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

D3.2 Quantification of economic costs and benefits of river infrastructures (evaluation of natural capital)

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History of changes

Version	Date	Changes	Pages
1.0	30 July 2019		

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Executive summary

This is version 1.0 of the Deliverable D3.2 `Quantification of economic costs and benefits of river infrastructures (evaluation of natural capital)’. This report is a deliverable of the AMBER project. This project has received funding from the European Union’s Horizon 2020 research and innovation programme under grant agreement No 689682.

The problem/topic

The Durham University Business School team working on AMBER is focusing on three main areas: impacts on river geomorphology (Department of Geography), impacts on fish habitats and migration behaviour (Department of Biosciences), and impacts on a watershed’s performance and socioeconomics (Business School). Findings from this last topic are presented in this report.

Summary of key findings

With increased population pressure on land resources, we can expect the demand on river infrastructures to continue to change. Some of the existing river barriers will become redundant, others will need to be built. For example, to build new reservoirs for renewable generation from hydropower, but also to provide water storage when areas have to deal with increasing weather extremes leading to floods or droughts. While the environmental impacts of large dams on European rivers have been widely researched, what is less studied are the benefits and costs of such infrastructural change.

Intervention measures, especially physical modifications in rivers and wider catchment areas, are often implemented by institutions without adequate investigation of the total economic effect on end users. In other words, while the engineering cost for construction of interventions are quite well-known, what is missing is the measure and scale of public benefits. This would also enable society to identify unintended consequences of intervention.

To systematically investigate economic benefits in advance is a highly beneficial best-practice, which also enables the undertaking of ongoing monitoring of the cause-effect relation that exists between intervention measures and public benefits. The AMBER project does not include a budget for collecting socio-economic data from the case studies, making it impossible to assess nonmarket values and ecosystem services flows associated with potential interventions from primary data. The project hence focussed on secondary data on economic benefits and costs associated with dam removal and on how to make use of data generated by other research initiatives related to this investigation.

Data generated from other partners that had budgets to collect information on other subjects for each case study were only partially useful for undertaking socio-economic assessments. We therefore decided to undertake tests for assessing socio-economic data on a larger pan-European scale, by overlapping data from AMBER's partners Politecnico of Milan(POLIMI), who developed the atlas of barriers for European rivers, and data from the Joint Research Centre (JRC), who developed the atlas on Ecosystem Services.

A summary of conclusions

The engineering costs of river barrier removal are highly variable, and this reflects the broad typology of engineering solutions implemented in the realization of these barriers, as well as the vast variation of contexts in which these are located. The most consistent result is that the sediment volume and type (especially when this is contaminated) are the two strong determinants of barrier removal cost, via the cost of sediment relocation. Another aspect that has emerged as important is that when the removal affects some other existing infrastructure, such as viaducts, bridges and roads. In some instances, like in the presence of a hydropower plant, or a series of plants, located downstream of the barrier to be removed, removal may lead to additional costs due to loss of hydropower generation during removal works.

On the benefit side, the picture that emerged is much more articulate. When benefits are mostly derived by the natural environment in the form of nonmarket goods, different techniques can be used to derive estimates of nonmarket value. There is no clear evidence of factors that affect benefits in a systematically important way, except perhaps for the density of population in the area that can directly enjoy the increased supply of nonmarket goods. Specifically, we found it useful to focus on the study of the economic impacts of river fragmentation on a watershed's ecosystems performance from an ecological and socio-economic point of view. Extensive river fragmentation has a profound impact on a watershed's habitats, mainly negative for those communities thriving in flood plains and ecotones (the transition zone between different ecosystems, including riparian and hyporheic zones). This often results in degrading the potential performance of several of the watershed's communities, including human, especially downstream of the barrier. The benefits provided by natural systems to humans are referred to as ecosystem services. Artificial barriers often alter ecosystem services, benefiting some and reducing others. We hence concluded that a case-by-case study is the only way to address the net effect.

As part of our study we developed a partnership with a research team from European's Joint Research Centre (JRC) specialising in ecosystem services. Together we developed the idea of using geographic information system tools to overlay AMBER's data with those from the JRC, to model ecosystem services demand and supply for the EU. The idea is to test an innovative approach that would enable stakeholders to undertake a first screening, and determine if a barrier, or a set of barriers, is possibly having an impact, positive or negative, on a selected number of ecosystem services for that region. For the research, the team has chosen the following three ecosystem services: flood regulation, outdoor recreation and water purification, with focus on the UK. The proposed approach, together with other tools developed by AMBER partners, is aiming to assist decision makers in developing barrier adaptation strategies and implementing mitigation measures, and to monitor the extent of gains and losses produced by river restoration projects. Preliminary findings are included as an AMBER deliverable, while further collaboration between Durham and JRC is envisaged to expand on this pilot test.

A summary of recommendations

Assessing the costs for river barrier removal interventions from secondary data, including the potential loss of hydropower generation in case of the presence of a plant or a series of plants located downstream of the barrier to be removed, is a relatively straightforward process, although much of this information pertains to non-academic sources, such as grey literature and to consultancy reports. What is more difficult is to measure public benefits, which are often unevenly distributed in terms of time and space. For this reason, in future investigations we advise directing efforts towards collection of specific datasets. The AMBER project did not include a budget for collecting socio-economic data from the case studies, making it difficult to assess nonmarket values and ecosystem services flows associated with potential interventions. To know public benefits in advance is a highly beneficial best-practice, which also enables the undertaking of ongoing monitoring of the relationship of cause-effect that exists between intervention measures and public benefits. For future similar projects, it is advisable to allow sufficient budget for the collection of data, and for development of socio-economic information systems to enable exploring the links between solutions and the wider society. This will probably require a categorisation of river barrier types, based on the features that determine economic benefit values.

We suggest that further studies should be undertaken to develop a socio-economic information system that can systematically explore the links between implemented engineering solutions (including nature-based solutions) and the wider society.

Authors

Sergio Vallesi (DU), Riccardo Scarpa (DU), Simone Bizzi (POLIMI), Wouter van de Bund (JRC), Carlos Garcia de Leaniz (SU), Jim Kerr (SOTON) from the AMBER Consortium.

Acknowledgements

Alessandra la Notte, Georgia Kakoulaki, Sara Vallecillo from JRC for support with ecosystem services data, and Francesco Rossato from Water Environment Limited for support with GIS mapping.

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1 LITERATURE REVIEW

1.1 River restoration vs. watershed repairs

In the European Union, the ambitious Water Framework Directive (WFD, 2000/60/EC) acknowledges the opportunity to improve the quality of all water bodies, including surface waters and groundwater, and there is strong evidence of a society that tries to reinstate a water-oriented culture. Water protection is one of the priorities of the Commission, and the aim of the European Water Policy is to reduce the pollution in European rivers and ensure that they remain clean (European Commission, 2019). The aim of the WFD are:

- expanding the scope of water protection to all waters;
- achieving "good ecological status" for all waters by a set deadline water management based on river basins;
- "combined approach" of emission limit values and quality standards;
- getting the prices right;
- getting the citizen involved more closely;
- streamlining legislation.

The novelty of the approach is the recognition that making Europe's waters cleaner requires managing water on a river basin scale. This model requires cooperation and joint objective-setting across Member State borders, and in the case of i. a. the Rhine, even beyond the EU territory (European Commission, 2019).

An increasing interest for river restoration is evidenced by intense academic activity from the beginning of the millennium (Brouwer R. , 2017), as shown in **Figure 1**.

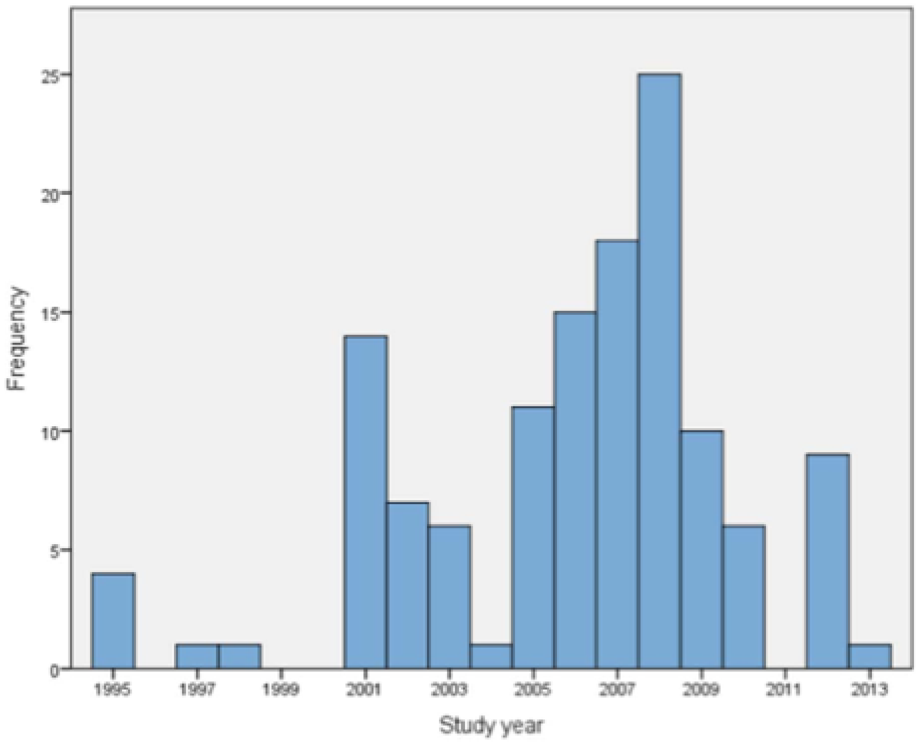

Figure 1. Distribution of river restoration valuation studies across years (Brouwer R. , 2017)

Limiting the action to the restoration of rivers alone is not going to achieve the targets of the WFD. A broader context is required to include restoration of other elements of watersheds. Furthermore, the word “restoration” alludes to the ideal of restoring a system to either its original status or to some earlier more natural status. This is often unrealistic, due to the level changes on a physical, biological, and climate basis. As discussed in the introduction chapter, participative watershed mapping should be the first step toward the reconstruction of how the present status quo was reached (ecogenesis) and determination of what watershed repairs are required to enhance water quality, water balance, and ecological performance across the system.

There are few watershed restoration case studies. Some of these are of significance, such as the Loess Plateau Rehabilitation Project, where more than 2.5 million people in four of China’s poorest provinces were lifted out of poverty through the introduction of sustainable farming practices to repair land which was heavily degraded by unsustainable land-uses, such as intensive grazing (World Bank, 2007). According to the results published by the World Bank, the households involved by the project saw their incomes grow by almost 200%; from about US$70 per year per person to about US$200. Most of the increase came about through agricultural productivity enhancement and diversification. The project required total bans on grazing, regeneration of grassland and shrub cover on previously grazed/cultivated slope-lands, the construction of a countless number of terraces to retain soil and water and for replanting perennial trees. Aside from the large poverty relief and increase in employment rate, other key outcomes include a remarkable improvement in the volume of sediments entering the Yellow River, which was reduced by more than 100 million tons each year, and a significant reduction of risk of flooding. (World Bank (2007).

Projects of this scale can achieve outstanding results, and boost in a relatively short time the watershed’s ecological performance as well as the number of sustainable livelihoods per hectares. Although these projects are still rare, during recent years they have been clearly on the rise, especially as early successful attempts have been implemented in critical parts of the world, such as in Sub-Saharan Africa. The model of a watershed restoration, or better, watershed repairs, can be also applied to the river restoration initiatives. Regardless of if the repair actions are going to be limited to the river area only or not, the idea of assessing the entire watershed performance and potential remains valid, especially if the ultimate goal is not only improving water quality but also water availability for the agricultural sector.

1.2 Costs-benefit of river infrastructures

This chapter focuses on the key literature that studies costs and benefits related to river infrastructures in the context of river restoration, as barrier removals are often are part of such restoration projects. Particular attention was given to the fact that, although there is an interest in evaluating the benefits and costs of a specific intervention, a common methodology is still lacking, simply because the range of circumstances manifested so far do not lend themselves to a single systematic approach.

Ideally, when analysing a restoration project, an array of standardised indicators should be provided to decision makers and stakeholders involved in a restoration scheme. The indicators should consider the different and varied aspects of the interaction of the infrastructure within the watershed. These are bound by a common thread that sees a river barrier from its structural point of view, its function in the flood regulation context, and the monetary and non-monetary values associated with its removal or maintenance (**Figure 2**).

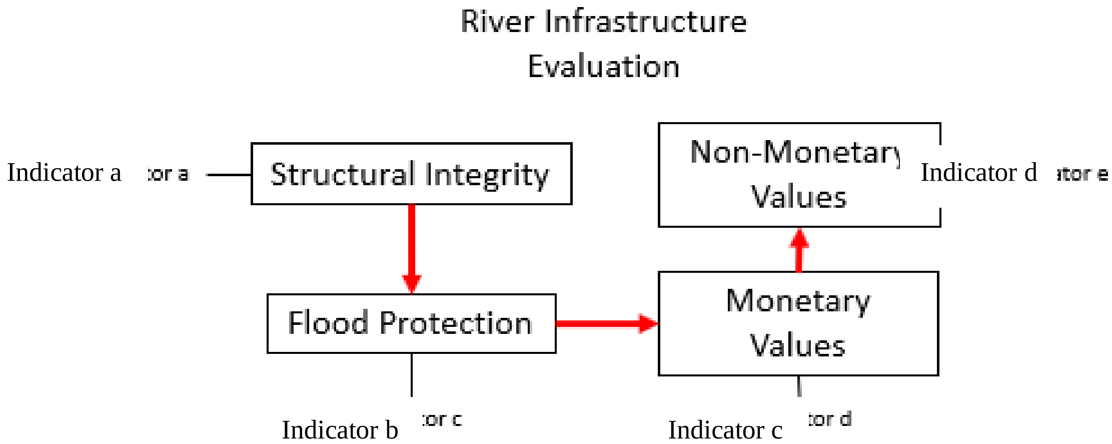

Figure 2. Standardised indicators (a graphical simplification)

A first step before assessing costs and benefits of any small barrier or large dams, should be determining its level of safety, from a structural point of view. As discussed in the introduction chapter, dam failures have multiple causes and those directly related to its nature of structure are:

- Design errors;
- Cost savings (in materials and techniques) during the construction, in spite of safety;
- Dam structure and spillway systems inadequate to withstand large seismic events and floods;
- Wrong choice of location for geological reasons;
- Poor maintenance.

Large dam failures are not rare in Europe. Some example of how large economic damages from dam disasters can be are presented below.

- On October 4, 2010, the embankment of Basin X of the red mud reservoir in Hungary failed and released a mixture of 600-700 thousand cubic metres of red mud and water. The slurry flooded the lower sections of the settlements of Kolontár, Devecser and Somlóvásárhely via the Torna creek. Ten people were killed, and approx. 120 people were injured. The spilling red mud flooded 800 hectares of surrounding areas. (Wise-Uranium, 2014). Immediately following the 'Red sludge' alumina plant accident, the Hungarian government passed a law that would allow the state to take control over companies responsible for environmental catastrophes. (Srebotnjak, 2019).

- On April 25, 1998, a tailings dam failure of the Los Frailes lead-zinc mine at Aznalcóllar near Seville, Spain, released 4-5 million cubic meters of toxic tailings slurries and liquid into nearby Río Agrio, a tributary to Río Guadiamar. (Wise-Uranium, 2015). The deluge of heavy metal spread across nine municipalities of Seville province, including Europe's most important wetland reserve, the Doñana national park. The environmental consequences of the disaster were enormous: right after the dam's collapse the entire aquatic life of the nearby Guadiamar river vanished and over 30 tons of dead fish were collected from the shores. (Srebotnjak, 2019). After a significant investment of at least €300 million in decontamination travails, a decade later environmental scientists declared the soil surrounding Aznalcollar to have "recovered reasonably well". (Srebotnjak, 2019).

- In 1923, 356 people were killed in the Gleno Dam failure, in Italy, when an arch dam collapsed. This event deeply influenced the evolution of Italian regulations regarding dam design and hydraulic risk evaluation (Pilotti, 2011). The cause of the Gleno Dam disaster can be found in the willingness to save money in spite of dam safety: the original design (a gravity dam) was replaced during construction with a cheaper arch dam that, after a heavy rainfall event, collapsed, sliding partly on the structure and on the smooth bedrock (Herzog, 1999).

- A similar accident, of even greater consequence, was the failure of the Vajont dam. This was a 262m high concrete arch dam (at the time, one of the tallest dams in the world), which was completed in 1959. Four years later (1963) during lake filling, an active landslide accelerated its course and collapsed on the reservoir causing a massive wave in the lake, which went over the dam and caused the immediate destruction of several villages and the death of 1,910 people living downriver.

Increasing rainfall intensity and changing climate patterns can also find a dam inadequate, and therefore a second step should be that of assessing the flood passing capacity to ensure the structure has sufficient spillway to pass large floods, and that the level of protection and risk that is deemed acceptable is provided.

The outcome from these two reviews provides the basis for considering options for maintenance vs. removal. If there is clear evidence that a dam is unsafe and it represents a threat for human life, or if the costs for its upgrade are excessive, its removal should be prioritised.

Regardless of whether the intervention at hand is a new barrier, assessing the performance of an existing barrier, or the potential for its redundancy, the overall economic impact of an intervention is typically evaluated through the cost-benefit analysis (CBA). This technique is about measuring in 'money terms' the present value (i.e. the flow discounted to the present) of future benefits and costs of a project to society. The process consists of converting costs and benefits to the same unit, which is their monetary present-value in a given currency, and then reporting the ratio of costs over benefits. The underlying assumption is that utility to society can be measured by money. The EU Commission promotes the use of CBA for major infrastructure projects above €50 million, has specific requirements for conducting CBAs, and has developed a specific guide for Member States to follow; the European Guide to Cost-Benefit Analysis of Investment Projects (European Commission, 2014).

While the values of market goods and services are relatively easy to estimate, particular attention is needed to estimate the economic value of non-market or intangible impacts. Those services provided by restored rivers that are not marketable (and subsequently are not monetised) will be often left out of the analysis (Brouwer, 2017). The wider social value attached those benefits, such as the existence of free-flowing water that can be enjoyed (Sanders, 1990) is frequently missing.

Economists have developed a range of approaches and methods to measure non-market benefits and costs. The most common methods are briefly presented in the following paragraphs, extracted from the guide *Cost-Benefit Analysis and the Environment* (Pearce, 2006). More detailed and technical descriptions can be found in the specialised literature that has grown rapidly in the last decades.

a) Hedonic pricing

'Methods on hedonic pricing utilise the fact that some market goods are in fact bundles of characteristics, some of which are intangible goods (or bads). By trading these market goods, consumers are thereby able to express their values for the intangible goods associated with them, and these values can be uncovered through the use of statistical techniques applied to adequate data.

However, this process can be hindered by the fact that a market good can have several intangible characteristics, and that these can be collinear in the existing data. It can also be difficult to measure the intangible characteristics in a meaningful way.

b) Travel cost

Travel cost methods utilise the fact that market and intangible goods can be complements (weak complentarity), to the extent that purchase of market goods and services is required to access an intangible good. Specifically, people have to spend time and money travelling to recreational sites, and these costs reveal something of the value of the recreational experience to those people incurring them. However, the situation is complicated by the fact that travel itself can have value, that the same costs might be incurred to access more than one site, and that some of the costs are themselves intangible (for example, the opportunity costs of time). Nevertheless, with adequate time and data one can derive demand functions for nonmarket goods associated with recreational activities.

c) Stated preference valuation: contingent valuation

Stated preference techniques for nonmarket valuation use survey techniques based on in-person, web-based, computer-aided or phone surveys. Through these, survey respondents are taken through questionnaires which either directly ask respondents for their willingness to pay (accept) or offer them choices between “bundles” of attributes accessible at a cost (when they represent improvements) or for compensation (when they represent worsening). From the observed hypothetical choices, the analysts can infer estimates of average willingness to pay (WTP) or willingness to accept (WTA).

Stated preference methods more generally offer a direct survey approach to estimating an individual’s or a household’s preferences, and more specifically, WTP amounts for changes in provision of (non-market) goods, which are related to respondents’ underlying preferences in a consistent manner. Hence, this technique is of particular worth when assessing impacts on nonmarket goods, the value of which cannot be uncovered using revealed preference methods.

The growing interest in stated preference approaches has resulted in a substantial evolution of techniques over the past 20 to 40 years. For example, the favoured choice of elicitation formats for WTP questions in contingent valuation surveys has already passed through a number of distinct stages. This does not mean that uniformity in the design of stated preference surveys can be expected any time soon, nor is this particularly desirable. Some studies show how, for example, legitimate priorities to minimise respondent strategic bias by always opting for incentive compatible payment mechanisms must be balanced against equally justifiable concerns about the credibility of a payment vehicle. The point is the answer to this problem is likely to vary across different types of project and policy problems.

There remain concerns about the validity and reliability of the findings of contingent valuation studies. Indeed, much of the research in this field has sought to construct rigorous tests of the robustness of the methodology across a variety of policy contexts and non-market goods and services. By and large, one can strike an optimistic note about the use of the contingent valuation to estimate the value of non-market goods. In this interpretation of recent developments, there is a virtuous circle between translating the lessons from tests of validity and reliability into practical guidance for future survey design. Indeed, many of the criticisms of the technique can be said to be imputable to problems at the survey design and implementation stage rather than to some intrinsic methodological flaw. Taken as a whole, the empirical findings largely support the validity and reliability of contingent valuation estimates, when studies are well-conducted and validated. However, such studies tend to be expensive.

d) Stated preference valuation: choice modelling

Many types of environmental impact are multidimensional in character. Hence an environmental asset that is affected by a proposed project or policy often will give rise to changes in component attributes each of which command distinct valuations. The application of choice modelling (CM) approaches to valuing multidimensional environmental problems has been growing steadily in recent years. CM is now routinely discussed alongside the arguably better-known contingent valuation method in state-of-the-art manuals regarding the design, analysis and use of stated preference studies (Johnston et al. 2017). While there are a number of different approaches under the CM umbrella, it is arguably the choice experiment variant (and to some extent, contingent ranking) that has become the dominant CM approach with regard to applications to multi-attribute environmental goods.

In a choice experiment, respondents are asked to choose their most preferred option from a choice set of at least two options, one of which is the status quo or current situation. It is this CM approach that can be interpreted in standard welfare economic terms, an obvious strength where consistency with the theory of cost-benefit analysis is a desirable criterion.

Much of the discussion about, for example, validity and reliability issues in the context of contingent valuation (CV) studies applies in the context of the CM. While it is likely that on some criteria, CM is likely to perform better than CV–and vice-versa –the evidence for such assertions is largely lacking at present. While those few studies that have sought to compare the findings of CM and CV appear to find that the total value of changes in the provision of the same environmental good in the former exceeds that of the latter, the reasons for this are not altogether clear. However, whether the two methods should be seen as always competing against one another – in the sense of say CM being a more general and thereby superior method – is debatable. Both approaches are likely to have their role in cost-benefit appraisals and a useful contribution of any future research would also be to aid understanding of when one approach should be used rather than the other.

e) WTP versus WTA?

Traditionally, economists have been fairly indifferent about the welfare measure to be used for economic valuation: willingness to pay (WTP) and willingness to accept compensation (WTA) have both been acceptable. By and large, the literature has focused on WTP. However, the development of stated preference studies has, quite regularly, discovered divergences, sometimes substantial in magnitude, between WTA and WTP. These differences still would not matter if the nature of property rights regimes were always clear. WTP in the context of a potential improvement is clearly linked to rights to the status quo. Similarly, if the context is one of losing the status quo, then WTA for that loss is the relevant measure. By and large, environmental policy tends to deal with improvements rather than deliberate degradation of the environment. So, there is a presumption that WTP is the right measure. The problems arise when individuals can be thought of as having some right to a future state of the environment. If that right exists, their WTP to secure that right seems inappropriate as a measure of welfare change, whereas their WTA to forego that improvement seems more relevant. In practice, the policy context may well be one of a mixture of rights, for example, a right to an improvement attenuated by the rights of others not to pay “too much” for that improvement.

Finding out why, empirically, WTA and WTP differ also matters. If there are legitimate reasons to explain the difference, then the preceding arguments apply, and one would have to recommend that CBA should always try to find both values. The CBA result would then be shown under both assumptions. But if the observed differences between WTA and WTP are artefacts of questionnaire design, there is far less reason to be concerned at the difference between them. The “fallback” position of their approximate equality could be assumed. Unfortunately, the literature is undecided as to why the values differ. This again suggests showing the CBA results under both assumptions about the right concept of value.

f) Benefits transfer

Benefits or value transfer involves taking economic value estimates from one context in which a proper investigation has been conducted and at least theoretically validated (the study site) and applying them to another context (the transfer site), often after adjustment. Transfer studies (for costs and benefits) are the bedrock of practical policy analysis in that only infrequently are policy analysts afforded the luxury of having money and time sufficient for designing and implementing original studies. In general, analysts must fall back on the information that can be gleaned from past studies. This is likely to be no less true in the case of borrowing or transferring WTP values to policy questions involving environmental or related impacts. Almost inevitably, benefits transfer introduces subjectivity and greater uncertainty into appraisals in that analysts must make a number of additional assumptions and judgements to those contained in original studies. The key question is whether the added subjectivity and uncertainty surrounding the transfer is acceptable and whether the transfer is still, on balance, informative.

Surprisingly given its potentially central role in environmental decision-making, there are no generally accepted practical transfer protocols to guide analysts. However, a number of elements of what might constitute best practice in benefits transfer might include the following. First, the studies included in the analysis must themselves be sound. Initial but crucial steps of any transfer are very much a matter of carefully scrutinising the accuracy and quality of the original studies. Second, in conducting a benefits transfer, the study and policy sites must be similar in terms of population and population characteristics. If not, then differences in population, and their implications for WTP values, need to be taken into account during adjustment. Just as importantly, the change in the provision of the good being valued at the two sites also should be similar.

The holy grail of benefits transfer is the consolidation of data on nonmarket values in emerging transfer databases via the systematic construction and validation of “function transfer” (Leon-Gonzalez and Scarpa, 2008; Matthews at al. 2009). Yet, while databases are to be welcomed and encouraged, these developments still need to be treated with some caution. Thus, there is a widely acknowledged need for more research to secure a better understanding of when transfers work and when they do not, as well as developing methods that might lead to transfer accuracy being improved.

However, a competent application of transfer methods demands informed judgement and expertise and sometimes, according to more demanding critics, as advanced technical skills as those required for original research. At the very least, it suggests that practitioners should be explicit in their analysis about important caveats regarding a proposed transfer exercise as well as take account of the sensitivity of their recommendations to changes in assumptions about economic values based on these transfers.’

1.3 A paradigm shift

The value of restoring fragmented and channelized watercourses into free-flowing rivers is similar to restoring stormwater from urban surfaces and drains into living streams. In both of these contexts, a paradigm shift is required in order to be able to see the true socio-economic benefits of such a different management approach. The effects are mostly studied from the perspective of the investor, either public or private. What is generally missing is the full impact of a project on the wider society, intended as a whole, and most of the times these kinds of projects find it difficult to carry out comprehensive benefits valuations. An example is the difficult implementation of Water Sensitive Urban Design (WSUD) in Australia.

WSUD is a land planning and engineering approach that integrates the urban water cycle into design, in order to minimise environmental degradation of urban areas (JSCWSC, 2009) and provide restoration services. In practical terms, WSUD aims at naturally collecting and making use of stormwater to reduce the damage it causes to water bodies. In urban areas, rainwater falls onto impervious urban surfaces, where it collects and carries pollutants, and then enters natural waterways through stormwater drains. At the opposite spectrum of this scenario, as shown in **Figure 3**, is the natural environment, where rainwater is absorbed by plants and infiltrates into the ground (Melbourne Water, 2017). WSUD has the same concept of the Low-Impact Development used in Canada and in the United States, the Sustainable Drainage Systems used in the United Kingdom, and the sponge cities approach used in China. A reason of the limited success of these practices is the lack of retrospective evaluations that support their effective benefits, in terms of quantifiable values.

Conversely, a virtuous framework that testifies a nationwide environmental-economical commitment can be found in Switzerland: here the government is planning substantial investment to achieve a better water quality. With two Swiss representative rivers as a case study, Logar et al., 2019, translated ecological benefits in monetary and quantifiable terms, investigated public WTP for river restoration and conducted a CBA. With the intent to have a broader picture of costs and benefits at a national level, the results have been upscaled. What emerged is that, although budget allocated for Swiss river restoration might be insufficient, the local population is willing to pay more than they are required to. Moreover, when performing the CBA, benefits outweighed the costs, justifying any economic investment.

1.4 Papers reviewed

The value of restoring urban drains to living streams has been addressed in one of the papers we reviewed, which was published in a special issue of *Water Resources and Economics* (Polyakov & Al, 2017). The study focuses on the Bannister creek in Western Australia. With the intent to provide retrospective evaluation of the environmental benefits and impacts of a restoration project, the local property value has been estimated using the hedonic method (Rosen, 1974). As anticipated in the previous section, this method assumes that a quantifiable variable (the sale price of a property) is a function of the underlying property attributes, including amenity benefits. In simplistic terms, the author studied the contribution of the Bannister creek restoration to the increase in property values in the surrounding area, demonstrating that benefits are significantly larger than costs.

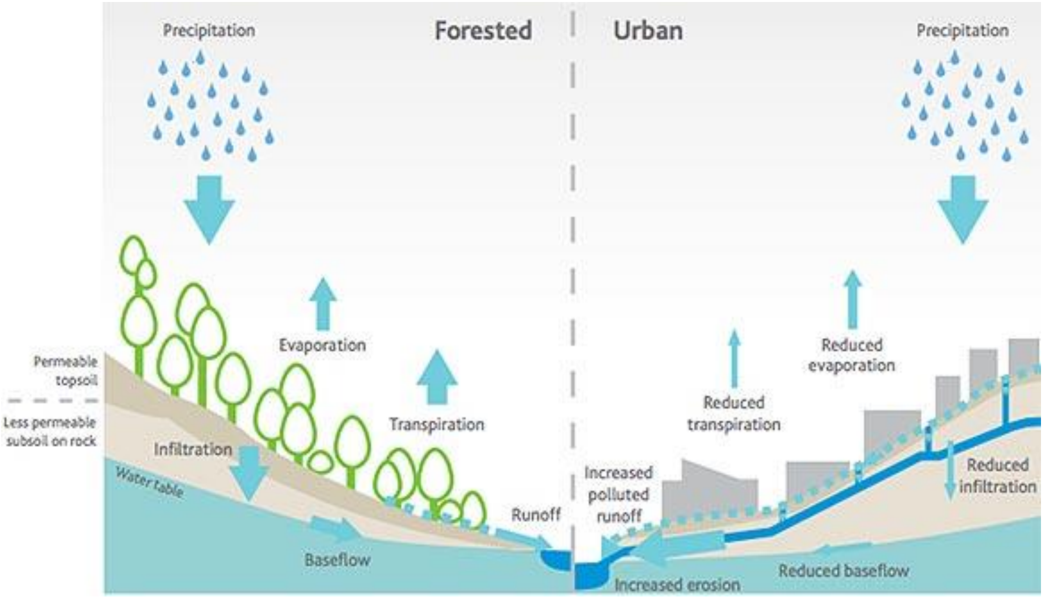

Figure 3. Different path of rainfall between natural and urban environment. Source: Melbourne Water (2017)

The hedonic valuation model has also been used within the context of another study we reviewed, the Edwards dam removal on the River Kennebec in the United States. Pollution and degradation, which in part is a result of damming, has been fuelled by the many industries located by the Kennebec and by other Maine rivers. This is reflected in a negative relationship between river closeness and housing prices. The Edwards Dam removal, a benchmark restoration project, has made it possible to investigate home sales pre- and post-removal of a dam, revealing significant penalty reduction in locations nearby the dam after restoration (Lewis, 2017). Another study (Lewis, 2018) analysed three different rivers and four markets: it has been found that vicinity to a river is negatively valued because of several factors, such as unpleasant odours during the hot season (due to stagnating water), and it is a reflection of Maine's rivers status. In the study, hedonic property value analysis with a sample of data spanning 10 years, is used to evaluate river characteristic and their influence on property price.

Another case study we reviewed was carried out in Norway, a country that produces 96% of its total electricity demand through hydropower (Graabak, 2017) - the largest production share in Europe. Despite the clean energy provided, ecological negative effects, including fish population reduction, have been noteworthy due to extensive damming and river fragmentation (Hansen, 2008). Ecological and economical effects are usually studied without accounting for their complementarity (Jager & Smith, 2008): either optimising hydropower is studied neglecting environmental issues, or sustaining fish population studies partly ignore hydropower generation. Within this context, we reviewed a comprehensive study that analylises the trade-offs between the production of Atlantic salmon smolts (young salmons) and hydropower generation (Bustos, 2017). The study was carried out by correlating the non-monetary values of the river restoration, such as the ecological status of a restored river expressed in fish production and recreational fishing, to the hydropower revenue.

Non-monetary values associated with river restoration interventions can be the subject of qualitative information and discussions annexed to CBA reports (Brouwer R. , 2017). However, an effort has been made to retrieve economic valuation studies that enable the assignment of a monetary value to nonmarket goods and services, thereby leading to their potential inclusion in the CBA, and hence in the decision-making process. For instance, a recent study (Bergstrom & Loomis, 2017) shows that out of 38 river restoration projects, non-monetary valuation methods had a decisive role in the decision-making of only 16% of the cases (Laurans & Al., 2013). In the evaluated studies, 10% of the non-monetary methods were used with a subsidiary but not decisive role (Laurans & Al., 2013). A more significant 74% of the valuation methods were applied with an informative role within the decision-making process, providing background information that might have been of influence.

What is interesting about the 2017 Bergstrom and Loomis study (Bergstrom & Loomis, 2017) is that they have conducted an extensive literature search on almost all the river restoration valuation studies, particularly in the US and in Europe. It therefore provides us with a comprehensive overview on the methodologies that are used to estimate the economic values in river restoration. In the 38 river restoration valuations, the following methods have been used:

- Contingent Valuation Method (CVM) – 13/38
- Choice Experiment (CE) – 14/38
- Hedonic Price Method (HPM) – 2/38
- Travel Cost Method (TCM) – 2/38
- Benefit Transfer – 3/38
- Simulation Models (SIM) – 2/38
- Replacement Cost Approach (RC) – 2/38

Note that TCM and HPM do not include non-use values, and neither do SIM and RC. That is, important with regards to the influence of those values in the decision-making process.

The specialised literature review on river restoration highlighted that willingness-to-pay (WTP) is the standard metric used to quantify economic value to market and non-market goods. These measures are estimated using the contingent valuation (CV), a stated preference method that creates a hypothetical market to prompt individual preferences and hypothetical willingness to pay (WTP). While it can be used for all public goods, it is frequently applied to the valuation of environmental resources (Bateman, 2002) and ecological restorations (Schultz, 2012). The CV method has been used, for example, to assess public values and preference from different stakeholder groups regarding a change in the ecosystem services for stream restoration in Finnish forested watersheds (Lehtoranta, 2017). By applying this method, it has been highlighted that when the benefits of restoration are more visible and easier to understand, the potential for funding river restoration activities is possible with the public and stakeholder support.

Mattman, Logar & Brouwer (Mattmann, 2016) conducted a meta-analysis on the existing literature concerning the economic valuation of the external effects of hydropower. In their study, it has been found that the public opinion is generally opposed to hydropower because of it causes degradation of a the river’s natural flow, and generally produces landscape to ecosystem worsening. However, ex-post, the WTP to mitigate impacts of hydropower is limited.

Transferability of the non-monetary benefits from a river restoration has been also considered at a cross-national watershed scale: for the River Danube (Brouwer, 2016). The authors conducted a choice experiment (DCE) to investigate the benefits of the influence of river restoration on floodwater detention (and related flood risk reduction) and on the river’s nutrient retention capacity (i.e. water quality). The study was across 3 countries: Austria, Hungary and Romania. Despite the greater socio-economic differences between Austria and the other two countries, transferability of the non-monetary benefits appeared to be feasible only between Austria and Romania. The importance of studying the transferability of the environmental benefits is also highlighted in a 2016 study by Lizin & Al. (Lizin, 2016), where spatial heterogeneity is tested as well as the transferability of river specific utility functions. Outcomes were increasing transfer errors when ignoring heterogeneity, but transferable river specific utility functions.

2 ECONOMIC ANALYSIS OF BARRIER REMOVAL

For thousands of years, millions of different kinds and sizes of river barrier have been built globally to i. a. alter freshwater courses from their natural itinerary and flow regimes, with the intention to benefit various aspects of human society. The construction of large dams and reservoirs in particular, has increased markedly in developing countries over the past 60 years, driven by the need to spur economic development and reduce poverty. The construction of large dams has significant positive and negative consequences on population and productivity, and the distribution of costs and benefits across population groups are issues that remain current (Duflo, 2005). Furthermore, the presence of approximately 50,000 large dams currently in operation worldwide – a large dams is defined as higher than 15m - resulted in the cumulative fragmentation of nearly half of the world's rivers, markedly disrupting their ecological connectivity and ability to sustain fish life. (Lehner, 2011).

For many of the existing river barriers, the initial reasons for building have ceased, or have evolved, or are in the process of evolving, due to changed and changing circumstances. Most of these reasons have an anthropogenic nature, including excessive sedimentation, insufficient inflows and increased safety risks posed by meteorological disruptions and climate change, more appropriately referred to as "climate chaos". At the same time, the same changes could also create the need for building new dams, e.g. for irrigation and drinking water. While data from some studies suggests that global temperatures might be on track to rise approximately 3°C above pre-industrial levels by the end of this century, new research is indicating that collapsing ice sheets at the poles may be ramping up the global impact of climate change (Gabbatish, 2019). The analysis published by an international research team predicted an era of climate chaos as more meltwater gushes into the oceans, producing a marked impact on weather within decades (Golledge, 2019). Meanwhile, major agencies have recognised the need to prepare for climate-related risks, assessing climate chaos as a "threat multiplier" (UN News, 2019).

Ideally, to undertake an effective quantification of costs and benefits of an existing or yet-to-be-build barrier, it is necessary to regularly assess socio-economic, cultural, political and environmental factors, including climate-related risks, against the performance of the barrier in delivering the intended outcomes, as well as the ecological performance of the river and of the watershed.

Benefits and costs analysis should include evaluation of marketable and non-market values. Including those related to whether the barrier was to be built or kept, and to whether the barrier was to be not-built or removed. Typical marketable values include water supply, irrigation, navigation, hydropower, commercial fishing. Examples of non-market values include flood regulation, water purification, biodiversity conservation, ecological continuity or river waters and outdoor recreation (some of which are ecosystem services). The two groups of values must be jointly considered to assess existing and newly proposed river barriers, so that the economically optimal solution can be identified. The assessment of solutions should include methods to mitigate barrier effects, for example, by implementing effective alternative corridors for a range of impacted species to mitigate the loss of habitat, mobility and ecosystem integrity.

To achieve reliable cost-benefit assessments, reduce the risk of conflicts, and produce socially desirable and environmentally sustainable results, a set of standard guidelines, procedures, and tools is required for stakeholders working on cross-boundary river management goals such as throughout Europe. Such unified approach would also enable linking existing projects and assist with restoring ecological connectivity across European rivers and watersheds.

2.1 Cost benefit analysis

Social CBA is carried out in order to systematically list, evaluate and compare the advantages and disadvantages of any large-scale infrastructure with wide impacts on society. River infrastructure projects of a certain size are routinely subject to such an analysis. This is because they impact the availability of public goods associated with river resources. The government is therefore interested in an evaluation of the consequences of river barrier removal from a societal, rather than a private view point.

In such analysis, for example, the benefits from a river restoration project are compared with the associated costs of barrier removal, as well as the loss of benefits suffered by those who enjoy the advantages of the existing river barrier. Importantly, this is done in a common analytical framework with clearly defined space and time boundaries. Money is used as the common denominator to compare costs and benefits related to a wide range of impacts, which are typically measured in differing units.

Restoration of river corridors to their 'natural status' may provide an enriched range of ecosystem services. While the ecological value of these additional services is often clear to ecologists and natural scientists, the societal value is often unknown. This is an obvious shortcoming in the information supporting river restoration policy and decision-making.

Collecting information on the value to society of ecosystem services is a challenge that the economics profession has only just started to tackle. It requires a set of skills that are quite different from the classic skills used in conventional economic analysis based on market data and market transactions. This is because many of the benefits and cost associated with river barrier removal are non-market. That is, they are not transacted in properly functioning markets, because such markets do not exist.

For example, it is unclear what market can value the change in sediment deposition patterns, or the increased movement of fish, to mention only two. This absence of market information on relevant economic values may more strongly penalize the completeness of information on the side of the potential benefits.

The results of this analysis can be interpreted as a B-C ratio, that is, total benefits divided by total costs, where a ratio larger than one indicates that the policy measure is beneficial from a social point of view and hence yields a welfare improvement. A CBA compares the costs and benefits of different restoration options in monetary terms. Strictly speaking, only those costs and benefits are included in a CBA that can be quantified in monetary terms. This is usually where most problems start for river restoration project appraisals since many effects, in particular ecological benefits, are often not priced in monetary terms. For many goods and services provided by restored or natural water resources, there is no market where they are traded, and therefore no market price is available, which reflects their economic value. Hence, it will hardly ever be possible to monetize all impacts all the time. Those impacts that cannot be monetized are therefore often left out of the analysis.

While a textbook CBA requires that all impacts be monetized, in practice, different approaches exist on how non-monetized impacts are included in CBA. Non-monetized impacts, if considered relevant, can for instance be included in a qualitative discussion accompanying the CBA results. In early CBA's, such impacts would have been either ignored entirely, left for a subsequent environmental impact analysis, or monetized only partly. Applying an approach of monetizing impacts where possible, and including them in another form where monetization impossible, marks a deviation from the textbook ideal, but it does not discredit the method as such. Nowadays, there are several economic valuation methods, which allow placing a monetary value on non-marketed goods and services. Including these non-market values in a CBA means that a wide range of environmental goods and services provided by river restoration are explicitly recognized in the process.

Table 1 lists possible scenarios. In each case, conducting a CBA would require different types of market and non-market data.

Market and non-market values of existing and yet-to-be-build river barriers are largely dependent on the functions (often multi-purpose) for which the barrier is designed. This is particularly evident for large dams, where substantial non-market costs originate from the environmental impacts associated with the particular use of the dam. Understanding the environmental and socio-economic impacts related to the different uses for large dams is therefore an essential prerequisite to undertake an effective quantification of costs and benefits of river infrastructures.

Depending on geography, time, economy, and socio-political factors, large dams were built worldwide for two main economic objectives:

- provide water storage and irrigation for agriculture uses; and
- provide reliable cost-effective base and peak load electricity.

Other common drives include: socio-economically benefitial functions like access to drinking water; electricity for energy intensive industries such as mining and aluminium smelters; flood control; navigation; job creation; fish farming; tourism; recreation; but can – especially on the global perspective – also be driven by special self-interests, individual careers and political prestige. The Global Water System Project (GWSP), a joint project of the Earth System Science Partnership (ESSP), 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: the Global Reservoir and Dam (GRanD) database. The current version of GRanD (v1.1) contains 6,862 records of reservoirs and their associated dams, with a cumulative storage capacity of 6,197 km^3. **Figure 4** shows their main use (Lehner, 2011).

Table 1. Different types of market and non-market data required to conduct a CBA under possible scenarios.

Starting point	What	Why	Who benefits (market)	Who benefits (non-market)
No existing barrier	Want to maintain the status quo with no existing barrier	To protect river ecosystem	Eco tourism	Angling club, small boating club, environmental groups
No existing barrier	Want to do river ecological restoration works	To restore river ecosystem	Eco tourism	Angling club, small boating club, environmental groups
No existing barrier	Want to do river engineering works	To protect assets	Households (river and coastal), insurance, local/regional council	
No existing barrier	Want to build a new barrier	To store/divert water	Hydropower, irrigation, navigation (industrial, tourism), aquaculture	Small boating club, flood protection authority
No existing barrier	Want to build a new barrier with the inclusion of adaptation and impact mitigation measures (i.g. fish pass)	To store/divert water and limt the impact to river connectivity	Hydropower, irrigation, navigation (industrial, tourism), aquaculture	Angling club, kayaking club, environmental groups
Existing barrier	Want to reduce the hight of the existing barrier or modify its configuration,	To restore river ecosystem		Angling club, kayaking club, environmental groups
Existing barrier	Want to complement the barrier with adaptation and impact mitigation measures (i.e. fish pass)	To restore river connectivity	Hydropower, irrigation, navigation (industrial, tourism), aquaculture	Angling club, kayaking club, environmental groups
Existing barrier	Want to increase the height of the existing barrier	To store water	Hydropower, irrigation, navigation (industrial, tourism), aquaculture	
Existing barrier	Want to remove the existing barrier	To restore river ecosystem		Angling club, kayaking club, environmental groups
Existing barrier	Want to maintain the status quo with the existing barrier	To store water	Hydropower, irrigation, navigation (industrial, tourism), aquaculture	

2.2 The CBA process

The European Commission promotes the use of Cost-Benefit Analyses (CBA) as an objective and verifiable method to measure in monetary terms all the benefits and costs of a project to society, and make successful investment decisions. This approach is essential for major infrastructure investment decisions, but it is also applicable for smaller projects. CBA is a useful management tool for national and regional authorities for planning barrier removal. Standard CBA is typically structured in seven steps (European Commission, 2014):

1. Description of the context
2. Definition of objectives
3. Identification of the project
4. Technical feasibility & Environmental sustainability
5. Financial analysis
6. Economic analysis
7. Risk assessment.

The following figure illustrates the scope of each step:

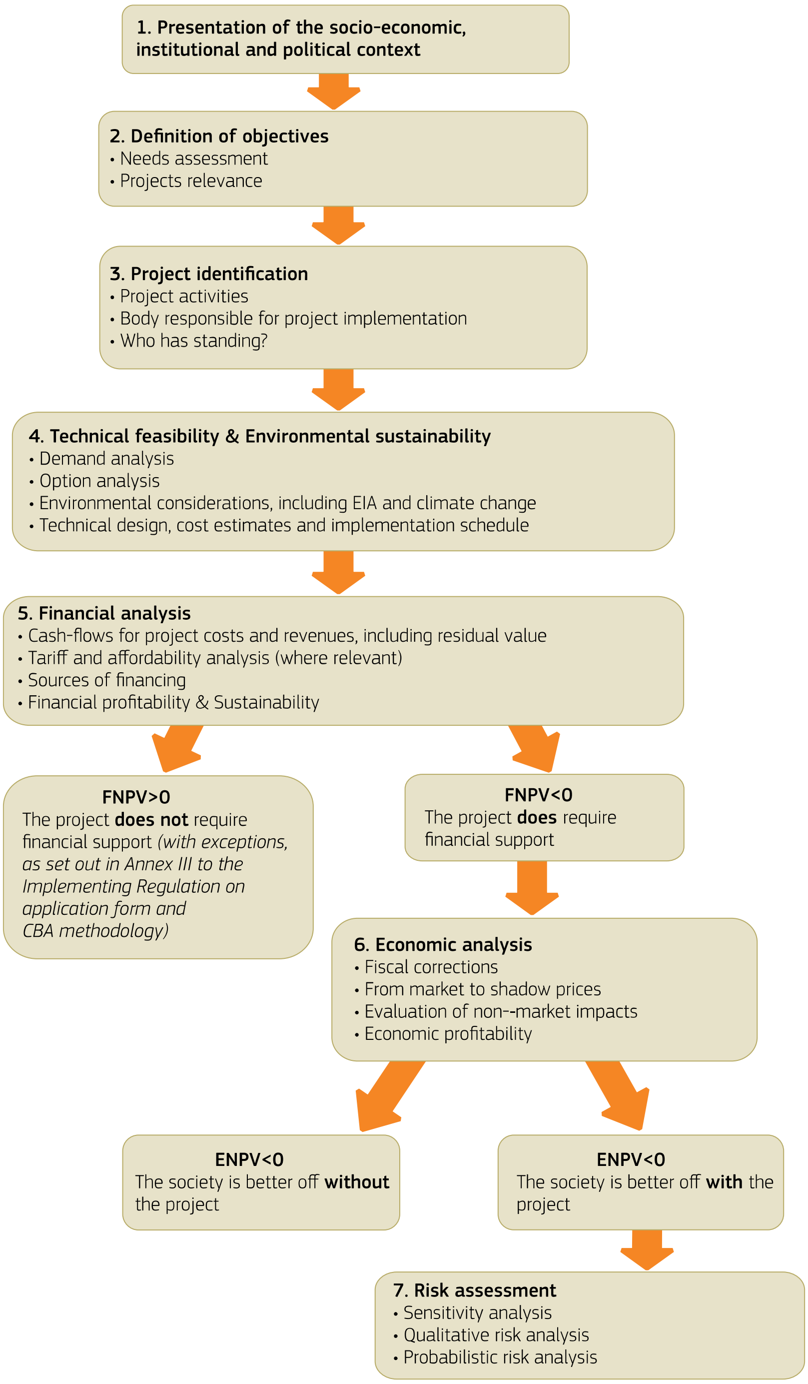

Figure 4. CBA recommended steps (Source: European Commission, 2014)

Typically, a CBA for an environmental project, including environmental remediation, protection and risk prevention, would involve the following type of objectives:

Table 2: Main types of objectives (Source: European Commission, 2014)

	Objectives
Remediation of polluted site areas	- To remove deep and surface pollution or contaminants from environmental assets such as soil, groundwater, sediment, or surface water bodies for the general protection of human health and the environment. - To remove pollution or contaminants from a brownfield site intended for redevelopment.
Preservation of natural assets	- To maintain Europe's biodiversity, for instance by ensuring the ecological coherence and connectivity of the Natura 2000 network[191]. - To safeguard and restore valuable natural ecosystems and assets at a broader landscape level so that they can continue to deliver valuable services to mankind.
Prevention of natural disasters	- To boost natural disaster resilience of the disaster-prone areas more vulnerable to extreme weather events and natural disasters, such as floods, landslides, avalanches, forest fires, storms, wave surge - To support local economies (e.g. in agriculture and forestry sectors) by reducing vulnerability to natural risks, adapting to climate change, maintaining sustainable livelihoods and fostering green growth.

In the case of barrier removals, an additional objective is safety for humans, especially in the case of old dams posing a dam safety risk. The easiest way to measure economic value of an environmental project, is to use any existing actual related market prices. For example, when the existence of a river barrier reduces freshwater fish catches, market values for the lost harvest could be linked to a fish market. When there is no market, the price can be derived through non-market evaluation procedures. This is the case of most of other aspects associated with barrier removals (outdoor recreation, biodiversity and water purification) since no direct market value can be associated with these activities.

The monetary measure of a change in an individual's well-being due to a change in environmental quality is called the total economic value of the change. The total economic value of a resource can be divided into use values and non-use values; i.e. total economic value = use values + non-use values (European Commission, 2014), as shown in **Figure 5**.

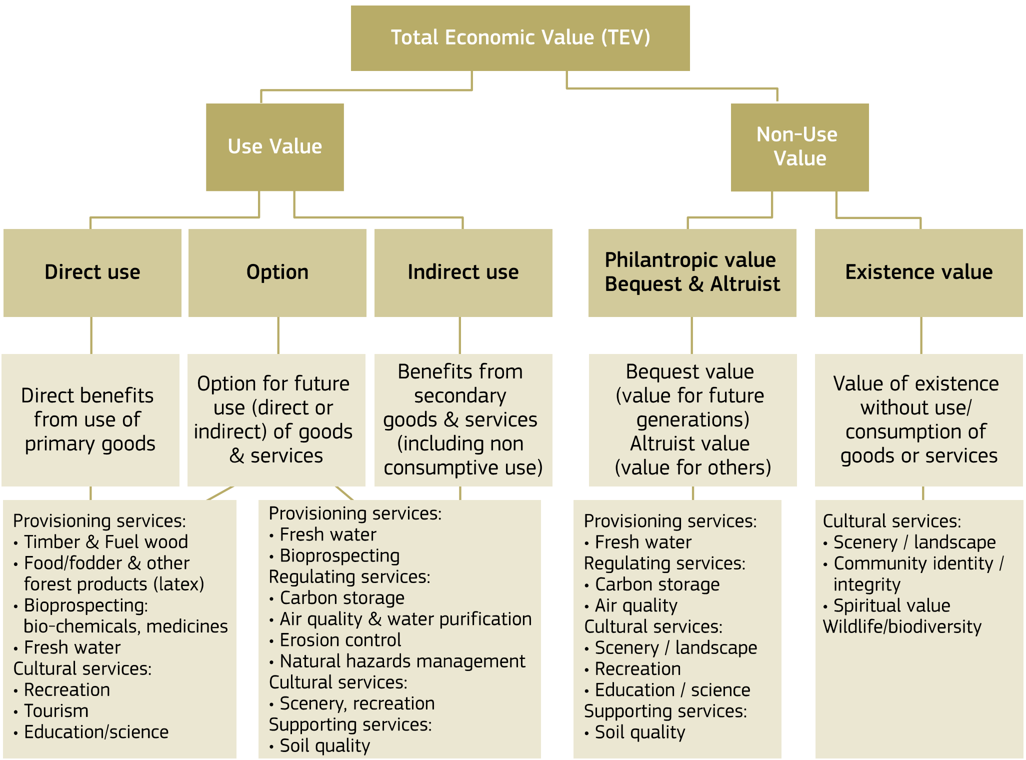

Figure 5: Total economic value (Source: European Commission, 2014)

The recommended approach is therefore to evaluate a proper willingness to pay (WTP) or willingness to accept (WTA) to quantify the cost/benefit for the environment (European Commission, 2014), as shown in **Figure 6**.

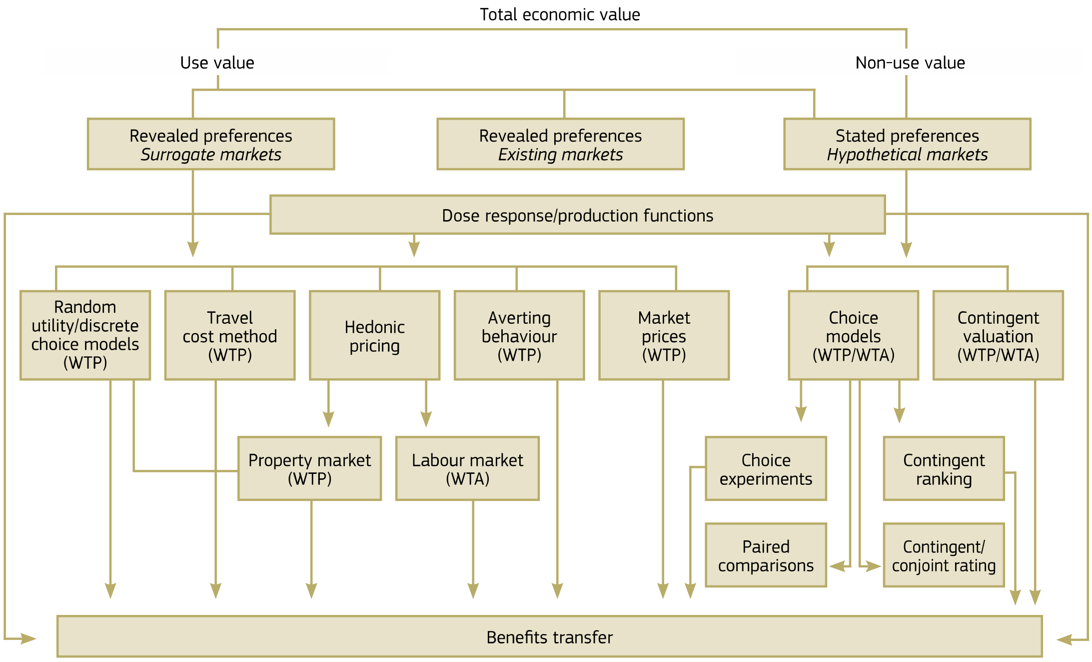

Figure 6: Main evaluation methods (Source: European Commission, 2014)

2.3 Non-market valuation procedures

There are three main methodologies for estimating the WTP: revealed preference methods, stated preference methods, and benefit transfer method. These are described below with an extract from European Commission's Guide to Cost-Benefit Analysis of Investment Projects (European Commission, 2014):

Revealed preference methods
'This approach implies that the valuation of non-market impacts is based on the observation of the actual behaviour and, especially, on the purchases made in actual markets. Consequently, the focus is on real choices and implied willingness to pay. The strength of these approaches is that they are based on actual decisions made by individuals. The main weakness is the difficulty of testing the behavioural assumptions upon which the methods rely. The main specific methods are: hedonic pricing/wage method; travel cost method; averting or defensive behaviour method.'

Stated preference methods
'Stated preference approaches are survey-based and elicit people's intended future or hypothetical behaviour in the markets. Through appropriately designed questionnaires, a hypothetical market is described where the good in question can be traded. A random sample of people is then asked to (a) either directly express their maximum willingness to pay for (or willingness to accept) a supposed change in the good's provision level; or (b) to consider making choices between alternative scenarios that imply payments or compensations.

The main strength of the methods based on this approach is represented by the flexibility they can assure. Indeed, they allow the evaluation of almost all non-market goods, both from an ex ante and from an ex post point of view. Moreover, this methodology is able to capture all types of benefits from a non-market good or service, including the so-called non-use values.

The main specific methods are the following: contingent valuation method, contingent ranking/rating and choice modelling method.'

Benefit transfer method
'This method consists of taking a unit value for a non-market good estimated in an original study and using this estimate, after some adjustments, to value benefits (or costs) that arise when a policy or project is implemented elsewhere. The benefit transfer method can be defined as the use of a good estimate in one site, the 'study site' as a proxy for values of the same good in another site, the 'policy site'. For example, the provision of a non-market good at a policy site could refer to a lake at a particular geographical location. If sufficient data is not available for that country, analysts can use values for similar conditions in data rich countries. The interest shown in this approach is due to the opportunity to reduce the need for costly and time–consuming original studies of non-market goods values. Moreover, benefit transfer could be used to assess whether or not a more in-depth analysis is worthwhile. Clearly, the main obstacle in using this approach is that benefit transfer can give rise to seriously biased estimates, though obviously judgement and insight are required for all the basic steps entailed in undertaking a benefit transfer exercise.'

2.4 Costs of barrier removals

The AMBER project did not include funds for undertaking the primary data collection for an empirical study on the economic valuation of non-market impacts of barrier removal. Furthermore, the benefit transfer method, to value benefits (or costs) that arise when a project is implemented elsewhere, was considered inappropriate due the very limited information available on removed barriers to date, and the uniqueness of each project. Every man-made barrier was put in place in a unique context of legal, regulatory, economic, biological, ecological, demographic and geological constraints. All these factors make the quantification of economic benefits of removal extremely difficult to determine and classify in an uncontroversial manner.

As for direct costs, we were able to approach a number of organisations in Europe and the USA that are involved with river restoration and collect data and insights on expenditures for barrier removal projects. These are presented in **Table 3**.

From the information provided on cost of barrier removal, it was possible to make the following observations:

- The size of the barrier certainly plays a role in determining the cost of a barrier removal project, however the three main game-changers are:
    - the presence of large volumes of sediments behind the dam,
    - the need for infrastructure retrofits, and
    - securing access and temporary land occupation

Sediments
Over time considerable amount of sediments build up behind most dams. This is a concern when the barrier is to be removed, as a sudden release of a large amount of sediment discharge in the river would negatively affect the quality of water and the overall river ecology. The presence of large

volumes of sediments would require expensive management plans, which can significantly escalate the overall cost of removal.

Retrofits
The presence of infrastructure retrofits, such as replacing a bridge, reinforcing a foundation or pier, developing flood control downstream, or replacing intakes, can also result in a considerable increase of cost of barrier removal.

Securing access and temporary land occupation
are additional critical factors that often result in increased cost of barrier removal. This is a well-known factor for planning new dams, but it is often an underestimated aspect in the case of dam removal.

Table 3. Organisations contacted to provide cost information

Country	Organisation	Data description
France	INRA - French National Institute for Agricultural Research	Cost of Vezin dam removal. For now, the cost of the removal of the Vezins dam is about 20M€ (15M€ for the sediment management, to prevent their movement downstream, and 5M€ for the dismantlement itself).
Spain	URA - La Agencia Vasca del Agua	Cost of dam removals in Spain. List of obstacles (GPS layers)
Spain	URA - La Agencia Vasca del Agua	Morphological information of the rivers in the Basque Country (Cantabrian and Mediterranean Sheds). List of obstacles.
Spain	Confederación Hidrográfica del Duero	Cost of Yecla de Yeltes dam removal. The last demolition was the Yecla de Yeltes dam. The cost was 106,207.75€.
Spain	Fish Migration Foundation	List of dams removed in Spain
Spain	Head of the environment department of the Tagus River Basin Authority (Spanish basin)	Cost of the Robledo dam removal (partial budgets only)
UK	Eden River Trust	Tender documentation for dam removal projects
USA	Colorado University	US literature review provides an estimate of total cost - based on the height of the dam. Another critical factor is access. For Elwha dam half of the cost for removal was for land acquisition.
USA	Princeton Hydro - New Jersey	Preliminary study consisting of assessing engineering construction costs for small dam removal projects

2.4.1 Costs of barrier removals in the U.K.

In the U.K., we received information from the Eden River Trust, which is a charity organisation dedicated to improving and protecting the Cumbrian River Eden, its tributaries and lakes. The data provided refer to the removal of six weirs, with the aim to restore natural river processes and sediment movement and open up fish passage for all native Eden fish species (**Figures 7** and **8**).

The work undertaken by the Eden River Trust involved completing several feasibility studies. In some cases, these assessments revealed the reduction of flood risk brought about by the weir removal. Especially strong reduction, if the project included other river restoration activities up and downstream of the barrier. In one instance, the presence of the weir was associated with damage to existing infrastructures including a bridge and water services, making a clear business case for removal. The economic benefits associated with the removal were unquantified, but a summary of the cost of removal is presented in **Table 4**.

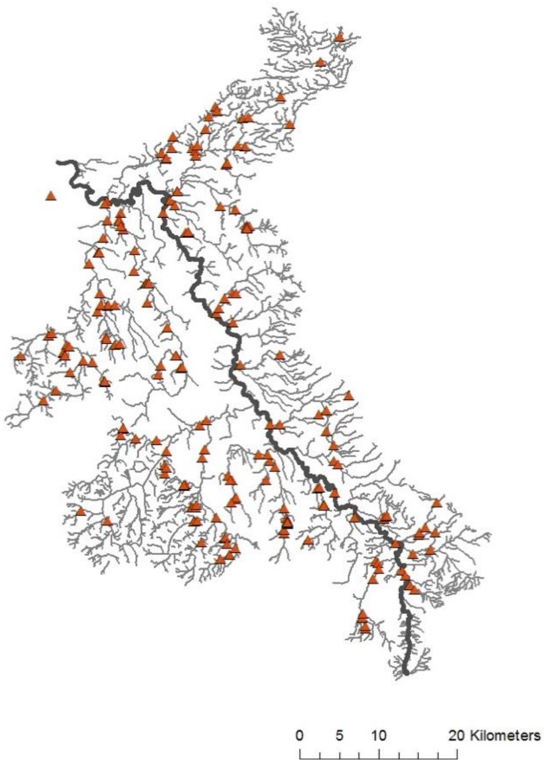

Figure 7. Map of Eden Catchment and barriers. Source: Eden River Trust.

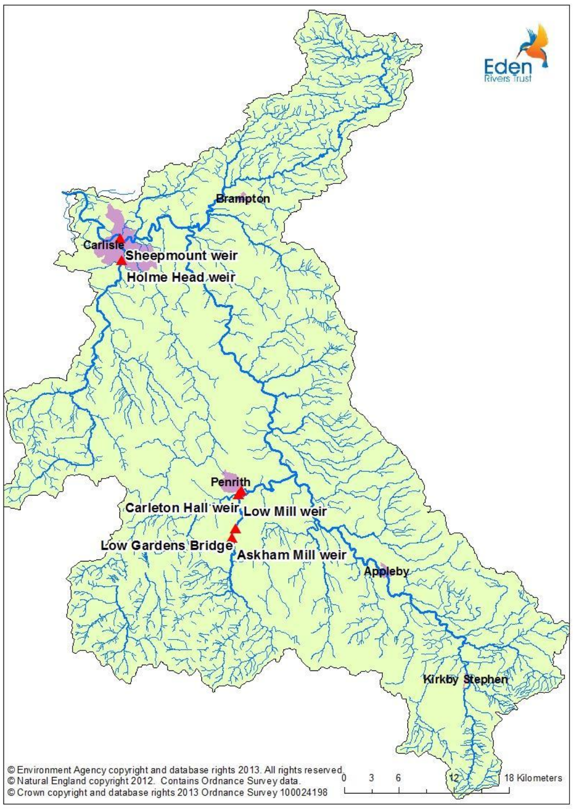

Figure 8. Location of the six weirs to be removed. Source: Eden River Trust.

Table 4. Dam removal projects in the Eden Catchment, UK

Weir name	Main reason for removal	Status	Cost for removal
Sheepmount weir	River connectivity, fish barrier.	Removal completed (30/06/2016)	Total cost of removal works (excluding studies) = £30,454.79 (incl. VAT) + 10k contingency (major part of cost was ground protection for access route (3rd of cost approximatively).
Holme Head weir	River connectivity, fish barrier.	Under assessment.	NA
Corby weir	River connectivity, fish barrier.	Currently planning (dealt by EA).	NA
Jackdaw Scar Ford	Public highways access. River connectivity, fish barrier.	Removal completed (15/01/2017). New design being developed.	Cost for removal £5900.
Carleton Hall weir	River connectivity, fish barrier.	Removal completed (25/08/2016).	Total cost of removal (excluding study) = £29,149 (incl. VAT) + 10k contingency (in case it needed revisiting to install any bank protection).
Low Mill weir	River connectivity, fish barrier. Impact on local flooding. Impact on Eamont Bridge. Impact on local water services.	Planning completed, tendering started.	Yet to be tendered but expected cost is between £200-260k.
Low Garden weir/ford	River connectivity, fish barrier (mainly to native fish).	Planning completed (14/08/2017), tendering started.	Yet to be tendered, but expect cost to come in between £50-70k.
Askham Mill weir	River connectivity, fish barrier.	On hold (depends on results from Low Garderns).	Cost for removal to be confirmed. On hold.
Selside Weir	River connectivity, fish barrier.	Removal completed (12/06/2017).	Unknown.
Applebee	River connectivity, fish barrier.	Unknown	Unknown

2.4.2 Costs of barrier removals in the U.S.A.

In the USA, the National Dam Inventory includes 76,953 dams which are either higher than 6 feet and impound more than 50 acre-feet or are 25 feet high and impound more than 15 acre-feet, but State level inventories list hundreds if not thousands of dams outside these dimensions. For example, Wisconsin lists 3,700 dams of which only 1,200 are listed in the NID. Since 1920, there has been interest in removal of approximately 400 dams, or only 0.5% of the NDI listed dams. This signals that the cost of removal is still higher than the benefits for most of the dams.

Data provided by the association American Rivers indicate that out of the estimated 2.5 million river barriers that were built in the U.S.A. (84,000 of which are classified as dams), more than 700 dams have been removed (**Figure 9**). According to these studies, the main factors influencing price were regional differences in sediment character, adopted sediment management methodologies, post-removal activities and permitting and social issues (Melchior, 2006). Amongst these, the cost of sediment is critical, with management costs ranging from USD 1 – 25/yd^3 (0.76 m^3) in the case of clean sediment, to USD 50 – 500/yd^3 (0.76 m^3) in case of contaminated sediments. (Melchior, 2006).

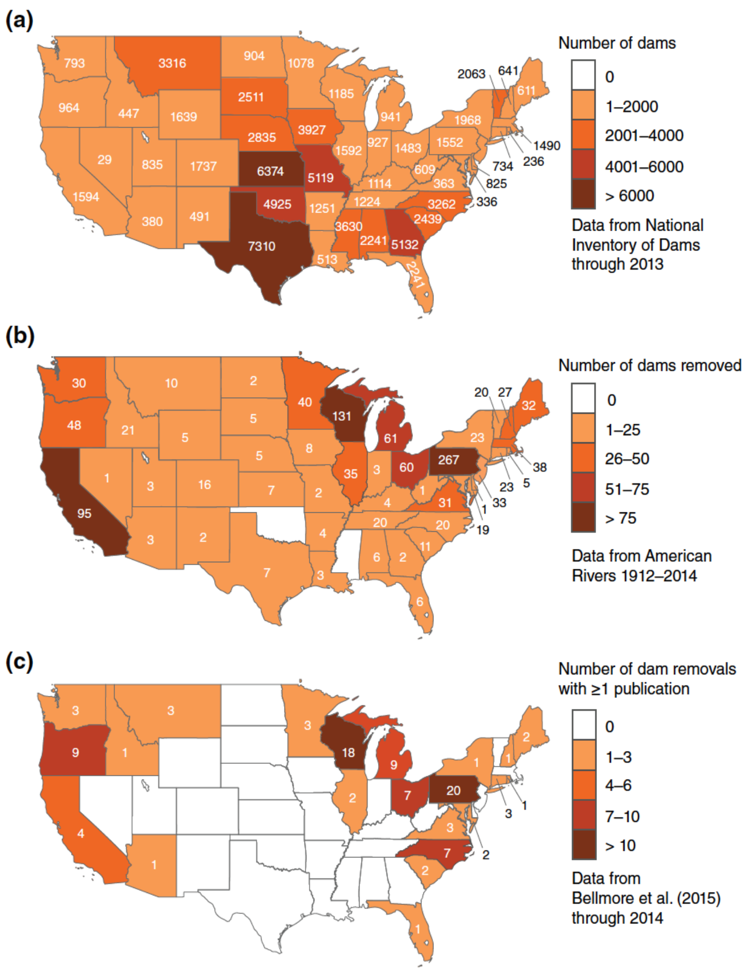

Figure 9. Distribution of dams in the contiguous U.S. (a), the number of dams removed; (b) the number of published dam removal studies, (c) by state. Source: WIRESs Water. DOI: 10.1002/wat2.1164

The cases we studied provide only an overview of the average dam removal cost from projects undertaken in recent years in the Northeast of the US and in Pennsylvania, which are presented in **Tables 5** and **6** (Merchior, 2006)

Table 5. Average removal cost for recent projects in the Northeast and Pennsylvania (Source: Merchior, 2006)

Northeast			
Phase	**Range**	**Mean**	**Number**
Feasibility	USD 9,000 – 236,000	USD 106,000	30
Design/ Permit	USD 9,000 – 188,000	USD 88,000	11
Construction	USD 6,500 – 720,000	USD 144,000	20

Table 6. Average removal cost for recent projects in Pennsylvania (Source: Merchior, 2006)

Pennsylvania		
Dam height (feet)	**Cost range**	**Median cost**
1-3 (<1m)	USD 1,500 – 95,000	USD 17,200
4-6 (<2m)	USD 5,000 – 300,000	USD 38,500
7-9 (<3m)	USD 3,200 – 187,000	USD 45,651
10-15 (<5m)	USD 50,000 – 195,000	USD 70,000
16-25 (<8m)	USD 30,000 – 440,000	USD 117,000

It is worth noting that the numbers for Pennsylvania appear low. This is because the State of Pennsylvania has made strong efforts to keep costs down for this type of projects.

Name	River	Location	Project Stage	Completion Date	Dam Dimensions	Full Dam Removal Cost (includes all infrastucture reconstruction and sediment management)	Volume of Concrete (yd^3)	Estimate for Dam Removal/Mobilization/ Demobilization/Site Work only (took out infrastucture and sediment management)	Total for Dam Removal/Mob/Demob/Site Wk with 30% added for design and 15% for construction contingencies	Total Cost for Dam Removal only in 2018 $ (based on Morgan Friedman Inflation calculator)
Matilija Dam	Ventura	Southern CA	Final Removal Designs in Progress	Estimated 2025	Concrete- 168 ft high and 620 ft wide	(2014 estimated) $35-100 Million depending on option	51,000	19,009,009	$ 28,418,468	$ 69,754,266
Klamath Dam Removal - Copco 1	Klamath	Northern CA and Southern OR	Final Removal Designs in Progress	Estimated 2021	132 ft high (Copco 1)	(2011 estimated) 290 million for all four dams	53,810	23,515,516	$ 35,155,696	$ 86,110,971
San Clemente Dam	Carmel	Central CA	Complete	2015	Concrete-106 ft high x 300 ft wide	(2009 estimated) 83 million	7,500	3,922,840	$ 5,864,646	$ 107,749,530
Condit Dam	White	Southern WA	Complete	2011	125 ft high	(1996 initial estimate) $37 million				
Elwa Dam Removal - Glines Canyon	Elwa	Puget Sound WA	Complete	2014	210 ft High (Glines)	(1991 FERC estimate) $37 million (Glines)				

Table 7. Total cost of dam removal for 5 large USA dams. Source: Princeton Hydro

The total cost of dam removal for 5 large USA dams was provided by Princeton Hydro and is shown in **Table 7**. For three dam removals, detailed cost data was obtained and compiled for infrastructure and sediment management costs (taken out of the total and dividing that total by the volume of concrete) to come up with a present-day unit value for the dam removal, including a 30% contingency for design and 15 % contingency for construction.

Additional data were provided by Princeton hydro, as part of a preliminary study consisting of assessing engineering construction costs for small dam removal projects (<10 m, mostly between 2 and 4m) completed by the company during recent years. This data is presented in **Table 8**.

Further data has been provided by Princeton Hydro for three large dam removal projects in the US, two planned and one already removed (**Table 9**). For each site, the data provides a view of the total cost estimate for removing the dam, and the cost estimates without the infrastructure and sediment management costs. This last value is then increased by 30% to take into account the design costs, and by 15% to take into account the construction contingency. The obtained cost, converted to present day dollars value, is then divided by the volume of concrete to be removed to obtain an indication of present-day unit value for the dam removal by m^3 of concrete removed.

Table 8. Engineering vs. Construction costs for dam removal projects undertaken by Princeton Hydro

No.	Engineering Cost			Construction Cost		% of Engineering Costs (Actuals) of Construction Cost (Actuals)	
	Estimated	Actual	Estimated vs. Actual, %	Estimated	Actual		
1	$7,016	$5,138	36.6%	n/a	n/a	n/a	
2	$16,155	$16,155	Same	$82,124	n/a	19.7%	
3	$50,135	$17,868	180.6%	n/a	n/a	n/a	
4	$22,415	$22,415	Same	$30,011	n/a	74.7%	*
5	$25,200	$31,588	-20.2%	n/a	n/a	n/a	
6	$29,933	$34,293	-12.7%	$253,243	n/a	13.5%	*
7	$52,000	$37,480	38.7%	n/a	n/a	n/a	
8	$43,565	$38,460	13.3%	n/a	n/a	n/a	
9	$52,397	$38,522	36.0%	n/a	n/a	n/a	
10	$45,096	$40,143	12.3%	n/a	n/a	n/a	
11	$74,341	$44,065	68.7%	$157,057	n/a	28.1%	*
12	$49,311	$51,373	-4.0%	n/a	n/a	n/a	
13	$42,930	$57,038	-24.7%	n/a	$174,650	32.7%	
14	$44,367	$57,198	-22.4%	$334,477	n/a	17.1%	*
15	$50,000	$58,210	-14.1%	n/a	n/a	n/a	
16	$80,898	$58,920	37.3%	n/a	n/a	n/a	
17	$65,360	$62,823	4.0%	$156,945	n/a	40.0%	*
18	$61,836	$65,578	-5.7%	$399,743	n/a	16.4%	*
19	$48,703	$65,937	-26.1%	n/a	$283,000	23.3%	
20	$70,027	$68,973	1.5%	n/a	$18,000	383.2%	
21	$84,875	$75,179	12.9%	n/a	n/a	n/a	
22	$70,005	$81,866	-14.5%	n/a	$250,182	32.7%	
23	$101,696	$101,149	0.5%	n/a	$469,000	21.6%	
24	$150,209	$140,219	7.1%	$1,263,777	n/a	11.1%	*
25	$154,797	$155,026	-0.1%	n/a	n/a	n/a	
26	$220,301	$188,304	17.0%	n/a	n/a	n/a	
27	$17,367	n/a	n/a	n/a	n/a	n/a	
28	$41,177	n/a	n/a	n/a	n/a	n/a	
29	$43,122	n/a	n/a	n/a	n/a	n/a	
30	$45,900	n/a	n/a	n/a	n/a	n/a	
31	$80,778	n/a	n/a	n/a	$58,250	138.7%	**
32	$119,987	n/a	n/a	$684,034	n/a	17.5%	***
Average							
	$64,434	$62,074	3.8%	$373,490	$208,847	29.7%	

Data source: Princeton Hydro: * Using Estimated Construction, ** Using Estimated Engineering *** Using Estimated Engineering and Estimated Construction

Table 9. Engineering vs. Construction costs for dam removal projects undertaken by Princeton Hydro (Source: Princeton Hydro)

Name/ Project Phase	Size/ Volume of concrete	Full Removal Cost / Removal Cost* *Mob, demob and site works only, no infrastructure reconstruction works, no sediment management works.	Expected Removal Cost**/ Expected removal cost in 2018 dollars*** **Removal Cost + 30% for design, + 15% for construction contingency. ***Based on Morgan Friedman Inflation calculator	Present day (2018) unit value for dam removal
Matilija Dam/ Final design in progress (estimated completion, 2025)	51 m high, 189 m wide/ 38,99m^3	$35-100 millions, depending on option (2014 estimate)/ $19,009,009	$28,418,468/ $30,899,571	$606/m^3
Klamath Dam 1/ Final design in progress (estimated completion, 2021)	40 m high/ 41,140m^3	$ 290 million for all four dams (2011 estimate)/ $23,515,516	$35,155,696/ $40,246,95	$748/m^3
San Clemente Dam/ Removal completed (2015)	32 m high, 91 m wide/ 5,734.2 m^3	$83 million (2009 estimate)/ $3,922,840	$5,864,646/ $7,019,208	$936/m^3

Data source: Princeton Hydro

2.4.3 Costs of barrier removals in Spain

Tables **10** and **11** report budget data for 38 river barriers removal included in the *LIFE* project *CIPRÍBER*, as reported in the Ministerial report (Gobierno de España (2017)). The 38 barriers, have been identified for partial or total removal.

Table 10. Summary of costs (Source: Gobierno de España, 2017)

Item	Cost in Euros	Sub-cost	Notes
Barrier removal	**395,893.42**		
Access		25,300.15	
Construction works		270,367.65	
Landscape restoration works		29,890.36	
Road restoration works		5,157.50	
Environmental control		65,177.76	
Services	**178,747.00**		It does not seem to include works for reinstating services such as gas, telecommunication, water
Works signs		63,301.54	
Fish control		62,595.12	
Ecosystem services study		31,020.95	
Final communication		21,829.39	
Sediment management	**22,694.57**		Seems too low
Non contaminated sediments		7,972.51	
Contaminated sediments		11,817.24	
Filtration and separation tanks		2,904.82	
Health & Safety	**20,022.76**		
Total planned for works	**617,357.75**		
Indirect costs (16%)		98,777.24	
Profit (6%)		37,041.47	
TOTAL BUDGET BEFORE TAXES	**753,176.46**		
Taxes (21%)	**158,167.06**		
TOTAL BUDGET WITH TAXES	**911,343.52**		
Cultural heritage tax (1%)	**6,173.58**		
TOTAL BUDGET WITH HERITAGE TAX	**917,517.10**		

Table 11 presents a description of the barriers to be removed, including an estimate of the volume of material to be demolished.

Table 11. Description of barriers to be removed (Source: Gobierno de España, 2017)

Location	Barrier Number	Barrier height (m)	Barrier length (m)	Volume estimate (m^3)	Notes
Río Agadón (irrigation canal)	1	5.2	19	0.0945	Includes construction of a fish pass
	2	6.5	50	0.0405	Includes construction of a fish pass
	3	2.6	26.25	48.3	
	4	1.4	18	0.0405	Includes construction of a fish pass
	5	1.7	18.6	28.33	
	6	2.5	20	0.0945	Includes construction of a fish pass
	7	2	29.5	35.4	
	8	3.1	35	35.54	
	9	1.2	12	0.0945	Includes construction of a fish pass
	10	1	13	2.077	
Río Agadone	11	3.5	10	11.54	
Arroyo Peones	12	0.4	4	2.1	
Burguillo	13	4	27	302.4	Seem high, although total barrier removal
	14	4	20	26.61	
Camaces	15	2.3	50	140.03	
Gavilanes	16	1.8	30	0.0585	Includes construction of a fish pass
Huebra	17	3	40	0.0945	Includes construction of a fish pass
	18	1.5	20	9.45	
	19	2.5	30	36.75	
	20	2.5	150	0.585	Includes construction of gate and fish pass
	21	4.5	50	89.6	
	22	4	45	125.4	
	23	4	30	76	
	24	2.5	92	2.88	Includes construction of gate and fish pass
Morasverdes	25	1	12	0.081	Includes construction of a fish pass
	26	5.5	57	0.0765	Includes construction of a fish pass
	27	1.5	20	0.0765	Includes construction of a fish pass
	28	1.5	100	0.045	Includes construction of a fish pass
	29	1.5	40	0.1575	Includes construction of a fish pass
Riofrío	30	2.5	90	169.5	Total barrier removal
	31	3.8	14	95.76	
	32	0.3	10	3.45	

Location	Barrier Number	Barrier height (m)	Barrier length (m)	Volume estimate (m^3)	Notes
Ribera de la Cabeza de Iruelos	33	2.5	25	0.375	Includes construction of a fish pass
Rivera de Dos Casas	34	4.5	25	107.46	
Rivera del Maíllo	35	1	12	0.0765	Includes construction of a fish pass
Yeltes	36	1.6	44	-	Includes construction of gate and fish pass
	37	0.9	40	9.45	
	38	1.5	55	25.02	

Statistics		Notes
Number of barriers	38	
Volume estimate (m^3)	1,385.035	
Average volume (m^3)	37.433	
Total cost estimate	€917,517.10	
Total cost estimate per m^3	€662.45	Similar to the US cost
Cost for sedimentation management	**€22,694.57**	Seems too low
Cost for sedimentation management per m^3	**€16.39**	Seems too low

2.4.4 A Spanish case study

Cost for the removal of the Dam of Robledo De Chavela (**Figure 10**).

Background

Location: The dam - between the municipalities of Valdemaqueda and Santa María de la Alameda, about 60 kilometers east of Madrid - is in the middle of the protected natural area of the Basins and Encinares of the Alberche and Cofio rivers. In addition, the area is key to the conservation of species such as the Iberian imperial eagle or the vulture, according to the project's environmental documentation.

Technical data: 22.7 meters high, 60 meters of coronation, 9,000 m^3 of concrete and 220,000 m^3 of reservoir volume.

History: The wall of the dam of Robledo de Chavela on the River Cofio was built in 1968 to create a water reservoir and had been in disuse since 1990, when the Canal de Isabel II took over this service. Since, it has suffered various breakdowns of the drainage structures of the hydraulic structure, which led to the uncontrolled release of sediments into the riverbed in 2012. This sludge spill downstreamed towards the Alberche, where the San Juan reservoir is located. The spill emptied the reservoir and made it difficult to extinguish a fire that burned 1,200 hectares of trees and scrubland in the area; the emergency helicopters had to go further to collect the water to put out the fire. It also caused the development of other serious environmental problems.

The barrier removal project: The Confederation of the Tagus demolished the 23m wall and ended the reservoir. The river restoration work, which includes the restoration of the channel, cost 1.4 million Euros. This amount covered the stabilisation of the accumulated sediments (where willow cuttings and other riparian vegetation have been planted); finishing the reforestation (also with shrub and ash, among others); the two large silt deposits that have been placed at the ends of the old reservoir, demolishing the wall (the focus of the costing reported below) and rebuilding the last part of the channel.

Prices in EURO April, 2014,
Originally compiled by Lidia Arenillas

Total cost of wall removal and demolition 288.671 Euros.

Earth Movements (total €36,969)

- Excavation and relocation of clay soils (9,500 m3) - Unitary Price = 0.72
- Excavation machine load and transport (9,500 m3) - UP = 0.96
- Transport of loose materials in dumper truck (7,500 m3) - UP = 1.85
- Earth laying with backhoe (7,500 m3) - UP = 0.95

Demolition of the Dam (total €84.361)

- Controlled demolition (6,539.66 m3) - Unitary Price = 12.90

Environmental Restoration of the Area (total €75,879)

- Sub-Chapter 3.1 - Layout and soil stabilization 54,057
- Sub-Chapter 3.2 - Supply and planting 5,188
- Sub-Chapter 3.3 - Surrounding (temporary) works 16,634

Demolition material (rip-rap) management (total €81,726)

- Sub-Chapter 4.1 – Chipping waste 67,526
- Sub-Chapter 4.2 - Crushing previously chopped material 14,200

Health and Safety (total €9,744)

- Sub-Chapter 6.1 - Collective safety 4,418
- Sub-Chapter 6.2 - Individual safety 780
- Sub-Chapter 6.3 - Hygiene and well-being 4545

Considering the length of the barrier, the linear cost per m^3 of concrete is €288000/ $m^3$9000=€32/m^3, which compares very favourably with the US data. However, the overall cost of the river remediation project per m^3 of concrete is €1500000/ $m^3$9000=€167/m^3, which still is a comparatively low amount compared to the US data.

Figure 10.The dam of Robledo De Chavela (https://elpais.com/ccaa/2014/02/09/madrid/1391967968_595758.html)

3 AMBER ATLAS AND JRC ECOSYSTEM SERVICES APPROACH

3.1 Introduction

AMBER partners have developed a number of tools to assess the impacts of stream fragmentation across relevant time and spatial scales. In particular, these will be used to estimate the loss of potential habitat for fish population, mainly endangered species and salmonids.

While these tools will provide useful information for assisting with decision making processes, a tool for assessing socio-economic benefits/costs is required for effective adaptive management of river barriers. The DUBS team worked together with AMBER partner Joint Research Centre (JRC) to explore possible ways to combine the AMBER barrier atlas with the JRC existing ecosystem services maps. The idea to combine these two sets of data was advanced as we were looking at developing a tool that could graphically show available socio socio-economic data in relation to the presence of a barrier. The aim is to develop a mapping tool that is able to assess ecosystem services flows across Europe in relation to the presence of river barriers, with the aim of assisting decision makers working towards barrier adaptive management and optimum ecosystem services.

The process of combining ecosystem service flow maps and barrier maps has never been tested before and it requires further development. An initial test has been undertaken as part of this work using flood regulation ecosystem service maps. The preliminary results are presented below.

3.2 AMBER ATLAS

The barrier atlas, produced by AMBER, presents the first attempt to map river barriers at a pan European level, with the aim of planning barrier adaptation strategies. The work, undertaken by AMBER's partner Polytechnic of Milan (POLIMI), consisted of combining data from existing databases, ground-truthed via field surveys run by AMBER partners, to produce the map and estimate the extent of river fragmentation. For the project, artificial barriers were defined as all those manmade structures that can interrupt ecological processes as described by the River Continuum Concept. (Vannote, Minshall, Cummins, Sedell, & Cushing, 1980)

Figure 11, extracted from `A comprehensive assessment of stream fragmentation in Great Britain', recently published by AMBER, provides the first set of results from the atlas, with focus on the UK (Jones, et al 2019). Findings from the study support AMBER's hypothesis that existing barrier databases considerably underestimate barrier density, and suggests that the majority of the rivers and streams in the EU are fragmented with only a small portion (in the case of UK around 3%) are free of artificial barriers.

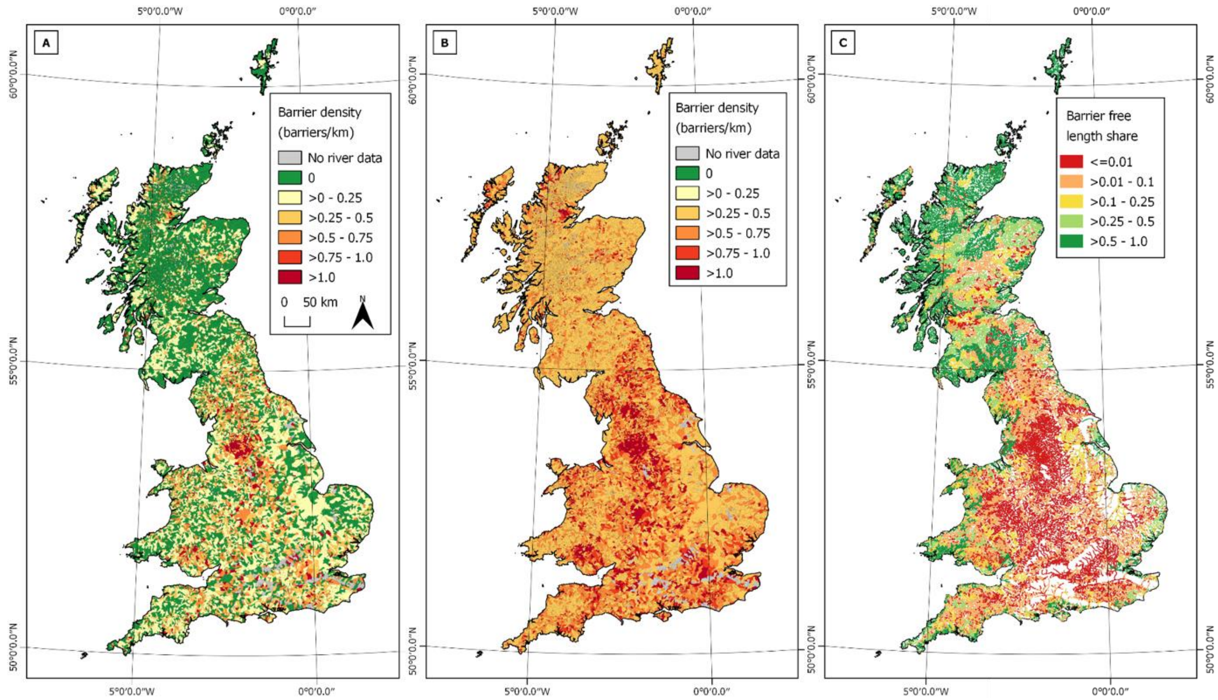

Figure 11. A) Existing records of barrier density (barriers/km) in Great Britain at CCM 2.1 catchment scale (ca. 9 km2) derived from Environment Agency, Scottish Environmental Protection Agency, GRanD and Ecrins barrier databases and OS Open Rivers river network. B) Estimated barrier density corrected by data from field barrier surveys across 19 catchments (303 km). C) Barrier-free length shown as a proportion of total network length in Great Britain based on records of dams and weirs.

3.3 JRC Ecosystem Services Data

JRC produced a number of pan-European ecosystem services maps as part of the European project "Integrated systems for Natural Capital Accounting" (INCA). The purpose of INCA is to generate satellite accounts that can be integrated into the System of National Accounts. The specific task undertaken at the JRC is to develop Supply and Use tables in physical and monetary terms for a series of ecosystem services. INCA accounts are spatially explicit; for most services, biophysical models are employed to assess annual flows of services that are then "translated" in monetary terms, with the aim is to assist planning for freshwater ecosystem management.

3.4 Flood Regulation

Amongst the Ecosystem Services (ES) that JRC worked on, we selected three which are most related with river fragmentation: flood regulation, water purification and outdoor recreation. Out of these, we selected flood regulation to undertake some initial testing which are presented below.

Flooding is a natural hazard that can be of different types: coastal flooding (for example, as a result of exceptionally high tides), flash flooding (for example, as a result of heavy and sudden rainfall which the ground cannot absorb), groundwater flooding (for example, as a result of slow ground saturation process), drain and sewer flooding (for example, . as a result of a blockage or failure within the drainage system), and fluvial (or riverine) flooding (for example, when a body of water exceeds its capacity). Out of these, fluvial is the most frequent and costly (UNISDR, 2011).

Fluvial flooding is controlled by anthropogenic interventions (i.e. man-made river banks and dams) as well as by natural systems (for example, grasslands, forests, shrubs, and wetlands). The ecosystem's capacity to lower flood hazards caused by heavy precipitation events by reducing the runoff fraction is called flood regulation ecosystem service. Enhancing the flood regulating capacity of ecosystems is especially valuable in the identified hot spots of flood regulation demand in central Europe (Stürcka, 2013).

The JRC collected data of flood control by ecosystem in a timeframe of six years (2006-2012), coinciding with years where imperviousness data were available. Although the series is not long enough to perform statistical analysis, data have been compared and interpreted to spot changes relevant for natural capital and policy decision support (JRC, 2019). The JRC studied the role of ecosystem services in flood control, mapping them in specific components:

a) Flood control potential;
b) Flood control demand;
c) Actual ES flow;
d) Unmet demand for flood control.

In the following, the above components are summarised, with a brief insight on the principles that define flood control by ecosystem services (curtesy of JRC).

The approach towards flood protection and ecosystem services is based on three assumptions, or principles:

1) Flood control is delivered at all times and not only during extreme rainfall events: the basis idea is that if an ecosystem is not present, flooding occurs also for rainfall events which have a greater duration or are less intense;
2) The ecosystem service potential and socio-economic demand for the service are the main drivers of changes in the service used;
3) Ecosystems play a key role in flood control by themselves but also provide support to hydraulic structures in place for defence. In this sense, ES flow can be split in two different values: when flood control is provided only by natural capital (NC) and when is controlled by NC and defence measures (NC+).

ES flow of flood control requires the assessment of the ES potential and ES demand to determine the service providing areas (SPA) and service demanding areas (SDA), respectively.

Actual ES flow depends on the spatial relation between SPA and SDA that ultimately depends on the direction of the water flow (slope-dependant): if SPA is upstream of SDA, actual ES flow is verifiedFinally, accounting tables are produced valuating the actual service flow from an economic perspective. This is schematised in **Figure 12**.

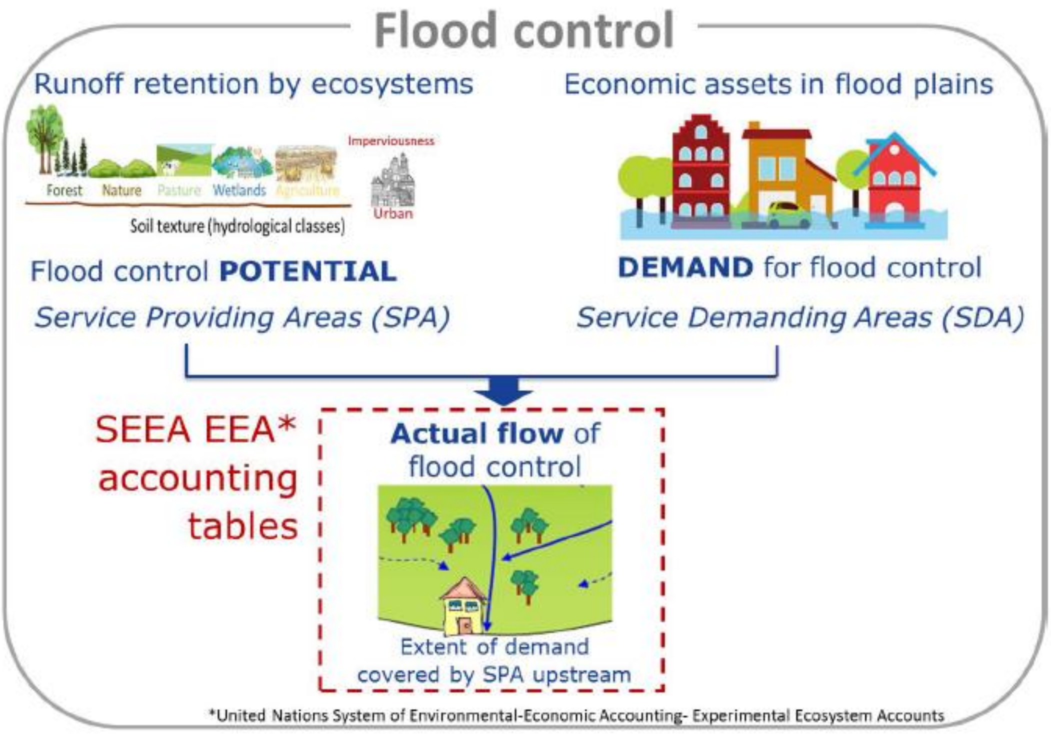

Figure 12. Flood control by ecosystems: main components (JRC, 2019)

Input data adopted in the assessment of ecosystem services and their different components are:

- ES potential;
- demand for flood control;
- actual ES flow.

Spatial analysis has been performed with a 100m x 100m grid cell and population data have been used at a resolution of 250m x250m. To visualise data and results, sub-catchment was used as a spatial reference unit. River catchment data have a size of 180 square kilometres, in line with the Arc Hydro model. (Bouraoui & al, 2009)

All maps and outcomes are produced only when all datasets presented data for the defined sub-catchment.

With regard to ecosystem potential to control floods, a dimensionless indicator was used to determine the extent of SPA per sub-catchment. That is the potential runoff detention, determined in five steps from the Curve Number (CN):

1) CN scoring for land-cover classes;
2) The above CN corrected by imperviousness;
3) Slope is introduced to adjust CN;
4) Natural and semi-natural land covers in riparian zones are integrated;
5) Potential runoff retention calculated as the difference between the maximum CN value obtained for the reference year 2012 and the CN score in a given location.

When focussing on demand for flood control, the attention has been addressed to the extent of the economic assets that can be potentially affected by a 1 in 500 years flood event. Economic assets can be identified as follows:

- Agricultural land (land principally occupied by agriculture that could be vineyards, fruit trees, agro-forestry areas, etc);
- Artificial land (areas with non-agricultural related activities, such as infrastructures, mines, etc)

Also, demand for flood control requires accounting for the amount of population living in the SDA.

In order to determine the actual ecosystem service for flood control, calculated as an area, the following operation has to be computed: the ratio between the upstream surface area covered by SPA and the total upstream surface area (variable between 0 and 1, with 1 indicating that the whole area upstream of the considered grid cell is covered by SPA), namely "Ratio SPAup", has to be multiplied with the grid cell size to calculate the actual ES flow per grid cell of SDA (see equation below).

$$\text{Actual ES flow (ha)} = \text{Ratio SPAup} \, * \, \text{SDAGrid cell size (ha)}$$

The idea behind this equation is that the actual ES flow is based on the runoff depending on the slope of the terrain, which creates the spatial relationship between SPA and SDA. It is then expressed as the number of hectares of the demand (SDA) covered by the ecosystem (SPA) in a given year.

Generally speaking, low values of actual ES flow are expected where the demand for flood control is high (such as highly anthropized areas) and the flood control potential is low (both by ecosystem services and defence infrastructures). As an example, it is possible to see the Po Valley (highlighted in red), highly industrialised with low ecosystem service protection (**Figure 13**).

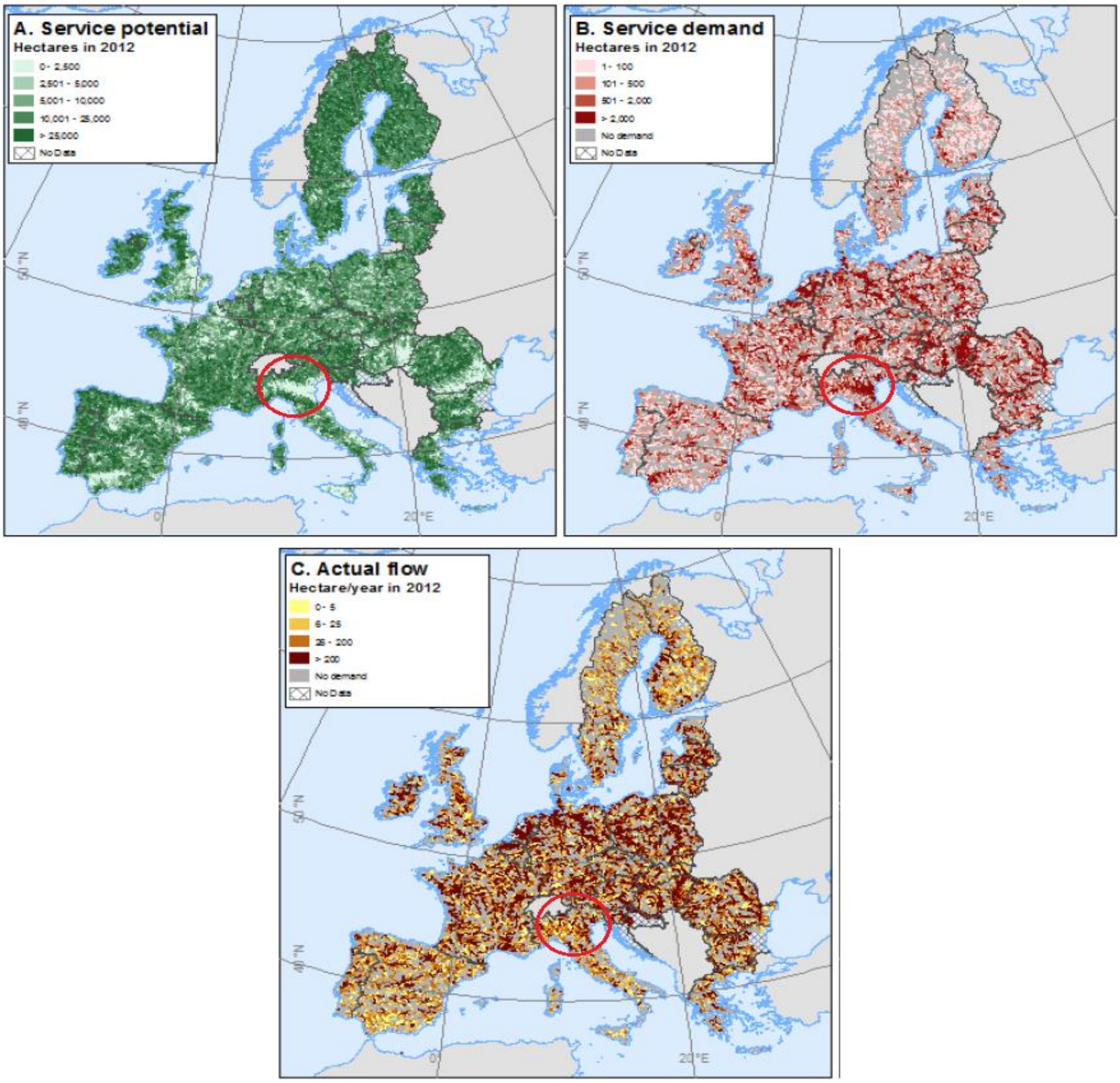

Figure 13. SPA, SDA and Actual Flow (JRC, 2019)

Another important parameter that can be mapped is the unmet demand: it is quantified as the demand less the actual flow (in hectares). It is the economic asset and population that does not receive benefits by the upstream ecosystems (the Netherlands are an exception, being the unmet demand not calculated: defences such as dikes already met the demand for flood control).

3.5 Visual Data Analysis – GIS Software

In order to find correlation between flood ecosystem services and the barriers presence, a graphical information system (GIS) software has been used. Maps have been created using ArcGIS, the software produced by ESRI (Environmental Systems Research Institute). Among other GIS products, ArcGIS allows to share big data projects in clouds managed by ESRI itself (ArcGIS online).

To find visual correlation between barriers in a catchment and ecological services provided in flood control, a two-pronged approach has been proposed:

- Searching for singularity in the barrier distribution and plotting beneath the JRC datasets previously described: this allows to have a barrier focused point of view that is unbiased from the ecosystem datasets;
- Searching for singularity in the JRC datasets and plotting above the dam layer. Dual with respect to the above-mentioned approach, this allows to focus on ecosystem services provided without being influenced by dam presence;

The first approach can be described by looking at **Figure 14**, where the barriers along the Trent River (UK) appear to be equally distributed in the last part of its length, highly affecting the river itself. On the other hand, east of the Trent River there is a particularly wide area (sub catchment 12000696 in particular, but also 12000815, 12000777, 12000757, 12000724) with sparse barriers. The second step Is plotting SPA, SDA, actual ES flow and unmet demand to understand whether there is correlation or not with barriers distribution and ecosystem services.

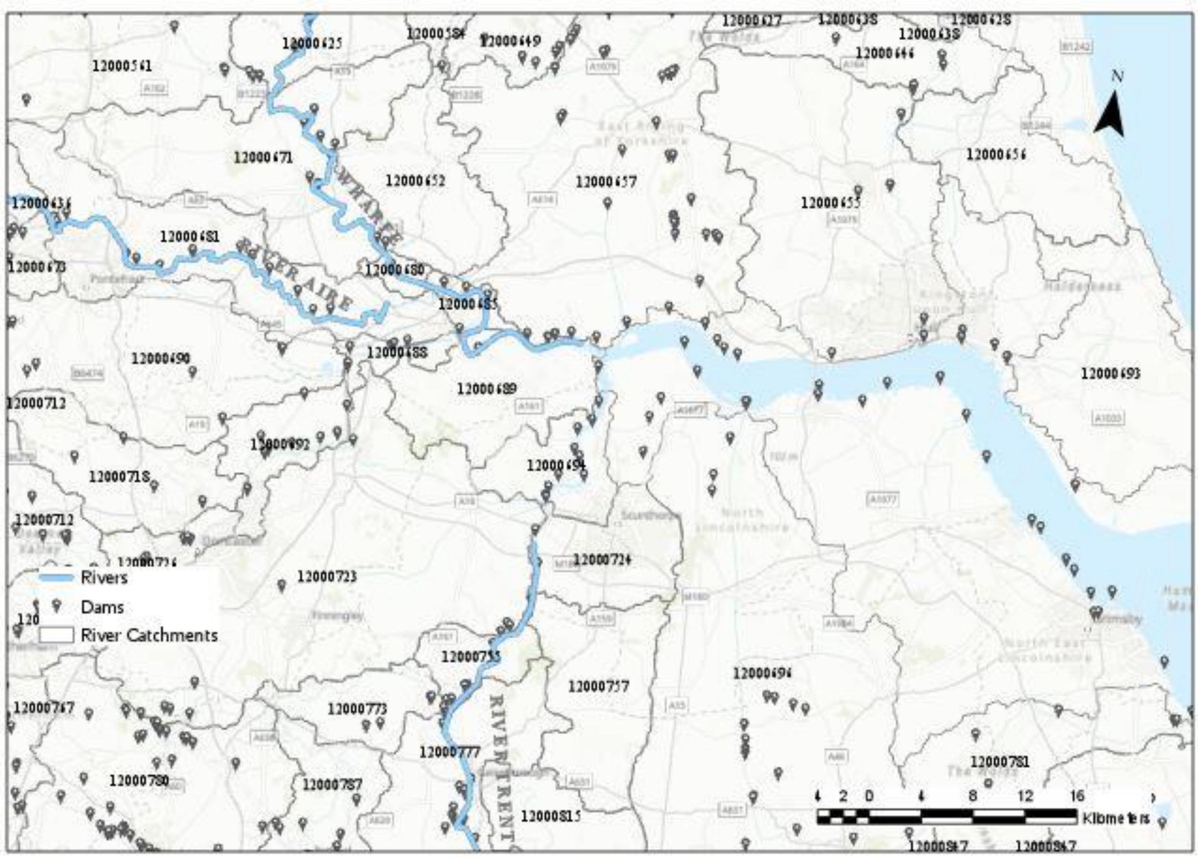

Figure 14. Dam singularity along the Trent River

The following figures represent the very early attempt to correlate the presence of barriers, and fragmented rivers, with ecosystem services, for the same target area in UK (**Figure 15)**.

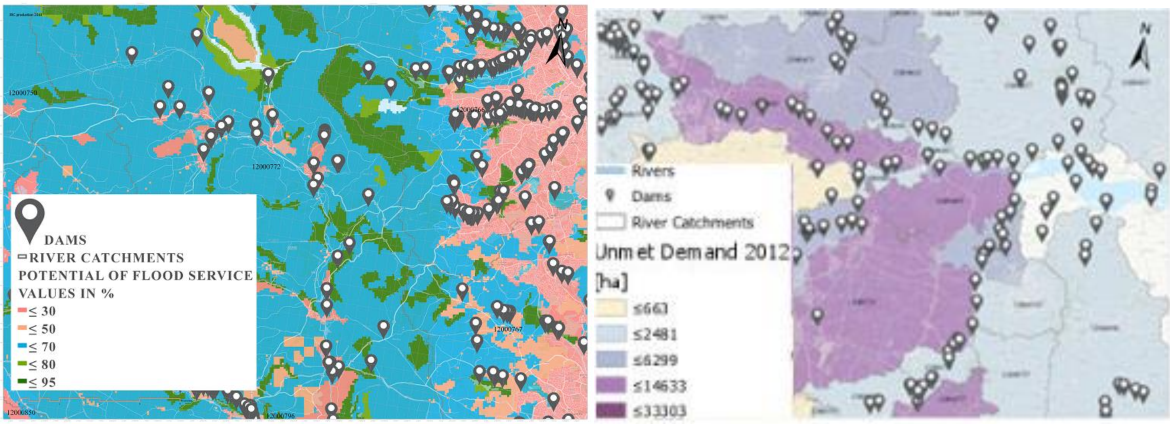

Figure 15. Correlating the presence of barrier with Potential of Flood Service, 2012 (left), and Unmet Demand, 2012 (right)

From the testing, areas with high potential of flood service were identified. Those can be seen in **Figure 16**, **17** and **18**.

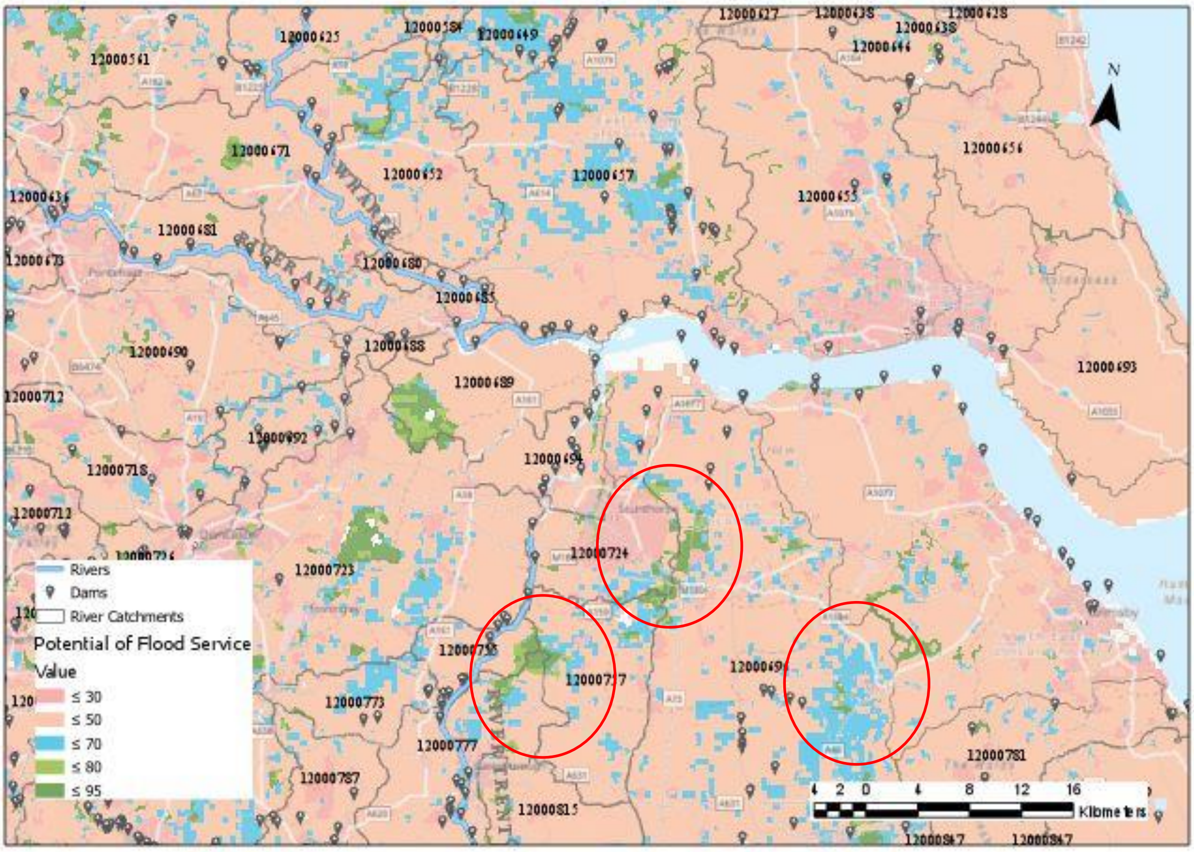

Figure 16. SPA in the test area (%)

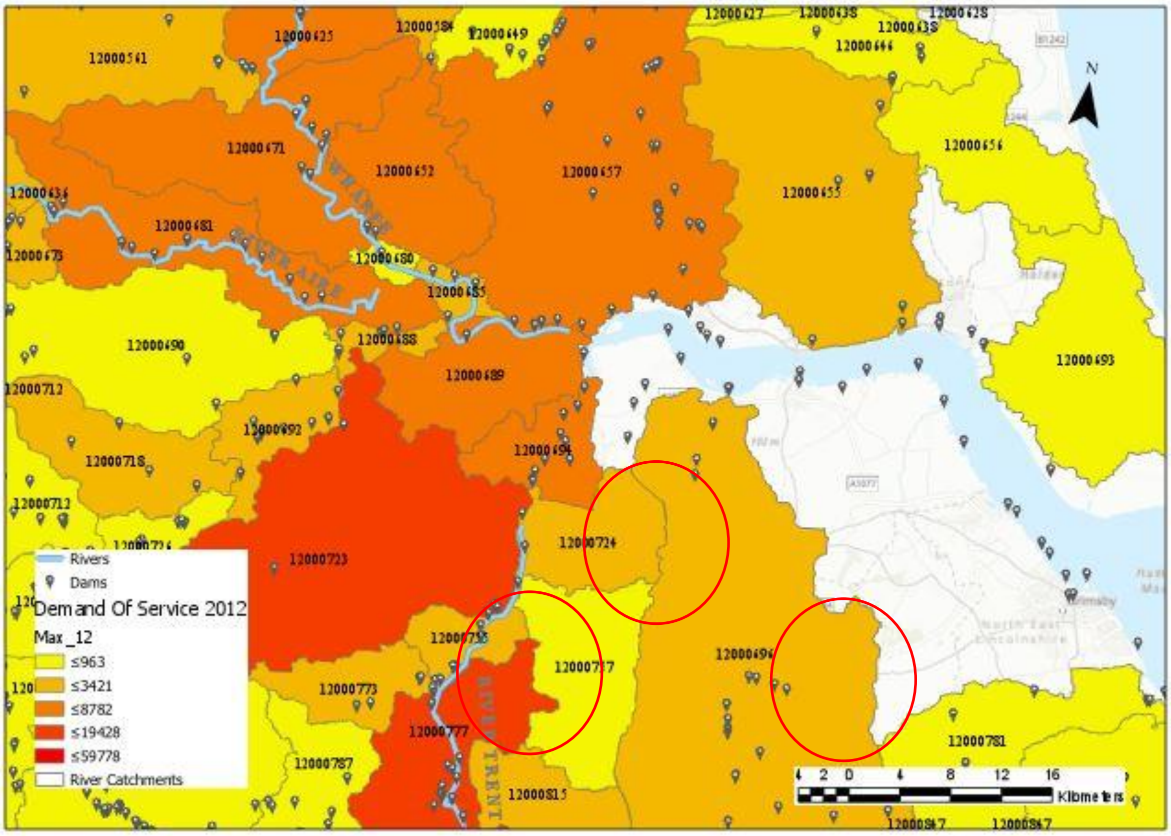

Figure 17. SDA in the test area (ha)

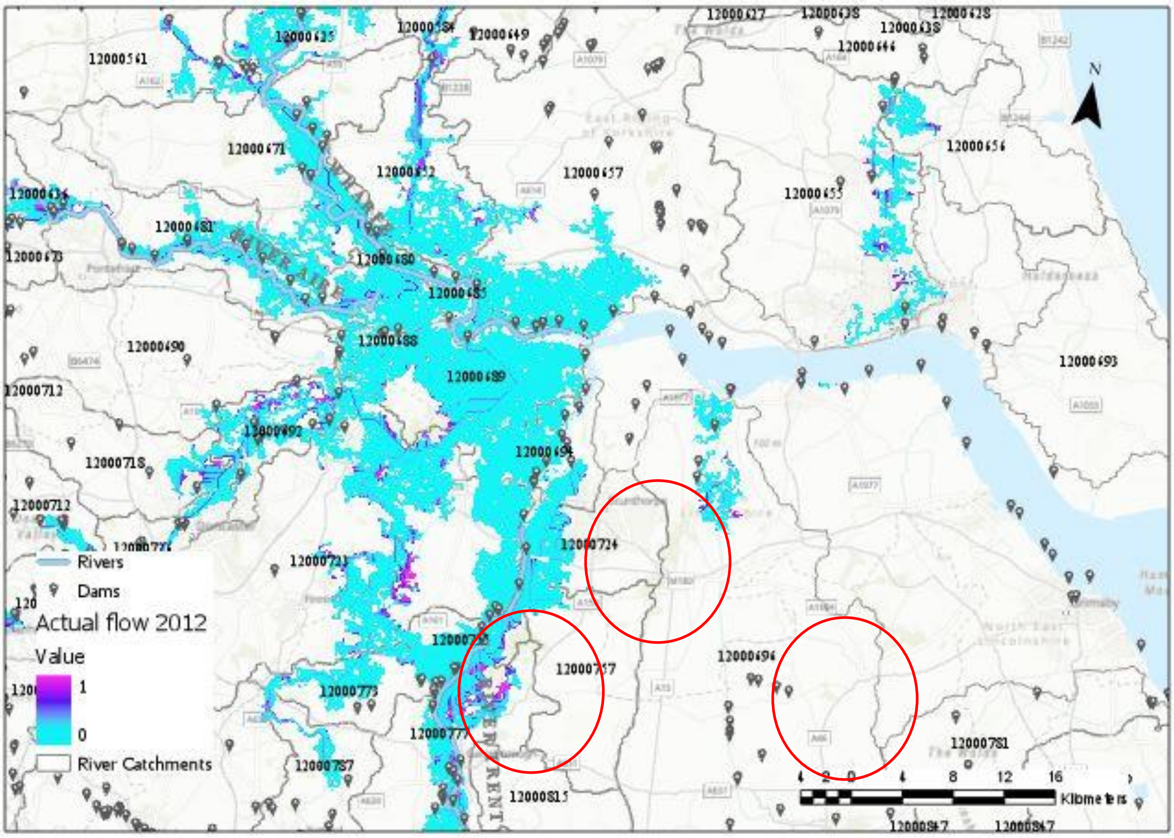

Figure 18. Actual ES flow in the test area (0-100%)

Looking at the downstream areas of the highlighted high potential zones (red circles) in **Figure 17,** and mainly in the 12000696-sub catchment, it appears that, although SPA is significant, no actual flow is provided. Therefore, it seems feasible to question what the role of the barriers is in that area: if not for flood defence, are those barriers useful for some reason in that area, maybe for navigation? If not, it is possible to start thinking about a dam removal to enhance ecosystem services.

However, it has to be notedthat the area of interest is characterised by a not particularly high demand of flood control service (SDA in **Figure 18**) that is reflected in the unmet demand in **Figure 19**.

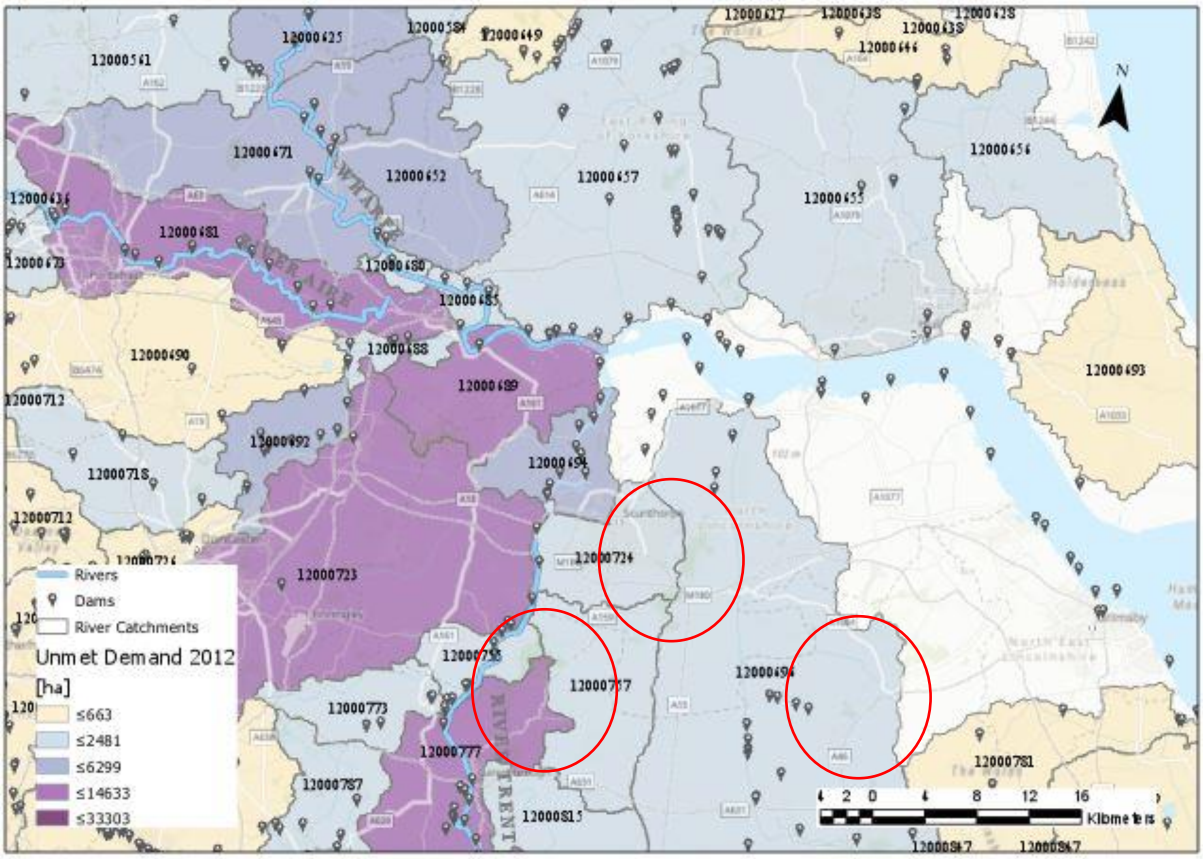

Figure 19. Unmet demand in the test area

The second approach consisted of looking at the SPA in Emilia Romagna (Italy) in **Figure 21**. What grabs the attention, focussing on the JRC SPA map, is the clear distinction between an area of high ecosystem service potential, namely the Apennines region (lower part of the image) and the low potential lowland of Emilia Romagna. The sub catchments drain the water in a south-to-north direction towards the Po valley. In dark blue is the River Po.

Starting form this observation, it is possible to analyse the other JRC maps and their relationship with barrier presence.

It is expected that those areas with lower SPA are those with higher SPD, and that can be seen in **Figure 20**. When looking at the barriers, the density in the southern sub catchments of **Figure 21** is very high and immediately raises questions about their role in the sub catchment (**Figure 22**). In further research It would be worth investigating if dams are impeding to provide service downstream.

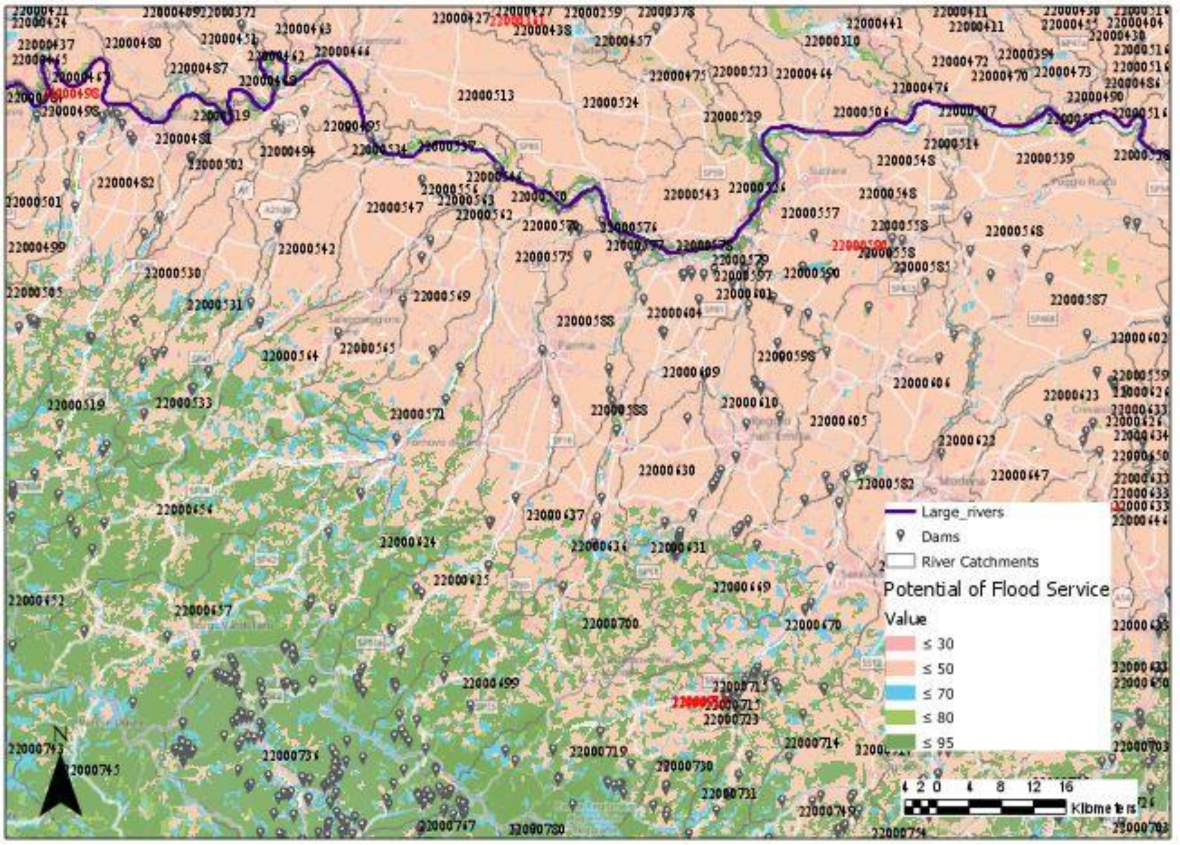

Figure 20. SPA in Emilia-Romagna (Italy)

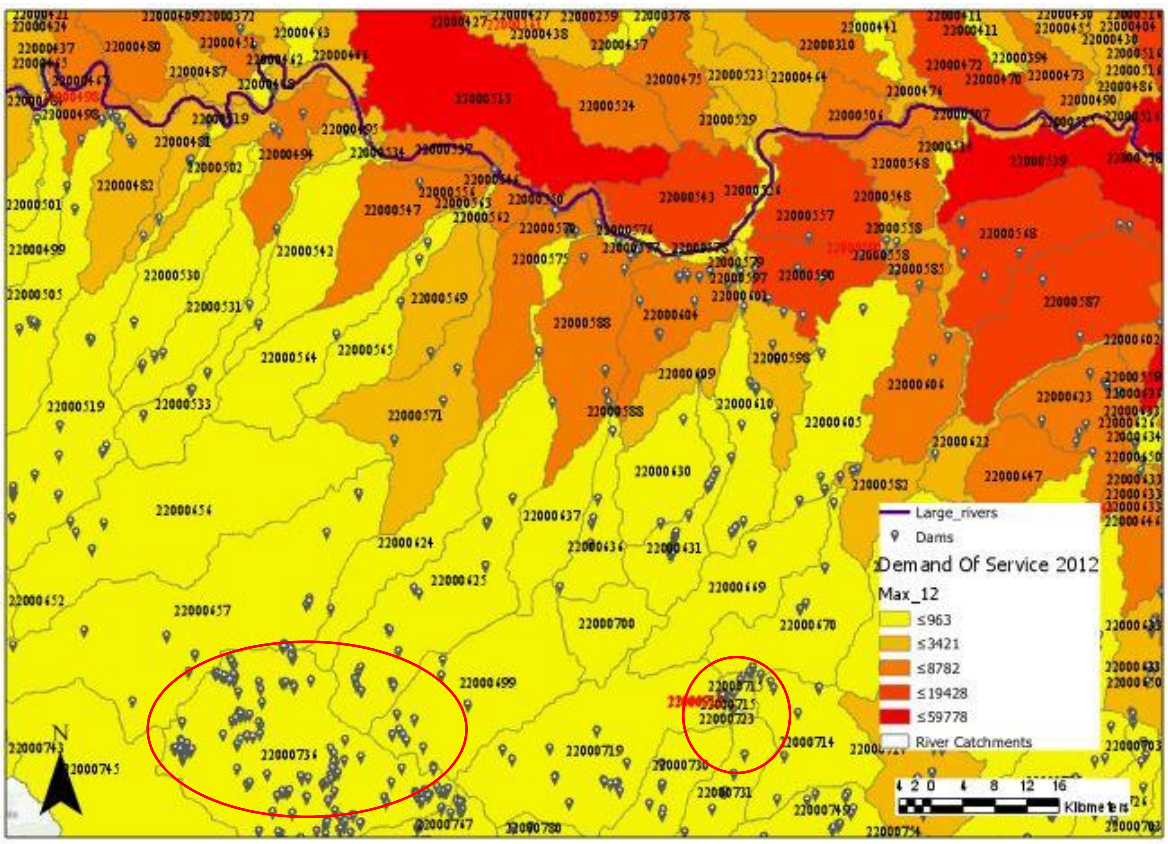

Figure 21. SDA in Emilia-Romagna (Italy)

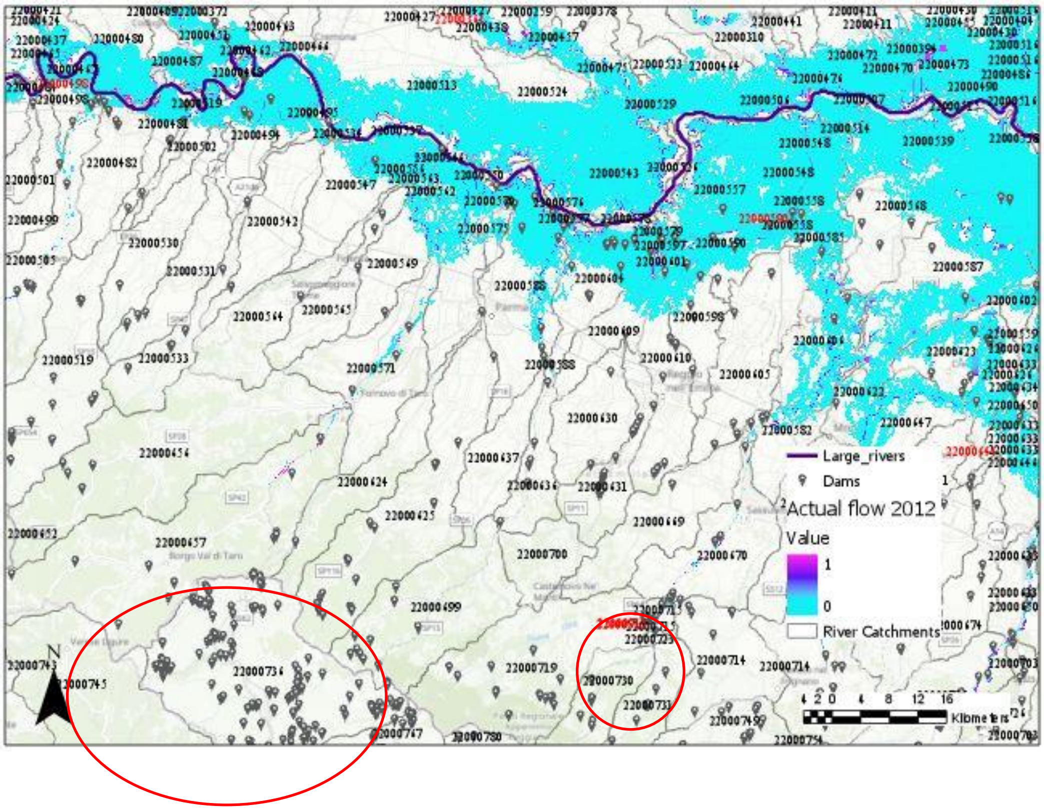

Figure 22. Actual ES flow in Emilia-Romagna

Figure **Figure 23** shows that there is no particular actual flow and therefore, as in the previous example, one can question whether the barriers are significant in terms of flood protection downstream or not.

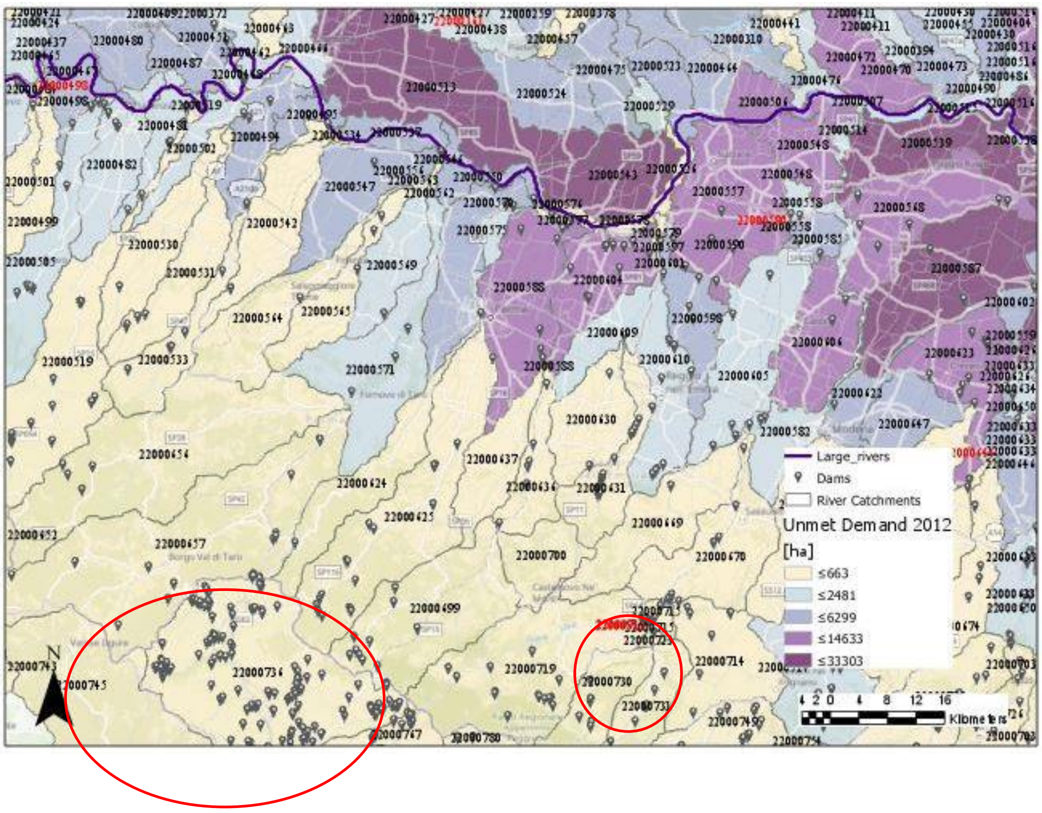

Figure 23. Unmet demand in Emilia-Romagna

As shown in **Figure 23**, unmet demand increases towards the Po valley, starting from low values in the Apennines region. This is expected, due to the drastic difference in land usage and coverage, however the highlighted areas (red circles) indicate zones with a possible relationship between high potential of ecosystem services and the presence of barriers, hence a possible case for dam removal.

It must be noted that both the above-mentioned examples refer to the year 2012, which is the reference year for the JRC method.

3.6 Future Development and Potential Use of Algorithms

Every procedure that derives from a rigorous and methodical approach can be translated in algorithm. Potentially, the visual analysis of barrier and ecosystems services in flood control can be aided by a tool that consents to speed up the process by, for example, reducing the number of dams in a dataset if not relevant.

For example, a number of barriers may not provide any flood defence and regulation downstream, and the upstream catchment might be able to mitigate peak flow events. In this instance, these barriers could possibly be removed from the map with an aim to enhance the ecosystem and reinstate environmental assets. However, if the dam in question is for hydropower production, the production of electricity would possibly overrule the flood regulation ecosystem service in monetary terms.

Although yet to be proved, the “hydroelectric dam case” could provide an insight on how a possible algorithm could work: for instance, the conditional statement “IF the dam provides hydropower THEN highlight it” can be written in coding terms, aiding the visual analysis towards a dam that can be removed from the database (being necessary for hydropower generation). As a result, the whole process can be less time consuming and more efficient.

It has to be said that the use of algorithms is effective if the starting datasets (dams and ecosystem services maps) are detailed and clear in their information: catchments carrying spatial information, its area, and the ecosystem services (SDA, SPA, actual ES flow) can be ranked and classified in order to be compared.

The reasoning explained above can be applied to any conditional statement that can be postulated: instead of hydroelectric dams, it is possible to apply to redundant structures for examples.

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