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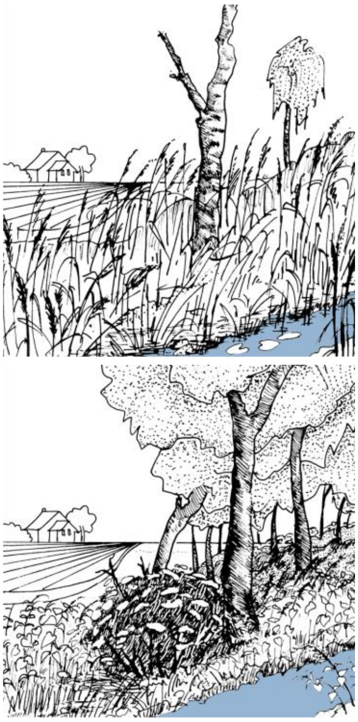
CircuLar Economy Approach to River pollution by Agricultural Nutrients with use of Carbon-storing Ecosystems (CLEARANCE)



REPORT

T5.4. Evaluate cultural ecosystem services and non-use values of rivers in case catchments under contrasting scenarios

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CONTENT

1. Introduction

Although the concept of ecosystem services dates back to the 1970s, it gained impetus in 1990s (e.g., Costanza 1997). After the release of the Millennium Ecosystems Assessment (MEA 2005) it became mainstream in life sciences and economics and influenced the discussion on nature conservation, natural resource governance and other adjacent academic disciplines (de Groot et al. 2010). Although considerable literature has emerged on definition, qualification, and classification of ecosystem services (e.g., Haines-Yong & Potschin 2012), much less advance have been achieved in their quantification and valuation which are important for management and governance (Ref. TEEB 2008). Considering the majority of the most recently proposed typologies, quantification and valuation of some categories of ecosystem services appear particularly challenging, the tendency which applies inter alia to the cultural ecosystem services (Milcu et al. 2013).

Moreover, with cultural ecosystems services, considerable relativism exists already on the classification stage. For instance, whilst the majority of studies routinely classify recreation as a cultural service (e.g., de Groot et al. 2010), others have argued that recreation should instead be classified as provisioning service (Abson & Termansen 2011). Recreational functions of water ecosystems like rivers, lakes, or seaside are highly dependent on water purity whereas water purification is mostly treated as a regulatory ecosystem service. Furthermore, identification of recreation with cultural ecosystem services could be misleading for researchers and policymakers who might assume that recreation exhausts the list of cultural ecosystem services, thereby contributing to marginalisation of aesthetic, spiritual, and other important services of this class (Milcu et al. 2013). Quantification and valuation of cultural ecosystem services face many methodological issues including lack of unambiguous indicators, difficulty of splitting into discrete units, phenomena of bundling, leading towards their frequent omission in quantitative analyses, cost-benefit analysis included. Thus, in their pivotal contribution on valuation of global ecosystem services, Costanza et al. (1997) failed to assign any finite monetary value to cultural services of the world's rivers and lakes.

Meanwhile, in some cases, of which management of small rivers in the farmland landscape is a typical example, quantification and/or valuation of cultural ecosystem services might be key for balanced decision-making. Prospects of rewilding of human-transformed small rivers for the sake of providing riverine water purity which is essential for recreation, impeding eutrophication and floods downstream, and supporting biodiversity depend on whether benefits gained as a result outweigh costs of their restoration and profits lost because of abandoning some agricultural practices. In the absence of any reliable indicators it is difficult to judge upon if the people aesthetically prefer wilderness-like and somewhat spontaneous appearance of small rivers over ordered and regular look of polder systems with canalised rivers and uniform vegetation plots, as the both patterns in landscape preferences are legitimate. Are people willing to pay for restoration of the former or are they sticking to maintenance of the latter? In the Baltic Sea Basin and elsewhere throughout lowland Europe river restoration belongs to the top of the landscape management agenda.

To shed light on those questions, we conducted a discrete choice experiments (DCE) in Denmark, Germany and Poland. Following Bieling et al. (2012), we employ visible manifestations of cultural ecosystem services on the physical landscape, such as riverbed shape and type of riparian vegetation – as means of quantification and valuation thereof. Our article is aimed at gaining empirical evidence of people's preferences towards the characteristics of small lowland rivers of the case-catchments which are essential for generating various ecosystem services enjoyed both on local and upper levels, including in the international Baltic Sea Basin. We try to learn how those characteristics reflect people's preferences, and thus if they could be used as indicators of the corresponding cultural ecosystem services within the framework of their quantification and valuation for the purposes of landscape management and governance. Besides, we examine the impact of information provided in the survey scenario and in the DCE exercise itself on shaping the respondents' tastes.

2. Small rivers transformation in the Lowland Europe

Since the early twentieth century, the majority of small rivers in Europe's lowlands have been artificially straightened, and riparian vegetation sealed or transformed to arable fields. This heavy intervention into the natural riverine ecosystems has made agricultural and forestry activities more profitable and served as a measure to deal with population growth and increased requirements for inland water transportation and flood protection (Addy et al. 2016). In total, up to 90 % of small rivers in the European Union have been transformed and lost their natural character (Szoszkiewicz et al. 2015).

Aside of certain improvement of rural conditions, the mass transformation of small rivers brought negative consequences in terms of ecosystem services supply Ecosystem services are the products of the natural environment, that directly or indirectly benefit human activities, such as water purification, landscape aesthetics, or carbon sequestration (MEA 2005, TEEB 2008). Artificially straightened riverbeds imply acceleration of naturally small flow velocity of lowland rivers entailing reduction of their capacity for water purification and assimilation of nutrients (mainly nitrogen and phosphorus) originating from the agricultural fertilisers.

Additionally, the removal of natural vegetation limit nutrients' interception from the runoff takes place in the farmland ecosystems due to removal of natural vegetation from the riverbanks. Therefore, increased direct influx of nutrients from the riverbank agricultural fields' runoff combined with the reduction of services of riverine ecosystems has led to a deterioration of riverine water quality locally, to a higher soil erosion, and to downstream eutrophication in medium and big rivers, estuaries, lakes and at the seashore. As an (often neglected) consequence, the naturally shallow and geographically inland Baltic Sea (Fig. 1) faced severe eutrophication

(Rönnberg&Bonsdorff 2004) caused by nutrient pollution influx from the mainland rivers exacerbated with a limited inflow of the cleaner marine and oceanic water (Fig. 2).

Apart from worsening water quality, regular and ordered riparian landscapes possess considerably smaller variety of natural conditions, leading to a loss of biodiversity in terms of both living habitats and migration corridors for dependent species including endangered and rare ones. Small river transformation is detrimental for associated recreational, aesthetic, intrinsic and other cultural services of riverine ecosystems. At the end of the day, many small rivers are not recognised by local people as such any more – but rather as anonymous canals or ditches, many of traditional/historical hydronyms have been changed (Brown et al. 2018) or went out of everyday use correspondingly. Finally, the local flood risk mitigation effect at the river's upstream is outbalanced by the reverse tendency downstream, where more severe and rapid floods can appear (e.g. Klaiber 1996, Bojarski et al. 2005, Florsheim et al. 2008).

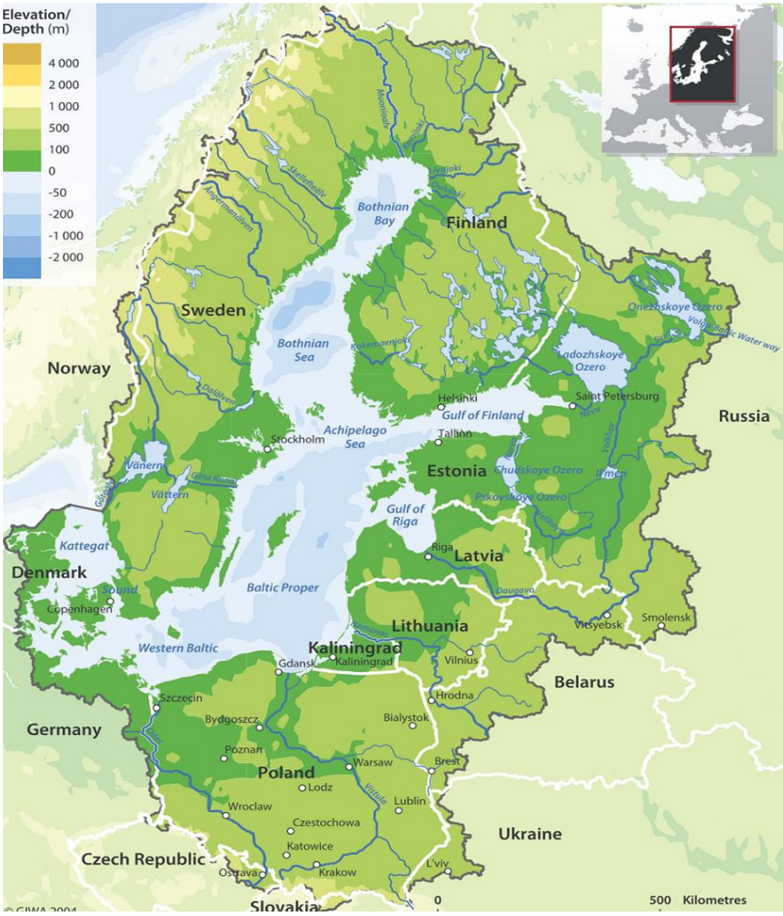


Fig. 1 – Baltic Sea Basin

Source: http://www.beras.eu/implementation/index.php/en/about-us/baltic-sea-basin

Having recognized the negative consequences of non-natural rivers, several governments in Europe have initiated measures to – at least partly – restore rivers to its natural shape. Most prominently, the European Union has published the Water Framework Directive, which requires all member states to ensure a good water quality of all surface waters (Directive 2000/60/EC, Hanley&Black 2006, Hanley et al. 2006, Meyerhoff&Dehnhardt 2007, Metcalfe et al. 2012). Restoration measures can mitigate negative consequences to riverine ecosystem services (Madsen & Debois 2006), including nutrients' cycling, water quality e.g. for recreational purposes (both locally and downstream including in the Baltic Sea), biodiversity, flood risk control, and the very aesthetical appearance of the small rivers in the farmland landscape.

The condition of the above listed ecosystem services are considered dependent on two basic characteristics of the riverine ecosystems: (1) degree of the riverbed tortuosity and (2) presence and type of the wetland buffer zones (WBZ), which are stripes (or differently shaped zones) of wetland ecosystems located in between the farmland and the riverside. The WBZ main functions are to intercept nutrients from the runoff, facilitate water purification, impede soil erosion and downstream eutrophication, and provide habitats and migration corridors for species (Tomer et al. 2015, Youn 2015). Moreover, some available WBZ solutions such as paludiculture (Wichtmann et al. 2010) provide extra economic yield out of their functioning, for instance in the form of biomass harvested for energy and/or construction purposes. Therefore, proposed measures are aimed at (1) reducing of the small rivers' velocity through making riverbeds more sinuous (or letting them meandering freely) and (2) establishing WBZ by separating the small rivers from adjacent farmland.

Satellite image of the cyanobacteria blooming scale in the Baltic Sea in 2018

Aero photo of the cyanobacteria bloom entering the beach

The beach covered by the cyanobacteria bloom

Fig. 2 -- Cyanobacteria blooms in the Baltic Sea (summer 2018)

Such measures have a high impact of the appearance and aesthetics of the landscape, which in turn can have welfare effects for the local population and tourists, who frequently use the riverine landscape for recreation. In the context of societal challenges associated with river restoration, a relevant question appears whether their wilder appearance and re-establishment of wetlands make small rivers more attractive for recreation and revives cultural farmland landscapes – benefits that are often neglected in conventional cost-benefit analyses. The neglect of these rather intangible benefits (often referred to as cultural ecosystem services) in the political decision making process can distort welfare measures and lead to inefficient outcomes (Milcu et al. 2013).

Landscape science literature on people's preferences towards the river restoration and management points at rather mixed evidence of their willingness to adopt riverine landscape restoration measures contemplated by landscape experts. Thus, Kenwick et al. (2009) examined preferences of rural and suburban communities population of Illinois towards the element of riparian buffers with the help of questionnaire survey. They found the overwhelming approval amongst both rural and suburban population with no differences between the preferences of professional landscape planners and common residents towards tree buffer at the water edge. On the contrary, the grass buffers appeared rather negatively seen by the residents while being endorsed by the landscape planners. They also found positive preferences towards the meandering streambed, whereas the preferences towards the earthen banks over human-made edges were not expressed confidently.

3. Cultural services of ecosystems

The Millennium Ecosystem Assessment (Sarukhán & Whyte 2005) defined cultural ecosystem services as *"the nonmaterial benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences"*. Cultural ecosystem services have been included in many other typologies of ecosystem functions and services. One essential characteristic of cultural ecosystem services is their intangibility, which often makes the physical, emotional, and mental benefits produced by them subtle and intuitive in nature (Kenter et al. 2011) and requiring indirect manifestations to be expressed through (Anthony et al. 2009). Therefore, the value assigned to cultural ecosystem services is highly dependent on cultural assessments of their contribution to individuals' well-being.

Cultural ecosystem services can be classified as ether non-consumptive direct use values (e.g. chamber non-invasive recreation) or non-use values (e.g., existence, bequest, or spiritual values). They typically suffer from poor quantification and integration in cost-benefit analysis including for policy and/or strategic documents (de Groot et al. 2005). With the rare exceptions, values of cultural ecosystem services are seldom captured by market transactions as this services are rarely marketable (e.g., Carpenter et al. 2009, Martín-López et al. 2009).

More wealthy and urbanised societies often value cultural ecosystem services in the first turn, and demand for cultural ecosystem services is expected to further grow in them (e.g., Carpenter et al. 2009, Guo et al. 2010, Ingold & Zimmermann 2011), whereas in traditional communities, cultural ecosystem services are essential for cultural identity and even survival (e.g., Le Maitre et al. 2007, Voora & Barg 2008, Brown & MacLeod 2011). Nevertheless, they are sometimes neglected and/or sacrificed by decision makers (de Groot et al. 2005, Chan et al. 2011, Hendee 2011).

Explicit accounting for cultural services in assessment-based decision-making is essential, in order to avoide bias toward other ecosystem services and unwanted trade-offs in land management. However, their integration, quantification and valuation are challenging tasks (Schaich et al. 2010, Milcu et al. 2013). Given their highly individual nature, empirical studies of perceptions, values, attitudes, and beliefs are required in order to generate more meaningful insights regarding the contributions of ecosystem services to human well-being. In particular, they give more precise understanding of the relevance of ecosystem services for common citizens and local stakeholders, allowing better recognition of trade-offs between different users and their groups. Many perception and preference-based studies have revealed that cultural ecosystem services are at least as preferred as other (more tangible) categories of ecosystem services. Therefore, valuation methods should be used enabling direct comparison of the scenarios comprising different proportions of cultural vs other categories of ecosystem services.

As the Millennium Assessment does not formulate explicit definitions of cultural ecosystem services, in our study we employ visible manifestations of cultural ecosystem services of the physical landscape, such as riverbed shape and type of riparian vegetation – as means of quantification thereof in our valuation study – in order to render them more intelligible for layman decision-makers, such as common citizens and/or local population.

4. Valuation of riverine ecosystem services amidst prospects of their restoration

4.1. Theoretical grounds of economic valuation of non-market goods

Valuation is the process of attributing a value (either economic or non-economic) to something. Non-economic valuation, mostly avoids using monetary terms, whereas the aim of economic valuation is to measure, in monetary terms, people's preferences for the benefits they obtain from, say functioning of intact ecosystems. The fact that ecosystems are valuable in monetary terms should encourage decision makers on various levels – ranging from individuals to governments – to take them into account in their decisions.

Valuing all ecosystem services in monetary terms might be not feasible. Thus, only a small subset of ecosystem processes and components are priced and incorporated in transactions as commodities or services in the real world (Pascual et al. 2010). In economics every decision about choice is preceded by comparison of benefits and costs associated with the choice alternatives. Measurement in monetary terms provides common denominator enabling direct comparison of benefits provided by ecosystem services (typically unobservable to decision-makers) against costs of their preservation and maintenance (mostly observable through the market transactions). The essence of ecosystem valuation, therefore, is to predict and show in monetary units how human decisions would affect ecosystem services' values, allowing their incorporation in public decision-making processes (Mooney, Cooper & Reid, 2005).

There are two main valuation approaches – biophysical and preference-based. They refer to different (but complementary) dimensions of ecosystem values. Biophysical methods address the concept of ecosystem resilience (i.e. the capacity to remain in a given ecological state). However, many challenges and limitations exist when valuing ecosystem resilience is difficult or impossible (i.e., when sudden and uncertain ecosystem transitions impede using marginal values). Therefore, current valuation efforts predominantly focus on preference-based methods, built on the concept of total economic value (TEV) and comprising two big families of methods, namely revealed preferences and stated preferences.

Revealed preference methods (e.g., travel-cost method, hedonic pricing) value non-market environmental goods by examining the consumption of related market-priced private goods. A number of variants of the revealed preference approach exist, depending on whether the environmental good and the related market good are complements, substitutes or one is an attribute of the other. A principle limitation of the revealed preferences approach is inability to elicit non-use (existence and/or bequest) values.

While revealed preference methods estimate original values by looking at actual behaviour, eliciting values by looking at intended behaviour is the province of stated preference methods. This is an umbrella term for a range of survey-based methods that use constructed or hypothetical markets to elicit preferences for specified changes in provision of environmental services. By far the most widely applied stated preference techniques are the contingent valuation method (e.g., Alberini & Kahn 2006) and, choice modelling (e.g., Hanley et al. 2002; Kanninen 2007). In the latter method, respondents are required to choose their most preferred alternative out of a (possibly relatively large) set of alternative policy or provision options offered at different prices and their willingness to pay is revealed indirectly through their choices. Ability to elicit both use and non-use values, by far remains an exclusive property of stated preference methods. Therefore, since cultural ecosystem services by definition comprise a substantial proportion of non-use value, use of stated preference approach is deemed an obvious choice as a tool of their monetary valuation. In this report, we also follow this approach through application of the Discrete Choice Experiment (DCE).

4.2. Review of literature on riverine ecosystem services' valuation

Restorations of the small rivers on the mass scale, carried out in various countries of Europe and in North America (e.g., Wheaton 2005, Madsen&Debois 2006, Bańkowska et al. 2010, Wohl et al. 2014, Binder et al. 2015), have introduced considerable changes of farmland landscapes and have entailed a broad range of consequences for the rural communities and businesses. From the economic perspective, despite the apparent win-win character, tendencies in people's preferences and benefits for different stakeholder groups are ambiguous. Affected land-users might encounter harvest and profits lost, being reluctant towards the restoration programmes even despite economic incentives (Buckley et al. 2012; Dworak et al. 2009). Local communities – if not informed properly – may fear higher flood hazard as that a 'softer' approach associated with rivers' restoration would have been seen by the public as offering a lower level of flood protection than the 'hard' engineering solutions which prevailed in the past (Tunstall et al. 2000). Furthermore, some people might prefer more regular, uniform and ordered farmland landscapes over 'chaotic' appearance associated with the meandering and WBZ because of their ideological beliefs or aesthetical tastes – the preferences sometimes observed with farmers and rural population (Ryan 1998; Nassauer 1989, 1997). Finally, people's preferences towards restoration of particular small rivers might be affected by the distance to the site, or by presence of other small rivers providing similar ecosystem functions and services in other (possibly, more convenient or less costly) locations. However, the higher costs and losses associated with stream restoration can in some cases be justified by its aesthetic and recreational benefits (Kenney et al. 2012) elicited with economic valuation tools.

To improve decision making in the context of natural resources, economists have established a set of methods to measure the value of cultural ecosystem services, or more generally, non-marketed goods and services (Freeman III et al, 2014). A frequently used method are discrete choice experiments (DCE). DCE have recently gained popularity in ecosystem services valuation, as they DCEs are embedded in questionnaires and allow respondents to trade off multiple elements in a policy choice involving biodiversity conservation or other public goods (Carson 2012). The respondents are typically presented with hypothetical policy scenarios and asked to choose their most preferred variant or to provide a full ranking of the variants, given the associated cost to their personal or household budget. Previous studies have applied DCE to landscape restoration management of various land cover types, including natural grassland habitats (e.g. Christie et al. 2006; Dallimer et al. 2015), forests (e.g. Meyerhoff et al. 2009, Liekens et al. 2013, Giergiczny et al. 2015, Valasiuk et al. 2017, 2018a), and wetlands (Carlsson et al. 2003, Birol et al. 2006, Newell & Swallow 2013, Valasiuk et al. 2018b).

There is also a substantial corpus of DCE literature addressing the various problems of rivers conservation and management with a mixed evidence of appropriate people's stated preferences. Until now, several studies used DCEs to elicit the economic value of riverine ecosystem services provided evidence of overwhelming public support for rivers' preservation and restoration, however sometimes with significant spatial, directional and other effects underlying taste heterogeneity.

Thus, Willis & Garrod (1999) found negative utility with declared anglers, local households and recreation visitors associated with increasing flow management on low-flow rivers of South-West England. Holmes et al. (2004) state that WTP for total restoration of the Little Tennessee river in North Carolina was greater than the sum of WTP for the contemplated partial restoration programs. Morrison & Bennet (2004) indicate the importance of valuing improved river health within five specific catchments of the New South Wales by sampling local populations. Kragt & Bennet (2009) claim that Tasmanians state, on average, positive values for protecting native riverside vegetation and rare native species in the George river catchment. Zander & Straton (2010) found support of Aboriginal and Non-Aboriginal respondents towards the conditions and ecosystem services of rivers in tropical Australia, including their spiritual values in traditional culture and beliefs. Bennet et al. (2008) estimate the benefits associated with improvements in the environmental health of Goulburn, Moorabool, and Gellibrand rivers in Victoria. They found that people irrespective of their location value the ecosystem services of rivers. Rayanov et al. (2018) revealed a high appreciation of Saxony's population for an increased naturalness of river banks and floodplains with WTP of more than 120 EUR per household and year, combined with heterogeneous preferences towards further infrastructural development stated by users vs non-users.

At the same time, Paulrud & Laitila (2013) when eliciting the citizens' WTP for restoration of Em and Mörrum rivers in Sweden (for the main purpose of recreational angling), found that costs of hydro-construction outweigh the benefits across all scenarios, even if they may provide significant ecosystem services. Kataria et al. (2012) found that retrieved welfare estimates over the ecosystem services of Odense river in Denmark were affected by disbelief in survey information. Khan et al. (2019) found significant heterogeneity across sub-basins of the Heihe river in the Northwest of China in case of various ecosystem services including water quality, farmland irrigation, sandstorm days, lake area, biodiversity, landscape and leisure & entertainment. Likewise, Brouwer et al. (2010) as well as Martin-Ortega et al. (2012) found significant spatial heterogeneity of preferences towards improvement of water quality across the Guadalquivir River Basin in Spain, as despite the respondents positive preferences towards water quality improvements throughout the entire river basin, they appeared not willing to pay extra to reach a more than good condition elsewhere, but in their own sub-basin. Likewise, significant spatial preference heterogeneity measured through distance decay and substitution effects was found by Lizin et al. (2016) in the case of Oude Kale and Leie rivers in Flemish part of Belgium. Moreover, Schaafsma et al. (2013) found significant directional effects in their DCE of ecosystem services of the Dutch Scheldt estuary, whereas Logar&Brouwer (2018) report significant distance decay, directional, and substitutional effects differently affecting preferences of local population and visitors in the case of the rivers Thur and Töss, located in north-eastern Switzerland.

Simultaneously, DCEs have also been used to put an economic value on water quality improvements at the Baltic Sea. Markowska&Żylicz (1999) report in their seminal stated preference contribution on Baltic Sea as an international public good an essential asymmetry in the distribution of costs of, and benefits from, the reduced eutrophication of the Baltic Sea between the littoral countries. In a recent EU project, Athiainen et al. (2015) have used a closely related method, contingent valuation and identified a high willingness to pay of the countries' population. Several other studies have found similar values; Sagebiel et al. (2016) and Athiainen & Vanhatalo (2012) summarise economic valuation studies regarding the Baltic Sea. On the basis of their meta-analyses covering 76 empirical studies conducted in the Baltic Sea countries, Sagebiel et al. (2016) found predominance of the valuation studies addressing eutrophication reduction and seaside recreation over other marine ecosystem services. This seems to reflect the main focus of scrutiny attached to Baltic Sea from the environmental economics' viewpoint. Given the causal dependence of the Baltic Sea eutrophication dynamics on the water purity of the discharging rivers systems, these issues should be approached holistically.

The studies mentioned above mostly address either a local change of a specific river (or several specific rivers), or a national or multinational programme; some others studies (e.g. Metcalfe et al.

2012) address hypothetical riverine water quality improvement both on local and national levels, whereas other studies involve datasets from several countries (e.g. Markowska&Żylicz 1999, Brouwer et al. 2009, Bateman et al. 2011, Athiainen et al. 2016).

However, according to the best of our knowledge, there is no study so far that simultaneously investigates preferences in the several countries for the total chain of interconnected water bodies' governance starting from local small rivers management – via national level programmes – to multinational measures for water quality and other ecosystem services' improvements in the contexts of the small rivers restoration measures. It appears a challenging task to separate the people's preferences towards the visible features of semi-intact small rivers such as curvy riverbeds and semi-natural WBZ vegetation from their preferences in favour of desirable ecosystem services being generated in the consequence of the WBZ restoration, e.g. improved river water quality, or reduced eutrophication of the sea. Indeed, some people might be willing to pay for cleaner water suitable for recreational purposes in their country's rivers and at the seaside, and at the same time be reluctant to restoration of sinuous small riverbed shape and/or WBZ in their close neighbourhood. Technically, these preferences are not mutually exclusive.

Therefore, in the present study, we try to fill this gap by simultaneously investigating preferences for local changes in the selected case-catchments and for national and multilateral programmes. Only in this setting, we can disentangle the relative importance of the different effects of small river restoration, including changes under contrasting scenarios in the local landscape as well as in international public goods such as reduction of eutrophication in the Baltic Sea. In particular, we concentrate on the following research questions:

- Firstly, we estimate willingness to pay of the local population for local river restoration in the selected case-catchments under contrasting scenarios as well as for nation-wide and international level programmes to improve water quality of larger rivers and the Baltic Sea. We thereby differentiate between measures that imply different impacts on the landscape. The results serve as indicators for preferences and can be used in the decision making process at various policy levels.

- Secondly, we measure how the population of the case-catchments located in lowland parts of Denmark, Germany and Poland, distribute their preferences between local-level issues such as the small rivers' appearance and functioning, and the consequences emerging on the higher levels of governance, for example reduction of eutrophication in the Baltic Sea.

To shed light on those questions, we conduct DCE in Denmark, as well as in the lowland parts of Germany and Poland. Following Bieling et al. (2012), we employ visible manifestations of ecosystem services on the physical landscape, such as riverbed shape and type of riparian vegetation – as means of quantification and valuation thereof on the local level. Besides, we confront them in the contrasting scenarios with complex indicators combining qualitative and quantitative aspects to elicit the value of water purity on the upper (both national and international) levels.

Additionally, we are addressing the problem of whether questionnaire-based state preference studies, which demand substantial transfer of information to the respondents before the decision-making exercise really discover the preferences, or create them. To do so, in addition to the DCE we ask simple intermediate questions and we employ comparative ranking of the photographs depicting the visual manifestations of the ecosystem services under scrutiny, presented to the different treatments of respondents, as well as simple intermediate questions.

5. Methodology

Ecosystem services of rivers are particularly challenging objects for the stated preference valuation exercises because of their spatial characteristics which impede a clear definition of the valued goods. The rivers' length complicates their holistic perception by the respondents – few people are assumed to properly accommodate any river in its entire flow from the source and to the mouth. Instead, most people are assumed to identify a river with its particular spatial sections and/or certain places of the riverside, mostly those located close to their places of residence or those being frequently visited with different purposes such as recreation. In case of the transformed small rivers of lowland Europe, their identification as valuation objects is further complicated because of their current anonymity – some human-transformed small rivers are currently perceived rather as merely anonymous ditches or canals.

The study scope covers the small rivers of the lowland part of the three European countries belonging to the Baltic Sea Basin fully – as Poland is, or partly – as Denmark and Germany are. Unlike Denmark where the entire country's area is lowland, in Germany and Poland we had to exclude the predominantly upland and mountainous administrative units from the study spatial focus.

5.1. Case Study Areas

The study scope covers the small rivers belonging to the three case-catchments of the lowland part of the three European countries belonging to the Baltic Sea Basin, namely Odense in Denmark, Ryck in Germany, and Narew in Poland. As the selected case-catchments are typical for the lowland parts of the countries involved, we assume that preferences of the population of lowland parts of the countries involved serve as a good proxy for preferences of the selected case-catchments' population. Therefore, we use preferences of the lowland population of Denmark, Germany, and Poland as proxy for preferences of the population of case-catchments of Odense, Ryck, and Narew correspondingly. Unlike Denmark where the entirely country's area is lowland, in Germany and Poland we had to exclude the predominantly upland and mountainous administrative units from the study spatial focus.

Thus, in Poland the study covers the entire area of Kujawsko-Pomorskie, Lubelskie, Lubuskie, Łódzkie, Mazowieckie, Opolskie, Podlaskie, Pomorskie, Warmińsko-Mazurskie, Wielkopolskie, Zachodniopomorskie voivodships[1] as well as Dolnośląskie voivodship excluding the mountainous powiats Lubański, Lwówecki, Jeleniogórski, Kamiennogórski, Wałbrzyski, and Kłodzki.

In Germany the study focus covers the entire area of the federal lands of Schleswig-Holstein, Mecklenburg-Vorpommern, Brandenburg, Bremen, Hamburg, Berlin, Hessen. In the federal land Nordrhein-Westfalen the following predominantly upland landkreises[2] were excluded: Euskirchen, Hochsauerlandkreis, Märkischer Kreis, Oberbergischer Kreis, Olpe, Siegen-Wittgenstein; in the federal land of Niedersachsen the landkreis Goslar was excluded; in the federal land Sachsen-Anhalt landkreis Harz was excluded; in the federal land Sachsen the landkreises Erzgebirgskreis i Sächsische Schweiz-Osterzgebirge were excluded. On the contrary, the following lowland parts of the predominantly upland federal lands were included in the study: of the federal land of Baden-Württemberg landkreises Rhein-Neckar-Kreis, Karlsruhe, Rastatt; of the federal land of Rheinland-Pfalz landkreises Mainz-Bingen, Mainz, Alzey-Worms, Worms, Rhein-Pfalz-Kreis, and Germersheim.

5.2. The questionnaire

The survey questionnaire was prepared as a result of interdisciplinary consultations and trial in-depth interviews. The questionnaire consisted of five parts, namely (1) introduction, (2) travel cost method questions, (2) survey scenario description, (3) choice experiment part, (4) attitudinal questions, and finally (5) socio-demographic questions. In addition to these five parts, we tested the impact of provided information on respondents' preferences.

5.2.1. DCE survey scenario

The survey scenario began with an explanation of the impact of the increasing use of fertilisers entering the transformed small rivers from the surrounding agricultural land and other sources on the eutrophication processes in rivers, lakes and in the Baltic Sea resulting in the increasingly more frequent and long-lasting blue-green algal (cyanobacterial) blooms there.

Small rivers' bed shape and the type of riverside vegetation restoration were presented as key elements of their management and governance. Their anticipated impact was explained on ecosystem services at the country level, such as sufficiency of water quality in rivers for recreational purposes, as well as water quality in the Baltic Sea. Likewise, the impact of the small rivers' bed shape and type of riverside vegetation on their water purity and biodiversity in the respondents' immediate neighbourhood were explained.

A hypothetical small rivers restoration programme was subsequently introduced. The programme aimed at improving water quality in the rivers throughout the respondents' country of residence,

[1] Administrative division hierarchy of Poland comprises voivodships and powiats, where a voivodship consists of several powiats

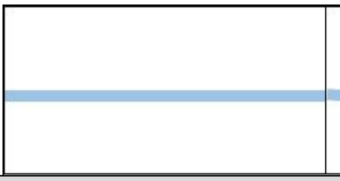
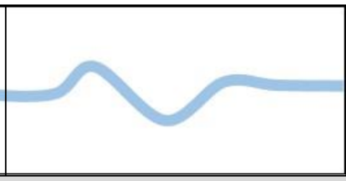
[2] In Germany's administrative division hierarchy a federal land comprises several landkreises.

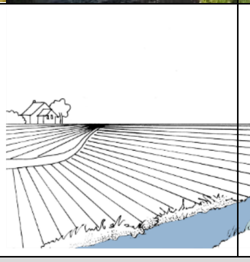
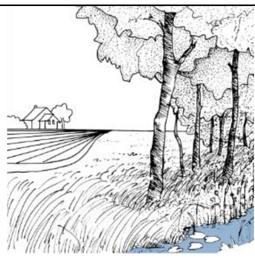
and in consequence of widespread implementation also in the Baltic Sea, as well as at changing of the local landscape and environmental condition of small rivers in the respondents' neighbourhood. The suggested programme included five key elements, referred to as attributes: (I) water purity in the country's lowland rivers, (II) water purity in the Baltic Sea, (III) the riverbed type of the small rivers flowing within 20 km distance from the respondent's place of residence, (IV) the riverine vegetation type at the banks of the same small river, and (V) cost.

The attributes (III) and (IV) reflect proposed changes of the small rivers' management and governance at the local level in the vicinity of the respondent's place of residence, whereas the attributes (I) and (II) describe the changes proposed at the higher (i.e. national and international) levels. The proposed programme attributes and their levels together with their visualisation are summarised in the Table 1.

Table 1 – Attributes, their levels and description

<table>
<tr><th colspan="4">ATTRIBUTE I – WATER QUALITY IN THE RIVERS ON THE COUNTRY LEVEL</th></tr>
<tr><td>GOOD
(improvement)</td><td colspan="3">Bathing possibility without limits</td></tr>
<tr><td>MEDIUM
(current state)</td><td colspan="3">Recreational use by adults only – not suitable for children</td></tr>
<tr><td>BAD
(worsening)</td><td colspan="3">Water is not suitable for bathing</td></tr>
<tr><th colspan="4">ATTRIBUTE II – WATER QUALITY IN THE BALTIC SEA</th></tr>
<tr><td></td><td colspan="2">Cyanobacteria blooms</td><td>Prohibition of bathing</td></tr>
<tr><td>GOOD
(improvement)</td><td colspan="2">Rare, locally</td><td>1-3 days a year</td></tr>
<tr><td>MEDIUM
(current state)</td><td colspan="2">Possible almost every summer, medium extent</td><td>4-10 days a year</td></tr>
<tr><td>BAD
(worsening)</td><td colspan="2">Possible every summer, widespread</td><td>11-20 days a year</td></tr>
<tr><td>VERY BAD
(strong worsening)</td><td colspan="2">Possible every summer, extremely widespread</td><td>The water is not suitable for bathing</td></tr>
<tr><th colspan="4">ATTRIBUTE III – RIVERBED SHAPE AND DYNAMICS</th></tr>
<tr><td>Levels:</td><td>regulated straightened riverbed</td><td>regulated curvy riverbed</td><td>naturally meandering riverbed</td></tr>
<tr><td>Illustrative photo depicting a given vegetation type</td><td></td><td></td><td></td></tr>
</table>

Icon representing a given vegetation type			
EXPLANATION: POTENTIAL FOR SUPPORTING OF			
riverine water purification	- Very low	medium	high
high biodiversity	- Very low	medium	high
water retention upstream and flood defence downstream	- Very low	medium	high

ATTRIBUTE IV—RIPARIAN VEGETATION TYPE				
Levels:	***Low-intensity agriculture***	***Intensive agriculture***	***Wild marshes***	***Wetland agriculture***
Illustrative photo depicting a given vegetation type				
Icon representing a given vegetation type				
EXPLANATION: POTENTIAL FOR SUPPORTING OF				
riverine water purification	Low	- Very low	High	High
high biodiversity	Low	- Very low	High	High
water retention upstream and flood defence downstream	Low	- Very low	High	High

ATTRIBUTE V: COST		
Annual change in your income as a result of the programme implementation	The levels of change in income were country specific	Germany: 0, 25, 50, 100, 200, 300 (in EUR) Poland: 0, 50, 100, 200, 400, 600 (in PLN) Denmark: 0, 175, 350, 700, 1400, 2100 (in DKK)

The cost attribute was framed as a new annual compulsory tax which would be imposed for all the country's citizens. The payment vehicle justification was given that financial means would be necessary for transformation of the riverbeds shape and restoration of the riverine vegetation stripes as well as for reimbursement of the lost profits to land users in cases when establishment of WBZ would entail shrinking of their farmland grounds.

The two country-level attributes (attributes I and II) were assigned different time perspectives. It was stated in the scenario description that the levels of the attribute indicating water quality in the rivers throughout the country reflect its prospective state in ten years from the start of the programme implementation, whereas the corresponding levels of the attribute indicating water quality in the Baltic Sea reflect its state in thirty years from the start of the programme implementation.

The business-as-usual was defined as medium water quality throughout the country's rivers both now and in ten years, whereas it was stated that lack of change now would entail growing accumulation of pollutants in the Baltic Sea, making the maintenance of the current state impossible and leading to the bad state. Regarding the local level attributes (attributes III and IV), it was assumed for the business-as-usual that the riverbed type was straight whereas the riparian vegetation type was set out as an Intensive agriculture for Denmark and Germany whilst Low-intensity agriculture – for Poland. Business-as-usual did not imply any changes in the respondents' annual income.

The four programme attributes (and their impacts) were described as being independent from each other. Although the local rivers' water quality in the respondent's residence vicinity could improve due to the riverbed and/or WBZ restoration, the rivers' water purity on the country scale might not improve and vice versa. At the same time, the quality of water in the Baltic Sea was framed as dependent on the co-ordinated efforts of all the Baltic Sea Basin countries. Therefore, programme combinations were said attainable where improvement of the water purity of the country's rivers did not lead to improvement of the water purity in the Baltic Sea and the other way around.

Each choice task included the status quo (no change) option together with two more programme alternatives with an associated annual change in the respondents' income.

5.2.2. Testing of impact of information on preferences

A very important issue rarely studied in the non-market valuation literature is whether during SP surveys we create preferences or discover them. This question is obviously not easy to answer. As in DCE we expect respondents to make trade-offs between studied attributes and their levels it's necessary that people understand what these attributes and their levels stand for. So a standard procedure is to explain these attributes and their levels to respondents before they trade them off. This often requires providing additional information on biological process or links between different components of environment that respondents could not be aware before. So it's likely that when a complex environmental good is being valued (as in our case) in the course of the survey the preferences are rather formed than discovered. In order to test how strong is this effect and to what extent the provided information shifts respondents' preferences we conducted an exercise in which respondents were asked to order photographs depicting different types of riverbeds and vegetation types.

Fig.3 – Rivers which differ from each other in their bed shape

Therefore, the aim of this part of our survey was to assess to what extent and in what direction the information provided in our survey shifts the respondents' preferences. To do so, the questionnaire included two series of photos depicting three different types of river bed shape (Fig.3) and five different riverine vegetation types (Fig.4). The half of respondents (i.e. 500 respondents in each country) were asked to rank the photos from most preferred to least preferred at the very beginning of the interview, before any information on the topic was provided (1st treatment). Whereas the second half of respondents (i.e. another 500 respondents in each country) was given the same task at the end of the survey, after reading the full scenario description (2nd treatment).

Fig.4 – Rivers which differ from each other by the riverine vegetation type

We would like to stress that exactly the same photographs which were used in the ordering exercises were also used in the survey when different vegetation types/riverbeds were introduced to respondents. Therefore, respondents in the 2^{nd} treatment when ordering the different landscapes were aware of ecosystem functions and services provided by each riverbed type and vegetation type associated with each photograph. This setting enabled us to test to what extent and in what direction (if any) the provided information shifted preferences regarding different riverbed and vegetation types.

5.3. DCE design and survey administering

The questionnaire was first elaborated in English (Supplement 1) and then translated into Danish, German and Polish (Supplement 2). The survey was administered as computer-assisted web interviews (CAWI) on representative samples of 1,000 respondents in each country in September of 2019. Each respondent was presented with twelve choice tasks. The combinations of attribute levels presented in each choice task were prepared in a way which maximised the amount of information revealed by respondents, conditional on our expectations regarding their preferences. These expectations (priors) were obtained through the pilot study conducted on a sample of 100 respondents in each country.

The final design was optimised for median Bayesian D-error of the MNL model (Scarpa & Rose 2008) based on the data from 300 interviews (100 from each of the countries). The design used Bayesian priors to account for the uncertainty associated with our imperfect knowledge of the true parameters (Bliemer et al. 2008). We randomized the order of choice tasks presented to each respondent to counter-balance possible ordering and anchoring effects (Day & Prades 2010). The same design composed of 36 choice-sets, divided into three blocks, has been used in the three studied countries. An example of a choice card is provided in Fig. 5.

A series of debriefing questions and socio-demographic data finalised the questionnaire in order to examine the respondents' attendance to DCE elements, their reasons for consequent opting out where appropriate, as well as respondents' attitudes to broad environmental issues.

5.4. Econometric modelling

In a DCE exercise, individuals are asked to identify their preferred choice *i* among a given set of *J* alternatives. The data analysis follows the Random Utility Model (RUM) (McFadden 1974). Under RUM, it is assumed that the observed choice from an individual *n* is the one she expects to provide her with the highest utility. Her utility function, U_{ni}, can be decomposed into a systematic part, V_{ni}, and a stochastic part, ε_{ni}. The probability P_{ni} that the decision maker *n* chooses alternative *i* instead of another alternative *j* of the choice set is $P_{ni} = \Pr(V_{ni} + \varepsilon_{ni} > V_{nj} + \varepsilon_{nj} \forall j \neq i)$. If ε_{ni} is assumed to be an independently and identically distributed extreme value type I (Train 2003), this probability has a closed form multinomial logit (MNL) expression,

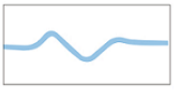
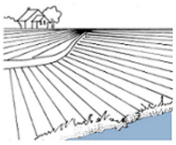
	Current management continuation	Programme A (Change in management)	Programme B (Change in management)
On the country's scale and in the Baltic Sea			
Water quality of the rivers in 10 years **Water fit for bathing**	Medium For adults only	Good For adults and elder children	Very good For everyone without limitations
Water quality of the Baltic Sea in 30 years **Cyanobacteria blooms** **Days of bathing prohibition**	Bad Possible every summer, widespread 11-20 days	Very bad Possible every summer, extremely widespread Water is not suitable for bathing	Very bad Possible every summer, extremely widespread Water is not suitable for bathing
An appearance of the agricultural landscape in your place of residence neighbourhood (within 20km)			
Riverbed type	 Regulated straightened	 Regulated curvy	 Naturally meandering
Riverine vegetation type	 Intensive agriculture	 Wetland agriculture	 Wild marshes
Annual change in your income as a result of the programme implementation	0 EUR	Every German citizen will have to pay the tax of 50 EUR per year	Every German citizen will have to pay the tax of 25 EUR per year
Your choice	○	○	○

Fig. 5 – Choice-card example from the German questionnaire

$$P_{ni} = \frac{e^{\beta' x_{ni}}}{\sum_j e^{\beta' x_{nj}}} \qquad [1]$$

where x is a vector of variables and β is a vector of parameters.

Besides the MNL, the more advanced mixed logit models (MMNL) (McFadden 1974, Train 2003) were estimated for every country involved in order to account for preference heterogeneity. MMNL is any model whose choice probabilities take the form

$$P_{ni} = \int \frac{e^{\beta_n' x_{ni}}}{\sum_j e^{\beta_n' x_{nj}}} \phi(\beta | b, \Omega) d\beta, \qquad [2]$$

where: $\frac{e^{\beta_n' x_{ni}}}{\sum_j e^{\beta_n' x_{nj}}}$ is a standard logit formula, $\phi(\beta | b, \Omega)$ is the density of the random coefficients with mean *b* and covariance *Ω*. Thus, the logit expression can be treated as a special mixed logit case with β being fixed. Limitation of the standard MNL that it can represent only the systematic taste variation but not random taste variations is relaxed by assuming a mixing distribution that is not degenerated at fixed parameters. In the MMNL model we accounted for panel structure of the data and systematic taste variation.

For the MMNL model, all the non-monetary attributes were assumed to follow normal distribution while the cost coefficient was assumed to follow log-normal distribution in order to impose the theory-driven restriction that marginal utility of money is positive. We note that all parameters were allowed to be freely correlated. Since the integral in equation [2] cannot be evaluated analytically the probabilities have to be simulated; in each run 1000 Halton random draws were generated. The MMNL models for the three countries were estimated in preference space (Train & Weeks, 2005).

The utility function specification for each of the country-specific models of the both types included two dummy-coded variables associated with the levels of water purity in the rivers on country level in 10 years, three dummy-coded variables standing for the levels of water purity in the Baltic Sea in 30 years, two dummy-coded variables for the levels of riverbed sinuosity, three dummy-coded variables standing for the levels of riverside vegetation, a continuous monetary cost coefficient, and an alternative-specific constant for the status quo.

6. Results and discussion

6.1. Discrete Choice Experiment

All models were initially estimated in preference space and WTP were calculated. Aside from their nominal values, all WTP values were denominated in 2019 Euros adjusted by the purchasing power parity (PPP) factor in order to enable their direct comparison[3] across the three countries. The WTP estimates for MNL and MMNL models adjusted for purchasing power parity are reported in the Tables 2 and 3. Besides, modelling results for MNL models for the three countries in the

[3] Country specific PPP factors are calculated as ratios of the appropriate country's GDP 2019 estimate denominated in PPP-adjusted international dollars, over the same GDP denominated in nominal dollars (both values retrieved from World Economic Outlook Database, 2019. IMF.org. International Monetary Fund accessed on 25th October 2019). PPP factors equal 0.9, 1.15, and 2.27 for Denmark, Germany, and Poland accordingly.

preference space as well as WTP estimates denominated in the nominal Euros for the two models are reported in the Supplement 3 for reference purposes.

Since the results of the two models are basically in line, below we concentrate by default on the results of MMNL, a model considered superior as unlike MNL it enables addressing random taste heterogeneity and panel structure of the data.

Table 2 – WTP (MNL), EUR'2019 PPP

	Denmark		Germany		Poland	
	Coefficient	SE	Coefficient	SE	Coefficient	SE
Bad river water quality in 10 years[1]	-108.96***	12.07	-93.80 ***	8.93	-195.69***	20.09
Good river water quality in 10 years[1]	63.59***	9.77	61.47 ***	6.84	65.91***	18.23
Very bad Baltic water quality in 30 years[2]	-57.36**	13.81	-42.70 **	9.34	-63.56**	25.76
Medium Baltic water quality in 30 years[2]	102.60***	12.53	91.72 ***	9.19	229.70***	22.45
Good Baltic water quality in 30 years[2]	151.80***	13.00	119.09 ***	9.56	323.41***	22.07
Regulated curvy riverbed shape[3]	51.16***	10.27	20.64 ***	6.64	78.85***	18.39
Naturally meandering riverbed shape[3]	89.69***	10.81	50.37 ***	6.99	171.44***	18.64
Low-intesity agriculture[4]	119.54***	14.07	50.97 ***	8.39	114.92***	21.67
Wild marshes[4]	163.71***	15.55	94.97 ***	9.93	220.61***	24.22
Wetland agriculture[4]	172.13***	15.29	89.60 ***	9.33	231.14***	23.37

[1] Medium river quality is the reference level, [2] Bad Baltic water quality is the reference level, [3] regulated straightened riverbed is the reference level, [4] Intensive agriculture is the reference level.
***, **, * significance at 1%, 5%, 10% level.

All estimated WTP values but one (i.e. the mean of regulated curvy riverbed for Poland) are highly statistically significant, whereas the signs of all estimated parameters obtained for the three countries are consistent with a priori expectations. Statistical significance of all the estimated standard deviations across all the models (only one at the 5% level whilst the rest of them at 1% level) points at considerable heterogeneity of preferences.

Table 3 – WTP (MMNL), EUR'2019 PPP

	Denmark		Germany		Poland	
	Coefficient	SE	Coefficient	SE	Coefficient	SE
Bad river water quality in 10 years[1]	-93,80***	11,70	-101,46***	11,43	-166,83***	19,31
Good river water quality in 10 years[1]	101,42***	10,13	76,50***	9,50	153,64***	16,10
Very bad Baltic water quality in 30 years[2]	-68,40***	11,79	-49,53***	13,13	-106,14***	18,96
Medium Baltic water quality in 30 years[2]	80,12***	10,12	106,93***	10,60	153,03***	16,83
Good Baltic water quality in 30 years[2]	123,33***	13,33	150,28***	12,63	197,96***	19,63
Regulated curvy riverbed	24,02***	7,03	22,27***	7,47	14,15	9,46

	Denmark		Germany		Poland	
	Coefficient	SE	Coefficient	SE	Coefficient	SE
shape[3]						
Naturally meandering riverbed shape[3]	62,42***	8,91	63,59***	9,33	66,50***	12,17
Low-intesity agriculture[4]	86,02***	12,07	62,25***	10,60	90,79***	16,21
Wild marshes[4]	131,23***	15,07	107,11***	12,29	158,94***	19,12
Wetland agriculture[4]	135,11***	14,87	114,58***	11,93	155,25***	16,87
Standard deviations						
Bad river water quality in 10 years	78,65***	11,96	77,08***	11,04	130,18***	16,41
Good river water quality in 10 years	57,59***	11,06	101,99***	12,11	107,29***	15,77
Very bad Baltic water quality in 30 years	134,29***	17,17	141,49***	16,31	200,70***	25,43
Medium Baltic water quality in 30 years	75,79***	12,09	38,84***	11,38	39,63**	16,69
Good Baltic water quality in 30 years	127,17***	15,69	95,14***	11,69	163,33***	16,74
Regulated curvy riverbed shape	40,95***	11,08	73,03***	12,18	44,76***	16,31
Naturally meandering riverbed shape	78,61***	11,31	102,98***	13,00	110,69***	16,20
Extensive agriculture	82,03***	9,89	99,36***	9,54	172,65***	16,80
Wild marshes	110,12***	12,91	87,32***	13,19	90,70***	16,12
Wetland agriculture	58,80***	12,86	39,81***	11,56	78,69***	17,69

[1] Medium river quality is the reference level, [2] Bad Baltic water quality is the reference level, [3] regulated straightened riverbed is the reference level, [4] Intensive agriculture is the reference level.
***, **, * significance at 1%, 5%, 10% level.

6.1.1. WTP for water purity on the country and international level

Surprisingly, despite the fact that the Polish GDP per capita (PPP) is only about 55% of the Danish and 58% of the German, the WTP estimates of Polish respondents for contemplated improvement of ecosystem services at least have the comparable order of magnitude with regards to WTP of the Danish and German respondents. Moreover, WTP stated by Polish respondents in the most cases significantly outperforms that of Danish and German respondents. For instance, the Polish respondents are on average willing to pay 153.03 PPP-adjusted Euros per year for maintenance of the medium water quality in the Baltic Sea in 30 years from now, which is 1.91 times as much as the Danish respondents' WTP for the same measure, and 1.43 times as much as that of German respondents; the similar tendency applies to the majority of ecosystem services under scrutiny. Therefore, the Poles seem to be ceteris paribus willing to devote a higher proportion of their welfare in order to deliver improvements in ecosystem services of lowland small rivers, as compared to the citizens of wealthier countries of Western Europe.

As expected, the respondents in all three countries are willing to pay for improvement of the water quality in their country's rivers. Similarly, they are willing to pay for maintaining the current level of the Baltic Sea water quality, or its improvement to a good level with WTP values rising monotonically, which is an important aspect of the theoretical validity of the results. Indeed, the WTP increases with increasing quantity or quality of the good under consideration (Carson & Mitchell 1993), i.e. the results obtained in the three countries almost perfectly obey the scale (with the only exception of riverbed shape in case of Poland). Thus, in every country WTP monotonically rises with the greater water quality improvement both in the rivers and in the Baltic Sea. At the same time, the levels of water quality worse than status quo are associated with negative WTP.

Respondents in all the three countries have stated their substantial WTP for improvement of water purity in the Baltic Sea which is a typical example of international public good (Markowska&Żylicz 1995). We also see that consistently in all the studied countries the WTP estimates for improvement of the water quality in the Baltic Sea are substantially larger than in the countries' rivers. For example, the WTP of German respondents for the highest level of water quality in the Baltic Sea is 150.28 EUR (with bad water quality being the base) and is 1.96 times higher than their WTP for the highest level of water quality in the rivers. In Poland the WTP for the highest level of the water quality in the Baltic Sea is 197.96 EUR and is 1.29 higher than the WTP for the corresponding level in the rivers, whereas in Denmark the WTP is 123.33 EUR and is 1.22 larger than the WTP for the highest level of the water quality in the rivers. The positive WTP estimates for water quality in the Baltic Sea are in line with results of earlier studies on WTP for improvements of Baltic Sea water quality (Athiainen & Vanhatalo 2012, Athiainen et al. 2016, Sagebiel et al. 2016).

6.1.2. WTP for the local-level management vs higher levels management scenarios

Another important finding is the uniform pattern in difference of WTP for improvement of ecosystem services between the local and country levels. Following the scenario, the improvement on the country level would be achieved by means of the same transformations of the small rivers as are contemplated on the local level, namely by restoration of riverbed shape tortuosity and riparian vegetation in the WBZ, and/or by reduction of use of fertilisers. Furthermore, it was explicitly communicated that an improvement of ecosystem services on the country level not necessarily entails the same improvement on the local level, and the other way around. Therefore, the respondents were assumed to make trade-offs in the DCE exercise between the greater delivery of local-scale and country-scale improvements of ecosystem services generated by the lowland small rivers.

If all the contemplated attributes are considered, the respondents state greater WTP for the country-scale improvement of ecosystem services. Thus, regarding the programme implying biggest possible improvement (i.e. the programme comprising achievement of the good quality in the country's rivers in 10 year and in the Baltic Sea in 30 years, as well as restoration of the natural meandering and either wild marshes or wetland agriculture type of riparian vegetation within the 20 km from the representative consumer's place of residence), WTP for local improvement would make ca. 47 per cent in case of Denmark, 44 per cent for Germany, and 39 per cent for Poland.

However, if the Baltic Sea water purification attribute was dropped and improvements of ecosystem services of small rivers alone were considered, WTP for improvement of riverine ecosystem services on the local level strikingly outperforms the WTP for improvement of water quality in rivers on the country level: the appropriate propensity ratio ranges from 2.33 times in favour of the local programme attributes in case of Germany to 1.44 in case of Poland, which means that respondents in all the three countries put having semi-intact small rivers (be they natural or restored) in their closest neighbourhood before anywhere else.

In this respect, we observe a very similar pattern across three countries regarding preferences for the two attributes at the local level (i.e. within a radius of 20 km within the consumer's place of residence), namely riverbed shape and vegetation type. We see that in all the countries stated WTP for restoration of natural meandering significantly outperforms WTP for introduction of the regulated curvy riverbed shape. In turn, in two of three countries (with the exception of Poland) regulated curvy riverbed type is consistently preferred over the regulated and straightened riverbed type. We also see that Intensive agriculture is the least preferred vegetation type across the countries. On the contrary, Wild marshes and Wetland agriculture – the options implying the highest level of ecosystem services of those contemplated in the DCE – have been assigned the highest and very similar WTP by the respondents in the three studied countries. The 95% confidence intervals for these two types of vegetation largely overlap with the WTP values being over 100 EUR in each of the countries. We hypothesise that an insignificant difference between WTP stated for Wild marshes and Wetland agriculture can be explained by similar level of ecosystem services (i.e. water purity, biodiversity, and flood control) associated with these two options in the survey scenario.

Interestingly, we observe substantial WTP values for the most preferred levels of the riverbed and vegetation types. For example, WTP for meandering rivers with respect to regulated straightened rivers varies from 62.42 EUR in Denmark to 66.50 EUR in Poland with WTP for Germany equal to 63.59 EUR being in between. Therefore, very similar, large and highly significant WTP for re-meandering of the riverbeds and restoration of WBZ to small lowland rivers is the overall pattern in all three Baltic Sea Basin countries involved, which makes their re-naturalisation a socially desirable policy. This finding is in line with other recent contributions from European countries, which contemplate rewilding or re-naturalisation of historically human-transformed ecosystems and restoration of their functions and services (e.g., Rayanov et al. 2018, Logar&Brouwer 2018, Valasiuk et al. 2018). Moreover, respondents in three countries put restoration of naturally meandering riverbeds and Wild marshes (or Wetland agriculture) WBZ on the local level before improvement of water quality in rivers on the country level: the appropriate WTP ratio in favour of the local programme attributes ranges from 1.44 times in case of Poland to 2.33 in case of Germany.

6.1.3. WTP for local-level small rivers' management as indicator of cultural ecosystems' value

Therefore, stated WTP for the locally-contemplated programme attributes is to be brought under particular scrutiny. Unlike the narrowly defined country-scale attributes reflecting two well-specified services of riverine ecosystems, namely – nutrient cycling and water purification in rivers country-wide, and down in the Baltic Sea, the local-scale attributes employed in this study acquire a whole bundle of ecosystem services. Obviously, aside from nutrients cycling and water purification for recreational means, riverbed shape and riparian vegetation type on the local level affect supply of such ecosystem services as supporting biodiversity (i.e. being a habitat, refugium and/or migration corridor for biota), flood control and defence locally and downstream – which was explicitly communicated in the survey scenario. However, it might be reasonable to assume, that notwithstanding their aforementioned ecosystem functions, these two characteristics of lowland small rivers are valuable cultural ecosystem services per se, thus bearing a non-zero intrinsic value.

Indeed, meandering riverbed as well as mosaic and scenic appearance of wild riverine marshes might be of positive aesthetic value for people with appropriate landscape tastes. Furthermore, they might be of sentimental value for people who grew up in the farmland environment then comprising semi-natural small rivers. Last but not least, the very traditional names of small rivers (currently forgotten by many as a side-effect of their transformation into the likes of canals or ditches) might be a valuable part of the local legacy and thus constitute a substrate for appropriate cultural ecosystem services. Quantification and valuation of cultural ecosystem services face many methodological issues, leading towards their frequent omission in quantitative analyses, cost-benefit analysis included (Milcu et al. 2013). For instance, in their pivotal contribution on valuation of global ecosystem services, Costanza et al. (1997) failed to assign any finite monetary value to cultural services of the world's rivers and lakes. Meanwhile, with small rivers in the farmland landscape, it is difficult to a priori judge upon the people aesthetic preferences, as any pattern in them is legitimate.

Obviously, on the grounds of this study's results we are not in a position to isolate the value of the aforementioned cultural ecosystem services out from the total WTP stated in favour of the restoration of semi-intact small rivers in the closest proximity from the respondents' places of residence. However, given that a corresponding WTP stated (on average) for the local improvement of riverine ecosystem services appeared positive, substantial, and several times higher than the WTP for improvement of rivers' water quality on the country level, it would be a reasonable assumption that at least with some proportion of respondents the former WTP acquires a value of cultural ecosystem services generated by the semi-intact small lowland rivers, including some aesthetical value. In this connection, a high degree of riverbed tortuosity and high naturalness of the riparian vegetation with the small rivers could be considered useful indicators of cultural ecosystem services of the farmland landscapes in the Baltic Sea Basin and elsewhere in lowland Europe, when higher costs and losses associated with stream restoration can be justified by its aesthetic benefits (Kenney et al. 2012).

6.2. Verification – answers to simple intermediate questions

Our DCE allows for quantifying people's preferences in monetary terms. The validity of this method in the context of abstract public goods which people are not familiar with is still subject of a scientific debate (Train & McFadden 2017, Johnston et al. 2017). There are many potential biases which are associated with stated preference methods. For example, comparing three alternative programmes across five attributes and numerous levels in twelve choice situations could be simply a cognitively challenging task for some people. In order to make sure that the obtained results are not an artefact of potential biases associated with stated preference methods, in addition to the DCE we asked the respondents simple, straightforward questions about their preferences regarding the studied attributes. The questions and the answers to these questions are presented in Table 4.

Table 4 – Answers to the intermediate questions

	Poland	Denmark	Germany
If you had influence on the landscape look in the closest agriculturally used surrounding of your place of residence, would you prefer that the small rivers in the rural landscape were close to natural or rather be regulated?			
Regulated	12,7%	7,5%	4,8%
Natural	83,4%	73,4%	88,4%
Don't know	3,9%	19,1%	6,8%
Should the government in your opinion undertake mitigation measures aimed at improvement of the water purity in Danish/German/Polish rivers, even if it would imply the necessity of raising taxes which you pay?			
Yes	77,4%	62,6%	71,8%
No	9,4%	15,3%	14,2%
Don't know	13,1%	22,0%	13,9%
Should the government in your opinion undertake mitigation measures aimed at improvement of the water purity in the Baltic Sea even if it would imply the necessity of raising taxes which you pay?			
Yes	74,4%	58,7%	68,4%
No	10,0%	16,8%	16,1%
Don't know	15,6%	24,5%	15,5%
Would you agree that restoration activities should be undertaken in the nearest agricultural neighbourhood of your place of residence with the aim of restoring the natural look of small rivers even if it would mean necessity of increasing taxes you pay?			
Yes	85,3%	69,2%	76,9%
No	6,3%	8,8%	9,1%
Don't know	8,4%	22,0%	13,9%
Would you like the wetland buffer zones separating fields from the rivers to be created in the closest agriculturally used neighbourhood of your place of residence even if it would mean necessity of increasing taxes which you pay?			
Yes	74,6%	66,6%	76,1%
No	8,9%	7,3%	9,3%
Don't know	16,6%	26,1%	14,7%

The respondents were asked these questions in separation that is throughout the survey after each block that introduced a given topic. The answers are in line with the DCE results, indicating a high support for re-naturalisation of small rivers in all three countries. Furthermore, people agree to tax increases to finance re-naturalisation.

6.3. Orderings landscapes and impact of information on preferences

For each riverbed type and vegetation type we report the share of respondents who chose a given type of landscape as the most preferred. Subsequently the landscapes were ranked from the most attractive – RANK 1 (i.e. the highest share of respondents chose this landscape as the most attractive) to the least attractive RANK 3.

In Table 5 we present the results of orderings of the photographs for different riverbed types. As in none of the treatments the rank has changed, only one number for both treatments is reported.

Table 5 – Ranking of different riverbed types

% of being most attractive	Before	After
PL	6,6%	8,3%
DK	11,6%	7,9%
DE	12,2%	9,4%
	Rank 3	
Borda count		
PL	1370	1386
DK	1310	1355
DE	1280	1345
	Rank 3	

% of being most attractive	Before	After
PL	8,9%	10,6%
DK	16,8%	18,7%
DE	12,4%	18,3%
	Rank 2	
Borda count		
PL	1019	983
DK	981	961
DE	1054	969
	Rank 2	

	Before	After
% of being most attractive		
PL	84,6%	81,1%
DK	71,7%	73,4%
DE	75,5%	72,3%
	Rank 1	
Borda count		
PL	611	631
DK	710	685
DE	666	686
	Rank 1	

We see that consistently in all three countries in both treatments naturally meandering rivers are by far the most preferred riverbed type. The shares of respondents who chose regulated curvy riverbed as the most preferred is slightly higher than regulated straightened riverbed. Interestingly, the same ordering is obtained in all three countries for both treatments.

In addition to calculate the shares we also tested whether the frequency of selecting a given riverbed type as the most preferred varied in the two treatments. To do so we carried out Pearson's chi-squared test. It tests a null hypothesis stating that the frequency distribution of frequencies observed in 1st treatment is consistent with a distribution in the 2nd treatment. The obtained test statistic is Pearson $\chi^2(8) = 14.04$, translating into a p-value of 0.081. Hence we fail to reject the null hypothesis that frequency distributions in both treatments are homogenous at a significance level of 5%.

Ranking riverbed types based on the shares of respondents who chose a given landscape as the most preferred does not take into account the respondents' complete ordering of all alternatives. An alternative method of determining a social preference relation is the Borda count, also known as rank-order voting (Adelsman & Whinston,1977)[4]. Each voter is asked to rank all of the alternatives. If there are for example three alternatives, then first choice gets a 1, second choice a 2, and on the third choice gets a 3. The voters' scores for each alternative are then summed up over all voters. As can be seen in Table 4 the ranking based on selecting the preferred option and the Borda count is the same. We also note that especially for Poland the Borda count for the photo depicting a naturally meandering riverbed is very close to its lower limit indicating very strong preferences for this riverbed type.

[4] The total score for an alternative is called its Borda count. For any two alternatives, *x* and *y*, if the Borda count of *x* is smaller than or the same as the Borda count for *y*, then *x* is socially at least as good as *y*. Given that there are 500 respondents in our exercise in each treatment the lower limit for Borda count is 500 and the upper limit is 1500.

Exactly the same steps as to analysing riverbed types have been applied to vegetation types. In Table 6 we present the results of orderings of the photographs for different vegetation types.

Looking at Table 6, we see that, consistently in all three countries, in both treatments wild marshes overgrown by shrubs and trees[5] is by far the most preferred vegetation type, whereas the second most preferred landscape was a photograph which was presented to respondents in the survey as wetland agriculture. This result holds for both ordering methods that we used. The least attractive landscape type was Intensive agriculture, which was not ranked as the least attractive landscape only for Germany in the 2nd treatment. When looking at Borda count, exactly the same ordering is obtained in both treatments i.e.

Wild marshes overgrown by shrubs and trees >Wetland agriculture >

Reeds-dominated wild marshes > Low-intensity agriculture > Intensive agriculture.

Interestingly, relatively little appreciation got Reeds-dominated wild marshes. This type of vegetation was perceived as substantially less attractive than Wetland agriculture and as only slightly more attractive than Low-intensity agriculture using Borda count and in one case (i.e. Poland in 1st treatment) had a smaller number of most attractive choices than Low-intensity farmland landscape.

It's beyond the scope of this article to answer what factors drive visual attractiveness of different landscapes but the obtained results seem to indicate that people tend to dislike landscapes which are regular and ordered, it looks that more appreciation gets landscapes which are mosaic with diverse vegetation patches. Our results also tend to indicate that the presence of trees and shrubs is an important positive factor.

Finally, as in the case of riverbed type we conducted a Pearson's chi-squared test. The obtained test statistic is Pearson $\chi^2(14) = 39.93$. This test statistic is much higher than the critical value for the 0.05 level and we reject the null hypothesis that both frequency distributions are the same. This implies that in the case of vegetation types we observe a significant impact of provided information on obtained orderings of different landscapes. On average this shift is positive, that is providing information on ecosystem services associated with different vegetation types results in higher appreciation of landscapes which are associated with higher provision of ecosystem services.

Table 6 – Ranking of different vegetation types

[5] Unlike in the photoset shown to the respondents before or after the DCE part, this type of riparian vegetation was not presented on the DCE choice-cards, since it belongs to the wider Wild Marshes class of vegetation, and it is less widespread as compared to the more homogenous reeds-dominated Wild Marshes.

Most attractive

	Before	After
PL	4,9%	2,6%
DK	4,1%	5,6%
DE	3,2%	4,9%

Borda count

	Before	After
PL	2016	2181
DK	2036	2099
DE	2134	2212

RANK

	Before	After
PL	5	5
DK	5	5
DE	5	4

RANK=5

Most attractive

	Before	After
PL	6,3%	6,0%
DK	6,8%	6,5%
DE	6,3%	4,5%

Borda count

	Before	After
PL	1807	1863
DK	1831	1854
DE	1822	1847

RANK

	Before	After
PL	3	4
DK	4	4
DE	4	5

RANK=4

Most attractive

	Before	After
PL	27,4%	22,6%
DK	30,4%	25,5%
DE	26,6%	20,4%

Borda count

	Before	After
PL	1047	1170
DK	1035	1159
DE	1090	1158

RANK

	Before	After
PL	2	2
DK	2	2
DE	2	2

RANK=2

Most attractive	Before	After
PL	56,6%	60,9%
DK	51,9%	48,4%
DE	57,0%	58,4%

Borda count	Before	After
PL	909	850
DK	947	1003
DE	883	872

RANK	Before	After
PL	1	1
DK	1	1
DE	1	1

RANK=1

Most attractive	Before	After
PL	4,9%	8,0%
DK	6,8%	14,0%
DE	7,0%	11,5%

Borda count	Before	After
PL	1721	1436
DK	1651	1385
DE	1571	1412

RANK	Before	After
PL	4	3
DK	3	3
DE	3	3

RANK=3

7. Conclusions and policy implications

Firstly, the citizens of Denmark, Germany, and Poland are willing to pay substantial amounts for improvement of ecosystem services and functions dependent of the intactness of the small rivers in the farmland landscape in the lowland part of their countries. The Baltic Sea and rivers country-wide, but also local attributes i.e. riverbed shape and riparian vegetation were assigned positive and high welfare estimates. For the overwhelming majority of small rivers, it implies re-meandering of their riverbed shapes, rewetting of floodplains, and restoration of wild marshes or development of paludiculture. We see that rewilding/re-naturalisation of rivers could get popular support, which conforms with the findings of the earlier studies (e.g. Rayanov et al. 2018). Besides, the citizens of the three littoral countries state high willingness to pay, for maintenance or improvement of the current level of water purity in the Baltic Sea in a perspective of thirty years from the start of the programme implementation. Their considerable preferences toward the Baltic Sea water purity lay grounds for the multilateral action in this respect.

Secondly, the preferences in favour of small rivers restoration stated in three countries can be characterised as "counter-NIMBY[6]", since the respondents clearly put re-meandering and restoration of the semi-natural riverine ecosystems in their neighbourhoods before the river water purification contemplated on the country scale. This tendency can be explained by the bundle of ecosystem services arising from the local small rivers' restoration and/or conservation action, including typically difficult-to-quantify aesthetic values. Therefore, the observable natural characteristics, such as meandering riverbeds and wild-looking riparian vegetation are highly attractive for the people and they can serve as proxy indicators of cultural ecosystem services.

Finally, the obtained results do not seem to be an artefact of DCE as the patterns in preferences are in line with the results of the alternative methods used. Thus, ranking of the photos depicting visual characteristics of small rivers yielded the very similar results irrespectively of whether the respondents were asked to provide their ordering in the very beginning of the interview or after learning the scenario and completing DCE task. Information provided in the questionnaire triggered only minor changes in the respondents' landscape tastes, so that the preferences appeared robust across the treatments and they are in favour of more spontaneous appearance of small rivers. It seems that wild-looking rivers are simply attractive for people. Furthermore, this preference pattern has also been confirmed by respondents' answers to direct intermediate questions, where reluctant and neutral attitudes towards contemplated rewilding/re-naturalisation were substantially less frequent than approval thereof. This circumstance might point at good knowledge level of the respondents in three countries about small rivers, their current state and restoration prospects, riverine ecosystem services, and perhaps more generally – about the urgency to mitigate the accelerating environmental crisis.

[6] NYMBY is an abbreviation standing for „Not in my backyard" statement typically meaning negative preferences for a local action combined with indifference or even positive preferences for the same action carried out elsewhere.

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Supplementary material

Supplement 3

MNL modelling results in preference space

	Coefficient		s.e.	z	Prob |z|>Z*	95%	Confidence interval
				Denmark			
SQ	-0,25844	***	0,06481	-3,99	0,0001	-0,38547	-0,13142
Change of income (tax)	-0,00053	***	0,00003	-16,16	0,0000	-0,00059	-0,00046
Bad river water quality in 10 years	-0,48847	***	0,04549	-10,74	0,0000	-0,57763	-0,39931
Good river water quality in 10 years	0,28509	***	0,04496	6,34	0,0000	0,19698	0,3732
Very bad Baltic water quality in 30 years	-0,25717	***	0,06289	-4,09	0,0000	-0,38044	-0,13391
Medium Baltic water quality in 30 years	0,45997	***	0,04935	9,32	0,0000	0,36325	0,55669
Good Baltic water quality in 30 years	0,68055	***	0,04651	14,63	0,0000	0,5894	0,7717
Regulated curvy river bedshape	0,22935	***	0,04382	5,23	0,0000	0,14346	0,31524
Naturaly meandering river bedshape	0,40209	***	0,04318	9,31	0,0000	0,31745	0,48673
Extensive agriculture	0,53594	***	0,04991	10,74	0,0000	0,43812	0,63376
Wild marshes	0,73394	***	0,05548	13,23	0,0000	0,6252	0,84267
Wetland agriculture	0,77171	***	0,05426	14,22	0,0000	0,66536	0,87807
				Germany			
SQ	-0,08614		0,06467	-1,33	0,1829	-0,21288	0,04061
Change of income (tax)	-0,00177	***	0,00011	-15,77	0,0000	-0,00199	-0,00155
Bad river water quality in 10 years	-0,62754	***	0,04581	-13,7	0,0000	-0,71732	-0,53775
Good river water quality in 10 years	0,41129	***	0,04466	9,21	0,0000	0,32375	0,49883
Very bad Baltic water quality in 30 years	-0,28571	***	0,06317	-4,52	0,0000	-0,40951	-0,16191
Medium Baltic water quality in 30 years	0,61362	***	0,04918	12,48	0,0000	0,51723	0,71002
Good Baltic water quality in 30 years	0,79678	***	0,04655	17,12	0,0000	0,70555	0,88802
Regulated curvy river bedshape	0,13806	***	0,04356	3,17	0,0015	0,05269	0,22343
Naturaly meandering river bedshape	0,33697	***	0,04305	7,83	0,0000	0,25259	0,42135
Extensive agriculture	0,34098	***	0,04907	6,95	0,0000	0,2448	0,43717
Wild marshes	0,63537	***	0,05449	11,66	0,0000	0,52858	0,74217
Wetland agriculture	0,59948	***	0,05329	11,25	0,0000	0,49503	0,70392
				Poland			

SQ	-0,22479	***	0,06514	-3,45	0,0006	-0,35246	-0,09712
Change of income (tax)	-0,00554	***	0,00024	-23,23	0,0000	-0,00601	-0,00507
Bad river water quality in 10 years	-0,47678	***	0,04534	-10,52	0,0000	-0,56565	-0,38791
Good river water quality in 10 years	0,16057	***	0,04559	3,52	0,0004	0,07121	0,24993
Very bad Baltic water quality in 30 years	-0,15485	**	0,06347	-2,44	0,0147	-0,27925	-0,03046
Medium Baltic water quality in 30 years	0,55963	***	0,05068	11,04	0,0000	0,4603	0,65897
Good Baltic water quality in 30 years	0,78793	***	0,04814	16,37	0,0000	0,69357	0,88228
Regulated curvy river bedshape	0,1921	***	0,0441	4,36	0,0000	0,10566	0,27854
Naturaly meandering river bedshape	0,41768	***	0,04313	9,68	0,0000	0,33314	0,50222
Extensive agriculture	0,27999	***	0,04982	5,62	0,0000	0,18236	0,37763
Wild marshes	0,53748	***	0,05546	9,69	0,0000	0,42878	0,64618
Wetland agriculture	0,56313	***	0,05372	10,48	0,0000	0,45783	0,66843

WTP (MNL), EUR'2019 Nominal

	Coefficient		s.e.	z	Prob |z|>Z*	95%	Confidence interval
				Denmark			
Bad river water quality in 10 years	-120,914	***	13,39596	-9,03	0,0000	-147,17	-94,658
Good river water quality in 10 years	70,5707	***	10,84092	6,51	0,0000	49,3229	91,8185
Very bad Baltic water quality in 30 years	-63,6597	***	15,32599	-4,15	0,0000	-93,6981	-33,6213
Medium Baltic water quality in 30 years	113,858	***	13,90695	8,19	0,0000	86,601	141,115
Good Baltic water quality in 30 years	168,46	***	14,42131	11,68	0,0000	140,195	196,725
Regulated curvy river bedshape	56,7718	***	11,39176	4,98	0,0000	34,4443	79,0992
Naturaly meandering river bedshape	99,5304	***	11,99483	8,3	0,0000	76,0209	123,0398
Extensive agriculture	132,664	***	15,61569	8,5	0,0000	102,058	163,27
Wild marshes	181,676	***	17,2619	10,52	0,0000	147,843	215,508
Wetland agriculture	191,026	***	16,97308	11,25	0,0000	157,759	224,293
				Germany			
Bad river water quality in 10 years	-81,5334	***	7,7591	-10,51	0,0000	-96,741	-66,3259
Good river water quality in 10 years	53,4377	***	5,94668	8,99	0,0000	41,7824	65,093
Very bad Baltic water quality in 30 years	-37,1212	***	8,11818	-4,57	0,0000	-53,0326	-21,2099
Medium Baltic water quality in 30 years	79,726	***	7,98546	9,98	0,0000	64,0748	95,3772

	Coefficient		s.e.	z	Prob |z|>Z*	95%	Confidence interval
Good Baltic water quality in 30 years	103,523	***	8,30857	12,46	0,0000	87,239	119,808
Regulated curvy river bedshape	17,9374	***	5,77387	3,11	0,0019	6,6208	29,2539
Naturaly meandering river bedshape	43,7813	***	6,0799	7,2	0,0000	31,865	55,6977
Extensive agriculture	44,3028	***	7,29113	6,08	0,0000	30,0125	58,5932
Wild marshes	82,5519	***	8,62865	9,57	0,0000	65,64	99,4637
Wetland agriculture	77,8879	***	8,10795	9,61	0,0000	61,9966	93,7791
				Poland			
Bad river water quality in 10 years	-86,0456	***	8,83475	-9,74	0,0000	-103,3614	-68,7298
Good river water quality in 10 years	28,9787	***	8,01649	3,61	0,0003	13,2667	44,6907
Very bad Baltic water quality in 30 years	-27,9469	**	11,32465	-2,47	0,0136	-50,1428	-5,751
Medium Baltic water quality in 30 years	100,999	***	9,8703	10,23	0,0000	81,653	120,344
Good Baltic water quality in 30 years	142,2	***	9,70427	14,65	0,0000	123,18	161,22
Regulated curvy river bedshape	34,6688	***	8,08662	4,29	0,0000	18,8193	50,5183
Naturaly meandering river bedshape	75,3802	***	8,19501	9,2	0,0000	59,3182	91,4421
Extensive agriculture	50,5315	***	9,52633	5,3	0,0000	31,8603	69,2028
Wild marshes	97,0003	***	10,65158	9,11	0,0000	76,1236	117,877
Wetland agriculture	101,63	***	10,27529	9,89	0,0000	81,491	121,769

WTP (MMNL), EUR'2019 Nominal

	Coefficient		s.e.	z	Prob |z|>Z*	95%	Confidence interval
				Denmark			
Bad river water quality in 10 years	-104,096	***	12,98562	-8,02	-129,547	-78,644	-104,096
Good river water quality in 10 years	112,556	***	11,2406	10,01	90,524	134,587	112,556
Very bad Baltic water quality in 30 years	-75,9045	***	13,08799	-5,8	-101,5565	-50,2525	-75,9045
Medium Baltic water quality in 30 years	88,9183	***	11,23299	7,92	66,9021	110,9346	88,9183
Good Baltic water quality in 30 years	136,867	***	14,79577	9,25	107,868	165,866	136,867
Regulated curvy river bedshape	26,6557	***	7,79847	3,42	11,371	41,9404	26,6557
Naturaly meandering river bedshape	69,2702	***	9,88519	7,01	49,8956	88,6448	69,2702
Extensive agriculture	95,4647	***	13,39968	7,12	69,2019	121,7276	95,4647
Wild marshes	145,637	***	16,72202	8,71	112,862	178,411	145,637

	Coefficient		s.e.	z	Prob |z|>Z*	95%	Confidence interval
Wetland agriculture	149,934	***	16,49684	9,09	117,6	182,267	149,934
			Standard deviation				
Bad river water quality in 10 years	87,2854	***	13,27709	6,57	61,2628	113,3081	87,2854
Good river water quality in 10 years	63,9139	***	12,27247	5,21	39,8603	87,9674	63,9139
Very bad Baltic water quality in 30 years	149,031	***	19,05783	7,82	111,678	186,383	149,031
Medium Baltic water quality in 30 years	84,1105	***	13,41926	6,27	57,8092	110,4117	84,1105
Good Baltic water quality in 30 years	141,125	***	17,4142	8,1	106,994	175,257	141,125
Regulated curvy river bedshape	45,4489	***	12,30127	3,69	21,3388	69,5589	45,4489
Naturaly meandering river bedshape	87,2375	***	12,55277	6,95	62,6345	111,8405	87,2375
Extensive agriculture	91,0299	***	10,97919	8,29	69,5111	112,5488	91,0299
Wild marshes	122,206	***	14,32846	8,53	94,123	150,289	122,206
Wetland agriculture	65,249	***	14,26713	4,57	37,286	93,2121	65,249
			Germany				
Bad river water quality in 10 years	-88,192	***	9,93788	-8,87	-107,6698	-68,7141	-88,192
Good river water quality in 10 years	66,496	***	8,25405	8,06	50,3184	82,6737	66,496
Very bad Baltic water quality in 30 years	-43,0513	***	11,41565	-3,77	-65,4255	-20,677	-43,0513
Medium Baltic water quality in 30 years	92,9535	***	9,21635	10,09	74,8898	111,0172	92,9535
Good Baltic water quality in 30 years	130,633	***	10,97537	11,9	109,122	152,145	130,633
Regulated curvy river bedshape	19,3592	***	6,4971	2,98	6,6252	32,0933	19,3592
Naturaly meandering river bedshape	55,2792	***	8,10989	6,82	39,3841	71,1743	55,2792
Extensive agriculture	54,1158	***	9,21156	5,87	36,0614	72,1701	54,1158
Wild marshes	93,1115	***	10,68556	8,71	72,1681	114,0548	93,1115
Wetland agriculture	99,5987	***	10,36835	9,61	79,2771	119,9203	99,5987
			Standard deviation				
Bad river water quality in 10 years	67,0043	***	9,59345	6,98	48,2015	85,8071	67,0043
Good river water quality in 10 years	88,655	***	10,52661	8,42	68,0232	109,2867	88,655
Very bad Baltic water quality in 30 years	122,99	***	14,17943	8,67	95,199	150,781	122,99
Medium Baltic water quality in 30 years	33,7633	***	9,88904	3,41	14,3811	53,1454	33,7633
Good Baltic water quality in 30 years	82,7056	***	10,16516	8,14	62,7823	102,6289	82,7056
Regulated curvy river bedshape	63,4829	***	10,59023	5,99	42,7264	84,2393	63,4829
Naturaly meandering	89,5161	***	11,30246	7,92	67,3636	111,6685	89,5161

	Coefficient		s.e.	z	Prob |z|>Z*	95%	Confidence interval
river bedshape							
Extensive agriculture	86,3744	***	8,28859	10,42	70,1291	102,6198	86,3744
Wild marshes	75,9023	***	11,46223	6,62	53,4367	98,3679	75,9023
Wetland agriculture	34,6032	***	10,04521	3,44	14,9149	54,2914	34,6032
Poland							
Bad river water quality in 10 years	-73,3565	***	8,49084	-8,64	-89,9983	-56,7148	-73,3565
Good river water quality in 10 years	67,557	***	7,07935	9,54	53,6817	81,4322	67,557
Very bad Baltic water quality in 30 years	-46,668	***	8,33806	-5,6	-63,0103	-30,3257	-46,668
Medium Baltic water quality in 30 years	67,2874	***	7,39824	9,1	52,7871	81,7877	67,2874
Good Baltic water quality in 30 years	87,0423	***	8,62939	10,09	70,129	103,9555	87,0423
Regulated curvy river bedshape	6,22244		4,1616	1,5	-1,93415	14,37903	6,22244
Naturaly meandering river bedshape	29,24	***	5,35057	5,46	18,7531	39,7269	29,24
Extensive agriculture	39,9193	***	7,12918	5,6	25,9463	53,8922	39,9193
Wild marshes	69,8872	***	8,40557	8,31	53,4126	86,3618	69,8872
Wetland agriculture	68,2644	***	7,41585	9,21	53,7295	82,7992	68,2644
Standard deviation							
Bad river water quality in 10 years	57,2406	***	7,2166	7,93	43,0963	71,3849	57,2406
Good river water quality in 10 years	47,1754	***	6,93237	6,81	33,5882	60,7625	47,1754
Very bad Baltic water quality in 30 years	88,2484	***	11,18152	7,89	66,3331	110,1638	88,2484
Medium Baltic water quality in 30 years	17,4232	**	7,33679	2,37	3,0434	31,8031	17,4232
Good Baltic water quality in 30 years	71,8162	***	7,3599	9,76	57,3911	86,2413	71,8162
Regulated curvy river bedshape	19,6821	***	7,17202	2,74	5,6252	33,739	19,6821
Naturaly meandering river bedshape	48,6713	***	7,12267	6,83	34,7112	62,6315	48,6713
Extensive agriculture	75,9145	***	7,38521	10,28	61,4397	90,3892	75,9145
Wild marshes	39,8815	***	7,08596	5,63	25,9932	53,7697	39,8815
Wetland agriculture	34,5997	***	7,77956	4,45	19,3521	49,8474	34,5997