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D.1.1 Guidance on Stream Barrier Surveying and Reporting.
Part A: Locating, Surveying and Prioritising Mitigation Actions for Stream Barriers.

This is version 1.0 of Guidance on Stream Barrier Surveying and Reporting. Part A: Locating, Surveying and Prioritising Mitigation Actions for Stream Barriers. This document is a deliverable of the AMBER project, which has received funding from the European Union's Horizon 2020 Programme for under Grant Agreement (GA) #689682.

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Preamble

Humans have been modifying river systems for millennia (Goudie, 2013). Reasons for modification include flood prevention, irrigation, power generation, navigation, gauging and to provide a reliable source of water (Goudie, 2013). Modification usually takes the form of damming (including weirs and barrages), channelization or water abstraction (often in vast quantities for power plant cooling) (Goudie, 2013). The rate of modification has increased dramatically in recent years and it is now estimated that over half of the world's large river systems are fragmented (Nilsson *et al.,* 2005). This includes over 45,000 dams greater than 15 m high (World Commission on Dams, 2000) and orders of magnitude more smaller obstructions. Rivers now rank among some of the most threatened ecosystems in the world (Dudgeon *et al.* 2006), and are the focus of restoration programmes which cost taxpayers billions (Palmer *et al.* 2005). Much of Europe depends on water from rivers for drinking, food production and the generation of hydropower, which is essential for meeting the European Union (EU) renewable energy target. Yet only half the EU surface waters have met the Water Framework Directive's (WFD) 2015 target of good ecological status, due in part to the fragmentation of habitats caused by tens of thousands of dams and weirs. Improving stream connectivity has been flagged as one of the priorities for more efficient stream restoration but effective rehabilitation of ecosystem functioning in European rivers needs to take the complexity and trade-offs imposed by barriers into account. However, strikingly, the location of the majority of barriers on European river systems is not known, there is no central inventory of existing barriers and methods to quantify the impact of barriers on stream connectivity are in their infancy.

This report is part of a deliverable that provides Guidance on Stream Barrier Surveying and Reporting (D1.1) to aid users within AMBER and elsewhere in decision making and for the development of Adaptive Barrier Management. It is split into two sections:

- **Part A**: Locating, Surveying and Prioritising Mitigation Actions for Stream Barriers.
- **Part B**: Towards a Pan-European ATLAS on Stream Barriers

Part A focusses on existing methodologies for surveying stream barriers. It includes sections on locating barriers, methods for assessing a barriers potential to influence longitudinal connectivity and provide socio-economic benefits (e.g. hydropower) and methods for prioritising mitigation actions. **Part B** focusses on evaluating the current state of existing barrier inventories throughout Europe and provides a road-map for the development of a pan-European ATLAS on stream barriers. This is **Part A** of the deliverable.

Executive summary

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Barriers to the free movement of aquatic organisms can negatively impact freshwater ecosystems. A key problem is that the number of river barriers in existence, their location and impact on connectivity is currently unknown. Numerous methods have been trialled to efficiently locate and document riverine barriers including compilation of historic databases (e.g. flood defence databases and long profile drawings of river reaches) and evaluation of remote sensing technology such as aerial and satellite imagery, Light Detection And Ranging (LIDAR) and Synthetic Aperture Radar (SAR). This has enable 1000's of barriers to be identified, relatively efficiently, with minimal time spent in the field. The databases and remote sensing technologies occasionally also allow additional useful data (e.g. barrier head difference) to be identified with little additional effort. In the UK citizen scientists are also being encouraged to help locate barriers through a specially developed smart phone application. However, the contribution to the UK barrier database from the application is currently small.

To assess the impact of individual barriers on habitat connectivity, numerous rapid barrier assessment protocols have been developed. In Europe the most well developed and widely accessible protocols, that are fully or partially available in English, are the SNIFFER (United Kingdom), ICE (French) and ICF (Spanish) protocols. These protocols compare the topographical and hydraulic characteristics of a barrier with the physical capabilities (e.g. swimming, jumping or crawling) of one or several fish species, to predict the passability of the structure for that species. However, the swimming abilities utilised for specific fish species and the methods used to calculate the final passability scores differ between the protocols. Trialling of the three protocols at two small barriers produced different passability scores for one of the barriers under identical flow conditions. Despite extensive field testing by a range of operators, limited field validation means that it remains unclear if any of the protocols produce passability scores that accurately reflect the actual passability of barriers for fish. There is a need to undertake more comprehensive validation of the current protocols at a range of barrier types and for a range of species/lifestages. The three protocols are however extensively field tested by a range of operators and represent a useful baseline for future development.

The socio-economic benefits of riverine barriers should be considered in conjunction with their impact on longitudinal connectivity so that informed decisions on mitigation actions can be undertaken. The potential for developing existing river infrastructure for the generation of hydropower has been evaluated for thousands of barriers at several locations within Europe. Examples of investigation undertaken in England and Wales and in Austria are discussed in detail within this report. The potential power that can be generated at an existing barrier is generally assumed to be a function of flow (P) and hydraulic head (Q). Factors such as the

environmental sensitivity of potential sites have also been frequently considered alongside power generation potential to prioritise opportunities for sustainable low-impact development. The potential benefits produced by barriers, other than hydropower, are numerous (e.g. flood control, food production, recreation and cultural significance) and should also be considered when planning management options. Although detailed methods to assess market and non-market values of infrastructure do exist, currently no coarse-scale rapid method for assessing the socio-economic benefits/costs of barriers is available. However, important information such as listing the current use(s)/purpose(s) of the barrier will aid in later management decisions.

To help direct barrier mitigation efforts (e.g. barrier removal, bypass construction, flow management schemes etc.) a variety of prioritisation methodologies have been formulated in recent years. These prioritisation models aim to highlight the most appropriate barriers to undertake mitigation actions on, to maximise the benefits produced, dependent on resource availability. There are numerous different prioritisation models currently available. They differ significantly from one another in a number of ways, including: (1) how they consider the issue of connectivity, (2) how the passability of barriers is included, (3) the species considered, (4) the parameters included in the model and (5) how the prioritisation process is undertaken. It is believed that optimisation models or methods that utilise greedy type heuristic selection present the best option for prioritising mitigation actions in large complex systems. Prioritisation models must also take into account sensible predictor variables which present an opportunity to markedly improve the cost-benefit return of mitigation actions and improve resource prioritisation. Commonly incorporated variables include type, quantity, and quality of habitat, distance between habitat, existing fish stocks and direct financial (e.g. in relation to construction work to remove a structure or build a fish pass) and economic costs (e.g. hydropower generation, water storage capacity and harvesting by fishermen). At present, no single prioritisation model is likely to meet the needs of all projects but there is a diverse range of models currently available. It is likely that through appropriate model selection and modification, options are available to suit most prioritisation needs.

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1 IMPACT OF RIVERINE BARRIERS ON AQUATIC ECOSYSTEMS

Riverine barriers can alter flow and sediment regimes (Nilsson *et al.,* 2005; Xu and Milliman, 2009), channel morphology (Gordon and Meentemeyer, 2006), and nutrient and oxygen availability (Bellanger *et al.,* 2004; Gresh *et al.,* 2000). Ecological impacts include changes in invertebrate communities (Boon, 1988), and for motile organisms the loss of, or reduced access to, critical habitat (Pess *et al.,* 2008), delayed migration (Caudill *et al.,* 2007), population isolation (Morita and Yamamoto, 2002), and reduced productivity and diversity (Agostinho *et al.,* 2008; Matzinger *et al.,* 2007). Many freshwater fish species undertake lifecycle essential migrations, seasonally or on an ontogenetic basis, for spawning, feeding, or refuge (Lucas and Baras, 2001). Iconic fish migrations include those of diadromous salmonids, of which millions of adults return to their native rivers to spawn annually (Gresh *et al.,* 2000), and the European eel, *Anguilla anguilla*, of which adults and juveniles undertake a *ca.* 6000 km journey to/from spawning grounds (thought to be in the Sargasso Sea) to freshwater rearing habitat throughout Europe (van Ginneken and Maes, 2005). Even so called 'resident' species that are considered nonmigratory and have very small home ranges (< 100 m), undertake short distance migrations and fish movements sporadically over time that are critically important components of metapopulation dynamics, resource management, evolution, and speciation (Fausch *et al.,* 2002). Poor river connectivity is considered one of the main reasons for declines in many European (Larinier, 2001; Kroes *et al.,* 2006) and other freshwater fish populations worldwide (Jungwirth *et al.,* 1998; Thorncraft and Harris, 2000; Marmulla & Welcomme, 2002). In extreme cases, populations have already become extinct. For example, in the Western United States, habitat degradation has been linked with the extinction of 29% of historic Pacific salmon populations (Gustafson *et al.,* 2007); in China, the construction of the Gezhouba dam on the Qiantang River resulted in the extinction of the reeves shad, *Tenualosa reevesii*; and in France, obstructions have caused the extinction of entire stocks of salmon in the Rhine, Seine and Garonne rivers (Larinier, 2001). As such, to conserve vulnerable freshwater species and adhere to legislative requirements there are pressures to mitigate for the negative impact of stream barriers.

2 LOCATING BARRIERS

A key problem in understanding where to focus mitigation resources is that the number and location of riverine barriers is not always known. In well studied parts of the Europe very high densities of barriers to the free movement of aquatic organisms have been identified. For example, in specific river basins in Catalonia thousands of infrastructure projects have been identified that could alter longitudinal river connectivity including large dams (over 15 m in height), weirs (under 15 m), crossings, gauging stations, bed sills and road and railway bridge pillars (ACA, 2005; Ordeix *et al.,* 2006; 2011), in France *ca.* 84,000 potential barriers to the free movement of aquatic organisms have been identified (ROE) and in the United Kingdom *ca.* 26,000 riverine barriers have been officially documented (EA, 2010). However, in these locations considerable resources have been allocated to locating and documenting barriers, a situation not reflected throughout Europe. In addition, even in these locations it is highly unlikely that all of the barriers present have been documented. For example, in the UK a focussed effort to identify barriers in the River Wey catchment, South East England, using previously unevaluated databases and a small amount of field work, identified 565 additional barriers not included in the national inventory (Eakins *et al.,* 2012). This suggests that the UK's most up to date inventory of riverine barriers could include fewer than 30% of potential obstacles to the free movement of aquatic organisms.

The resources required to locate barriers can be significant, especially in large river systems covering hundreds of thousands of square kilometres of land. On a European level, identifying the locations of hundreds of thousands of potential barriers to the free movement of aquatic organisms over such a large area is likely to be unfeasible without considerable coordination of effort and resources, or the development of advanced new data collection methods.

For this report a barrier is defined as: "Any structure in a river system that prevents or delays, or increases an organism's energy expenditure and susceptibility to disease and predation, during passage upstream or downstream past the structure compared to unobstructed conditions."

2.1 Existing databases

In a project to document possible sites for hydropower development in England and Wales, riverine barriers were systematically located by identifying objects in high resolution maps (OS MasterMap – Ordnance Survey, UK) that crossed the Detailed River Network database held by the national Environment Agency (EA, 2010). These barriers consisted of waterfalls, barrages, weirs, dams, mills and locks. Barrier type was inferred from the site name, description or text string contained within the OS MasterMap. The hydraulic head of each structure was estimated through several methods (e.g. Light Detection And Ranging [LIDAR] and Synthetic Aperture Radar [SAR]) and the WFD ecological classification status of the waterbody associated with each barrier was recorded. The final 'River Barriers' dataset identified 25,935 barriers, which at the time represented the most definitive list of potential barriers to the movement of aquatic organisms in England and Wales.

Although the dataset and methods utilised by the EA (2010) provided a resource efficient way to locate a large number of potential barriers it did not produce an exhaustive list of all barriers present. As part of a project to assess and prioritise barriers for removal in the River Wey catchment, south east England, Eakins *et al.* (2012) used two additional sources of data to identify potential barriers as well as some catchment walkovers. In addition to the River Barriers database, the National Flood Coastal Defence Database (NFCDD) and long profile drawings for sections of the river Wey were assessed. The NFCDD database was queried to select features > 5 m distant from structures present within the River Barriers database and features listed as either a 'culvert', 'weir', 'sluice' or 'screen' were extracted. The NFCDD database contained some data on structure height, width and length as well as general information on the structure's condition and recommendations for repairs and monitoring. Where possible these data were assigned to each structure. The long profile drawings were collected by various consultancies between 1990 and 2011 and were presented as a database of computer-aided design (CAD) profiles. By manually searching the profiles for labelled structures and sudden drops in water level, the locations of potential barriers were extracted to form a third structural database. It was possible to calculate head height from water level change and, where surveys were detailed enough, it was also possible to obtain water depth in the pools below the barrier. Additional information provided for some structures included culvert length, slope and diameter.

The River Barriers database detailed 242 barriers in the Wey catchment. An additional 456 barriers were identified in the NFCDD, 197 in the levelling data and 53 from field surveys that weren't listed in the River Barriers (total additional barriers identified without duplication = 565). It is also likely that not all barriers were identified as the levelling data covered only 48% of the primary and secondary river channels and the field surveys covered a very small fraction of the catchment.

This example shows that existing databases can be utilised to produce detailed inventories of a significant proportion of potential barriers within a river system at relatively little cost. In addition, many existing databases contain further useful information (e.g. head height, pool depth, barrier type) that can be used to make a preliminary assessment of a barriers impact on longitudinal connectivity. Where possible the availability of existing datasets should be investigated as a primary step in locating, assessing and reporting on stream barriers.

2.2 Satellite/aerial imagery

In the Republic of Ireland a catchment based assessment methodology to locate, identify and evaluate barriers has been formulated by Inland Fisheries Ireland (IFI). The methodology has been used throughout Ireland (King *et al.,* 2016a) and is currently being utilised to identify and assess barriers in a number of catchments as part of an Environmental Protection Agency (EPA) funded project in Ireland (called 'Assessment of extent and impact of barriers on freshwater hydromorphology and connectivity in Ireland' – The ReCONNECT Project). The IFI strategy has been formulated to ensure 'cost-effectiveness' and consistency in terms of time, effort and manpower. Tier 1 of the methodology involves efficiently locating barriers by reviewing where road networks cross river networks within a Geographic Information System (GIS) and by assessing satellite and aerial imagery and historic maps to identify potential barriers. Each data source is evaluated systematically by first assessing the main stem and then incrementally assessing all tributaries and channels until the entire river network, for all data sources, has been reviewed. Aerial/satellite imagery is used in conjunction with current and historic maps and provides a useful extra data source for identifying the presence of barriers not noted on maps. The imagery also provides opportunity to quickly assess whether potential barriers highlighted in historic maps still exist.

The IFI procedure leads to the creation of a 1:50,000 scale map of a catchment with all potential barriers marked, colour-coded and labelled with a discrete code. This is accompanied by a series of Excel tables where each structure is listed. This desk-based procedure provides a rationalised 'road map' for a survey team to go out into an area and view all locations marked as potential barriers. The list of potential barriers compiled from the desk study is likely to be excessive, in so far as all road crossings will be recorded but not all will be barriers. However the procedure provides a method for efficiently locating a large number of potential barriers and focussing efforts in the field. The IFI methodology also includes standardised procedures for capturing basic barrier information using ruggedised laptops (Tier 2). For example, geo-referencing of location and photographs along with dimensional measurements and an assessment of structure passability in the conditions prevailing on the day of survey, for a range of fish species are undertaken.

2.3 Citizen Science

A possible solution to locate a large amount of data with limited resources is through Citizen Science. Citizen science involves engaging volunteers to collect and/or process data as part of a scientific enquiry (Silvertown, 2009). It has flourished in recent years due to the increased availability and capability of the internet and everyday technological devices (Silvertown, 2009). Citizen science programmes are widely used for gathering information on the presence and abundance of species and for assessing changes in species distribution and population size in a number of taxa (Barlow *et al.,* 2015; Dickinson *et al.,* 2010; Gregory *et al.,* 2005; Penone *et al.,* 2013; Sewell *et al.,* 2012; Wright *et al.,* 2013). Data generated by monitoring schemes has been used to inform conservation priorities, assess responses to anthropogenic change, determine and evaluate management actions and produce bioindicators to assess the state of ecosystems (see Barlow *et al.,* 2015). For example, a UK National Bat Monitoring Program was successfully used to estimate spatial changes in the populations of 10 bat species based on a range of data collected at over 3200 separate sites by Citizen Scientists (Barlow *et al.,* 2015).

Recently in the United Kingdom a smartphone application (app), 'RiverObstacles', has been developed to allow barriers to be located and assessed using Citizen Science (launched on May 15th 2015) (www.river-obstacles.org.uk). The free to use mobile app for iPhone and Android devices was developed by the Nature Locator team in collaboration with the Scottish Environment Protection Agency (SEPA), the Rivers and Fisheries Trust for Scotland (RAFTS) and the Environment Agency (EA). The app enables people to send in photos and details of riverine barriers that they see when they are "out and about". Users can easily log several compulsory or optional variables at each barrier. At its simplest the app allows citizens to take a photo of a barrier, log location (geographic coordinates) and upload this to the internet for later verification. Advanced options include recording the origin, type and physical properties of the barrier and making a subjective opinion of the barrier's passability for salmon, trout, eels, lampreys, grayling, and/or coarse fish. The user clarifies the likely accuracy of their subjective opinion by stating whether their experience level is 1) non-expert, 2) local knowledge, 3) expert opinion, or 4) the results are based on a survey assessment (presumed to be the SNIFFER protocol or similar). Once the data is uploaded the records are verified by either SEPA or the EA and the relevant data extracted to be used by each agency as they see fit. The verified data is displayed, along with the EA and SEPA inventories of barriers in the UK, on the RiverObstacles website (www.river-obstacles.org.uk) and can be viewed and downloaded by members of the public. The app is easy to use and the website enables users to view their contribution to the national inventory in the form of a leader board. To date there are approximately 300 hundred registered users, with 60 being considered 'active' (pers. comm. Dave Kilby). A total of 225 verified barrier records have been added to the national inventory through the app (data accurate as of 05/09/2016).

To enable Citizen Scientists to locate and report on unknown barriers the app seems well designed and functional. However, the low level of participation suggests a potential marketing issue. It is also noteworthy that many of the most active users currently work in or are associated with fisheries conservation and hence are not the primary target audience. In addition the advanced options do not currently conform to any specific survey assessment methodology (although the scoring system most closely relates with the SNIFFER methodology – see section 4.1.1). The mismatch of the advanced variables logged compared to those recommended by the SNIFFER protocol has come about due to compromises between the requirements of the founding partners (SEPA, EA and RAFTS) (pers. comm. David Kilby). Hence, it should probably be considered a separate barrier assessment methodology in itself and as such, it contributes to problems associated with having multiple different protocols in Europe.

3 METHODS TO ASSESS THE IMPACT OF BARRIERS ON RIVER CONNECTIVITY

Within Europe, the impact of individual or multiple barriers on river connectivity for migratory fish is usually undertaken at large hydropower dams with fish passes and is generally evaluated by undertaking empirical telemetry studies (e.g. Chanseau & Larinier 1998; Gowans *et al.* 2003; Rivinoja *et al.* 2006). These studies are generally carried out by experts, usually acting in accordance with management plans formulated at the catchment scale or smaller. Although these empirical studies provide valuable information on barrier passability at a small scale they are generally very resource intensive and with some notable exceptions (e.g. Winter & Van Densen 2001; Ovidio *et al.* 2007; Lucas *et al.* 2009) they generally focus on salmonids and/or larger barriers. There is a need for coarse-resolution techniques to assess the passability of a wide range of barriers quickly and easily that can be implemented at catchment, national or international scales to facilitate prioritisation of restoration actions. These barrier assessment protocols must be appropriate for a wide range of species and multiple different barrier types. In this report the term 'passability' broadly refers to the proportion of fish that encounter an impediment which can successful pass it (during either an upstream or downstream migration). Within specific protocols, 'passability' may be considered as either binary (i.e. a barrier is passable [1] or impassable [0]), probabilistic (e.g. 0-1) or as a score, with additional factors often incorporated in its definition (e.g. delay, energetic expense or predation risk).

Identification of complete physical barriers to migration (e.g. large dams without a fish pass) is often relatively straight-forward based on knowledge of the physical dimensions of the structure. However, the impacts of temporal and/or partial barriers, such as culverts and weirs, that may not necessarily physically obstruct fish movement, but create impediments under specific flow conditions when depths are insufficient or velocities exceed swimming capabilities are much harder to accurately quantify without resource intensive empirical studies. A solution to bypass the need for costly empirical studies is to use rule-based and/or statistical models to estimating passability based on survey data of physical and/or hydrodynamic properties of in-stream structures. This can be achieved either through (1) basic knowledge relating to the maximum swimming speed and jumping heights of fish (e.g. SNIFFER, 2010a, Sola *et al.,* 2011; Baudoin *et al.,* 2014) or (2) empirical data on passability of a small sample of barriers (e.g. Coffman, 2005).

An in depth synthesis on methods to evaluate and prioritise removal of barriers to fish movement was undertaken by Kemp and O'Hanley (2010). They outlined that to support decision makers, tasked with planning river restoration efforts, appropriate methodologies are needed to: (1) carry out a rapid assessment of instream barriers; (2) store, maintain, and access relevant data pertaining to barrier networks; and (3) prioritise barrier removal and repair decisions to maximise restoration gains. Kemp and O'Hanley (2010) noted that as a result of regional differences in policy objectives, there has been a tendency for uncoordinated development of these tools, generally in an ad hoc fashion, resulting in major inefficiencies, and duplication of effort and frequent repetition of past mistakes. Despite these observations it would seem that little coordination of effort has occurred in the following years and that the development of assessment, reporting and prioritisation tools has generally continued separately between regions. For example, separate barrier assessment protocols currently exist in several European countries despite efforts primarily being driven by the same legislation – the Water Framework Directive (2000/60/EC).

3.1 Existing barrier assessment protocols in Europe

This section of the report focuses on existing barrier assessment protocols that have been developed in Europe. The United Kingdom (SNIFFER), French (ICE) and Spanish (ICF) protocols are discussed in most detail as these are well developed, widely accessible and are fully or partially available in English. Barrier assessment protocols that have been developed outside of Europe are not considered here (e.g. the FishXing model - Love *et al.,* 1999; Furniss *et al*., 2006 - and protocols developed by Coffman (2005) and Clarkin *et al.* (2003), in North America) as they are not always directly relevant to the European situation (e.g. they are generally heavily focussed on road crossings such as culverts and bridge footings and tend to ignore other barrier types) (Kemp and O'Hanley, 2010). For an in depth review on these barrier assessment protocols see Kemp and O'Hanley (2010).

3.1.1 United Kingdom

In the UK, the generally accepted method to accurately assess the passability of barriers for aquatic organisms is the WFD111 method commissioned by the Scotland and Northern Ireland Forum for Environmental Research (SNIFFER) funded by SEPA and NIEA and in partnership with several other UK based agencies and trusts (Table 1) (Kemp *et al.,* 2008; SNIFFER, 2010a,b,c). The methodology, commonly referred to as the SNIFFER methodology, aimed to produce a 'coarse resolution rapid-assessment methodology to assess obstacles to fish migration'. The method was designed for barriers and species encountered in the UK, and can be used to assess natural barriers and a range of man-made structures. Species and life stages considered include adult salmon, adult trout, adult grayling, adult eel, cyprinids, juvenile eel, juvenile salmonids and juvenile lamprey. The criteria for determining passability scores are based on published data describing the swimming and leaping abilities of different fish species/life stages.

Table 1. Entities involved in the formation of the SNIFFER protocol for assessing passability of riverine barriers

Role	Entity
Commissioned by:	Scotland and Northern Ireland Forum for Environmental Research (SNIFFER)
Funded by:	Scottish Environment Protection Agency (SEPA)
	Northern Ireland Environment Agency (NIEA)
Partners:	Environment Agency (EA)
	Fisheries (Electricity) Committee
	Loughs Agency
	Marine Scotland
	Rivers and Fisheries Trusts of Scotland
	Scottish National Heritage
	Scottish Water

It is recommended that the SNIFFER protocol be undertaken in summer low-flow conditions. This is to (1) promote ease of access, (2) so that assessments are undertaken under the most severe conditions in relation to fish passage and (3) to maximise the chance that most of the data can be collected by measurement and not estimation. Non-uniform structures often consist of hydraulically distinct “transversal sections” (TS) which present multiple passage options for fish (Figure 1). Each TS is assessed independently in the protocol. Five physical measurements are recorded along three transects at each TS (Crest/inlet, Mid-point and Foot/outlet) in order to inform passability. At its simplest the assessment protocol requires each velocity and depth measurement location within a TS to be assessed in relation to a fish’s swimming and leaping abilities and ranked as either 0 (complete barrier), 0.3 (partial barrier high impact), 0.6 (partial barrier low impact) or 1 (no barrier) (Table 2). The overall passability score for each TS is the lowest score of the easiest route to ascend the TS. The overall passability score for the entire barrier is equal to the TS with the highest score.

Figure 1. Examples of riverine barriers where transversal variation in flow depth and velocity may lead to provision of alternative passage routes for ascending and descending fish. Red arrows are used to indicate separate traversal sections that provide possible routes for fish passage that would be considered independently within the SNIFFER assessment protocol. Figure taken from SNIFFER (2010a).

The protocol considers multiple different barrier types by categorising barriers as one or a combination of either a (1) jump, (2) swim or (3) depth barrier. As such it is versatile in relation to the types of barriers that can be assessed. It also separately considers barriers with fixed dimension gaps (e.g. culverts), abstraction points and screens (for downstream moving fish).

Table 2. The scores and definitions assigned as part of the SNIFFER protocol to indicate a barriers passability to fish

Passability score	Description
1.0 - Passable barrier	The barrier does not represent a significant impediment to the target species / life-stage, or species guild, and the majority of the population will pass during the majority of the period of migration (movement). This does not mean that the barrier poses no costs in terms of delay, e.g. increased energetics, or that all fish will be able to pass.
0.6 - Partial low impact barrier	The barrier represents a significant impediment to the target species / life-stage, or species guild, but most of the population (e.g. > two-thirds) will pass eventually; or the barrier is impassable for a significant proportion of the time (e.g. < one-third).
0.3 - Partial high impact barrier	The barrier represents a significant impediment to the target species / life-stage, or species guild, but some of the population (e.g. < one-third) will pass eventually; or the barrier is impassable for a significant proportion of the time (e.g. > two-thirds).
0.0 - Complete barrier	The target species / life-stage, or species guild cannot pass the barrier (e.g. impassable falls with no fish pass present) or it is known that fish are unable to pass because the species distribution comes to an abrupt halt at that point.

Based on how the barrier is categorised a number of additional variables (e.g. step height, pool depth, gap dimensions) are required to be logged and assessed in addition to generic additional factors such as the presence of debris. Although the protocol, at its simplest, is objective, final scores are likely to be heavily influence by the experience of the assessor as the impact of additional variables on passability, such as the presence of turbulence, a standing wave and/or debris, are required to be subjectively assessed. Hence overall the protocol is comprehensive but has some limiting factors (Table 3). Using two surveyors and readily available equipment, approximately 5.7 barriers can be surveyed a day using the SNIFFER protocol (King *et al.,* 2016b).

Table 3. Advantages and limitations of the SNIFFER methodology

Advantages	Limitations
• Based on evidence of swimming performance. • Considers multiple barrier types. • Considers multiple species/life stages. • Considers up and downstream passage separately. • Produces passability scores in format useful for later prioritisation. • Produces a subjective assessment of passability under high flow conditions. • Factors in flow heterogeneity at barriers by splitting barrier into separate transversal sections.	• Only truly objective if useful variables are not considered (e.g. presence of debris). • Species considered are UK centric – not appropriate for whole of Europe. • Cyprinids are grouped together – not realistic, range of swimming abilities within UK cyprinids. • The swimming performance data used to define passability is based on forced experiments – when fish are allowed to swim in large channels volitionally swimming performance has been shown to be greater. • Does not consider fish passes.

The SNIFFER methodology is the primary barrier assessment protocol used in the UK, having been used by multiple organisations to assess barriers. However, there is currently no central repository of the survey results, hence the level of use it receives is hard to quantify. In the development stages the protocol was trialled at 45 barriers and validated against expert visual opinion and a subset of fish data at a smaller number of locations (SNIFFER, 2010b). Fish data was limited but where possible the presence and/or abundance of fish upstream and downstream of a barrier were used to assess the validity of the passability scores assigned using the assessment protocol. The total number of barriers where fish data were available for salmon, trout, cyprinids and eels/lamprey was N = 20, 26, 8 and 1, respectively. The results of the testing revealed that the methodology required significant modification in order to enable meaningful scores to be obtained across a wide range of potential riverine barriers (SNIFFER, 2010b). Numerous methodological changes were implemented before the final protocol was released.

The final protocol has been used to assess barriers to fish movement in the River Wey catchment (Eakins *et al.,* 2012; King *et al.,* 2016b). The methodology is used by Inland Fisheries Ireland in cases where barrier mitigation work is planned – with both pre- and post- mitigation surveys undertaken. Although user feedback, expert opinion and presence/absence fish data were used to refine the protocol no comprehensive validation of barrier passability (e.g. using telemetry studies) has been undertaken. Further refinements to the protocol are being formulated currently, based on feedback from a dedicated workshop held in May 2016 and ideas presented in the French ICE protocol (pers comm. Colin Bull). These refinements will be proposed to SEPA later this year (2016).

3.1.2 France

In France the National Agency for Water and Aquatic Environments (ONEMA – Office national de L'eau et des Milieux Aquatiques) has developed the ICE protocol for ecological continuity (Baudoin *et al.,* 2014). It is based on a major review of the current scientific knowledge and on the scientific and technical progress made by a working group composed of French (ONEMA and ECOGEA) and Belgian (University of Liège) experts in this field. Much like the SNIFFER protocol, the ICE protocol is based on a comparison of the topographical and hydraulic characteristics of barriers with the physical capabilities (swimming, jumping or crawling) of the fish species analysed. The protocol requires the identification of the potential passageway(s) through which fish can pass for each barrier (similar to the transversal sections utilised in the SNIFFER protocol) and characterises their geometric features and the hydraulic conditions, and compares the results with the physical capabilities of a given species. Long profiles of each potential passageway are recorded by collecting altimetric data for each specific point in a structure corresponding to a significant change in the profile, e.g. a break in a slope. Although, occasionally, velocimetric data is required to be collected, often it is not required as hydrodynamic equations and modelling have been used to set specific physical thresholds (e.g. head height and slope) above which velocity is estimated to restrict passage.

It differs from the SNIFFER protocol in that a larger range of fish species are assessed (47 separate species / lifestages). These species are grouped into 11 groups with sub-groups according to physical swimming capabilities (Table 4). However, the ICE protocol does not assess the passability of barriers in the downstream direction. A major limitation when considering the vulnerability of certain downstream moving lifestages to river barriers (e.g. silver eel: Piper *et al.,* 2015).

Within the ICE protocol the passability of barriers is defined on a similar scale to the SNIFFER protocol with possible scores being 0 (total barrier), 0.33 (high-impact partial barrier), 0.66 (medium impact partial barrier), 1 (low-impact passable barrier) or NC (barrier having indeterminate impact). Generally, passability scores (PS) are assigned based on the threshold physical values (e.g. depth, velocity, slope etc.) present at the barrier compared to the minimum, average and maximum swimming abilities assigned to each fish group (e.g. Table 4). For example, if the velocity is lower than a fishes minimum swimming speed then PS = 1, if it is between the minimum and average PS = 0.66, if it is between average and maximum PS = 0.33 and if it is above the maximum then PS = 0. To reduce time in the field, thresholds are also outlined (extreme values) whereby the barrier is instantly classed as impassable and no further measurements are required to be recorded.

Table 4. Species and lifestages considered and their assigned swimming capabilities within the ICE protocol.

<table>
<tr><th rowspan="2">ICE species group</th><th rowspan="2">Species [size range mm]</th><th rowspan="2">Jumping species</th><th colspan="3">Swimming speed (m/s)</th><th colspan="3">Height of jump (m)</th></tr>
<tr><th>Min.</th><th>Avg.</th><th>Max.</th><th>Min.</th><th>Avg.</th><th>Max.</th></tr>
<tr><td>1</td><td>Atlantic salmon (Salmo salar)
Brown or sea trout [50-100] (Salmo trutta)</td><td>Yes</td><td>4.5</td><td>5.5</td><td>6.5</td><td>1</td><td>1.5</td><td>2.5</td></tr>
<tr><td>2</td><td>Mullets (Chelon labrosus, Liza ramada)</td><td>Yes</td><td>4</td><td>4.75</td><td>5.5</td><td>0.8</td><td>1.1</td><td>1.8</td></tr>
<tr><td>3a</td><td>Allis shad (Alosa alosa)</td><td rowspan="3">No</td><td>3.5</td><td>4</td><td>5</td><td rowspan="3">/</td><td rowspan="3">/</td><td rowspan="3">/</td></tr>
<tr><td>3b</td><td>Twaite shad (Alosa fallax fallax)</td><td rowspan="2">3</td><td rowspan="2">3.75</td><td rowspan="2">4.5</td></tr>
<tr><td>3c</td><td>Sea lamprey (Petromyzon marinus)</td></tr>
<tr><td>4a</td><td>Brown or sea trout [25-55] (Salmo trutta)</td><td rowspan="2">Yes</td><td>3</td><td>4</td><td>5</td><td>0.5</td><td>0.9</td><td>1.4</td></tr>
<tr><td>4b</td><td>Brown trout [15-30] (Salmo trutta)</td><td>2.5</td><td>3</td><td>3.5</td><td>0.3</td><td>0.5</td><td>0.8</td></tr>
<tr><td>5</td><td>Asp (Aspius aspius)
Pike (Esox lucius)</td><td>No</td><td>3.5</td><td>4.25</td><td>5</td><td>/</td><td>/</td><td>/</td></tr>
<tr><td>6</td><td>Grayling (Thymallus thymallus)</td><td>Yes</td><td>3</td><td>3.75</td><td>4.5</td><td>0.4</td><td>0.75</td><td>1.2</td></tr>
<tr><td>7a</td><td>Barbel (Barbus barbus)
Chub (Squalius cephalus)
Nase (Chondrostoma nasus)</td><td rowspan="2">No</td><td>2.5</td><td>3.25</td><td>4</td><td rowspan="2">/</td><td rowspan="2">/</td><td rowspan="2">/</td></tr>
<tr><td>7b</td><td>River lamprey (Lampetra fluviatilis)</td><td>2</td><td>2.75</td><td>3.5</td></tr>
<tr><td>8a</td><td>Common carp (Cyprinus carpio)</td><td rowspan="4">No</td><td rowspan="4">2</td><td rowspan="4">2.75</td><td rowspan="4">3.5</td><td rowspan="4">/</td><td rowspan="4">/</td><td rowspan="4">/</td></tr>
<tr><td>8b</td><td>Common bream (Abramis brama)
Pikeperch (Sander lucioperca)</td></tr>
<tr><td>8c</td><td>White bream (Blicca bjoerkna)
Ide (Leuciscus idus)
Burbot (Lota lota)
Perch (Perca fluviatilis)
Tench (Tinca tinca)</td></tr>
<tr><td>8d</td><td>Daces (Leuciscus spp. except Idus)</td></tr>
<tr><td>9a</td><td>Bleak (Alburnus alburnus)
Schneider (Alburnoides bipunctatus)
Mediterranean barbel (Barbus meridionalis)
Blageon (Telestes souffia)
Crucian carp (Carassius carassius)
Prussian carp (Carassius gibelio)
Roach (Rutilus rutilus)
Rudd (Scardinius erythrophthalmus)
South-west European nase
(Parachondrostoma toxostoma)</td><td rowspan="2">No</td><td rowspan="2">1.5</td><td rowspan="2">2.25</td><td rowspan="2">3</td><td rowspan="2">/</td><td rowspan="2">/</td><td rowspan="2">/</td></tr>
<tr><td>9b</td><td>Streber (Zingel asper)
Bullheads (Cottus spp.)
Gudgeons (Gobio spp.)
Ruffe (Gymnocephalus cernuus)
Brook lamprey (Lampetra planeri)
Stone loach (Barbatula barbatula)
Spined loach (Cobitis taenia)</td></tr>
<tr><td>10</td><td>Sunbleak (Leucaspius delineatus)
Bitterling (Rhodeus amarus)
Threespine stickleback (Gasterosteus gymnurus)
Smoothtail ninespine stickleback (Pungitius laevis)
Minnows (Phoxinus spp.)</td><td>No</td><td>1</td><td>1.5</td><td>2</td><td>/</td><td>/</td><td>/</td></tr>
<tr><td>11a</td><td>European eel [yellow eel] (Anguilla anguilla)</td><td rowspan="2">No</td><td colspan="3"><1.5</td><td rowspan="2">/</td><td rowspan="2">/</td><td rowspan="2">/</td></tr>
<tr><td>11b</td><td>European eel [glass eel] (Anguilla anguilla)</td><td colspan="3"><0.5</td></tr>
</table>

The protocol outlines decision trees for determining the overall passability scores of the following barrier types for each species group based on recorded physical data:

- Vertical or sub-vertical barriers (slope > 150%);
- Weirs with inclined downstream face (slope ≤150%);
- Rock weirs;
- Barriers comprising gates or where underflows occur;
- Road/rail structures.

The protocol also provides guidance for assigning a PS to complex structures consisting of more than one of the above barrier types. A separate generic decision tree is outlined for eels (glass and yellow eels) for which the presence/absence of a crawl way is considered. A usable crawl way is characterised as a continuous section where the water depth is very low, less than 10 mm for glass eels and less than 20 mm for yellow eels. Tables are provided to estimate the PS based on slope and length of the crawl way.

The ICE protocol also provides guidance to assessing whether the type and characteristics of any fish passes that are present are suitable for a given species. Although the authors outline that this is a "pre-assessment" and that further in-depth analysis is required to accurately assess the efficacy of a fish pass. The pre-assessment method outlines the required sizing parameters for several types of fish pass required to enable the passage of a given species group based on factors such as minimum size of pools, presence/absence of slots/notches, water depths, discharge for given slots/notches and maximum flow-velocities / head-drops between pools. The protocol outlines decision trees for determining whether the following fish pass types are likely to function effectively:

- Pool type passes;
- Pool type passes with skimming flows (vertical slots, deep lateral notches or triangular notches);
- Pool type passes with plunging jets (rectangular notches, pre-barrages, triangular notches);
- Rock-chute fish passes;
- Rock-chute fish passes with successive rows of elements;
- Rock-chute fish passes with staggered arrays of elements;
- Denil fish passes;
- Fish passes designed specifically for eels.

For each species group the fish pass is classed as either 1) negative - does not comply with the general sizing criteria (i.e. the pass is not well suited or not at all suited to the given purpose) or 2) positive - complies with general sizing criteria however it is necessary to proceed with an in-depth study of the hydraulic conditions in the fish pass and of its attractiveness.

It is recommended that the ICE protocol is carried out under the hydrological conditions most common during the migratory period of the given species. Baudoin *et al.* (2014)

acknowledge that although low-flow conditions make it easy to access structures and measure the various structural features, they also significantly alter the passability of the structure (e.g. maximum head-drop, minimum depths, less depth in the plunge pool etc.) and hence are not representative of the hydrological conditions most common during migratory periods. Table 5 summarises some of the advantages and limitations of the ICE protocol.

Table 5. Advantages and limitations of the ICE protocol.

Advantages	Limitations
• A very wide range of species are considered. • Species are grouped based on evidence of physiological capabilities. • Considers multiple barrier types. • Produces passability scores in format useful for later prioritisation. • Requires assessment to be undertaken at likely peak migration times – i.e. more applicable results. • Species groupings easy to add to if used elsewhere. • Considers fish passes. • Factors in flow heterogeneity at barriers by splitting barrier into separate transversal sections.	• Does not consider downstream passage. • Certain barrier types / fish pass classed as to difficult to assess accurately without more in-depth analysis by experts. • The swimming performance data used to define passability are based on forced experiments – when fish are allowed to swim in large channels volitionally swimming performance has been shown to be greater. • Standardised recording sheets and full guidance protocols are currently only available in French.

The ICE protocol is used on the whole French territory, specific studies have utilised it in Belgium and Portugal and it has been adapted to be used in the French overseas departments and territories with different fish species (pers. comm. Michael Ovidio). The general methodology is available in English (Baudoin *et al.,* 2014) but the field implementation guidance and recording sheets are currently only available in French (ONEMA, 2015). Similar to the SNIFFER protocol, although it is known to be widely used, the results are not currently compiled centrally and hence the level of use it receives is difficult to quantify. Currently the location and simple characteristics of over 90,000 barriers in France are logged on a database called ROE (Référentiel des Barriers à l'Ecoulement – Repository of Barriers to Flow). This inventory is scheduled to be updated with existing ICE data towards the end of 2016 / beginning of 2017 (pers. comm. Karl Kreutzenberger) providing an interactive tool for viewing and utilising the barrier passability scores. Although no direct validation of the ICE protocol has been undertaken the results of a molecular study, aimed at measuring the adaptive potential of fish populations under climatic influence in fragmented versus not-fragmented environments, have proven to be complementary to the ICE protocol results when compared (pers. comm. Karl Kreutzenberger).

3.1.3 Spain

In Catalonia, an autonomous community of the Kingdom of Spain, the index of river connectivity (ICF, from the Catalan name Index de Connectivitat Fluvial) has been formulated to evaluate the influence of barriers on fish movement (Sola *et al.,* 2011). Originally conceived in 2006 the ICF was designed as part of a procedure to assess the hydromorphological quality of Catalan rivers (HIDRI protocol - ACA, 2006). Application of this index by several consultancies and research centres (Ferrer *et al.,* 2009; Ordeix *et al.,* 2006; Rocaspana *et al.,* 2009) revealed the existence of deficiencies that yielded a final result that did not coincide with real longitudinal connectivity evaluated independently and it has been corrected since then (Sola *et al.,* 2011). The index is based on the comparison of the physical characteristics of a barrier and/or fish pass with the capabilities of the fish present in the river section to overcome the barrier. Applying the ICF index involves five stages: 1) determining fish present in the river, 2) classifying fish fauna according to specific groups, 3) assessing the barrier and fish pass (if present), 4) assessing capacity for fish groups to overcome barriers and 5) assessing some final modulators. The index categorises barriers based on the chance it can be crossed by all species groups, only by some species groups, or by no species groups.

The index currently considers 23 distinct species as well as *Cobitus* sp. (loaches) combined into 4 main groups with subgroups (Table 6). The fish species on which the index is based are those characteristic to Catalan continental waters. However, the author's state that the index is easily adaptable to be used with aquatic fauna from other geographical regions (Sola *et al.,* 2011). Similar to both the SNIFFER and ICE protocol the index considers the physical capabilities of each species / species group against physical measurements recorded at the barrier (e.g. depth and velocity). Specific threshold values for each species group are provided and if these thresholds are surpassed then that species group is considered not able to pass the structure. Unlike the SNIFFER and ICE protocols the ICF index considers barrier passability for each species group to be binary, either they can or cannot pass. In addition, unlike the other protocols a single depth and/or velocity measurement is taken at each structure (assumed to be a representative measurement). These representative measurements are used to assess the passability of the structure to each species group. As such, flow heterogeneity across the structure is ignored.

Within the index, barriers are categorised according to (1) structures that water passes over to create a small waterfall, (2) structures in which water passes through one or more holes and (3) structures with very little slope where water flows over but does not generate a water fall. Fish passes are categorised according to (A) those comprising of close-to-nature like conditions (such as fish ramps, bed ramps, lateral rivers or canals), (B) technical fish passes (further differentiated into devices that consist of steps or those that are ramp like) and (C) mechanised or specific technical fish pass devices (e.g. gates, lifts, locks and fish pumps, or devices specific to one or few species, such as eel ramps). Physical measurements and associated water velocities of the barrier and fish pass are required and sampling sheets are available to allow data to be easily logged in the field (see Sola *et al.,* 2011). Different barrier / fish pass types require different measurements to be taken and these are detailed within the recording sheets. Surveyors are also required to decide whether the barrier is suitable for creeping fish species to pass (i.e. eels) based on whether there are rough margins (e.g., presence of vegetation, roots, substrate heterogeneity), short slopes that are not too steep and particular wetted conditions exist (i.e. a subjective assessment).

Table 6. Species groups utilised in the ICF protocol to produce a passability score for a riverine barrier.

Group	Group Definition	Subgroup (if applicable)	Subgroup Definition (if applicable)	Species	
				Common	**Latin**
Group 1 (G1) - **Littorals and similar**	Migratory species (anadromous or amphidromous) with short or long distance movements, with a moderate or low capacity to overcome barriers	Group 1a (G1a)	Large species, with a moderate capacity to overcome barriers	Allis shad Twait shad Thinlip mullet Thicklip mullet Flathead grey mullet	*Alosa alosa* *Alosa fallax* *Liza ramada* *Chelon labrosus* *Mugil cephalus*
		Group 1b (G1b)	Small or benthic species, with a low capacity to overcome barriers	Big-scale sand smelt European flounder Sea lamprey	*Atherina boyeri* *Platichthys flesus* *Petromyzon marinus*
Group 2 (G2) - **eels and similar**	Migratory species (catadromous), with long distance movements and high capacity to overcome barriers but not able to jump			European eel	*Anguilla anguilla*
Group 3 (G3) - **cyprinidae and similar**	Intra-river migratory species (potamodromous) with a moderate or low capacity to overcome barriers	Group 3a (G3a)	Large species, with a moderate capacity to overcome barriers	Mediterranean barbel Catalonian barbel Ebro barbel Catalan chub Iberian chub Ebro nase Chabot des Pyrénées	*Barbus meridionalis* *Barbus haasi* *Luciobarbus graellsii* *Squalius laietanus* *Squalius pyrenaicus* *Parachondrostoma miegii* *Cottus hispaniolensis*
		Group 3b (G3b)	Small species, with little capacity to overcome barriers	Adour minnow Eurasian minnow Lanquedoc stone loach Freshwater blenny Loaches Bermejuela Three-spined stickleback	*Phoxinus bigerri* *Phoxinus phoxinus* *Barbatula quignardi* *Salaria fluviatilis* *Cobitis sp.* *Achondrostoma arcasii* *Gasterosteus aculeatus*
Group 4 (G4) - **trout and similar**	Intra-river migratory species (potamodromous) with a high capacity to overcome barriers, by swimming and/or jumping			Brown trout	*Salmo trutta*

The index does not generate a passability score for each species group. The barrier passability score is formulated depending on whether they are permeable to all potential fish groups (75), only to some groups (50), only to one fish group (25) or if it is not permeable to any group (0) considering only the groups present in the local area. When there is only one potential fish group present (e.g. in a high mountain stream where only group 4 may be present – Table 4) the maximum score is assigned if fish can pass (75) and the minimum if they cannot pass (0). If there are only two potential groups in a section and only one of them can pass, a score of 50 points is given. The score (0, 25, 50 or 75) is then modulated based on inspection of complementary attributes (modulators) that are likely to increase or decrease the passability of the barrier (Table 6). For example, the presence of natural substrate in a fish pass will increase the final score of the fish pass by 10 points, whilst the absence of guidance mechanisms to a downstream bypass (if present) will decrease the final score by 5 points (Table 6).

Table 6. ICF modulators and score adjustment weights

Category	Modulator	Score
Barrier modulators	The morphology of the barrier allows, in high flows situation or temporarily, water to pass through one or both sides, allowing the fish to go upstream.	+5
	Only in low slope barriers (<45%), if its surface is rough and irregular	+5
	Presence of any overhanging structure at any point of the infrastructure	-5
Fish pass modulators	Presence of a natural substrate, with similar characteristics to the one in the river, inside the fish pass	+10
	Correct location of the fish pass entrance (from downstream to upstream)	+5
	Wrong location of the entrance (from downstream to upstream)	-5
	Width of the wet part of the fish pass below 1/20 average width of the river	-5
	Fish pass with gates or cross-walls that need a constant maintenance to guarantee its functionality	-5
	Fish pass in a bad condition of preservation or maintenance	-10
Downstream movement modulators	Fish can migrate downstream safely and directly through the barrier (i.e., low height barrier (<10m), sufficient water depth, or close to nature fish pass)	+5
	If there is a derivation canal, a mechanism exists to help fish avoid or minimise the risk of entering into the derivation canals (mechanical, light, sound or electrical). Or if there is not derivation canal.	+5
	If there is a derivation canal, no mechanisms exists to help fish avoid or minimise the risk of entering into the derivation canals.	-5
	Downstream migration directly through the barrier is possible but with risk of injury or death (i.e. fall of more than 10 m)	-5

The final ICF index value for the barrier can range from 0 to 110 with barriers being classified into one of five quality levels ranging from ‘very good’ to ‘bad’ depending on the degree of passability for the different fish groups present (Table 7).

Table 7. ICF classification categories.

Range	Quality	Interpretation
≥ 95	Very Good	All the potentially present groups of fish can pass in nearly any hydrological situation. Absence of barriers for fishes or existence of a partial or total demolition of a barrier for fishes.
75-94	Good	The majority of the potentially present fish groups can pass in nearly any hydrological situation. Presence of a small barrier or with a good fish pass for fishes.
50-74	Moderate	The majority or some of the potentially present fish groups can pass, in any or in some hydrological conditions. Presence of a relatively permeable barrier for fishes with too specific or little functional fish pass for fishes.
25-49	Poor	Only one or few species of the potentially present fish groups can pass, and in determined hydrological situations. Presence of a barrier with very specific or very little functional fish pass.
< 25	Bad	No species of the potentially present fish groups or only some in very exceptional hydrological situations can pass. Presence of a quite big barrier without any fish pass or with little or non-functional fish pass.

The original ICF index (HIDRI protocol - ACA, 2006) was validated by several consultancies and research centres with deficiencies in the protocol identified that yielded a final result that did not coincide with real longitudinal connectivity (Ferrer *et al.,* 2009; Ordeix *et al.,* 2006; Rocaspana *et al.,* 2009). The new index has been updated in numerous ways with the main difference in the results being a general increase in the final score awarded to barriers with fish passes (Sola *et al.,* 2011). Validation of the new index at seven barriers with fish passes, using a mix of direct observation, fish pass intake traps, up and downstream electrofishing and trapping and mark-recapture studies, revealed comparable results to *in situ* measurements of fish fauna movements (Ordeix *et al.,* 2011). The ICF index is widely used throughout Catalonia (e.g. Ordeix et al., 2011) and Portugal (e.g. Bochechas, 2015) and has been used in Estonia (Pensa, 2016). Advantages and limitations of the protocol are summarised in Table 8.

Table 8. Advantages and limitations of the ICF protocol.

Advantages	**Limitations**
• A wide range of species are considered. • Species are grouped based on multiple criteria (swimming capabilities and likely position in catchment). • Considers multiple barrier types. • User friendly (simple). • Produces a single passability score (very useful for prioritisation models). • Species groupings relatively easy to add to if used elsewhere. • Considers fish passes.	• Downstream passage considered in a simplistic manner. • Considers passage to be binary. • Passability score based on a simplistic comparison of swimming capability versus a barriers physical attributes. • The swimming performance data used to define passability are based on forced experiments – when fish are allowed to swim in large channels volitionally swimming performance has been shown to be greater. • Not suitable for barriers which display transversal flow heterogeneity.

It is worth noting that a limitation of the ICF protocol, the fact that transversal flow heterogeneity is ignored, could easily be overcome by divided a barrier into spate hydrodynamic sections (e.g. Figure 1), with each section being treated as a separate barrier. The final whole barrier passability score would be equivalent to the traversal section with the highest score (e.g. analogous to how structures which display transversal flow heterogeneity are treated in the SNIFFER and ICE protocols).

3.1.4 Germany

A methodology for assessing barriers to fish migration in Germany is provided in the Handbuch Querbauwerke (Dumont, 2005) and was translated from German to English for the purpose of a review undertaken by Kemp *et al.,* (2008) and Kemp and O'Hanley (2010).

The German protocol outlines a methodology to standardise barrier assessment, collate and maintain existing data on barriers to fish migration, and enables identification of sites where further surveying is required (Dumont, 2005). It functions at both the site and catchment scale and facilitates water bodies being designated an ecological status in line with the requirements of the WFD (Dumont, 2005). At the site scale, upstream migration is assessed in relation to the following factors: (1) passability or porosity of the barrier itself; (2) attraction of a fish pass if present; and (3) efficiency of the fish pass for enabling upstream movement (Dumont, 2005). For upstream passage to be considered feasible at least one route must be passable. For downstream migrating life-stages, passage past barriers is considered feasible if a free route is available and fish can disperse undamaged. To assess this the following factors are considered: (1) ratio of discharged diverted flow to overall flow of the river (used to indicate the probability of downstream migrants entering the operational channel or hydropower plant); (2) existence of a safe route of passage; (3) injury rate associated with barrier passage (e.g., injury associated with passing weir and collision in the tail water zone); and (4) injury rate associated with passage through a hydropower facility or water diversion system. For both upstream and downstream passage, each factor is assessed by using tables to assign passage routes to well-defined categories, with categories ranging from A (unobstructed) to E (completely blocked) (e.g. Table 9).

Unlike other protocols, the German considers a temporal component of fish passage, i.e. guidelines provide a general rule that fish passage facilities should guarantee flawless functioning for at least 300 days per year (Dumont, 2005). However, the exact timings of fish migration are not considered which could influence passability. The strengths of the methodology relate to the considerations of species other than salmonids (including juvenile life-stage) moving both upstream and downstream. Of particular interest is that the German methodology combines barrier passability assessment with an evaluation of hydropower potential at the sites surveyed. The primary factors of interest were the technical and economic feasibility of developing hydropower potential of each site and for catchments as a whole (discussed further in section 5.1.2). Thus, the development of new hydropower is considered in conjunction with the removal or repair of barriers. This system should not, therefore, be considered solely as a tool for prioritising restoration actions, but also as a planning tool for maximising the economic gain of future river development in an environmentally sensitive manner. One limitation is that much of the classification system seems to be based on user opinion and hence a high level of subjectivity is likely to exist in the results.

Table 9. Example classification table from Dumont, 2005. Table for assessing attraction and porosity for downstream migrating fish at a barrier. From Kemp *et al.,* 2008.

Class	Ecological status for fish	Technical criteria regarding a barrier	Technical criteria regarding a fish pass
A	Unimpaired downstream migration of fish.	There is no diversion channel.	No hydropower plant or water diversion system installed.
B	The attraction of migration corridor is only slightly impaired.	Only slight diversion of the water (maximum 25% mean flow MQ), so most of the fish migrating downstream can pass the barrier. The passability of the bypass meets or exceeds the minimum flow according to classification B in Table 9.	The fish passage facility is next to water diversion system according.
C	The attraction of migration corridor is only moderately impaired.	Diversion of the water is up to 50 % mean flow (MQ), so only a small proportion of the fish migrating downstream can pass the barrier. The passability of the bypass is at least as classification C in Table 9.	The position and flow of the fish passage facility next to the water diversion facility deviates moderately from the criteria provided.
D	The attraction of migration corridor is strongly impaired.	Water diversion up to 100 % MQ (mean flow).	The position and flow of the fish passage facility next to the water diversion facility deviates strongly from the criteria provided.
E	No migration corridor exists or attraction is extremely poor	Water diversion over 100 % MQ (mean flow).	No, or no functioning, fish passage facility next to the water diversion system **or** fish pass attraction is extremely poor.

The methodology has been utilised throughout Germany (Dumont *et al.,* 2006) and is used as part of the assessment protocol to assign the ecological status of a water body as part of legislative requirements under the WFD. However, it is uncertain if any direct validation against telemetry studies at specific sites has been undertaken.

3.1.5 South Eastern Europe

A recent EU funded program to aid in the development of hydropower in South Eastern Europe, the 'SEE HYDROPOWER' project, assessed the methodologies of Austria, Slovakia, Romania and Italy for prioritising and restoring longitudinal connectivity in the Danube river basin (Mielach *et al.,* 2012). The methods for each country were as follows:

3.1.5.1 Austria

In Austria a standardised prioritisation approach for the restoration of river continuity is present. They implement a stepwise achievement of objectives, beginning with undertaking mitigation activities for long and medium distance migratory species (continuity warranted until 2015) and ending with short distance migrators (continuity warranted until 2021/2027). Legislation requires that fish passes be utilised on barriers to fish migration but a formalised way to assess the passability of barriers does not seem to exist. However, a standardised scoring system has been formulated to assess the efficiency of fish passage facilities ranging from I (totally functioning) to V (not functioning) based on an average score taken from combined qualitative and quantitative assessments of fish passage efficiency. Elements of the assessment procedure require hydraulic and physical measurements of the fish pass to be taken and compared against thresholds formulated for different species based on morphology and physical swimming capabilities. However, as this assessment protocol only considers fish passes and not the barrier itself it is not discussed further. The assessment protocol is detailed by Woschitz *et al.* (2003) but is currently only available in German (see Mielach *et al.,* 2012 for more detail).

3.1.5.2 Slovakia

In Slovenia, there is no standardised prioritisation approach for the restoration of river continuity. Mitigation decisions are currently based on expert knowledge. For rivers with trout as the dominate species, barriers higher than 1m must be equipped with a fish pass (Mielach *et al.,* 2012).

3.1.5.3 Romania

In Romania, barriers are prioritised for mitigation if the height of barrier is below 15 m (i.e. a barrier height for which fish passes are considered a technical feasible solution) and they are located on a watercourses with migratory fish species present. Outside these two categorisations mitigation actions are assessed case by case (Mielach *et al.,* 2012).

3.1.5.4 Italy

No standardised prioritisation or barrier assessment approach for the restoration of river continuity (Mielach *et al.,* 2012).

3.2 Statistical methods for predicting barrier attributes and estimating passability

An alternative to physically measuring barrier attributes is to estimate them based on environmental variables. These can then be related to fish swimming abilities to predict barrier passability (e.g. Januchowski-Hartley *et al.,* 2014). Januchowski-Hartley *et al.* (2014) used a statistical model based on data from 2235 road culverts, to predict expected outlet drop and water velocity at over 249,310 potential road crossing locations throughout the Laurentian Great Lakes Basin, United States, based on the following environmental predictors: (1) upstream area draining to the culvert, (2) stream segment gradient, (3) stream reach gradient and (4) slope at the site of the culvert. They used the estimated outlet drop and water velocity at each road crossing to predict culvert passability based on the swimming ability of three fish groups (weak, moderate and strong swimmers). The results of their model enabled them to predict the probability of passage for each fish group at every potential culvert within the Laurentian Great Lakes Basin and highlight areas that might be particularly problematic for the dispersion of aquatic organisms. This type of analysis has the potential to be very powerful especially if used in combination with prioritisation models (e.g. King *et al.*, 2016b). However, it should be noted that the model was based on very simplistic passage criteria (i.e. only outlet drop and velocity) and only culvert type road crossings were considered. The model would not be suitable for assessing passability and prioritising mitigation in areas with a high abundance non-culvert type barriers without considerable adaptation.

3.3 Comparison of existing protocols

This section of the report focuses on directly comparing the three protocols that are most well developed, widely accessible and fully or partially available in English: The SNIFFER, ICE and ICF protocols.

3.3.1 Subjectivity

A key rational for having a standardised protocol is to remove subjectivity in the assessment process and hence reduce the requirement for experts to undertake the assessments. Although the SNIFFER, ICE and ICF protocols provide a standardised format to assess barriers, a certain amount of subjective opinion is still required to complete some sections. The ICE protocol is the most objective as it provides clearly defined decision trees based predominantly on specific threshold values for a large range of barriers and species. However, a by-product of this strict objectivity is that complex passage problems which are not easily classified are not assessed within the ICE framework. For example, it does not consider downstream passage assessable due to the "complexity of the biological mechanisms involved and the in-depth knowledge required on the local hydrology, on draw-off conditions and on the hydro-mechanical characteristics of each structure" (Baudoin *et al.,* 2014). The ICF and SNIFFER protocols require more subjective opinions to be given. For example, the SNIFFER protocol requires opinions on the impact of debris and turbulence on passage to be made with no physical measurements taken and ICF protocol requires certain modulators, such as 'whether fish can pass downstream with no physical injury or mortality', to be subjectively assessed with minimal guidance. However, both these protocols do consider downstream passage as part of their standardised format. Although it should be acknowledged that this is considered in a very simple way within the ICF protocol.

3.3.2 Species considered

There is very little cross over in relation to the individual species considered for each protocol. Out of the combined 55 individual species named in the ICE, SNIFFER and ICF protocol only 11 are named in more than 1 protocol and only 2 species (brown trout and European eel) are named in all three. However, this simplistic assessment should be treated with caution as the ICE protocol also considers bullheads (*Cottus spp.*), gudgeons (*Gobio spp.*) and Daces (*Leuciscus spp.* except *Idus*), the ICF considers Loaches (*Cobitus spp.)* and the SNIFFER protocol considers lamprey species (assumed to be *Petromyzon marinus* and *Lampetra spp.*) together. Hence there is likely to be greater crossover in the species considered than is simply quantifiable. What is clear is that the species considered in each protocol are country-specific and they will not be directly applicable everywhere in Europe. However, the species groupings formulated in the ICE and ICF protocols offer a useful foundation to add and remove species. The ICE groupings have been formulated solely on physical movement capabilities (e.g. swimming, leaping and crawling) whilst the ICF groupings have been formulated based on the likely locations that the species are found in a catchment and then subdivided according to physical movement capabilities. Hence, it could be argued that the ICF grouping system is more comprehensive. However, it is uncertain whether the current proposed 4 groups with subgroups outlined in the ICF protocol are suitable for a greater range of species to be added without modification (i.e. whether they would be applicable at a European level). One potential task for the future may be to formulate a list of generic European species / species groups that would act as suitable proxies for all of Europe.

As mentioned previously, the ICF protocol groups fish both on physical movement capabilities and considering likely changes in species distributions. For example, littoral species that need only to be considered near the coast form one group whilst intra-river migratory species (e.g. trout) that are likely to be found throughout the entire catchment form another. The benefit of this system is that the ICF index considers barrier passability against the species likely to be present in the local area and produces a single barrier passability score for all species. The ICE and SNIFFER protocol, which require individual species / species groups to be considered separately and hence multiple barrier passability scores are generated, may make later prioritisation models more complicated. However, it should also be considered that to generate the single barrier passability score the ICF protocol considers passability for each species group as binary: a species group can either pass or it cannot (the final score is a function of what proportion of species groups present in the area can pass). The SNIFFER and ICE protocols grade a barrier for each species / species group according to its likely level of impact on passability (4 grades for both the SNIFFER and ICE protocol – e.g. Table 2). Hence, for each species / species group, the ICE and SNIFFER protocols provide higher resolution data than the ICF protocol.

3.3.3 Swimming capabilities

Below we consider the swimming capability thresholds utilised for the two species that are incorporated into all three protocols, brown trout and European eel. Only threshold levels beyond which fish are considered no longer able to pass upstream are compared as these are the only values that are directly comparable between the protocols.

3.3.3.1 Brown trout (Salmo trutta)

The swimming capability thresholds of brown trout utilised in each protocol are outlined in Table 10. The major differences are that the SNIFFER protocol considers brown trout to be able to jump *ca.* 0.2 m higher with a much shallower pool depth than the other protocols, the ICE protocol considers trout to be able to swim between 0.5 and 1.1 m s^{-1} faster than the SNIFFER and the ICF protocols, respectively, and the SNIFFER protocol considers trout to be able to swim in water 5 cm shallower than the other two protocols. However, it is important to consider that these comparisons deal only with the thresholds beyond which fish are considered no longer able to pass upstream. Both the SNIFFER and ICE protocol further categorise physical values below these threshold levels according to a likely scale of impact on passability (e.g. Table 2) whereas the ICF protocol just considers any values below the thresholds as simply within the ability of a species to pass. In addition, the exact life stage each protocol is considering is not always clear. For example, the SNIFFER protocol refers to brown trout but does not indicate which size of fish these swimming capabilities have been formulated on and the ICF protocol only states that swimming capabilities were formulated based on the average size of individuals within the population. Whereas the ICE protocol differentiates between three size classes of brown trout (brown or sea trout [50-100cm], brown or sea trout [25-55cm] and brown trout [15-30cm]). For comparability to the other two protocols the smallest size of brown trout was considered but in reality the swimming capabilities used in each protocol may not be comparable as they have been formulated specifically for the lifestages/sizes of fish in the country of origin. What is apparent is that if the protocols were used to assess the same barrier they may produce different passability results for the same species. Users should consider these factors and assess the relevance of the threshold values assigned to each species / species groups within each protocol before selecting the most appropriate protocol to use.

Table 10. Threshold physical limits beyond which brown trout, *Salmo trutta*, are considered not to be able to pass a barrier within the SNIFFER, ICE and ICF protocols.

Physical variable	Threshold	SNIFFER	ICE*	ICF
Head difference (*h*)	Max.	1.0 m	0.8 m	0.75 m
Water velocity (*v*)	Max.	3.0 ms^{-1}	3.5 ms^{-1}	2.4 ms^{-1}
Slope (*s*)	Max.	60%, 40% and 15% at barriers with effective lengths of ≤ 3 m, 4-9 m and ≥ 10 m, respectively.	N/A	30 %
Water depth (d_w)	Min.	0.05 m	0.1 m	0.1 m
Pool depth (d_p)	Min.	0.3*h*	Ranges from 0.7*h* when *h* > 2 m to 1.2*h* when *h* ≤ 0.25 m	1.25*h*

*for the ICE protocol data was extracted for the 15-30 cm brown trout group.

3.3.3.2 European eel (*Anguilla anguilla*)

Due to the European eel's unique ability to overcome barriers by climbing/crawling, all the protocols primarily assess passability for this species in a similar way (i.e. in relation to whether a usable crawlway is present). However, key differences in assessment methods still exist between each protocol. For example, the SNIFFER protocol considers juvenile eel to be able to pass any barrier where a climbing substrate is present, with the presence of a climbing substrate being subjectively assessed. When no climbing substrate is present, SNIFFER considers structures over 30 m long, with high levels of turbulence (subjectively assessed), water depths of < 0.02 m and water velocities of > 0.8 m s^{-1} to be complete barriers to the upstream movement of eel. The ICF protocol considers the eel to be able to pass if the bank morphology is suitable for creeping fish (subjectively assessed). If the banks are not suitable for creeping fish then slope must be < 45 %, water depth > 0.01 m, water velocity < 1.7 m s^{-1} and head difference < 0.2 m for eel to be able to pass the barrier. The ICE protocol considers glass eel (60 - 120 mm) and yellow eel (120 – 400 mm) separately. Both lifestages are considered able to pass if a continuous crawl way exists over the barrier with very low flow (< 10 mm and < 20 mm water depth for glass and yellow eel, respectively). If no crawl way exists then glass eels are considered not to be able to pass, whilst yellow eel can only pass if water depth is > 0.02 m, water velocity is < 1.5 m s^{-1} and the head drop (if skimming flow) is < 0.5 m. Hence, although similar methods are used to assess the passability of a barrier by eels (i.e. the presence of a crawl way, followed by more general swimming capabilities) the way this is assessed and the thresholds used varies between the protocols. For example, the presence of a crawlway is assessed subjectively in the ICF and SNIFFER protocols compared to being assessed according to defined water depth limits in the ICE protocol. In addition, the SNIFFER protocol considers eels to need much slower velocities to be able to overcome barriers (< 0.8 m s^{-1}) compared to the ICF (< 1.7 m s^{-1}) and the ICE (< 1.5 m s^{-1}) protocols. Hence, as with brown trout, each protocol could produce a different passability score for European eel if used to assess the same barrier. Protocol users should check the defined swimming limits for each species and check they are suitable for the size/lifestages of species in the relevant water body / country before using a protocol.

3.3.4 Results

Two riverine barriers were assessed using the SNIFFER, ICE and ICF protocols to provide some evidence on the comparability of the results produced. The two barriers consisted of a Flat V gauging weir (50.953858°, -1.370071°, Figure 2) and a ford road crossing (50.955059°, -1.370958°, Figure 3), both on Monk's Brook, a tributary of the River Itchen, Hampshire, UK. The flat v gauging weir spanned the width of the river (15.2 m), the head difference between upstream and downstream was very low (0.06 m), the wetted width at the weir crest was 4 m and the plunge pool depth below the structure was 0.23 m. The ford road crossing consisted of a flat 17.2 m long concrete plinth with a stepped weir at the downstream end (6 exposed steps of uniform dimensions: length: 0.24 m, height 0.03 m). The ford and weir spanned the width of the river (20.3 m) and produced very shallow flows (*ca.* 4-6 cm). The head difference across the ford and weir was 6 and 26 cm, respectively (total: barrier head difference: 32 cm). The plunge pool depth below the stepped weir was 0.25 m. All assessments were undertaken on 13 September 2016 under summer low flow conditions. Guidelines for each protocol were followed. Where subjective opinions were required they were heavily tailored towards guidelines incorporated in each protocol rather than user opinion to make the assessments as objective as possible. For simplicity, passability scores for fish moving downstream are not considered.

Figure 2. Flat V gauging weir (50.953858°, -1.370071°) on Monk's Brook, a tributary of the River Itchen, Hampshire, UK.

Figure 3. Ford road crossing (50.955059°, -1.370958°) on Monk's Brook, a tributary of the River Itchen, Hampshire, UK.

3.3.4.1 Flat V weir results

Results of the ICE protocol indicate that the Flat V weir would be impassable to all species / species groups considered except European yellow and glass eel (groups 11a and 11b, respectively). Results of the SNIFFER protocol indicate that the flat V would be impassable to all species / species groups considered accept juvenile eel for which the barrier was passable (PS = 1) and adult Lamprey and juvenile salmonids for which it would be a partial high impact barrier (PS = 0.3). Results of the ICF protocol indicate that the flat V was impassable to only small or benthic littoral or similar species with a low capacity to overcome barriers (group 1b) and small cyprinidae and similar species with little capacity to overcome barriers (group 3b). All other species / species groups considered could pass. If all of the species groups are considered to be present in the area the overall ICF score assigned to the barrier would be 55 (Moderate impact – see Table 7). As such, there is a clear difference in the results produced by each protocol for this barrier. The key differences in the results stem from the ICF protocol not considering the depth of water flowing over the crest of an barrier to be an obstacle to jumping fish if the crest is < 50 cm wide (longitudinally). Whereas despite having a very narrow crest the SNIFFER and ICE protocols considered the limited water depth flowing over the crest of the flat V to present a barrier to upstream jumping fish. The further subtle differences between the results of the ICE and SNIFFER protocols (e.g. for adult lamprey and juvenile salmonids) stem from differences in the minimum swimming depths required for the successful upstream passage of certain species during normal swimming (i.e. adult lamprey and juvenile salmonids are considered to be able to swim in shallower water in the SNIFFER protocol than in the ICE protocol). In this circumstance the ICF protocol is likely to be a more accurate reflection of actual passability of the barrier as it is our opinion that highly motivated jumping species would probably be able to jump or swim over the flat V weir under the conditions present.

3.3.4.2 Ford road crossing results

For this barrier, results of all three protocols were in agreement with the only species being able to pass being the European eel. In this circumstance the ford presented a depth barrier for the upstream movement of most species and the structure was considered too long for fish to be able to successfully jump past the barrier. Eel were considered able to pass by each protocol due to the presence of shallow margins on either bank that provided a usable crawlway.

3.3.4.3 Overview

These results emphasise the potential for the protocols to disagree on the passability score assigned to a potential barrier, highlighting that these types of rapid assessments should be used with caution until further validation is undertaken. It should also be noted the above assessments were undertaken under low flow summer conditions, as recommended for the SNIFFER protocol. However, the ICE protocol suggest that assessments should be undertaken under “hydrological conditions most common during the migratory period of the given species” and the ICF should be undertaken under conditions which are “representative of the normal conditions of the evaluated river section”. Although this does not impede the comparability of the protocols under the same conditions it is worth noting that each protocol has been designed to function under certain conditions. For example both the, ICE and ICF protocols expect the results to be directly relevant as they should have been taken at peak migration times or during average flow conditions. The SNIFFER protocol seeks to maximise the possibility of collecting data by recommending that assessments be undertaken under low flow conditions and then asks the assessor to estimate passability under high-flow conditions (not considered here).

3.4 Conclusions

The primary application of course-scale rapid barrier assessment protocols is to enable passability data to be collected on a large number of riverine barriers cost effectively and efficiently. They are not intended to be a direct replacement for more accurate and comprehensive quantitative assessments of barrier passability using techniques such as telemetry (e.g. PIT, radio or acoustic) or mark-recapture studies. They should however function to provide an accurate indication of whether a structure is likely to negligibly or significantly affect passability in order to guide future management decisions. The above mentioned European protocols have mostly been developed independently of one another and although they are fundamentally based on the same principle (i.e. the comparison of the swimming capabilities of fish to the physical and hydraulic conditions present at a barrier) the processes and methods used to produce the final passability scores are very different. Importantly, comprehensive validation and subsequent iterative improvement should be a key consideration during the development of assessment tools (Coffman 2005; Kemp and O'Hanley, 2010). Comprehensive validation should be undertaken using telemetry studies rather than using more coarse scale techniques (e.g. presence/absence of fish above a barrier), as well designed telemetry studies provide data on passage efficiency (the proportion of fish that pass an barrier that wish to do so), rather than a more generic assessment of whether some fish can pass or not. Although some initial testing has been undertaken, the current versions of the most comprehensive protocols (SNIFFER, ICE, and ICF) have not been adequately validated for a range of species at a range of barrier types. As inconsistencies between predicted and actual passability could lead to suboptimal management decisions with resulting economic and ecological costs (King *et al.,* 2016b) comprehensive validation of the above protocols is considered essential before they more widely used.

Although the ICF is more simplistic than the SNIFFER or ICE protocol a key advantage is that it produces a single score that represents barrier passability for all species. Trialling of the assessment protocols on two local barriers (Section 4.3.4) indicated that the ICF performed well (based on our expert opinion, rather than any actual validation) for predicting the passability of barriers despite its simplicity. In addition, with a simple adaptation of the protocol (undertaking separate assessments for differing transversal sections, e.g. Figure 1, and selecting the highest of the resulting passability scores) the protocol would be appropriate for use on more complex structures that display transversal flow heterogeneity. Hence for the purpose of producing data for effective prioritisation models that aim to produce results relevant for all fish species the ICF protocol may be the most appropriate. It is also the simplest to undertake and likely to be the quickest and therefore cost effective of the three protocols. For specifically targeting restoration of a key species, the SNIFFER or ICE protocols are most appropriate as they provide comprehensive passability scores tailored to individual species / species groups.

Importantly, a key finding when comparing the protocols was that there were differences in the threshold swimming capabilities used to predict passability. Although the swimming capabilities used are relevant to the species and lifestages considered in the region where the protocol was developed, they are unlikely to be appropriate for use elsewhere without modification. One of the key challenges in the future development of the existing protocols would be to devise swimming capability thresholds for either (1) all European species or (2) a subset of proxy species / species groups that would be appropriate for use throughout Europe. This will likely be challenging as a number of assumptions and compromises will be required to produce data applicable for all areas. In addition, accurate swimming performance data is not always available for all species / lifestages. A possible way to bypass this issue, and to factor in that barrier assessment protocols are continually being developed, is to focus on collecting applicable data on a barriers physical attributes that can then be used to assess relative passability independent of fish swimming capability data or which can feed into a project specific barrier assessment protocol selected at a later date.

The assessment protocols tend to consider barriers as either a jump, swim or depth barrier with the presence of a crawlway considered separately. An important idea that was utilised in both the SNIFFER and ICE protocols is that flow heterogeneity can be accounted for by splitting a barrier into separate transversal sections (TS), with passability assessed separately for each TS. Below is a list of the physical attributes that are considered to influence passability within the SNIFFER, ICE and/or ICF protocols:

- Type of barrier
- Depth of flow over barrier (a representative value or measured at multiple points)
- Flow velocity at barrier (a representative value or measured at multiple points)
- Total head height
- Height of any vertical drops
- Plunge pool depth below vertical drops
- Upstream water depth
- Longitudinal width of barrier crest
- Slope
- Total effective length
- Number and dimensions of steps (length and height)
- Presence of lip at crest or foot of barrier
- Effective resting locations
- Presence of standing wave
- Presence of crawl way (plus dimensions of crawl way)
- Presence of debris likely to impede passage
- Presence of structures damaging to downstream migrating fish
- Presence and estimation of level of turbulence (low, medium, high)
- Flow conditions (low, medium or high flow)
- Notch/gap dimension (i.e. at culverts or undershot weirs)
- Presence of abstraction points (location and proportion of flow diverted) and any screen (plus angle and mesh size of screen)

If these attributes are recorded at a barrier then it is likely that passability can be calculated at a later date according to project specific aims. A coarse scale passability assessment of any fish passes present is considered difficult to ascertain accurately (Baudoin *et al.,* 2014) but the following measurements can be used to indicate if they are likely to be effective or not:

- Type of fish pass and dimensions (width, length)
- Total head height
- Height of any vertical drops
- Pool dimensions
- Head difference between pools
- Water depth in pool
- Depth of water over baffles/weirs
- Dimensions of notches, orifices and slots
- Total slope
- Flow velocity (representative or at key location e.g. through orifices)
- For eel passes: Slope, length, substrate type and water depth and velocity.
- Water flowing through entrance, inside and at outlet of pass.
- Fish pass condition (e.g. poor/adequate/good)

Importantly, well annotated field sketches of the barrier and/or fish pass are required (aerial and longitudinal view points) to identify different TSs and all measurement locations. Pictures of the barrier from multiple angles should also be collected.

Currently it is unclear whether the evaluated protocols produce accurate passability scores for the range of barrier types and species / lifestages they cover. However, they are all based on sound assumptions and have been trialled by a diverse range of operators in the field. As such they represent a very useful baseline for future development. The above information on barrier assessment protocols can be used to help managers select the most appropriate protocols to use, educate users on the potential drawbacks of the existing protocols and guide future research. A logical next step would be to comprehensively validate the existing protocols (e.g. through telemetry or mark and recapture studies) and compare outputs at a broader range of barrier types.

4 ASSESSING SOCIO-ECONOMIC IMPORTANCE OF BARRIERS

The socio-economic benefits of riverine barriers should be considered in conjunction with their impact on longitudinal connectivity so that informed decisions on mitigation actions can be undertaken. Growing pressures on energy and water resources mean that riverine barriers can provide benefits that are important for meeting economic requirements and legislative goals. For example in the UK, the Renewable Energy Strategy has set a legally binding target that 15 percent of energy production comes from renewable sources by 2020, with hydropower playing a key role in delivering this target (HM Government, 2009). As such, there are competing legal pressures to both implement new hydropower facilities and mitigate for the impact of riverine barriers. Accurately assessing the current or potential socio-economic benefits that existing barriers produce is a crucial step in how barriers are managed.

4.1 Hydropower production/potential

The hydropower development potential of a barrier is difficult to fully quantify without experts undertaking comprehensive site assessments, modelling and feasibility studies (e.g. Koc *et al.,* 2016). Even for a preliminary feasibility study for a mini-hydropower development, the list of recommended variables that should be investigated is extensive (British Hydropower Association, 2012):

- The existence of a suitable waterfall or weir and a turbine site,
- A consistent flow of water at a usable head,
- The likely acceptability of diverting water to a turbine,
- Suitable site access for construction equipment,
- A nearby demand for electricity, or the prospect of a grid connection at reasonable cost,
- The social and environmental impact on the local area,
- Land ownership and/or the prospect of securing or leasing land for the scheme at a reasonable cost,
- Outline scheme layout and equipment specifications,
- An initial indication of design power, annual energy output and revenue,
- Ball-park costs for developing the scheme.

However to focus resources for hydropower development, data is required at a large number of barriers to enable prioritisation to take place. Hence a simplified method of assessing hydropower potential at pre-existing barriers is required. Generally, the available energy at a site (P) is the product of water density (ρ - kg/m^3), acceleration due to gravity (g - m/s^2), discharge (Q - m^3/s) and head difference (H - m):

$$P = \rho\, g\, Q\, H$$

However, multiple other factors contribute to the feasibility of hydropower projects and there are practical limitations to what sites can be developed (pers. comm. Dr Gerald Muller). For example:

- Sites with P less than 10 kW are very probably not economical;
- Sites with H less than 0.3 m are not economical;
- Remote sites where an electricity line would have to be added are only economical if the power generated is sufficient;
- Hydropower usually requires a part of the cross section of the river. If this affects flood alleviation (i.e. if the installation blocks the flow and if there are houses etc. upstream), then hydropower development is likely not to be appropriate.

Several projects have been undertaken to assess and prioritise the hydropower potential of existing riverine barriers, often in conjunction with other factors (e.g. conservation pressure), and two of these are outlined below.

4.1.1 England and Wales

In the England and Wales a list of potential sites for the development of hydropower has been compiled by assessing Ordnance Survey maps to locate current barriers and then assessing their power potential (EA, 2010). As per guidance given by the British Hydropower Association, the power potential was calculated by using the following formula:

$$P_t = \mu\ \rho\ g\ Q\ H$$

Where P_t is the mechanical power produced at the turbine shaft (Watts), μ is the hydraulic efficiency of the turbine, ρ is the density of water (1000 kg m^{-3}), g is the acceleration due to gravity (9.81 m s^{-2}), Q is the volume flow rate passing through the turbine ($m^3\ s^{-1}$) and H is the head (height from top to bottom of the barrier - m). Assuming 7 per cent as a typical water-to-wire efficiency for the whole system, the above equation was simplified to:

$$P_t\ (kW) = 7 \times Q\ (m^3/s) \times H\ (m)$$

As such the potential power generation at a site was assumed to be a function of the flow (P) and hydraulic head (Q) and both were allocated, where possible, for each of the 25,935 barriers identified during the preliminary mapping phases of the project (see section 3.1). Barrier head differences were assessed using LIDAR (vertical resolution ± 0.25 m) and/or SAR (vertical resolution ± 1.0 m) from upstream and downstream water elevations 5 m distant from the barrier. Flow estimates were obtained based on the most accurate data available at each site. This including data from gauging stations, flow duration statistics from around 11,000 regular ‘Outflow Points’ and/or mean flow values from a Continuous Estimation of River Flow dataset (based on catchment size). The power potential was calculated for each barrier and barriers grouped according to power potential bands most appropriate for different turbine types (Table 11).

Table 11. Power potential of riverine barriers in England and Wales

Power potential category	Maximum Power Potential	
	Number of barriers	Total Power (kW)
0 - 10 kW	15,653	48,090
10 - 20 kW	3,418	48,680
20 - 50 kW	3,384	107,127
50 - 100 kW	1,497	104,903
100 - 500 kW	1,548	324,678
500 - 1500 kW	360	294,128
> 1500 kW	75	250,219

In addition to the power potential of the structure the project also categorised the sensitivity of the barrier in relation to the potential environmental impact of hydropower development at the site. This was done by assessing modelled fish population data (Fish Classification Scheme 2 – FCS2) and the presence of Special Areas of Conservation (SAC) in three stages. The first stage of the sensitivity classification process determines if a diadromous species was highly likely to be present (greater than 0.6 probability of presence according to the FCS2 model) and/or if a relevant SAC for a diadromous fish species intersected the water body. If either or both of these criteria are met, the water body and any water bodies immediately downstream of it were classified as high sensitivity. In the second stage, all remaining water bodies were categorised according to whether there is a low, medium or high probability of presence of any 'diadromous', 'migratory' or 'mobile' species according to Table 12. The final water body sensitivity score is the sum of the mobile, migratory and diadromous species probability of presence scores (Table 13). The third stage involves promoting by one category any barriers identified in stage 2 that fall within the boundary of a SAC for any freshwater habitats or species.

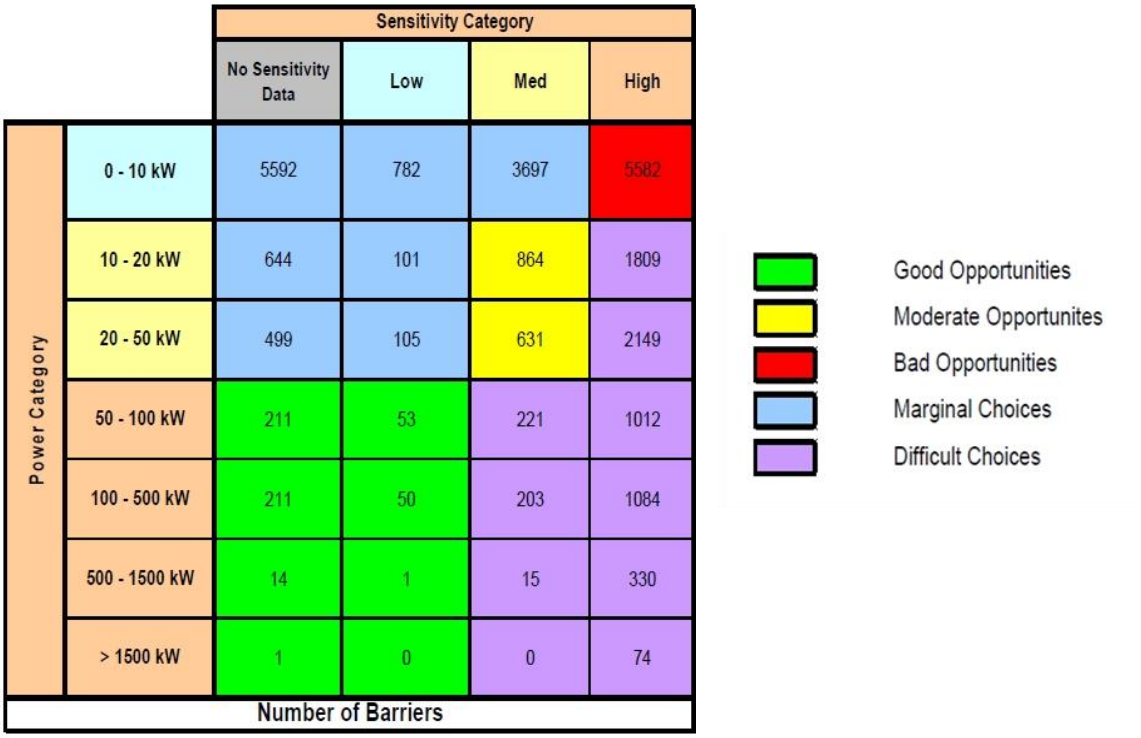
Table 12. Scores given to each water body based on the FCS2 probability of presence of species in different migratory groups.

Group	Probability of Presence	Score
Diadromous Species	High	Accounted for in first stage of assessment.
	Medium	3 pts
	Low	0 pts
Migratory Species	High	4 pts
	Medium	2 pts
	Low	0 pts
Mobile Species	High	2 pts
	Medium	1 pts
	Low	0 pts
Non-migratory species	Presence of non-migratory Species is not considered	

Table 13. Mapping of probability of presence scores to sensitivity bands

Total Scores	Sensitivity Band
6 – 9	High
3 – 5	Medium
0 – 2	Low

After sensitivity assessment, 64.42%, 21.71% and 4.21% of barriers were categorised as being in the High, Medium and Low sensitivity bands, respectively. The remaining 27.65% were categorised as 'No Sensitivity' due to a lack of data for those locations. Ultimately the relationship between power potential and environmental sensitivity was assessed categorically (Figure 4) with the best opportunities existing at locations where there is a high hydropower potential and a low sensitivity categorisation, whilst the least attractive opportunities are those with low hydropower potential and high sensitivity.



Power Category	No Sensitivity Data	Low	Med	High
0 - 10 kW	5592	782	3697	5582
10 - 20 kW	644	101	864	1809
20 - 50 kW	499	105	631	2149
50 - 100 kW	211	53	221	1012
100 - 500 kW	211	50	203	1084
500 - 1500 kW	14	1	15	330
> 1500 kW	1	0	0	74

Figure 4. Numbers of barriers identified in England and Wales according to their hydropower potential and environmental sensitivity, colour coded in relation to hydropower development potential. Adapted from EA (2010).

Only 2% of barriers were classified as good opportunities for development under this scheme, representing only 7% of the potential maximum energy generation (kW) if all opportunities were developed. Barriers classed as 'difficult choices', indicating further assessment would be necessary, accounted for 84% of the total power generation opportunities (kW). This scheme highlights the potential to simply and with relatively limited resources classify a very large number of barriers and identify a small range of initial barriers that present good opportunities for hydropower development.

The scheme further identified barriers with medium to high power potential that were located within one of the 2708 heavily modified water bodies in England and Wales (classified under the WFD) as potential 'win-win' situations. The theory being that these heavily modified water bodies require hydromorphological improvements regardless, hence, hydropower development in combination with adequate fish passage opportunity could be an efficient use of resources (i.e. a 'win-win' situation). The highlight of factoring in the influence of heavily modified water bodies was that potential "win-win" opportunities accounted for nearly half (49%) of the national total potential power.

4.1.2 Austria

In Austria a decision support tool (Hy:Con) has been developed to identify hydropower projects with the highest energy economic value combined with least conservation concerns (Seliger *et al.,* 2015). Each hydropower plants is evaluated on the basis of its energy economic criteria with regard to (1) economic attractiveness, (2) security of supply, (3) quality of supply and (4) climate protection. Concurrently, the ecologically value and sensitivity of the river stretches associated with the hydropower plant is evaluated based on 39 single criteria grouped into the categories: (1) ecological status, (2) hydro-morphological status, (3) length of free flowing river sections and migration corridors, (4) key habitats, (5) key species, (6) floodplain forests, (7) legal and (8) other designated protected areas. Different conservation scenarios are then used to assess which hydropower projects should be developed based on changing socio-economic pressures: (S1) maximal conservation, (S2) WWF energy revolution (S3), moderate conservation, (S4) minimal conservation, (S5) AWC and (S6): WWF eco-master-plan. The conservation conflict of a hydropower project is assigned by defining a conflict rating for each ecological criterion depending on the scenario. The highest conflict rating is a so-called "exclusion criteria" which indicates the presence of conservation values incompatible with HP development. Non-excluded projects are classified as very high, high, medium or low conservation value.

The decision support tool was used to classify 102 planned hydropower plants in Austria. The combined annual production of all the plants was calculated as 4,304 GWh/a with an installed capacity of 4,742 MW. Thirty five projects were rated with medium, 22 with high and only five with very high hydropower attractiveness, while the remaining 40 hydropower plants were considered as not attractive from an energy-economic point of view (rating low or moderate). In relation to the ecological value and sensitivity, the number of projects excluded varied depending on the conservation scenario, ranging from 65 hydropower projects in S1 to zero projects in S4 and S5. In almost all scenarios a high share of the analysed projects were in conflict with conservation needs at some level. This assessment methodology provides a useful overview of the likely socio-economic benefits of hydropower development and includes environmental factors that combine to make it a comprehensive assessment protocol. However, it should be noted that no final ranking protocol was devised that quantitatively combined the results of the 'energy economic' and 'ecologically value and sensitivity' assessments of for each plant. Rather, the location of each development project was graphically visualised (hydropower attractiveness against conservation needs) for each conservation scenario.

4.2 Other socio-economic benefits

The potential benefits produced by barriers, other than hydropower, are numerous (e.g. water security, navigation, flood control, food production, recreation and cultural significance) and should be considered when planning management options. Although many direct-use benefits (e.g. food production) are relatively easy to quantify from an economic point of view (e.g. they have a market value - Tagziehchi *et al.,* 2013), non-direct values (e.g. cultural significance or recreation) are much harder to quantify. Typically non-direct values are ascertained based on what people would be willing to pay for the service to exist despite no direct benefit to themselves (e.g. a non-market value - MacDonald *et al.,* 2011). In depth socio-economic cost-benefit analyses to help inform management decisions for riverine infrastructure are primarily only undertaken for larger dam removal projects (e.g. Kruse and Scholz, 2006) but the principles if applied to other mitigation options and smaller barriers are applicable. Kruse and Scholz (2006) incorporated estimates of the economic impact of dam removal on the value of local jobs, property and fish stocks and the non-use values of species conservation and restoring a free-flowing river, cultural and tribal values (e.g. return of traditional fishing grounds and increased salmon harvests for ceremonial, subsistence and commercial use) and other recreational activities as part of a preliminary cost-benefit analysis of removing 4 dams in Oregon, USA. The study outlines the complicated ways in which dams and dam removal can produce a range of social-economic benefits/costs and how these can be quantified.

Although these factors should play a key role in decision making, implementing such extensive socio-economic assessments at all barriers within Europe will not be feasible. To help focus mitigation actions a coarse scale method for assessing the socio-economic benefits/costs of barriers is required. To these authors knowledge no such assessment methodology currently exists. At a course scale, the benefit/cost of a barrier is related to how it is used. As such, recording simple information on how the barrier is used will help infer whether the barrier has associated socio-economic benefits/costs and will help direct future investigations. For example: 1) the barriers original intended purpose, 2) is it still being utilised as intended, 3) is there evidence that the barrier is currently being utilised for any other purpose or has alternative non-direct significance, and 4) a list of the other purposes or non-direct significances

5 PRIORITISATION OF MITIGATION EFFORTS

To efficiently improve longitudinal connectivity, mitigation measures should be assessed and implemented on a catchment scale (Kroes *et al.* 2006; Pinsky *et al.* 2009). However, limited resources often mean that mitigation measures must be carefully applied to maximise the benefits that are obtained (Kemp and O'Hanley, 2010). To help direct barrier mitigation efforts, a variety of prioritisation methodologies have been formulated (e.g. Paulsen and Wernstedt, 1995; Kuby *et al.,* 2005; O'Hanley and Tomberlin, 2005; Cote *et al.,* 2009; Kocovsky *et al.,* 2009; Zheng *et al.,* 2009). These were comprehensively reviewed by Kemp and O'Hanley (2010) and the field has continued to grow since then (e.g. Diebel *et al.,* 2010; 2015; Bourne *et al.,* 2011; O'Hanley, 2011; Nunn and Cowx, 2012; Anderson *et al.,* 2012; O'Hanley *et al.,* 2013; Branco *et al.,* 2014; King *et al.,* 2016b).

The multitude of different prioritisation models currently available differ significantly from one another in a number of ways including: (1) how they consider the issue of connectivity, (2) how barrier passability is included, (3) the species considered, (4) the parameters included in the model and (5) how the prioritisation process is undertaken. For any given prioritisation model, habitat connectivity is generally considered in reference to the life history of the focal species. For example many early prioritisation systems focussed exclusively on diadromous fish species and hence their indices were heavily biased towards connectivity between stream reaches and the sea (e.g. Nunn and Cowx, 2012). Later models have tended to focus on connectivity for potadromous (or resident) species and, as such, studies have focused on maximising connectivity between different areas within a river network, regardless of their proximity/connectivity to the Sea (e.g. O'Hanley *et al.,* 2013; Branco *et al.,* 2014; Diebel *et al.,* 2015, King *et al.,* 2016b). A key study in the development of these separate indices was that undertaken by Cote *et al.* (2009) who presented the Dendritic Connectivity Index (DCI) for both diadromous and potadromous species (DCI_D and DCI_P, respectively), an index that has been used and improved upon in a number of more recent studies (e.g. Diebel *et al.,* 2015).

To accurately prioritise barriers for mitigation, appropriate parameters that predict the likely benefits and costs associated with mitigation must be utilised (Kemp and O'Hanley, 2010). Although longitudinal connectivity has been shown to significantly influence the presence or absence of a number of migratory and resident species in river systems, environmental variables generally have a larger impact (Mahlum *et al.,* 2014). For example, the presence and abundance of target species will be influenced by availability, quantity and quality of key habitats, local river hydrology and geomorphology (e.g. stream order and/or slope) and by the wider aquatic and riparian ecosystem (e.g. resource competition). Whilst socio-economic factors, such as the cost of mitigation actions or whether the barrier is currently or could potentially provide other benefits such as recreation, sport fishing, hydropower, water supply, and flood control, will heavily influence the likelihood of actions being undertaken. Prioritisation models that fail to consider key confounding variables are likely to focus mitigation resources inappropriately. However, barrier prioritisation models, as with other environmental management decision tools, are inherently limited by the availability and accuracy of data and a trade-off between functionality and complexity (i.e. overly complex models tend to be difficult to interpret – Van Nes and Scheffer, 2005). Commonly incorporated model parameters include the type, quantity and quality of habitat, distance between habitats, existing fish stocks, direct costs (e.g. in relation to construction work to remove a structure or build a fish pass) and associated economic costs (e.g. hydropower generation, water storage capacity and harvesting by fishermen) (Kemp and O'Hanley, 2010).

Barrier passability is often considered as binary within prioritisation models (e.g. Kuby *et al.,* 2005; Branco *et al.,* 2014), likely for simplicity but also due to lack of available passability data. However Anderson *et al.* (2012) in a study to investigate the sensitivity of prioritisation models to differing barrier passability scores identified that probabilistic barrier scores (e.g. 0-1) are more appropriate than binary scores as it is important to incorporate uncertainty into criteria underlying remediation decisions. Such probabilistic scores are directly produced for multiple species using course-scale rapid assessment protocols such as the SNIFFER or ICE methodologies discussed previously (Sections 4.1.1 and 4.1.2, respectively) and have been utilised in more complex prioritisation models (e.g. King *et al.,* 2016b). However, the lack of available passability data for the majority of barriers can reduce the applicability of using probabilistic scores to inform prioritisation models in large systems. A number of models have overcome this issue by estimating passability scores at barriers based on either regression analysis in relation to environmental factors (Januchowski-Hartley *et al.,* 2014) or extrapolation from a limited amount of field data (King *et al.,* 2016b). These estimations present a powerful way to prioritise barrier removal in the absence of comprehensive field data on barrier passability. However, the accuracy of such estimation tools has yet to be verified.

In some models, connectivity is also scaled by the distance between habitat patches to promote prioritisation actions in regions where key habitat patches are closer together and also to realistically limit the importance of habitat patches that are beyond the migration range of target species (e.g. Diebel *et al.,* 2015; King *et al.,* 2016b). For example, Diebel *et al.* (2015) used a distance weighting function to scale the accessibility of nearby habitats toward 1 and distant habitats toward 0. A value of 20 km was selected as the distance at which the weight equals 0.5, which approximates to the typical spatial autocorrelation function of the distributions of several fish species in a comparable study area (Diebel *et al.,* 2010). When applied in a prioritisation model, this weighting function should be altered to represent the frequency distribution of seasonal movements in an unimpeded stream network of the focal species (Diebel *et al.,* 2015). Similar distance weighting functions are utilised in prioritisation models focusing on anadromous species with connectivity scaled according to the distance from the tidal limit (e.g. Nunn and Cowx, 2012).

The process by which barriers are prioritised can also heavily influence the model efficacy (Kemp and O'Hanley*,* 2010). In general models can be grouped according to whether they prioritise mitigation actions based on a scoring and ranking type system (e.g. Nunn and Cowx, 2012), greedy type heuristic selection (e.g. Diebel *et al.,* 2010; 2015), complete enumeration (e.g. Cote *et al.,* 2009) or formal optimisation based models (e.g. mixed integer linear programs) (e.g. O'Hanley et al., 2013; King et al., 2016b). Scoring and ranking systems are considered too simplistic as barriers are treated independently, which can lead to a highly inefficient set of barriers being selected for mitigation (O'Hanley & Tomberlin 2005). At the other extreme, although guaranteed to provide optimal solutions, complete enumeration is considered extremely limited and only appropriate for solving relatively small problems with a handful of barriers (Kemp and O'Hanley, 2010; Mackey *et al.,* 2016). With the greedy type heuristic approach, mitigation actions are not evaluated independently nor are barrier rankings generated in a static fashion. Instead, rankings are derived by (1) calculating benefit-cost ratios for each barrier while taking into account passabilities at other barriers, (2) mitigating the barrier with the highest ratio and then (3) repeating steps (1) and (2) for all remaining barriers until the budget has been exhausted (O'Hanley *et al.* 2013). A key advantage of greedy type heuristics, in comparison to simple scoring and- ranking methods, is that they can often produce optimal to near optimal solutions very quickly (O'Hanley and Tomberlin, 2005). Optimisation models also produce exact optimal solutions like complete enumeration, but do so in a much more efficient manner. For example, King *et al.,* (2016b) present a complex optimisation model, incorporating a number of parameters to estimate species richness as a function of the connectivity status of a river. They overcome the problem non-linearity (which makes models notoriously difficult to solve) by approximating the relationship between species richness and connectivity as a piece wise linear curve. The final model produces near optimal solutions very quickly. These solutions, in turn, can be implemented *in toto* or be used as the starting point for further fine-tuning and analysis.

Despite the varied impact that artificial structures can have on different fish, few methodologies account for multiple species (exception: King *et al.,* 2016b: Neeson *et al.,* 2015). Most prioritisation models are tested for a focal species or a proxy species that is assumed to be an appropriate representative for the whole community (e.g. King and O'Hanley, 2016). Indeed, there is some evidence to suggest that although passability scores may have some effect on connectivity indices, they may have little effect on final restoration priorities (Bourne *et al.,* 2011). Such findings may have important implications for how passability scores are ascertained and whether relative passability scores, calculated independently of a specific species swimming capabilities, may be adequate to provide near optimal restoration priorities.

Importantly, when prioritising mitigation actions, the passability of multiple barriers in a river network is usually assumed to be independent (e.g. Dumont, 2005; O'Hanley and Tomberlin, 2005; Cote *et al.,* 2009), with connectivity between habitat patches generally calculated as the product of individual passability scores. For example if barriers A, B and C have passability scores of 0.3, 0.6 and 1, respectively the passability of the three barriers combined (P_{ABC}) is considered to be 0.18 (P_{ABC} = A*B*C). This assumes, however, that the likelihood of a fish passing one barrier does not affect the probability of the same fish passing any successive barriers. This is unlikely as the subset of fish that pass a barrier tend to be those which are more able to overcome physical barriers, such as vertical drops and high flow velocities (e.g. larger fish with better swimming ability) and those which exhibit behaviours that lend themselves to locating and moving past artificial barriers. As such, passability scores assigned to later barriers encountered on a migration path may underestimate the likelihood of fish passing them (Kemp and O'Hanley, 2010).

5.1 Prioritisation examples

Three detailed examples of prioritisation models tested on real systems are described below: A relatively simple scoring and ranking system for the entire Danube River Basin

(Mielach *et al.,* 2012), a complex optimisation model for the River Wey catchment, Southeast England (King *et al.,* 2016b) and a second optimisation model focused on small scale hydropower development in England and Wales which factors in habitat connectivity (Ioannidou and O'Hanley, 2016).

5.1.1 Danube River Basin

As part of the EU SEE HYDROPOWER project, a scoring and ranking prioritisation index was developed and trialled for the entire Danube River Basin (Mielach *et al.,* 2012). The methodology prioritises barriers for removal by assessing the modelled distributions or historic occurrences of fish species grouped by migration type (long or medium distance migrants or resident species). Rivers capable of supporting or being used by long distance migrants (LDM) were derived from historic records. The potential locations of medium distance migrants (MDM) were derived based on a model formulated from occurrence data documented across Europe as part of the EU-project EFI+ database. The prioritisation principle follows the idea that LDM within the Danube receive the highest priority (weight 4) followed by LDM within tributaries to the Danube (weight 2). MDM receive even less priority (weight 1) and head waters are excluded from the prioritisation process (weight 0). Within this prioritisation framework, barriers near the mouth of a river receive higher priority than upstream barriers; similarly, barriers on the Danube receive higher priority than those on tributaries. To give higher weight to river segments that are less fragmented by continuity interruptions, the length of the reconnected habitat is weighted depending on the length of river segments. In addition, the final selection criterion is related to the protection status of river areas, with barriers within protected areas of the NATURA2000 network receiving higher priority, as it is more likely that those river segments should be maintained in good habitat status and will be restored to greater extent than unprotected river segments. The criteria are combined by computing an overall prioritisation index (PI) by weighting the first criterion, migratory habitat, by the cumulated weight of the 4 other criteria as follows:

$$\text{PI} = \text{migratory habitat} \times (1 + \text{first barriers upstream} + \text{distance from mouth} + \text{reconnected habitat} + \text{protected site})$$

The maximum possible value of the PI is 36 and the minimum is 0 (only in head waters). Finally, the PI was grouped into 5 classes: top priority (PI >13), very high priority (PI 10- 12), high priority (PI 7-9), medium priority (4-6) and low priority (PI 1-3).

In total, 946 barriers were considered. More than a quarter (27%) were not a priority (PI=0) because they are located in headwaters or canals. Out of the 681 remaining barriers, 39 barriers (4%) had a high to top priority, 99 barriers (10%) were of medium priority and 543 barriers were of low priority (58%). The results revealed ecological priorities for continuity restoration within the Danube River Basin and represent one of the largest direct applications of a prioritisation model. Importantly though, a scoring and ranking system was utilised which treats barriers independently and has been shown to produce highly inefficient solutions (O'Hanley and Tomberlin, 2005). In addition, barriers were treated somewhat simplistically as having a binary passability score, which might further limit the accuracy of the recommendations produced. The authors acknowledge that the proposed prioritisation should be used as a guideline and final mitigation decisions would need to be further investigated before resources were dedicated to any one barrier (Mielach *et al.,* 2012). This is primarily because the prioritisation system does not take into account factors such as the technical feasibility and/or cost of implementing mitigation at each barrier (e.g. building a fish passes or removing the barriers) (Mielach *et al.,* 2012). The authors also noted that, as the Danube river basin crosses multiple countries, mitigation actions would likely be heavily influenced by the national restoration and conservation programmes which the barriers come under.

5.1.2 River Wey Catchment, UK.

The model presented by King *et al.* (2016) is designed to deal with within-river movements of resident species. It factors in barrier passability scores derived from real SNIFFER data, habitat type, amount of habitat available, a distance decay factor (habitat that is further away is considered less desirable than habitat nearby) and cost of mitigation actions. It was tested on the River Wey catchment, Southeast England, a system that comprises of two main tributaries and covers approximately 900 km^2. Cumulative barrier passage is derived by multiplying the individual barrier passability scores together. The aim of the model is to select barriers for repair or removal to maximise mean resident fish species richness (R) within the study area. The authors assume that R within a river area is determined, at least in part, by its connectivity status (C). R (the dependent variable) is estimated based on C (the independent variable) using a generalised Poisson regression model. They overcome the non-linearity introduced by using a Poisson regression model by approximating the relationship between R and C as a piece-wise linear curve. They use species absence as a proxy for species richness in the final model, as this produced a better fit with the connectivity index.

Barriers to fish movement within the catchment (n = 669) were identified using three existing databases and some catchment walkovers. Ninety three structures were directly considered as impassable (PS = 0) due to exceeding various thresholds (e.g. head height greater than 1 m) and 35 navigation locks were assigned a PS of 0.3 due to insufficient knowledge on passability through these structures. The remaining 478 barriers were assigned upstream/downstream passability scores according to the median values for each structure type derived from the results of SNIFFER rapid barrier assessments undertaken within the same catchment (n = 63). The costs of barrier mitigation were estimated on the basis of costs for work at similar structures and from information provided in Armstrong *et al.* (2010). The cost of mitigating all 669 candidate barriers within the River Wey was estimated to be £53,355,000.

The results indicated an overall pattern of diminishing returns, whereby increases in species richness become progressively smaller with increased budget. Given a budget of just £5M, for example, mean richness can increase by roughly 50% (2.3 species) above the baseline value. To achieve nearly a doubling in species richness, however, required a fourfold increase in the budget (i.e. £20M for an increase of 5.0 in species richness). Initial gains in species richness were primarily seen first in the upper reaches, followed by gains in the middle to lower sections of the river catchment. Barriers targeted for mitigation at lower budget levels (≤£25M) tended to be large (≥1m head height), have lower than average initial passability and were generally more costly to mitigate compared to barriers as a whole.

The model provides a useful tool for prioritising mitigation but also for investigating the dynamics of resource distribution. Interestingly despite almost equal proportion of culverts and weirs in the catchment, the model targeted weirs for mitigation much more often at lower budgets (≤£25M). In addition, for budgets ≤£25M improvements were predominantly seen in areas with high levels of bifurcation.

Key advantages of the model are: 1) The model does not treat barriers independently, 2) key environmental variables are incorporated for predicting results of barrier removal, 3) near optimal solutions are produced, and 4) it adopts a multi-species focus by maximising overall species richness. As such, it is fairly comprehensive and versatile. The model is noteworthy for integrating statistical methods to maximise gains in mean species richness across a catchment. In this regard, it provides a simplified way of focusing on an ecologically relevant goal (species richness) without the need to integrate data hungry and computationally intensive population/ecosystem simulation models (King *et al.,* 2016b).

5.1.3 England and Wales

Ioannidou and O'Hanley (2016) present a prioritisation model not for the removal or repair of barriers but for the eco-friendly location of small hydropower plants (SHPs). Their model assesses the potential impact of hydropower development on longitudinal connectivity for diadromous species and includes indicators such as potentially available habitat, passability of the existing barriers and new hydropower facilities and the impact of "backwater effects" (a raising of the water surface profile due to the presence of in-stream structures) on hydropower potential and barrier passability. The model ensures that a minimum amount of accessible habitat following siting of dams is maintained and maximises the total hydropower produced across all SHPs. The model is relevant for manging existing barriers as it factors in cases through which development of hydropower at an existing barrier improves the passability of the barrier through appropriate mitigation (e.g. construction of a fish pass), resulting in a "win-win" situation in which both increased river connectivity and hydropower production are achieved. The model has the potential to be a very powerful planning tool for balancing the ever increasing need for renewable energy while limiting or even reducing the connectivity impairment of rivers.

The model was tested for the whole of England and Wales. The River Barriers database (see section 3.1) was used to obtain the location, type and head height of 25,935 existing riverine barriers (EA, 2010). Discharge at each site was calculated through a regression model that predicted mean flow based on mean annual precipitation within the upstream catchment area. All existing barriers were assigned upstream passability scores based on a simplified relationship between head height (H) and passability for brown trout extracted from the SNIFFER protocol based on the following rule base:

$$Passability\ Score = \begin{cases} 1 & if\ \mathrm{H} \leq\ 0.6\ \mathrm{m} \\ 0.6 & \text{if } 0.4\ \mathrm{m}\ <\ \mathrm{H}\ \leq\ 0.6\ \mathrm{m} \\ 0.3 & \text{if } 0.6\ \mathrm{m}\ <\ \mathrm{H}\ \leq 1\ \mathrm{m} \\ 0 & if\ \mathrm{H} > 1\ \mathrm{m} \end{cases}$$

Downstream passability for existing barriers was assumed to be 1. Within the model, the development of hydropower at a site and concurrent installation of a fish pass was assumed to increase upstream passability to 0.5.

Interestingly, due to the very large number of barriers present on rivers in England and Wales, many of which are believed to be impassable, the model predicted that up to 14,607 SHPs could be installed with a hydropower potential of 691.9 megawatts (MW) while at the same time increasing accessible habitat by 229%. When the model was adjusted to select only sites that produce 5 kilowatts (KW) or greater (typically termed “pico” scale plants and assumed to be the smallest output economically feasible) the model predicted 7672 SHPs could be installed, resulting in a maximum hydropower potential of 681.9 MW and a 177% increase in accessible habitat. Further analysis revealed that only a small subset of the candidate sites (n = 100, 0.6% of the total possible) was required to produce almost 25% of the total capacity (174.4 MW). Importantly, if the location of those 100 candidate sites was optimised, a 100% increase in accessible habitat could be produced while still producing a potential 154.7 MW. The results emphasise the potential ‘win-win’ situation surrounding small hydropower development. Interestingly the model highlights that small (< 5m head height) dams and weirs are the preferred choice for SHP placement.

5.2 Conclusions

It is vital that prioritization methods, if they are to be applied in the real world, are capable of producing cost-effective solutions using easy-to-obtain data. Ideally, they should also be fairly easy to implement, computationally efficient, and flexible in meeting different planning goals (King *et al.,* 2016b). Importantly, identifying all potential barriers in a system is imperative to accurately assess connectivity (Cote *et al.* 2009; Januchowski-Hartley *et al.* 2013; O’Hanley 2011; Branco *et al.,* 2014). As such, significant effort should be allocated to locating all barriers within a system before prioritisation is undertaken. In addition, investigation into the cumulative impact of barriers on habitat connectivity needs to be undertaken, as currently prioritisation models consider barrier passability to be independent but this is unlikely to be realistic.

It is believed that optimisation models (e.g. King *et al.,* 2016b) or methods that utilise greedy type heuristic selection (e.g. Branco *et al.,* 2014) present the best option for prioritising mitigation actions in large complex systems. Prioritisation models must also take into account sensible predictor variables which present an opportunity to markedly improve the cost-benefit return of mitigation actions and improve resource prioritisation (Branco *et al.,* 2014). Commonly incorporated variables include type, quantity, quality of habitat, distance between habitats, existing fish stocks and direct financial (e.g. in relation to construction work to remove a structure or build a fish pass) and economic costs (e.g. hydropower generation, water storage capacity and harvesting by fishermen) (Kemp and

O'Hanley, 2010). The selection of which predictor variables to incorporate will likely be influenced by project specific aims and data availability. In addition, the selection of either a diadromous or potadromous connectivity indices will depend on the focal species of the project. If connectivity for the whole fish community is required, connectivity indices for different migration types (e.g. diadromous and potadromous) will have to be utilised and recommended mitigation actions produced based on combining these indices.

At present, no single prioritisation model meets the needs of all projects but there is a diverse range of models currently available. It is likely that through appropriate model selection and modification, options are available to suit most prioritisation needs.

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D.1.1 Guidance on Stream Barrier Surveying and Reporting.

Part B: Towards a Pan-European ATLAS on Stream Barriers

This is the 1.0 version of Guidance on Stream Barrier Surveying and Reporting. Part B: Towards a Pan-European ATLAS on Stream Barriers. This document is a deliverable of the AMBER project, which has received funding from the European Union's Horizon 2020 Programme for under Grant Agreement (GA) #689682.

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Preamble

Humans have been modifying river systems for millennia (Goudie, 2013). Reasons for modification include flood prevention, irrigation, power generation, navigation, gauging and to provide a reliable source of water (Goudie, 2013). Modification usually takes the form of damming (including weirs and barrages), channelization or water abstraction (often in vast quantities for power plant cooling) (Goudie, 2013). The rate of modification has increased dramatically in recent years and it is now estimated that over half of the world's large river systems are fragmented (Nilsson *et al.,.,* 2005). This includes over 45,000 dams greater than 15 m high (World Commission on Dams, 2000) and orders of magnitude more smaller obstructions. Rivers now rank among some of the most threatened ecosystems in the world (Dudgeon *et al.,.* 2006), and are the focus of restoration programmes which cost taxpayers billions (Palmer *et al.,.* 2005). Much of Europe depends on water from rivers for drinking, food production and the generation of hydropower, which is essential for meeting the European Union (EU) renewable energy target. Yet only half the EU surface waters have met the Water Framework Directive's (WFD) 2015 target of good ecological status, due in part to the fragmentation of habitats caused by tens of thousands of dams and weirs. Improving stream connectivity has been flagged as one of the priorities for more efficient stream restoration but effective rehabilitation of ecosystem functioning in European rivers needs to take the complexity and trade-offs imposed by barriers into account. However, strikingly, the location of the majority of barriers on European river systems is not known, there is no central inventory of existing barriers and methods to quantify the impact of barriers on stream connectivity are in their infancy.

This report is part of a deliverable that provides Guidance on Stream Barrier Surveying and Reporting (D1.1) to aid users within AMBER and elsewhere in decision making and for the development of Adaptive Barrier Management. It is split into two sections:

- **Part A**: Locating, Surveying and Prioritising Mitigation Actions for Stream Barriers.
- **Part B**: Towards a Pan-European ATLAS on Stream Barriers

Part A focusses on existing methodologies for surveying stream barriers. It includes sections on locating barriers, methods for assessing a barriers potential to influence longitudinal connectivity and provide socio-economic benefits (e.g. hydropower) and methods for prioritising mitigation actions.
Part B focusses on evaluating the current state of existing barrier inventories throughout Europe and provides a road-map for the development of a pan-European ATLAS on stream barriers. This is **Part B** of the deliverable.

Executive summary

This is the 1.0 version of Guidance on Stream Barrier Surveying and Reporting. Part B: Towards a Pan-European ATLAS on Stream Barriers. This document is a deliverable of the AMBER project, which has received funding from the European Union's Horizon 2020 Programme for under Grant Agreement (GA) #689682.

The state of river fragmentation of European rivers is largely unknown. Additionally, an exhaustive ATLAS of stream barriers at pan-European scale currently does not exist despite the critical impacts of barriers on freshwater ecosystems and ecosystem services associated with their uses. From an informal survey based on 38 European countries, it has emerged that databases exist at national and regional levels. However, their consistency in terms of typology of mapped barriers and list of variables stored vary significantly. Some countries have a national inventory that in most cases concerns only major dams, few have an exhaustive mapping also of minor barriers, and other countries have no information at national level.

One of the AMBER project aims is to create the first pan-European ATLAS of river barriers that impact river connectivity. The ATLAS aims to establish a common framework for barrier mapping, data collection and storage for any type of barrier that is likely to have an impact on river ecosystem connectivity (including water, sediments and organisms), and to support barrier reporting in a consistent and homogeneous way throughout Europe. This will be achieved through an extensive exercise of existing database compilation and a critical analysis of the compiled data. This analysis will include a robust validation process that combines statistical approaches, field-survey, remote sensed data and the use of a smartphone APP developed within the project. The ATLAS will provide a wide picture of the state of river fragmentation at pan-EU scale together with an overall picture on data accessibility across Europe.

Annex A of this document reports the guidelines for data compilation of existing national (or regional/provincial) databases.

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1 INTRODUCTION

An exhaustive ATLAS of stream barriers at pan-European scale currently does not exist despite an established recognition of their critical impacts on freshwater ecosystems and societal values of key ecosystem services associated with their uses, e.g. hydropower production, flood protection and water supply (Petts and Gurnell 2005; McCartney 2009; Yin *et al.,.,* 2014). In Europe various stream barrier databases exist at different geographical levels such as national, regional and provincial. However, their consistency varies largely in terms of type of stream barriers surveyed, which can range from major dams (with a storage > 0.1 km^3) to minor barriers (height < 5 m) such as run of the river power plants with no storage, weirs and culverts. For some geographical areas multiple independent databases exist for different typologies of barriers. Moreover, variables and information stored for each barrier vary significantly in between countries and regions in Europe. Aiming at building a pan-European ATLAS of stream barriers, we need to cope with these data gaps, find an effective way to integrate existing datasets and develop an open-ended user-friendly platform for future data maintenance and update. To this aim, this report first compares some publically available information on stream barriers in Europe at intercontinental, national and regional scales; and secondly it presents the results of an informal survey conducted amongst available contacts in various European Member States to describe the state of monitored barriers (not necessary publicly accessible). The intent is to provide the basis for discussing quality of available information in order to design suitable and feasible objectives for the ATLAS, and finally to provide a road map to its development.

1.1 Existing data availability in Europe about Dams and Stream barriers: continental, national and regional scales

The most up to date and peer-reviewed database of large reservoirs existing at world-wide scale is named the Global Reservoir and Dam (GRanD) database (Lehner *et al.,* 2011). The dataset has been developed by the Global Water System Project (GWSP), a joint project of the Earth System Science Partnership (ESSP), which initiated an international effort to collate the existing dam and reservoir data sets with the aim of providing a single, geographically explicit and reliable database for the scientific community. Datasets and documentation are freely available at http://www.gwsp.org/products/grand-database.html. Although the main focus was to include all reservoirs with a storage capacity of more than 0.1 km^3, many smaller reservoirs were added if data were available. The GRanD database include 3,793 barriers for Europe provided by the European Environment Agency (EEA). This number is up to date to the year 2007, and the EEA is currently working to release a new version of this lake and barriers database as a component of the EEA's river and catchment GIS (ECRINS[1]).

We compare this existing information at pan-European scale with some national and regional databases on stream barriers publically available. First, we compare the density of barriers amongst different source of information. We calculated the number of other barriers present approximately every 300 km^2 around each barrier using the function

[1] http://www.eea.europa.eu/data-and-maps/data/european-catchments-and-rivers-network#tab-gis-data

'heatmap' of QGIS. Figure 1, 2 and 3 show the results using the GRanD database, the French national database (available at: link 1. Also see: link 2) the Swedish national inventory (available at: link 1. Also see: link 2), and two regional databases in Spain and one in Italy. The regional databases have been provided from specific water authorities and not all of the original metadata archived is publically available.

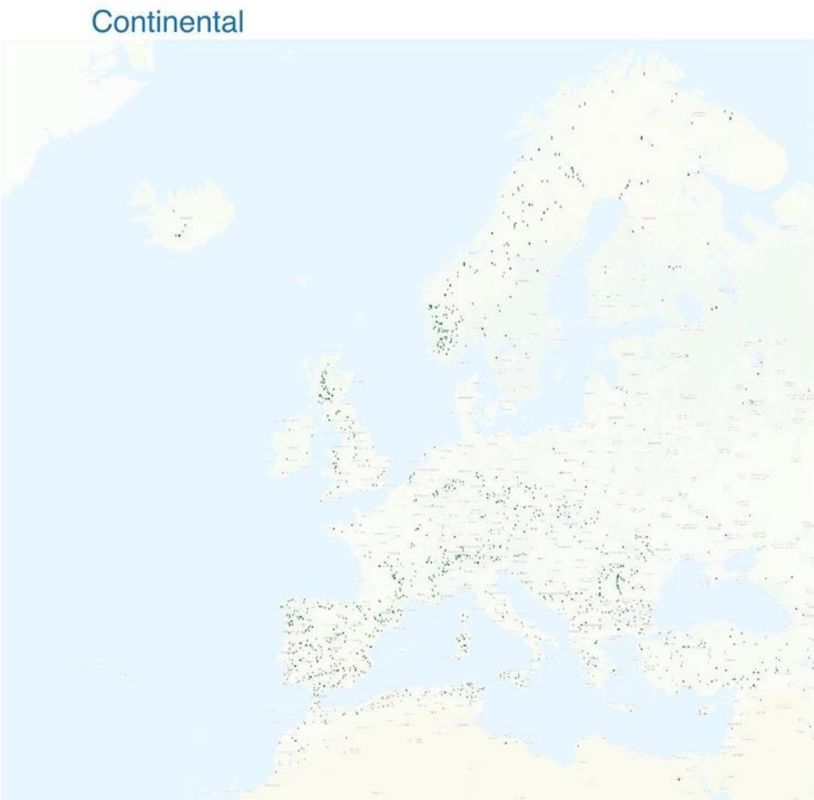

Figure 1. Barriers density for the GRanD database

At the continental level, the Grand database has mostly a density of 1 at the location of every barrier and zero elsewhere. This means that the GRanD database has a barrier density lower than 1 every 300 km^2 (Figure 1). The pictures emerging from the national and regional databases are drastically different (Figures 2 and 3). The density of dams are significantly higher reaching peaks of 40-50 units every 300 km^2. Moreover, the density is positive everywhere in the two countries (France and Sweden) and in the regions analysed. This indicates that in these areas an unimpeded basin of larger than 300 km^2 no longer exists.

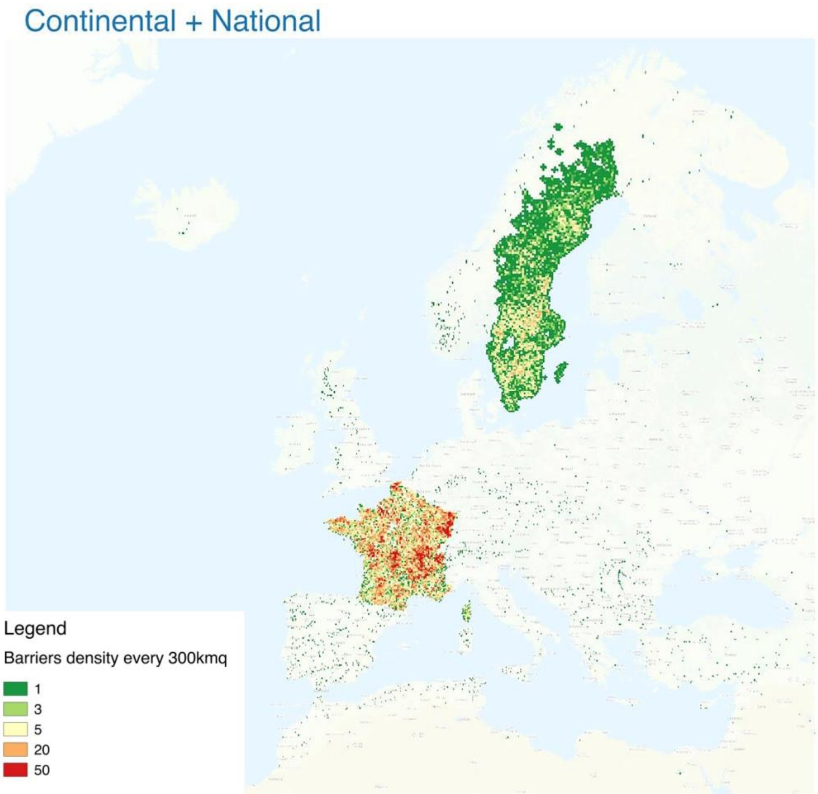

Figure 2. Barrier density merging together the GRanD database and the national databases of France and Sweden

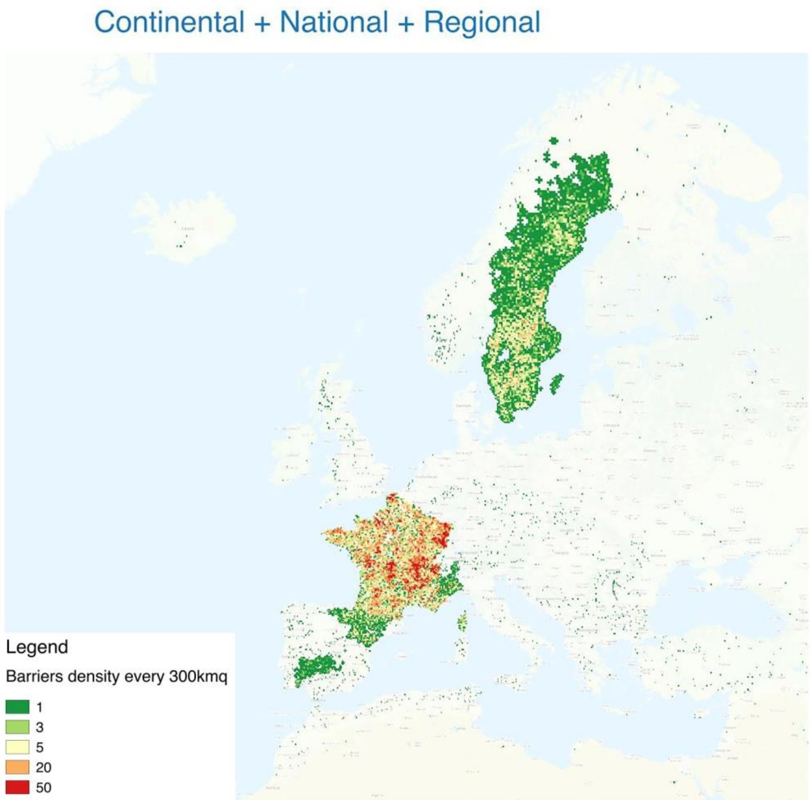

Figure 3. Barrier density merging together the GRanD database, the national databases of France and Sweden, and the regional databases from Spain and Italy

Figure 4 shows the distribution of barrier heights for the GRanD, French, Swedish and US[2] databases. It clearly emerges that the GRanD database neglects most of the barriers lower than 10 m, which is the size of the majority of structures impacting river connectivity documented in the French, Swedish and US databases. For this reason, the real number of existing barriers is orders of magnitude higher than what is reported in the GRanD database.

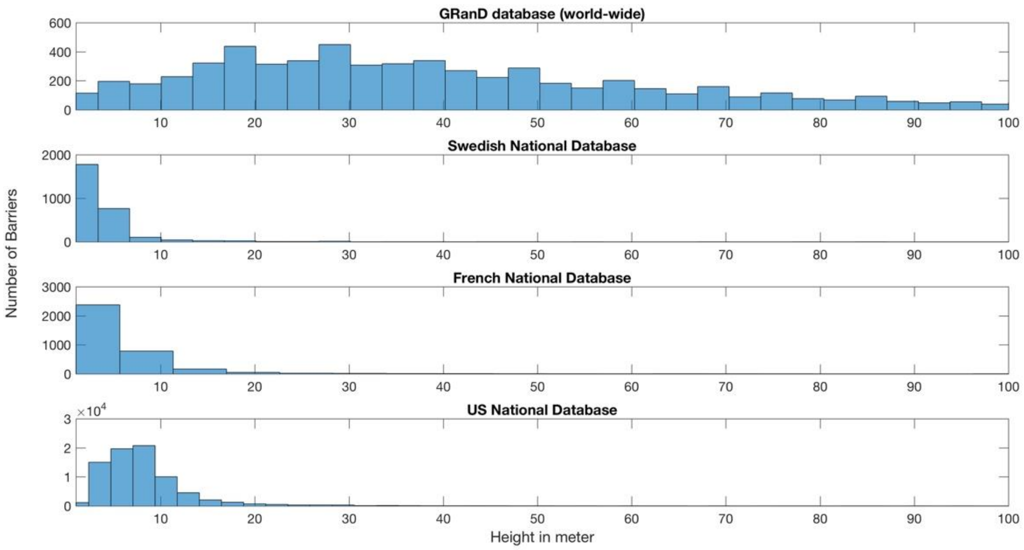

Figure 4. Histogram of stream barriers height lower than 100 m for the GRanD database, the French, Swedish and US national databases.

These results clearly show that the real degree of river fragmentation around the world is largely unknown. Previous studies, which assess fragmentation indexes of freshwater systems world-wide are likely to be significantly underestimating the real scale of the problem (Vörösmarty *et al.,* 2010). Focusing on Europe, it has to be noted that the more exhaustive national databases such as the French and Swedish ones are very rare (almost unique in Europe). Most of the European countries have centralized database only for major dams (e.g. Spain and Italy) and rely, when existing, on regional if not provincial databases for smaller barriers. Moreover, data on culverts, which are well known barriers to ecosystem connectivity, are often completely absent. For example, in the UK a focussed effort to identify barriers in the River Wey catchment using previously unevaluated databases and a small amount of fieldwork identified 565 additional barriers not included in the national inventory (Eakins *et al.,* 2012). As such, the UK's most up to date inventory for riverine barriers could possibly include fewer than 30% of potential obstructions to the free movement of aquatic organisms.

It is not only the location but also the parameters stored for each barrier that significantly varies between national or regional databases and between Member States. Table 1 reports a synthesis of some key variables archived for each barrier for the databases analysed so far.

[2] National Inventory of Dams, publically available at http://nid.usace.army.mil

Most of the databases include basic information related to location (e.g. basin and river names) and design parameters such as stored volume, barriers= height and age of construction. However, fewer have information on dam material, obstacle condition and installed facilities, for restoration purposes. Moreover, the list of relevant variables required for a barrier assessment would be significantly longer including information on hydrology and channel morphology upstream and downstream the barrier. These variables may exist for specific case studies where these assessments have been carried out. However, this information is very rare and often it is not embedded into existing databases, and of limited accessibility. To design the framework of the first ATLAS on European stream barriers we need then to take these evidences derived by publicly accessible data into carefully consideration.

Table 1. Selected list of stored variables for existing continental, national and regional databases on stream barriers

	World	National				Regional	
	GRanD	**Sweden**	**France**	**Switz.**	**US**	**Guadiana Basin**	**Ebro Basin**
Latitude/Longitude	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Name of obstacle	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Name of River Basin	Yes	No	Yes	No	Yes	Yes	No
Name of river	Yes	No	Yes	No	Yes	Yes	Yes
Height	Yes	Yes	Yes	Yes	Yes	Yes	Yes
Width	Yes	Yes	No	Yes	Yes	Yes	Yes
Storage Volume	Yes	Yes	No	Yes	Yes	No	No
Date built	Yes	Yes	No	Yes	Yes	No	No
Type of obstacle	No	Yes	Yes	Yes	Yes	Yes	No
Origin of obstacle (natural / artificial)	Yes	Yes	No	Yes	No	Yes	No
Construction type/material	No	Yes	No	No	Yes	Yes	No
Obstacle condition (good, bad)	No	No	No	No	Yes	Yes	No
Use (hydropower, flood mitigation etc.)	Yes	No	Yes	?	Yes	No	No
Fish passage/other mitigation	No	Yes	Yes	No	No	Yes	Yes

1.2 Survey on existing data availability

In order to develop a preliminary understanding of current data availability on European stream barriers, which are not necessarily publicly available, we have started an informal survey to contact 38 countries, i.e. all EEA countries plus some others relevant ones, which belong to the European continent such as Switzerland, Iceland and Norway. A complete list is reported in Table 2 and Figure 5.

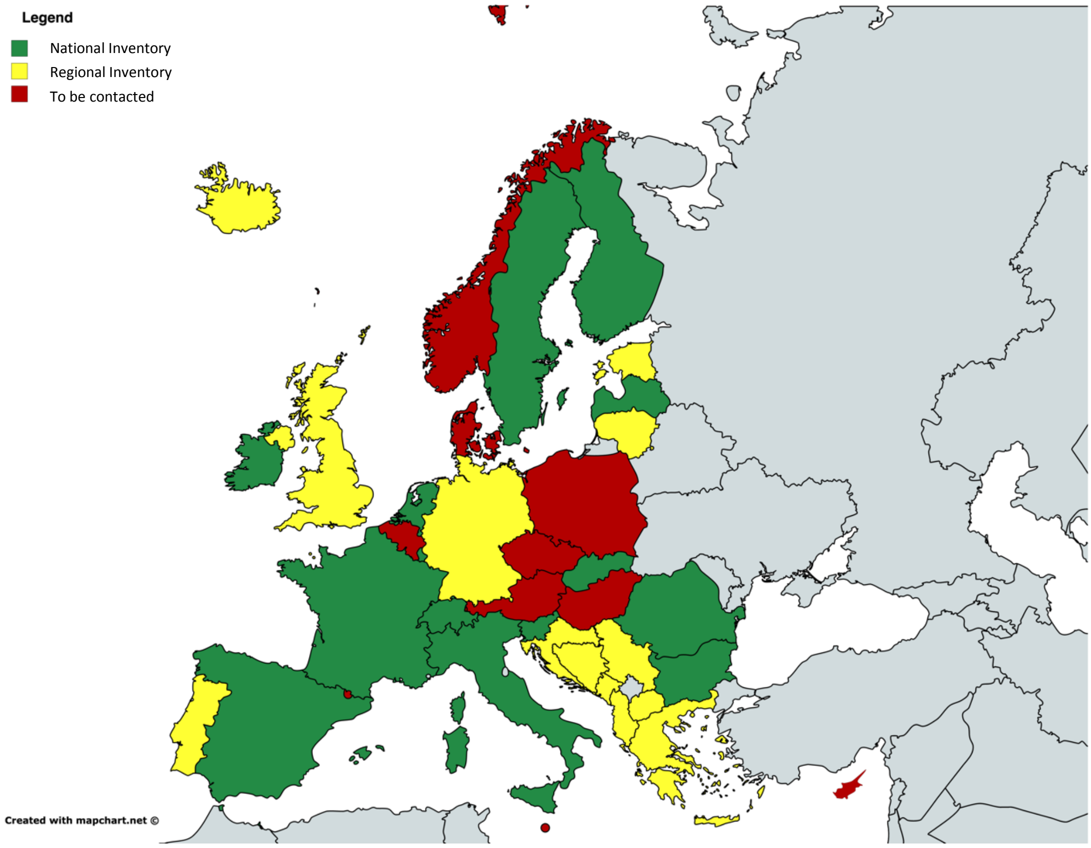

Figure 5. Map of European countries included in the ATLAS, it distinguishes between: i) Contact established, national inventory existing (colour green); ii) Contact established, national inventory not existing (colour yellow); iii) Yet to be contacted (colour red).

We contacted experts in the field and professionals working for water local authorities, universities or research institutes from environment and/or agriculture departments. At the time this report was drafted 27 out of the 38 countries listed have been contacted. For these countries we have an established link to support the data compilation process. The contacts can be part of the AMBER project consortium or not (see Table 2 and Figure 5 for details). They provided information on the availability of existing databases on stream barriers. Table 2 reports the information if national inventories exist or not. Fourteen out of 27 have a national inventory. However, these inventories vary largely between countries. For instance, the national database of Italy included only major dams; barriers higher than 15 m or with a storage volume bigger than 10^6 m^3. It includes 541 dams over the Italian territory (last update July 2015[3]). The France national inventory also includes minor barriers (see also Figure 4) and stores 83795 entries. The difference between these two types of information is substantial and it will hold for all the remaining countries, each of them is likely to have a significantly different accuracy in stream barrier mapping. In order to expand national inventories, which include only large dams, they will have to be integrated with regional databases mapping smaller typologies of barriers normally under the legal responsibility of the regional government authorities. A similar type of exercise has also to be expected for all countries (13) which currently do not have a national database. The

[3] Source: http://www.registroitalianodighe.it

protocol adopted by each country to map stream barriers is a legacy of its legal framework concerning water management. The regional and national levels have to be integrated for each country based on case specific context.

Table 2. Summary of countries contacted, or to be contacted (TBC), and available information on existing stream barrier inventories.

Country	AMBER country	Inventory
Albania	no	Regional
Andorra	no	TBC
Austria	no	TBC
Belgium	no	TBC
Bosnia and Herzegovina	no	Regional
Bulgaria	no	National
Croatia	no	Regional
Cyprus	no	TBC
Czech Republic	no	TBC
Denmark	yes	TBC
Estonia	no	Regional
Finland	no	National
France	yes	National
Germany	yes	Regional
Greece	no	Regional
Hungary	no	TBC
Iceland	no	Regional
Ireland	yes	National
Italy	yes	National
Latvia	no	National
Liechtenstein	no	TBC
Lithuania	no	Regional
Luxemburg	no	National
Macedonia	no	Regional
Malta	no	TBC
Montenegro	no	Regional
Netherland	yes	National
Norway	no	TBC
Polonia	yes	TBC
Portugal	no	Regional
Romania	no	National
Serbia	no	Regional
Slovakia	no	National
Slovenia	no	National
Spain	yes	National
Sweden	yes	National
Switzerland	yes	National
United Kingdom	yes	Regional

2 A ROAD MAP FOR THE FIRST ATLAS OF EUROPEAN STREAM BARRIERS

The disruption of river longitudinal connectivity, together with other hydromorphological degradation, is one of the main causes of impairment of the ecological state according to the WFD. Barriers impact organism, water and sediment connectivity with different magnitude and extent depending on barrier type, but any type of barrier that may have an impact on the ecological state should be reported for the WFD. However, there is a need to limit the survey to a representative, relevant, and yet manageable subset of all European stream barriers. For these reasons, the ATLAS developed within AMBER (see below) will provide, in a first stage, an extensive and exhaustive data compilation of existing databases in order to ensure the coverage of the spatial scale that includes at least all the water bodies as defined by the WFD. This exercise of data compilation, also supported by the results from other Work Packages about barrier impacts on river systems, will allow us to provide more detailed indication on which kind of barriers can and should be monitored at the pan-European level at a later date.

The state of river fragmentation of European rivers is largely unknown. Some databases exist from continental to national and regional levels however, their consistency in terms of typology of mapped barriers and list of variables stored, vary significantly. Some countries have a good national inventory, others are working in this direction, and others have no information at national level. The degree of mapping produced by the ATLAS will not be consistent amongst countries, since some of them have a legacy of previous work with similar objectives whereas other do not. However, **AMBER's ATLAS** has the aim to establish a **common framework** for barrier mapping, data collection and storage, which overtime should create a homogenous and comparable degree of barrier mapping at **pan-European scale**.

In order to guarantee **consistency** in terms of **barrier mapping** within the ATLAS, the basic spatial data that will be used come from the **European Catchment and Rivers network system** (ECRINS; http://www.eea.europa.eu/data-and-maps/data/european-catchments-and-rivers-network#tab-gis-data), that is a composite geographical information system (including river network, river catchments and lakes) where the river networks have been defined on common criteria using a 100 m resolution Digital Elevation Model.

Towards this goal and premise, the ATLAS will have to fulfil important **requirements**, that are summarised in the following sections.

2.1 Data compilation procedure and standards

In a first stage, it is planned to compile and report information on **all barrier types** (any height) because they are relevant and requested by the WFD (i.e. impact on ecological state). We are aware that the results at a pan-EU scale may not be directly comparable as barriers (particularly small barriers such as culverts) are not reported systematically by all EU countries. However, this strategy will allow us to provide a wide picture of the **state of the art** on **river fragmentation** at a pan-EU scale together with an overall picture of **data availability** across Europe. It will also provide the EC important information on **data gaps** regarding different barrier types, and for specific counties, thus allowing future monitoring efforts to be better focussed. A validation procedure (see Section 2.2 below) will test the accuracy of some of the provided databases and of the estimation of river fragmentation as well as the accuracy of data coverage and gaps at pan-EU scale.

The shared metadata **format** will be characterized by general basic information which should be fulfilled (at least in part) by all the database entries. This concerns **11 pre-defined key variables** that are likely to be broadly available in existing databases (see Tables 3 and 4). These variables allow the identification and localisation of a barrier (e.g. source ID, spatial location, river name, etc.) and provide a general description of the main features (i.e. barrier type, height and age). They have been selected after a first survey of already available databases and, in our opinion, these represent the basic information to be reported about the existence, type and physical dimensions of a barrier. The 11 key variable are divided in two types. **Constrained variables (CO)** are mainly numerical variables that are defined a priori within AMBER, as for example the barrier height, that is defined as the vertical distance between the lowest point on the crest of the barrier and the lowest point in the original streambed. **Case specific variables (CS)** are mainly categorical variables that depend on the source database (e.g. categories of barrier type). At this stage, we prefer to collect all the information, even if there is some heterogeneity.

In a second stage, once the ATLAS starts to be populated, a **data quality check** will be performed to analyse the information collected at regional and national level from contacted countries, including variable post-processing. For example, CS parameters will be processed to make them more homogeneous between countries (e.g. provide a unique EU classification for categories of barrier types). This further step would also allow us to evaluate whether additional variables are worth including in the ATLAS from the existing databases (e.g. the barrier status, the barrier use, information on passability, etc.). The final result will be a coherent and reasonably homogeneous pan-European ATLAS that integrates all existing and accessible (to the AMBER project) databases at regional and national scales.

The described procedure and selected variables are compliant with the scopes of a European ATLAS for **barrier reporting.** A proposal of standard protocol (including key variables) that should be adopted by Member States for barrier reporting will be provided during the project. This barrier reporting protocol will use the APP developed within AMBER (see Section 2.5). The ATLAS and displayed variables could in case support barrier assessment but are not exhaustive for assessment purposes. If an assessment of likely impacts is required more detailed information should be collected. For a detailed and suitable list of variables for barrier assessment refer to D1.1 Part A.

It is foreseen that existing data that will not fit within this structure (e.g. barrier age, storage volume, other barrier height values) will be stored within AMBER and used by AMBER partners for research purposes.

The data compilation exercise will be covered by AMBER partners. Specific **guidelines** for data compilation and information gathering are provided in **Annex A**. This includes information for example on the spatial scale used for barrier reporting, the scope and dimension (national, regional, provincial) of the existing databases, and aims to standardize and ease the process of data compilation.

Based on these preliminary stages, project resources will be used to cover poorly monitored countries or to support the compilation of data in those countries where data is scattered amongst regional and provincial authorities consequently requiring more resource to build a national inventory.

Table 3. Key parameters that we propose to be compiled for the ATLAS. CO, constrained; CS, case specific.

Key parameters	*Definition*	*Type*
ATLAS_ID	*New ID defined within* AMBER	*CO*
Source_ID	*ID of the source (national, regional) database*	*CS*
URL	*Link to data source. It can be, e.g.: the web address of the owner institution, the available web address of the national/regional DB*	*CS*
Country	*EU country or EU area, e.g. Balkans, Danube...*	*CO*
X_coord	*Latitude*	*CO*
Y_coord	*Longitude*	*CO*
River	*Name of the river*	*CS*
Basin	*Name of river basin*	*CS*
Height	*Barrier height (m), i.e. the vertical distance between the lowest point on the crest of the barrier and the lowest point in the original streambed*	*CO*
Type	*Dam, weir, spillway, etc.*	*CS*
Year	*Date of building (end)*	*CO*

Table 4. Two example data entries for the ATLAS. CO, constrained; CS, case specific; NA, indicate that the information is not available.

ATLAS_ID	Source_ID	URL	Country	X_coord	Y_coord	River	Basin	Height	Type	Year
CO	***CS***	***CS***	***CO***	***CO***	***CO***	***CS***	***CS***	***CO***	***CS***	***CO***
1	ROE22500	http://www.sandre.eaufrance.fr/ATLASCatalogue/	France	712311	6977061	rivière l'omignon	Artois-Picardie	NA	spillway	NA
2	{1CD63311-EF6D-4694-BB76-97754D76CE67}	http://www.smhi.se/klimatdata/ladda-ner-data/villkor-for-anvandning-1.30622	Sweden	6716679	633862	SE671710-158979	SE3	1	NA	NA

2.2 Data validation

In parallel with the database compilation process, we will develop a **methodology for data validation** and data quality check that will include:
(i) A statistical validation at pan-EU-scale to estimate data coverage and degree of harmonization between countries;
(ii) A field-based validation by means of (a) about 10 spot-checks on sub-basins about the dam existence, plus (b) an extensive survey on selected river systems about the effective data coverage, for 3-4 countries, also making use of the APP (see section 2.5).

The strategy for data validation will be detailed in the report D1.2, due June 2017, and field-based surveys for data validation will be conducted starting summer 2017.

2.3 Data curation, storage and updating

The ATLAS will be made available through the project web site and the JRC data portals, as well as stored and managed by JRC, allowing long-term preservation.

Moreover, the ATLAS needs to include a specific programme for the regular update of the datasets (possibly to be fed directly by Member States). In so doing, the degree of barriers mapping within Europe should overtime become comparable in between countries and increasing cases of dam removal updated and monitored. The updating process will be made feasible by keeping the **source ID** and a link to the source database (e.g. an URL or mail address of the institution). We also propose to define an ATLAS ID that will uniquely identify a barrier at the European scale. Means for updating are, for e.g. the APP, the Member States involvement. Given our resources, we are not able to guarantee a regular updating process by the consortium after the project end.

2.4 Data accessibility and compliance with EU standards

According to H2020 policies, the data in the ATLAS will be **freely available** and **public**. This will be mentioned to the different institutions that will provide us data during the data compilation phase.

The general structure of the ATLAS will be built in order to guarantee the compliance with EU standards (i.e. INSPIRE; http://inspire.ec.europa.eu/). In the case of collected data, it will be reported if the source data is INSPIRE compliant or not.

2.5 The ATLAS and the APP

The AMBER project is also developing a mobile phone APP for collecting barrier information using a Citizen Science programme. The 11 key parameters proposed in Table 3 for barrier reporting are consistent with the design of the APP framework. In a second phase of data compilation when the ATLAS will be already structured and partly populated experiments will be carried out to test if and how the information derived from the APP may be used to populate the ATLAS. The field validation exercise (see Section 2.2) may utilise the second tier of APP development for collecting more detailed information by experts.

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Vörösmarty CJ, McIntyre PB, Gessner MO, *et al.,* (2010) Global threats to human water security and river biodiversity. *Nature* **467**:555–61. doi: 10.1038/nature09440

Yin X-A, Yang Z-F, Petts GE, Kondolf GM (2014) A reservoir operating method for riverine ecosystem protection, reservoir sedimentation control and water supply. *Journal of Hydrology* **512**:379–387. doi: 10.1016/j.jhydrol.2014.02.037

ANNEX A: GUIDELINES FOR DATA COLLECTION

Each expert responsible for data compilation of barriers in one or more countries will be instructed to adhere to the following guidelines.

1) The data provided will be openly accessible and 'FAIR', that is findable, accessible, interoperable and re-usable (EU 2016). Institutions providing information must agree with this data management policy.
2) The existing national (or regional) databases (DBs) provided to the AMBER project must be delivered in any easily readable table format (e.g. xls, db, cvs, etc.). Variable (or column) names must be written in English. A short description/technical definition of each variable should also be reported (see for example Table 1). A list of key priority variables and their description is reported in Table 3 of the main D1.1 Part B report (also reported here in Table A1 for clarity). A check list to be filled by the expert (last column in Table A1) has been added to indicate if these variables exist or not in the DB provided. This task will support the following steps of data harmonization at pan-EU scale.

Table A1. Proposed key parameters for the ATLAS. CO, constrained; CS, case specific; regarding the last column to be filled, in case the variables is not existing in the source DB analysed please indicate NA.

Key parameters	*Definition*	*Type*	***Variable name in the Source DB***[4]
ATLAS_ID	*New ID defined within AMBER*	*CO*	
Source_ID	*ID of the source (national, regional) database*	*CS*	
URL	*Link to data source. It can be, e.g.: the web address of the owner institution, the available web address of the national/regional DB*	*CS*	
Country	*EU country*	*CO*	
X_coord	*Latitude (WGS84)*	*CO*	
Y_coord	*Longitude (WGS84)*	*CO*	
River	*Name of the river*	*CS*	
Basin	*Name of river basin*	*CS*	
Height	*Barrier height (m), i.e. the vertical distance between the lowest point on the crest of the barrier and the lowest point in the original streambed*	*CO*	
Type	*Dam, weir, spillway, etc.*	*CS*	
Year	*Date of building (end)*	*CO*	

[4] This column must be filled by expert reporting for each distinctive Database

3) All existing documents and reports about the DB regarding data structures, data standards, updating policy must be provided together with the DB. Here, list their references or indicate URL link:
   ……
   ………
   ……
   ……
   ……

4) Please answer the following questions:
   a) What has the database built for? Has the DB been used for the WFD reporting about 1st and/or 2nd RBM plan?
      ………
      ………
      ………
      ………
      ………
   b) What spatial scale or criteria have been adopted to identify the river network used to report barriers existence? If a report on existing protocols exists please attach it (or indicate URL link if available online). If this information is not existing it is important to provide details on the scale of the river network map used for data reporting; or alternatively, if a Digital Elevation Model was used to derive the river network, provide the criteria (e.g. minimum drained area) used to define a river channel.
      ………
      ………
      ………
      ………
      ………
   c) Is the database INSPIRE compliant? (Yes/No)
      (please see: http://inspire.ec.europa.eu/documents/inspire-data-specification-hydrography-%E2%80%93-technical-guidelines-31)?
      ………
      ………
      ………
      ………
      ………

5) In case the database for a country is composed of more than one database (e.g. a National plus available Regional/Provincial databases), the previous steps must be replicated for each independent source of information provided (since each is likely to adopt different standards for data collection).