat gain for phytophilic and limnophilic fish species.
16. **canopy shading** – decrease of shaded water surface, loss of less exposed areas for fish and increase of water insolation level and temperature – threat for stenothermic species.
17. **overhanging vegetation** – decrease of shaded and covered water areas, loss of cover for fish some increase of water insolation level and temperature – diminished suitability for stenothermic species.
18. **undercut bank** – loss of structural habitat elements and cover for fish in impounded areas, decrease of habitat diversity and diminished suitability for rheophilic, benthic and water column fish species, especially for large specimens.
19. **floodplain accessibility** – altered connection between river bed and floodplain waterbodies, loss of spawning, nursery and feeding habitat for limnophilic and phytophilic fish species.
20. **habitat continuity** – direct effect of barrier on river continuity alteration and fish migration routes, causing local population fragmentation, loss of spawning grounds accessibility and decline of diadromous species.
21. **habitat stability** – change in natural flow pattern, sudden changes in flow conditions and water level, loss of stable accessible habitat, higher frequency of extreme events caused by barrier operation.

The goal of attribute selection is to assess the potential impact of different barrier types on riverine habitat in relation to specified fish macro-habitat guilds in an expected fish community. However, some of those criteria have synergistic effects – e.g. increase of depth results in velocity reduction and sediment granulation change. On the other hand, pairs of listed attributes may have contrary effect – e.g. decrease of high water velocity habitats and increase of lentic areas, an increase of mud in bottom sediments can cause a decline in coarse substrate.

The attributes listed above frequently have opposing effects on different fish guilds (rheophilic, limnophilic, benthic, water column species). Current scientific knowledge allows for the estimation of the magnitude of change of particular attribute caused by various barrier types (dam, weir, sluice, culvert, ford, ramp), helping us to assess the impact on fish community structure.

2.3 Creation of the Fish Community Macrohabitat Types for European rivers

The Fish Community Macrohabitat Types developed and presented in the deliverable 2.1 underwent thorought internal review, which revealed considerable shortcomings of the model with regard to expected spatial distribution of identified types. For example, the model did not distinguished Scottish Highlands indicating that is insensitive to such changes. Therefore, the model has been revised according to the description below. Specifically, we included new classification of Environemental Zones to better take into account the climatic variability of Europe. The result is presented below.

2.3.1 Data source

The European Intercalibration (IC) database consists of of fishery data gathered during the intercalibration process conducted between 2006-2011 under the auspices of the European Commission Joint Research Centre – JRC (WFD Intercalibration 2011). It contains 19 tables with data and metadata for 4561 sites. Fish samples were taken with electrofishing between 1958 and 2008. The following countries represented are Austria, Belgium (Flanders and Wallonia), Czech Republic, Germany, Denmark, Spain, Estonia, United Kingdom (England and Wales, Scotland, Northern Ireland), Finland, France, Greece, Ireland, Lithuania, Luxembourg, Latvia, Netherlands, Norway, Portugal, Romania, Sweden, Slovenia and Slovakia.

This database was complemented with the data from Poland (938 fished sites sampled between 2011 and 2015), used with the permission of Chief Inspectorate for Environmental Protection, Poland. The total dataset available was 5499 fished sites, including 1315 not disturbed sites.

There was a requirement for individual permissions for data use, which caused some difficulties. The majority of the EU countries (19) granted permission. Germany, France, Denmark and Greece did not respond to data use requests, and thus data from those countries were not used. Romania refused to permit use of its data within the AMBER project and thus was also completely excluded from analyses.

A subset consisting of 1099 sites classified as representing low anthropogenic pressure (NDS) was used for FCMacHT model calibration. The metadata consist of selected physical variables for each site as well as relative proportions of each captured species in the sample. The selected physical variables of IC are as follows (**Table 4**):

- Geomorphic river type
- Size of catchment
- Altitude
- Geological typology
- Actual river slope
- Natural sediment
- Wetted width
- Floodplain connectivity (E_floodplain)
- Intercalibration Region (IC_Region)
- Stream Order (Strahler_SO)

However, the Intercalibration data set is limited to only a portion of European rivers, i.e. the above attributes are not available at the same level of accuracy for the remaining portion of European waterbodies. Our goal is to create a predictive map of macrohabitat distribution across Europe. To supplement missing information, we searched for an equivalent of the above physical variables in public pan-European datasets (**Table 4**).

Table 4. Physical variables of Intercalibration dataset (IC), their relative importance factor estimated by CART and unified new physical variables used for CART analysis.

IC Variable	Description	Importance	New physical variablee	Description
Actual.river.slope	Slope of the river (‰) measured on a stretch of length corresponding to catchment size: 1 km (for	15	W_SLOPE	Slope of the river topological segment derived from the CCM2 database (‰). Topological segment is a river portion inbetween junctions with another river, source or estuary

IC Variable	Description	Importance	New physical variablee	Description
	catchment area up to 100 km^2), 5 km (catchment area 100-1000 km^2) and 10 km (catchment area above 1000 km^2)			
Altitude	Altitude of a site (m above sea level)	14	W_ALT	Mean altitude of the river segment as a mean altitude between the altitudes of its beginning and its end (m a.s.l.), derived from the CCM2
Size.of.catchment	Size of the river catchment above the site (km^2)	13	W_CATCH	Size of catchment of a river segment above it calculated as a sum of the immediate catchment of the river segment and all catchments above the given river segments derived from the CCM2 database
IC_GROUP	Intercalibration regional group: Nordic, Lowland-Midland, Alpine-type Mountains, Mediterranean South-Atlantic, Danubian (WFD Intercalibration, 2011)	12	EnZ	The following Environmental Zones of Europe were used: Alpine North, Boreal, Nemoral, Atlantic North, Alpine South, Continental, Atlantic Central, Pannonian, Lusitanian, Anatolian, Mediterranean Mountains, Mediterranean North, Mediterranean South (Metzger, 2018). However, not all zones had reprezentatives in NDS.
Geological.typology	Calcerious, Silicious, Organic type of prevailing geology derived from available geological maps	9	Z_GEO3	Geological classification derived from European Soil Database v2.0 (ESDB v 2.0; Panagos *et al*. 2012) and International Hydrogeological Map of Europe 1:1,500,000 (IHME1500, 2018) classified into three geological types: calcareous, silicious, organic (Table 5, Table 6).
Wetted.width	Average river width derived from several	9	Attribute not considered	

IC Variable	Description	Importance	New physical variablee	Description
	measurements at a given site			
Strahler.stream.order	Stream order according to Strahler (1957) – small streams has order "1", when two of them meet – the river has order "2", when two rivers of the order 2 met – the order growth to "3" etc.	8	W_STRAHLER	Strahler stream order classification derived from CCM2 database
Geomorph.river.type	Information in 5 categories to be selected, data based on field observations and maps, for river segment including sampled site: Naturally constraint no mob, Braided, Sinuous, Meander regular, Meander tortous.	7	Attribute not considered	
Natural.sediment	Granulation of naturally predominant sediment in 3 classes: 1- coarse (rocks, boulders, stones, gravel), 2 – medium size (sand), 3 – small (mud, clay, peat).	6	Attribute not considered	
E_water_source_type	Predominant source of water: pluvial, pluvio-nival, glacial, groundwaters	5	Attribute not considered	
E_floodplain	Presence of the floodplain (in	4	Attribute not considered	

IC Variable	Description	Importance	New physical variablee	Description
	natural, non-disturbed conditions)			

Each fishing site from the IC dataset is given an attribute value of the river segment it belongs to, based on the Catchment Characterisation and Modelling River and Catchment Database, version 2.1 (CCM2, Vogt *et al.* 2007). The following physical river and catchment characteristics are chosen:

- size of catchment,
- Strahler stream order,
- slope of the river segment,
- mean altitude of a river segment.

The Environmental Stratification of Europe (Metzger, 2018) is used to assign an environmental zone to the IC sites and consequently to all European river networks. The following environmental zones were used:

- Alpine North,
- Boreal,
- Nemoral,
- Atlantic North,
- Alpine South,
- Continental,
- Atlantic Central,
- Pannonian,
- Lusitanian,
- Anatolian,
- Mediterranean Mountains,
- Mediterranean North,
- Mediterranean South (**Table 4**).

Calcerious, silicious and organic types of prevailing geology were derived from available geological maps, i.e. European Soil Database v2.0 (European Soil Data Centre; Panagos et al., 2012) and the International Hydrogeological Map of Europe 1:1,500,000 (IHME1500, 2018). The IHME1500 has been used to classfy the bedrock into silicious and calcerious types based on the lithological formations classification presented therein (**Table 5**) The ESDC data were used to obtain the most accurate available organic substrate data across Europe and superimposed on the silicious and calcareous data. We also note that “The peatland map of Europe” (Tanneberger *et al*., 2017) has been created, which to date seems to be the most accurate map of peatland accross Europe. This map would complement the areas where ESDC data are missing. However, reuse of the data for AMBER, neither in vector nor in raster for Europe in its entirety, does not seem feasible (pers. com. F. Tanneberger, 03.2019). Therefore, the general attitude towards obtaining the organic soil dataset was to use the ESDC data with the custom rules for selection (**Table 6**). If at least one attribute value listed in **Table 6** was true it, was classified as organic type substrate. We compare these results with those of Tanneberger *et al*. 2017 in **Figure 12**. It is apparent that not all regions are equally well mapped with organic soli information data. The best representation seems to be for Central and Eastern-European countries as well as for Ireland, the United Kingdom, Finland and Alpine regions. The least adequate organic type substrate mapping results were obtained for Scandinavia (Sweden and Norway) as well as South and West European countries (**Figure 12**) as compared to the European peatland map (Tanneberger et al. 2017).

Table 5. IHME1500 v.1.1 lithological descriptions and their classification into silicious and calcareous geological types used for this research.

Silicious geological type	Calcareous geological type
Calcarenites and sands	Chalkstones and marls
Clays	Chalkstones, limestones (jointed, karstified)
Clays and claystones, marlstones	Clays and dolomitic limestones
Clays and claystones, sandstones, conglomerates	Clays and limestones, sandstones
Clays and shales (combustible)	Clays, marls and limestones
Clays, boulder clays, silts, sands, gravels	Clays, sands and dolomitic limestones, marlstones, sandstones
Clays, marls and sandstones	Clays, sands and marlstones, pyroclastic rocks with gypsum
Clays, marls and sandstones, conglomerates	Clays, sands, gravels, marls and limestones, sandstones, conglomerates, pyroclastic rocks
Clays, marls and sandstones, siltstones, limestones	Conglomerates (calcareous), sandstones and sands, clays, gravels
Clays, marls and sandstones, siltstones, limestones with gypsum	Dolomitic limestones
Clays, marls with gypsum	Dolomitic limestones (jointed, karstified)
Clays, sands	Dolomitic limestones and marls
Clays, sands and sandstones	Dolomitic limestones and marls, clays
Clays, sands and sandstones with gypsum	Dolomitic limestones and marls, clays with gypsum
Clays, sands and siltstones, sandstones	Dolomitic limestones, marlstones and clays with gypsum
Clays, sands, gravels	Dolomitic limestones, marlstones, claystones
Clays, sands, gravels and sandstones with gypsum	Dolomitic limestones, marlstones, siltstones, sandstones and sands
Clays, sands, gravels, marls and claystones, sandstones, conglomerates	Dolomitic limestones, plutonic rocks
Clays, sands, marls and sandstones, shales	Dolomitic limestones, shales, sandstones
Clays, silts and sandstones, marlstones	Gypsum, anhydrite and clays
Clays, silts, sands	Gypsum, anhydrite, dolomitic limestones
Clays, silts, sands, gravels	Limestones
Claystones, sandstones, limestones and clays	Limestones (jointed, karstified)
Claystones, sandstones, siltstones and clays	Limestones (sandy), sandstones and sands, silts
Conglomerates	Limestones and clays, fine sands
Conglomerates, limestones, sandstones and marls, clays	Limestones and marls
Conglomerates, limestones, sandstones, marlstones	Limestones and sands
Conglomerates, quartzites, sandstones, shales, dolomitic limestones	Limestones, calcarenites, sandstones and marls
Conglomerates, sandstones and gravels, sands	Limestones, cherts, sandstones, shales
Conglomerates, sandstones and marls, clays	Limestones, claystones, sandstones, conglomerates
	Limestones, claystones, sandstones, conglomerates and marls, sands
	Limestones, claystones, shales

Silicious geological type	Calcareous geological type
Conglomerates, sandstones and marls, clays with gypsum	Limestones, conglomerates, sandstones and clays
Conglomerates, sandstones, cherts, shales, dolomitic limestones, ophiolitic series	Limestones, conglomerates, sandstones, marlstones and sands
Conglomerates, sandstones, claystones and clays	Limestones, marlstones
Conglomerates, sandstones, limestones and sands, clays	Limestones, marlstones and clays, sands, silts with gypsum
Diatomaceous rocks	Limestones, marlstones, sandstones, conglomerates
Fine sands	Limestones, marlstones, schists
Fine sands, silts, clays, gravels	Limestones, ophiolitic series and marls, clays
Gneisses, mica schists, amphibolites	Limestones, sandstones
Gneisses, mica schists, migmatites	Limestones, sandstones and marls
Gneisses, plutonic rocks (acid)	Limestones, sandstones and sand, gravel
Gravels, sands	LImestones, sandstones and sands, clays
Gravels, sands covered by clays, silts	Limestones, sandstones and sands, clays with gypsum
Gravels, sands, clays	Limestones, sandstones and sands, silts, clays
Gravels, sands, clays, marls and sandstones, conglomerates, limestones	Limestones, sandstones, claystones
Marls and sandstones	Limestones, sandstones, conglomerates, ophiolitic series and clays
Marls and sandstones, limestones with gypsum	Limestones, sandstones, siltstones and marls
Marls and sandstones, limestones, claystones	Limestones, shales
Marls, clays and sandstones, conglomerates, limestones with gypsum	Limestones, shales, sandstones
Marls, sands, clays and sandstones	Limestones, shales, sandstones and marls
Phyllites, gneisses, shales, sandstones, volcanic rocks	Limestones, siltstones, sandstones and marls, clays
Phyllites, schists, quartzites	Marbles
Plutonic rocks	Marbles, schists, quartzites
Plutonic rocks (acid to intermediate)	Marls and claystones, limestones
Plutonic rocks (acid to intermediate, gneissic)	Marls and limestones
Plutonic rocks (basic)	Marls and limestones, sandstones
Plutonic rocks (ultrabasic)	Marls, clays
Pyroclastic rocks	Marls, clays and limestones with gypsum and anhydride
Pyroclastic rocks and sands, clays	Marls, clays, sands and limestones, sandstones
Pyroclastic rocks, volcanic rocks, marlstones	Marlstones, claystones with gypsum and salt
Quartitzes, sandstones, shales, limestones	Marlstones, claystones, shales, phyllites
Quartzites	Marlstones, limestones, sandstones and sands, clays, marls
Quartzites, conglomerates, phyllites, shales	Marlstones, sandstones
Quartzites, conglomerates, sandstones, shales (jointed)	Marlstones, sandstones and marls, clays
Quartzites, sandstones	Marlstones, sandstones and sands, clays
Quartzites, sandstones, phyllites	Marlstones, sandstones, conglomerates with lignites and clays
Quartzites, sandstones, shales	Marlstones, sandstones, limestones and clays
Quartzites, sandstones, shales, volcanic rocks	Travertines

Silicious geological type	Calcareous geological type
Quartzites, shales	
Sands	
Sands (glauconitic)	
Sands and sandstones	
Sands, clays	
Sands, clays and sandstones	
Sands, clays and sandstones, conglomerates	
Sands, clays and sandstones, limestones	
Sands, clays, marls and sanstones, phosphorites, lignites	
Sands, gravels	
Sands, gravels covered by clays, silts	
Sands, gravels, boulders, clays, silts	
Sands, gravels, silts, clays	
Sands, silts, clays	
Sands, silts, clays and sandstones	
Sandstones	
Sandstones and clays, marls	
Sandstones, claystones	
Sandstones, claystones, lignites	
Sandstones, claystones, marlstones, limestones with gypsum	
Sandstones, conglomerates	
Sandstones, conglomerates, claystones, shales, marlstones	
Sandstones, conglomerates, shales, quartzites	
Sandstones, limestones and clays	
Sandstones, limestones, shales, lignites	
Sandstones, marlstones	
Sandstones, marlstones, limestones, volcanic rocks (basic)	
Sandstones, phyllites, quartzites	
Sandstones, shales	
Sandstones, shales (combustible) and clays	
Sandstones, shales and silts	
Sandstones, shales, conglomerates, limestones and marls	
Sandstones, shales, conglomerates, phyllites, volcanic rocks (basic)	
Sandstones, shales, limestones	
Sandstones, siltstones, claystones	
Sandstones, siltstones, claystones with gypsum	
Sandstones, siltstones, claystones, limestones	
Sandstones, siltstones, conglomerates and clays	

Silicious geological type	Calcareous geological type
Schists, gneisses Serpentinites, ophiolitic series Shales Shales, imestones Shales, phyllites, schists, sandstones Shales, quartzites, sandstones Shales, quartzites, volcanic rocks Shales, sandstones Shales, sandstones, cherts, volcainc rocks Shales, sandstones, conglomerates Shales, sandstones, limestones Shales,quartzites, sandstones, phyllites, schists Silts, clays, gravels, boulders Silts, clays, sands Silts, clays, sands, gravels and conglomerates Silts, fine sands Siltstones, claystones, sandstones Siltstones, sandstones and sands, clays Valley fillings Volcanic rocks Volcanic rocks (acid to intermediate) Volcanic rocks (acid) Volcanic rocks (acid), pyroclastic rocks, sandstones, shales Volcanic rocks (basic to intermediate) Volcanic rocks (basic) Volcanic rocks (basic), ophiolitic series Volcanic rocks (jointed) Volcanic rocks, pyroclastic rocks Volcanic rocks, sandstones, shales, dolomitic limestones Volcanic rocks, shales, sandstones, conglomerates, claystones, limestones	

Table 6. Soil typological units (STU) of the European Soil Database v. 2.0 used to derive organic substrate data for European rivers classification. If at least one attribute value was true for the considered STU attributes, it was classified as organic type substrate.

Attribute name	Attribute description	Attribute value
FAO90-LEV1 (FAO90LV1)	Soil major group code of the STU from the 1990 FAO-UNESCO Soil Legend	4- Marsh H - Histosols (includes: Fibric Histosol, Gelic Histosol, Folic Histosol, Terric Histosol, Thionic Histosol)

FAO85-FULL (FAO85FU)	Full soil code of the STU from the 1974 (modified CEC 1985) FAO-UNESCO Soil Legend. 444, O (includes: Od, Odp, Oe, Ox)	444 – Marsh O - Histosol Od - Dystric Histosol Odp - Placi-Dystric Histosol Oe - Eutric Histosol Ox - Gelic Histosol
WRB-FULL (WRBFUL)	Full soil code of the STU from the World Reference Base (WRB) for Soil Resources	4 4 44 4 - Marsh, HSax - Alcalic Histosol HScy - Cryic Histosol HSdy - Dystric Histosol HSeu - Eutric Histosol HSfi - Fibric Histosol HSfo - Folic Histosol HSgc - Glacic Histosol HSge - Gelic Histosol HSom - Ombric Histosol HSrh - Rheic Histosol HSsa - Sapric Histosol HSsz - Salic Histosol HSti - Thionic Histosol HStx - Toxic Histosol
TEXT_SRF_DOM (TXSRFDO) TEXT_SRF_SEC (TXSRFSE)	Dominant surface textural class of the STU Secondary surface textural class of the STU	9 - No mineral texture (Peat soils)
PAR-MAT-DOM (PARMADO) PAR-MAT-SE (PARMASE)	Code for dominant parent material of the STU Code for secondary parent material of the STU	8000 - organic materials 8100 - peat (mires)

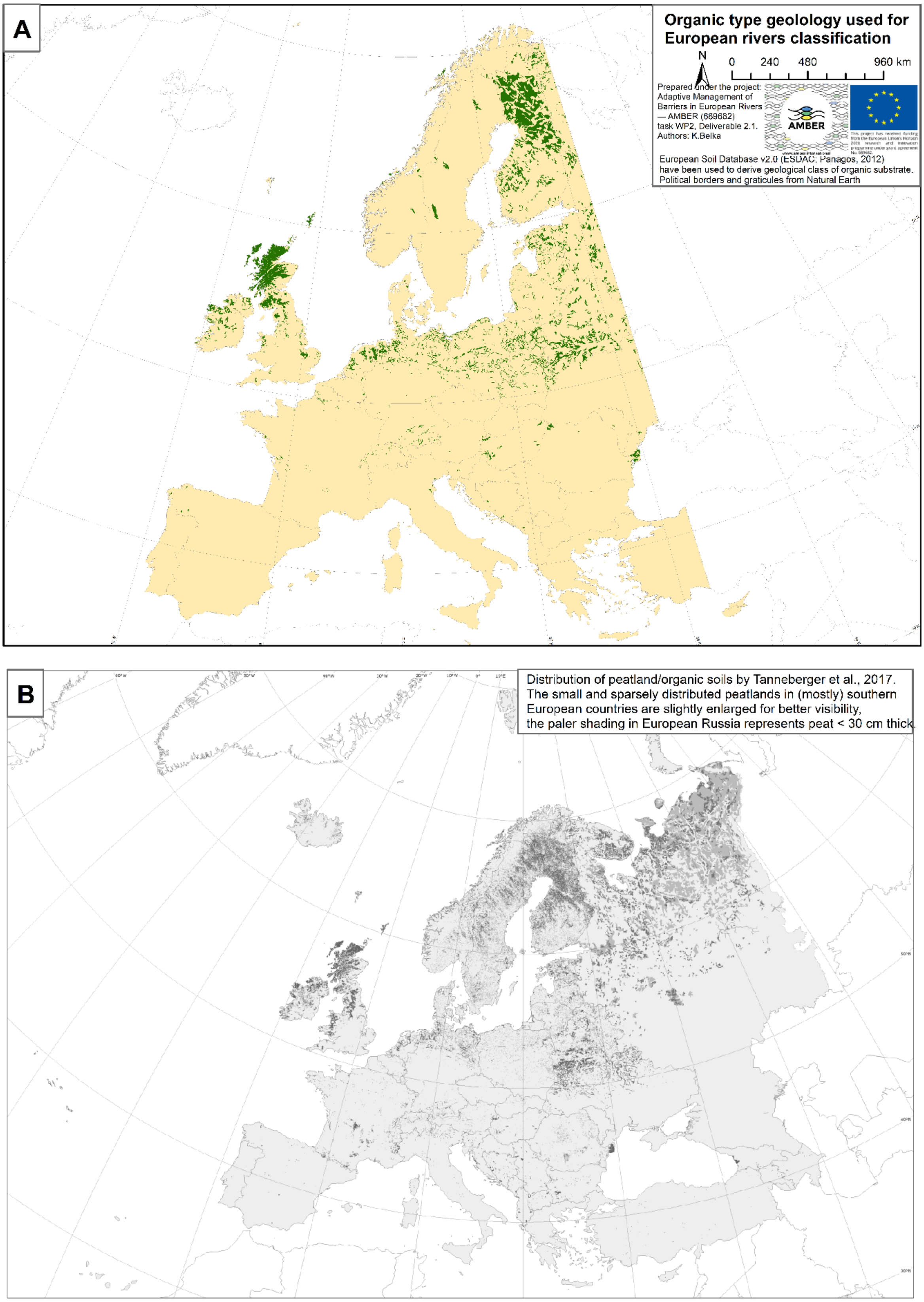


Figure 12. Organic substrate distribution as derived from ESDC data for European soils (A) in comparison with the peatland map of Europe by Tanneberger *et al*. 2017 (B).

Eventually, the following geomorphic descriptors are assigned to the sampling sites:

- Catchment area (km^2)
- River segment Slope (‰)
- Strahler Stream Order (W_STRAHL)
- Mean altitude of the river segment (m a.s.l.) (W_ALT)
- Environmental Zones of Europe (Metzger, 2018)
- Geological type

The remaining descriptors, initially included in the IC dataset, i.e. wetted width, geomorphological river type, natural sediment, source of water and presence of floodplain, are considered either unavailable in necessary quality for all European rivers or too sensitive to human induced alteration (e.g. Geomorphic river type) and are therefore excluded from further analysis (see **Table 4** for description). Spatial distribution of the used geomorphic descriptors are presented in **Figure 13**.

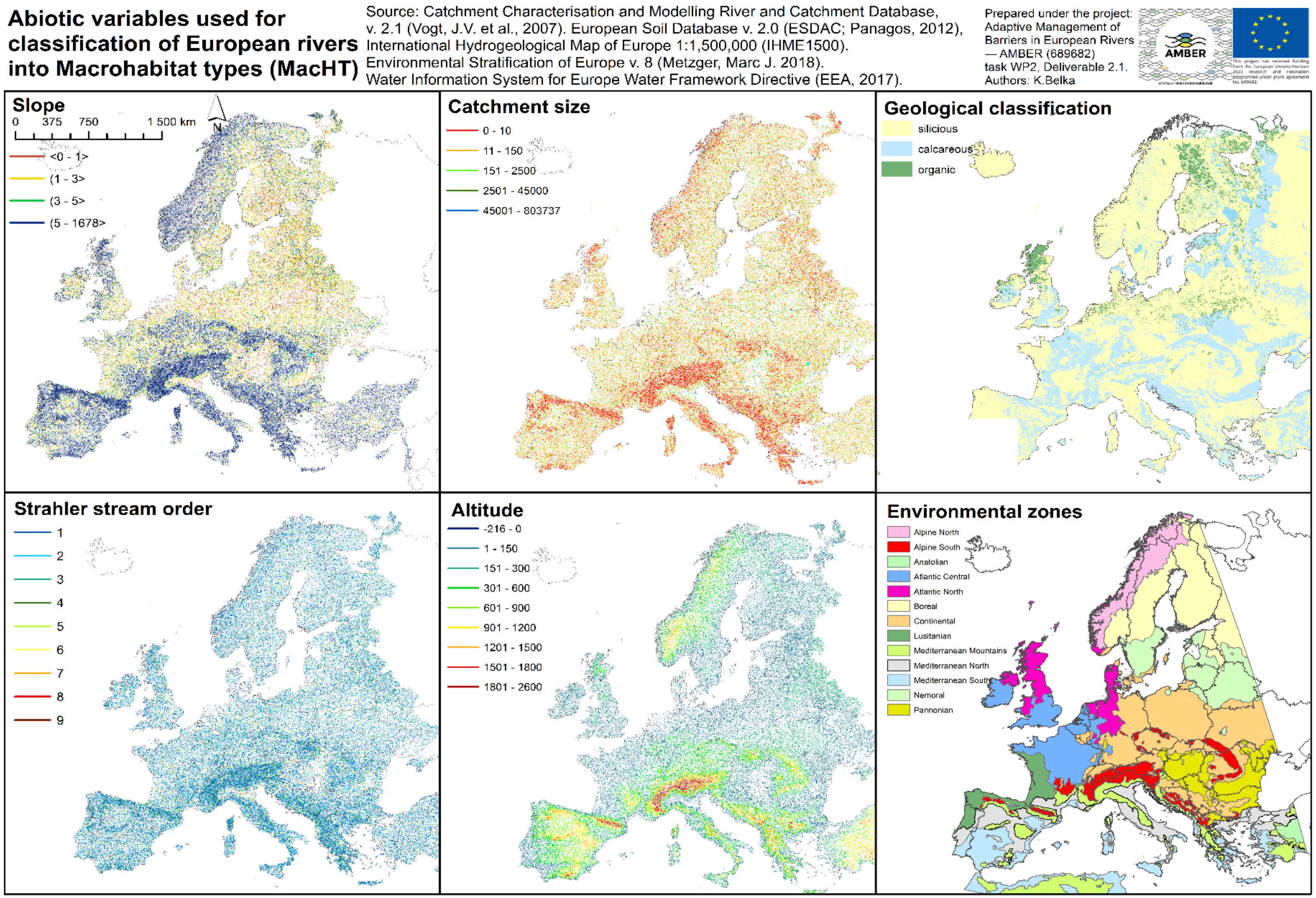


Figure 13. Distribution of abiotic variables used for European rivers classification across Europe.

2.3.2 Fish habitat use guilds

Habitat use guilds (HUT) were determined by modifying the fish guild classification created for the EFI+ Project (Melcher *et al.* 2007, Holzer 2008, EFI+ Manual 2009, Logez *et al*. 2013), as well as some other literature positions – review in Brylińska 2000, Kottelat and Freyhof 2007, www.fishbase.org. According to which, each of the 302 species occurring in European rivers (EFI+ Manual 2009) are ascribed to the following guilds:

- Intolerant species
- Tolerant species,
- Benthic species,
- Rheophilic species,
- Lithophilic species,
- Phytophilic species,
- Insectivorous species,
- Omnivorous species.

For the purpose of this project, eleven HUT guilds were established by combining particular guild characteristics (**Table 7**):

1) Highly rheophilic, intolerant species,

Includes intolerant, rheophilic and lithophilic species. Chosen species were mainly salmonids, bullhead species and strongly rheophilic cyprinids.

2) *Rheophilic benthic species, preferring sandy-gravel bottom substrate*

Only benthic and rheophilic species. Neither lithophilic nor omnivorous species are included, regardless of other assignments. These are mainly sturgeons, barbels, gudgeons and some benthic cyprinids.

3) *Rheophilic water column species, preferring sandy-gravel bottom substrate*

Rheophilic and lithophilic species inhabiting the open water column. These are mainly species such as chub, rainbow trout and common minnow.

4) *Limnophilic benthic species of moderate tolerance*

Limnophilic benthic species, excluding those which are rheophilic, litophilic, phytophilic or omnivorous. Mainly includes loaches, as well as ruffe and some flatfish occurring in estuaries.

5) *Limnophilic water column species of moderate tolerance*

Species that are not tolerant, not benthic, not rheophilic and not phytophilic. This is a large guild consisting mainly of various Cyprinids, including some shads, nases, daces, minnows and roaches. It also contains some Coregonids.

6) *Intolerant, rheophilic benthic species, preferring detritus or pelal bottom substrate*

This guild is composed of lamprey species, with specific biology and habitat requirements. Larvae are detritivorous and inhabit shallow areas. Some of them are also long-migratory species, with a marine, parasitic adult phase.

7) *Intolerant, water column species*

Species that are intolerant and not benthic such as: allis shad, some Corregonids and Salmonids.

8) Limnophilic lithophilic species of moderate tolerance

Lithophilic species that are neither highly tolerant nor intolerant. Also, they are not benthic nor rheophilic nor phytophilic. Mainly species such as asp, Macedonic shad and some daces.

9) Limnophilic phytophilic species of moderate tolerance

Phytophilic species which are neither tolerant, nor rheophilic, nor lithophilic. These are such fish as tench, some loaches, weatherfish, pike, catfish, rudd and bitterling.

10) Benthic species of moderate tolerance

Benthic species that are not tolerant. This guild consists of European eel, burbot, some barbels and Neogobiids.

11) Generalists - tolerant species

Only species that are tolerant are chosen regardless of if they belonged to any other guild. These are mainly cyprinids, such as roach, bleak, common carp, or bream, but also perch and most alien invasive species.

Table 7. Habitat use and tolerance guilds distinguished basing on Melcher's guilds classification (columns) used for Target Fish Communities development. Symbols: 1 – species belong to a guild, 0 – species not belong to a guild, x – guild not taken into account.

Guild No	Guild	Intolerant	Tolerant	Benthic	Rheophilic	Lithophilic	Phytophilic	Omnivorous	N species
1	Highly rheophilic, intolerant species	1	0	x	1	1	0	0	21
2	Rheophilic benthic species, preferring sandy-gravel bottom substrate	x	x	1	1	0	x	0	52
3	Rheophilic water column species, preferring sandy-gravel bottom substrate	x	x	0	1	1	0	x	20
4	Limnophilic benthic species of moderate tolerance	x	0	1	0	0	0	0	29
5	Limnophilic water column species of moderate tolerance	x	0	0	0	x	0	x	70
6	Intolerant, rheophilic benthic species, preferring detritus or pelal bottom substrate	1	0	1	1	1	0	0	10
7	Intolerant, water column species	1	0	0	x	x	x	x	12
8	Limnophilic lithophilic species of moderate tolerance	0	0	0	0	1	0	x	9
9	Limnophilic phytophilic species of moderate tolerance	x	0	x	0	0	1	x	28
10	Benthic species of moderate tolerance	x	0	1	x	x	x	x	14
11	Generalists - tolerant species	0	1	x	x	x	x	x	37

The detailed list of species assignment to different guilds provides **Table 8**.

Table 8. List of fish species occurring in Europe with assigned guild according to EFI+ Manual and based on Melcher's guilds classification and literature data (review: Brylińska 2000, Kottelat and Freyhof 2007, www.fishbase.org).

Species	Guild No	Species	Guild No	Species	Guild No
Barbus caninus	1	*Cobitis narentana*	4	*Alosa alosa*	7
Chondrostoma miegii	1	*Cobitis ohridana*	4	*Alosa fallax*	7
Cobitis calderoni	1	*Cobitis punctilineata*	4	*Anaecypris hispanica*	7
Cottus gobio	1	*Cobitis rhodopensis*	4	*Aphanius iberus*	7
Cottus koshewnikovi	1	*Cobitis stephanidisi*	4	*Coregonus albula*	7
Cottus petiti	1	*Cobitis strumicae*	4	*Coregonus lavaretus*	7
Cottus poecilopus	1	*Cobitis taenia*	4	*Leuciscus keadicus*	7
Hucho hucho	1	*Cobitis tanaitica*	4	*Pungitius hellenicus*	7
Leuciscus souffia	1	*Cobitis trichonica*	4	*Salmo trutta lacustris*	7
Oncorhynchus gorbuscha	1	*Cobitis zanandreai*	4	*Salvelinus alpinus*	7
Oncorhynchus tschawytscha	1	*Gymnocephalus cernuus*	4	*Valencia hispanica*	7
Romanichthys valsanicola	1	*Hemichromis bimaculatus*	4	*Valencia letourneuxi*	7
Salmo salar	1	*Knipowitschia milleri*	4	*Alosa macedonica*	8
Salmo trutta fario	1	*Knipowitschia panizzae*	4	*Aspius aspius*	8
Salmo trutta trutta	1	*Neogobius fluviatilis*	4	*Chalcalburnus chalcoides*	8
Salmothymus obtusirostris	1	*Neogobius gymnotrachelus*	4	*Coregonus peled*	8
Salvelinus fontinalis	1	*Platichthys flesus*	4	*Leuciscus borysthenicus*	8
Salvelinus namaycush	1	*Pleuronectes platessa*	4	*Leuciscus carolitertii*	8
Thymallus thymallus	1	*Sabanejewia bulgarica*	4	*Leuciscus pyrenaicus*	8
Zingel streber	1	*Sabanejewia larvata*	4	*Polyodon spathula*	8
Zingel zingel	1	*Sander volgensis*	4	*Thymallus baicalensis*	8
Abramis sapa	2	*Alburnus albidus*	5	*Aphanius fasciatus*	9
Acipenser baeri	2	*Alosa immaculata*	5	*Atherina boyeri*	9
Acipenser gueldenstaedtii	2	*Alosa killarnensis*	5	*Clarias gariepinus*	9
Acipenser naccarii	2	*Alosa maeotica*	5	*Economidichthys pygmaeus*	9
Acipenser nudiventris	2	*Alosa tanaica*	5	*Economidichthys trichonis*	9
Acipenser oxyrinchus	2	*Alosa vistonica*	5	*Esox lucius*	9
Acipenser ruthenus	2	*Atherina hepsetus*	5	*Eupallasella perenurus*	9
Acipenser stellatus	2	*Atherina presbyter*	5	*Knipowitschia caucasica*	9
Acipenser sturio	2	*Chalcalburnus belvica*	5	*Leucaspius delineatus*	9
Ambloplites rupestris	2	*Chelon labrosus*	5	*Leuciscus idus*	9
Barbatula barbatula	2	*Chondrostoma arrigonis*	5	*Misgurnus anguillicaudatus*	9
Barbatula bureschi	2	*Chondrostoma genei*	5	*Misgurnus fossilis*	9
Barbatula pindus	2	*Chondrostoma knerii*	5	*Pseudophoxinus stymphalicus*	9
Barbus barbus	2	*Chondrostoma phoxinus*	5	*Pungitius platygaster*	9

Species	Guild No	Species	Guild No	Species	Guild No
Barbus cyclolepis	2	*Chondrostoma prespense*	5	*Rhodeus sericeus*	9
Barbus euboicus	2	*Chondrostoma soetta*	5	*Rutilus aula*	9
Barbus guiraonis	2	*Coregonus autumnalis*	5	*Scardinius acarnanicus*	9
Barbus haasi	2	*Coregonus muscun*	5	*Scardinius erythrophthalmus*	9
Barbus macedonicus	2	*Coregonus oxyrinchus*	5	*Scardinius graecus*	9
Barbus meridionalis	2	*Coregonus pidschian*	5	*Scardinius racovitzai*	9
Barbus peloponnesius	2	*Coregonus trybomi*	5	*Scardinius scardafa*	9
Barbus plebejus	2	*Dicentrarchus labrax*	5	*Silurus aristotelis*	9
Barbus steindachneri	2	*Dicentrarchus punctatus*	5	*Silurus glanis*	9
Barbus tyberinus	2	*Hypophthalmichthys nobilis*	5	*Squalius alburnoides*	9
Chondrostoma lemmingii	2	*Ictiobus niger*	5	*Tinca tinca*	9
Chondrostoma nasus	2	*Ladigesocypris ghigii*	5	*Tropidophoxinellus hellenicus*	9
Chondrostoma polylepis	2	*Lepomis auritus*	5	*Tropidophoxinellus spartiaticus*	9
Chondrostoma toxostoma	2	*Lepomis cyanellus*	5	*Umbra krameri*	9
Chondrostoma vardarense	2	*Leuciscus microlepis*	5	*Anguilla anguilla*	10
Chondrostoma willkommii	2	*Leuciscus svallize*	5	*Barbus graecus*	10
Cobitis elongata	2	*Leuciscus turskyi*	5	*Barbus microcephalus*	10
Cobitis vardarensis	2	*Leuciscus ukliva*	5	*Barbus prespensis*	10
Cobitis vettonica	2	*Leuciscus zrmanjae*	5	*Cobitis meridionalis*	10
Gobio albipinnatus	2	*Liza aurata*	5	*Knipowitschia goerneri*	10
Gobio banarescui	2	*Liza saliens*	5	*Knipowitschia thessala*	10
Gobio benacensis	2	*Micropterus dolomieui*	5	*Lota lota*	10
Gobio elimeius	2	*Morone saxatilis*	5	*Neogobius kessleri*	10
Gobio gobio	2	*Mugil cephalus*	5	*Neogobius melanostomus*	10
Gobio kesslerii	2	*Mylopharyngodon piceus*	5	*Proterorhinus marmoratus*	10
Gobio uranoscopus	2	*Odonthestes bonariensis*	5	*Salaria fluviatilis*	10
Gymnocephalus baloni	2	*Oreochromis mossambicus*	5	*Triglopsis quadricornis*	10
Gymnocephalus schraetser	2	*Oreochromis niloticus*	5	*Zosterisessor ophiocephalus*	10
Huso huso	2	*Osmerus eperlanus*	5	*Abramis brama*	11
Pachychilon pictum	2	*Pachychilon macedonicum*	5	*Alburnus alburnus*	11
Rutilus pigus	2	*Parabramis pekinensis*	5	*Ameiurus melas*	11
Sabanejewia aurata	2	*Pelecus cultratus*	5	*Ameiurus nebulosus*	11
Sabanejewia balcanica	2	*Phoxinellus adspersus*	5	*Barbus bocagei*	11
Sabanejewia romanica	2	*Phoxinellus alepidotus*	5	*Barbus comizo*	11
Vimba melanops	2	*Phoxinellus croaticus*	5	*Barbus graellsii*	11
Vimba vimba	2	*Phoxinellus epiroticus*	5	*Barbus sclateri*	11
Zingel asper	2	*Phoxinellus ghetaldii*	5	*Blicca bjoerkna*	11

Species	Guild No	Species	Guild No	Species	Guild No
Zingel balcanicus	2	*Phoxinellus metohiensis*	5	*Carassius auratus*	11
Abramis ballerus	3	*Phoxinellus prespensis*	5	*Carassius carassius*	11
Alburnoides bipunctatus	3	*Phoxinellus pstrossii*	5	*Carassius gibelio*	11
Iberocypris palaciosi	3	*Poecilia reticulata*	5	*Chondrostoma lusitanicum*	11
Leuciscus aradensis	3	*Pseudophoxinus beoticus*	5	*Cichlasoma facetum*	11
Leuciscus burdigalensis	3	*Pseudophoxinus minutus*	5	*Clarias batrachus*	11
Leuciscus cephalus	3	*Rutilus arcasii*	5	*Cobitis paludica*	11
Leuciscus illyricus	3	*Rutilus basak*	5	*Ctenopharyngodon idella*	11
Leuciscus leuciscus	3	*Rutilus heckelii*	5	*Cyprinus carpio*	11
Leuciscus lucumonis	3	*Rutilus karamani*	5	*Fundulus heteroclitus*	11
Leuciscus montenigrinus	3	*Rutilus lusitanicus*	5	*Gambusia affinis*	11
Leuciscus muticellus	3	*Rutilus meidingeri*	5	*Gambusia holbrooki*	11
Leuciscus pleurobipunctatus	3	*Rutilus ohridanus*	5	*Gasterosteus aculeatus*	11
Leuciscus polylepis	3	*Rutilus prespensis*	5	*Hemichromis fasciatus*	11
Leuciscus torgalensis	3	*Rutilus ylikiensis*	5	*Hypophthalmichthys molitrix*	11
Oncorhynchus kisutch	3	*Sander lucioperca*	5	*Ictalurus nebulosus*	11
Oncorhynchus mykiss	3	*Sygnathus abaster*	5	*Ictalurus punctatus*	11
Phoxinellus fontinalis	3	*Tilapia zillii*	5	*Lepomis gibbosus*	11
Phoxinus phoxinus	3	*Vimba elongata*	5	*Liza ramada*	11
Rutilus frisii	3	*Eudontomyzon danfordi*	6	*Micropterus salmoides*	11
Rutilus rubilio	3	*Eudontomyzon hellenicus*	6	*Perca fluviatilis*	11
Aulopyge huegelii	4	*Eudontomyzon mariae*	6	*Perccottus glenii*	11
Barbus albanicus	4	*Eudontomyzon stankokaramani*	6	*Pimephales promelas*	11
Cobitis arachthosensis	4	*Eudontomyzon vladykovi*	6	*Pseudorasbora parva*	11
Cobitis bilineata	4	*Lampetra fluviatilis*	6	*Pungitius pungitius*	11
Cobitis dalmatina	4	*Lampetra planeri*	6	*Rutilus macrolepidotus*	11
Cobitis elongatoides	4	*Lethenteron camtschaticum*	6	*Rutilus rutilus*	11
Cobitis hellenica	4	*Lethenteron zanandreai*	6	*Umbra pygmaea*	11
Cobitis megaspila	4	*Petromyzon marinus*	6		

2.3.3 Statistical analysis

River water bodies were classified according to the fish assemblages expected in specific geomorphologial settings at non-disturbed (reference) sites. Subsequently, we used physical attributes associated with the classes to expand this classification to all water bodies in the European database. We applied h two techniques: non-hierarchical cluster (NHC) analysis and discrimination with Analysis of Group Similarities (ANOSIM) for step one and Classification and Regression Trees (CART, Breiman *et al.* 1984, Clarke 1993, De'ath & Fabricius 2000) for step two.

2.3.3.1 Cluster Analysis

Non hierarchical cluster (NHC) analysis was conducted using the IC data from 1099 river sites classified as nondisturbed. Cluster analysis was applied to two data sub-sets sequentially. The first sub-set consisted of geomorphic descriptors of sampling sites, and was clustered into samples with similar habitat characteristics. These cluster groupings were then added as an additional variable to the biological data of HUT fish guild proportions captured in each site to produce a mixed data set (guilds/physical clusters), which was then also clustered. In both cases, the clustering procedure is the same. The distance matrix is created by standardizing the data using *Gower* and *Manhattan* similarity distances for the physical and the mixed data set respectively (Gower 1971, Krause 1987). The number of clusters is determined with help of scree and silhouette plots. A Partitioning Around Medoids (PAM) clustering model was applied (Kaufman & Rousseeuw 1987, Hastie *et al.* 2001, Park & Jun 2009). The cluster plots and silhouette plots were created, as well as box plots for each variable. Subsequently data discrimination with ANOSIM was performed to verify model performance. The calculated FCMacHT classes are assigned to each site. The above decision tree is applied to all water bodies of Europe to determine their FCMaCHT class.

2.4 Quantification of barrier impact on fish habitat

To estimate the influence of each barrier type on a particular habitat attribute presented in chapter 2.2 we undertook an extensive literature review (AMBER MS1 Appendix), and discussed our conclusions as part of a workshop event with experts involved in the project. The change of the relative habitat area where an attribute occurred was quantified using five categories:

1) 0.0 major reduction
2) 0.5 small reduction
3) 1.0 no change
4) 1.5 small increase
5) 2.0 major increase

The results were summarized and the effects of each barrier type (*Bi*) in numeric values between 0 and 2 were given in **Table 10**. The *Bi* values were used to calculate the weighted remaining habitat proportion (w*RHp*) in each FCMacHT after barrier construction, according to the formula:

$$wRHp = \frac{\sum_1^{21}(GSI * Bi)}{\sum_1^{21} GSI} * GP * 100\%$$

Where:

wRHp – weighted remaining habitat proportion,
GSI – guild specific suitability of habitat attribute,
Bi – barrier impact,

GP – guild proportion in EFC.

The *wRHp* value ranges from 0 to 200%. In the case of permanent negative impact on all habitat attributes, it equals 0, while in a theoretical situation of only positive impact – it reaches 200%. In the case of no barrier impact, the *wRHp* will show no change in habitat availability and equals 100%.

This formula was used to assess the impact of each barrier type on habitat suitability and availability for particular FCMacHT river type, expressed as percentage of the remaining suitable habitat for EFC**.**

3 RESULTS

3.1 Cluster Analysis

Figure 14 presents scree and silhouette plot for the physical site descriptors, with help of which 16 river type clusters are selected. **Figure 15** shows the cluster plot for two principle components and **Figure 16** a silhouette plot. The first two components explain 69.9% of variability and average silhouette width is 0.69. The ANOSIM discrimination with obtained clusters is significant with R=0.967 with p<0.001 (**Figure 17**).

The identified clusters are then added to the biological data for each site (HUT proportions). **Figure 18** is the scree and silhouette plot for biological descriptors associated with physical clusters from **Figure 14**. Based on this figure we selected 15 clusters for NHC procedure. **Figure 19** shows the cluster plot with FCMacHT clusters well separated in 2-D space. This separation explains only 16.33 % of variability indicating that other components are important in the process. **Figure 20** shows silhouette plot with average silhouette width of 0.62. The ANOSIM discrimination with obtained clusters offers R=0.98 with p<0.001 (**Figure 21**).

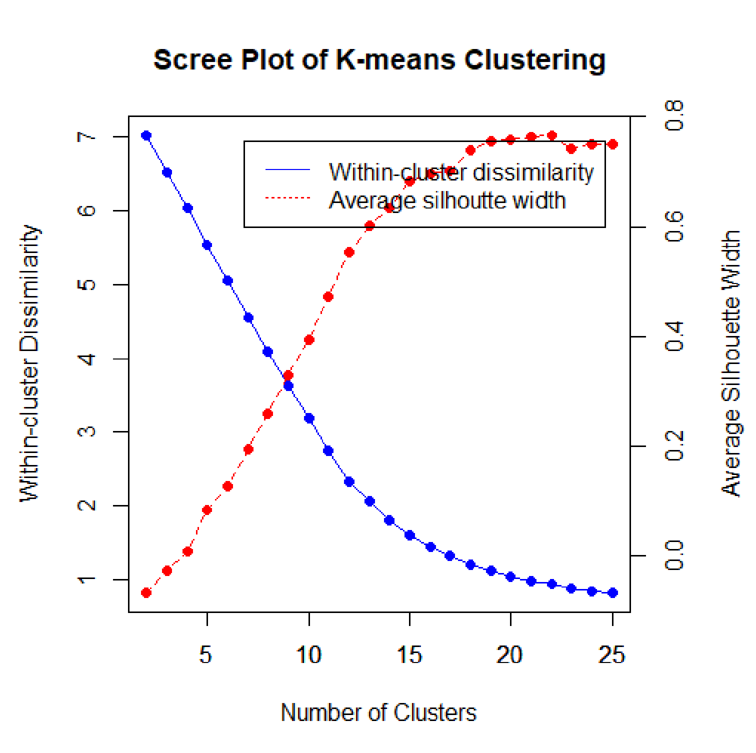


Figure 14. Scree and silhouette plot of environmental attributes of the sampled sites.

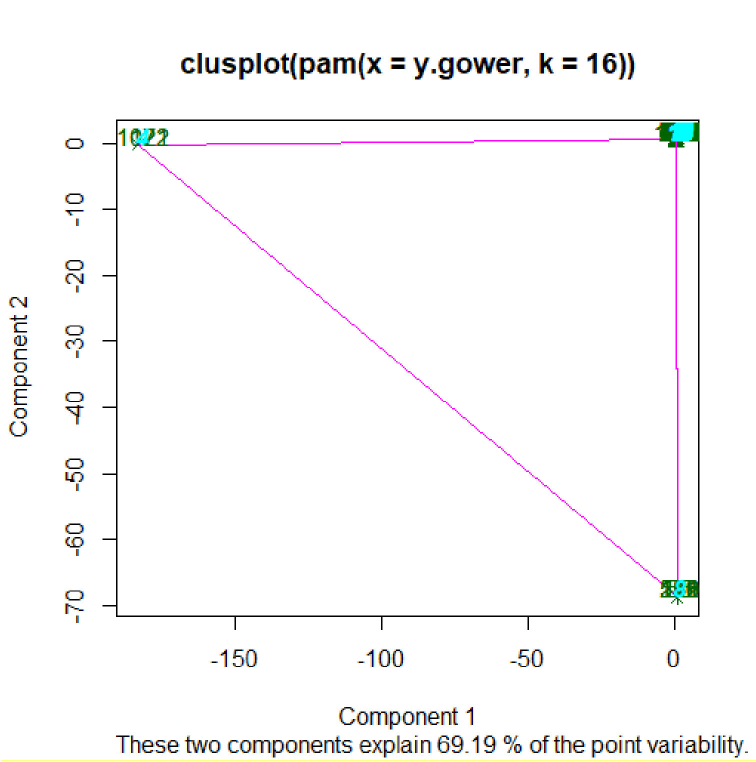


Figure 15. Cluster plot of environmental attributes of the sampled sites for two main components. The numbers represent the site number and pink lines distance between the ovoids.

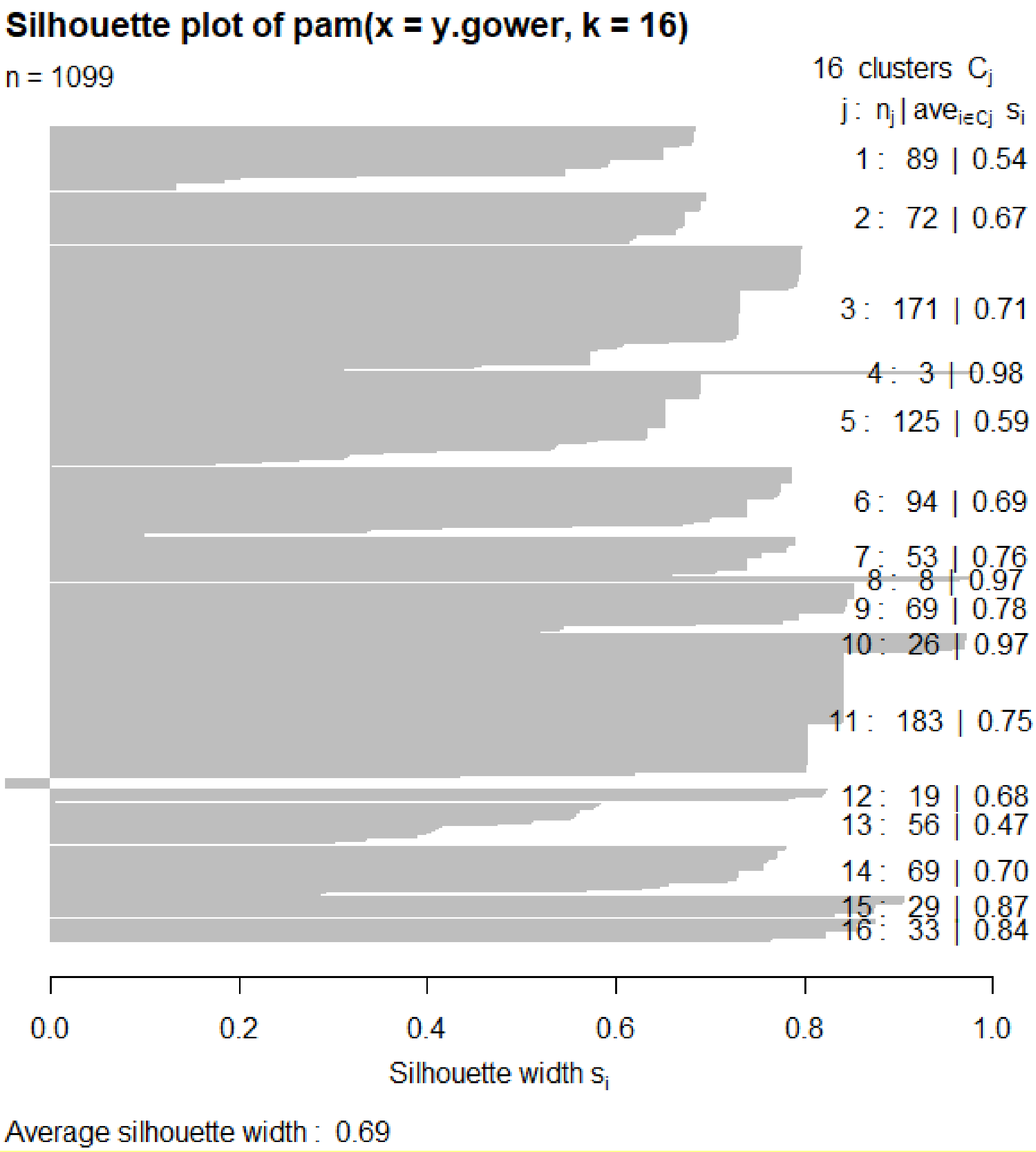


Figure 16. Silhouette plot for clusters of environmental attributes of the sampled sites.

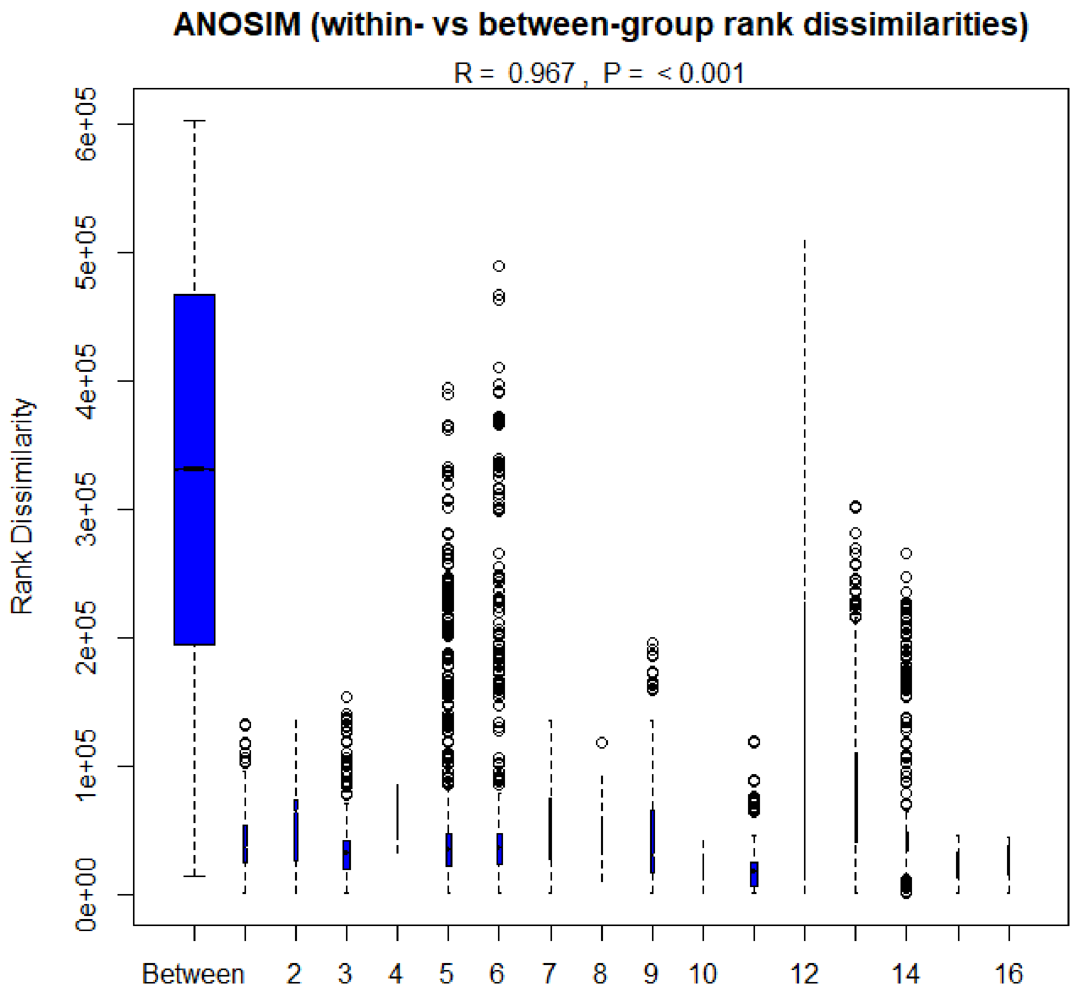


Figure 17. Whisker diagram representing distances between and within the clusters of environmental attributes of sampled sites.

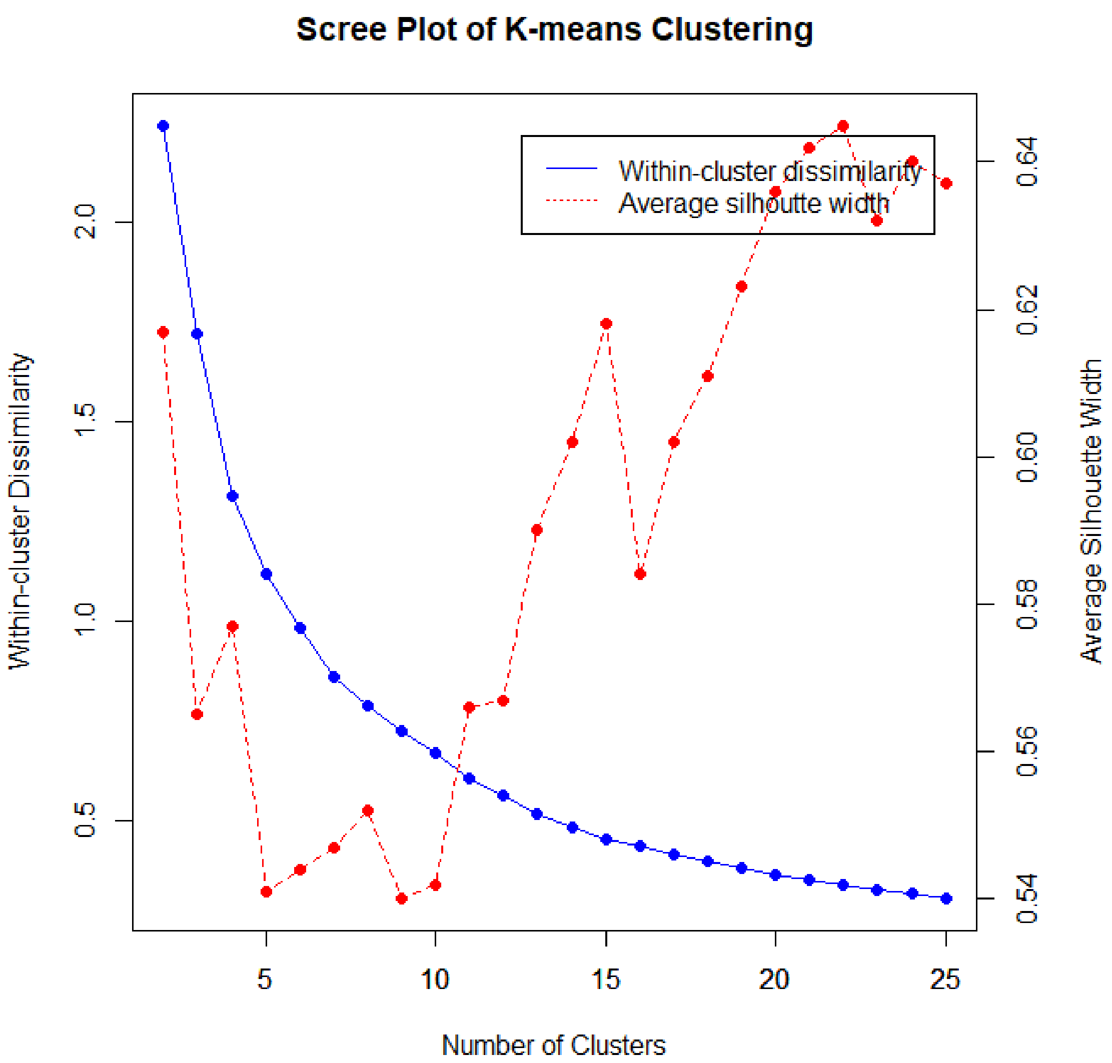


Figure 18. Scree and silhouette plot of habitat attributes (environmental class and guild composition) of sampled sites.

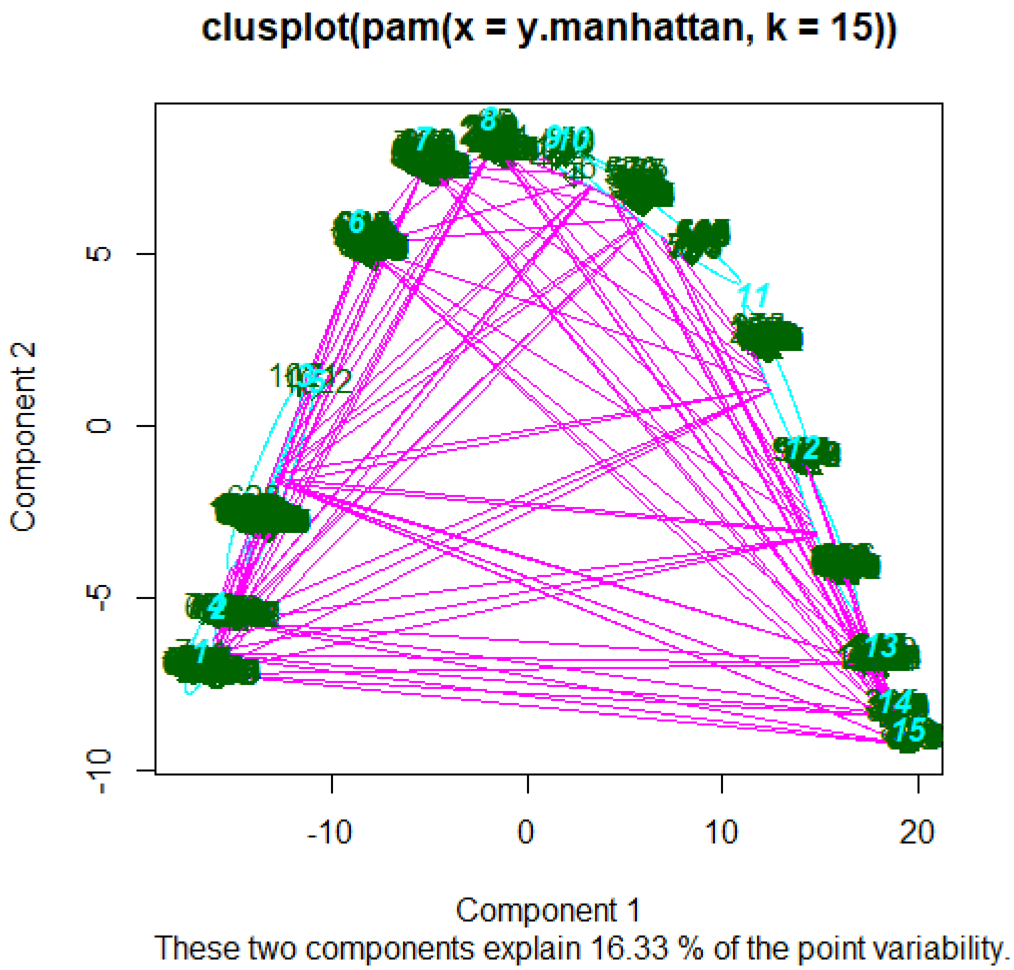


Figure 19. Cluster plot of habitat attributes of the sampled sites for two main components. The numbers represent the site number and pink lines distance between the ovoids.

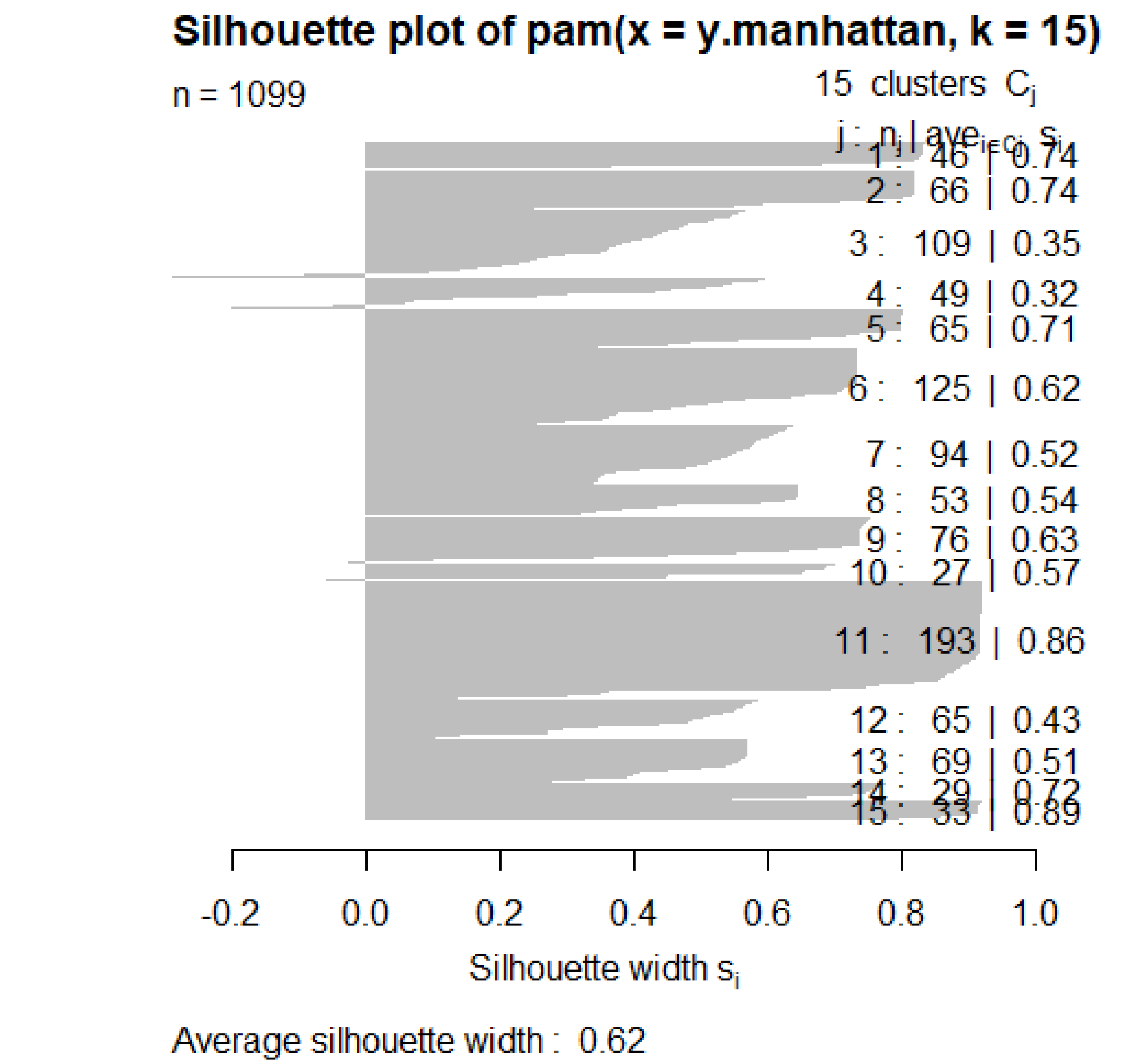


Figure 20. Silhouette plot for clusters of habitat attributes of the sampled sites.

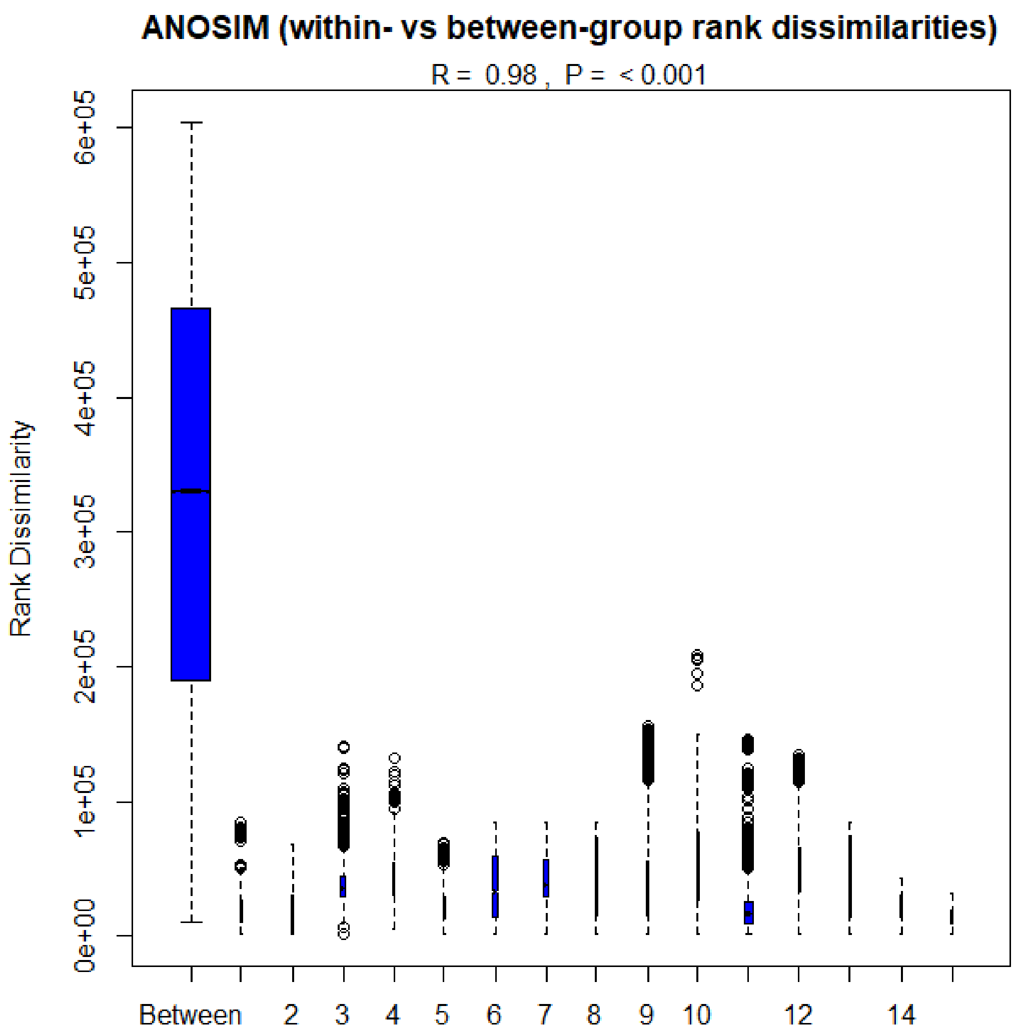


Figure 21. Whisker diagram representing distances between and within the clusters of habitat attributes of sampled sites.

3.1.1 CART

The FCMacHT class is then added to the physical descriptors of each site as a grouping variable to perform Classification and Regression Tree (CART) analysis to determine how the FCMacHT distribution is shaping along the gradient of these variables. The Complexity Parameter plot is used to determin the acceptable relative error for pruning the decision tree.

Figure 22 presents the Complexity Parameter plot and **Figure 23** the Classification tree, which was pruned at 0.003 relative error. The 38 leaves long tree has 86% of correct reclassification (Kappa=0.912).

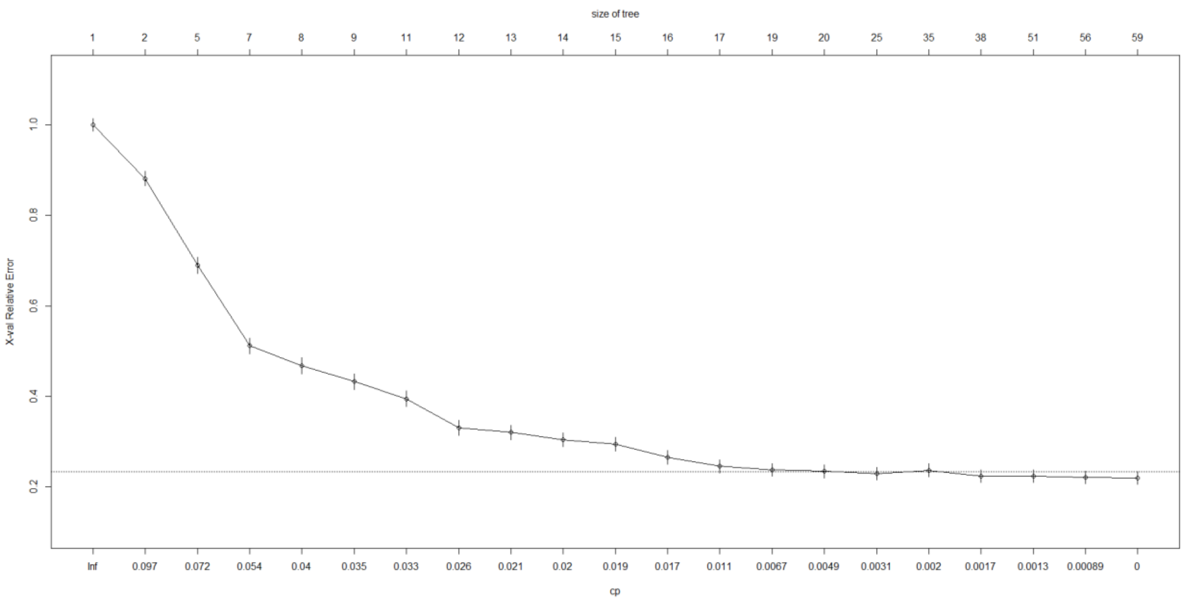


Figure 22. Complexity Parameter plot for environmental attributes of sampled sites.

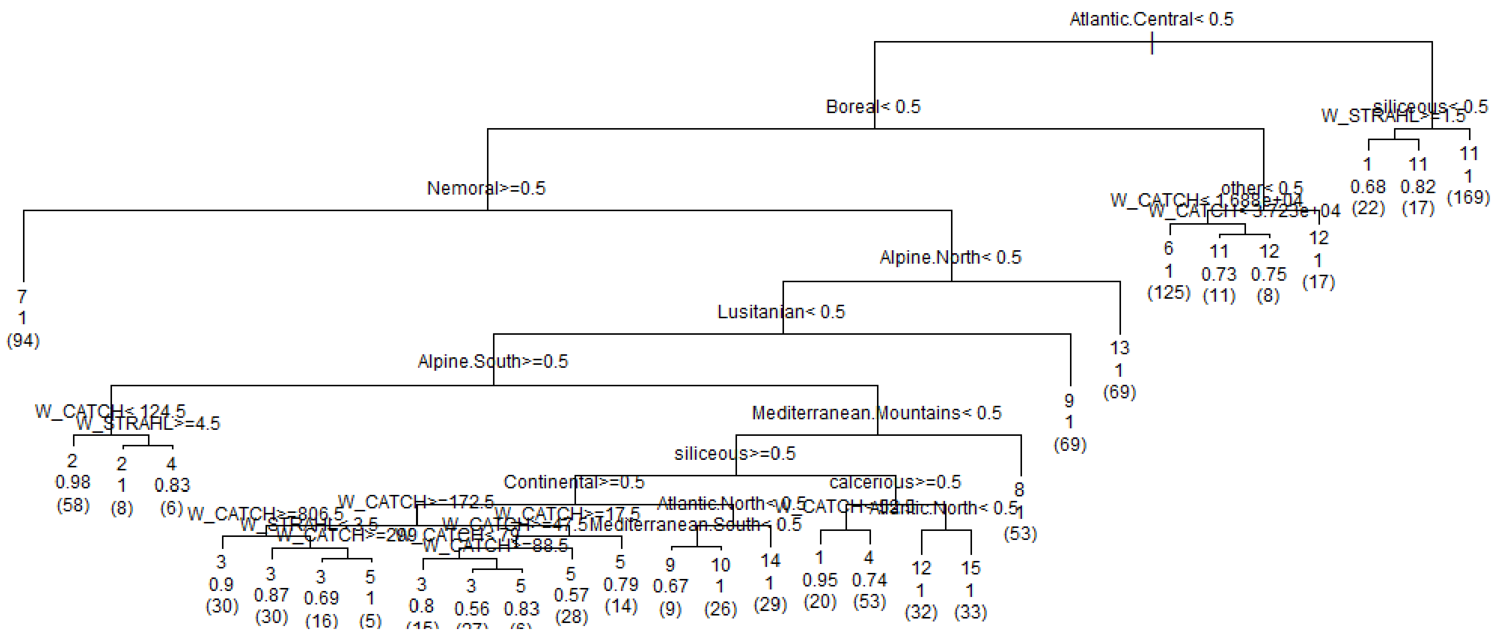


Figure 23. Classification tree of environmental attributes according to the FCMacHT class.

The model first separated Atlantic Central Environmental Zone and assigned water bodies to group 11 (Western European and Atlantic Rivers) with the exception of first order streams non-siliceous, which were put to group 1 (Highland medium sediment rivers). Rivers of Boreal Environmental Zone with non-siliceous and non-calcareous geology catchments larger than 16 880 km^2 have similar habitat distribution as group 11, but streams with smaller catchments are again in group 1. Boreal rivers with organic geology and those with catchments larger than 37230 km^2 are classified as group 12 (Lowland medium sediment and organic rivers). All Nemoral Environmental Zone river's fish community macrohabitat types are distributed as group 7 (Boreal Lowland Rivers), Alpine North group 13 (Boreal Atlantic Large-medium sediment rivers) and Lusitanian are a majority of group 9 (South European highland rivers). Alpine South Environmental Zone rivers are divided into 2 groups: these with catchments smaller than 124.5 km^2 and a stream order of 5 and higher into group 2 (Mountain alpine and subalpine rivers) and other rivers into group 4 (Central European lowland, large-medium sediment rivers). Rivers of Mediterranean Mountain Zones build group 8 (Mediterranean mountain and upland rivers). In remaining zones, geology drives further divisions as such siliceous rivers in Continental Environmental Zone all belong either to group 3 (Central European lowlands, medium sediment rivers) or 5 (Highland and lowland, large-medium sediment rivers). Atlantic North EZ creates group 14 (Atlantic medium-large sediment rivers) and Mediterranean South Zone group 10 (Mediterranean lowland rivers). Remaining siliceous rivers are in group 9. Calcareous rivers belong to group 1 and 4 depending on catchment size. Rivers with other geology in Atlantic North Zone are in group 15 (North Atlantic lowland medium-large sediment rivers and in other zones in group 12.

3.2 Map

Figure 24 represents the distribution of rivers classified according to FCMacHT sampled in the Intercalibration exercise.

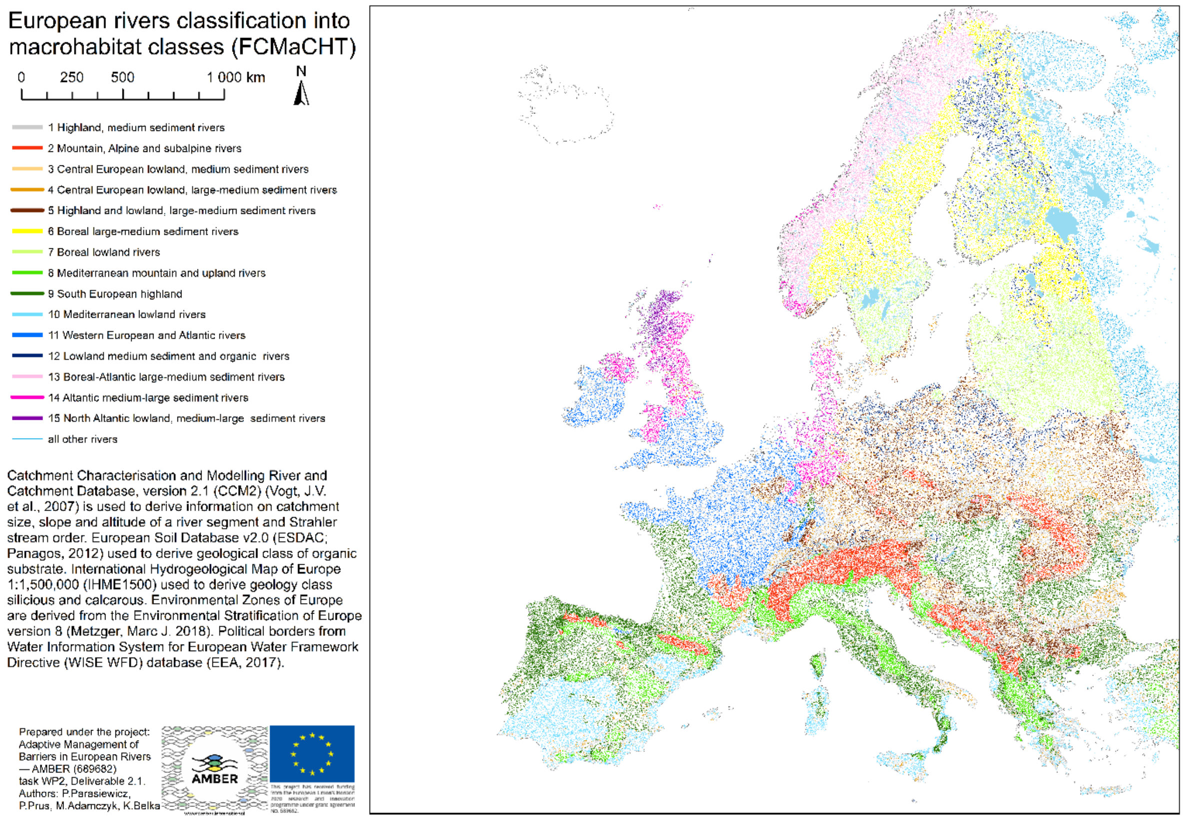


Figure 24. **European rivers** classified into 15 **Macrohabitat types** (**MacHT**) (AMBER D2.1, version 2.0).

3.3 Expected Fish Communities

Using a guilds proportion in fish assemblage calculated for a set of 10 randomly chosen NDS for each type, an Expected Fish Community (EFC) was developed (**Figure 25**). Basic abiotic and geographic characteristics of NDS ascribed to 15 MacHT types are described for each river type, together with the expected fish guild composition.

Type 1. Highland, medium sediment rivers (Figure 24, 25)
The first river-type distinguished groups rivers and streams of highland and submountain type, with dominating medium size sediment fraction. NDS sites are located up to 686 m a.s.l. (average 317 m a.s.l.) and characterized by moderate slope values - up to 20‰ (average 8.2‰). Those rivers are distributed across Europe, except Boreal regions. Fish communities of this type consist mainly of Highly rheophilic, intolerant species (48%), with a share of 24% of Rheophilic water column species and 12% of Rheophilic benthic species, preferring sandy-gravel bottom substrate. Those guilds are complemented by Generalists (16%). Such composition of fish assemblage is related to moderate and high slope values, and corresponds with cold, well aerated water and medium or coarse bottom sediment.

Type 2. Mountain, Alpine and subalpine rivers (Figure 24, 25)
This river-type group streams and small rivers of Alpine and subalpine type located in mountain areas. NDS sites are located up to 1212 m a.s.l. (average 729 m a.s.l.) and characterized by high slope values - up to 90‰ (average 23‰). Those rivers are distributed across Central and Southern Europe in Mountains: Alps, Carpathians, Pyrenees, Dinaric Alps and Balcan Mts. Fish communities of this type consist mainly of Highly rheophilic, intolerant species (55%), with a share of 27% of Rheophilic water column species and 18% of Rheophilic benthic species, preferring sandy-gravel bottom substrate. Such composition of fish assemblage is related to high slope values, and corresponds with cold, well-aerated water and coarse bottom sediment.

Type 3. Central European lowland, medium sediment rivers (Figure 24, 25)
This type comprises Continental rivers and streams located mianly in Central and Eastern Europe plains. NDS sites are located up to 380 m a.s.l. (average 186 m a.s.l.) and characterized by low slope - up to 14‰ (average 2.9‰). Those rivers are distributed across the Central Europe – from Eastern France, Germany and Belgium through Poland, Czech Republic and Slovakia to Ukraine, Romania and Balcans, in Continental bio-geographical regions. The fish community of these rivers is composed of 37% of Rheophilic benthic species, preferring sandy-gravel bottom substrate. All four rheophilic fish guilds constitute 72% of fish community, while two limnophilic guilds contribute to 13%. Benthic species of moderate tolerance represent 5%. Generalists are moderately abundant with a 9% share in fish community. A high diversity of guilds composition reflects high habitat variability in rivers and streams distributed wdely across the Continental bio-geographical region.

Type 4. Central European lowland, large-medium sediment rivers (Figure 24, 25)
Type 4 groups rivers of similar geographical location as the group 3, in Continental bio-geographical regions. Some of them are also located in the Mediterranean region. In this group are smaller streams and rivers that are distributed on higher altitudes. NDS sites are located up to 521 m a.s.l. (average 317 m a.s.l.) and characterized by moderate slope - up to 10‰ (average 4.6‰). The fish community of these rivers is composed of 41% of Rheophilic benthic species, preferring sandy-gravel bottom substrate. All four rheophilic fish guilds constitute 75% of fish community, while limnophilic phytophilic and Benthic species of moderate tolerance guilds contribute to 8% and 7% respectively. Generalists are moderately abundant with 10% share in fish community. Guild diversity is similarly

related to riverine habitat variability, with a higher representation of rheophilic species as in Type 3 due to more upland character of the considered streams.

Type 5. Highland and lowland, large-medium sediment rivers (Figure 24, 25)
This type comprises of Continental streams and small rivers located mainly in Central and Eastern Europe submountain and upland landscapes. NDS sites are located up to 827 m a.s.l. (average 286 m a.s.l.) and characterized by high slope variability - up to 39‰, with an average of 10.4‰ and a minimum of 0.6‰. The rivers considered are distributed across the Continental bio-geographical region, with some sites in South Scandinavia. The fish community of these rivers is composed of 39% Highly rheophilic intolerant species. All four rheophilic fish guilds constitute 76% of fish community, while two limnophilic guilds contribute to 14%. Generalists are moderately abundant with a 10% share in fish community. Strong domination of rheofilic guilds is connected to a more upland character of those rivers.

Type 6. Boreal large-medium sediment rivers (Figure 24, 25)
Type no 6 groups lowland and upland Boreal rivers and streams. Chosen NDS sites are located on altitudes up to 708 m a.s.l. (average 239 m a.s.l.), with variable slope values – maximum of 40.8‰, and an average of 10.1‰. Those rivers are located in Northern Europe, in the Boreal bio-geographical region. The fish assemblage is dominated by three rheophilic guilds amounting together to 65% of the fish community, with dominance of highly rheophilic intolerant species (39%). The next important guild – Intolerant, water column species make 13%, while two guilds of limnophilic species together have a 13% share. Generalists are present with a moderate share of 6%. The fish community of this river type is quite complex, with six guilds present. Domination of rheofilic guilds is connected to more a frequent occurrence of coarse bottom substrate in those rivers.

Type 7. Boreal lowland rivers (Figure 24, 25)
This group consists of lowland Boreal rivers. Chosen NDS sites are located on altitudes up to 200 m a.s.l. (average 96 m a.s.l.), with low slope values – maximum of 16.6‰, and an average of 2.8‰. Those rivers are located in Northern Europe, in the Boreal bio-geographical region, Nemoral Environmental Zone, hence in more southern locations than Type 6. The fish assemblage is dominated by four rheophilic guilds amounting together to 76% of fish community, with dominance of highly rheophilic intolerant species (39%). Only one limnophilic guild has a 6% share, while benthic species of moderate tolerance have 8%. Generalists represent a share of 10%. The fish community of this river type is quite complex, with six guilds present. Domination of rheofilic guilds in these lowland rivers is connected to more severe climate of boreal zone and location of a number of rivers in hilly a landscape of lakelands.

Type 8. Mediterranean mountain and upland rivers (Figure 24, 25)
In river-type 8, a set of Mediterranean mountain and upland rivers is grouped. Maximum height of NDS site is 1208 m a.s.l. with an average of 785 m a.s.l. River slopes are quite high – up to 27,6‰ (average 5,6‰). Those rivers are distributed widely across the Mediterranean region; from Portugal to Greece. Rheophilic guilds strogly dominate the fish community with a total share of 91%. Within this group higly rheophilic, intolerant species are most numerous (44%). Limnophilic lithophilic species of moderate tolerance makes 9% of fish community and generalists are absent. A high share of rheophilic guilds and lack of generalists results from the mountaineous character of the rivers grouped in this type.

Type 9. South European highland rivers (Figure 24, 25)
The rivers grouped in this type represent the highland character of the Lusitanian, Mediterranean and Pannonian zone. Maximum height of NDS site is 572 m a.s.l. with an average of 229 m a.s.l. River

slopes are high – up to 63‰ (average 13.8‰). Those rivers are distributed widely across Southern Europe from Western France, Portugal and Spain, through Italy and Greece to Pannonia and Black Sea coasts. Two guilds of reophilic species dominate, with common share of 58%. Next groups are limnophilic litophilic and limnophilic phytophilic species of moderate tolerance (22% and 11% respectively). The Generalists share in fish community is moderate – 9%. A higher share of limnophilic guilds, which are tolerant to higher temperatures and lower oxygen concentrations in these highland rivers is connected with lower altitudes and warm climate conditions of South European regions.

Type 10. Mediterranean lowland (Figure 24, 25)
This river-type is distributed across Mediterranean region, especially in costal zones and in Southern Spain. Maximum height of NDS site is 572 m a.s.l. with an average of 231 m a.s.l. River slopes are quite high – up to 20,5‰ (average 7,7‰), due to the costal character of numerous rivers in this type. Limophilic groups strongly dominate in these rivers (48%), with 41% of Limnophilic phytophilic species of moderate tolerance. Two guilds of reophilic species (water column and benthic) constitute together only 24%, highly rheophilic species are absent. Next group are benthic species of moderate tolerance (8%). The generalist share in fish community is high – 20%. A high share of limnophilic guilds and generalists, which are tolerant to higher temperatures and lower oxygen concentrations in these rivers is connected with the warm climate conditions and periodic draughts characteristic of the South Mediterranean zone.

Type 11. Western European and Atlantic rivers (Figure 24, 25)
In this type, a set of lowland and costal streams of Atlantic region is grouped. For NDS sites altitude reaches up to 220 m a.s.l. (average 76 m a.s.l.). Slope values are very high, reaching 181‰ with an average of 24.6‰. Rivers of this type are located in the Atlantic environmental zone from Ireland, through England to North-central France, Belgium and Holland. The fish community is composed only of three guilds: Highly rheophilic, intolerant species and Rheophilic benthic species, making 55% and 27% respectively, with a substantial share of generalists – 18%. Fish assemblage corresponds to stream character with extremely high slope gradients, but a high share of generalists reflects the lowland location of this river type.

Type 12. Lowland medium sediment and organic rivers (Figure 24, 25)
This type comprises lowland and organic rivers and streams located mainly in Central and Eastern European plains and in Scandinavia. NDS sites are located up to 351 m a.s.l. (average 115 m a.s.l.) and characterized by low slope - up to 10.3‰ (average 2.15‰). Those rivers are distributed across Central Europe – from Austria, Switzerland and Germany through Poland, to Ukraine and Belarus, in the Continental bio-geographical region. Another group of sites is located in Finland, Latvia, Estonia, Russia and locally in Sweden and Norway – in the Boreal region. The fish community of these rivers is higly diversified (nine guilds). It is composed of 35% of highly rheophilic intolerant species, while rheophilic benthic and water column species represent 9% and 18% respectively. All four rheophilic fish guilds constitute together 66% of fish community, while two limnophilic guilds contribute to 10%. Benthic species of moderate tolerance represent 7% and intolerant water column species 5%. Generalists are quite abundant with a 12% share in fish community. This complex guild structure reflects high habitat variability in rivers and streams distributed widely across the Continental and Boreal bio-geographical regions.

Type 13. Boreal-Atlantic large-medium sediment rivers (Figure 24, 25)
Type 13 groups upland and mountain Boreal rivers and streams, located in Scandinavia, the Alpine North Environmental Zone. Chosen NDS sites are placed on altitudes up to 530 m a.s.l. (average 337 m a.s.l.), with variable slope values – maximum of 30‰, and an average of 5.7‰. Those rivers are

distributed mainly in Norway and Sweden in costal and mountaineous regions, in the Alpine North bio-geographical region. The fish assemblage is specific, with domination of highly rheophilic intolerant species (48%), a substantial share of rheophilic water column species (16%) and intolerant water column species (24%). The last guild present in rivers of this type are benthic species of moderate tolerance with a 12% share. Generalists are absent and all guilds found have high environmental demands. Domination of rheophilic and lithophilic guilds is connected to coarse bottom substrate in those rivers and severe climatic conditions of the Alpine North zone.

Type 14. Altantic medium-large sediment rivers (Figure 24, 25)
This lowland and costal river type is distributed across the Atlantic North biogeographical region, except Northern Scotland. Maximum height of NDS site is 71 m a.s.l. with an average of 51 m a.s.l. River slopes are quite high – up to 22.6‰ (average 11.3‰), due to the costal character of a number of rivers in this type. Rheophilic groups strongly dominate in these rivers, making altogether 76%, with 48% of highly rheophilic, intolerant species. Two other guilds of reophilic species (water column and benthic) constitute 16% and 12% respectively. The generalists share in fish community is high – 24%. Domination of rhoeophilic guilds is connected to noticeable slopes and coarse sediment, while high share of generalists result from lowland locations of rivers.

Type 15. North Altantic lowland, medium-large sediment rivers (Figure 24, 25)
This lowland and costal river type is located exclusively in Northern Scotland. The highest location of NDS site is 266 m a.s.l. with an average of 103 m a.s.l. River slopes are moderate – up to 14.6‰ (average 9.3‰). Two rheophilic groups strongly dominate in these rivers, making altogether 73%, with 55% of highly rheophilic, intolerant species and 18% reophilic water column species. The generalists share in fish community is very high – 27%. Similarly as in type 14, domination of rhoeophilic guilds is connected to considerable river slopes and occurrence of coarse sediment, while a high share of generalists result from lowland locations of rivers and specific organic geology.

3.4 Guild specific habitat suitability criteria

Three suitability classes, 0 – not important, 0.5 – partially important and 1 – important, were established in order to estimate the level of influence of each attribute on every fish guild and assess the guild specific suitability criteria (*GSI*) for each attribute (**Table 9**):

1. **Highly rheophilic, intolerant species** – require high water velocity, gravel bottom substrate and interstitial space, low water trophy and temperature and high oxygen concentrations. These species require good cover and longitudinal connectivity, therefore this guild is very sensitive to most factors related to impoundment and river continuum disruption.
2. **Rheophilic benthic species, preferring sandy-gravel bottom substrate** – this guild shows similar preferences to habitat attributes as the previous one. In addition, these species require increased depth, the presence of rheophilic macrophytes and mosses as well as habitat stability. This guild is also strongly vulnerable to impounding.
3. **Rheophilic water column species, preferring sandy-gravel bottom substrate** – show contrary preferences to those described above, and benefit from more depth and slightly slower velocity. This guild has relaxed requirements regarding water quality; however, these species require gravel areas and interstitial spaces for spawning, as well as cover and habitat stability. Barriers therefore have both positive and negative effects on this guild.
4. **Limnophilic benthic species of moderate tolerance** – this guild is associated with lentic habitats with low water velocity, high depth and soft bottom sediment. It has low water

quality and habitat continuity requirements. Therefore, this fish group benefits from impoundment and shows low vulnerability to barrier effects.

5. **Limnophilic water column species of moderate tolerance** – this group is associated with soft bottom substrates, macrophytes and floodplain water bodies. It tolerates habitat fragmentation and instability, so it is not sensitive to the impact of barriers and can easily accommodate impoundment conditions.
6. **Intolerant, rheophilic benthic species, preferring detritus or pelal bottom substrate**– this guild is composed of several lamprey species with specific biology. Feeding habitats in rivers are specific for detritivorous larval stadium, adults highly differ in feeding strategy, as parasitic forms (usually marine) or do not feed (resorbed alimentary track) as short living stage. This guild requires moderate to high water velocity, accompanied by shallow margins or backwaters with more lentic conditions. It needs muddy or detritus substrate, good water quality and oxygen conditions. This group is dependent on natural hydromorphologic conditions and vulnerable to habitat modifications, especially changes in water depth and substrate composition. It is also sensitive to river fragmentation; therefore, it is highly sensitive to barrier effects.
7. **Intolerant, water column species** – this guild needs good water quality, low temperature cover and preserved longitudinal connectivity. It also needs moderate water velocity and coarse sediment and is sensitive to changes in composition towards sand and mud. Therefore, it is rather vulnerable to the effects of barriers.
8. **Limnophilic lithophilic species of moderate tolerance** – the guild is associated with coarse bottom substrate with less regard to low water velocity and higher depth. It requires some shelter, especially woody debris. This guild is also very sensitive to river fragmentation and habitat instability.
9. **Limnophilic phytophilic species of moderate tolerance** – this guild prefers lentic habitats with aquatic vegetation, low water velocity, greater depth and soft bottom sediment. It tolerates increased water trophy, higher temperatures and lower oxygen content. The guild is less sensitive to disruption of longitudinal river continuity, but it is strongly dependent on floodplain waterbodies, so lateral connectivity is highly important. In general, this guild benefits from river impoundment, especially if the floodplain waterbodies remain accessible.
10. **Benthic species of moderate tolerance** – this guild is associated with medium water velocity and depth and bottom habitats. It prefers coarse bottom sediment and is strongly dependent on shelter. It has moderate requirements to water quality and habitat continuity, but requires stable habitat conditions. Therefore, this guild is affected by river impoundment.
11. **Generalists – tolerant species** – this guild has no clear habitat condition preferences. It tolerates lentic and moderately lotic habitats, preferring higher depth and the presence of aquatic vegetation. It is not affected by change of substrate composition towards soft bottom sediments. The guild tolerates high water trophy, higher temperatures and low oxygen concentration. It is not sensitive to disruption of longitudinal and lateral river continuity or to unstable habitat conditions. Clearly, this guild may benefit from river impoundment.

As can be seen above, different aspects of barriers affect each guild in different ways. The sum value in the bottom line of **Table 9** reflects the level of importance of all listed habitat attributes for a given fish guild. This value is high for specialist guilds and lowest for generalists. However, it only reflects the strength of guild reaction to attribute change, not the direction of response. So impoundment can be seen as having a strong effect on some groups through the loss of lotic habitats and environmental stability, as well as a disruption in river continuity, while other guilds (generalists and limnophilic phytophilic species), can, to some extent, benefit from changes caused by the presence of the barrier. Therefore, the guild response to a barrier can be either negative or positive.

As described above, the fish guild composition varies for different FCMacHT (**Figure 24, 25**), so the assessing barrier impacts on rivers with regard to habitat suitability and availability should be type specific. This was achieved by using the proportion of fish guilds for each of 15 FCMacHT to calculate weighted importance of all 21 habitat attributes on fish community characteristics for an undisturbed river.

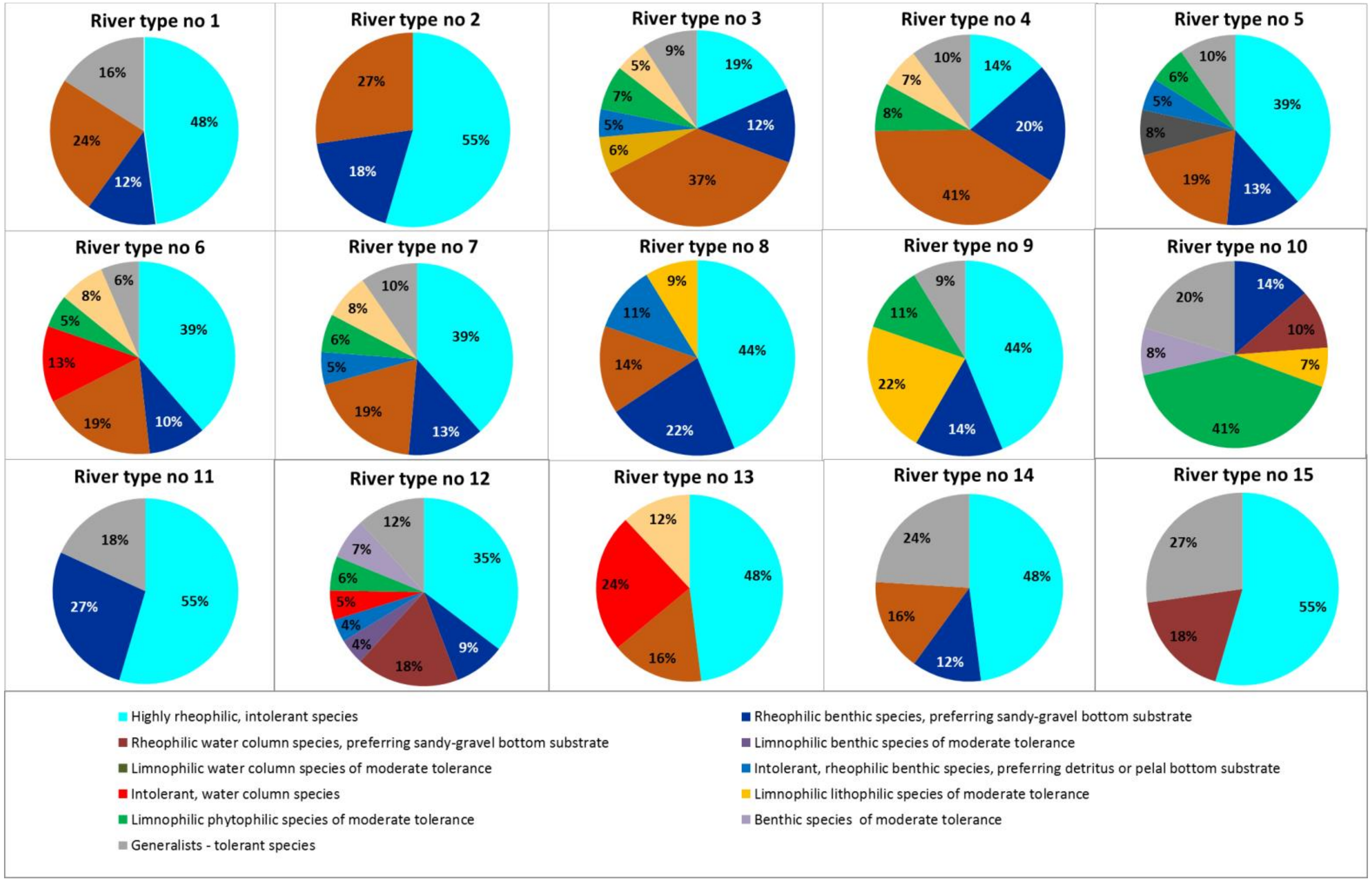


Figure 25. Fish guilds as a percentage for each of 15 FCMacHT. Guilds are ordered from more rheophilic and intolerant to generalists – tolerant species. (D2.1, version 2.0).

3.5 Quantification of barriers impacts on fish habitat

Table 9 shows that highly rheophylic intolerant species and intolerant, rheophilic benthic species, preferring detritus or pelal bottom substrate are the most sensitive to the occurrence of the habitat attributes considered, with an overall score of 13. In contrast, generalist species are least sensitive to physical factors, and their populations are regulated by other mechanisms such as food availability. All rheophylic species are more sensitive to physical habitat attributes.

With regards to the overall impact of barrier type on changes in habitat area, the results in **Table 10** indicate that the strongest change is expected from dams. Other barrier types have less pronounced effects, and ramps were found to have the lowest influence. This is expressed by higher number of “no impact” cases in the bottom row of **Table 10**, ranging from 0 for dams and weirs to 13 and 15 for ramps and culverts respectively.

Consequently, the lowest ***wRHp*** is at dams and weirs (**Table 11**), although it is not the same for all river types. The most impacted are Mountain, Alpine and subalpine, Mediterranean mountain and upland and Boreal-Atlantic large-medium sediment rivers. The lowest level of habitat impact is expected for Mediterranean lowland – moderate impact of dams and a bit stronger, significant impact of weirs. Low impact level is also characteristic for Atlantic and North Atlantic lowland, medium-large sediment rivers communities. This trend can be observed across all barrier types. Weirs indicate similar patterns like dams with a bit lower impact. The exception is observed for Mediterranean lowland rivers, which show less remaining habitat for weirs than for dams. The largest habitat areas remain with ramps indicating only slight habitat loss for most river types, except the three most sensitive mentioned (moderate impact). The fords produce mainly moderate habitat loss cases. For most river types, it is at a border line, and for three types – even in a low habitat loss class. For the majority of culvert cases, moderate habitat loss is estimated. Exceptions are Mountain, Alpine and subalpine and Boreal-Atlantic large-medium sediment rivers, which show significant habitat reduction. In contrast, for Mediterranean lowland rivers the model estimates slight impact level. For sluice habitat loss level is diversified from moderate (Mediterranean lowland) through significant (eight river types) to major (six river types – mainly of mountaineous and Boreal character).

Table 9. Guild specific suitability (***GSI***) to habitat attributes: 0 – not important; 0.5 – moderately important; 1 – very important.

Habitat attributes		FCMacHT guilds - suitability										
		Highly rheophilic, intolerant species	Rheophilic benthic species, preferring sandy-gravel bottom substrate	Rheophilic water column species, preferring sandy-gravel bottom substrate	Limnophilic benthic species of moderate tolerance	Limnophilic water column species of moderate tolerance	Intolerant, rheophilic benthic species, preferring detritus or pelal bottom substrate	Intolerant, water column species	Limnophilic lithophilic species of moderate tolerance	Limnophilic phytophilic species of moderate tolerance	Benthic species of moderate tolerance	Generalists - tolerant species
	Guild no.	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**
1	high velocity	1	0,5	0,5	0	0	1	1	0,5	0	0,5	0
2	low velocity	0	0	0,5	1	1	0,5	0	1	1	0	0,5
3	deep areas	0,5	0	1	1	1	0	1	0,5	0,5	0,5	1
4	shallows	0,5	0,5	0,5	0,5	1	1	0	0	1	0	0
5	interstitial space	1	1	0,5	0	0	0,5	0,5	0,5	0	1	0
6	sand	0	0	0	0,5	1	0,5	0	0	1	0	0,5
7	mud	0	0	0	1	1	0,5	0	0	1	0	1
8	gravel	1	1	1	0	0	0,5	1	1	0	1	0
9	boulder	1	1	0,5	0	0	0,5	0,5	0,5	0	0,5	0
10	woody debris	1	0,5	1	0,5	0	1	1	1	0,5	1	0
11	oxygenated water	1	0,5	0,5	0	0	1	1	0,5	0	0,5	0
12	cold water	1	1	0,5	0	0	1	1	0	0	0,5	0
13	low trophic level	1	1	0,5	0,5	0,5	0,5	1	0,5	0	0,5	0
14	rheophilic macrophytes, mosses	0,5	1	0,5	0,5	0	0	1	0	0	1	0
15	macrophytes	0	0	0	0,5	0,5	0	0	0	1	0	0,5
16	canopy shading	1	0,5	1	0	0	1	0,5	0,5	1	0,5	0
17	overhanging vegetation	0,5	0	0,5	0	0	0,5	0	0	1	0	0
18	undercut bank	0,5	1	1	0,5	0	0,5	0,5	0,5	0	1	0
19	floodplain accessibility	0	0	0	0	1	0,5	0	0	1	0	0,5
20	habitat continuity	1	1	1	0	0	1	1	1	0,5	0,5	0
21	habitat stability	0,5	1	0,5	0,5	0	1	0,5	0,5	0,5	1	0
	Sum = Habitat sensitivity	**13**	**11,5**	**11,5**	**7**	**7**	13	**11,5**	**8,5**	**10**	**10**	**4**

Table 10. Impact of different barrier types on habitat area with habitat attributes: 0 – major reduction, 0.5 – slight reduction; 1 – no change; 1.5 – slight increase; 2.0 – major increase.

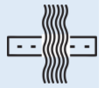
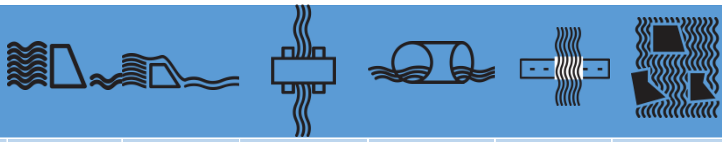
Habitat attributes		Barrier impact					
		Dam	Weir	Sluice	Culvert	Ford	Ramp
Barrier pictogram							
1	high velocity	0	0	0	0.5	0.5	0.5
2	low velocity	2	1.5	1.5	1.5	1.5	1.5
3	deep areas	2	1.5	1.5	1	1	1.5
4	shallows	0	0	0	1	1	1
5	interstitial space	0	0	0	0.5	0.5	0.5
6	sand	2	1.5	1.5	1.5	1	1
7	mud	2	1.5	1.5	1.5	1	1
8	gravel	0	0	0	0.5	1	0.5
9	boulder	0	0.5	0.5	1	1	1.5
10	woody debris	0	0.5	0.5	0.5	1	1
11	oxygenated water	0	0.5	0.5	1	1	1
12	cold water	0	0	0	1	1	1
13	low trophic level	0	0.5	0.5	1	1	1
14	rheophilic macrophytes, mosses	0	0	0	0.5	0.5	0.5
15	macrophytes	2	1.5	1	1	1	1
16	canopy shading	0	0.5	0.5	1	1	1
17	overhanging vegetation	0	0	0.5	1	1	1
18	undercut bank	0	0	0.5	0.5	1	1
19	floodplain accessibility	0	0.5	0.5	1	1	1
20	habitat continuity	0	0	0.5	0	0.5	0.5
21	habitat stability	0	0.5	0.5	1	0.5	1
	Number of "no change" cases	**0**	**0**	**1**	**11**	**15**	**13**

Table 11. Weighted remaining habitat proportion (***wRHp***) with regard to barrier type and FCMacHT. Red – severe habitat loss (≤10), orange – major habitat loss (11-50%), yellow – significant habitat loss (51-75%), green – moderate habitat loss (76-90%), blue – low habitat loss (>90).



no.	River FCMacHT type	Dam *wRHp* %	Weir *wRHp* %	Sluice *wRHp* %	Culvert *wRHp* %	Ford *wRHp* %	Ramp *wRHp* %
1	Highland, medium sediment	38	46	52	81	91	94
2	Mountain, Alpine and subalpine	11	28	37	73	85	89
3	Central European lowland, medium sediment	42	49	56	83	90	95
4	Central European lowland, large-medium sediment	38	45	53	80	89	94
5	Highland and lowland, large-medium sediment	42	49	55	84	90	95
6	Boreal large-medium sediment	27	39	46	77	87	91
7	Boreal lowland	33	43	50	80	88	93
8	Mediterranean mountain and upland	13	30	38	74	85	89
9	South European highland	36	46	52	81	89	94
10	Mediterranean lowland	78	74	77	97	95	100
11	Western European and Atlantic	36	45	50	82	87	93
12	Lowland medium sediment and organic	41	48	54	82	89	94
13	Boreal-Atlantic large-medium sediment	13	30	38	72	84	88
14	Altantic medium-large sediment	50	54	59	85	90	96
15	North Altantic lowland, medium-large sediment	57	59	63	87	91	98

4 DISCUSSION AND CONCLUSIONS

The framework and models developed here meet the requirements of the WFD, pointed out by the ECOSTAT Working Group by quantitatively defining the impact of barriers on fish communities and distinguishing between specific facilities and regions. The impact of each barrier type on habitat attributes (*Bi*) was defined for all river types. However, ecological fish groups that occurred in an earlier EFC and their sensitivity to habitat attribute modifications were used to calibrate it. In the final w*RHp* formula, the type specific barrier impact was calculated using the fish guild proportion (*GP*) for each FCMacHT. The model did not take into account the technical and operating conditions of the facility and the resulting impact on habitat conditions. Therefore, **Table 11** presents a picture where impact is differentiated by expected fish community types, but does not include all potential impacts on habitat continuity and stability. These impacts are grouped into two functional categories:

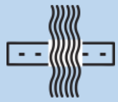
1. Migration impact caused by technical inadequacies at the location and by facilities located downstream of the investigated one: dams, weirs and sluices without a fishpass, purging culverts, step forming fords, inadequate construction of ramps (**Table 12**). These factors cause disruption of migratory routes and, for anadromous fish species, a single permanent barrier between sea and spawning grounds results in total population decline. The presence of barriers on a catchment scale is also detrimental for potamodromous fish when feeding habitats are separated from spawning areas.

2. Loss of habitat stability caused by the operating condition of the barrier facility: hydropeaking, lack of environmental flows or thermal alterations. These factors influence both migratory and local fish species by rapidly changing water levels, modifying natural flow regimes and physio-chemical water parameters. The extent of such changes depend on the barrier operational scheme used by the water management authority **(Table 12).**

We propose to incorporate a migration penalty of 25% in order to allow for the additional impacts, crucial for most vulnerable groups - diadromous, potamodromous fish especially. This will provide incentives for reaching higher classes of ecological potential through the introduction of appropriate mitigation measures. **Table 13** presents the *wRHp* after introducing the penalty for all cases. We consider that all barriers can be significant in generating major impact categories. In five cases for dams and three for weirs, we observed severe habitat loss. In all other cases for dams, weirs and sluices, major habitat loss occurs, with an exception of Mediterranean lowland rivers, showing lower, but significant impact for dams and sluices. For this river type, the highest value of *wRHp* 75% for ramps was also observed.

Table 12. Description barriers used in Conceptual Model with migration penalty **-** where HMWB is not able to achieve a GEP.

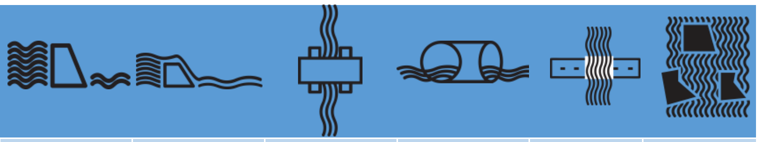
	BARIER DESCRIPTION	PHOTO	PICTOGRAM
1	**Dam** - a barrier that blocks or constrains the flow of water and raises the water level. **Not reaching the requirements for GEP due to:** Lack of fish passage.		

BARIER DESCRIPTION		PHOTO	PICTOGRAM
2	**Weir** - a barrier aimed at regulating flow conditions and water levels. **Not reaching the requirements for GEP due to:** Lack of fish passage.		
3	**Sluice** - a movable barrier aimed at controlling water levels and flow rates in rivers and streams. **Not reaching the requirements for GEP due to:** Lack of fish passage. Sluice built on unpassable weir.		
4	**Culvert** - a structure aimed at carrying a stream or river under an obstruction. **Not reaching the requirements for GEP due to:** Lack of connection to riverbed and substrate, with the water depth and velocity not passable for fish all year. Culvert crest is blocking the upstream migration.		
5	**Ford** - a structure in a river or stream which creates a shallow place for crossing the river or stream by wading or in a vehicle. **Not reaching the requirements for GEP due to:** Unpassable and blocking the river most of the time, and can be passed only during high flow period.		

	BARIER DESCRIPTION	PHOTO	PICTOGRAM
6	**Ramp** - a ramp or a bed sill is a structure aimed at stabilizing the channel bed and reducing erosion and is recognizable by its stair-like shape. **Not reaching the requirements for GEP due to:** Unpassable and blocking the river most of the time, and can be passed only during high flow period.		

Photos of barriers (from top): DAM: 1- I. Wagner (ERCE); WEIR: 2 – AMBER; SLUICE: 3 - Z. Kaczkowski (ERCE); CULVERT: 4 – AMBER; FORD: 5 – AMBER; RAMP: 6 – SSIFI.

Table 13. Migration penalized weighted remaining habitat proportion (*wRHp*) with regard to barrier type and FCMacHT. Red – severe habitat loss (≤10), orange – major habitat loss (11-50%), yellow – significant habitat loss (51-75%), green – moderate habitat loss (76-90%), blue – low habitat loss (>90).



no.	River FCMacHT type	Dam *wRHp* %	Weir *wRHp* %	Sluice *wRHp* %	Culvert *wRHp* %	Ford *wRHp* %	Ramp *wRHp* %
1	Highland, medium sediment	13	21	27	56	66	69
2	Mountain, Alpine and subalpine	0	3	12	48	60	64
3	Central European lowland, medium sediment	17	24	31	58	65	70
4	Central European lowland, large-medium sediment	13	20	28	55	64	69
5	Highland and lowland, large-medium sediment	17	24	30	59	65	70
6	Boreal large-medium sediment	2	14	21	52	62	66
7	Boreal lowland	8	18	25	55	63	68
8	Mediterranean mountain and upland	0	5	13	49	60	64
9	South European highland	11	21	27	56	64	69
10	Mediterranean lowland	53	49	52	72	70	75

11	Western European and Atlantic	11	20	25	57	62	68
12	Lowland medium sediment and organic	16	23	29	57	64	69
13	Boreal-Atlantic large-medium sediment	0	5	13	47	59	63
14	Altantic medium-large sediment	25	29	34	60	65	71
15	North Altantic lowland, medium-large sediment	32	34	38	62	66	73

This conceptual model shows the expected impact of a single barrier on local habitat conditions with all possible mitigation measures applied (including barrier passability), as shown in **Table 11**. It also allows for an estimate of how this impact is increased by multiple barriers. For a set of barriers located in a single water body (defined according to WFD), we recommend calculating the cumulative impact by multiplying the proportion values of remaining habitat, calculated for each barrier. In those cases the wRHp indicates that a "GEP" defined for those water bodies and water uses would imply a poor quality of habitat availability and suitability (which has to be validated by actual biological monitoring). For example, one dam and one weir reduce the remaining habitat for Mountain Alpine and subalpine rivers to 3% of the impounded stretches (severe habitat loss), despite maintained barrier passability. In a case of Mediterranean lowland rivers, such loss is considerably smaller, but significant – 58% remaining habitat in impounded stretches. However, it should be noted that even a single permanent barrier may generate severe or major impact (**Table 13**). Hence, the presented model may serve as a tool for adaptive management of barriers towards fulfilling the requirements of WFD.

Although standardisation at European level of the concepts and methods developed here is clearly necessary, solutions for mitigation measures will have to be largely site-specific. Thus, our conceptual model of ecological impacts of barriers in EU considering habitat selection criteria for running waters is only a starting point to elaborate site-specific scenarios, especially in terms of climate change scenarios for Europe (Roudier *et al.,* 2016, van Vliet *et al.,* 2015; Kundzewicz *in prep.*). We predict that future changes in the magnitude and duration of hydrological droughts will show contrasting patterns across Europe. For large areas of Italy, France, Spain, Greece, the Balkans, Ireland, and the UK, droughts are expected to become more frequent and last longer (Roudier *et al.,* 2015), mainly due to a reduction in rainfall and higher evapotranspiration. Such predictions are robust for southern France, parts of Spain, Portugal and Greece. For the rest of Europe, changes in droughts are not expected to be significant or there is a reduction in their length and magnitude. This is especially the case in northern Fenno-Scandinavia and Western Russia, where modelled changes are very robust. Results achieved by van Vliet *et al.* (2015) show a distinct north–south divide in terms of climate change impacts; in the south, water availability will be reduced, while in the north it will increase. Moreover, across different climate models, precipitation and streamflow will increase in northern Europe and decrease in southern Europe.

Our conceptual model is based on the Ecohydrology concept, and is defined as a sub-discipline of hydrology that focuses on ecological processes occurring within the hydrological cycle and attempts to utilise such processes to enhance environmental sustainability. The concept of Ecohydrology (Zalewski, 2000), considers a hydrologic pattern as the primary driver in river ecosystem functioning (e.g., Minshall, 1988; Zalewski and Naiman, 1995; Zalewski, 2000), shaping the template of the geomorphic habitat of the river, and being directly linked with the ecosystem carrying capacity (Southwood, 1977; Frissell *et al.,* 1986; Townsend and Hildrew, 1994; Hudson, 2002; Thorp *et al.*, 2010). Ecohydrology not only provides scientific understanding of the interplay between hydrology and biota, but also a systemic framework on how to use ecosystem processes as a new tool for river basin management, complementary to hydrotechnical solutions already applied, like barriers on rivers. Thus, especially when a barrier cannot be removed, the Ecohydrolohy concept and tools (ecohydrological biotechnologies) help to harmonize existing or a new build infrastructure with ecosystem and societal needs (WBSRC - **W**ater, **B**iodiversity, Ecosystem **S**ervices, **R**esilience, **C**ultural Heritage - Zalewski, 2014).

The approach presented here should help to better assess the impacts of barriers on ecosystems. It is an important step for the achievement of good ecological potential (GEP) of heavily modified water bodies (HMWB) as required by the WFD. It also accommodates recent scenarios of climate change for Europe (Zalewski, 2010ab), after calibration for different regions due to differences in climate, geomorphology, human impact and cultural heritage. More detailed models developed in the AMBER case studies should allow for better stratification of the penalty level associated with flow alterations at barriers in different FCMacHT classes.

For the river basin management practice under the WFD, we see the potential contribution of the developed concept and its results as follows:

The proposed concept of "weighted remaining habitat proportion" (wRHp) with its focus on (fish) habitat availability and suitability can serve as an indicator for the expected value of the biological quality element (BQE) "fish fauna" (assuming that there are no other, non-barrier related effects on fish fauna) and using an "umbrella group" approach also for other BQE-s. Of course the actual status of the fish and other BQE-s needs to be validated by biological monitoring.

If a water body is completely undisturbed (no barrier), 100% of the habitat is present for the reference fish community. Assuming that there are no other relevant effects, the wRHp thresholds/ranges could then provide indications with regard to the different classes of the fish fauna in the context of water body classification, i.e. wRHp > 90% could potentially indicate a fish fauna in line with High Ecological Status, wRHp 76-90%, a fish fauna in line with Good Ecological Status and so on. One value-added of the modelling approach could be in predicting changes of the fish fauna quality before implementing new modifications to the river. For, example, if a new barrier is to be built in a water body with good ecological status, the wRHp could be used to estimate the degree of deterioration and the expected class of the fish fauna, also taking into account possible mitigation measures. It may detect cases leading to a deterioration of a BQE class from good to moderate, which is important for the non-deterioration principle stated in the WFD.

4.1 Divagation regarding potential application of wRHp concept in Water Framework Directive

The wRHp concept can also add value with regard to HMWB designation and ecological potential. In this context the specific framework for HMWB designation has to be taken into account. Basically, the designation of HMWB under the WFD enables river basin management to balance ecological restoration and water use interest in a specific water body. According to Article 4.3 WFD two conditions have to be met to designate a HMWB:

a. water uses such as drinking water provision or power generation would suffer "significantly negative effects" from targeting a good ecological status which would apply in a natural water body.
b. these uses cannot be substituted by alternative means which are technically feasible, a significantly better environmental option and are not disproportionately costly.

If both conditions are met, the water body can be classified as HMWB and instead of the "good ecological status" the target is "good ecologic potential" (GEP). Depending on the actual and specific modifications (e.g. existing dam), the GEP for the water body needs to be derived on a case by case basis. The GEP is to be derived from the maximum ecological potential (MEP), as conditions of GEP are described in the WFD as "there are slight changes in the values of the relevant biological quality elements as compared to the values found at maximum ecological potential". And the conditions for MEP are in turn determined by the physical conditions with result from heavy modification of a water body; "the values of the relevant biological quality elements [should] reflect, as far as possible, those associated with the closest comparable surface water body type, given the physical conditions which result from the artificial or heavily modified characteristics of the water body". With regard to the supporting hydromorphological quality elements this requires that "hydromorphological conditions are consistent with the only impacts on the surface water body being those resulting from the artificial or heavily modified characteristics of the water body once all mitigation measures have been taken to ensure the best approximation to ecological continuum, in particular with respect to migration of fauna and appropriate spawning and breeding grounds." Assuming these theoretical considerations, in the example of the dam this means in basic terms, if dam removal is impossible – the MEP is constrained by the negative impacts of a dam (impoundment of river stretch changing the river characteristic and reducing fish habitat availability). In such a case, best approximation of river continuity has to be established based on mitigation measures (such as building a fish pass, or change dam operation to reduce habitat alteration). The key question arises: what are "all" mitigation measures which lead to the "best" approximation? Of course, technical feasibility seems a logical condition to define the set of "all" mitigation measures (for example, is it possible to design a fish pass for a dam of 150m height and what would be its effectiveness?). However, in addition, technically feasible mitigation measures have to be evaluated with respect to their impact on the water uses, which were the reason to designate a HMWB. In order to ensure a coherent concept of HMWB designation no mitigation measures should be implemented, which would result in a "significantly negative impact" on the relevant water uses because the threat of "significantly negative impacts" is the basis for a HMWB designation in the first place. The legal term of "significantly negative impacts" is quite unclear in the WFD and needs to be further specified by Member States. For example, in context of hydropower a "significantly negative impact" could be defined as a 5% loss of electricity production in a water body. Depending on the actual definition of significance certain (combinations of) mitigation measures would exceed significance. To avoid these compromises with regard to the set of mitigation measures, the "best" approximation could be made (e. g. a non-optimal fish pass design). Hence, the "best approximation" and therefore the resulting MEP and GEP, are not absolute standards in ecologic terms, but are dependant on the maximum set of mitigation measures that are technically feasible and avoid negatively significant effects for the relevant water uses. Reaching GEP might then in some cases include an optimal, state-of the art fish pass, and in other cases a fish pass with reduced effectiveness (e. g. fish elevators for high dams) or in extreme cases no fish pass at all (if the dam is too high). Assuming that all evaluations have been done properly and measures executed accordingly in all cases, the ecological result will be "good" ecological potential by definition, although the actual ecological quality with regard to the fish fauna can vary substantially.

Against this theoretical background, the wRHp concept and its classes cannot be used directly to assess whether a GEP is achieved or not. If designation of HMWB has been done accurately and mitigation measures properly evaluated, defined and executed, GEP is achieved by definition. However the wRHp concept and its classes can indicate the actual level of GEP deviation from conditions of GES in a specific water body. This approach is based on the resultant of modification and mitigation measures taken transferred into percentage of remaining habitat (wRHp). Even if all typical mitigation measures are taken (e.g. construction of upstream/downstream fish passes), the results of the study reveal that considerable habitat loss may occur (**Table 11**, for example, dam at highland river, only 38% remaining habitat). In addition, the penalty approach (though it does not yet reflect gradual compromises like a less effective type of fish pass), can account for cases where not all typical mitigation measures can be taken and the resulting level of GEP in relation to GES conditions is lower. As a consequence, the wRHP concept can illustrate that a (by definition) “good” ecological potential can be quite poor in ecologic terms with regard to fish fauna (although still to be validated by biological monitoring).

Another point to keep in mind for the practical application is that in our concept the calculated habitat loss refers to the impounded area only (e.g. from dam upstream until the beginning of the impoundment). However, in WFD practice a water body can be much longer and thus the impounded area may be only a part of it. The monitoring sites for the biological quality elements then sometimes are located within the impoundment and sometimes out of it. The adopted scheme of \their aggregation then leads to the status class of the respective water body.

In conclusion – the concept of wRHp has the potential to better inform decision making authorities in general or adaptive barrier management. The value-added of the wRHp approach with regard to river basin management is in particular:

- It can be used as an indicator to estimate the habitat loss and consequently fish fauna degradation resulting from existing and new river modifications such as a new dam construction and therefore deliver additional information for the **evaluation of the effects of barriers on ecological status and potential.**
- It can be used as an indicator to deliver additional information during HMWB designations (or their periodical reviews). The expected degradation of habitat and fish fauna under a “GEP” can inform the second condition of the designation test, which evaluates whether an alternative to the water use is a significantly better environmental option (ECOSTAT 2016).

5 REFERENCES

Balon, E. K. (1992) How dams on the River Danube might have caused hybridization and influenced the appearance of a new cyprinid taxon. *Environmental Biology of Fishes,* **33**, 1-2, 167-180.

Baudoin, J-M., Burgun, V., Chanseau, M., Larnier, M., Ovidio, M., Sremski, W., Steinbach, P., and Voegtle, B. (2014) The ICE protocol for ecological continuity. Assessing the passage of obstacles by fish. Concepts, design and application. ONEMA, The National Agency for Water and Aquatic Environments, ISBN: 979-10-91047-29-6.

Belmar, O. *et al.* (2013) Effects of flow regime alteration on fluvial habitats and riparian quality in a semiarid Mediterranean basin. *Ecological Indicators*, **30**, 52–64. doi: 10.1016/J.ECOLIND.2013.01.042

Benejam, L. *et al.* (2016) Ecological impacts of small hydropower plants on headwater stream fish: from individual to community effects. *Ecology of Freshwater Fish*, **25**(2), 295–306. doi: 10.1111/eff.12210

Bernez, I. *et al.* (2004) Combined effects of environmental factors and regulation on macrophyte vegetation along three rivers in western France. *River Research and Applications,* **20**(1), 43–59.

Brooks, S. S. *et al.* (2002) Assessing stream ecosystem rehabilitation: limitations of community structure data. *Restoration Ecology,* **10**(1), 156–168.

Bunn, S.E., Arthington, A.H. (2002) Basic Principles and Ecological Consequences of Altered Flow Regimes for Aquatic Biodiversity. *Environmental Management,* **30**, 492–507.

Brylińska M. 2000. Ryby Słodkowodne Polski. [Freshwater fishes of Poland]. PWN Warszawa, 521 pp.

Camargo, J. A., Alonso, A. and De la Puente, M. (2005) Eutrophication downstream from small reservoirs in mountain rivers of Central Spain. *Water Research,* **39**(14), 3376–3384.

Capers, R. S. (2003) Macrophyte colonization in a freshwater tidal wetland (Lyme, CT, USA). *Aquatic Botany*, **77**(4), 325–338.

CCM River and Catchment Database, version 2.1 (CCM2) [dataset]. European Commission - JRC, 2007.

Céréghino, R., Legalle, M. and Lavandier, P. (2004) Drift and benthic population structure of the mayfly Rhithrogena semicolorata (Heptageniidae) under natural and hydropeaking conditions. *Hydrobiologia*, **519**(1–3), 127–133.

Ceschin, S. *et al.* (2015) The effect of river damming on vegetation: is it always unfavourable? A case study from the River Tiber (Italy). *Environmental Monitoring and Assessment,* **187**(5). 301. doi: 10.1007/s10661-015-4521-7

Chao, B. F. (1995) Anthropogenic impact on global geodynamics due to reservoir water impoundment. *Geophysical Research Letters*, **22**, 3529-3532.

de Jalón, D.G *et al.* (2013) *REFORM - REstoring rivers FOR effective catchment Management.* Deliverable D1.2 Effects of pressures on hydromorphology. Review on pressure effects on hydromorphological variables and ecologically relevant processes. http://reformrivers.eu.

de Leaniz, C. G. (2008) Weir removal in salmonid streams: implications, challenges and practicalities. *Hydrobiologia,* **609**: 83–96.

Dean, J. *et al.* (2002) Effects of lowhead dams on freshwater mussels in the Neosho River, Kansas, Transactions of the Kansas Academy of Science. *BioOne,* **105**(3), 232–240.

Dudgeon, D. (2000). Going with the flow: Large-scale hydrological changes and prospects for riverine biodiversity in tropical Asia. *BioScience,* **50**, 793-806.

Dynesius, M. and Nilsson, C. (1994) Fragmentation and flow regulation of river systems in the northern third of the world. *Science,* **266**, 753-762.

EFI+, (2009) Manual for the application of the new European Fish Index - EFI+. A fish-based method to assess the ecological status of European running waters in support of the Water Framework Directive. June 2009. *EFI+ Consortium*. http://efi-plus.boku.ac.at/software/doc/EFI+Manual.pdf [With Annexes at: http://efi-plus.boku.ac.at/software/doc/Annexes.pdf]

ECOSTAT, 2016. WG ECOSTAT report on common understanding of using mitigation measures for reaching Good Ecological Potential for heavily modified water bodies. Part 1: Impacted by water storage. Drafted by ad hoc GEP/Water storage group for CIS WG ECOSTAT. Editors: E. KAMPA & S. DÖBBELT-GRÜNE, pp 90.

European Soil Data Centre (ESDAC), esdac.jrc.ec.europa.eu, European Commission, Joint Research Centre, accessed: 2018-02-16.

Fausch, K.D., Lyons, J., Karr, J.R., and Angermeier, P.L. (1990) Fish communities as indicators of environmental degradation. *American Fisheries Society Symposium,* **8**, 123–144.

Frissell, C.A., Liss, W.J., Warren, C.E., Hurley, M.D. (1986) A hierarchical framework for stream habitat classification: viewing streams in a watershed context. *Environmental Management,* **10**, 199-214.

Gillespie, B. R. *et al.* (2015) A critical analysis of regulated river ecosystem responses to managed environmental flows from reservoirs. *Freshwater Biology,* **60**(2), 410–425.

Growns, I.O. and Growns, J.E. (2001) Ecological effects of flow regulation on macroinvertebrate and periphytic diatom assemblages in the Hawkesbury – Nepean River, Australia. *Regulated Rivers Research & Management,* **17**, 275–293.

Harby, A. and Halleraker, J. H. (2001) Ecological impacts of hydro peaking in rivers. *International Journal of Hydropower and Dams*, **8**, 132–135.

Harrison, T.D., Whitfield, A.K. (2004) A multi-metric fish index to assess the environmental condition of estuaries. *Journal of Fish Biology*, **65** (3), 683–710. DOI: 10.1111/j.0022-1112.2004.00477.x

Hilderbrand, R. H. *et al.* (1997) Effects of large woody debris placement on stream channels and benthic macroinvertebrates. *Canadian Journal of Fisheries and Aquatic Sciences,* **54**(4), 931–939.

Hudson, H.R. (2002) Linking the physical form and processes of rivers with ecological response. Pp. 121-134, In: F.J. Dyer, M.C. Thomas & J. Molley (eds). The Structure, Function and management Implications of Fluvial Sedimentary Systems *(Proceedings of an International Symposium held at Alice Springs, Australia, September 2002).* IHAS Publ., **276**, 484 pp.

IHME1500, 2018. Spatial Data of the International Hydrogeological Map of Europe 1 : 1,500,000 in Vector Format (Version IHME1500 v1.1). Duscher, K., Günther, A. Federal Institute for Geosciences and Natural Resources (BGR), Hannover, Berlin, Germany.

Ittekkot, V., Humborg, C., and Schafer, P. (2000) Hydrological alterations and marine biogeochemistry: A silicate issue? *BioScience,* **50**, 776-782.

Junk, W.J., P.B. Bayley, and Sparks, R.E. (1989) The flood pulse concept in river-floodplain systems. p. 110-127. In D.P. Dodge [ed.] *Proceedings of the International Large River Symposium. Canadian Special Publication of Fisheries and Aquatic Sciences,* **106**.

Kelly, C. A., Rudd, J. W. M., St. Louis, V., and Moore, T. (1994) Turning attention to reservoir surfaces, a neglected area in greenhouse studies. EOS, *Transactions of the American Geophysical Union,* **75**, 332-333.

Kottelat M., Freyhof J. 2007. Handbook of European freshwater fishes. Publications Kottelat, Cornol, Switzerland. 646 p.

Laasonen, P., Muotka, T. and Kivijärvi, I. (1998) Recovery of macroinvertebrate communities from stream habitat restoration. *Aquatic Conservation: Marine and Freshwater Ecosystems*, **8**(1), 101–113.

Larson, M. G., Booth, D. B. and Morley, S. A. (2001) Effectiveness of large woody debris in stream rehabilitation projects in urban basins. *Ecological Engineering,* **18**(2), 211–226.

Lauters, F. *et al.* (1996) Influence of hydropeaking on invertebrates and their relationship with fish feeding habits in a Pyrenean river. *Regulated Rivers: Research & Management,* **12**(6), 563–573.

Lenat, D. R., Penrose, D. L. and Eagleson, K. W. (1981) Variable effects of sediment addition on stream benthos. *Hydrobiologia,* **79**(2), 187–194.

Madsen, J. D. *et al.* (2001) The interaction between water movement, sediment dynamics and submersed macrophytes. *Hydrobiologia,* **444**(1–3), 71–84.

Martínez, A. *et al.* (2013) Stream regulation by small dams affects benthic macroinvertebrate communities: from structural changes to functional implications. *Hydrobiologia,* **711**(1), 31–42.

Master, L. L., Flack, S. R., and Stein, B. A. (1998) Rivers of Life: Critical Watersheds for Protecting Freshwater Biodiversity. Arlington (VA): *Special Publication of the US Nature Conservancy.*

McCully, P. (1996) Silenced Rivers. The Ecology and Politics of Large Dams. London: Zed Books.

Metzger, Marc J. (2018). The Environmental Stratification of Europe, [dataset]. University of Edinburgh. https://doi.org/10.7488/ds/2356.

Minshall, G.W. (1988) Stream ecosystem theory: a global perspective. *Journal of North American Benthological Society,* **7**, 263-288.

Moura, A. *et al.* (2013) The role of a cascade of reservoirs and seasonal variation in the phytoplankton structure in a tropical river. *Brazilian Journal of Biology,* **73**(2), 291–298. doi: 10.1590/S1519-69842013000200009

Mueller, M., Pander, J. and Geist, J. (2011) The effects of weirs on structural stream habitat and biological communities. doi: 10.1111/j.1365-2664.2011.02035.x

Neraas, L. P., and Spruell, P. (2000) Fragmentation of riverine systems: the genetic effects of dams on bull trout (Salvelinus confluentus) in the Clark Fork River system. *Molecular Ecology,* **10**(5), 1153-64.

Nilsson, C., and Berggren, K. (2000) Alterations of riparian ecosystems caused by river regulation. *BioScience*, **50** 783-792.

Panagos P., Van Liedekerke M., Jones A., Montanarella L. 2012. "European Soil Data Centre: Response to European policy support and public data requirements"; Land Use Policy, 29 (2), pp. 329-338. doi:10.1016/j.landusepol.2011.07.003

Parasiewicz, P. (2003) Upscaling: Integrating habitat model into river management. *Canadian Water Resources Journal*. Special Issue: State-of-the-Art in Habitat Modelling and Conservation of Flows, **28**(2), 283-300.

Parasiewicz, P. (2007a) The MesoHABSIM Model Revisited. *River Research and Application,* **23**, (8), 893-903.

Parasiewicz, P. (2007b) Developing a reference habitat template and ecological management scenarios using the MesoHABSIM model. *River Research and Application,* **23**(8), 924-932.

Petts, G.E. (1984) Impounded rivers: perspectives for ecological management. Chichester: John Wiley, 326 pp.

Pont, D., Hugeney, B., Beier, U., Goffaux, D., Melcher, A., Noble, R., Rogers, C., Roset, N., Schmutz, S. (2006) Assessing river biotic condition at a continental scale: A European approach using functional meristics and fish assemblages. *Journal of Applied Ecology,* **43** (1), 70–80. DOI: 10.1111/j.1365-2664.2005.01126.x

Postel, S. L., Daily, G. C., and Ehrlich, P. R. (1996) Human appropriation of renewable fresh water. *Science,* **271**, 785-788.

Pringle, C. M. (1997) Exploring how disturbance is transmitted upstream: Going against the flow. *Journal of the North American Benthological Society*, **16**, 425-438.

Pringle, C. M., Freeman, M. C. and Freeman, B. J. (2000) Regional effects of hydrologic alterations on riverine macrobiota in the New World: Tropical-temperate comparisons. *BioScience,* **50**, 807-823.

Richardson, S. M., Hanson, J. M. and Locke, A. (2002) Effects of impoundment and water-level fluctuations on macrophyte and macroinvertebrate communities of a dammed tidal river. *Aquatic Ecology*, **36**(4), 493–510. doi: 10.1023/A:1021137630654

Roni, P. *et al.* (2006) Rehabilitation of bedrock stream channels: the effects of boulder weir placement on aquatic habitat and biota. *River Research and Applications,* **22**(9), 967–980.

Rosenberg, D. M., Berkes, F., Bodaly, R. A., Hecky, R. E., Kelly, C. A., and Rudd, J. W. M . (1997) Large-scale impacts of hydroelectric development. *Environmental Reviews,* **5,** 27-54.

Rosenberg, D. M., McCully, P. and Pringle, C. M. (2000) Global-Scale Environmental Effects of Hydrological Alterations: Introduction. *BioScience,* **50**(9), 746 -751. doi.org/10.1641/0006-3568(2000)050[0746:GSEEOH]2.0.CO;2

Roudier, P., Andersson, J. C. M., Donnelly, C., Feyen, L., Greuell, W. and Ludwig, F. (2016) Projections of future floods and hydrological droughts in Europe under a +2°C global warming. *Climatic Change*, **135**(2), 341-355.

Schinegger, R., Trautwein, C., Melcher, A., Schmutz, S. (2011) Multiple human pressures and their spatial patterns in European running waters. *Water and Environment Journal,* **26** (2), 261–273. DOI: 10.1111/j.1747-6593.2011.00285.x

Schmutz, S., Cowx, I.G., Haidvogl, G., Pont, D. (2007) Fish-based methods for assessing European running waters: A synthesis. *Fisheries Management and Ecology,* **14** (6), 369–380. DOI: 10.1111/j.1365-2400.2007.00585.x

Scrimgeour, G., Jones, N. and Tonn, W. M. (2013) Benthic macroinvertebrate response to habitat restoration in a constructed Arctic stream. *River research and applications,* **29**(3), 352–365.

SNIFFER. (2010a) WFD111 (2a) Coarse resolution rapid-assessment methodology to assess obstacles to fish migration. Field manual level A assessment. 78 pp. Available at: http://www.sniffer.org.uk/knowledge-hubs/resilient-catchments/waterframework-directive-and-uktag-co-ordination/fish-obstacles-porosity/ (accessed 02/09/2016)

SNIFFER. (2010b) WFD111 (2a) Trialling of the methodology for quantifying the impacts of obstacles to fish passage. 46 pp.

Southwood, T.R.E. (1977) Habitat, the templet for ecological strategies? *Journal of Animal Ecology,* **46**, 337-365.

St. Louis, V. L., Kelly, C. A., Duchemin, E., Rudd, J. W. M., and Rosenberg, D. M. (2000) Reservoir surfaces as sources of greenhouse gases to the atmosphere: A global estimate. *BioScience,* **50**, 766-775.

Tanneberger, F. et al. (2017) ‘The peatland map of Europe’, Mires and Peat, 19(22), pp. 1–17. doi: 10.19189/MaP.2016.OMB.264.

Thorp, J.H, Flotemersch, J.E., Delong, M.D., Casper, A.F., Thoms, M.C., Ballantyne, F., Williams, B.S., O’Neill, B.J., and Haase, C.S. (2010) Linking Ecosystem Services, Rehabilitation, and River Hydrogeomorphology. *BioScience,* **60**(1), 67-74.

Tiemann, J. S. *et al.* (2004) Effects of Lowhead Dams on Riffle-Dwelling Fishes and Macroinvertebrates in a Midwestern River. *Transactions of the American Fisheries Society,* **133**(3), 705–717.

Tikkanen, P. *et al.* (1994) Short-term recovery of benthos following disturbance from stream habitat rehabilitation. *Hydrobiologia,* **273**(2), 121–130.

Tockner, K., Malard, F., and J. Ward. (2000) An extension of the flood pulse concept. *Hydrological Process*, 2861-2883.

Townsend, C.R., Hildrew, A.G. (1994) Species traits in relation to a habitat templet for river systems. *Freshwater Biology,* **3**, 265–275.

van Vliet, M.T.H., Donnelly, Ch., Strömbäck, L., Capell, R., Ludwig, F. (2015) European scale climate information services for water use sectors. *J. Hydrol.,* **528**, 503-513.

Vannote, R. L., Minshall, G. W., , Cummins, K. W., Sedell, J. R., and Cushing, C. E. (1980) The River Continuum Concept. *Canadian Journal of Fisheries and Aquatic Sciences*, **37**(1), 130-137. doi.org/10.1139/f80-017

Vogt J V *et al.* (2007) A pan-European River and Catchment Database. *European Commission - JRC*, Luxembourg, (EUR 22920 EN) 120 pp.

Vörösmarty, C. J., and Sahagian, D. (2000) Anthropogenic disturbance of the terrestrial water cycle. *BioScience*, **50**, 753-765.

Wang, X. *et al.* (2013) Assessing impacts of a dam construction on benthic macroinvertebrate communities in a mountain stream. *Fresenius Environmental Bulletin.* Parlar Scientific Publications, Angerstr 12 Freising 85354 Germany, **22**(1), 103–110.

WCD, (2000) - World Commission on Dams, (2000) Dams and development. A new framework for decision-making. The report of the World Commission on Dams. Earthscan Publications Ltd., London, UK. 404 pp.

Welcomme, R.L. (1985) River Fisheries. *FAO Fisheries Technical Paper 262*, FAO, ROME.

Welcomme, R.L. (2001) Inland Fisheries: Ecology and Management. Fishing News Books/Blackwell Science, 358 pp.

WFD Intercalibration (2011) Phase 2: *Milestone report – October 2011.* European Commission Directorate General, JRC Joint Research Centre, Institute of Environment and Sustainability. JRC documents, ss. 105.

Wildman, L., Parasiewicz, P., Katopodis, Ch. And Dumond, U. (2003) An Illustrative Handbook on Nature-Like Fishways - Summarized Version. *American rivers*, Washington , DC. 21pp.

Winter, H.V., and Van Densen, W.L.T. (2001) Assessing the opportunities for upstream migration of non-salmonid fishes in the weir-regulated River Vecht. *Fisheries Management and Ecology*, **8**, 513-532.

Wu, N., Tang, T., Fu, X., Jiang, W., Li, F., Zhou, S., Cai, Q. and Fohrer, N. (2010) Impacts of cascade run-of-river dams on benthic diatoms in the Xiangxi River, China. *Aquatic Sciences,* **72**, 117-125.

Zalewski, M. (2000) Special Issue on Ecohydrology. Ecological Engineering (The Journal of Ecotechnology) 16 nr 1, Ecohydrology - the scientific background to use ecosystem properties as management tools toward sustainability of water resources. Guest Editorial, *Ecological Engineering,* **16**, 1-8.

Zalewski, M. (2002) Ecohydrology-the use of ecological and hydrological processes for sustainable management of water resources. *Hydrological Sciences Journal,* **47**(5), 825-834

Zalewski, M. (2010a) Ecohydrology for compensation of Global Change. *Brazilian Journal of Biology. Suppl.,* **70**(3): 689-695.

Zalewski, M. (2010b) Ecohydrology for implementation of the EU water framework directive. *Water Management,* 1-11, doi: 10.1680/wama.1000030.

Zalewski, M. (2014b). Ecohydrology and Hydrologic Engineering: Regulation of Hydrology-Biota Interactions for Sustainability. *Journal of Hydrologic Engineering,* 10.1061/(ASCE)HE.1943-5584.0000999, A4014012.

Zalewski, M., Naiman, R. (1985) The regulation of riverine fish communities by a continuum of abiotic-biotic factors. In: Alabaster, J.S. (ed.) FAO UN. Habitat Modification and freshwater Fisheries. *Proceedings of Symposium of the EIFAC FAO*. Butterworths Scientific, pp. 3-9.

6 APPENDIX A: REVIEW OF BARRIER IMPACT ON BIOTA

Table 14. Descriptions cited after authors

Reference	Study Location	IMPACT ON BIOTA (acc. to WFD BQEs)		
		Fish	**Macroinvertebrates**	**Macrophytes**
		DAM		
Addy *et al.* 2012. *Science of the Total Environment* 432: 318-328	Scottish montane river		Impacts of flow regulation on freshwater pearl mussel (*Margaritifera margaritifera*): stable and suitable substrate habitats for adult mussels exist despite the potential geomorphic effects of impoundment but a degree	

			of compaction was observed that could adversely affect juvenile mussel habitat.	
Almodóvar & Nicola. 1999. *Regul. Rivers: Res.Mgmt.* 15: 477-484	Hoz Seca, Tagus Basin, Spain	The downstream estimated population densities and biomass of trout showed a decrease of about 50 and 43%, respectively, following regulation. Examination of length-for-age tables revealed no obvious change in growth but a significant difference in age structure. The main consequence of the imposed fluctuating flow regime was a serious reduction in trout production caused by a loss of suitable habitat and a loss of juveniles.	The effects of disturbance on benthic macroinvertebrates were also analysed but no changes in abundance were detected	
Almeida *et al.* 2009. *Int. Rev. Hydrobiol.* 94: 179-193	Atlantic forest area of south-eastern Brazil		The dam did induce changes in the composition of benthic communities, especially in the dry period. However, overall, the fauna seemed to be able to persist during periods when the flow was absent.	
Angilletta *et al.* 2008. *Evolutionary Applications* 1: 286-299	4 streams in Washington and Oregon, USA	Chinook salmon likely suffered a decrease in mean fitness after the construction of a dam in the Rogue River.		
Antonio *et al.* 2007. *Neotropical Ichthyology* 5 (2): 177-184	Parana River, Brazil	The fish were caught in the dam forebay downstream, marked, and released upstream and downstream. Nearly half of the recaptures downstream occurred in tributaries, indicating that in the presence of an obstacle the fish are able to locate alternative migration routes. The remainder stayed in the main channel of the Parana River, at a mean distance of less than 50 km from the release point. Of the fish released upriver from the dam, approximately half were recaptured downriver. Although the river was only partly dammed, the movement of the fish downriver suggests that they became disoriented after being transferred.		
Archer *et al*. 2008. In: *Sustainable Hydrology for the 21st Century*. Proceedings of the 10th BHS	River Tyne, North-East England	Not all salmon move in response to a water release from the reservoir and some remain in the estuary until a natural spate occurs.		

National Hydrology Symposium, Exeter				
Armitage 2006. *River Research and Applications* 22: 947-966	Northern England		Nineteen of the 31 common taxa in the regulated sites declined in abundance by a factor of 5 or more.	
Baldigo & Smith 2012. *River Research and Applications* 28: 858-871	Northern New York		Function and apparent health of macroinvertebrate communities were generally unaffected by atypical flow regimes and/or altered water quality at study reaches downstream from both dams in the Indian, Cedar and Hudson Rivers. The lentic nature of releases from both impoundments, however, produced significant changes in the structure of assemblages at Indian and Cedar River sites immediately downstream from both dams, moderate effects at two Indian River sites 2.4 and 4.0 km downstream from its dam, little or no effect at three Cedar River sites 7.2–34.2 km downstream from its dam, and no effect at any Hudson River site.	
Barrella & Petrere. 2003. *River Research and Applications* 19 (1): 59-76	Tietê and Paranapanema rivers, Brazil	The damming of the superior reaches of Paranapanema River provoked a decrease in fish diversity.		
Bejarano *et al.* 2010. *Freshwater Biology* 56 (5): 853-866	Boreal stream in northern Sweden.			Trees and shrubs had migrated towards the mid-channel along the entire study reach, but the changes were largest immediately downstream of the dam. Species richness after regulation increased for trees but decreased for shrubs.
Benitez-Mora & Camargo. 2014. *Hydrobiologia* 728: 167-178	Central Spain		Abiotic changes, particularly the downstream nutrient enrichment, apparently affected the macroinvertebrate	Total coverage and taxa richness of submersed macrophytes increased downstream from dams.

				communities. Total density and total biomass of benthic macroinvertebrates increased downstream, but taxa richness tended to decrease. Scrapers appeared to be the macroinvertebrate feeding group most favoured downstream from dams.	
	Benjankar *et al.* 2012. *Ecological Engineering* 46: 88-97	Kootenai River, USA.			A dynamic vegetation model was used to analyse the change in floodplain area occupied by individual vegetation types and vegetation succession dynamics as consequences of river modification and dam operation. The area occupied by colonization and cottonwood young transition vegetation decreased gradually from the pre-dam to post-dam conditions. In contrast, the cottonwood old transition vegetation area increased significantly from the pre-dam to post-dam conditions.
	Benstead *et al.* 1999. *Ecol. Appl*. 9: 656–668	Rio Espiritu Santo, Puerto Rico, USA		The low-head dam caused large numbers of post larval shrimps to accumulate directly downstream of the structure. Mortality of drifting first-stage larvae by entrainment into the water intake during downstream migration averaged 42% during the 69-d study period.	
	Birnie-Gauvin *et al.* 2017. *Journal of Environment al Management* 204 (1): 467-471	Gudenaa River, Jutland, Denmark	The dam removal has led to a dramatic increase in trout density, especially in young of the year. Surprisingly, this increase was not just upstream of the barrier, where the ponded zone previously was, but also downstream of the barrier, despite little changes in habitat in that area.		

	Boles 1981. *Hydrobiologia* 78: 133-146	Trinity River, northwestern California, USA		Temperature alteration by the hypolimnial release reservoir prevented colonization by all but a few organisms able to tolerate such conditions in the riffles below the dam. Large but unstable populations of *Simulium, Baetis,* and the Chironominae developed below the dam, aided by an abundant periphyton and detrital food supply and lack of competition and predation.	
	Bradford *et al.* 2011. *Freshwater Biology* 56: 2119-2134	Bridge River, British Columbia, Canada	The total number of salmonids did increase after the instream flow release, but most of the gains could be attributed to the rewatering of a previously dry channel located immediately below the dam. In reaches that had flowing water during the baseline period, the response of individual salmon species to the increase in flow was variable, and there was little change in total abundance after the flow release. Results were inconsistent with both habitat modelling, which predicted a decrease in habitat quality with increasing flow, and holistic instream flow approaches, which imply greater benefits with larger flows.		
	Branco *et al.* 2014. *J. Appl. Ecol.* 51: 1197-1206	Tagus River, Spain	The proposed prioritization method, using spatial graphs and habitat suitability modelling, makes it possible to model the impact of the removal or placement of an insurmountable barrier on the overall functional connectivity of a river network, facilitating resource allocation and minimizing the impact of new barrier implementation.		
	Brandimarte *et al.* 2016. *Braz. J. Biol.* http://dx.doi.org/10.1590/1519-6984.16814	Mogi-Guaçu river, São Paulo State, Brazil		Damming impact on the Chironomidae was indicated by the reduction of both genera richness in the margins and relative abundance of groups typical of faster waters.	

	Bredenhand & Samways. 2009. *J. Insect Conserv*. 13: 297–307	Eerste River, Cape Floristic Region, South Africa.		Macroinvertebrate species diversity below the dam was only half of that in the pristine catchment area above the dam. Furthermore, Ephemeroptera, Plecoptera and Trichoptera diversity and abundance dropped to almost zero as a result of the impoundment. In contrast, the abundance of the Diptera family Chironomidae increased substantially below the dam.	
	Burroughs *et al.* 2010. *Trans. Am. Fish. Soc.* 139: 1595-1613	Pine River, Michigan, USA	Following removal, 8 species formerly found only below the dam utilized newly available portions of the river above the dam. Most fish species (18 of the 25 evaluated) showed an increase in abundance following removal, strongly supporting the idea that dam removal reduces multiple factors limiting riverine fishes. Brown trout and rainbow trout were the primary sport fishes present in the river, and the abundance of both species increased by more than twofold over the course of the study. The abundance of white suckers also increased significantly due to increased reproductive success. The results of this study illustrate how dam removal is a useful tool for restoration of habitat connectivity and habitat conditions and how the fish community in a coldwater stream responded to the removal.		
	Casado *et al.* 2016. *Geomorphology* 268: 21-34	Sauce Grande River, Argentina.			Whilst the unregulated river exhibited active lateral migration with consequent adjustments of the channel shape and size, the river section below the dam was characterized by (i) marked planform stability (93 to 97%), and by (ii) vegetation encroachment leading to alternating yet localized contraction

				of the channel width (up to 30%).
Catalano *et al.* 2007. *North Am. J.Fish. Manag.* 27: 519-530	Baraboo River, Wisconsin, USA	After dam removal, 10 of the 11 species were collected at new sites upstream from the dam, indicating that recolonization of reconnected upstream sites had occurred. Some species recolonized rapidly and in large numbers.		
Caudill *et al.* 2013. *Plos One* 8, e85586	Snake River, Washington, USA	Gradients in fishway water temperatures present a migration obstacle to many anadromous migrants. Unfavourable temperature gradients may be common at reservoir-fed fish passage facilities, especially those with seasonal thermal layering or stratification. Understanding and managing thermal heterogeneity at such sites may be important for ensuring efficient upstream passage and minimizing stress for migratory, temperature-sensitive species.		
Cereghino & Lavandier. 1998. *Regulated Rivers-Research and Management* 14: 297-309	River Oriege, French Pyrenees.		The lowest densities and biomasses of Plecoptera were estimated at 700 m downstream from the plant, underlining the prominent role of hydropeaking.	
Clarke *et al.* 2008. Canadian Technical Report of Fisheries and Aquatic Sciences 2784.	Review	Flow management practices can have direct (survival) and indirect (growth, reproduction, bioenergetics) impacts on fishes. Specific impacts on fishes resulting from flow alteration included changing total gas pressure, habitat access, stranding, water temperature, nutrient dynamics, bioenergetics and food supply. Flow management also affects physical habitat, altering erosion and deposition patterns resulting in changes in habitat structure and cover.		
Cortes *et al.* 1998. *Hydrobiologia* 389: 51-61	The Poio and Balsemão rivers in Northern Portugal		The composition of the invertebrate fauna was only clearly modified downstream of the impoundment used to divert water to a small town (with longer retention of the water). Decrease of fauna diversity was more pronounced below the	

				impoundment used for hydro-power generation reflecting the stress caused by the relatively frequent fluctuations in water flow.	
	Cumming . 2004. *Ecol. Appl.* 14: 1495-1506	Wisconsin, USA	Although downstream dams have a significant effect on fish species richness, this effect is small by comparison to the influence of water quantity and summer maximum temperatures.		
	Dorobek *et al.* 2015. *River Systems* 21: 125-139	2 rivers of urban Columbus, Ohio, USA	Dam removal may act as a pulse disturbance with quantitative short-term impacts on fish assemblages. Fish responses to dam removal likely operate along a temporal trajectory wherein short-term responses will be critical in shaping longer-term responses.		
	Doyle *et al.* 2005. *Geomorphology* 71: 227-244	Wisconsin, USA		Each of the ecosystem attributes responded to the disturbance of dam removal in different ways and recovered at very different rates, ranging from months to decades. Riparian vegetation appeared to require the greatest time for recovery, while macroinvertebrates had the least. Mussel communities were the most adversely affected group of species and showed no signs of recovery during the time period of the study.	
	Finch *et al.* 2015. River Res. Applic. 31: 156-164	Colorado River, USA	Results are counterintuitive and show that more natural steady flows water release reduced growth rates of juvenile humpback chub compared with fluctuating flows when both treatments occurred within the same year.		
	Franchi *et al.* 2014. *Journal of Limnology* 73: 203-210	Tiber River, Italy	The structure of the communities downstream from Montedoglio Reservoir changed radically after the construction of the barrier, while the fish community upstream from the barrier appeared unaffected by the dam. The release of hypolimnetic water had a marked impact on the fish community of the Tiber River because it interrupted the typical longitudinal zonation of the species in the river. In addition, this reservoir can pose an increased threat to the native fish species, since it is a source of diffusion of various exotic species.		
	Gehrke *et al.* 2002. *River Research and Applications* 18: 265–286	Shoalhaven River, Australia	Species richness was greater downstream of the dam, with 21 species, compared to 16 species upstream of the dam. Ten diadromous species are believed to be extinct above the dam because of obstructed fish passage.		

		Another four migratory species capable of climbing the wall have reduced abundances upstream. Accumulations of fish, particularly juveniles, directly below the dam were evident for nine species. Fish communities upstream and downstream of the dam differed significantly, identifying the dam as a significant discontinuity in the available fish habitats within the system.		
Gibbins *et al.* 2001. *Fisheries Management and Ecology* 8: 463–485	Kielder Reservoir, Scotland	Physical Habitat Simulation (PHABSIM) models suggest that the reservoir compensation flow provides adequate habitat for older 0+ fish but limits the availability of spawning habitat. Transfers of water to the River Wear result in short-term changes in the abundance of certain invertebrate species, although there is no evidence that the river's invertebrate fauna has been permanently altered. Consequently, it is unlikely that fish are affected indirectly by transfers through reductions in the abundance of their invertebrate prey.		
Gillespie *et al.* 2015. *River Research and Applications* 31: 953-963	River Humber in north-east England		This study has identified key impacts of regulation on macroinvertebrate community composition in upland sites within a large British catchment, specifically: reduced relative abundance of Coleoptera and Ephemeroptera and enhanced relative numbers of Trichoptera, Chironomidae and Oligochaeta.	
Głowacki *et al.* 2011. *River Research and Applications* 27: 612-629	Warta River, Poland	Fish exerted consumption pressure on chironomids upstream by foraging mostly on benthic insects, but not downstream where they fed on microcrustaceans of reservoir origin or on epiphytic fauna (mainly Chironomidae). The reservoir impact on chironomids decreased their diversity upstream and increased downstream, while much the opposite was true in fish.		
Głowacki & Penczak. 2012. *Journal of Fish Biology* 80: 2213–2235	Warta River, Poland	The results indicate that dam reservoirs may be oscillators of fish diversity and that part of the literature controversy over the effect of a reservoir on fish populations may be due to a too short sampling period: it is shown that both increase and decrease in diversity may be observed on the basis of samples selected from one diversity time series obtained at the tailwater.		
Godinho & Kynard. 2009. *J. River Research and Applications*	Brasil	In addition to providing passage for pre-spawning migrants, upstream fishways also provide passage for other fish migrations (e.g. foraging), and all up- and downstream migrations during life history need to be addressed at		

	25 (6): 702-712		dams to conserve fish resources. An upstream fishway is important even if the upstream reach does not have spawning or nursery habitats.		
	Gore. 1980. *Hydrobiologia* 69: 33-44	Tongue River in Montana and Wyoming		The hypolimnion release reservoir and associated downstream areas act as barriers, both thermal and geographical, to aquatic insect dispersal resulting in an impoverished benthic community in thermally recovered areas downstream of the reservoir.	
	Grant. 2001. *Hydrological Processes* 15: 1531–1532	Author 's opinion on dam removal	From past work on the effects of dams on rivers, we know that not all dams are created equal. The same will be true of dam removal: some will stimulate dramatic effects on river and ecosystem processes, others will have no effects, and some may open Pandora's boxes of new problems. The latter will be particularly true in cases where reservoirs are filled with sediments contaminated by organic or inorganic compounds, such as the PCB-contaminated sediments.		
	Grzybkowska *et al.* 2012. *Fauna Norvegica* 31: 25-33	3 streams in central Poland		In the upstream reaches due to small fluctuations of abiotic parameters (discharge) large populations of a small number of chironomid species (mainly Chironomini) dominated in the benthos, while in the downstream reaches a moderate disturbance enabled a much higher number of species to develop and coexist, but at a lower level of density than at the upstream sites (the Bzura and Mrożyca Streams). In turn when the size of inorganic substrate particles was larger (gravel) at the downstream site not only a higher number of species but also their higher density than in upstream site (the Mroga Stream) might be observed.	

	Grzybkowska *et al.* 2017. *Ecological Engineering* 99: 256–264	Warta River (Poland	In the tailwater of the Warta River, SAM and their associated organisms disappeared at the end of each summer as an effect of water level management in the reservoir. In addition, reconstruction of this diverse and rich biota began every year at the end of May, resulting in a similar abundance of the main biological groups, such as benthos, epiphytes and zooplankton. This phenomenon applies to all of the above-mentioned ecological groups, except for fish assemblages with domination of roach, perch and ruffe, which change in terms of abundance from season to season.		
	Halleraker *et al.* 2007. *River Research and Applications* 23: 493-510	River Surna, Norway	A temperature-adjusted running of the hydropower plant was found to have a major influence on the production of salmonids. Downstream of the hydropower station, rapid ramping has often occurred. This is possibly harmful for juvenile fish inhabiting the shallow parts of the river due to stranding.		
	Hansen & Hayes. 2012. *River Research and Applications* 28: 1540-1550	Michigan and Wisconsin rivers, United States		Generally, the macroinvertebrate community recovered 3–7 years following dam removal both in terms of taxonomic similarity and richness, although densities could take decades to recover.	
	Helms *et al.* 2011. *Journal of the North American Benthological Society* 30: 1095-1106	20 streams, Alabama, USA	Fish assemblages are influenced by small, low-head mill dams in south-eastern US streams and these structures continue to influence fish after they have been breached. Fish recovery from the effects of dams was evidenced by longitudinal similarity of reaches in streams with relict dams. This evidence suggests fish assemblages above an existing structure probably would not benefit from its breaching but might benefit from its removal. In contrast, removal of breached structures could pose significant risks to other sensitive species downstream.		
	Horne *et al.* 2004. *River Research and Applications* 20: 185-203	Manistee River, Michigan, USA	Considering only temperature effects, bottom withdrawal provides the greatest promise for increasing natural steelhead recruitment by decreasing the likelihood of year-class failures in the warmest summers.		

	Jansson *et al*. 2000. *Ecology* 81: 899–903	Northern Sweden			The effect of dams as barriers to plant dispersal along rivers was assessed by comparing the flora of vascular plants between pairs of run-of-river impoundments in northern Sweden. Adjacent impoundments in similar environmental settings develop different riparian floras because species with poor floating capacity become unevenly distributed among impoundments. Such discontinuities were not found along a free-flowing river, suggesting effective dispersal of riparian plants in the absence of dams.
	Käiro *et al.* 2011. *River Research and Applications* 27: 895-907	Lowland streams, Estonia		Damming affected the stream biota significantly and negatively in cases where fine sediments were accumulated above the dam.	
	Kanehl *et al.* 1997. *North Am. J. Fish. Manag.* 17: 387-400	Milwaukee River, Wisconsin, USA	Five years after dam removal, habitat quality was good to excellent, smallmouth bass abundance and biomass had increased substantially, common carp abundance and biomass had declined dramatically, and biotic integrity was good.		
	Kil & Bae. 2012. *Anim. Cells Syst*. 16: 69-76	Korean stream		Even a partial removal of a dam, resulting in increased substrate diversity in the upper site, could sufficiently help rehabilitate lost ecological integrity of streams without major habitat changes.	
	Kingsford. 2000. *Austral Ecology* 25: 109–127	Floodplain wetlands in Australia			Dams diversions and river management have reduced flooding to wetlands, altering their ecology, and causing the death or poor health of aquatic biota. This has

					changed wetland biota from one tolerant of a variable flooding regime, to one that withstands permanent flooding.
	Korman & Campana. 2009. *Transactions of the American Fisheries Society* 138: 76-87	Colorado River, Arizona, USA	Hourly variation in flow caused by hydropeaking alters patterns of nearshore habitat use for age-0 rainbow trout and reducing hourly variation in flow can lead to increased otolith growth.		
	Kruk & Penczak. 2003. *Annales de Limnologie - International Journal of Limnology* 39 (3): 197-210	Warta River, Poland	Not only obligatory riverine species suffer from the effect of the dam. Facultative riverine eel, burbot, wels and pike considerably decreased in number and standing crop, including their apparent absence in some years following the damming. Nevertheless, median densities of perch and roach, two generalists thriving in European regulated rivers, increased from tens and hundreds, respectively, to thousands per hectare. Significant increases in density were also recorded for other four species: zander, ruffe, bream and silver bream.		
	Kuby *et al.* 2005. *Adv. Water Resour.* 28: 845-855	Willamette River, Oregon, USA	A variant of the basic model considered fish-passage systems to be effective at connecting upstream and downstream habitat without removing the dam and losing its economic benefits. Because the fish-migration benefits of dam removal depend on which other dams are removed within the river system, multi-dam analysis is important.		
	Kui *et al.* 2017. *Ecohydrology* 10:e1839	Bill Williams River, Arizona, USA.			In the decades after the dam was built, woody plant cover within the river's bottomland nearly doubled, narrowing the active channel by 60% and transforming its planform from wide and braided to a single thread and more sinuous channel. Compared with native

				cottonwood-willow vegetation, non-native tamarisk locally induced a twofold greater reduction in channel braiding.
Lessard & Hayes. 2003. *River Research and Applications* 19: 721–732	Michigan, USA	Increasing temperatures below surface release dam coincided with lower densities of several cold-water fish species, specifically brown trout, brook trout and slimy sculpin while overall fish species richness increased downstream. Density of mottled sculpin, another cold-water species, was not related to temperature changes below the dams.	Macroinvertebrates showed shifts in community composition below dams that increased temperature.	
Li *et al.* 2012. *Forest Ecology and Management* , 284: 251–259	Middle-lower Lancang-Mekong River, China			Cascading hydropower dams can enhance habitat fragmentation, reduce the distribution ranges (latitude and altitude) of primary vegetation and reduce the complexity of the vegetation types along the river as well as induce the loss of primary vegetation in the whole watershed.
Lucas *et al.* 2009. *Freshw. Biol.* 54: 621–634	River Derwent, North East England	Access in to the Derwent appeared severely restricted by a tidal barrage, beyond which lamprey migrated rapidly in unobstructed reaches. Of all lamprey tagged in the lower 4 km of river, or ascending the barrage, 64% and 17% passed the first and second weirs respectively, with high flows crucial for this. Although over 98% of lamprey spawning habitat occurred more than 51 km upstream, on average just 1.8% of river lamprey.		
Lucas *et al.* 2016. *Ecosphere* 7 (5), (e01235)	Santa Ana River, California, USA			Successional changes in the Santa Ana River floodplain due to the construction of the Seven Oaks Dam has had a deleterious impact on the potential long-term survival of endangered subshrub (*Eriastrum densifolium*).

Mackay & Waters. 1986. *Ecology* 67: 1680-1686	Valley Creek, Minnesota, USA		Annual production by filter-feeding caddisfly larvae was significantly higher below impoundments than above. The most likely factor was postulated to be an increase in abundance or quality of seston. Other factors were better growths of moss and filamentous algae, which provided attachment sites for retreat construction, and the absence of abrasive sand after deposition in impoundments.	
Magilligan *et al.* 2016. *Geomorphology* 252: 158-170		Dam removal has provided important results and insights. It shows that ecological benefits can be achieved rapidly: (i) several species, which were never observed upstream of the former dam, have now made it upstream of the former barrier, and (ii) sea lamprey are constructing redds up to and beyond the former dam - all within the first year of removal.		
Maloney *et al.* 2008. *Freshw. Biol.* 53: 1055–1068	Fox River, Illinois, USA	Following the breach fish assemblage only slightly shifted to free-flowing characteristics 3 years after the breach.	Following the breach, relative abundance of Ephemeroptera, Plecoptera and Trichoptera increased, whereas relative abundance of Ostracoda decreased, in the former impoundment to levels comparable to free-flowing sites. Overall macroinvertebrate assemblage structure shifted to a characteristically free-flowing assemblage 2 years following the breach.	
Marchant & Hehir. 2002. *Freshwater Biology* 47 (5): 1033-1050	19 dams in South-east Australia		All the dams seemed to cause much the same disruption to the fauna. Of the families predicted to have widespread occurrence, 9–12 were found at most sites (tolerant taxa), while a larger number (14–24) were missing (intolerant taxa). The limited recolonisation below dams may well be because of the fact that dams act as barriers to	

				drift, the most prominent route for invertebrate colonists.	
	Martínez *et al.* 2013. *Hydrobiologia* 711: 31-42	5 streams in Northern Spain		The regulation by surface release small reservoirs negatively affects density, richness and diversity of macroinvertebrates in the reaches below the dam. The main driver is probably the variability of the flow regime due to the absence of any ecological flow, which creates droughts below the dam in summer. Furthermore, downstream reaches are characterized by loss of riparian forest quality and reduction of benthic habitat heterogeneity.	
	McCarthy *et al.* 2008. *Hydrobiologia* 609: 109-124	Shannon River, Ireland	Decline in juvenile eel recruitment resulting from the installation of hydroelectric facilities. In addition to the long-term effects the hydroelectric facilities have had on the stock levels, there is also an annual effect on the migratory patterns of downstream migratory silver eels.		
	Merritt &Cooper. 2000. *Regulated Rivers: Research and Management* 16: 543–564	Green River, north-western Colorado, USA.			Vegetation patterns reflect a dichotomy in moisture conditions across the floodplain on the regulated Green River: marshes with anaerobic soils supporting wetland species and terraces having xeric soil conditions and supporting communities dominated by desert species. The probable long-term effects of channel and hydrologic changes at Browns Park include the eventual replacement of *Populus*-dominated riparian forest by drought tolerant desert scrublands, and the enlargement

					of in-channel fluvial marshes.
	Merritt *et al.* 2010. *Freshwater Biology* 55: 206–225	Model			Riparian vegetation composition, structure and abundance are governed to a large degree by river flow regime and flow-mediated fluvial processes. Streamflow regime exerts selective pressures on riparian vegetation, resulting in adaptations (trait syndromes) to specific flow attributes. Widespread modification of flow regimes by humans has resulted in extensive alteration of riparian vegetation communities.
	Mortenson & Weisberg. 2010. *Global Ecology and Biogeography* 19 (4): 562-574	South-western USA.			Rivers with a large drainage area and low flow variability are inherently more vulnerable to invasions. River regulation does not necessarily increase the cover of non-native, invasive species. Instead, changes in flow allow proliferation of species that have life-history traits suited to modified flow regimes.
	Muraoka *et al*. 2017. Boulder *Limnologica* 62:188–193	Experiment	Diverse arrangement of boulders is required on individual rocky ramp fishways to allow the movement and migration of multiple species.		
	Noonan *et al*. 2012. *Fish Fish*. 13: 450-464	Review	Downstream passage efficiency was 68.5%, slightly higher than upstream passage efficiency of 41.7%, and neither differed across the geographical regions of study. Salmonids were more successful than non-salmonids in passing upstream (61.7 vs. 21.1%) and downstream (74.6 vs. 39.6%) through fish passage facilities. Passage efficiency differed significantly between types of fishways; pool and weir, pool and slot and natural fishways had the		

		highest efficiencies, whereas Denil and fish locks/elevators had the lowest. Upstream passage efficiency decreased significantly with fishway slope, but increased with fishway length, and water velocity.		
Orr *et al.* 2008. *River Research and Applications* 24: 804-822	Boulder Creek, Wisconsin, USA		Algal and invertebrate populations increased in the weeks after the dam removal, but did not reach densities similar to the upstream reference reach. In the following year, both periphyton and invertebrate densities were lower in the downstream reach, suggesting a long-term effect of the removal.	
Osmundson *et al.* 2002. *Ecol.Appl.* 12: 1719–1739	Colorado River, USA	Numbers and biomass of fish corresponded with biomass of detritus, periphyton, and invertebrates, strongly suggesting that their numbers are limited by available food. Flows of sufficient magnitude are frequently required to winnow silt and sand from the bed and transport it downstream where it can be deposited on floodplains or channel margins. However, river regulation, primarily in the headwaters has reduced the magnitude of these flows during the past 50 years and thereby reduced the frequency of flushing events.		
Ovidio & Philippart. 2002. *Hydrobiologia* 483: 55–69	River Meuse, Belgium	Fish were captured several weeks before their spawning migrations and tagged with radio-transmitters. Some small obstacles are not as insignificant as initially thought and can significantly disrupt and/or obstruct their upstream movements.		
Ovidio *et al.* 2007. *Fish. Manag. Ecol.* 14: 41–50	Belgium rivers	The ability of trout and grayling to pass different typologies of physical obstacles in natural river systems is discussed in the context of enabling their free movement in rivers.		
Paller & Saul. 1996. *Environmental Biology of Fishes* 45: 151-160.	Savannah River, USA	The timing and spatial pattern of gizzard shad spawning were markedly affected by a temperature gradient caused by the release of hypolimnetic water from an upstream reservoir.		
Pelicice *et al.* 2015. *Fish*	Review, South America.	Large reservoirs are important barriers to fish migration in South America. Dams generally prevent		

Fish. 16: 697-715		upstream movements, whereas reservoirs impede mainly downstream movements.		
Penczak. 1992. *Hydrobiologia* 242: 87-93	Warta River, Poland	The mean biomass and production of fish populations in upstream and downstream sites of the Jeziorsko Reservoir, in four subsequent years after impoundment, never reached values as high as those before impoundment. Despite considerable differences between the morphological characters of the two sites, similar changes in fish biomass and production were recorded.		
Penczak & Kruk. 2005. *Journal of Applied Ichthyology* 21: 169-177	Warta River, Poland	The self-organizing map definitely proved profound changes in fish assemblage composition downstream of the dam: most lithophilous species declined and many phytolithophilous and phytophilous species became dominants, particularly in the tailwater site where downstream migration of 0+ of successfully spawned species from the reservoir took place.		
Penczak *et al.* 2006. Acta Oecologica 30: 312-321	Warta River, Poland		The deepest habitats below the dam were most resistant to water level fluctuations, while the formerly most productive habitat at the tailwater site became the most negatively impacted. Nevertheless, the reservoir has not negatively influenced chironomid density, because the latter increased closer the mid-river, where large patches of macrophytes developed.	
Penczak *et al.* 2012. Ecological Modelling 227: 64– 71	Warta River, Poland		The study indicated that the dam without a fish ladder has an essential impact on fish assemblage structure, which is more pronounced in the tailwater. However, the present study indicated that the rate of assemblage changes caused by the impoundment has much	

				abated in the first decade of the 21st century.	
	Pess *et al.* 2008. *Northwest Science* 82: 72-90	Elwha River, Washington , USA	Dams has disrupted salmon migration and reduced salmon habitat by 90%. Several historical salmonid populations have been extirpated, and remaining populations are dramatically smaller than estimated historical population size.		
	Principe. 2010. *Int. J. Limnol.* 46: 77-91	Mountain streams, Cordoba, Argentina		Small dams may have impact on the function and structure of the invertebrate community, though hydraulic habitats would not be affected. Changes in richness and diversity, in the abundance of filterers and shredders and in the abundance of certain species were shown. Small dams generate quite small reservoirs and their permeability may be greater than those of large dams.	
	Quinones *et al.* 2014. *Rev Fish Biol Fisheries*	California, USA	Net positive effects of dam removals are most likely in two situations. The first is where salmonid populations below the dams are still large making them more resilient to the temporary negative impacts of dam removal such as sedimentation. The second is where populations are imperilled but adverse effects of dam removal are few.		
	Renöfält *et al.* 2013. *River Research and Applications* 29: 1082-1089	Swedish stream		Dam removal reduced some macroinvertebrate taxa at the downstream site, but no effect on community composition was found. Reduction of taxonomic richness was also found. Some dam-removal effects persisted or even increased over time. The most likely explanation for the suppression of benthic macroinvertebrate richness following dam removal is a significantly increased sediment transport from the former reservoir and a subsequent loss of preferred substrates. Results indicate that adverse dam-removal	

			effects may be long lasting but taxon specific.	
Roberts *et al.* 2013. *Freshwater Biology* 58: 2050-2064	3 rivers, Virginia and North Carolina, USA	Study of *Percina rex* suggests that, in the absence of hydrological barriers, benthic, non-migratory species can undertake surprisingly extensive gene flow. Hydroelectric dams drastically accelerated genetic differentiation relative to historical, riverine connections.		
Sankey *et al.* 2015. *Journal of Geophysical Research: Biogeosciences* 120 (8): 1532-1547	Colorado River, Arizona, USA			Net increase in vegetated area since completion of the dam. Magnitude and timing of vegetation changes are river stage-dependent. Vegetation expansion is coincident with inundation frequency changes and is unlikely to occur for time periods when inundation frequency exceeds approximately 5%. Short pulses of high flow do not keep vegetation from expanding onto bare sand habitat.
Santos *et al.* 2005. *J. Appl. Ichthyol.* 21: 381-388.	Lima River, Portugal	Bypass discharge explained most of the variation in the number of cyprinids, whereas water temperature was more important fo diadromous species. This study proved the efficacy of the bypass for passage of almost all occurring species and life stages and also for providing suitable habitat for fish fauna..		
Santos *et al.* 2014. *Ecol. Eng.* 73: 335–344.	Experiments, Portugal	Fishway passage success was discharge-related and independent of boulder density. However, a high density boulder configuration combined with higher fishway discharge can be beneficial as it reduced fish transit time.		
Santucci *et al.* 2005. *North American Journal of Fisheries* 25: 975-992	Fox River, Illinois, USA	Little evidence of cumulative effects of dams was found; however, data suggest that low-head dams adversely affect warmwater stream fish and macroinvertebrate communities by degrading habitat and water quality and fragmenting the river landscape.		

	Sethi *et al.* 2004. *Hydrobiologia* 525: 157–165	Koshkonong Creek, Wisconsin, USA		Removal of the dam led to mortality both within the former impoundment and in downstream reaches. Transport and deposition of reservoir sediments likely contributed to downstream mussel mortality.	
	Shafroth *et al.* 2016. *Ecosphere* 7 (12)	Elwha River, Washington, USA			Riparian forest responses to the recent removal of the two dams on the Elwha River will depend largely on channel and geomorphic adjustments to the release, transport, and deposition of the large volume of sediment formerly stored in the reservoirs, together with changes in large wood dynamics.
	Sharma *et al.* 2005. *Aquat. Ecosyst. Health Manag.* 8: 267–275	Tinau River, Nepal		The dam building had significant impacts on the macroinvertebrate composition just above the dam site, probably as a result of deposition of inorganic material within the small reservoir and changes in water speed. Damming of the Tinau River thus seems only to have a relatively minor impact on the river biota downstream of the dam site.	
	Singer & Gangloff. 2011. *Freshwater Biology* 56: 1904–1915	Alabama stream, USA		Data suggest that some small impoundments enhance conditions for freshwater mussel growth in downstream reaches. Restoring deteriorated mill dams may be, in some cases, a better mussel management option than dam removal, especially if large aggregations of imperilled mussels are present immediately downstream.	
	Smith & Goeckler. 2015. *Hydrobiologia* 755: 1-12	Neosho River, Kansas, USA		Density and abundance of zebra mussel declined downstream from reservoir sources but repeatedly increased at sites	

				inundated by lowhead dams compared to free-flowing areas. Upstream reservoirs remain the main source of zebra mussel larvae, but population sinks at lowhead dams could produce veligers that could recruit downstream and progressively colonize downstream reaches.	
	Spence & Hynes. 1971. *Journal of the Fisheries Research Board of Canada* 28: 35–43	Grand River, Ontario, Canada		Pronounced differences were found in the macroinvertebrate riffle fauna upstream and downstream of a flood control impoundment.	
	Steffensen *et al.* 2013. *Ecol. Freshw. Fish* 22: 374-383.	Small stream, Ontario	Nature-like fishway improved connectivity for stream fishes to approximately 85% of the creek upstream of the barrier.		
	Sullivan & Manning. 2017. *PeerJ* 5:e3189; Doi10.7717/peerj.3189	Olentangy River, Ohio, USA		Variability in macroinvertebrate response trajectories by season was observed, providing initial evidence that ecological responses to dam removal may be temporally variable and follow seasonally distinct recovery trajectories.	
	Takahashi & Nakamura. 2011. *Landscape Ecol Eng* 7: 65–77.	Satsunai River, Japan			Land cover types were associated with flood frequency below the dam. The reduced flood frequency of the upper site resulted in increased area of riparian vegetation and decreased area of active channel.
	Tealdi *et al.* 2011. *Journal of Hydrology* 396 (3-4): 302-312	Modelling			Differences in vegetation between the pre- and post-dam conditions are non-negligible and nonlinear behaviour occurs. Changes in the hydrologic regime, due to anthropic reservoirs and morphologic variations, are able to drastically modify the delicate and complex dynamics of the

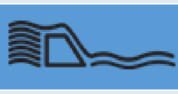
				riparian vegetation, which can proliferate or decline and shift to more suitable floodplain locations.
Tiemann *et al.* 2004. *Transactions of the American Fisheries Society* 133: 705-717	Neosho River, Kansas, USA	Macroinvertebrate richness did not differ among site types, but abundance was lowest at downstream treatment sites and evenness was lowest at upstream treatment sites. Fish species richness did not differ among site types, but abundance was highest at downstream reference sites and evenness was highest at upstream sites. The abundance of some benthic fishes was influenced by the dams, including that of the Neosho madtom, which was lowest immediately upstream and downstream from dams, and those of the suckermouth minnow, orange throat darter, and slenderhead, which were highest in downstream treatment sites. This study suggests that the effects of lowhead dams on fishes, macroinvertebrates, and habitat are similar to those reported for larger dams.		
Tiemann *et al.* 2005. *Journal of Freshwater Ecology* 20: 519-525	Neosho River, Kansas, USA		Differences in habitat around lowhead dams are unfavourable for ephemeropterans, plecopterans, and trichopterans taxa.	
Tiemann *et al.* 2007. *Northeastern Naturalist* 14: 125-138	Fox River, Illinois, USA		Results suggest a negative effect of lowhead dams on habitat characteristics and freshwater mussel assemblages. Dams limit the upstream distribution of 5 species.	
Tombolini *et al.* 2014. *Knowledge and Management of Aquatic Ecosystems* 412, (03)	Tiber River, central Italy			The main vegetation changes occurred during the first decades after the dam construction. The dam operation caused significant local expansion of the upstream river waters, causing the flooding of most of the surrounding lands, and the formation of new sub-lentic wetlands. The altered

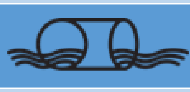
					hydrogeomorphic conditions favoured the development of natural riparian and macrophyte communities typical of lacustrine ecosystems and therefore mostly different from those expected for the analysed river typology.
	Travnichek *et al.* 1995. *Transactions of the American Fisheries Society* 124: 836–844.	Tallapoosa River, Alabama, USA	After the minimum flow was initiated, species richness 3 km below the dam more than doubled, and over half of the species collected were classified as fluvial specialists.		
	Tullos *et al.* 2014. *PLOS ONE* 9:e108091.	Calapooia River, Oregon		The presence of the dams constituted a strong ecological disturbance to the near-downstream reaches on investigated rivers, despite the fact that both rivers passed unregulated flow and sediment during the high flow season.	
	Vaikasas *et al.* 2013. *Journal of Environmental Engineering and Landscape Management* 21(4): 305-315	Virvyte river, Lithuania		Dam building has significant impacts on macroinvertebrate composition. Both of the disturbed sites (in and below dams) have significant lower taxa number of macroinvertebrates (both total and EPT taxa) compared to the control sites. The total abundance of macroinvertebrates was significantly higher in control sites of river than in the sites in and below dams.	
	Vallania & Corigliano.2007. *Environ. Monit. Assess.* 124: 201–209	Grande River, Argentina		The collector-filterers, scrapers and predators increase whereas the collector-gatherers and shredders decreased below the dam. There were significant differences at the level of gatherers and shredders.	
	Vaughn & Taylor. 1999. *Conservation*	Little River, USA		Mussel extinction gradient downstream from impoundments was observed: with increasing	

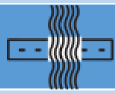
	Biology 13: 912–920.			distance from the mainstem reservoir there was a gradual, linear increase in mussel species richness and abundance.	
	Vinson. 2001. *Ecological Applications* 11: 711–730.	Flaming Gorge Dam, northeastern Utah, USA		Results suggest that we should not only evaluate traditional habitat attributes, but biological interactions as well, when determining or monitoring the effects of river regulation on aquatic biota.	
	Wang *et al.* 2011. *River Research and Applications* 27: 473-487	Rivers in Michigan and Wisconsin, USA	Impacts of dams on river fishes are more complex than just blocking fish migratory pathways. The negative influences of downstream dam density and upstream dam abundance on species richness and diversity may reflect the cumulative impacts of dams in both upstream and downstream directions.		
	Wang *et al.* 2013. *Fresenius Environ. Bull.* 22: 103-110.	Jiuchong River, China		Significant impacts of flow regulation on macroinvertebrate species richness and number of Ephemeroptera, Plecoptera and Trichoptera taxa were detected at impacted habitats, where observed species richness were lower and EPT taxa also were lower.	
	Wang *et al.* 2014. *Freshwater Biology* 59: 1343-1360	Yangtze River, China	Gradual delay in spawning time coupled with a lack of long flow increase (duration >5 days), the occurrence of dissolved gas super-saturation and the disappearance of most spawning grounds in the Three Gorges Reservoir, resulted in a substantial decline in carp larval abundance of the middle Yangtze River.		
	Watters. 1996. *Biol. Conserv.* 75: 79-85.	Five river systems, USA		Dams, even lowhead structures, may contribute to the overall depletion of unionoids by artificially restricting their distributions and isolating populations from each other.	
	Williams. 2008. *Hydrobiologia* 609: 241-251.	Columbia River, USA	Efforts to save large runs of salmon: e.g. modifying dam operations to provide more constant flow and providing additional flow from storage reservoirs to create more natural		

			flow through areas inundated by dams.		
	Williams *et al.* 1992. *Bulletin of the Alabama Museum of Natural History* 13: 1–10	Black Warrior and Tombigbee rivers, Alabama, USA		Habitat destruction associated with impoundments has severely depleted mussels of the main channels of both rivers (reduction from 54 to 11 species)	
	Williams *et al.* 2012. *J. River Research and Applications* 28: 407-417	Review	Even well designed fish ladders or nature‐like bypass channels for upstream migrants, even those with good attraction flows, will fail if incorrectly sited. Developing successful installations for downstream migrants remains much more difficult, probably because downstream fish move with the flow and have less time to assess cues at entrances to any bypasses that they encounter.		
	Wu *et al.* 2010. *Aquatic Sciences* 72: 117-125	Xiangxi River, China			
	Xiaocheng *et al.* 2008. *Acta Ecol. Sin.* 28: 45-52	Xiangxi River, China		All the characteristics of the macroinvertebrate community were more or less affected by the construction, especially by the abundance, filter-collector percentage, predator percentage, and the stations. The results also suggested that the sites beneath the dam had the most different community structures, indicating that diverting the water current completely is harmful to the protection of macroinvertebrate diversity of the river.	
	Zhong & Power. 1996. *Regulated Rivers-Research and Management* 12: 81-98	Changjiang River, Quiantang River and Han River, China	Migrations of anadromous and semi-migratory fish were blocked by the Gezhouba Dam, although some species adapted to the new environment by reproducing downstream. Below the Xinanjiang and Danjiangkou dams spawning was delayed 20-60 days by lower water temperatures. Reduced water velocities and less variable discharges caused spawning grounds below the dams to be		

		abandoned. Marked changes in the hydrological regime caused the extinction of *Macrura reevesii,* a highly valued fish, in the Qiantang River. The fish communities in the Qiantang estuary were affected by the regulated discharge.		
Žganec *et al.* 2013. *Limnologica* 43 (6): 460-468	Gojacka Dobra River, Croatia		Drastic change in water quality and increased load of suspended particles that accumulated in mosses soon after the dam closure were the main factors that contributed to the endemic moss specialist *Echinogammarus cari* rapid population decline at sites downstream of the new dam. Widespread microhabitat generalist *Gammarus fossarum* was much less affected by such changes, exhibiting no drastic population decline and even an increase in population size after disturbance event at three examined sites.	
Fjellheim & Raddum. 1996. *Regulated Rivers Research and Management* 12: 501-508	River Ekso, Norway	During the first years after regulation and weir building, brown trout density increased from 2.5 to 11.1 individuals 100 m^{-2}. In 1983 a density of 23.0 trout 100 m^{-2} was achieved. Weir basins increase the area of pool habitats in strongly regulated rivers, and are of major benefit for trout populations, especially by segregating size classes and increasing winter survival. The presence of intermittent riffle sections is also very important, both as spawning and nursery areas and for fish food production.	In the first years after weir building, biomass was greatest in the riffles due to a higher abundance of lotic species like the mayfly, *Baetis,* blackflies and many stonefly larvae. The biomass of oligochaetes and chironomids was similar both in the riffles and in the deeper and more lentic weir basin. In the following years the biomass of lentic chironomid species increased dramatically in the basin. In 1984-1988 net benthic animal production in the basin had increased 10-fold compared with 1975-1976.	
Greet *et al.* 2011. *Freshwater Biology* 56: 2514–2528	5 upland streams, south-eastern Australia			Flow regulation by weirs, which results in reduced discharge and reduced non-flood flow variability,

					facilitated the encroachment of ‘dry’ species, both native and exotic, into the stream channel. Grass cover was also greater downstream of the weirs.
	Wiśniewolski, W., Prus, P. 2009. *Annals of Warsaw University of Life Sciences – SGGW Land Reclamation* 41(2): 131-142.	Poland	Differences in fish fauna composition between impoundments and free flowing rivers, increase of share of limnophilic and eurytopic species in reservoirs.		
			WEIR		
	Alexandre *et al.* 2010. *River Research and Applications* 26: 977–994	Two Iberian streams	Lentic upstream sites presented higher density of limnophilic, omnivorous and exotic species, like gudgeon, which are well adapted to this type of habitat. Downstream and between obstacles sites were characterized by the dominance of rheophilic and insectivorous taxa, especially barbell. Richness metrics did not differ among site types, but diversity was higher in sites located between the obstacles away from its direct influence, where the habitat diversity was higher.		
	de Leaniz. 2008. *Hydrobiologia* 609: 83–96	Review	Weirs can significantly increase the vulnerability of migratory fish to anglers, alter natural migration patterns, and exacerbate the effects of opportunistic predators. Overcrowding of fish at downstream pools can also facilitate the spread of parasites and infectious diseases, magnify the impact of pollution incidents, and increase the risk of mass mortalities, particularly at low flows.		
	Winter & Van Densen. 2001. *Fisheries Management and Ecology* 8: 513-532	River Vecht, Netherlands	Migratory opportunities along the six weirs were extremely limited. Only 10 of 32 species were able to ascend all weirs in 5-30% of the years. Opportunities were the greatest for large-sized species during November-March at the downstream-situated weirs,		

			whereas small-sized species had no opportunities year-round.		
	Khan & Colbo. 2008. *Hydrobiologia* 600: 229–235	Newfoundland, Canada		The study indicated that longstanding point source physical disturbance (road culverts) primarily impacted taxa abundance rather than community present/absent data, which will recolonize the disturbed zone by downstream drift.	
			CULVERT		
	Cocchiglia *et al.* 2012. *Freshwater Reviews* 5: 141-168	Literature review	The construction of river crossings has the potential to generate elevated inputs of sediment that may impact adversely on aquatic environments, both in the short and long term.		
	Peterson. 2010. *PSU McNair Scholars Online Journal* 4 (1), Article 8	2 streams, Oregon, USA		This study showed correlations between the composition of the benthic macroinvertebrate community and the presence of road culverts. Fundamental shift of the local lotic ecology below road culverts.	
	Tummers *et al.* 2016. *Sci. Total Environ*. 569–570: 850–860	River Deerness, England	This study demonstrates that habitat connectivity restoration at engineered in-stream structures has been effective for both strong swimmers (brown trout) and for those with limited swimming abilities (bullhead).		
	Gibson *et al.* 2005. *Fisheries* 30(1):10-17		In the light of declines in Atlantic salmon (Salmo salar) stocks, the study demonstrated the extent to which stream crossings along a newly constructed section of the Trans Labrador Highway (TLH Phase II) in southern Labrador accorded with government regulations for fish habitat protection. Researchers surveyed crossings of permanent streams over a 210 km road segment, containing 4 bridges and 47 culverts. Fifty-three percent of culverts posed problems to fish passage, due to poor design or poor installation.		
			SLUICE		

	Richardson, Hanson & Locke, 2002. *Aquatic Ecology*, 36(4):493–510			Consecutive reductions in water levels, controlled by tidal sluices, has been shown to be catastrophic for macrophytes communities and macroinvertebrates and molluscs inhabiting the littoral zone.	
	Capers. 2003. *Aquatic Botany.* 77(4):325–338				Operation of barriers controlling water levels often have implications for macrophytes taxa reliant on being submerged
	Zamanm et al. 2013. *Natural and Social Sciences* 1 (2): 99-110	Bangladesh	The sluice gate pessimistically affected the fish diversity reducing the type and number of fish species, decreasing their overall and local status and thrown them in extinct, endangered, vulnerable and threatened position where abundance, availability and breeding of most species dominantly hampered, changed or reduced.		
	Madsen *et al.* 2001. *Hydrobiologia*444(1–3):71–84				Operation of barriers controlling water levels dynamics impacts submerged macrophytes.
			FORD		
	Williams *et al.* 2005. Water & Atmosphere 13(1)		Problems with fords: Typical ford construction impedes migratory native fish; 'Climbing' fish, such as redfin bullies and eels, are less affected. Improved ford design can help remedy the problem.		
			RAMP		
	Plesiński *et al.* 2018. *Science of the Total Environment* 631-632: 1201–1211	Mountain stram, Poland	The study shows that the block ramp cannot provide longitudinal connectivity and migration of fish occurring in the mountain stream. Some crevices in the side zones of the ramp could be parts of the migration corridor, but only for small and medium-sized fish.		
	Weibel &Peter. 2013. *Aquat. Sci.* 75: 251–260	7 streams, Switzerland	Block ramps with slopes of >5 % are ineffective for the small-sized cyprinid species and vertical drops within step-pool ramps can hinder successful upstream passage of bullhead.		
	Scrimgeour, Jones & Tonn. 2013.			Rock ramps similar in size to low head weirs support lower macroinvertebrate and macrophyte densities than reference reaches. Thus	

River research and applications 29(3): 352–365			highlighting the importance of barrier head height contributing to their impacts on macroinvertebrates through their control on sedimentation and flow velocity.

7 REFERENCES IN APPENDIX A

Addy, S., Cooksley, S.L. and Sime, I. (2012) Impacts of flow regulation on freshwater pearl mussel (*Margaritifera margaritifera*) habitat in a Scottish montane river. *Science of the Total Environment,* **432**, 318-328.

Alexandre,C.M., Almeida, P.R. (2010) The impact of small physical obstacles on the structure of freshwater fish assemblages. *River Research and Applications,* **26**, 977–994.

Almeida, E., Oliviera, R., Mugnai, R., Nessimian, J., and Baptista D. (2009) Effects of small dams on the benthic community of streams in an Atlantic forest area of Southern Brazil. *Int. Rev. Hydrobiol.,* **94**, 179-193.

Almodóvar, A., Nicola, G.G. (1999) Effects of a small hydropower station upon brown trout S*almo trutta L*. in the river Hoz Seca (Tagus Basin, Spain) one year after regulation. *Regul. Rivers: Res.Mgmt.,* **15**, 477-484.

Angilletta, M.J., Jr., Steel, E.A., Bartz, K.K., Kingsolver, J.G., Scheuerell, M.D., Beckman, B.R. and Crozier, L.G. (2008) Big dams and salmon evolution: changes in thermal regimes and their potential evolutionary consequences. *Evolutionary Applications,* **1,** 286-299.

Antonio, R.R., Agostinho, A.A., Pelicice, F.M., Bailly, D., Okada, E.K. and Dias, J.H.P. (2007) Blockage of migration routes by dam construction: can migratory fish find alternative routes? *Neotropical Ichthyology,* **5**(2), 177-184.

Archer, D.R., Rippon, P., Inverarity, R. and Merrix, R. (2008) The role of regulating releases and natural spates on salmonid migration in the River Tyne, North-East England. In: *Sustainable Hydrology for the 21st Century*. Proceedings of the 10th BHS National Hydrology Symposium, Exeter. 15-18th Sept 2008.

Armitage, P.D. (2006) Long-term faunal changes in a regulated and an unregulated stream – Cow Green thirty years on. *River Research and Applications,* **22**, 947-966.

Baldigo, B.P., and Smith, A.J. (2012) Effects of Flow Releases on Macroinvertebrate Assemblages in the Indian and Hudson Rivers in the Adirondack Mountains of Northern New York. *River Research and Applications,* **28**, 858-871.

Barrella, W., and Petrere, Jr. M. (2003) Fish community alterations due to pollution and damming in Tietê and Paranapanema rivers (Brazil). *River Research and Applications,* **19**(1), 59-76.

Bejarano, M. D., Nilsson, Ch., del Tánago, M. G. and Marchamalo, M. (2010) Responses of riparian trees and shrubs to flow regulation along a boreal stream in northern Sweden. *Freshwater Biology,* **56**(5), 853-866.

Benitez-Mora, A. and Camargo, J.A. (2014) Ecological responses of aquatic macrophytes and benthic macroinvertebrates to dams in the Henares River Basin (Central Spain). *Hydrobiologia,* **728**, 167-178.

Benjankar, R., Jorde, K., Yager, E.M., Egger, G., Goodwin, P. and Glenn, N.F. (2012) The impact of river modification and dam operation on floodplain vegetation succession trends in the Kootenai River, USA. *Ecological Engineering,* **46,** 88-97.

Benstead, J.P., March, J.G., Pringle, C. and Scatena, F. (1999) Effects of a low-head dam and water abstraction on migratory tropical stream biota. *Ecol. Appl*., **9**, 656–668.

Birnie-Gauvin, K., Larsen, M.H., Nielsen, J., and Aarestrup, K. (2017) 30 years of data reveal dramatic increase in abundance of brown trout following the removal of a small hydrodam. *Journal of Environmental Management,* **204**(1), 467-471.

Boles, G.L. (1981) Macroinvertebrate Colonization of Replacement Substrate Below a Hypolimnial Release Reservoir. *Hydrobiologia,* **78**, 133-146.

Bradford, M.J., Higgins, P.S., Korman, J. and Sneep, J. (2011) Test of an environmental flow release in a British Columbia river: does more water mean more fish? *Freshwater Biology,* **56**, 2119-2134.

Branco, P., Segurado, P., Santos, J.M., and Ferreira, M.T. (2014) Prioritizing barrier removal to improve functional connectivity of rivers. *J. Appl. Ecol.,* **51**, 1197-1206.

Brandimarte, A.L., Anaya, M. and Shimizu, G.Y. (2016) Impact of damming on the Chironomidae of the upper zone of a tropical run-of-the-river reservoir. *Braz. J. Biol.* http://dx.doi.org/10.1590/1519-6984.16814

Bredenhand, E. and Samways, M.J. (2009) Impact of a dam on benthic macroinvertebrates in a small river in a biodiversity hotspot: Cape Floristic Region, South Africa. *J. Insect Conserv*., **13**, 297–307.

Burroughs, B.A., Hayes, D.B., Klomp, K.D., Hansen, J.F., and Mistak J. (2010) The effects of the Stronach Dam removal on fish in the Pine River, Manistee County, Michigan. *Trans. Am. Fish. Soc.,* **139**, 1595-1613.

Capers, R. S. (2003) Macrophyte colonization in a freshwater tidal wetland (Lyme, CT, USA). *Aquatic Botany*, **77**(4), 325–338.

Casado, A., Peiry J.-L. and Campo, A. M. (2016) Geomorphic and vegetation changes in a meandering dryland river regulated by a large dam, Sauce Grande River, Argentina. *Geomorphology,* **268**, 21-34.

Catalano, M., Bozek, M., and Pellett, T. (2007) Effects of dam removal on fish assemblage structure and spatial distributions in the Baraboo River,Wisconsin. *North Am. J. Fish. Manag.,* **27**, 519-530.

Caudill, C.C., Keefer, M.L., Clabough, T.S., Naughton, G.P., Burke, B.J. and Peery C.A. (2013) Indirect Effects of Impoundment on Migrating Fish: Temperature Gradients in Fish Ladders Slow Dam Passage by Adult Chinook Salmon and Steelhead. *Plos One* **8**, e85586.

Cereghino, R. and Lavandier, P. (1998) Influence of hydropeaking on the distribution and larval development of the Plecoptera from a mountain stream. *Regulated Rivers - Research and Management,* **14**, 297-309.

Clarke, K. D., Pratt, T. C., Randall, R. G., Scruton, D. A., and Smokorowski, K. E. (2008) Validation of the Flow Management Pathway: Effects of Altered Flow on Fish Habitat and Fishes Downstream from a Hydropower Dam. Canadian Technical Report of Fisheries and Aquatic Sciences, **2784**.

Cocchiglia, L., Purcell, P. J. and Quinn, M.-K. (2012) A critical review of the effects of motorway river-crossing construction on the aquatic environment. *Freshwater Reviews,* **5**, 141-168. (Review)

Cortes, R.M.V., Ferreira, M.T., Oliveira, S.V., and Godinho, F. (1998) Contrasting impact of small dams on the macroinvertebrates of two Iberian mountain rivers. *Hydrobiologia,* **389**, 51-61.

Cumming, G.S. (2004) The impact of low-head dams on fish species richness in Wisconsin, U.S.A. *Ecol. Appl.,* **14**, 1495-1506.

de Leaniz, C. G. (2008) Weir removal in salmonid streams: implications, challenges and practicalities. *Hydrobiologia,* **609**: 83–96. (Review)

Dorobek, A.C., Sullivan, S.M.P., and Kautza, A. (2015) Short-term consequences of lowhead dam removal for fish assemblages in an urban river system. *River Systems,* **21**, 125-139.

Doyle, M.W., Stanley, E.H., Orr, C.H., Selle, A.R., Sethi, S.A., and Harbor, J.M. (2005) Stream ecosystem response to small dam removal: lessons from the Heartland. *Geomorphology,* **71**, 227-244.

Finch, C., Pine III, W.E. and Limburg, K.E. (2015) Do hydropeaking flows alter juvenile fish growth rates? A test with juvenile humpback chub in the Colorado river. *River Res. Applic.*, **31**, 156-164.

Fjellheim, A., and Raddum, G.G. (1996) Weir building in a regulated west Norwegian river: long term dynamics of invertebrates and fish. *Regulated Rivers Research and Management,* **12**, 501-508.

Franchi, E., Carosi, A., Ghetti, L., Giannetto, D., Pedicillo, G., Pompei, L. and Lorenzoni, M. (2014) Changes in the fish community of the upper Tiber River after construction of a hydro-dam. *Journal of Limnology,* **73**: 203-210.

Gehrke, P.C., Gilligan, D.M. and Barwick, M. (2002) Changes in fish communities of the Shoalhaven River 20 years after construction of Tallowa Dam, Australia. *River Research and Applications,* **18**, 265–286.

Gibbins, C.N., Soulsby, C., Jeffries, M.J. and Acornley, R. (2001) Developing ecologically acceptable river flow regimes: a case study of Kielder reservoir and the Kielder water abstraction system. *Fisheries Management and Ecology,* **8**, 463–485.

Gibson, R.J., Haedrih, R. L. and Wernerheim, C.M. (2005) Loss of Fish Habitat as a Consequence of Inappropriately Constructed Stream Crossings. *Fisheries,* **30**(1), 10-17.

Gillespie, B.R., Brown, L.E., and Kay, P. (2015) Effects of Impoundment on Macroinvertebrate Community Assemblages in Upland Streams. *River research and Applications,* **31**(8), 953-963.

Głowacki, Ł. and Penczak, T. (2012) Large dam reservoirs are probably long-period oscillators of fish diversity. *Journal of Fish Biology,* **80**, 2213–2235.

Głowacki, Ł., Grzybkowska, M., Dukowska, M., and Penczak, T. (2011) Effects of damming a large lowland river on chironomids and fish assessed with the (multiplicative partitioning of) true/Hill biodiversity measure. *River Research and Applications*, **27**, 612-629. DOI: 10.1002/rra.1380

Godinho, A.L., and Kynard, B. (2009) Migratory fishes of Brazil: life history and fish passage needs. *J. River Research and Applications*, **25**(6), 702-712.

Gore, J.A. (1980) Ordinational Analysis of Benthic Communities Upstream and Downstream of a Prairie Storage Reservoir. *Hydrobiologia,* **69**, 33-44.

Grant, G. (2001) Dam removal: panacea or Pandora for rivers? *Hydrological Processes,* **15**, 1531–1532.

Greet, J., Webb, J.A., and Downes, B.J. (2011) Flow variability maintains the structure and composition of in-channel riparian vegetation. *Freshwater Biology,* **56**, 2514–2528.

Grzybkowska, M., Kucharski, L., Dukowska, M., Michyio Takeda, A., Lik, J., and Leszczynska, J. (2017) Submersed aquatic macrophytes and associated fauna as an effect of dam operation on a large lowland river. *Ecological Engineering,* **99**, 256–264.

Grzybkowska, M., Kurzawski, M. and Dukowska M. (2012) Response of Chironomidae (Diptera) to impoundments in lowland streams. *Fauna Norvegica,* **31**, 25-33.

Halleraker, J.H., Sundt, H., Alfredsen, K.T. and Dangelmaier, G. (2007) Application of multiscale environmental flow methodologies as tools for optimized management of a Norwegian regulated national salmon watercourse. *River Research and Applications,* **23**, 493-510.

Hansen, J.F., and Hayes, D.B. (2012) Long-term implications of dam removal for macroinvertebrate communities in Michigan and Wisconsin rivers, United States. *River Research and Applications,* **28**, 1540-1550.

Helms, B.S., Werneke, D.C., Gangloff, M.M., Hartfield, E.E., and Feminella, J.W. (2011) The influence of low-head dams on fish assemblages in streams across Alabama. *Journal of the North American Benthological Society,* **30**, 1095-1106.

Horne, B.D., Rutherford, E.S. and Wehrly, K.E. (2004) Simulating effects of hydro-dam alteration on thermal regime and wild steelhead recruitment in a stable-flow Lake Michigan tributary. *River Research and Applications,* **20**, 185-203.

Jansson, R., Nilsson, C. and Renofalt, B. (2000) Fragmentation of riparian floras in rivers with multiple dams. *Ecology,* **81**, 899–903.

Käiro, K., Mols, T., Timm, H., Virro, T., and Jarvekulg, R. (2011) The effect of damming on biological quality according to macroinvertebrates in some estonia streams, central - baltic europe: a pilot study. *River Research and Applications,* **27**, 895-907.

Kanehl, P.D., Lyons, J., and Nelson, J.E. (1997) Changes in the habitat and fish community of the Milwaukee River, Wisconsin, following removal of the Woolen Mills Dam. *North Am. J. Fish. Manag.,* **17**, 387-400.

Khan, B., and Colbo, M.H. (2008) The impact of physical disturbance on stream communities: lessons from road culverts. *Hydrobiologia*, **600**, 229–235(B).

Kil, H.K., and Bae, Y.J. (2012) Effects of low-head dam removal on benthic macroinvertebrate communities in a Korean stream. *Anim. Cells Syst*., **16**, 69-76.

Kingsford, R.T. (2000) Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. *Austral Ecology,* **25**, 109–127.

Korman, J. and Campana, S.E. (2009) Effects of Hydropeaking on Nearshore Habitat Use and Growth of Age-0 Rainbow Trout in a Large Regulated River. *Transactions of the American Fisheries Society,* **138**, 76-87.

Kruk, A., and Penczak, T. (2003) Impoundment impact on populations of facultative riverine fish. *Annales de Limnologie - International Journal of Limnology,* **39**(3), 197-210.

Kuby, M.J., Fagan, W.F., ReVelle, C.S., and Graf, W.L. (2005) A multiobjective optimization model for dam removal: an example of trading off salmon passage with hydropower and water storage in the Willamette basin. *Adv. Water Resour.,***28**, 845-855.

Kui, Li , Stella, J. C., Shafroth, P. B., House, P. K., and Wilcox, A. C. (2017) The long-term legacy of geomorphic and riparian vegetation feedbacks on the dammed Bill Williams River, Arizona, USA. *Ecohydrology,* **10**:e1839.

Lessard, J.L. and Hayes, D.B. (2003) Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. *River Research and Applications,* **19**, 721–732.

Li, Jinpeng, Dong, S., Yang, Z., Peng, M., Liu, S. and Li, X. (2012) Effects of cascade hydropower dams on the structure and distribution of riparian and upland vegetation along the middle-lower Lancang-Mekong River. *Forest Ecology and Management*, **284**, 251–259.

Lucas, M.C., Bubb, D.H., Jang, M., Ha, K. and Masters, J.E.G. (2009) Availability of and access to critical habitats in regulated rivers: effects of low-head barriers on threatened lampreys. *Freshw. Biol.,* **54**, 621–634.

Lucas, S.D., Wheeler, J.A., Atallah, Y.C., Walker, S.E., Jones, C.E. and Burk, J.H. (2016) Long-term impacts of dam construction on plant succession and survival of an endangered species. *Ecosphere*, **7**, 5, (e01235).

Mackay, R., and Waters, T. (1986) Effects of small impoundments on hydropsychid caddisfly production in valley creek, Minnesota. *Ecology,* **67**, 1680-1686.

Madsen, J. D. *et al.* (2001) The interaction between water movement, sediment dynamics and submersed macrophytes. *Hydrobiologia,* **444**(1–3), 71–84.

Magilligan, F.J., Nislow, K.H., Kynard, B.E., and Hackman, A.M. (2016) Immediate changes in stream channel geomorphology, aquatic habitat, and fish assemblages following dam removal in a small upland catchment. *Geomorphology,* **252**, 158-170.

Maloney, K.O., Dodd, H.R., Butler, S.E. and Wahl, D.H. (2008) Changes in macroinvertebrate and fish assemblages in a medium-sized river following a breach of a low head-dam. *Freshw. Biol.,* **53**, 1055–1068.

Marchant, R. and Hehir, G. (2002) The use of AUSRIVAS predictive models to assess the response of lotic macroinvertebrates to dams in south-east Australia. *Freshwater Biology,* **47**(5), 1033-1050.

Martínez, A., Larranaga, A., Basaguren, A., Perez, J., Mendoza-Lera, C., and Pozo, J. (2013) Stream regulation by small dams affects benthic macroinvertebrate communities: from structural changes to functional implications. *Hydrobiologia,* **711**, 31-42.

McCarthy, T.K., Frankiewicz, P., Cullen, P., Blaszkowski, M., O'Connor, W. and Doherty, D. (2008) Long-term effects of hydropower installations and associated river regulation on River Shannon eel populations: mitigation and management. *Hydrobiologia,* **609**, 109-124.

Merritt, D.M., and Cooper, D.J. (2000) Riparian vegetation and channel change in response to river regulation: a comparative study of regulated and unregulated streams in the Green River Basin, USA. *Regulated Rivers: Research and Management,* **16**, 543–564.

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

Mortenson, S. G. and Weisberg, P.J. (2010) Does river regulation increase the dominance of invasive woody species in riparian landscapes? *Global Ecology and Biogeography,* **19**(4), 562-574.

Muraoka, K., Nakanishi, S., and Kayba, Y. (2017) Boulder arrangement on a rocky ramp fishway based on the swimming behaviour of fish. *Limnologica,* **62**, 188–193.

Noonan, M.J., Grant, J.W.A., and Jackson, C.D. (2012) A quantitative assessment of fish passage efficiency. *Fish Fish*., **13**, 450-464.

Orr, C.H., Kroiss, S.J., Rogers, K.L., Stanley, E.H. (2008) Downstream benthic responses to small dam removal in a coldwater stream. *River Research and Applications,* **24**, 804-822.

Osmundson, D.B., Ryel, R.J., Lamarra, V.J., and Pitlick, J. (2002) Flow–sediment–biota relations: implications for river regulation effects on native fish abundance. *Ecol.Appl.,* **12**, 1719–1739.

Ovidio, M., and Philippart, J.C. (2002) The impact of small physical obstacles on upstream movements of six species of fish – synthesis of a5-year telemetry study in the River Meusebasin. *Hydrobiologia,* **483**, 55–69.

Ovidio, M., Capra, H., and Philippart, J.C. (2007) Field protocol for assessing small obstacles to migration of brown trout *Salmo trutta*, and European grayling *Thymallus thymallus*: a contribution to the management of free movement in rivers. *Fish. Manag. Ecol*., **14**, 41–50.

Paller, M.H. and Saul, B.M. (1996) Effects of temperature gradients resulting from reservoir discharge on *Dorosoma cepedianum* spawning in the Savannah River. *Environmental Biology of Fishes,* **45**, 151-160.

Pelicice, F.M., Pompeu, P.S., and Agostinho, A.A. (2015) Large reservoirs as ecological barriers to downstream movements of Neotrophical magratory fish. *Fish Fish.,* **16**, 697-715.

Penczak, T. (1992) Fish production in the Warta River, Poland: a postimpoundment study. *Hydrobiologia,* **242**, 87-93.

Penczak, T., Głowacki, Ł., Kruk, A., and Galicka, W. (2012) Implementation of a self-organizing map for investigation of impoundment impact on fish assemblages in a large, lowland river: Long-term study. *Ecological Modelling*, **227**, 64– 71.

Penczak, T., Kruk, A. (2005) Patternizing of impoundment impact (1985-2002) on fish assemblages in a lowland river using the Kohonen algorithm. *Journal of Applied Ichthyology,* **21**, 169-177.

Penczak, T., Kruk, A., Grzybkowska, M. and Dukowska, M. (2006) Patterning of impoundment impact on chironomid assemblages and their environment with use of the self-organizing map (SOM). *Acta Oecologica,* **30**, 312-321.

Pess, G.R., McHenry, M.L., Beechie, T.J., and Davies, J. (2008) Biological impacts of the Elwha River dams and potential salmonid responses to dam removal. *Northwest Science,* **82**, 72-90.

Peterson, T. (2010) The Effect of Road Culverts on the Benthic Macroinvertebrate Community in Wadeable Lotic Ecosystems. *PSU McNair Scholars Online Journal,* **4** (1), Article 8. 10.15760/mcnair.2010.52

Plesiński, K., Bylak, A., Radecki-Pawlik, A., Mikołajczyk, T., and Kukuła, K. (2018) Possibilities of fish passage through the block ramp: Model-based estimation of permeability. *Science of the Total Environment,* **631-632**, 1201–1211.

Principe, R. (2010) Ecological effects of small dams on benthic macroinvertebrate communities of mountain streams (Cordoba, Argentina). *Int. J. Limnol.,* **46**, 77-91.

Quinones, R. M., Grantham, T. E., Harvey, B. N., Kiernan, J. D., Klasson, M., Wintzer, A. P., and Moyle, P.B. (2014.) Dam removal and anadromous salmonid (*Oncorhynchus* spp.) conservation in California. *Rev Fish Biol Fisheries.* DOI 10.1007/s11160-014-9359-5 (+Review)

Renöfält, B.M., Lejon, A.G.C., Jonsson, M., and Nilsson, C. (2013) Long-term taxon-specific responses of macroinvertebrates to dam removal in a mid-sized Swedish stream. *River Research and Applications,* **29**, 1082-1089.

Richardson, S. M., Hanson, J. M. and Locke, A. (2002) Effects of impoundment and water-level fluctuations on macrophyte and macroinvertebrate communities of a dammed tidal river. *Aquatic Ecology*, **36**(4), 493–510. doi: 10.1023/A:1021137630654

Roberts, J.H., Angermeier, P.L., and Hallerman, E.M. (2013) Distance, dams and drift: what structures populations of an endangered, benthic stream fish? *Freshwater Biology,* **58**, 2050-2064.

Sankey, J.B., Ralston, B. E., Grams, P. E., Schmidt, J. C. and Cagney, L. E. (2015) Riparian vegetation, Colorado River, and climate: Five decades of spatiotemporal dynamics in the Grand Canyon with river regulation. *Journal of Geophysical Research: Biogeosciences,* **120**(8), 1532-1547.

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

Santos, J.M., Ferreira, M.T., Godinho, F.N., and Bochechas, J. (2005) Efficacy of a naturelike bypass channel in a Portuguese lowland river. *J. Appl. Ichthyol*., **21**, 381-388.

Santucci, V., Gephard, S., and Pescitelli, S. (2005) Effects of multiple low-head dams on fish, macroinvetebrates, habitat, and water quality in the Fox River, Illinois. *North American Journal of Fisheries,* **25**, 975-992.

Scrimgeour, G., Jones, N. and Tonn, W. M. (2013) Benthic macroinvertebrate response to habitat restoration in a constructed Arctic stream. *River research and applications,* **29**(3), 352–365.

Sethi, S.A., Selle, A.R., Doyle, M.W., Stanley, E.H. and Kitchel, H.E. (2004) Response of unionid mussels to dam removal in Koshkonong Creek, Wisconsin (USA). *Hydrobiologia,* **525**, 157–165.

Shafroth, P.B., Perry, L. G., Rose, Ch. A. and Braatne, J. H. (2016) Effects of dams and geomorphic context on riparian forests of the Elwha River, Washington. *Ecosphere,* **7**, (12), Article e01621

Sharma, C.M., Sharma, S., Borgstrom, R. and Bryceson, I. (2005) Impacts of a small dam on macroinvertebrates: A case study in the Tinau River, Nepal. *Aquat. Ecosyst. Health Manag.,* **8**, 267–275.

Singer, E.E. and Gangloff, M.M. (2011) Effect of a small dam on freshwater mussel growth in an Alabama (U.S.A.) stream. *Freshwater Biology,* **56**, 1904–1915.

Smith, B.R., Edds, D.R., Goeckler, J.M. (2015) Lowhead dams and the downstream dispersal of zebra mussels. *Hydrobiologia,* **755**, 1-12.

Spence, J.A. and Hynes, H.B.N. (1971) Differences in benthos upstream and downstream of an impoundment. *Journal of the Fisheries Research Board of Canada,* **28**, 35–43.

Steffensen, S.M., Thiem, J.D., Stamplecoskie, K.M., Binder, T.R., Hatry, C., Langlois-Anderson, N., Cooke, S.J. (2013) Biological effectiveness of an inexpensive nature-like fishway for passage of warmwater fish in a small Ontario stream. *Ecol. Freshw. Fish,* **22**, 374-383.

Sullivan, S.M.P. and Manning, D.W.P. (2017) Seasonally distinct taxonomic and functional shifts in macroinvertebrate communities following dam removal. *PeerJ 5*:e3189; DOI 10.7717/peerj.3189

Takahashi, M., and Nakamura, F. (2011) Impacts of dam-regulated flows on channel morphology and riparian vegetation: a longitudinal analysis of Satsunai River, Japan. *Landscape Ecol. Eng.,* **7**, 65–77.

Tealdi, S., Camporeale, C. and Ridolfi, L. (2011) Modeling the impact of river damming on riparian vegetation. *Journal of Hydrology,* **396** (3-4), 302-312.

Tiemann, J.S., Dodd, H.R., Owens, N., and Wahl, D.H. (2007) Effects of lowhead dams on unionids in the Fox River, Illinois. *Northeastern Naturalist,* **14**, 125-138.

Tiemann, J.S., Gillette, D.P., Wildhaber, M.L., and Edds, D.R. (2004) Effects of lowhead dams on riffle-dwelling fishes and macroinvertebrates in a midwestern river. *Transactions of the American Fisheries Society,* **133**, 705-717.

Tiemann, J.S., Gillette, D.P., Wildhaber, M.L., and Edds, D.R. (2005) Effects of lowhead dams on the ephemeropterans, plecopterans, and trichopterans group in a North American river. *Journal of Freshwater Ecology,* **20**, 519-525.

Tombolini, I., G. Caneva, L. Cancellieri, S. Abati, and Ceschin, S. (2014) Damming effects on upstream riparian and aquatic vegetation: the case study of Nazzano (Tiber River, central Italy). *Knowledge and Management of Aquatic Ecosystems,* **412** (03).

Travnichek, V.H., Bain, M. B. and Maceina, M. J. (1995) Recovery of a warmwater fish assemblage after the initiation of minimum-flow release downstream from a hydroelectric dam. *Transactions of the American Fisheries Society,* **124**, 836–844.

Tullos, D.D., Finn, D.S., and Walter, C. (2014) Geomorphic and ecological disturbance and recovery from two small dams and their removal. *PLOS ONE* **9**:e108091.

Tummers, J.S., Hudson, S., Lucas, C.S. (2016) Evaluating the effectiveness of restoring longitudinal connectivity for stream fish communities: towards a more holistic approach. *Sci.Total Environ*., **569–570**, 850–860.

Vaikasas, S., Palaima, K., Pliūraitė, V. (2013) Influence of hydropower dams on the state of macroinvertebrates assemblages in the Virvyte river, Lithuania, *Journal of Environmental Engineering and Landscape Management,* **21**(4), 305-315.

Vallania, A.E. and Corigliano, M.C. (2007) The effect of regulation caused by a dam on the distribution of the functional feeding groups of the benthos in the sub basin of the Grande River (San Luis, Argentina). *Environ. Monit. Assess*., **124**, 201–209.

Vaughn, C.C. and Taylor, C.M. (1999). Impoundments and the decline of freshwater mussels: a case study of an extinction gradient. *Conservation Biology,* **13**, 912–920.
Vinson M.R. 2001. Long-term dynamics of an invertebrate assemblage downstream from a large dam. *Ecological Applications* 11: 711–730.

Wang J., Li C., Duan X., Chen D., Feng S., Luo H., Peng Q. and Liao W. 2014. Variation in the significant environmental factors affecting larval abundance of four major Chinese carp species: fish spawning response to the Three Gorges Dam. *Freshwater Biology* 59: 1343-1360.

Wang L., Infante D., Lyons J., Stewart J., and Cooper A. 2011. Effects of dams in river networks on fish assemblages in non-impoundment sections of rivers in Michigan and Wisconsin, U.S.A. *River Research and Applications* 27: 473-487.

Wang X., Cai Q., Jiang W., and Qu X. 2013. Assessing impacts of a dam on benthic macroinvertebrate communities in a mountain stream. *Fresenius Environ. Bull.* 22: 103-110.

Watters, G.T. (1996) Small dams as barriers to freshwater mussels (Bivalvia, Unionoida) and their hosts. *Biol. Conserv.,* **75**, 79-85.

Weibel, D., and Peter, A. 2013. Effectiveness of different types of block ramps for fish upstream movement. *Aquat. Sci.,* **75**, 251–260.

Williams, E., Boubee, J., Smith, J. (2005) When fish meet fords. *Water & Atmosphere,* 13(1).

Williams, J.D., Fuller, S.L.H. and Grace, R. (1992) Effects of impoundments on freshwater mussels (Mollusca: Bivalvia: Unionidae) in the mainstem of the Black Warrior and Tombigbee rivers in western Alabama. *Bulletin of the Alabama Museum of Natural History,* **13**, 1–10.

Williams, J.G. (2008) Mitigating the effects of high-head dams on the Columbia River, USA: experience from the trenches. *Hydrobiologia,* **609**, 241-251.

Williams, J.G., Armstrong, G., Katopodis, C., Larinier, M., and Travade, F. (2012) Thinking like a fish: a key ingredient for development of effective fish passage facilities at river obstructions. *J. River Research and Applications,* **28**, 407-417.

Winter, H.V. and Van Densen, W.L.T. (2001) Assessing the opportunities for upstream migration of non-salmonid fishes in the weir-regulated River Vecht. *Fisheries Management and Ecology,* **8**, 513-532.

Wiśniewolski, W., Prus, P. (2009) Preliminary criteria for ecological status assessment index for dam reservoirs on the basis of the ichthyofauna assemblages. *Annals of Warsaw University of Life Sciences – SGGW Land Reclamation,* **41**(2), 131-142.

Xiaocheng, F., Tao, T., Wanxiang, J., Fengqing, L., Naicheng, W., Schuhan, Z., and Qinghua, C. (2008) Impacts of small hydropower plants on macroinvertebrate communities. *Acta Ecol. Sin.,* **28**, 45-52.

Zamanm, A., Jewelm, A.S., Hossainm, Y., Provhat, S. J., Jasmine, S., Alim, M. and Hossainm, I. (2013) Impact assessment of sluice gate on fishing activity, fisheries diversity, riverine habitatability and livelihood stability of the fishermen in the Northwestern Bangladesh. *International Journal of Research in Applied, Natural and Social Sciences (IJRANSS)*, **1**(2), 99-110.

Žganec, K., Đurić, P., Hudina, S. and Gottstein, S. (2013) Population and distribution changes of two coexisting river amphipods after the closure of a new large dam. *Limnologica,* **43**(6), 460-468.

Zhong, Y.G. and Power, G. (1996) Environmental impacts of hydroelectric projects on fish resources in China. *Regulated Rivers-Research and Management,* **12**, 81-98.