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Abstract

European policy agendas related to mitigation of water related risks and the building of a resilient environment include Nature Based Solutions (NBS). However, the difficulties to evaluate the economic value of NBS can affect their implementation. Meta-Regression Analysis (MRA) function transfer is a low cost solution allowing assessment of the benefits of public or environmental goods. Currently, none of the numerous MRA studies of the economic value of ecosystems and biodiversity focuses on NBS applied in a river basin context. The main contribution of this paper is to provide a first MRA function transfer to assess the value of NBS restoration measures and identify their primary and co-benefits. Our data are based on 179 observations related to 49 studies conducted in America, Europe, and Asia-Oceania since the mid-1990s. Our estimates show that the primary benefit of water regulation is insignificant, but that individuals value co-benefits, such as recreation and aesthetics, and the scenario giving more room for the nature in river restoration. These co-benefits have a significant and positive effect on the Willingness To Pay (WTP). Among NBS measures, river stream is shown to be the most valued. Among the contextual and methodological variables, river length, river location, Contingent Valuation Method (CVM) and national tax vehicle are significant and robust. The median error rates related to our MRA are around to 17% which suggests that the value transfer function could be a cost effective policy tool to enable economic assessment of NBS restoration measures in a river basin context.

Keys words: Nature Based Solution (NBS), Meta-Analysis Regression, Ecosystem services, Willingness To Pay.
JEL Codes: Q51, Q57, O13

Introduction

Over many years, river ecosystems have dried up and/or been damned to allow land development, which, in periods of heavy rainfall, can cause costly and even catastrophic flooding. At the same time, higher temperatures have led to drought risk, which is equally damaging. The current ongoing climate change is highlighting the need for Nature Based Solutions (NBS) to reduce risks and build a resilient environment. In contrast to alternative solutions, which "are designed and managed to be as simple, replicable and predictable as possible" (Eggermont et al., 2015, p. 243), NBS are complex solutions that address the complex problems of socio-ecological systems resilience, climate change and sustainability (Nesshöver et al., 2017). NBS is an umbrella term (Cohen-Shacham et al., 2016), which includes several ecosystem-based concepts. Research is needed to allow a better understanding (Nature, 2017) and better evaluation of these solutions. Also, policy makers need reliable estimates of the impacts of NBS to inform their decision making, while NBS to reduce climate risk must be cost-effective (Meyer, 2016).

However, assessing the benefits of NBS is not straightforward. First, Nesshöver et al. (2017, p.1221) show that NBS provide "substantive", "instrumental" and "normative" benefits. In addition, they increase the socio-economic benefits derived from Ecosystem Services (ES) (e.g., biodiversity and recreational services) and adaptions to reduce climate change and increase food security (Eggermont et al., 2015; Cohen-Shacham et al., 2016; Maes & Jacobs, 2017; Raymond, et al., 2017; Albert, et al., 2019). Second, most of these ES, governance and societal benefits have public good characteristics and, therefore, are not tradeable on the market. Thus, different market and non-market valuation approaches are needed to assess the economic value of the benefits induced by NBS (De-Groot & co-authors, 2012; Grizzetti et al., 2016). Since such approaches are costly in terms of competences, time and money, policy makers and academics are interested in transferring the value from existing evaluations to implement new primary surveys (Brander et al., 2012; Chaikumbung et al., 2016), to allow assessment of benefits of NBS, at lower cost.

The literature includes numerous studies that employ Meta- Regression Analysis (MRA) to estimate the economic value of ecosystems and biodiversity. For example, Chaikumbung et al. (2016) report the findings from 17 valuation meta analyses for wetlands and Barrio and Loureiro (2010) present the findings for 4 forest land studies. Brander & Koetse (2011) and Perino et al., (2014) conduct a MRA on green infrastructures in urban areas while Brouwer

(2017) and Bergstrom and Loomis (2017) focus on river restoration projects. However, none of these works specifies the type of NBS evaluated in the primary studies. Smith and Pattanayak (2002, p. 282) highlight that, in the context of MRA function transfer "the central issue that distinguishes the studies is consistency in both the measure of value summarized across studies and in the environmental commodity or service". Therefore, it is important to consider similar NBS measures to achieve consistency (Smith & Pattanayak, 2002; Bergstrom & Taylor, 2006) and reduce generalization errors in the MRA (Rosenberger & Stanley, 2006). To our knowledge, only Bockarjova and Botzen (2017) and Bockarjova et al. (2018) apply the concept of green infrastructure to study different types of NBS in an urban context. No published research focuses on NBS restoration measures in a river basin context. We try to fill this gap by (i) identifying and classifying NBS restoration measures in primary studies; (ii) considering a large number of NBS benefits in order to include a wider set of ES; (iii) illustrating the potential of our MRA function transfer to evaluate NBS river restoration measures on the Brague River basin in France.

The objective is to conduct a MRA to derive value functions to evaluate different NBS restoration measures and their benefits. Our dataset is built on 179 observations from 49 studies, conducted in America, Europe, and Asia-Oceania from 1996 to 2018. We analyse and identify the determinants of the Willingness To Pay (WTP) for NBS, based on primary studies that use stated preferences such as Contingent Valuation Method (CVM) and Choice Experiment (CE).

The paper is organized as follows. Section 1 discusses the analytical framework. Section 2 describes data selection, standardization, and coding. Section 3 presents and analyses the results of the MRA models. Section 4 discusses our MRA function transfer in the context of evaluation of NBS restoration measures on the Brague River basin. Section 5 discusses the findings and concludes.

1 Methodology

1.1 NBS or not: specifying the object of analysis

Adopting a value transfer approach to evaluation of NBS requires identification of the types of NBS evaluated in primary studies. This respects the requirement for commodity consistency[1] in benefit transfer studies and reduces generalization errors[2] (Smith & Pattanayak, 2002; Rosenberger & Stanley, 2006). However, NBS are defined in different ways in relation to different ecosystems (Annex 1). According to Albert et al. (2017), the vagueness of the notion of NBS requires scientific input (Nature, 2017). It is important to define a similarity criterion since the term NBS is an umbrella term and emerged at the interface between science, policy, and practice (Nesshöver et al., 2017).

Based on the literature review and the various definitions of NBS (Annex 1), we identified three important characteristics of NBS. First, they try to mitigate a targeted risk. The context in which NBS are applied - finding a solution to climate change - is important. The effects of climate change (e.g., extreme weather) are increasing the risk of "abrupt and, in some cases, irreversible environmental changes detrimental to human development" (Eggermont et al., 2015, p. 243). Andersson et al. (2017) consider risk reduction to be the main objective of NBS.

The second characteristic is that they exploit nature (at least in part), to achieve self-regulation of ecological functions. Albert et al. (2019 p.14) suggest that "NBS must fulfill the criteria of challenge-orientation, ecosystem process utilization, and practical viability". Eggermont et al. (2015) suggest that NBS should be designed to address species diversity wherever possible. They highlight, also, that "clones from one or very few plant species could increase the risk of biological invasion and lead to poor resistance and resilience to future extreme events" (Eggermont et al., 2015, p. 245). These studies hightlight that exploiting nature benefits society through the provision of ES and biodiversity.

The third characteristic is involvement of stakeholders to allow the mapping of different dimensions of risk and consideration of multi-interests and conflicts. Stakeholder involvement is essential for the design, implementation, and management of appropriate, efficient and socially acceptable NBS. Society'speople's perceptions of NBS and their management can

[1] This requirement is satisfied by the inclusion of estimated values for goods and services that are similar across studies.
[2] Generalization errors are related inversely to the degree of correspondence between study and policy location.

determine their long term use (Andersson et al., 2017). NBS require flexible and open-ended management (Andersson et al., 2017, Raymond et al., 2017, Kabisch et al., 2016).

In this study, the interest is in natural measures to mitigate the risk of river floods by "altering the catchment scale runoff regime through the manipulation of hydrological flow pathways throughout the catchment" (Wilkinson et al., 2014, p.1245). River restoration measures differ and are urban or rural context dependent, and may be aimed at several objectives such as improving the hydromorphology, longitudinal connectivity and slowing or storing of water flows (Zingraff-Hamed et al., 2017; Wilkinson et al., 2014). Based on the categorization provided on the European Natural Water Retention Measures (NWRM) platform,[3] we can identify four types of NBS measures (Table 1). They include measures targeting: the river stream (e.g., stream bed restoration, dam removal); the riparian vegetation (e.g., riparian buffers, natural bank stabilization); the floodplain (e.g., wetland restoration, floodplain restoration); and achievement of a good ecological status of the river ecosystem (e.g., ecological management, sewage interception).

Table 1: Types of NBS measures to mitigate natural risks in the context of climate change

Types of NBS measures	Natural risks	Using nature	Stakeholder involvement
River stream	Flood; drought; lost biodiversity	Measures aim to restore the river bed to its (nearly) original state to support biodiversity (e.g. bank re-naturalization, dam removal)	Identifying important benefits for stakeholders from nature.
Riparian vegetation	Flood; Drought; Biodiversity lost	Measures to replant or allow regrowth of vegetation to create an interface between the land and the river (e.g. plant engineering, riparian buffer)	Identifying important benefits for stakeholders from nature.
Floodplain	Flood; drought; biodiversity lost	Measures to provide a natural space for the retention of flood and rainwater (e.g. removing legacy sediment, creating lakes or ponds in the floodplain)	Identifying important benefits for stakeholders from nature.
Others	Flood; lost biodiversity.	Measures to manage the river ecology (e.g., ecological management, sewage interception	Identifying important benefits for stakeholders from nature.

1.2 Involvement of stakeholders to identify stakeholder benefits

According to Raymond et al. (2017), it is essential to engage stakeholders in the selection and assessment of NBS. In the context of economic assessment, the focus, often, is on the impact of NBS on stakeholders' wellbeing, based on the benefits they derive. Several studies see

[3] http://nwrm.eu/

provision of ES as one of the key co-benefits of NBS (Cohen-Shacham et al., 2016; Maes & Jacobs, 2017; Nesshöver et al., 2017; Gulsrud et al., 2018; Albert et al., 2019). We suggest that stakeholder involvement can allow identification of relevant ES to avoid transferring ES that stakeholders do not value.

In the context of river restoration, a change to the natural environment can result in several, mostly non-market, ES benefits (Holmes et al., 2004). In the present study, we adopt the ES framework (Millennium Ecosystem Assessment, 2003) and the Common International Classification of Ecosystem Services (CICES, V5.1, 2017).[4] Primary studies were scrutinized, taking account of seven benefits (see Table 2), from the bundle of ES affected by river restoration measures. These benefits can be classified according to value type and potential beneficiaries. The three types of potential beneficiaries we consider are people living in the river basin area; visitors to the area; and the population worldwide. Some ES provide direct use benefits (e.g., food and materials, water regulation, recreation) and others provide indirect use benefits (e.g., local environmental regulation, global climate regulation, aesthetic properties). There are also non-use benefits (e.g., habitats and biodiversity). ES related to hydrological functions (Brouwer et al., 1999) are a major focus due to our interest in use of NBS to mitigate risk of river flooding. In this study, we consider water regulation benefits as the primary benefit and the other benefits as co-benefits.

[4] https://cices.eu/

Table 2: Benefits of NBS for river restauration

Benefits	ES categories of CICES	Definition	Values and Potential beneficiaries
Water regulation (primary benefit)	Regulating and maintenance	Hydrological cycle and water flow regulation, including drought and flood.	Non-consumption use value for residents
Food & Material (co-benefit)	Provisioning	Cultivated or wild plants, reared or wild animals for nutritional or processing purposes	Consumption use and option value for residents and visitors
Local environmental regulation (co-benefit)	Regulating and maintenance	Resilience of local environment to stresses including high temperature, fires, sandstorms, land salinization, erosion, pollutants, and mass movement.	Indirect use value for residents
Global climate regulation (co-benefit)	Regulating and maintenance	Carbon storage	Indirect use value for the population worldwide
Habitat quality and biodiversity (co-benefit)	Regulating and maintenance	Maintaining nursery populations and species protection	Non-use value for residents and visitors
Recreation (co-benefit)	Cultural	Activities promoting health and enjoyment for residents or tourists (swimming, other water sports, angling, etc.)	Non-consumption use value for residents and visitors
Aesthetic appreciation (co-benefit)	Cultural	Natural landscape providing aesthetic benefits.	Indirect use value for residents and visitors

1.3 Value indicator and transfer method

The literature distinguishes three value transfer methods: unit value transfer (valuation of a single study or an average valuation derived from administratively approved valuations of several studies - with and without adjustments for inflation); benefit function transfer (based on an estimated value function from an individual primary study); and MRA value function transfer (using a value function estimated based on the outcomes of previous primary studies) (Richardson et al., 2015). The outcomes of all these methods are likely to include significant transfer errors (ranging from 0% to 7,080%) resulting from generalization, measurement errors, and publication selection bias (Rosenberger & Stanley, 2006). However, MRA is considered the most promising method since it is less sensitive to the problems related to individual studies (Rosenberger & Stanley, 2006; Richardson et al., 2015). According to Chaikumbung et al. (2016, p.164), MRA has the advantage of “summarising information from several studies and can be used to generate benefit transfer functions that are more widely applicable and less sensitive to the attributes of individual studies”.

Commodity consistency and the welfare consistency are minimum requirements for MRA function transfer (Bergstrom & Taylor, 2006; Smith & Pattanayak, 2002). Most recent MRA studies in environmental economics (e.g., Pettinottia et al., 2018; Chaikumbung et al., 2016; Brander et al., 2013, 2012), address the welfare consistency issue by controlling in the regression function for the Hicksian and Marshallian measures of the welfare (Bergstrom & Taylor, 2006; Smith & Pattanayak, 2002). According to Smith and Pattanayak (2002), this is not recommended for MRA related to value transfer; they consider that the MRA provides the "best" information to evaluate the policy site. Moreover, Bergstrom and Taylor (2006) argue that this approach controls on for the idiosyncrasies of each valuation method. Following Brouwer et al. (1999), we consider only theoretically consistent welfare measures (Bergstrom & Taylor, 2006). Therefore, we focus on work on the Hicksian WTP for river restoration or a mix of services, using CVM and CE method. In our analysis, WTP per household per year represents the consumer surplus derived from changes to the environment from nature restoration.

1.4 Meta-Analysis Protocol

In line with work on MRA for value transfer (Richardson, et al., 2015), we adopt a three step approach. In the first step, we identify relevant keywords, based on NBS measures listed on the NWRM platform, to identify NBS that affect hydro-morphological functioning on the catchment scale. The keywords are wetland restoration, riparian forest restoration, floodplain restoration, river restoration, stream restoration, stream rehabilitation, river ecological restoration. In the second step, we searched for relevant papers in Science Direct, Springer, Wiley and Google scholar, using the previously identified keywords combined with the terms "economic valuation", "contingent valuation" and "willingness to pay". The third step involved selecting the relevant articles, based on the criteria that they valued the impacts of ecological restoration projects on ES provision and that they reported an economic valuation using a stated preferences approach, based on primary and original survey data.

Thus, our final data for the meta-analysis come from 49 studies evaluating the impacts of ecological river restoration on the provision of ES, published between 1996 and 2018. The list of studies is presented in Table 3. We extracted 179 observations, corresponding to between 1 and 17 observations per study. The results in the selected studies are reported in various national currencies (€, £, US$, CAN$, SEK, NZ$, RMB, etc.) and cover different periods (from 1986 to 2016). To homogenize these results and adjust for inflation, in our study, all values are expressed in 2017 US$ ppp (Purchasing Power Parity). This is a commonly used approach to deal with temporal trends in MRA functions (Johnston & Rosenberger, 2010).

Table 3: Authors, NBS measures, country and number of observations for each study in the database

N°	Author	NBS measures	Types of NBS measures	Country	Obs.
1	Adams et al., 2004	Dam removal, riparian buffers.	Riparian vegetation and River stream	USA	1
2	Amigues et al., 2002	Riparian buffers.	Riparian vegetation	France	5
3	Bae, 2011	Natural stream restoration and recreational facilities	River stream	South Korea	2
4	Barak and Katz, 2015	Stream rehabilitation	River stream	Israel	1
5	Beaumais et al., 2009	Floodplain restoration	Floodplain restoration	France	1
6	Bell et al., 2003	Restoration for Coho Salomon recovery	River stream, other	USA	10
7	Berrens et al., 1996	Measures for protecting minimum instream	Other	USA	1
8	Bliem et al., 2012	Floodplain restoration, reconnecting tributaries	Floodplain restoration	Austria	6
9	Bliem & Getzner, 2012	Wetland restoration, reconnecting floodplain, removal of stabilizing blocks of rock	Floodplain restoration, River stream	Austria	4
10	Broadbent et al., 2015	Measures for water resource management and ecological management of riparian forests	Riparian vegetation, Other	USA	3
11	Brouwer et al., 2016	Re-establishing connectivity, removing barriers, enlarging floodplains	Floodplain restoration, River stream.	Austria, Hungary, and Romania	12
12	Che et al., 2014	Ecological restoration of river network	Other	China	6
13	Chen et al., 2014	Ecological restoration of riparian meadows	Riparian vegetation	Belgium	1
14	Colby & Orr, 2005	Conservation of Riparian buffers	Riparian vegetation.	USA	1
15	Collins et al., 2005	Stream restoration	River stream	USA	4
16	Doherty et al., 2014	Ecological restoration of water bodies	Other	Ireland	16
17	Farber & Griner, 2000	Ecological restoration of watershed	Other	USA	2
18	Hanley et al., 2006	Restoration of the ecological status of the river under Water Framework Directive	Other	United Kingdom	3
19	Hanemann et al., 1991	Measures for wetland maintenance and improvement, contamination and salmon improvement	Floodplain restoration, River stream.	USA	10
20	Holmes et al.,2004	Restoration of riparian buffers and natural bank stabilization	Riparian vegetation	USA	4

21	Johnston et al., 2011	Restoration of migratory fish passage, dam removal	River stream.	USA	5
22	Jørgensen et al., 2013	Measures for water quality restoration	Other	Denmark	1
23	Kahn et al., 2017	Restoration of riparian vegetation and regulation measures	Riparian vegetation	Brazil	2
24	Kenney et al., 2012	Restoration of riparian vegetation and meadows	Riparian vegetation, Floodplain restoration	Belgium	2
25	Kim et al., 2015	Measures for ecological restoration	Other	South Korea	1
26	Lehtoranta et al., 2017	Restoration of the original status of the stream, restoration of riparian forests	River stream, Riparian vegetation	Finland	1
27	Loomis, 1996	Dam removal	River stream.	USA	3
28	Loomis et al., 2000	Restoration of riparian buffers, conservation easement, water withdrawing reduction, wetlands restoration	Riparian vegetation, Floodplain restoration, Other	USA	1
29	Mansfield et al., 2012	Dam removal, water withdrawal regulation, fish restoration	River steam, Other	USA	9
30	Meyerhoff & Dehnhardt, 2007	Floodplain restoration, pollution reduction, construction of fish ladders	Floodplain restoration, River stream.	Germany	1
31	Milon & Scrogin, 2006	Change in land use from agriculture for natural reserve, water use restriction	Floodplain restoration.	USA	6
32	Ndebele & Forgie, 2017	Measures for ecological restoration	Other.	New Zealand	2
33	Ojeda et al., 2008	Restoration of wetlands and riparian buffers	Floodplain restoration and Riparian vegetation.	Mexico	1
34	Pattison et al., 2011	Measures for wetlands restoration.	Floodplain restoration.	Canada	3
35	Paulrud & Laitila, 2013	Measures for recreational angling	Other.	Sweden	1
36	Polizzi et al., 2015	Stream restoration	River stream.	Finland	2
37	Ramajo-Hernández & Saz-Salazar, 2012	Measures for water resource management	Other.	Spain	3
38	Rezende et al., 2015	Extending mangrove area, vegetation planting	Riparian vegetation.	Brazil	17
39	Saz-Salazar et al., 2009	Measures for ecological restoration	Other.	Spain	2
40	Schaafsma et al., 2012	Measures for ecological restoration	Other.	Netherlands	1
41	Seeteram et al., 2018	Hydrological and species restoration	River stream.	USA	1
42	Senzaki et al., 2017	Measure for ecological restauration	Other	Japan	4
43	Thomas & Blakemore, 2007	Riparian corridor management and fencing	Riparian vegetation.	USA	1
44	Trenholm et al., 2013	Restoration of riparian buffers	Riparian vegetation	Canada	4
45	Vollmer et al., 2015	Measures for ecological restoration	Other.	Indonesia	3
46	Wang & He, 2018	Sewage interception, waterway	River stream, Other	China	1

		dredging			
47	Weber & Stewart, 2009	Restoration of riparian forests and wetlands, bank removal	Riparian vegetation, Floodplain restoration, River stream	USA	5
48	Zhao et al., 2013	Restoration of riparian vegetation and channel morphology	Riparian vegetation	China	2
49	Zhongmin et al., 2003	Restoration of natural vegetation	Riparian vegetation	China	1

2 Model specification and Data description

2.1 Model and specification

We use the logarithm of the mean annual WTP per household[5] as the dependent variable and, following Boyle et al. (1994), we consider three categories of explanatory variable vectors: (i) NBS measures and benefits; (ii) contextual characteristics; and (iii) methodological characteristics. These variables are described in the succeeding sections and include two sets of independent variables used for the “weak structural utility” MRA approach for value transfer (Bergstrom & Taylor, 2006). The semi-log functional form has been used in other MRA studies to account for distributional issues in the reported WTP (Pettinottia et al., 2018; Ojea et al., 2016; Barrio & Loureiro, 2010). Moreover, because of the nested error structure stemming from the dependency of the reported WTP in a study (Bergstrom & Taylor, 2006; Smith & Pattanayak, 2002), we decompose the error terms into study and estimation errors. The specification is written as:

$$\ln y_{ik} = \beta_0 + \beta_1 X_{1ik} + \beta_2 X_{2ik} + \beta_3 X_{3ik} + \varepsilon_i + \varepsilon_k, \quad (1)$$

where i is the study; k is the number of WTP estimates in each study; β_0 and β_* are the respective constants and coefficients associated to the each vector of explanatory variables to be estimated; ε_i and ε_k are the within study panel error and the cross estimate error respectively. Pettinotti et al. (2018) identified several different estimators used in recent MRA studies based on the Ordinary Least Squares (OLS) and General Least Squares (GLS) techniques. According to Rosenberger and Stanley (2006), publication bias can affect the accuracy of the transfer function, which potentially would bias OLS estimates. Stanley and Doucouliagos (2017) demonstrate that unrestricted Weighted Least Squares (WLS) is superior for correcting

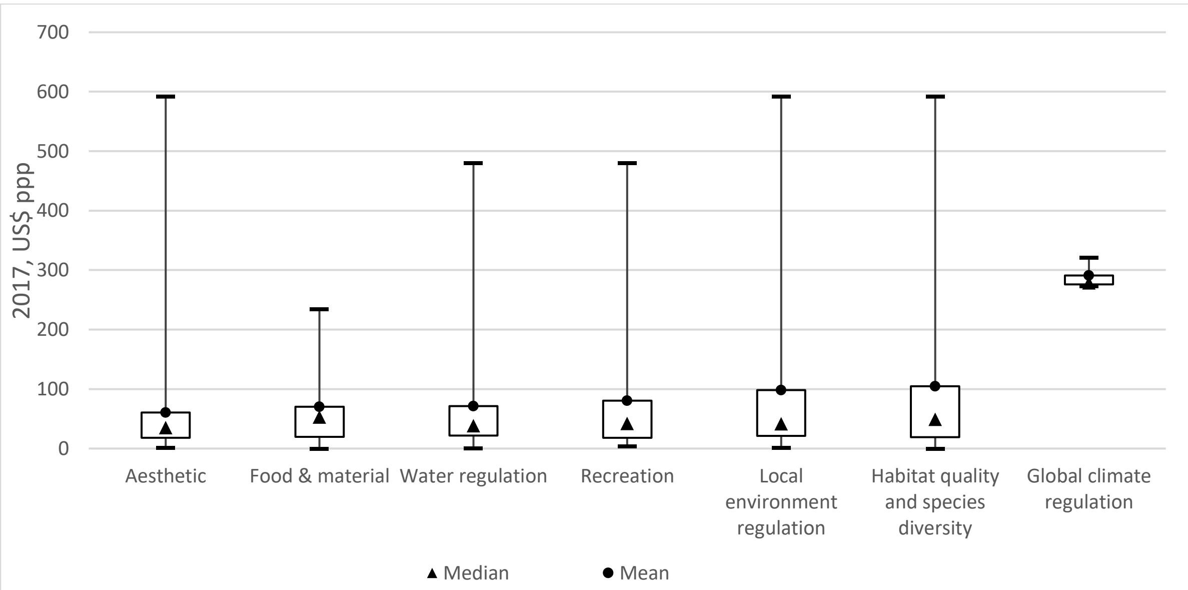
[5] Note that an alternative would be the median WTP. In the literature, median and mean WTP have own advantages for public decision-making. The former is considered a good predictor of public acceptability and real WTP (OECD, 2018), the latter is appropriate if the objective is to identify efficiency criteria in the decision-making process (Barrio and Loureiro, 2010). In the present study, we consider only the mean WTP because few primarily studies report median WTP.

publication (or small sample) bias because it is robust to excess between-study heterogeneity. In line with Chaikumbung et al. (2016), we use the effective sample size[6] to construct a proxy for the estimated standard error since none of the studies report the standard error needed for WLS estimation.

2.2 NBS measures and benefits

The first set of variables include three aspects. The first is the seven types of benefits provided by NBS, which constitute the bundle of ES evaluated in the primary studies (Table 2). Food and materials, water regulation, recreational activities, and aesthetics are classified as direct use values; local environment and global climate regulation are considered indirect use values; and habitats and biodiversity are considered non-use values. A primary study highlighting at least one of these benefits is coded 1 and 0 otherwise. Note that the benefit categories are not mutually exclusive; most studies attempt to value multiple ES. Also, most studies highlight the benefit related to habitats and biodiversity (72%) followed by local environmental regulation (57%), recreational activities (39%), aesthetic appreciation (33%), water regulation (28%) and food and materials (24%). Figure 1 shows the distribution of the WTP in our meta-data, across the different benefits provided by river ecological restoration. Note that although climate change regulation has the highest mean value, this is based on only three observations.

Figure 1: Distribution of the WTP cross benefits



Number of observations for each benefit: Aesthetic (60), Food and material (44); Water regulation (51); Recreation (70); Local environment regulation (104); Habitat quality and species diversity (129); Global climate regulation (3).

[6] It represents the sample size corrected for response rate.

The second aspect considered is type of NBS measure as defined in Table 1. The categories of river stream, riparian vegetation floodplain and other, are not mutually exclusive. Also, we consider the level of ambition in the restoration work or provision of ES. Some studies evaluate two river restoration scenarios with different levels of ambition; giving more room for nature by restoring a larger section of river (e.g., Amigues et al, 2002; De-Rezende et al.,2015) or providing a larger number of ES (e.g., Holmes et al., 2004; Che et al.,2014). We assign the value 1 to the most ambitious scenario (0 otherwise) and the one about 28% of the observations in the meta-data. We consider that this variable captures the scale effect in the provision of ES.

2.3 Contextual characteristics

Contextual variables are related to the socio-economic attributes of respondents and the characteristics of the study area, including income, population density, geographic location and length of local river. Income is the mean income of the respondents in the primary studies[7]. The different country areas include North America (41%), South America (11%), Europe (35% studies) and Asia-Oceania (13%).[8] The length of the river refers to the scale of the restoration and the number of km of river included in the restoration work. All the variables are taken from primary studies or relevant external sources.

2.4 Methodological characteristics

The methodological variables refer to the evaluation methods used in the stated preferences approach - CVM and CE- and are coded 1 (primary study employs CVM) and 0 otherwise. They refer also to the different payment means used in the evaluation exercise, for example, local tax, national tax, utility bill, and donation. National tax, local tax, and utility bill are the most frequent, each representing more than a quarter of the observations. These variables also represent the econometric method used to derive the mean WTP, distinguishing among non-parametric, semi-parametric and parametric models, and the survey mode, distinguishing among face-to-face, internet and a mix of the two. Finally, we control for whether the study was published and the impact factor of the publishing journal.

[7]If the income is not reported, it is proxied by per capita GDP at the time of the survey.
[8] Our research protocol did not identify any primary studies on the African continent.

Table 4: Descriptive statistics

Variable names	Variable description	Mean	Std. Dev	Min	Max
Dependent variable					
WTP	= Natural logarithm of the annual mean of the WTP per household in 2017 US$.	3.49	1.54	-2.54	6.38
NBS measures and benefits					
Water regulation	=1 if the restoration impacts the water regulation benefit, 0 otherwise.	0.28	0.45	0	1
Food & material	=1 if the restoration impacts the food and material benefit, 0 otherwise. Baseline category	0.24	0.43	0	1
Local environmental regulation	=1 if the restoration impacts the local environment regulation benefit, 0 otherwise.	0.57	0.49	0	1
Global climate regulation	=1 if the restoration impacts the global climate regulation benefit, 0 otherwise.	0.02	0.12	0	1
Recreation	=1 if the restoration impacts the water regulation benefit, 0 otherwise.	0.39	0.48	0	1
Aesthetic appreciation	=1 if the restoration impacts the aesthetic benefit, 0 otherwise.	0.33	0.47	0	1
Habitat quality and biodiversity	=1 if the restoration impacts the water regulation benefit, 0 otherwise. Baseline category.	0.71	0.45	0	1
Ambitious scenarios	= 1 if the most ambitious restoration scenario, 0 otherwise	0.27	0.44	0	1
Riparian vegetation	=1 if the restoration impacts the riparian vegetation, 0 otherwise.	0.27	0.44	0	1
Floodplain	=1 if the restoration impacts the floodplain, 0 otherwise.	0.28	0.45	0	1
River stream	=1 if the restoration impacts the river bed, 0 otherwise.	0.29	0.45	0	1
Other	=1 if the restoration impacts agricultural land or is related to other management practices. Baseline category. Baseline category.	0.36	0.48	0	1
Contextual characteristics					
Income	= Natural logarithm of income per household in 2017 US$	10.41	0.77	7.79	11.75
Population density	= Natural logarithm of the density of population in the studied area.	5.65	2.19	0.53	9.01
Length of the studied river	= Natural logarithm of km of river involved in the restoration work.	4.11	2.12	-1.20	10.23
South America	=1 if the restoration project is located in South America, 0 otherwise.	0.11	0.31	0	1
Asia-Oceania	=1 if the restoration project is located in Asia, 0 otherwise.	0.12	0.33	0	1
Europe	=1 if the restoration project is located in Europe, 0 otherwise.	0.35	0.47	0	1
North America	=1 if the restoration project is located in North America, 0 otherwise. Baseline category	0.40	0.49	0	1
Methodological variables					
CVM	=1 if contingent valuation method and 0 otherwise; with CE representing the baseline category.	0.43	0.49	0	1
Local tax	=1 if local tax is the payment vehicle, 0 otherwise.	0.24	0.43	0	1
National tax	=1 if national or income tax is the payment vehicle, 0 otherwise.	0.24	0.42	0	1
Donation	=1 if voluntary participation is the payment vehicle, 0 otherwise.	0.12	0.32	0	1

Unspecified taxes	=1 if the payment vehicle is not specified, 0 otherwise.	0.16	0.36	0	1
Utility bill	=1 if water bill or utility tax is the payment vehicle, 0 otherwise. Baseline category.	0.27	0.44	0	1
Face to face	=1 if face to face is the survey mode, 0 otherwise	0.34	0.47	0	1
Mix	=1 if two or more survey modes are combined, 0 otherwise.	0.25	0.43	0	1
Internet	=1 if web is the survey mode, 0 otherwise. Baseline category.	0.39	0.49	0	1
Semi-parametric	=1 if a semi-parametric model is used for the estimates, 0 otherwise	0.38	0.48	0	1
Non-parametric	=1 if a non-parametric model is used for the estimates, 0 otherwise.	0.06	0.24	0	1
Parametric	=1 if a parametric model is used for the estimates, 0 otherwise. Baseline category.	0.55	0.49	0	1
Impact factor	=the SCImago Journal Rank where the study has been published in the year of the publication	1.27	0.73	0	2.74
Report	= 1 if the study is not published in a journal, 0 otherwise; with Journal article representing the baseline category.	0.06	0.24	0	1

3 Results

3.1 Meta-regression results

The meta-regression results are presented in Table 5. Given our focus on NBS measures and their benefits, we start with the baseline models (1, 2 and 3), which include explanatory variables the NBS measures, benefits, and contextual characteristics. The extended models 4, 5 and 6 include methodological variables as a robustness check. Models 1 and 4 (2 and 5; 3 and 6) uses the OLS (unrestricted WLS; WLS random effects) estimator with standard errors (SE) adjusted for clustering of observations within studies.[9] We conducted several diagnostic tests. The Breusch-Pagan/Cook-Weisberg test rejects the null assumption of constant variance. The mean variance inflation factor statistic is 3.84, meaning that multicollinearity is not a problem.[10] The adjusted R^2 statistics are relatively high, suggesting a reasonable fit between the models and the data. With a few exceptions, the OLS and WLS estimations are fairly similar. The coefficients of the dummy variables show the percentage change in the dependant variable given an absolute change in the explanatory variables.

Starting with estimated coefficients of the benefits, the results support most of the literature (Pettinotti et al., 2018; Brander et al., 2013; 2012; Barrio & Loureiro, 2010). With the respect to habitat quality and biodiversity, as the omitted variable, the primary benefit of water

[9] The WLS is based on the effective sample size as the weight.

[10] The VIF ranges from 1.38 to 11.18. We also checked the Pearson correlation matrix; it confirmed the results of the VIF statistic. The Pearson correlation matrix is available on request.

regulation is negative and insignificant across models. This is in line with Brouwer (2017) and Bergstrom and Loomis (2017). This suggests that, in the primary studies, there is no perception of the self-regulation of nature. Hence, the population is not willing to pay more to reduce the risks related to provision of water. Alternatively, it might be that people assume that they have paid for this through their insurance mechanism. Regarding the co-benefits, we find that, with the exception of the variable for local environmental regulation, the coefficients of the other variables are positive and significant. This suggests that, in addition to the baseline category variable, NBS that provide food and materials, global climate regulation, recreation, and aesthetics are the most highly valued in the primary studies. However, when we consider methodological characteristics, the coefficients of the global climate regulation and food and materials variables are not robust. These results are in line with Johnston et al. (2011) which suggest that, in stated preferences studies, regulation ES are less perceived by people than the final ES and then are less valued. The results indicate, also, that the coefficients of the most ambitious scenario are positively significant and robust, confirming a scale effect in the provision of ES. This supports previous findings from contingent-based valuation methods when environmental changes are characterized by ES provision (Richardson & Loomis, 2009; Van Houtven et al., 2007; Smith & Osborne, 1996). For the NBS measures, the estimated coefficients of river stream are positive and significant across models. This suggests that restoration measures targeting the river stream are generally more valuable than the baseline category of other measures. The coefficients of the riparian vegetation and floodplain variables are not robust across models.

In the case of the contextual variables, with the exception of river length and location, the estimated coefficients are all insignificant. The coefficients of river length are positive and significant, indicating scope sensitivity, while the coefficients of location are not robust. Therefore, interpretation of the results is not straightforward; previous studies provide a mixed picture in relation to the importance of the contextual variables for explaining the huge heterogeneity in the reported WTP. Some studies find a significant effect, others find a non-significant effect. In the case of the income variable, for example, the estimated coefficients are significant in the analyses in Brouwer (2017) and Barrio and Loureiro (2010) and insignificant in the analyses in Brander and Koetse (2011) and Shrestha and Lomis (2002). We observe similarly mixed evidence for the location and ecosystem size variables (Bergstrom & Loomis, 2017; Brouwer, 2017; Bockarjova et al., 2017; Barrio & Loureiro, 2010; Brander & Koetse, 2011; Johnston & Duke, 2009; Brouwer et al., 1999; Lindhjem, 2007). One reason for this might be the data coding protocol employed. For instance, Ojea and Loureiro (2011) demonstrate that different measures of the environmental change being estimated, matter for testing the scope sensitivity hypothesis. Their results show, also, that the estimated coefficients of both the income and location variables are sensitive to the model specification. It could be argued, also, that this is explained by the standardization of the different types of reported values in WTP per hectare (e.g., Pettinottia et al., 2018; Bockarjova et al., 2017; Chaikumbung et al., 2016; Brander & Koetse, 2011). The conversion considers some contextual characteristics because it is based on information on the population and the size of the ecosystem being evaluated.

In the case of the methodological variables, their inclusion improves the explanatory power of the extended models. We find that only the estimated coefficients of the CVM and national tax variable are significant and robust. The coefficient of the CVM suggest that the CE studies in our database were estimated with a linear utility function. Adamowicz et.al. (1998) points out that a linear functional form of the indirect utility function generates a lower WTP for the CE compared to the CVM estimates. In contrast, a quadratic functional form produces a higher WTP compared to the CVM. Mogas et al. (2006) show that four of the seven articles reviewed confirm that CE generates a lower mean WTP than CVM. The coefficient of national tax indicates that, on average, studies using the payment vehicle report a higher WTP. For the remaining methodological variables, the estimated coefficients are either not robust or are insignificant.

Table 5: Meta-regression results (around here)

	Baseline models			Extended models		
	General OLS (1)	Unrestricted WLS (2)	Radom effects WLS (3)	General OLS (4)	Unrestricted WLS (5)	Radom effects WLS (6)
Water regulation	-0.10	-0.16	-0.10	-0.19	-0.33	-0.19
	(0.36)	(0.39)	(0.29)	(0.34)	(0.41)	(0.31)
Food & Material	0.75	1.07*	0.75*	0.01	0.46	0.00
	(0.57)	(0.61)	(0.39)	(0.77)	(0.99)	(0.50)
Local environment	0.33	0.32	0.33	0.21	0.11	0.21
regulation	(0.30)	(0.41)	(0.25)	(0.25)	(0.31)	(0.24)
Global climate	2.28***	2.62***	2.28**	1.32	1.63*	1.31
regulation	(0.59)	(0.64)	(0.87)	(0.88)	(0.95)	(0.93)
Recreation	0.57*	0.82**	0.57**	0.75*	1.17**	0.75**
	(0.30)	(0.36)	(0.23)	(0.40)	(0.46)	(0.25)
Aesthetic	0.60*	0.56	0.60**	0.61*	0.72*	0.61**
appreciation	(0.31)	(0.38)	(0.28)	(0.31)	(0.39)	(0.28)
Ambitious scenario	0.57**	0.73**	0.57**	0.63**	0.79**	0.63**
	(0.20)	(0.23)	(0.24)	(0.22)	(0.27)	(0.23)
Riparian vegetation	0.61	0.88*	0.61*	-0.13	0.19	-0.13
	(0.44)	(0.44)	(0.33)	(0.45)	(0.46)	(0.38)

Floodplain	0.72	0.81	0.72**	0.63*	0.44	0.63*
	(0.49)	(0.48)	(0.31)	(0.37)	(0.37)	(0.33)
River stream	0.98*	1.00*	0.98**	0.89**	1.18**	0.89**
	(0.49)	(0.50)	(0.31)	(0.44)	(0.42)	(0.36)
Income	0.18	0.08	0.18	0.04	0.12	0.05
	(0.27)	(0.28)	(0.18)	(0.23)	(0.22)	(0.21)
Population density	-0.04	-0.09	-0.04	0.08	0.11	0.08
	(0.08)	(0.09)	(0.05)	(0.10)	(0.11)	(0.07)
Length of the studied river	0.21**	0.26**	0.21***	0.13*	0.17*	0.13**
	(0.09)	(0.10)	(0.06)	(0.08)	(0.08)	(0.06)
South America	-2.04*	-1.86	-2.05**	-0.68	-1.18	-0.68
	(1.04)	(1.15)	(0.72)	(1.48)	(1.81)	(0.99)
Asia	-0.47	0.29	-0.47	-0.74	-0.28	-0.74*
	(0.74)	(0.83)	(0.44)	(0.52)	(0.57)	(0.44)
Europe	-0.07	0.17	-0.07	0.03	-0.01	0.03
	(0.45)	(0.47)	(0.31)	(0.45)	(0.45)	(0.33)
CVM				0.88**	1.01**	0.88**
				(0.42)	(0.47)	(0.31)
Local tax				0.12	-0.31	0.12
				(0.50)	(0.52)	(0.47)
National tax				0.86*	1.14**	0.86**
				(0.44)	(0.55)	(0.43)
Donation				1.03	1.28	1.03
				(0.80)	(0.91)	(0.62)
Unspecified				0.47	-0.11	0.47
				(0.67)	(0.75)	(0.51)
Face to face				0.31	0.71	0.31
				(0.71)	(0.73)	(0.39)
Mix				0.43	0.76	0.43
				(0.81)	(0.90)	(0.44)
Semi parametric				-0.44	-0.33	-0.44*
				(0.32)	(0.31)	(0.25)
Non parametric				-0.67	-0.75	-0.67
				(0.66)	(0.64)	(0.47)
Impact factor				-0.38	-0.75*	-0.38
				(0.31)	(0.39)	(0.24)
Report				-0.69	-1.28	-0.69
				(0.78)	(0.96)	(0.82)
Constant	3.44	0.14	3.46*	1.67	-0.56	1.69
	(2.99)	(3.05)	(1.99)	(2.79)	(2.66)	(2.44)
N	179	179	179	179	179	179
R^2 ad.	0.25	0.26	0.25	0.35	0.37	0.35

Standard errors in parentheses
* $p < 0.10$, ** $p < 0.05$, *** $p < .001$

3.2 Fitness for value transfer

Given our aim to provide a value transfer for NBS restoration measures, we perform in-sample and out-of-sample convergence validity tests to explore the usefulness of our MRA function. Brander et al. (2013) highlight that policy makers need to be aware of the potential errors involved when commissioning a value transfer application. According to Rosenberger and Loomis (2000), comparing a transfer value to the "true" value, in an original study of the site is one way to evaluate the performance of the MRA function. Note that there are various stastistics that can be used to access the in-sample and out-of-sample convergence validity of the the MRA function (Brander et al., 2006; Shrestha & Loomis, 2003; Rosenberger & Loomis, 2000). In our analysis, we use Mean Absolute Percentage Error (MAPE) to calculate the transfer error (Kaul et al.; 2013).[11] This evaluates the similarity between the predicted and observed values. The observed value represents the reported WTP in our database (in the case of the in-sample convergence validity test) and the original WTP obtained from studies out-of-database (in the case of the out-of-sample convergence validity test). A relatively smaller MAPE indicates convergence (Shrestha & Loomis, 2003).

Table 6 presents the distribution of the in-sample MAPE of the extended models and Figure 2 plots the observed and predicted values, in ascending order of the observed values.[12] The average MAPE across models represents about 48% with the median around 17%. We find that about 72%, 88% and 5% of our database have a MAPE respectively less than 30%, 50% and greater than 100%. Figure 5 presents similar results to the literature (e.g., Brander et al., 2006; Chaikumbung et al., 2016). It shows that our value transfer function tends to overestimate very low values and underestimate high values. The average and median MAPE are large, suggesting that the transferred values should be interpreted in relation to a particular policy context (Johnston & Rosenberger, 2010). However, they are within the range of those in the literature, from 30% to 186% for average MAPE and 36% to 39% for median MAPE (Kaul et al 2013; Chaikumbung et al., 2016). Consequently, we believe that any of functions of the three extended models could be used to estimate NBS values related to river restoration at policy sites.

Table 6: In-sample MAPE (%)

Extended models	Mean	Median	Min	Max	% of Obs. with MAPE<30%	% of Obs. with MAPE<50%	% of Obs. with MAPE>100%
General OLS (4)	48.14	17.75	0.34	1596.49	74%	88%	4%
Unrestricted WLS (5)	47.46	19.54	0.55	1624.82	70%	88%	5%
Radom effects WLS (6)	48.15	17.76	0.55	1596.31	74%	88%	5%

[11] $MAPE = \frac{|Value_{predicted} - value_{observed}|}{y_{observed}}.100$. Noting that the in-sample MAPE is based on the transformed value of WTP (natural logarithm) (e.g., Brander et al., 2006) while the out-out-sample MAPE is based on the value of WTP (e.g., Shrestha & Loomis, 2003).

[12] We also compute the MAPE for the baseline models, but fund that these are relatively higher; about 54% for the mean.

Figure 2: Observed and predicted values (Unrestricted WLS)

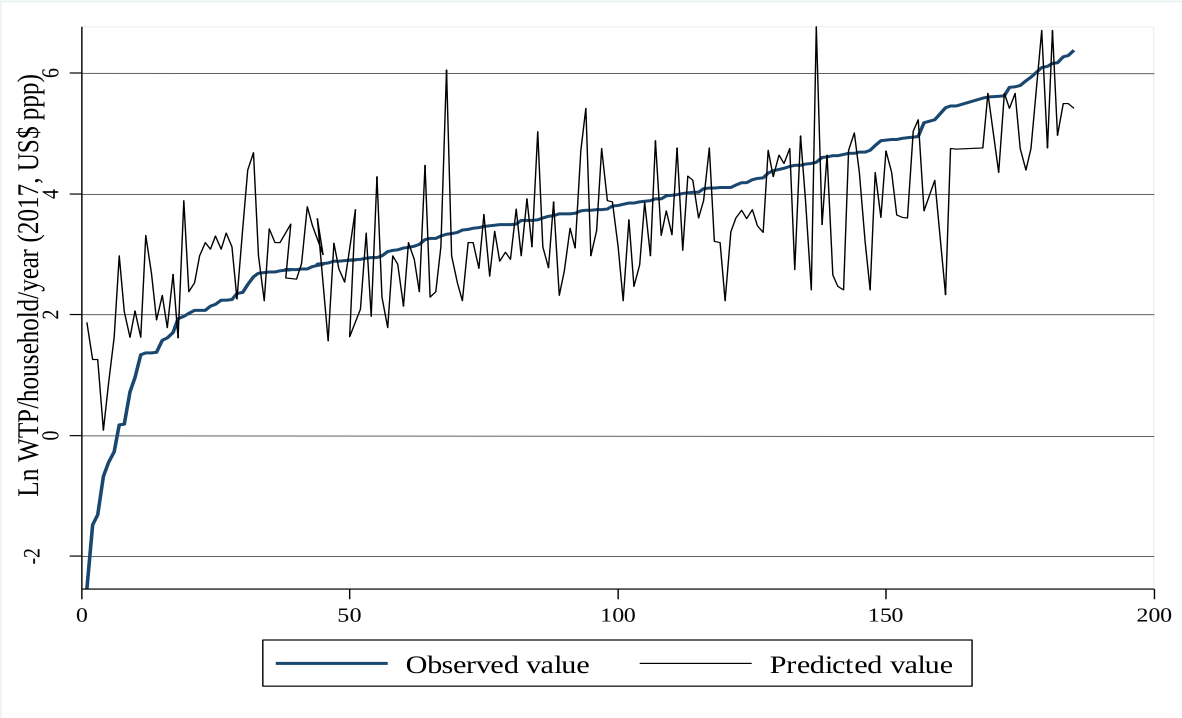


Table 7 presents the results of the out-of-sample convergent validity test for four studies evaluating ecosystem services in river restoration. These studies are "good candidates" (Kaul et al 2013 p. 102) because they fit the criteria defined in the meta-analysis protocol. To calculate the MAPE, we estimate WTP values incorporating the NBS measures and their benefits, and the contextual and methodological characteristics of the studies. As suggested by Wasserstein et al. (2019), we consider the p-values as effect sizes of the confidence intervals of the estimated coefficients. We find that MAPEs range from 8% to 75% and that the unrestricted WLS model performs relatively better than other models. We can conclude that the test performs relatively well compared to prior meta-studies. Indeed, Kaul et al. (2013) show that the median and mean absolute transfer error is 33% and 42% with a maximum of 172%.

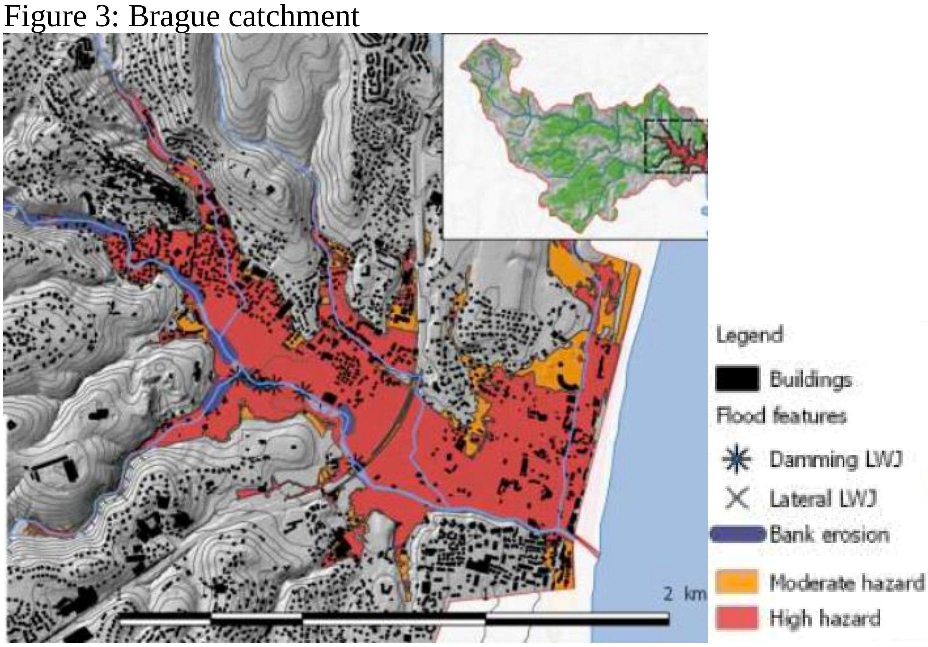
Table 7: Out-sample convergent validity (around here)

Study	Contextual and methodological characteristics	NBS measures	Benefits	Original WTP ($, 2017, ppp)	Estimated WTP ($, 2017, ppp)			MAPE		
					General OLS	Unrestricted WLS	Random effect WLS	General OLS	Unrestricted WLS	Random effect WLS
Stithou et al. (2012)	Boyne river catchment; Irland; CE; Semi parametric model	River stream	Habitat quality and biodiversity; Local envir. Regulation; Recreation.	49	16	45	21	67%	8%	57%
Stithou et al. (2012)	Boyne river catchment; Irland; CE; Semi parametric model	River stream	Habitat quality and biodiversity; Local envir. Regulation; Recreation; Ambitious scenario	77	30	99	40	61%	28%	49%
Shultz and Soliz (2007)	Comarapa watershed; Bolivia; CVM, Parametric model	River stream; Riparian vegetation	Water regulation; Food and material	28	7	10	8	75%	65%	72%
Meyer (2013)	Minnesota river; USA; CE; Parametric model	Others	Local envir. Regulation; Recreational	32	36	47	44	13%	46%	37%
Morardet et al. (2013)	Vistre watershed; France; CE, Parametric model	Riparian vegetation; Floodplain; River stream	Water regulation, local envir. Regulation, Habitat quality and biodiversity recreational	71	21	38	30	70%	46%	57%
Moradet et al. (2013)	Vistre watershed; France; CE; Parametric model	Riparian vegetation; Floodplain; River stream	Water regulation; Local envir. Regulation; Habitat quality and biodiversity; Recreation; Ambitious scenario	94	39	84	56	58%	11%	40%

4 NBS measures for the Brague

The objective of this section is to illustrate the potential of our MRA function transfer in the context of our policy site, the Brague. The Brague is a catchment of about 70 km² in the South-East of France, with a population of 1,300 habitants per km2 (Figure 3). Between 1970 and 2015, the Brague experienced 15 devastating and fatal floods. In the context of climate change, public authorities are planning to implement NBS restoration measures as an alternative to traditional grey measures, to mitigate this flood risk. The Brague catchment is one of nine of the H2020 NAIAD project demonstration sites (demos). The project aims to value of NBS to mitigate natural hazards.[13] In this context, we organized six focus groups and conducted 15 semi-structured interviews between July 2017 and December 2018. As underlined in Section 1, the direct involvement of stakeholders is important to design NBS and evaluate their benefits.

Figure 3: Brague catchment



Source: Pengal et al. (2017)

To design NBS, we used a selection of the NBS measures available on the NWRM[14] platform and discussions with national and local authorities. Two NBS scenarios emerged with one scenario

[13] See NAIAD deliverables D6.1 and D6.3 for details of the characteristics of the Brague demo, the process of the designing the NBS strategies and identification of relevant benefits with the participation of stakeholders (http://naiad2020.eu/).
[14] http://nwrm.eu/

giving more room for the Brague River. The scenarios exemplify different measures such as floodplain restoration and management, streambed re-naturalization, natural bank stabilization and coarse woody debris (Table 8). To identify the relevant benefits, stakeholder involvement was extended to representatives of local NGOs and citizens. These stakeholders put the highest value on flood risk mitigation and identified other ES as co-benefits. For stakeholders, it is important to enchance the ecological functioning of the Brague. They acknowledged the negative impacts on the Brague ecosystem of urbanization and flood risk mitigation infrastructures. They expessed their desire to protect and improve biodiversity, water availability and hydro-morthological equilibria. The stakeholders also identified the impacts of NBS measures on air quality, air cooling, and urban agriculture as important co-benefits.

We estimate the WTP by incorporating the value of the variables of the NBS measures, benefits and contextual characteristics using the unrestricted WLS model. We set methodological characteristics at the database mean. In line with Osborne (2000), we estimate lower and upper bound values using the 95% prediction intervals to account for the uncertainty associated to the estimated WTP. We found a mean WTP of $52 for NBS+ and $117 for the most ambitious NBS. However, the prediction intervals are large stressing on the need of using the MRA transfer function with caution. Johnston and Rosenberger (2010, p. 486) put it that “higher degrees of precision and consequently lower transfer errors are needed as one moves from broad benefit–cost analyses for information gathering or screening of projects and policies to calculation of compensatory amounts in negotiated settlements and litigation”.

Table 8: NBS scenarios and WTP estimated in the case of Brague (around here)

NBS strategies	Types of NBS measures	Types of benefits relevant for stakeholders	Estimated WTP $, 2017 ppp/ Households (Unrestricted WLS)		
			Mean	95% CI Lower bound	95% CI upper bound
NBS+	River stream: 2-40 m of river-bed widening; 5.5 km of natural bank stabilisation; cross road debris Floodplain: 7 ha of restored wetlands; 9.6 km of pedestrian path Others: 960 m² of road brige redisigning 1-2 m high of large wood trapping facilities	Water regulation: Flood risk; underground water availability Recreation: Recreational activities; soft mobility; eco-tourism Local environmental regulation: Air quality, hot temperature mitigation, Habitat quality and species diversity: Biodiversity; hydro-morthological equilibra Food & Material: Urban agriculture (garden community)	52	4	661
NBS++	River stream: 10-40 m of river bed widening; 5,5 km of natural bank stabilisation; cross road debris Floodplain: 11 ha of restored wetlands; 10.3 km of pedestrian path Others: 1,850 m² of road brige redisigning with respect to the Brague ecosystem; 3 m high of large wood trapping facilities	Water regulation: Flood risk; underground water availability Recreation: Recreational activities; soft mobility; eco-tourism Local environmental regulation: Air quality, hot temperature mitigation, Habitat quality and biodiversity: Biodiversity; hydro-morthological equilibra Food & Material: Urban agriculture (garden community)	117	9	1472

5 Discussion and conclusion

The term NBS was introduced in 2008 to refer to ways to mitigate and adapt to climate change effects whilst, simultaneously, protecting biodiversity and improving sustainable livelihoods (Eggermont et al., 2015; Nesshöver et al., 2017). NBS are high on policy agendas, to reduce risks and build a resilient environment. For instance, in the Brague catchment, the public authorities are planning to implement more NBS restoration measures to mitigate flood risk but require better estimates of the economic value of these measures. This information is essential for cost-benefit analysis related to in policy-making decisions (Mechler, 2016).

The paper explored the possibility of relying on the MRA transfer function to estimate the economic value of NBS. The paper addresses a gap in the literature related to NBS river restoration measures. To our knowledge, this is the first MRA of NBS and their benefits, in a river basin context. Since the NBS concept is rather vague, we proposed a framework to identify NBS related to river restoration. We collected information from 49 evaluation studies based on a rigorous protocol ensuring core economic variables, commodity, and welfare consistency.

Our estimates show that individuals value co-benefits, such as recreation and aesthetic appreciation, and the ambitious river restoration scenario. This supports consideration of an ambitious NBS scheme to provide recreational and aesthetic benefits. The primary benefit of water regulation was insignificant but length of the restored river was significant and positive, confirming the sensitivity of the WTP to the scope of the river restoration works. We found, also, that NBS measures affecting river streams are highly valued. The convergent validity tests indicate that the error rates are similar to those in prior studies. Hence, it seems that the value transfer function would be a cost effective policy tool for an economic assessment of NBS. The case study of the Brague catchment area shows that the prediction intervals of the estimated WTP are large because the MRA function suffers from sources of uncertainty (Rosenberger and Stanley, 2006).

The first source of uncertainty is related to publication bias (Rosenberger and Stanley, 2006). We cannot claim that the MRA function is free of publication bias. Our meta-analysis includes peer-review published papers written in English (annex 2). Generalization errors represent another source of uncertainty and arise when applying the MRA transfer function to a policy site whose the characteristics are not fully similar to the characteristics of our study (Rosenberger and Stanley, 2006). In MRA, capturing all the differences in quantity and quality across studies are not straightforward. In particular, we use binary variables to consider the differences arising from the idiosyncrasies of each study, but this cannot capture differences in quality. The categories of the NBS measures and benefits are dummy variables that capture this information only weakly. It might be that these variables do not take account of incremental differences between NBS and benefits. Generalization errors can result, also, from the assumption in the MRA transfer function that preferences remain unchanged over time. This hypothesis is even more important in the case of economic evaluation of NBS. Indeed, people's perceptions of NBS are a determinant of their survival over time (Andersson et al., 2017) and practitioners must remember that this assumption of stable preferences may not hold. Therefore, before employing the MRA transfer function, people's preferences need to be updated by identifying relevant ES to avoid transferring ES that stakeholders do not value.

In addition, uncertainties can result from the measurement errors in primary studies. For example, Johnston et al. (2011 p.1947) highlight that the lack of information on ecosystem functions and their ES may conduct respondents "to speculate as to whether there are additional effects of presented policy scenarios related to changes". This may lead to measurement biais on welfare change. Moreover many study provide insufficient information on the ecological, physical characteristics of the river, and on socio-demographic characteristics such as education, ethnicity, and attributes of local historical context (Johnston and Rosenberger, 2010). It is difficult to obtain this information at the river basin scale, although such data would greatly improve the MRA transfer function. There is stream of work (e.g. Kenter et al., 2015; Primmer et al., 2018) that highlights the critical role played by people's values, beliefs and norms in the evaluation of ecosystems. According to Brouwer (2000), these aspects can affect the outcomes of stated preferences studies, and consequently, increase generalization errors because they do not account for important differences between study sites and policy sites. Hence, primary studies should discuss how people's values, beliefs, and norms affect the WTP for NBS. This would increase the reliability and accuracy of the MRA transfer function for decision-making.

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Annex 1: Definition of NBS and related concepts

Concepts	Definition	Examples of measures
Nature-Based Solutions (NBS)	The IUCN defines NBS as "actions to protect, sustainably manage and restore natural or modified ecosystems that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits" (Cohen-Shacham, 2016, p.5). The EC defines NBS as "actions inspired by, supported by or copied from nature; both using and enhancing existing solutions to challenges, as well as exploring more novel solutions, for example, mimicking how non-human organisms and communities cope with environmental extremes" (Nesshöver & co-authors, 2017, p.1217).	¬ Riparian and wetlands restoration. ¬ Sustainable agricultural practices. ¬ Reconnect rivers and floodplains. ¬ Allow for meandering. ¬ Enhance water retainment. ¬ Extensity agricultural land use. ¬ Transform fields into grassland. ¬ Replacement of fossil fuel and fertilizer input by natural processes and jobs in agriculture. ¬ Green roofs, pockets of nature, or sustainable urban drainage systems in city
Natural Systems Agriculture (NSA)	"NSA is predicated on an evolutionary-ecological view of the world that is featured by an ecologically sound perennial food-grain-producing system where soil erosion goes to near zero, chemical contamination from agrochemicals plummets, along with agriculture's dependence on fossil fuels" (Jackson, 2002, p. 111).	¬ Polycultures of perennial grain crops ¬ Plant community ¬ Soil community
Natural Solutions	"Natural solutions refer to the use of protected areas to deal with the climate crisis. "Protected areas are geographical space, recognized, dedicated and managed, through legal or other effective means, to achieve the long term conservation of nature with associated ecosystem services and cultural values" (Dudley, et al., 2010, p. 8)	¬ Protected areas management ¬ Protected areas development
Ecosystem based Adaptation (EbA)	"EbA integrates the use of biodiversity and ecosystem services into an overall strategy to help people adapt to the adverse impacts of climate change" (Colls et al., 2009, p. 1)	¬ Sustainable management of river ecosystems; grasslands and rangelands; protected area systems ¬ Restoration of coastal habitats ¬ Conservation agriculture systems
Ecosystem Approach (EA)	EA is "a strategy for decentralised, participatory and systemic natural resource management." (Nesshöver & co-authors, 2017, p. 1219)	¬ Multi-stakeholder systemic Management
Green infrastructures	"Green infrastructure is an interconnected network of green space that conserves natural ecosystem values and functions and provides associated benefits" to human populations" (Benedict & McMahon, 2002, p. 12)	¬ Network of parks and wildlife refuges; ¬ Network of waterways, wetlands, woodlands, wildlife habitats ¬ Network of farms; ranches; forests ¬ Ecological corridor or greenways
Ecological engineering (EE)	"EE is defined as the design of sustainable ecosystems that integrate human society with its natural environment for the benefit of both" (Mitsch, 2012, p. 5)	¬ Restoration of river systems, minelands, prairies. ¬ Wetlands creation. ¬ Agro-ecological engineering ¬ Wastewater wetlands ¬ Bio-manipulation ¬ Soil bioremediation ¬ Solar aquatics ¬ Biosphere 2
Catchment Systems Engineering (CSE)	"CSE is an interventionist approach to altering the catchment scale runoff regime through the manipulation of hydrological flow pathways throughout the catchment". (Wilkinson et al., 2014)	¬ Bunds, ¬ Drain barriers ¬ Runoff storage features ¬ Large woody debris dams,

		¬ Buffer strip management, ¬ Willow barriers.
Ecosystem Services (ES)	"ES are the aspects of ecosystems utilized (actively or passively) to produce human well-being." (Fisher et al., 2009, p. 645)	¬ Landscape management ¬ Environmental education ¬ Protected areas management
Natural Capital (NC)	"NC is the sum of exhaustible resources, renewable resources, and what are called today regulating ecosystem services" (Missemer, 2018, p.90)	¬ Terrestrial ecosystems (e.g. forests, landscapes) ¬ Aquatic ecosystems (e.g. river and marine systems) ¬ Maintenance of the composition of the atmosphere ¬ Hydrological cycle regulation ¬ Waste assimilation, recycling of nutrients, generation of soils, pollination of crops. ¬ Scenery of the landscapes.

Annex 2: Studies used in the Meta data

Study	Type and language of publication
Adams, W. M., Cauzillo, M., Chiang, K., Deuling, S. L., & Tislerics, A. (2004). Investigating the feasibility of river restoration at Argo Pond on the Huron RIVER, Ann Arbor, Michigan.	Master's thesis; English
Amigues, J.-P., Boulatoff, C., Desaigues, B., Gauthier, C., & Keith, J. E. (2002). The benefits and costs of riparian analysis habitat preservation a willingness to accept/willingness to pay contingent valuation approach. *Ecological Economics 43*, 17-31	Peer-reviewed journal; English
Bae, H. (2011). Urban stream restoration in Korea: Design considerations and residents'willingness to pay. *Urban Forestry & Urban Greening 10*, 119–126.	Peer-reviewed journal; English
Barak, B., & Katz, D. (2015). Valuing instream and riparian aspects of stream restoration – A willingness to tax approach. *Land Use Policy 45*, 204–212.	Peer-reviewed journal; English
Beaumais, O., Chakir, R., & Laroutis, D. (2009). Valeur économique des zones humides de l'estuaire de la Seine (France) : Application de la Méthode d'Évaluation Contingente.	Working paper; French
Bell, K. P., Huppert, D., & Johnson, R. L. (2003). Willingness to Pay for Local Coho Salmon Enhancement in Coastal Communities. *Marine Resource Economics 18*, 15–31.	Peer-reviewed journal; English
Berrens, R. P., Ganderton, P., & Silva, C. L. (1996). Valuing the Protection of Minimum Instream Flows in New Mexico. *Journal of Agricultural and Resource Economics 21, 2*, 294-309.	Peer-reviewed journal; English
Bliem, M., & Getzner, M. (2012). Willingness-to-pay for river restoration: differences across time and scenarios. *Environ Econ Policy Stud, 14*, 241–260.	Peer-reviewed journal; English
Bliem, M., Getzner, M., & Rodiga-Laßnig, P. (2012). Temporal stability of individual preferences for river restoration in Austria using a choice experiment. *Journal of Environmental Management, 103*, 65-73.	Peer-reviewed journal; English
Broadbent, C. D., Brookshire, D. S., Goodrich, D., Dixon, M. D., Brand, L. A., Thacher, J., & Stewart, S. (2015). Valuing preservation and restoration alternatives for ecosystem services in the southwestern USA. *Ecohydrol. 8*, 851–862	Peer-reviewed journal; English
Brouwer, R., Bliem, M., Getzner, M., Kerekes, S., Milton, S., Palarie, T., . . . Wagtendonk, A. (2016). Valuation and transferability of the non-market benefits of riverrestoration in the Danube river basin using a choice experiment. *Ecological Engineering 87 (2016) 20–29*, 20–29.	Peer-reviewed journal; English
Che, Y., Li, W., Shang, Z., Liu, C., & Yang, K. (2014). Residential Preferences for River Network Improvement: An Exploration of Choice Experiments in Zhujiajiao, Shanghai, China. *Environmental Management 54*, 517–530.	Peer-reviewed journal; English
Chen, W. Y., Aertsens, J., Liekens, I., Broekx, S., & Nocker, L. D. (2014). Impact of Perceived Importance of Ecosystem Services and Stated Financial Constraints on Willingness to Pay for Riparian Meadow Restoration in Flanders (Belgium). *Environmental Management, 54*, 346–359	Peer-reviewed journal; English
Colby, B., & Orr, P. (2005). Economic Tradeoffs in Preserving Riparian Habitat. *Natural Resources Journal, 45*, 15-31.	Peer-reviewed journal; English
Collins, A., Rosenberger, R., & Fletcher, J. (2005). The economic value of stream restoration. *Water Resour. Res., 41, W2017*, 1-9.	Peer-reviewed journal; English
Doherty, E., Murphy, G., Hynes, S., & Buckley, C. (2014). Valuing ecosystemservicesacrosswaterbodies: Results fromadiscretechoice experiment. *Ecosystem Services 7*, 89–97.	Peer-reviewed journal; English
Farber, S., & Griner, B. (2000). Valuing watershed quality improvements using conjoint. *Ecological Economics 34*, 63–76.	Peer-reviewed journal; English
Hanley, N., Wright, R. E., & Alvarez-Farizo, B. (2006). Estimating the economic value of improvements in river ecology using choice experiments: an application to the water framework directive. *Journal of Environmental Management 78*, 183–193.	Peer-reviewed journal; English

Hanemann, M., Loomis, J., & Kanninen, B. (1991). Statistical Efficiency of Double-Bounded Dichotomous Choice Contingent. *American Journal of Agricultural Economics, 73, 4*, 1255-1263.	Peer-reviewed journal; English
Holmes, T. P., Bergstrom, J. C., Huszar, E., Kask, S. B., & III, F. O. (2004). Contingent valuation, net marginal benefits, and the scale of riparian ecosystem restoration. *Ecological Economics 49*, 19–30.	Peer-reviewed journal; English
Johnston, R. J., Segerson, K., Schultz, E. T., Besedin, E. Y., & Ramachandran, M. (2011). Indices of biotic integrity in stated preference valuation of aquatic ecosystem services. *Ecological Economics, 70*, 1946–1956.	Peer-reviewed journal; English
Jørgensen, S. L., Olsen, S. B., Ladenburg, J., Martinsen, L., Svenningsen, S. R., & Hasler, B. (2013). Spatially induced disparities in users' and non-users' WTP for water quality improvements—Testing the effect of multiple substitutes and distance decay. *Ecological Economics 92*, 58–66.	Peer-reviewed journal; English
Kahn, J. R., Vásquez, W. F., & Rezende, C. E. (2017). Choice modeling of system-wide or large scale environmental change in a developing country context: Lessons from the Paraíba do Sul River. *Science of the Total Environment, 598*, 488–496.	Peer-reviewed journal; English
Kenney, M. A., Wilcock, P. R., Hobbs, B. F., Flores, N. E., & Martınez, D. C. (2012). Is Urban Stream Restoration Worth It? *Journal of the American Water Resources Association, 48, 3*, 603-615.	Peer-reviewed journal; English
Kim, J. H., Kim, S.-N., & Doh, S. (2015). The distance decay of willingness to pay and the spatial distribution of benefits and costs for the ecological restoration of an urban branch stream in Ulsan, South Korea. *Ann Reg Sci 54*, 835–853	Peer-reviewed journal; English
Lehtoranta, V., Sarvilinna, A., Väisänen, S., Aroviita, J., & Muotka, T. (2017). Public values and preference certainty for stream restoration in forested watersheds in Finland. *Water Resources and Economics, 17*, 56–66.	Peer-reviewed journal; English
Loomis, J. B. (1996). Measuring the economic benefits of removing dams and restoring the Elwha River: Results of a contingent valuation survey. *Water Resources Research, 32, 2*, 441-447.	Peer-reviewed journal; English
Loomis, J., Kent, P., Strange, L., Fausch, K., & Covich, A. (2000). Measuring the total economic value of restoring ecosystem services in an impaired river basin: results from a contingent valuation survey. *Ecological Economics 33*, 103–117.	Peer-reviewed journal; English
Mansfield, C., Houtven, G. V., Hendershott, A., Chen, P., J. P., Nourani, V., & Kilambi, V. (2012). Klamath River Basin Restoration Nonuse Value Survey.	Report; English
Meyerhoff, J., & Dehnhardt, A. (2007). The European Water Framework Directive and Economic Valuation of Wetlands: the Restoration of Floodplains along the River Elbe. *European Environment 17*, 18–36.	Peer-reviewed journal; English
Milon, J. W., & Scrogin, D. (2006). Latent preferences and valuation of wetland ecosystem restoration. *Ecological Economics 56*, 162– 175.	Peer-reviewed journal; English
Ndebele, T., & Forgie, V. (2017). Estimating the economic benefits of a wetland restoration programme in New Zealand: A contingent valuation approach. *Economic Analysis and Policy 55*, 75–89.	Peer-reviewed journal; English
Ojeda, M. I., Mayer, A. S., & Solomon, B. D. (2008). Economic valuation of environmental services sustained by water flows in the Yaqui River Delta. *Ecological Economics 65*, 155-166.	Peer-reviewed journal; English
Pattison, J., Boxall, P. C., & Adamowicz, W. L. (2011). The Economic Benefits ofWetland Retention and Restoration in Manitoba. *Canadian Journal of Agricultural Economics 59*, 223–244.	Peer-reviewed journal; English
Paulrud, A., & Laitila, T. (2013). A cost-benefit analysis of restoring the Em River in Sweden: valuation of angling site characteristics and visitation frequency. *Applied Economics 45*, 2255–2266.	Peer-reviewed journal; English
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Ramajo-Hernandez, J., & Saz-Salazar, S. d. (2012). Estimating the non-market benefits of water quality improvement for a case study in Spain: A contingent valuation approach. *Environmental Science & Policy 22*, 47–59.	Peer-reviewed journal; English

Rezende, C. E., Kahn, J. R., Passareli, L., & Vásquez, W. F. (2015). An economic valuation of mangrove restoration in Brazil. *Ecological Economics 120*, 296–302.	Peer-reviewed journal; English
Saz-Salazar, S. D., Hernández-Sancho, F., & Sala-Garrido, R. (2009). The social benefits of restoring water quality in the context of the Water Framework Directive: A comparison of willingness to pay and willingness to accept. *Science of the Total Environment 407*, 4574–4583.	Peer-reviewed journal; English
Schaafsma, M., Brouwer, R., & Rose, J. (2012). Directional heterogeneity in WTP models for environmental valuation. *Ecological Economics 79*, 21–31.	Peer-reviewed journal; English
Seeteram, N. A., Engel, V., & Mozumder, P. (2018). Implications of a valuation study for ecological and social indicators associated with Everglades restoration. *Science of the Total Environment 627*, 792–801.	Peer-reviewed journal; English
Senzaki, M., Yamaura, Y., Shoji, Y., Kubo, T., & Nakamura, F. (2017). Citizens promote the conservation of flagship species more than ecosystem services in wetland restoration. *Biological Conservation 214*, 1–5.	Peer-reviewed journal; English
Thomas, R. H., & Blakemore, F. B. (2007). Elements of a cost–benefit analysis for improving salmonid spawning habitat in the River wye. *Journal of Environmental Management 82*, 471–480.	Peer-reviewed journal; English
Trenholm, R., Lantz, V., Martínez-Espiñeira, R., & Little, S. (2013). Cost-benefit analysis of riparian protection in an eastern Canadian watershed. *Journal of Environmental Management 116*, 81-94.	Peer-reviewed journal;
Vollmer, D., Prescott, M. F., Padawangi, R., Girot, C., & Grêt-Regamey, A. (2015). Understanding the value of urban riparian corridors: Considerations in planning for cultural services along an Indonesian river. *Landscape and Urban Planning 138*, 144–154.	Peer-reviewed journal; English
Wang, H., & He, J. (2018). Implicit individual discount rate in China: A contingent valuation study. *Journal of Environmental Management 210*, 51-70.	Peer-reviewed journal; English
Weber, M. A., & Stewart, S. (2009). Public Values for River Restoration Options on the Middle Rio Grande. *Restoration Ecology 17, 6*,762–771.	Peer-reviewed journal; English
Zhao, J., Liu, Q., Lin, L., Lv, H., & Wang, Y. (2013). Assessing the comprehensive restoration of an urban river: An integrated application of contingent valuation in Shanghai, China. *Science of the Total Environment 458–460*, 517–526.	Peer-reviewed journal; English
Zhongmin, X., Guodong, C., Zhiqiang, Z., Zhiyong, S., & Loomis, J. (2003). Applying contingent valuation in China to measure the total economic value of restoring ecosystem services in Ejina region. *Ecological Economics 44*, 345-358.	Peer-reviewed journal; English