Imperial College London

Developing green interventions for the implementation of the Water Framework Directive through systems-based economic approaches

A thesis submitted to Imperial College London for the degree of Doctor of Philosophy

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2022

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DECLARATION OF OWN WORK

I declare that this thesis “Developing green interventions for the implementation of the Water Framework Directive through systems-based economic approaches” is entirely my own work and that where any material could be construed as the work of others, it is fully cited and referenced, and/or with appropriate acknowledgement given.

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ABSTRACT

Across the globe, society faces significant environmental challenges. Deforestation, intensive agriculture, pollution, and overexploitation of natural resources among other pressures create the conditions for natural capital depletion, ultimately jeopardizing economic development. To reverse such trends in the water sector and avoid the collapse of inland water ecosystems, the European Commission introduced the Water Framework Directive. A distinct example of integrated water resources management, the Directive aims to provide a holistic approach of managing inland waters efficiently by substituting previous fragmented legislation that focused on managing different aspects of water resources. Still, despite progress in the Directive's implementation, approximately half of the EU's surface waters are not in good condition.

Analysing the efforts of Member States to assess the socio-economic dimensions of water ecosystems and develop programmes of measures to achieve the objectives of the Directive, indicates a lack of understanding of what and how should be assessed. The information included in the River Basin Management Plans denotes insufficient connections between pressures and interventions, as well as poor economic analysis that often disregards a great portion of benefits and costs associated with transitioning towards a desired state. To support the implementation of the Directive and to provide a systems thinking perspective on environmental management decisions, the presented work blends ecosystem services with economic methodologies, develops integrated tools and approaches and tests their applicability in real cases. Findings show that such tools improve the robustness of the socioeconomic assessments of proposed measures and have the potential to engage stakeholders in the process. Additionally, they enable the use of natural capital accounting methodologies to improve the understanding of the connection between management interventions and the overall system status. Furthermore, the included research assesses the effectiveness of nature-based approaches as interventions on the system to influence the interactions of components within socioecological systems and drive them towards the desired state. Additionally, given the way economic dimensions permeate decision-making processes and current debates on environmental management, the current thesis proposes an alternative vision for the sustainable use of natural resources.

Overall, the undertaken research demonstrates the need for using economic tools through a systems prism for the implementation of the Water Framework Directive and environmental policies in general to deliver socioecological improvements.

ACKNOWLEDGEMENTS

Though the cover states the name of the researcher who developed this thesis, I would also like to name all of those, who to varying degrees provided assistance, encouragement, and guidance, and without whom this work would not have been completed.

First and foremost, I would like to express my sincere gratitude to Professor Nick Voulvoulis for the motivation, supervision, and guidance he provided that enabled me to produce this study. I consider it a privilege to have had Nick as my supervisor constantly trying to push me forward in my research and providing solutions for any issues that would arise.

I would also like to extend my appreciation to Dr Theodoros Giakoumis for his support through sharing ideas, discussing problems, and sharing feelings, all necessary for making steps towards developing this thesis. Also, I would like to thank Dr Yannis Kountouris and Chetna Vaghela of the Centre for Environmental Policy at Imperial College London for their advice and enlightening discussions at different stages of my research, as well as my examiners Prof Mark Burgman and Prof Phoebe Koundouri for the time they put in reviewing and examining my thesis.

On a more personal note, I would like to thank my family, Rena (an economist at heart), Pavlos, and Thalia, whose love and support have been fundamental in my life. I could not have completed this dissertation without the support and care of Nicholas, who, no matter what, makes everything possible for me to progress. I would also like to thank my friends, and people from the “Monday team” for their support.

Lastly, I would like to express how grateful I am to George & Victoria Karelia Foundation for financially supporting my studies.

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BT	Benefit Transfer methods
CAP	Common Agricultural Policy
CBA	Cost-benefit analysis
CEA	Cost-effectiveness analysis
CICES	Common International Classification of Ecosystem Services
CIS	Common Implementation Strategy
CPI	Consumer Price Index
DEFRA	Department for Environment, Food & Rural Affairs
DOC	Dissolved Organic Carbon
DPSIR	Drivers- Pressures-Status-Impacts-Response Framework
EU	European Union
GDP	Gross Domestic Product
HP	Hedonic Pricing method
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services

IWRM	Integrated Water Resources Management
LEEP	Land, Environment, Economics and Policy Institute
MEA	Millennium Ecosystem Assessment
MENE	Monitor of Engagement with the Natural Environment
NbS	Nature-based Solutions
OECD	Organisation for Economic Co-operation and Development
ORVal	Outdoor Recreation Valuation Tool
PoMs	Programme(s) of measures
PPP	Purchasing Power Parity
RB	River Basin
RBMP	River Basin Management Plans
RP	Reveal Preference methods
SAC	Special Areas of Conservation
SDGs	Sustainable Development Goals
SEEA EEA	System of Environmental Economic Accounts – Experimental Ecosystem Accounting

SES	Social-Ecological Systems
SP	Stated Preference methods
SSSI	Sites of Special Scientific Interest
SuDS	Sustainable drainage systems
TC	Total Carbon
TCM	Travel Cost Method
TEEB	The Economics of Ecosystems and Biodiversity
WFD	Water Framework Directive
WTP	Willingness to pay

1. Introduction

Water is a vital resource for sustaining life on earth and a key factor of economic development (Forslund et al, 2009). In Europe, approximately 65% of the water used for different purposes is abstracted from rivers (European Environment Agency, 2018c). Besides direct consumption benefits, aquatic ecosystems provide indirect benefits that constitute key wellbeing factors (Costanza et al., 1997; Butler and Oluoch-Kosura, 2006; Harrison et al., 2010). However, freshwater ecosystems are affected by a variety of pressures (e.g., water abstractions, point and non-point pollution, etc.) (Vörösmarty et al., 2010), which result both in the depletion of the quantity and the deterioration of the quality of water. In addition, land-use changes and climate variability further exacerbate these phenomena (Ormerod, 2009).

Traditionally, water management practices have been dealing with the water use of each sector of the economy (domestic, agricultural, industrial, etc.) independent of each other, resulting in fragmented policy interventions (Vieira, 2020), which have proven to be inadequate in fostering economic development without damaging the environment. In the 1980s, the emergence of the Integrated Water Resources Management paradigm demonstrated a process of how to consider multiple and often conflicting objectives in order to manage water resources effectively (Grigg, 2008), which over time has become the norm in water management (Molle, 2009). In Europe, such an approach to manage wicked inland water problems (Defries and Nagendra, 2017) was formalized in 2000, through the adoption of the European Union's Directive 2000/60/EC, commonly known as the Water Framework Directive (WFD). Key to the implementation of the WFD is the design and development of cost-effective Programmes of Measures to minimize pressures identified in river basins that impact the status of aquatic ecosystems. Furthermore, one of the innovative features of the WFD is the incorporation of economics for assisting in describing the status of the system and informing policy decisions (Bouleau, 2008).

However, more than two decades after it was adopted, following delays in its implementation and misunderstandings regarding the introduced definitions and approaches (Voulvoulis, Arpon and Giakoumis, 2017), the WFD has not yet delivered its main objective to achieve good status (European Commission, 2000) of all inland waters. From a policy implementation perspective, the Member States' low level of experience in integrated approaches, has been raising concerns (Baaner, 2011) about the types of selected interventions to tackle identified pressures and the role of economic tools and principles to promote options aligned with improving the status of European waters. Therefore, although the Directive has been praised for its systemic nature (Everard and Longhurst, 2018), Member States have not been able to harvest its potential, as indicated by the fact that 56 % of the total number of water bodies are currently far from being in a good state (European Environment Agency, 2021).

Designing and selecting interventions to manage complex environmental problems that do not lead to unanticipated side issues (Sterman, 2015) requires approaches that adopt a holistic view of the interdependences among system components and enable the holistic evaluation of policy interventions (Bellamy et al., 2001). Systems thinking (Forrester, 1994) can shed light on the multiple facets of socioecological issues (Neely, 2019) and enable the identification of leverage points and points of vulnerability (Holling, 2001), both as a family of tools (Polaine et al., 2022) and a competence (Kioupi and Voulvoulis, 2019; Ratinen and Linnanen, 2022) to understand complex systems. Transitioning from a reductionist worldview to a paradigm that encompasses complex interactions, calls for a more sophisticated conceptualization of social and ecological systems as nested systems (Jaaron and Backhouse, 2019; Small, Owen and Paavola, 2022), that incorporate the coupling of changes in the natural world with human behaviour (Castro, 2022).

In recent years, the concept of ecosystem services, the tangible and intangible benefits provided by ecosystems to humans (Burkhard and Maes, 2017), has received increased recognition for improving environmental management, by providing a straightforward link between ecosystem functioning and its effects on socioeconomic systems (Vlachopoulou et al., 2014; Boulton et al., 2016); and describing nature as a system (Costanza et al., 2017). Ecosystem services integrated into economic analysis in the context of natural resources management enables policymakers to understand the environmental, social, and economic trade-offs that occur within the system, and through that improve decision-making by applying interventions that influence the performance of the system (Mathews and Jones, 2008). Nevertheless, while the contribution of ecosystem services to environmental management is widely acknowledged (Sukhdev Pavan, Heidi Wittmer, 2014; Díaz et al., 2015), practical applications are still limited (Costanza, 2020). Additionally, despite the increasing use of systems thinking in environmental research (Turner et al., 2016; Zomorodian et al., 2018), a significant gap still remains in understanding how economic values are affected by interventions that influence ecosystem health (Hernández-Blanco et al., 2022).

In response to increasing environmental pressures, increasing complexity in human-made systems (Seiffert and Loch, 2005) and reductionist implementation of holistic frameworks (Gregory and Keeney, 2002), the current thesis aims at improving the use of economics in environmental management to encourage holistic interventions. By incorporating ecosystem services into economic methods and using natural capital accounting, the undertaken research presents theoretical and practical approaches to improve our understanding of the performance of socioecological systems and design interventions that are able to minimize pressures, while providing a wide range of benefits to society. Case studies in the UK and Greece are used to demonstrate the applicability of these approaches and discuss their limitations. In addition to that, the current thesis assesses the cost-effectiveness of nature-based solutions, a new type of interventions that aims to restore nature and increase human welfare by generating multiple benefits. Finally, the contribution of the developed integrated approaches to sustainable development as well as a new role for economics are assessed to promote a new vision of sustainable development.

2. Background

2.1. The Water Framework Directive: An overview

The Water Framework Directive (WFD), a leading European Union policy adopted in 2000, developed a framework for the protection of inland surface waters, transitional waters, coastal waters, and groundwater. Overall, the WFD aims at preventing further deterioration of aquatic ecosystems and enhancing their status; promoting sustainable water use, enhancing the aquatic environment through the adoption of specific programmes of measures that correspond to human pressures on each catchment; ensuring the progressive reduction of pollution of groundwater and preventing further pollution; and contributing to mitigating the effects of floods and droughts. The WFD introduced an integrated river basin management approach that pursues managing simultaneously land and water ecosystems thus establishing a systems thinking approach to water management (Voulvoulis, 2012). To achieve its objectives, the Directive defines key milestones and sets out deadlines for realising them. Furthermore, it institutionalized the River Basin Management Plans, the cornerstone of the implementation of the WFD that incorporate all relevant information on the components and status of the system and the proposed management options to influence its function, which must be updated in fixed intervals, following each management cycle (Figure 2.1).

In relation to its governance aspects, the Directive establishes river basin managing authorities responsible for the management activities in each river basin district that constitute a territorial unit the boundaries of which are defined by the hydrologic characteristics of each area (Di Quarto and Zinzani, 2021). Following a hierarchical approach (Borja et al., 2006), preparatory activities include among others the identification of River Basin Districts across Europe, the characterization of water bodies according to six surface water categories (i.e., rivers, lakes, transitional waters, coastal waters, artificial waters, and heavily modified water bodies), and the sub-division of surface water categories into specific types according to pressures and resulting impacts. Having the catchment or river basin as the reference system, the status of each water body as a subsystem and its distance from a state under undisturbed reference conditions (European Commission, 2003c) must be assessed.

The state, or the ecological status or potential of water ecosystems according to the WFD is defined by its structure and function (Arnold and Wade, 2015), where the term structure refers to specific physical, chemical and biological characteristics defined in Annex V of the Directive. Functioning has been interpreted by EU Member States as the good quality of structural elements, though scholars claim that it corresponds to complex interactions between abiotic and biotic elements of the ecosystem (de Jonge, Elliott and Brauer, 2006; Solimini, Ptacnik and Cardoso, 2009).

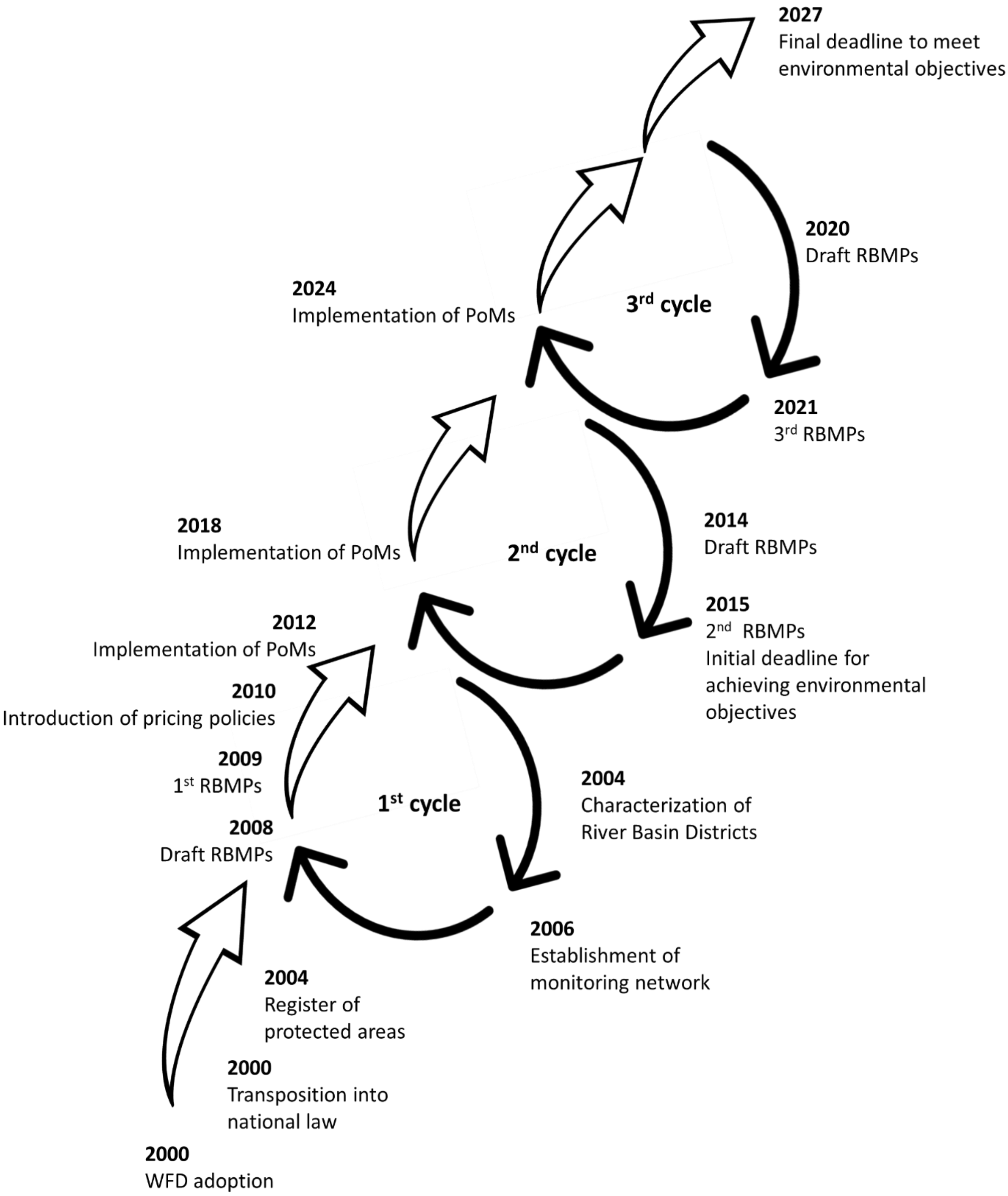

Figure 2.1 WFD milestones and management cycles

Following the principles of Integrated Water Resources Management (IWRM), managing authorities must take into account the environment and how it interacts with other components of the economy and society (Jager et al., 2016), evaluate how each sector of the economy relates with water resources through water uses and the generation of pressures over time and space, and assess the various economic, aesthetic, cultural, emotional, and environmental dimensions of water ecosystems (Hellegers and Davidson, 2021). Therefore, the WFD accommodates an analysis of the structural and functional characteristics of water bodies and through the identification and significance of non-water elements of the catchment, an assessment of the network of relationships among them. For instance, water is used extensively for agricultural activities and residential consumption (European Environment Agency, 2018a) in Europe, but it also plays the role of a sink for urban and industrial waste, which entails multiple pressures arising from various sectors with potentially competitive uses.

The interaction of aquatic ecosystems with the wider environment of the catchment is described through the notion of "pressures". Based on the "Drivers-Pressures-State-Impact-Response" framework (European Commission, 2003b) that has been adopted to harmonize the implementation of the Directive across heterogeneous Member States, pressures concern factors that cause perturbations on ecosystems and their components (Duel et al., 2005). Managing these negative interactions and their impacts requires the development of various technical, regulatory or economic instruments (Atkins et al., 2011; de Jonge, Pinto and Turner, 2012; Gregory et al., 2013). The involvement of non-state actors in the implementation of the WFD, one of its innovative features, aims at improving policy acceptance, facilitating the analysis of the state of the catchment and has the potential to improve decision making (Ker Rault and Jeffrey, 2008) and reduce uncertainty (Newig, Pahl-Wostl and Sigel, 2005), an inherent feature of water management.

2.1.1. The Water Framework Directive Programme of Measures

The goal of the WFD implementation is the sustainable use of water in each river basin by taking into account environmental, social, and economic considerations. Critical to achieving this is the design and implementation of holistic policy interventions. Article 11 of the Directive requires each Member state to establish a programme of measures for each river basin considering the information and analysis required under Article 5 (Characterization of the river basin districts) and the objectives established for each river basin.

Each programme of measures (PoM) consists of basic, and where necessary, supplementary measures. Basic measures have to be put into execution regardless of the status of water bodies to avoid degradation of ecologically undisturbed water bodies (Albrecht, 2013) due to water abstractions, input of pollutants from diffuse sources, accidental pollution incidents, and changes in the hydromorphological conditions among others; to secure the compliance with EU environmental legislation (e.g., the Habitat Directive, the Nitrates Directive, the Urban Wastewater Treatment Directive); and promote the efficient and sustainable use of water resources by recovering the total cost of water services. In addition to them, supplementary measures may be adopted at a local scale to provide further protection and improvement of the status of European waters where the basic measures prove insufficient. As described in Annex VI (B) they can include both technical and non-technical measures such as economic and fiscal instruments, emission and abstraction controls, advisory services, awareness and educational actions and subsidy programmes.

Besides their contribution towards reaching good water status, PoMs must be selected based on their economic impact. More specifically, Annex III (b) requires "judgements about the most cost-effective combination of measures in respect of water uses to be included in the programme of measures under Article 11 based on estimates of the potential costs of such measures". The Directive does not define a specific economic approach or criterion on how cost-effectiveness should be assessed, though it has been proposed that the cost assessment should be based on welfare economics valuation methods that consider the total cost of measures on the environment and the society (Jensen et al., 2013). Besides that, the results of PoMs should be assessed iteratively until the desired status for each water body is reached.

To simplify PoMs reporting requirements, Member States are asked to categorize the selected measures that correspond to Article 11.3 (b to l) and 11.4 into defined categories under the name Key Type of Measures (Table 2.1). These measures are expected to reduce significant pressures required to achieve good status or to prevent deterioration of the status of water bodies in high and good status (European Commission, 2021b).

Table 2.1 Overview of Key Type of Measures (KTM)

Code	Title
1	Construction or upgrades of wastewater treatment plants
2	Reduce nutrient pollution from agriculture
3	Reduce pesticides pollution from agriculture
4	Remediation of contaminated sites (historical pollution including sediments, groundwater, and soil)

Code	Title
5	Improving longitudinal continuity (e.g., establishing fish passes, demolishing old dams)
6	Improving hydromorphological conditions of water bodies other than longitudinal continuity
7	Improvements in flow regime and/or establishment of minimum ecological flow
8	Water efficiency measures for irrigation, industry, energy and households
9	Water pricing policy measures for the implementation of the recovery of costs of water services from households
10	Water pricing policy measures for the implementation of the recovery of costs of water services from industries
11	Water pricing policy measures for the implementation of the recovery of costs of water services from agriculture
12	Advisory services for agriculture
13	Drinking water protection measures (e.g., establishment of safeguard zones, buffer zones etc.)
14	Research, improvement of knowledge base reducing uncertainty
15	Measures for the phasing-out of emissions, discharges and losses of priority hazardous substances or for the reduction of emissions, discharges and losses of priority substances
16	Upgrades or improvements of industrial wastewater treatment plants (including farms)
17	Measures to reduce sediment from soil erosion and surface run-off
18	Measures to prevent or control the adverse impacts of invasive alien species and introduced diseases
19	Measures to prevent or control the adverse impacts of recreation including angling
20	Measures to prevent or control the adverse impacts of fishing and other exploitation/removal of animals and plants
21	Measures to prevent or control the input of pollution from urban areas, transport and built infrastructure
22	Measures to prevent or control the input of pollution from forestry
23	Natural water retention measures
24	Adaptation to climate change
25	Measures to counteract acidification
99	Other key type measures reported under PoM

2.1.2. The role of economics in the implementation of the WFD

In managing inland water resources, Member States must define their River Basin Districts and through a decentralised management approach, managing authorities must prepare the River Basin Management Plans (RBMP) and design and select the most appropriate Programmes of Measures. An important aspect of this process is the emphasis on the socioeconomic dimensions of river catchments (Unnerstall, 2007). Accurately measuring the state of the catchment as a system has two complex dimensions. The first concerns the environmental status (chemical and ecological), portrayed by an environmental indicator that measures the deviation between the current status and a state where anthropogenic pressures on water bodies are at their minimum (Voulvoulis, Arpon and Giakoumis, 2017). The second relates to the non-environmental characteristics of the catchment that are described by the economic analysis included in the RBMPs. Overall, the economic characterization of the river basin determined primarily under Articles 5 and 9 requires the assessment of economic analysis of water uses per sector and their accruing environmental, economic and resource costs (Koundouri et al., 2015).

Such aspects are crystallized in the WATECO (2002) document developed to guide Member States that had low or no experience in water management, which summarizes the three steps required for the successful implementation of the Directive (Figure 2.2): a) the economic characterization of the Riven Basin, b) the assessment of the current costs of water uses (i.e., financial, environmental and resources costs) and the extent of its recovery and c) development of potential programs of measures and assessment of their cost-effectiveness.

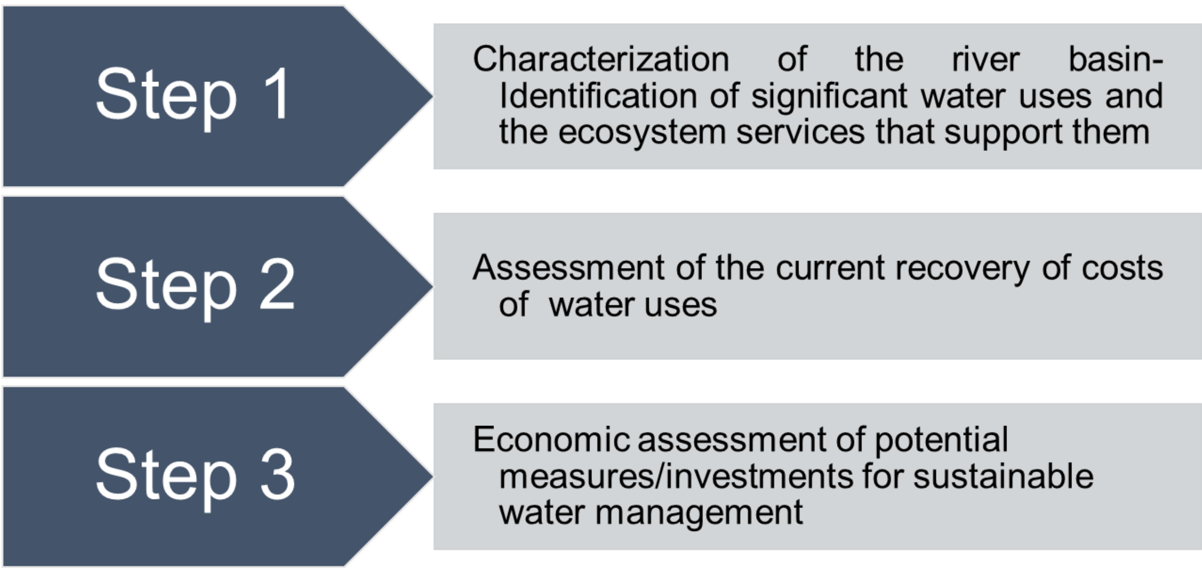

Figure 2.2 Steps of the economic analysis included in the WATECO document (Koundouri et al., 2015)

However, besides Articles 5, 9, 11 and Annex III (Balana, Vinten and Slee, 2011; Koundouri and Davila, 2013; Gutiérrez-Martín, Borrego-Marín and Berbel, 2017), reference to notions pertaining to economic disciplines can be traced elsewhere in the legal text (Table 2.2). The preface of the Directive mentions that economic and social conditions in the regions that constitute the Community should be considered along with the costs and benefits accruing from environmental policies aiming at correcting mismanagement or lack of actions that might exacerbate environmental issues (paragraph 12). In addition to that, the Directive calls for using economic instruments as programmes of measures when designing environmental policies (paragraph 38). Related to this are the principles of recovery of costs of water services and the polluter-pays principle. Additionally, the assessment of the PoMs must consider their costs as well as their effectiveness (paragraph 43) in achieving good ecological status of inland waters. Furthermore, justification for derogations from achieving set environmental objectives may be based on the disproportionality of costs of PoMs in relation to the benefits accruing from reducing pressures and improving water status (Martin-Ortega et al., 2014).

The Directive promotes the "sustainable" use of water resources (Article 1), which from a holistic perspective covers environmental, social, and economic aspects. To elucidate that, the Brundtland Commission (Brundtland, 1985) defines sustainable the "…development that meets the needs of the present without compromising the ability of future generations to meet their own needs". Whether this refers to weak or strong sustainability[1], determined in terms of the extent to which the society can substitute natural with manmade capital, it requires that the utility of current and future generations remains unchanged or at least that future generations do not enjoy a lower level of utility. Inter-generational equity is expressed in Article 7 of the Directive, which concerns water bodies currently providing water to humans, but also water bodies that are likely to be used for the same purpose in the future.

Finally, Annexes II, III, VI and VII, refer to the types of analysis and data that are necessary for adequately informing policy decisions. For instance, Annex II refers to the identification and assessment of anthropogenic pressures, among which are the water demand of industrial, agricultural, commercial, and residential users. While residential and agricultural demand (for example, Arbués et al., 2003; Dalhuisen et al., 2001; Gardner, 2010; Hooker and Alexander, 1998; Howe and Linaweaver, 1967; Makki et al., 2015; Martin and Wilder, 1992; Schoengold et al., 2006; Sebri, 2014; Zhou, 2016) have been more broadly studied than industrial demand (Rees, 1969; De Rooy, 1974; Renzetti, 1992, 2002; Reynaud, 2003), in-depth knowledge of the factors that influence water demand, thus may exacerbate pressure on water resource encourages understanding of complex management issues (Heinz et al., 2007). Involvement

[1] For a detailed explanation of the concepts, see Cabeza Gutés (1996)

of non-state actors in the implementation of the WFD, one of the innovative features of the Directive, aims at improving policy acceptance, facilitate the analysis of the analysis of the state of the catchment and thus has the potential to improve decision making (Ker Rault and Jeffrey, 2008) and reduce uncertainty (Newig, Pahl-Wostl and Sigel, 2005), an inherent feature of water management.

Table 2.2 References to economic tools and principles in the text of the WFD

Part of the Directive	Relevance to economics
Preamble 11	Introduction of the polluter should pay principle
Preamble 12	Economic and social conditions should be taken into account including benefits and costs accruing from actions or lack of actions
Preamble 36	Reference to the importance of undertaking economic analysis on water uses
Preamble 38	Reference to the use of economic instruments to achieve the objectives of the Directive focusing on the internalisation of all external costs in the pricing scheme
Preamble 43	Reference to identifying cost-effective measures to eliminate pollution
Article 1	Introduction of the sustainable use of water resources as one of the purposes of the Directive
Article 2	Reference to water services and uses by households, public institutions or any other economic agent
Article 4	Introduction of exemptions in cases where costs of achieving good ecological status are extremely high
Article 5	Economic analysis is formally introduced as a prerequisite for the implementation of the Directive
Article 9	Social, environmental, and economic aspects are introduced in the context of estimating the recovery of costs of water services
Article 11	Introduction of Programmes of Measures for addressing pressures that must be based on the analysis undertaken under Article 5
Article 13	Definition of the River Basin Management Plans and the types of analysis they must contain (including economic)
Article 16	Reference to the cost-effectiveness of measures regarding the progressive reduction of discharges, emissions, and losses of specific substances
Annex II	Description of the types of analyses Member States must undertake to identify and estimate significant water abstractions for urban, industrial, agricultural, and other uses
Annex III	Specifications concerning the types of economic analysis
Annex VI	Specifications concerning programmes of measures and economic tools
Annex VII	Specifications of the analysis that should be covered by the River Basin Management Plans, among which is the economic analysis of water uses and a summary of the programmes of measures

2.2. Economic tools and principles

Environmental goods and services encompass characteristics that do not leave a footprint on the market (Russell, 2001). Therefore, management decisions that influence their quality, quantity and in general their provision are driven by their significance to the economic and social systems. Given the complexity of environmental issues, management decisions may involve multiple objectives, or result in multiple effects that can be loosely described as socioeconomic gains and losses (Pearce, 2002). Economic valuation is a core element in informing environmental management decisions using tools, such as cost-benefit and cost-effectiveness analysis and national accounting. As expressed by Daily (1997) “We don’t protect what we don’t value”. Research in economics provides a wide range of techniques for the estimation of the value of non-marketed goods and services (see, for example, Hanley and Spash (1993)), related to natural resources and ecosystem services or the societal and economic effects of policy options. Using economic valuation in the context of environmental management decisions fosters understanding of the relationship between means (alternatives) and ends (objectives) (Keeney, 2002; Gregory et al., 2012); can facilitate trade-offs among any set of objectives (Hammond, Keeney and Raiffa, 1998); and when performed holistically can promote “value-focused thinking” (Keeney, 1997).

Central to economic valuation are “dose-response functions”, which link a change in the state of nature to some response in some other part of the system. Therefore, value is a measure of the satisfaction or dissatisfaction that individuals obtain from changes in their environment. The net sum of the monetized satisfaction and dissatisfaction constitutes the total economic value (Pearce, Atkinson and Mourato, 2006), which is decomposed into use and non-use values (Figure 2.3). Use value relates to the actual use of the good, or the possibility for actual use in the future, and is a quite straightforward concept. Actual use is divided into direct and indirect values (e.g., generation of incomes). Non-use values (also called passive-use values) can be classified into existence value, bequest value and option value. To elucidate more on that, if we consider the case of a river ecosystem, direct use values would concern benefits such as the provision of potable water; indirect use values would concern benefits, such as generation of income or increased productivity. In addition to these, individuals may be willing to bear some cost to preserve the ecological status of the river for future generations (bequest value), or they may be inclined to preserve the river because they anticipate future benefits from its use, such as medical benefits, recreational benefits, and others. Finally, individuals may appreciate the existence of the river although they do not actively use it, because they may appreciate the fact that others obtain benefits from it (for a more detailed description of the different types of economic value, see Pearce et al. 2006).

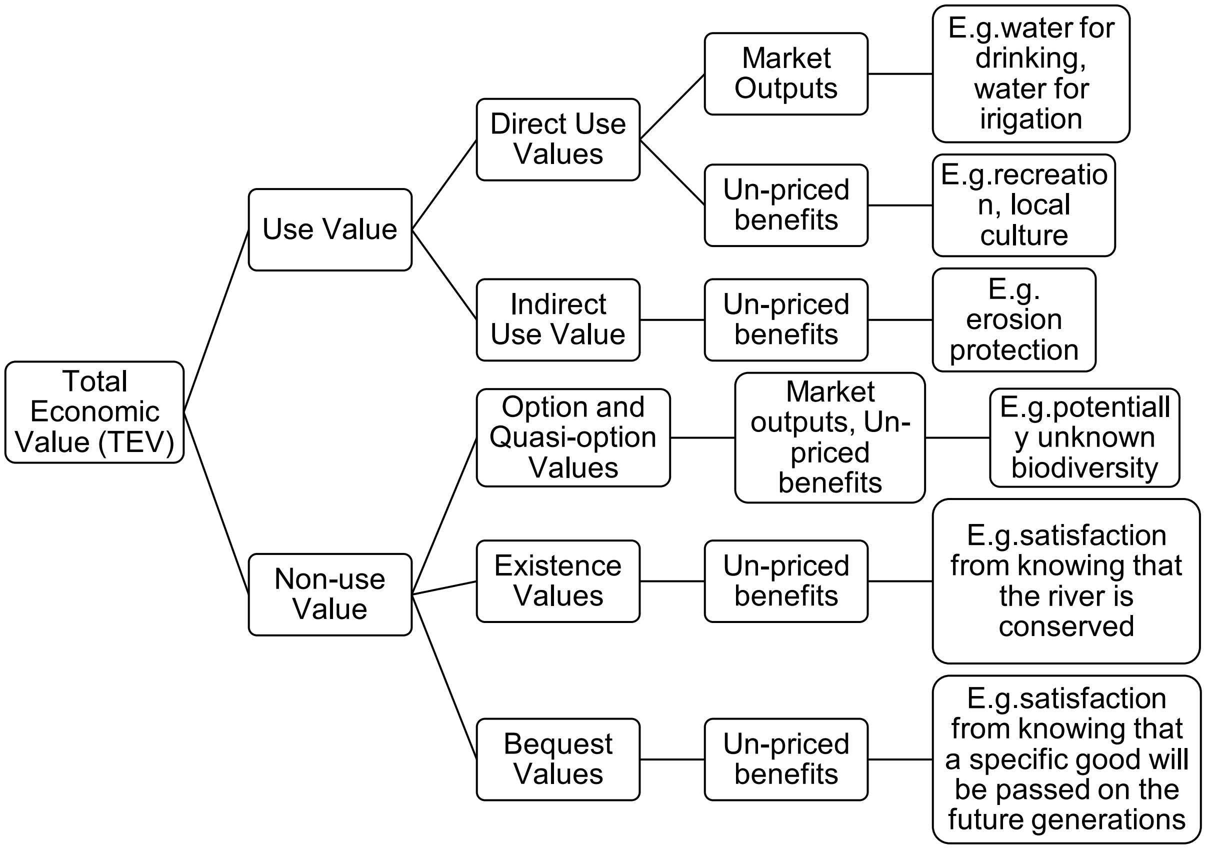

Figure 2.3 Total Economic Value (Adapted from: Pearce and Moran (1994))

Estimation of the aforementioned values in monetary terms is commonly performed through the use of revealed (RP) (also known as market techniques), stated preference methods (SP) and benefit transfer methods (BT). RP is a family of techniques that consider information dwelling from surrogate markets to estimate the value of a good through observing individuals' choices. According to Russell (2001), RP aim at quantifying an environmental good (or bad) by observing its "footprint" in surrogate markets. Such methods are the following: hedonic pricing (HP), travel cost method and averting behaviour/defensive expenditure.

The first method, HP, recognizes that the market price captures the value of a bundle of characteristics of a good. Therefore, HP attempts to identify, which part of the price of a good is attributed to environmental characteristics. For instance, two areas with identical socioeconomic characteristics that differ only in the levels of an environmental amenity (e.g., a clean river) will exhibit differences in property prices. Empirically, Colby and Wishart (2002) in their HP study estimated the economic value of the Tanque Verde Wash in Northeast Tuscany accruing from its scenic view, support of wildlife and benefits from acting as a noise and pollution buffer. They found that house prices dropped 0.45% for a 1% increase in the distance from the resource. Besides the HP, the travel cost method (TCM) is another approach that can be used to value the use of non-market benefits (mostly recreational) stemming from geographical areas, such as parks. To do that, TCM considers the number of visits to a specific location over a year and the expenses for such visits (fuel expenses, depreciation of vehicle, accommodation expenses, time spent travelling etc.). An example of the application of TCM is that of Loomis (2003) who used both on-site surveys and household surveys to estimate the benefit of visiting the Snake River in Jackson Hole, Wyoming and values varying from $9.67 to $23.92 per day. Furthermore, averting behaviour approaches are based on the notion that when individuals face risks or disutility due to a negative externality (non-market bad), are willing to pay for goods and services traded in actual markets to mitigate their utility loss. As explained by Garrod and Willis (1999), households would be willing to bear the cost of installing double-glazed windows to mitigate their exposure to noise pollution caused by traffic. The expenditure to avoid noise exposure indicates the value that households place on noise reduction.

Contrary to the aforementioned approaches, SP are survey-based methods that use constructed markets, where it is assumed that the good of interest can be traded (Mitchell and Carson, 1989). Appropriately designed questionnaires define the good (either as a whole or in terms of its characteristics), the institutional setting of the market and the pricing mechanism, through which the provision of the good will be secured (Bateman, Carson and Day 2002). Assuming that individuals are the best judges of their own preferences (Freeman III, 2010) and that they behave as though they were participating in a real market, economists can estimate how respondents´ willingness to pay (WTP) (i.e., the maximum cost they would be willing to bear for a good or a service) is affected under different hypothetical scenarios. Compared to RP which can only be used to estimate the use value of a good ex-post (real change), SP can be used to elicit both use and non-use values either ex-ante (hypothetical change) or ex-post (Garrod and Willis, 1999; Bateman, Carson and Day, 2002; Kjær, 2005). In general, two methods are categorized as SP, namely the contingent valuation method and the choice modelling method.

Finally, another type of tool, namely benefit transfer methods, described as the "…practice of adapting value estimates from past research to assess the value of a similar, but separate, change in a different resource" (Smith et al., 2012), have been used widely to monetize use and non-use values. The literature categorizes benefit transfers as either value transfers that include the use of a single value from a study performed in a similar site, or the average of values from a number of studies; or function transfers that use a valuation function to adjust the value that is transferred from a previous study based on the socioeconomic and ecological characteristics of the policy site (site of interest) (Boutwell and Westra, 2013). While value transfer methods are common in the economic valuation of the benefits of natural resources, the benefit transfer literature focuses on several types of transfer errors. More specifically, the estimates used in a value transfer exercise might suffer from validity errors that relate to the way the primary studies are conducted (Plummer, 2009). To mitigate this issue, Lawton et al. (2021) suggest collecting estimates of value at multiple sites to produce an average WTP. Furthermore, measurement errors relate to differences in socioeconomic characteristics (Bergstrom and Taylor, 2006); and to differences in resource substitutes between sites (Bateman et al., 2006). Brouwer (2000) proposes that studies considered in value transfers must be based on adequate data; sound empirical techniques; similar sites (e.g., in terms of their population and economic parameters); the change in the provision levels of environmental goods should be similar; the goods assessed in the studies from which estimates are transferred must not differ significantly, and the markets in the different sites must have a similar structure. These suggestions are supported by Bateman et al. (2011) that concluded that when the policy site and the sites from which values are collected are similar, simple mean value transfers minimise transfer errors.

The aforementioned techniques can be used to estimate the benefits or costs accruing from a change in the circumstances and could therefore be used to predict or validate the effectiveness of policy instruments (Stern, Common and Barbier, 1996; Garrod and Willis, 1999; Sterner and Coria, 2003). In addition to that, following the total economic value framework, the policymaker can examine the direct and indirect effects of a measure on the components of a system, either realized or expected (Bateman, Carson and Day, 2002; Vandermeulen et al., 2011). In recent years, ecosystem services have been used extensively in valuation exercises, which has increased significantly the relevant literature (Wilson, applications and 1999, 1999; Nelson et al., 2009; Keeler et al., 2012; Bateman et al., 2013; Koundouri et al., 2015, 2017; Haines-Young and Potschin, 2016).

2.3. The concept of ecosystem services and its relation to the WFD

Humans rely on nature for their survival and wellbeing. The importance of ecological processes and outputs on wellbeing has been researched under the concept of ecosystem services that describe the varied benefits ecosystems provide to humans (Fisher, Turner and Morling, 2009). The notion of ecosystem services can be traced back to c. 400 B.C. when Plato connected deforestation to soil erosion (Daily, 2013a). However, only around the end of the 1990s and the beginning of the 2000s, ecosystem services were brought into the limelight. Studies such as that of Costanza et al. (1997) and Daily (1997), as well as the publication of the Millenium Ecosystem Assessment (MEA) (2005), signified a boon to the literature on ecosystem services that followed both theoretical and empirical strands (Seppelt et al., 2011; Egoh et al., 2012; Potschin et al., 2016).

MEA points out that individuals as well as societies are embedded within ecosystems (Martínez-Harms and Balvanera, 2012) and together form a socio-ecological system whose components interact with each other. More specifically, its preface refers to an ecosystem as being “a dynamic complex of plant, animal, and microorganism communities and the non-living environment interacting as a functional unit” (Millenium Ecosystem Assessment, 2005). The interaction of elements of such complex systems generates goods and services, i.e., environmental outputs that influence human welfare. These goods and services have been grouped into four categories of services: *provisioning, regulating, supporting and cultural services*. This classification has been used widely; however, it has received notable criticism due to the double counting of some services (Fisher, Turner and Morling, 2009; Landers and Nahlik, 2013), which led to the development of other classifications, such as the Total Economic Value of Ecosystems and Biodiversity (TEEB 2010), the Common Classification of Ecosystem Services (CICES) (Haines-Young and Potschin, 2016; Haines-Young and Potschin-Young, 2018), and the Final Ecosystem Goods and Services classification system proposed by the US Environmental Protection Agency (Landers and Nahlik, 2013; Landers, Nahlik and Rhodes, 2016). Ecosystem services assessments undertaken at local, regional, or national levels rely on one of these classifications, however, given the lack of unified classification comparison across studies is difficult (Busch et al., 2012). Nevertheless, the traction that this concept gained over the years produced considerable evidence of the degradation of natural ecosystems (Quintas-Soriano et al., 2016) and triggered the kick-off of a number of governmental and non-governmental initiatives, such as the Intergovernmental

Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES), the UK National Ecosystem Assessment (UK National Ecosystem Assessment, 2014) and others.

Table 2.3 Types of ecosystem services based on the Economics of Ecosystems and Biodiversity classification (Sukhdev, 2012)

Types of services	Examples of Benefits
Provisioning	Food (e.g., fish), water (e.g., water for drinking, cooking), raw material (e.g., timber), medical resources (e.g., Biochemical products), ornamental resources
Regulating	Air quality regulation, climate regulation, erosion prevention, waste treatment, pollination, biological control
Habitat	Maintenance of life cycles of migratory species, sustenance of genetic diversity
Cultural	Cognitive development, spiritual experience, opportunities for recreation (e.g., swimming, fishing)

(Adapted from: de Groot, Wilson and Boumans (2002))

The notion of ecosystem services as flows generated by natural capital (i.e., the stock of natural resources that generate ecosystem goods and services) (Häyhä and Franzese, 2014) has created the bases for the integration of ecosystem services into national accounting systems (Capriolo et al., 2020). Natural capital accounting aims at explaining the dependence of human societies and the economy on nature, positing that healthy ecosystems are able to provide services in support of human well-being (De Groot, 1987). In Europe, natural capital accounts have been developed at a national scale in the UK and the Netherlands following the System of Environmental Economic Accounting - Ecosystem Accounting (SEEA EEA) integrated framework (UN, 2017). Eurostat and the European Commission have financed projects in EU Member States (e.g., Bulgaria, Denmark, Estonia etc.) and research projects such as the Mapping and Assessment for Integrated Ecosystem Accounting (Hein, et al., 2020) and the We Value Nature projects (Bagstad et al., 2021). At the catchment scale research has been conducted by Argüello et al.(2022), as well as in the context of the INCASE project in Ireland (Farrell et al., 2021) that have produced results on the condition, extent and environmental flows of ecosystem services. Nevertheless, the incorporation of economic considerations into natural capital accounts is still in an experimental phase (Farrell et al., 2022). Additionally, to my knowledge, results on natural capital accounts that integrate both economic and environmental values at river basin scale are limited, due to lack of empirical studies.

2.4. Socioecological systems, systems thinking and complex adaptive systems

Historically, policy decisions and as a result management of natural resources and the environment were based on limiting the focus of observation and on the reduction of complex problems into rationally manageable components (Chapman, 2002). In contrast with it, systems thinking -a way to thinking that views problems as phenomena relating to dynamic systems- highlights that the drivers of an event and the realised impacts (e.g., water pollution) may not be straightforwardly related. Systems thinking emerged in the twentieth century opposing reductionist approaches that had been followed until then (Flood, 2010). According to Scrieciu et al.(2021), the literature on systems thinking attributes a cognitive and a communication dimension to the relevant research. The first dimension relates to an iterative learning process or a mental framework about how the world functions, whereas the second relates to explaining interdependencies, feedbacks, and systems.

Comprehending a given state where different phenomena (expected or unexpected) take place, according to systems thinking, requires to observe the circumstances as emerging from the structure of various parts of the system that interrelate dynamically with each other (Meadows, 2008). The term "systems-thinking" was coined by Barry Richmond in 1987, who described it as the art and science of making reliable inferences about behaviour by developing an increasingly deep understanding of underlying structure. System thinking is a system of thinking about systems (Arnold and Wade, 2015), which consists of three factors: purpose, parts, and interconnections. Systems thinking methods of understanding and managing complex problems include among others Causal Loop Diagrams (CLDs), system archetypes, stock and flow diagrams, tree diagrams, fuzzy cognitive maps, and system dynamics models (Nyam et al., 2020). The literature on systems thinking and system dynamics has increased exponentially during the last 40 years (Nyam et al., 2020), perhaps due to the fit of this framework to deal with modern complex problems (Hossain et al., 2020) that relate to the increasing interdependences and the speed at which changes occur (Seiffert and Loch, 2005)

Systems theory recognizes that a system consists of a large number of elements, their non-linear interactions and feedback loops (positive and negative). Systems operate under conditions far from equilibrium and are constantly changing to respond to the constant flow of energy and information (Holden, 2005). Systems are constantly changing. Problems that arise within them are complex, dominated by uncertainty, affect various components and scales;

and demand flexibility in management decisions, embracing the knowledge of different actors and disciplines and the diversity of values of stakeholders (Reed, 2008). More specifically, research claims that participatory approaches in environmental management have the capacity to reduce conflict, and thus result in increased public support for the implementation of management interventions in the long-run (Devente et al., 2016; Swart et al., 2018). Several methods exist for facilitating stakeholder involvement in water resources planning (Luyet et al., 2012), such as the Multi-Criteria Decision Analysis (MCDA) that can be used in policy planning (e.g., Karjalainen et al., 2013); the Delphi method (e.g., Musa et al., 2015); focus groups (e.g., Wibeck, 2011); cognitive mapping (e.g., Dodouras and James, 2007); scenario-based stakeholder engagement (e.g., Tompkins, Few and Brown, 2008); conceptual system modelling (Magnuszewski, Sendzimir and Kronenberg, 2005); living labs (Leminen, Westerlund and Nyström, 2012; Schuurman, De Marez and Ballon, 2015; Potters et al., 2022); various types of workshops (e.g., Reed et al., 2013), among others.

Sustaining participation of stakeholders in management processes over time enables the evaluation of management practices (Gaillard, 2014) and promotes feedback learning, adaptive governance and resilience (Folke et al., 2005, 2016). Resilience refers to the ability of the system to continue to function despite the occurrence of endogenous or exogenous disturbances (Levin et al., 2013) and constitutes a property that the literature on complex adaptive systems has stressed to a great extent (Bhamra, Dani and Burnard, 2011; Martin-Breen and Anderies, 2011).

The adaptive cycle model (Holling, 2001) developed from the perspective of ecology has been used as an operative approach to foster understanding of the complexity and processes of systems. According to it, systems go through four phases of change, namely reorganization (α), exploitation (r), conservation (k) and release (Ω) as depicted in Figure 2.4. During the r phase, the system phases rapid growth, and ecological, economic, social, and cultural capitals are accumulated. Connections among the elements of the system are not yet established. Intense competition among them determines the configuration of the system. During the k phase, the connections among elements are determined, accumulation of capital slows down, which turns the focus towards efficiency. Given the tight connections in this phase, resilience declines, which makes the system less resistant to disturbances. In the event of a shock, the system enters the Ω phase. The system releases the stock that has been preserved and loses its structure. Upon the appearance of an opportunity for reorganization, the system enters the α phase. Lack of structure gives rise to the emergence of new configurations and high adaptability of the system.

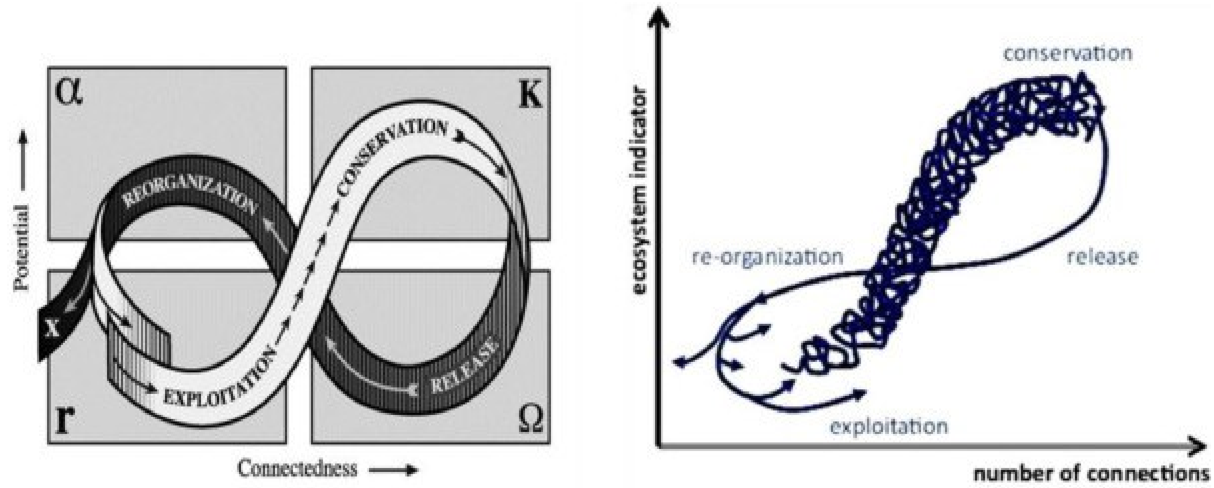

Figure 2.4 A stylized representation of the adaptive cycle metaphor and its four phases. The "x" label represents the leakage of potential that can generate a less productive system. (Source: Gunderson and Holling (2002) and the adaption by Burkhard, Fath and Müller (2011))

Besides the dynamics of resilience, another issue that increases the complexity of managing systems is the interaction of systems at different scales (Newell, 2003; Holdschlag and Ratter, 2013; Fath, Dean and Katzmair, 2015). To accommodate this issue, the adaptive cycle model has been extended through the concept of panarchy which postulates the nested nature of the adaptive cycles. A key element of this model is that complex systems are regarded as being nested across scales, which interact with each other through the exchange of energy and information. Connections across scales are influenced by the remember force when the *α* phase of a cycle is influenced by a higher-level *K* phase and the revolt force when the *Ω* phase of a lower level causes a crisis at a lower level (Gotts, 2007) (Figure 2.5).

The panarchy theory thus accounts for phenomena that happen at some level and resonate across higher levels (Sundstrom and Allen, 2019) (e.g., production of CO_2 emissions in a city that contributes to global warming, as well as the opposite i.e., global warming that raises the sea level in a coastal area). As a result, pressures on natural ecosystems may stem from other elements at the same level (e.g., pollution caused by agricultural activities within a river catchment) or systems at other levels (e.g., a higher concentration of residents in the region where a river catchment is located that causes changes in land uses). Additionally, the connectedness among levels introduced by this theory suggests that management interventions implemented at any level may unavoidably impact all other levels and nodes. However, a possible issue that may arise is the mismatching of scales between management practices and ecological systems (Cumming, Cumming and Redman, 2006). For example, Pelosi et al. (2010) mention that the mismatch between the implementation levels of measures, such as the Common Agricultural Policy and the levels of the ecological processes is one of the reasons why interventions at the farm scale are unsuccessful. Therefore, recognizing the source (reason and level) of symptoms noticed in a system is essential for adopting management paths that are fit for success.

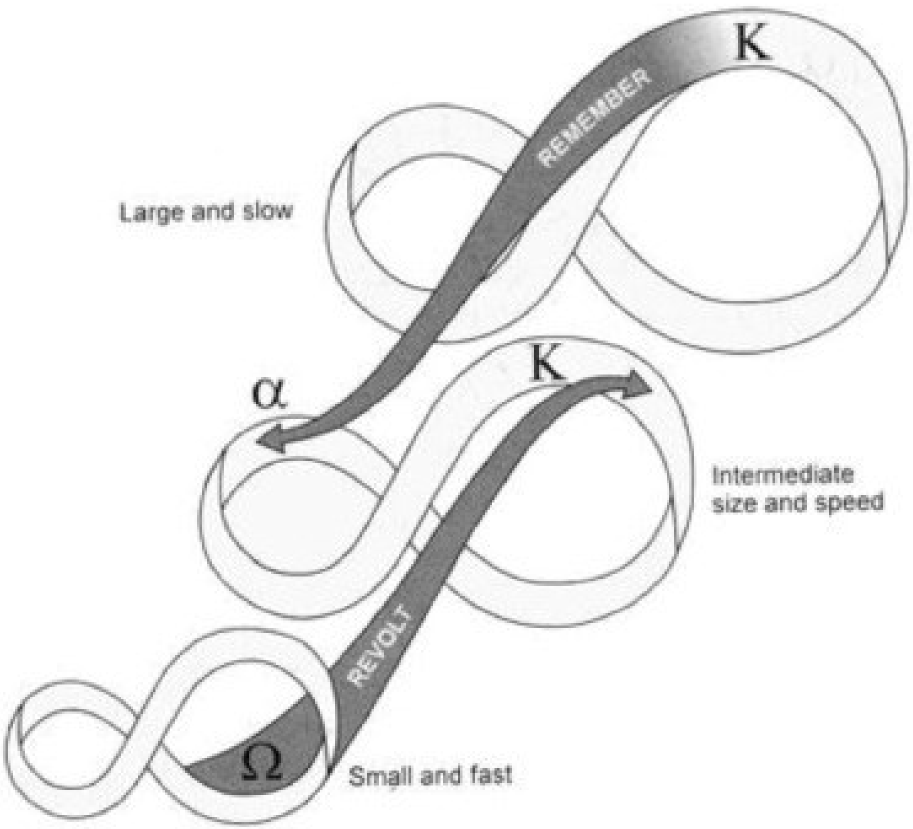

Figure 2.5 Nested adaptive cycles with cross-scale feedbacks (Gunderson and Holling, 2002)

Environmental problems take place at the intersection of natural and social systems (Berkes, Colding and Folke, 2008; Chaffin, Craig and Gosnell, 2019) and have often been studied through a Social-Ecological Systems (SES) perspective (Figure 2.6). Given the open system nature of SES, the processes and interactions of elements within them are influenced by political, economic, biochemical, and institutional conditions (Chapin, Kofinas and Folke, 2009). Being of systemic nature, SES methodology entails viewing how specific problems are connected to interconnected fields and scales and introduces an interdisciplinary approach to explaining complex phenomena, according to which humans and nature are interdependent, form complex relationships and shape each other (Gain et al., 2019). Commonly, approaches such as Integrated Water Resources Management (IWRM), Water-Energy-food Nexus, Nature-based Solutions (NbS) and socio-hydrology have been proposed for addressing current water management challenges (Gain et al., 2021).

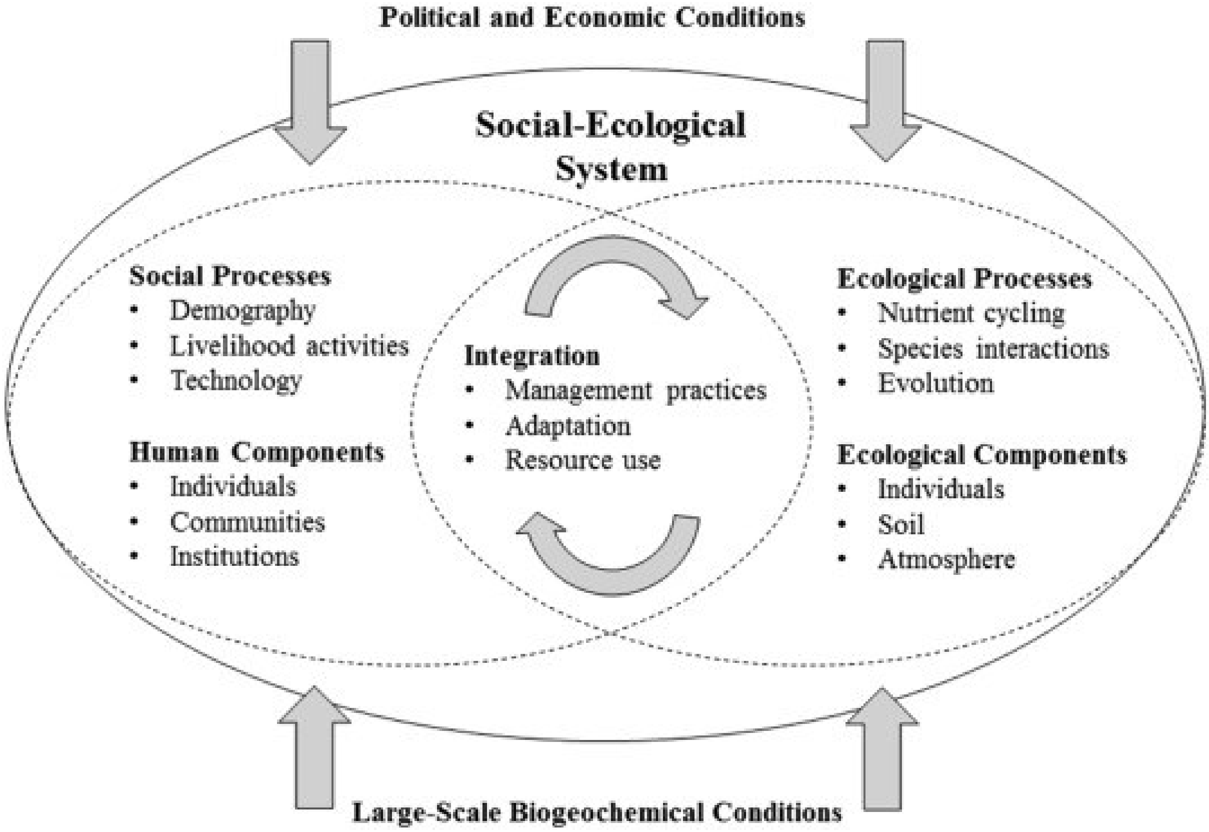

Figure 2.6 A conceptual representation of an SES (Virapongse et al., 2016)

IWRM has gained increasing popularity over the last two decades and currently constitutes one of the dominant water management concepts worldwide (Leidel et al., 2012; Heldt et al., 2017). The WFD has been the major driving force for the dissemination of the IWRM in Europe, as it requires not only the improvement of the status of surface waters and groundwaters in the EU Member States, but also the participation of relevant stakeholders in the several stages of its implementation (Mostert, 2009; Richter et al., 2013), and the integration of economic principles and tools to support management decisions. In relation to the latter, benefits and costs accruing from changes in the ecological conditions of rivers and natural resources in general describe interactions both within and across systems (Zhang et al., 2018). Economic tools and approaches can shed light on these interactions encouraging policymakers to consider the wider system when designing policy instruments. However, despite efforts to integrate economics into environmental management to improve understanding of the structure and processes that take place within them, the centre of gravity in analysing the human-nature interactions still remains in the natural sciences (Polasky et al., 2019), which in turn does not create the basis for developing pathways towards sustainable development.

2.5. Sustainability and its relation to systems thinking

Sustainability is a complex and vague concept. Its definition in literature and policy has been characterized as being ill-defined, not defined or contradictorily defined (Phillis and Andriantiatsaholiniaina, 2001). Across disciplines and scientific fields, different definitions of sustainability abound, amounting to more than 300 definitions (Dobson, 1996). Sustainability in social terms describes the satisfaction of basic human needs of the current and future generations (Littig and Grießler, 2005). Ecologists, on the other hand, define sustainability in terms of the integrity and proper functioning of the ecosystem, or the absence of disturbances (Aarts, 1999). Lastly, economists see sustainability in terms of allocating different types of capital (human, social, built, natural) to satisfy the needs of humans across generations.

In previous years, through a series of high-level events and conferences, the issue of sustainability and sustainable development attracted the interest of scientists and policymakers all over the world. This has fostered the development of a commonly accepted language, which has been used in public discourse. The 1972 United Nations Conference on the Human Environment in Stockholm expresses the paradigm shift that occurred in the 70s. Among the principles agreed upon at the conference, principle 3 stated that “The capacity of the earth to produce vital renewable resources must be maintained and, wherever practicable, restored or improved”, which indicates the amount of political capital that environmental groups gathered in the 1960s (Munn, 1992). Another major event was the publication of the World Conservation Strategy (IUCN, 1980) commissioned by the United Nations Environment Programme in 1980 which represented a consensus of the world’s biggest international organisations on sustainable development. After that, the Brundtland Report (WCED, 1987) introduced the sustainable development definition and a series of prestigious international events (Figure 2.7), positioned sustainable development at the centre of environmental policy and declared it as the purpose of the socio-environmental system. Most recently, the United Nations 2030 Agenda and the Sustainable Development Goals (SDGs) in 2015, made a significant step towards a more holistic approach related to how to put human wellbeing and ecosystem resilience at the core of global policy for achieving development (Costanza et al., 2016). The seventeen SDGs replaced the previously adopted Millennium Development Goals to introduce solutions that integrate social, economic, and environmental outcomes (Wood and Declerck, 2015; Folke et al., 2016; United Nations, 2018). Achieving the SDGs frames a problem where the goal is to maximize socio-economic development with respect to the biophysical constraints of nature. In other words, harvesting nature’s flow of goods and services needs to consider the critical natural capital stock that must be preserved in order to maintain the provision of ecosystem services (Rudolf De Groot et al., 2003; O’Neill et al., 2018). As discussions on sustainability and sustainable development are becoming increasingly common and fervid, naturally questions arise about what is to be sustained and how.

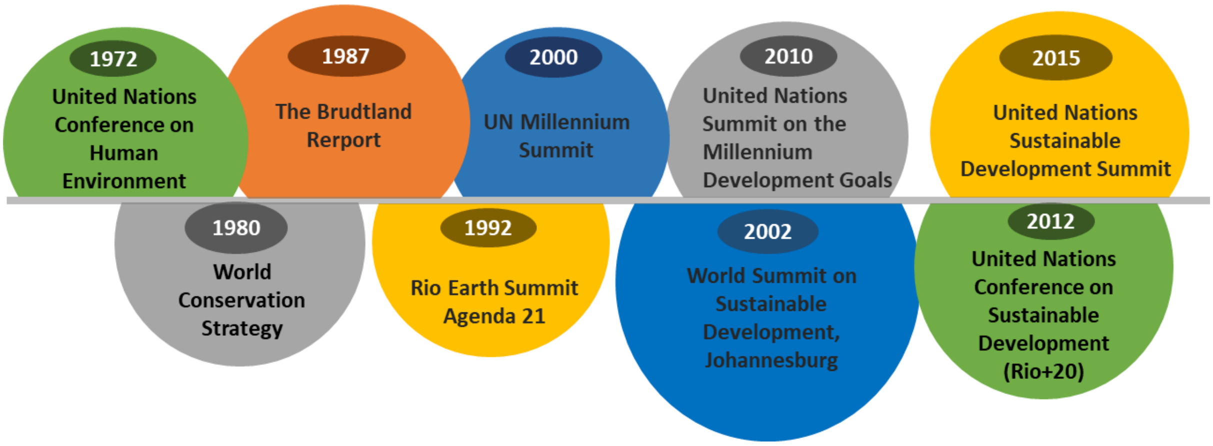

Figure 2.7 Major international events on sustainable development

In economic research, weak and strong sustainability interpretations (Turner, 1993; Wilson, Tyedmers and Pelot, 2007) emerged to address this issue. The distinction between the two relates to neoclassical and ecological economics thinking respectively (Ramos and Caeiro, 2010). Central to both is the extent to which natural capital can be substituted by manufactured (tools, machines and infrastructure that contributes to production), human (individuals' capacity to work), and social capital (networks and organisations that coordinate the actions of individuals) (Ekins, 1992). Natural capital contains all attributes of nature (e.g., the atmosphere, natural resources, species etc.) that are either vital to life (critical natural capital), important but not vital (constant natural capital), or not highly valued and replaceable (tradable natural capital) (Davies, 2013). At one end of the spectrum, weak sustainability (Solow, 1974, 1986, 1993; Hartwick, 1977, 1978, 1990) suggests that manmade capital can substitute natural capital and maintain welfare for future generations, through non-declining aggregate capital stock. Weak sustainability is based on strong commensurability (Spangenberg, 2013), therefore, in principle, it means that all types of capital can be expressed in the same monetary unit (De Groot et al., 2003). At the other end, strong sustainability proposes that natural capital is not substitutable, as its critical elements cannot be restored (Pearce, Markandya and Barbier, 1989) and it constitutes a component of manufactured capital (Daly, 1991). This implies an ordinal scale of measurement, which allows for assessing the contribution of natural capital to other types of capital and identification of cases where relaxing the non-substitutability assumption is permitted through satisfying the condition of non-declining aggregate capital stock. Advocates for strong sustainability argue that other types of capital cannot substitute the services provided by nature, therefore natural capital is not replaceable (Ahi and Searcy, 2014). Pearce and Turner (1990) and Ekins et al.(2003) describe that natural capital performs four distinct types of functions: a) it provides raw materials that either enter into the production of goods or are consumed directly (e.g., food and metals); b) it absorbs the waste products of production and consumption; c) it provides amenity services that contribute to human welfare; and d) it provides supporting services on which life-supporting functions depend. Furthermore, the realisation of the weak sustainability paradigm requires that either renewable and non-renewable resources are in abundance, or that the elasticity of substitution between natural and manufactured capital is at least equal to unity, or that technological progress can increase the productivity of the natural capital stock faster than it is being depleted (Dietz and Neumayer, 2007).

Weak sustainability has been criticized to be a quick fix that tackles symptoms (e.g., slow economic development and water pollution) rather than the real causes of the problems (Bielyet al., 2018). As a normative goal, sustainability has been framed as the maximization of objectives across economic, social, and environmental systems (Elliott, 2012; Barbier and Markandya, 2013; Costanza et al., 2016 among others). Barbier (1987) used a Venn diagram to describe sustainable development as the intersection of the objectives of these three systems. According to this view, maximizing the goals of one system does need to lead to sustainable results as the costs imposed on other systems are neglected. Though conceptual, this representation of sustainability strengthens the viewpoint of society, economy and environment composing a system with a specific purpose and elements that interact and produce certain outcomes. However, it does not explicate the emerging properties of the elements and the sub-systems and thus does not elevate to the level of an operational framework.

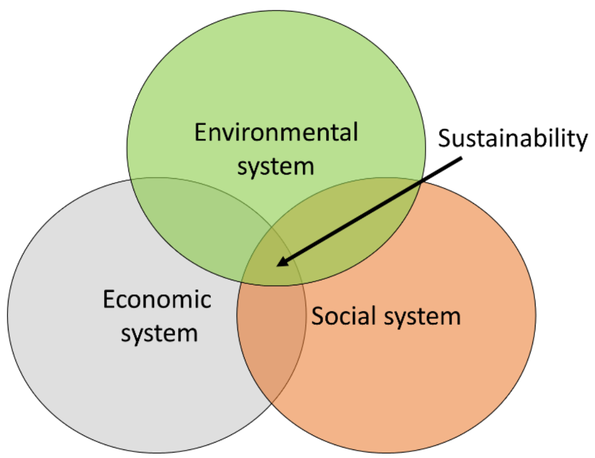

Figure 2.8 A systems view on sustainability. (Adapted from: Barbier and Burgess (2017))

Nevertheless, recently, sustainability has been defined by threshold values of social and ecological variables. For example, the "safe and just space" framework (Raworth, 2012) argues that the exploitation of the environment should be done to a degree that it does not cross the planetary boundaries but satisfies humans' basic needs. Based on this idea, Raworth (2012) developed the concept of "Doughnut" to describe the interaction between planetary boundaries and social foundation, however in just one study that assessed such links (Capmourteres et al., 2019) dependencies were proven to be diverse in terms of directionality, and the magnitude of effect and type of dependencies. Such a framework may constitute a diagnostic tool for the status of the system in terms of sustainability, however, the higher the level of analysis the most likely might be that the framework disregards heterogeneity across local socio-environmental systems. In line with this, it has been questioned whether planetary threshold values do reflect local conditions (Heistermann, 2017).

In the context of complex systems, the literature recognizes a range of tools, models, and approaches to describe the relationship of society with the natural world and the transition towards sustainability. For instance, Ecological Economics views the environment as a supplier of goods and services to the economy, a sink for waste products and as the larger system in which the economy is embedded (Kerschner and O'neill, 2015). On the contrary, Environmental Economics perceive ecosystems as an element of the economic system (Daly, 2001)[2]. Additionally, theories that conceptualize complex systems (e.g., Complex Adaptive Systems (Preiser et al., 2018)); assess the interactions in socio-ecological systems (e.g., the Social–Ecological Systems Framework (Partelow, 2018)); investigate the functioning of technical regimes and networks (e.g., Theories of Practice (Corsini et al., 2019) and Multilevel Perspective on transitions (Köhler et al., 2019)); view the economic system as an element embedded in the social system (e.g., the Social Provisioning Perspective (Jo, 2011), the Systems of Provision approach (Fine, Bayliss and Robertson, 2018)) have been considered to describe sustainability transitions in the context of provisioning systems[3] (Fanning, O'Neill and Büchs, 2020). Furthermore, sustainability is often studied in parallel with resilience, a concept that describes the response of the system to external shocks and persistent stress (Folke, 2016). As observed by Marchese et al. (2018) the approaches identified in the literature describe the two concepts as either being embedded in one another or being two separate objectives.

[2] Douai, Mearman and Negru (2012) provide a presentation of the dominant neoclassical and heterodox schools of economic thinking on sustainability.
[3] O'Neill et al. (2018) describe provisioning systems as interlinked complex physical and social systems that work together to transform resources to goods and services to satisfy human needs.

The engineering community has proposed Industrial Ecology (Korhonen and Snäkin, 2005) to elucidate how industrial processes interact with nature and to determine how industrial and manufacturing processes could be transformed so that they become compatible with nature's functioning (Commoner, 1997; Erkman, 1997; Ayres and Ayres, 2015). The fact that this approach does not provide a concise way of how uncertainty inherent in ecological processes could be portrayed in industrial processes (Jensen, Basson and Leach, 2011; Lifset and Graedel, 2015) created the space for other ideas, such as the Circular Economy paradigm, which has been receiving increasing attention due to its potential benefits for sustainable development (Homrich et al., 2018). In such an economy, production and consumption form a circle of material flow, contrary to the dominant linear model, where goods are produced, sold, used, and then discarded as wastes. The life cycle of products is extended through reducing, alternative reusing, recycling, and recovering materials aiming at relaxing the dependence of the economy on nature (Kirchherr, Reike and Hekkert, 2017). Underexplored issues attached to the Circular Economy model (Korhonen, Honkasalo and Seppälä, 2018), manifest the loose connection between that particular model and sustainable development (Millar, McLaughlin and Börger, 2019). Nevertheless, the idea of optimizing the system rather than its components to decrease the environmental impact of additional units of economic output (Ghisellini, Cialani and Ulgiati, 2016) has been promoted as a means of decoupling.

3. Aim, objectives and significance of research outputs

3.1. Aim

The aim of the thesis is to assess and improve the role of economic tools and principles in environmental management decisions in the context of the Water Framework Directive in order to promote green interventions through systems-based economic approaches that foster sustainability transformations of the socioecological system.

3.2. Objectives

The objectives of this thesis are the following:

- Support the improvement of ecological conditions by utilizing integrated socioecological approaches for the selection of policy interventions to address pressures.
- Explore opportunities for integrating natural capital accounting in environmental management.
- Investigate the potential of integrating nature-based solutions in natural resources management to improve the effectiveness of policy interventions.
- Investigate the role of economics in sustainability transitions and policy improvements.
- Provide recommendations for improving environmental management decisions through system-based approaches by utilizing the learnings of implementing the Water Framework Directive.

3.3. Significance of the thesis

Starting from the environmental and socioeconomic consequences of management practices developed due to the adoption of the Water Framework Directive in Europe, the goal of the undertaken research is to influence how environmental policies are developed and implemented by assessing the drawbacks of applied practices and identifying robust alternatives. The thesis is influenced by the GLOBAQUA project (funded by the 7th EU Framework Programme under the title 'Managing the effects of multiple stressors on aquatic ecosystems under water scarcity') that aimed to assess the interaction among pressures on freshwater resources under water scarcity conditions in order to improve knowledge on their relationship and improve water management practices. Using two of the GLOBAQUA case studies, the PhD thesis made original contributions to the scientific literature through the publication of two academic articles in peer-reviewed journals:

- *Souliotis, I. and Voulvoulis, N., 2021. Incorporating Ecosystem Services in the Assessment of Water Framework Directive Programmes of Measures. Environmental Management, 68(1), pp.38-52. Available at: https://link.springer.com/article/10.1007/s00267-021-01478-7*
- *Souliotis, I. and Voulvoulis, N., 2021. Natural Capital Accounting Informing Water Management Policies in Europe. Sustainability, 13(20), p.11205.* Available at: *https://www.mdpi.com/2071-1050/13/20/11205*

However, the scope of the study extended beyond that of the GLOBAQUA project and investigated how alternative integrated measures can be utilized to improve the overall performance of the socio-ecological system. This resulted in a publication that provides evidence of how a "green" intervention besides yielding ecological improvements of water resources was able to enhance socioeconomic conditions.

- *Souliotis, I. and Voulvoulis, N., 2022. Operationalising nature-based solutions for the design of water management interventions. Nature-Based Solutions, p.100015. Available at: https://www.sciencedirect.com/science/article/pii/S2772411522000076*

Moreover, a significant part of the thesis concerns the relationship between the use of environmental resources to sustain economic development and the human-made effects on the environment. Based on the findings of the undertaken research and the sustainability literature, the study has resulted in the development of an article that provides a new vision of sustainable development accompanied by systemic changes:

- *Souliotis, I. and Voulvoulis, N., 2022. A systems thinking perspective on the role of economics in transitioning towards sustainability. World Development Sustainability. Submitted in July 2022*

Furthermore, in the same context the study has contributed to the development of an academic article that proposes intentionally altering the components and structures of the system that are the root causes of complex socioecological challenges:

- *Voulvoulis, N., Giakoumis, T., Hunt, C., Kioupi, V., Petrou, N., Souliotis, I. and Vaghela, C., 2022. Systems thinking as a paradigm shift for sustainability transformation. Global Environmental Change, 75, p.102544. Available at: https://www.sciencedirect.com/science/article/pii/S0959378022000826*

Finally, the pitfalls in the implementation of the Water Framework Directive and the methodological approaches and findings developed and discussed in the study constitute the main body of a journal article currently being prepared, which, in line with systems thinking, proposes an integrated definition of a desired state to be considered by policymakers when designing policy objectives in environmental resources management:

- *Souliotis, I. and Voulvoulis, N., 2022. Learning how to improve environmental management decisions from implementing the Water Framework Directive. In preparation*

4. Methodological aspects of the study

4.1. Thesis structure

An overview of the research topic and the rationale for the conducted research is introduced in Chapter 1: General Introduction. Chapter 2 consists of five sections that explain concepts and provide detailed information on topics touched upon by the subsequent chapters. More specifically, section 2.1 and its subsections provide a general outline of the Water Framework Directive and explain how economics relates to its implementation; section 2.2 presents common tools and approaches used by economists to assess socioeconomic values; section 2.3 presents the concept of ecosystem services extensively used in this thesis and briefly discusses its connection to the implementation of the Water Framework Directive; section 2.4 presents the theoretical foundations of Complex Adaptive Systems, systems thinking and socioecological systems; and lastly section 2.5 discusses the concept of sustainability that is used extensively throughout this thesis and explains its relation to systems thinking. The subsequent chapters, all self-contained, are organised in a similar fashion which includes an introduction, development of the research topic, materials and methods, results, discussion, and main conclusions.

Chapter 5 discusses the issues of monitoring and evaluating the effectiveness of implemented programmes of measures and the challenges faced by EU Member States concerning the use of economic techniques in such assessments. The chapter recognizes that when PoMs are designed properly, they are able to reduce anthropogenic catchment pressures and improve the ecological conditions of the aquatic ecosystem, which in turn affects the provision of ecosystem services. Based on that, a methodology is developed for the evaluation of the cost-effectiveness of measures in terms of ecosystem services benefits identified through relevant stakeholders. The application of the methodology is demonstrated through a case study and its potential to facilitate the economic analysis required by the WFD is discussed. The findings of the chapter, demonstrate how the proposed methodology effectively incorporates ecosystem services in the assessment of the economic impact of proposed actions as well as its potential to engage stakeholders.

Parallel to reducing pressures, policy interventions exert influence on the social and economic dimensions of the catchments. Therefore, building on the international momentum natural capital accounting has gained, Chapter 6 adapts this methodology to the requirements of the Water Framework Directive. Besides presenting the key elements of this approach, Chapter

6 demonstrates how natural capital accounting can be applied in water management. Using two case studies, changes in the extent of aquatic ecosystems are discussed and the asset value of water for residential consumption and recreational purposes is estimated, through the use of economic techniques. Findings demonstrate that the asset value of benefits dwelling from rivers fluctuates from year to year, which indicates how measures that influence the use of water may impact the value of water. Mainstreaming natural capital accounting informs policymakers of the benefits that could be obtained through measures that focus not only on minimizing pressures but increasing a wide range of socioeconomic benefits.

Nature-based solutions constitute such interventions. Chapter 7 reviews their potential and evaluates their role in catchment management within the context of the Water Framework Directive. In essence, the focus of this chapter is to assess how nature-human interdependencies can be considered when policy interventions are designed to increase the overall performance of water management. Furthermore, Chapter 7 examines the effectiveness of a nature-based solution- a constructed wetland installed to improve the effluent quality of a water recycling centre in England. Besides the benefits related to the primary role of its construction, additional ecosystem services benefits are identified and evaluated. The undertaken assessment confirms that the operation of the nature-based solution in tandem with traditional grey infrastructure generated multiple benefits (e.g., carbon sequestration and habitat for species), which highlights the potential of such interventions to improve water management practices and unlock tailor-made private sector investments for the protection of the environment.

Additionally, Chapter 8 provides a systems thinking perspective on the issue of sustainability, in the core of which lies a new role of economics as a discipline that can assist in explaining the interactions of socioeconomic and environmental components of the socioecological system. Based on this, the chapter provides a new vision of sustainable development.

Finally, chapters 9 and 10 constitute the overall discussion and conclusions sections of the thesis. In these the findings of previous chapters are synthesized and critically reviewed, recommendations for improving environmental management decisions are provided and further avenues of research are suggested.

4.2. Methodology

Several methodological approaches were deployed for the development of the thesis depending on the research objectives of each chapter. In each chapter, a review of the literature was conducted to address gaps in relation to the addressed topic. Apart from the introduction, background, discussion and conclusions, the development of all other chapters included gathering and processing qualitative and quantitative information. The concept of economic value is central in this thesis, the estimation of which is done through the application of primary and secondary economic valuation methods. To obtain robust results, priority was given to official national databases and sources, either developed for the purposes of or used for satisfying the reporting requirements of the Water Framework Directive, though other publicly available sources were used in cases such information was not accessible. Detailed material and methods are described in individual chapters; however, approaches taken to methodology included:

- Policy analysis was based on European Union Directives and policy documents, WFD implementation reports, guidance documents and European Commission implementation reports, reports from the Organization of Economic Cooperation and Development, the United Nations, and the World Bank, academic and grey literature as well as reports from EU-funded projects.
- Data gathering from various sources including raw monitoring and WFD classification results data, information on the cost of measures and ecosystem services, Gross Domestic Product, Purchasing Power Parity, inflation rate, and spatial information from sources such as the UK Environment Agency; the Greek Ministry of Environment, Energy and Climate Change; the Broadland Catchment; and the Norfolk Rivers Trust.
- Generation of new data (e.g., estimation of the number of visits in the Broadland catchment) for the estimation of the value of ecosystem services in the Anglian river basin, the Broadland catchment, the Evrotas river basin; the value of natural capital in the Anglian and Evrotas river basins.
- Econometric analysis, including the estimation of the consumer surplus through the use of the travel cost method in R (Core Development Team, 2020), application of variations of the benefit transfer and avoided cost methods and to a lesser extent use of ArcGIS (ESRI, 2011) for obtaining and creating variables used in the economic valuation exercises.

4.3. Research limitations

The following chapters present interdisciplinary approaches to improve environmental management by utilizing a systems thinking perspective on the relationship between humans and nature. Besides extensively reviewing relevant academic literature, the work includes developing new practical and theoretical tools and testing hypotheses to obtain empirical results from case studies. Detailed comments on limitations are presented in the discussion section at the end of each chapter.

Overall, economic assessments of benefits and costs were based on the implementation of different economic techniques, either value transfers or revealed preference techniques. To achieve that, my main sources of data were official national or local databases, the River Basin Management Plans background documents, reports on specific aspects published by the catchment managing authorities, scientific reports as well as published research. Stated preference techniques, such as the contingent valuation method and choice experiments have not been employed for the purposes of this thesis. While the shortcomings of value transfer methods are acknowledged, this was a deliberate choice as its main focus was on developing new environmental management approaches and concepts and not on improving existing techniques used in economics. Furthermore, as Newbold et al. (2018) claim, any exercise that concerns the estimation of costs and benefits includes at least some form of value transfers. Even in the case where a new economic study is conducted for the assessment of policy measures, policymakers would need at least implicitly to assume that the value of costs/ benefits does not vary between the times at which the study is conducted and the policy interventions are implemented. Besides that, in some cases, such as in the estimation of benefits of the Ingol wetland, it could not have been done otherwise, as the gathering of data by local authorities was systematized only after the completion of the project. However, the strength of stated preference techniques to capture non-use values is greatly acknowledged. To this end, future research endeavours could focus more on employing such techniques to assess for instance the heterogeneity in individual preferences that are positively or negatively affected by such projects.

Furthermore, as far as data limitations are concerned, as discussed in Chapter 5, information concerning the status of the environment as well as socioeconomic variables, especially data restrictions related to the Greek case influenced the structure of the chapter. On one hand, due to this fact, extensive data processing and manipulation were needed for obtaining concrete results. On the other hand, the work demonstrated the importance of countries investing in gathering and maintaining high-resolution panel data inventories to allow for better monitoring of occurring changes. Nevertheless, the chapter uses the natural capital accounting methodology, as it is applied by the UK, which relates primarily to the use of ecosystem services. However, such type accounts do not exhaust the possibilities of monitoring a wider spectrum of natural capital characteristics, such as the land-use changes that relate to the extent of the ecosystems; ecosystem condition; and quantification of ecosystem service flows.

Finally, driven by the availability of information, some of which was obtained from the GLOBAQUA Project, a programme funded by the 7th EU Framework Programme, UK case studies were mainly used for empirical applications. While this does not weaken the obtained results, future research endeavours could test the methodologies developed in this thesis in other areas to further assess their applicability and performance. At the end of each chapter, the outcomes are compared with other academic studies. Where a limited number of relevant empirical studies exist (as in Chapters 6 and 7), the validity of the results is assessed through comparisons with findings from the economic valuation literature.

5. Incorporating Ecosystem Services in the Assessment of Water Framework Directive Programmes of Measures

5.1. Introduction

The EU Water Framework Directive recognizes that water ecosystems do not constitute stand-alone structures but are embedded within a wider socio-ecological system and proposes River Basin Management Plans (RBMPs) as the means of achieving the protection, improvement, and sustainable use of freshwater systems across Europe. At the core of RBMPs, Programmes of Measures (PoMs) aim to protect the environment and improve the overall status of the system (Voulvoulis et al., 2017; Vugteveen et al., 2006). PoMs constitute tailored actions implemented by the managing authorities to reduce catchment pressures to levels that are compatible with the achievement of the ecological objectives (i.e., good status of water bodies) introduced by the WFD (Giakoumis and Voulvoulis, 2018b). In developing PoMs, the WFD (Art. 11, par.1) requires Member States to utilize the information gathered in fulfilling the provisions of earlier articles (e.g., Article 5 on the characterisation of the river basin district) and the gap analysis between the current status and the reference conditions. Monitoring and evaluation of implemented measures are crucial for assessing their effectiveness and creating the agenda for the consecutive planning cycle (Europen Commission, 2012).

However, from the submitted river basin management plans, as well as published European Commission reports (European Commission, 2019), it is clear that significant gaps exist in the assessment of PoMs. The 4th Implementation Report (European Commission, 2015) published in 2015, raised concerns about the economic analysis and the link between pressures and PoMs in providing justification for their selection by most Member States. Cost-effectiveness analysis had been suggested by the WATECO group (European Commission, 2003) and had been adopted by most States (Martin-Ortega, 2012) as part of the WFD implementation process. However, its application, as reported in the 1st cycle of the RBMPs varied among countries, with only 8 (Germany, France, Lithuania, Luxembourg, Latvia, Portugal, Romania, and the United Kingdom) out of the 23 countries including this type of analysis when designing measures (European Commission, 2015). Even in these countries, it was not treated consistently across river basins, with some not mentioning it at all or including a general description (WRc, 2015). Differences in the depth of analysis among Member States were also confirmed by the 5th Implementation Report, published in February 2019. More specifically, ambiguity was observed about what costs should be included in the assessment of PoMs, with only one-third of the total number of assessed Member States providing full information (European Commission, 2019). Significant gaps remain in achieving more harmonised approaches to estimate and integrate environmental and resource costs, while it is acknowledged that the economic underpinning of PoMs would greatly facilitate water-related decisions and investments (Gómez-limón and Martin-Ortega, 2013). What often seems to lack in environmental management decisions is the connection between pressures and ecosystem functions (Schröter et al., 2019), which negatively influences economic decisions.

Ecosystems have the potential to supply a range of services that are of fundamental importance to human well-being, health, livelihoods, and survival (Costanza et al., 1997; MEA, 2005; TEEB, 2010), and these services can be described as the benefits that people obtain from ecosystems (MEA, 2005). Recent publications have defined ecosystem services as contributions of ecosystem structure and function (in combination with other inputs) to human well-being (Burkhar et al., 2012; Burkhard and Maes, 2017). The concept of ecosystem services was developed in the 1990s as a way to improve the effectiveness of biodiversity-protection policies (Fisher et al., 2008). Conceptually, it considers the links of biodiversity and ecosystems with socio-economics systems (Boulton et al., 2016). With a global initiative on the economics of ecosystems and biodiversity, which started in 2007, setting a framework for valuing ecosystem services (Bourguignon, 2015), their application to improve economic analysis and contribute to several aspects of the WFD implementation has been acknowledged (Grizzetti et al., 2016; Vlachopoulou et al., 2014). Their application has been shown to allow for a more systematic way to effectively prioritise significant pressures and therefore select appropriate programmes of measures for the WFD (Giakoumis and Voulvoulis, 2018a), and has been suggested for the assessment of policies (Nyborg, 2014). However, their potential to improve the economic underpinning of PoMs and evaluate their effectiveness in economic terms has been underexplored. The current chapter, therefore, develops a framework for the evaluation of the effectiveness of PoMs that considers ecosystem services, as the benefits from improvements in overall water status classifications. Its application is demonstrated through a case study where PoMS are evaluated by comparing expected costs and benefits associated with changes in the delivery of ecosystem services due to their implementation and its potential to facilitate the economic analysis required by the Directive is discussed. The developed methodology, accommodates stakeholders' perceptions and public preferences, as required by the WFD (Article 14) for the design and selection of PoMs (Perni et al., 2020) and the overall implementation of the Directive (Waylen et al., 2019).

5.2. Incorporating ecosystem services in the economic analysis concerning PoMs

Economic principles and instruments are at the core of the WFD and vital for its success, as several of its articles require Member States to undertake economic analysis. Article 4 requires an economic appraisal of disproportionate costs to assess the need for exemptions; Article 5 sets the deadline for the preparation of the economic analysis of water uses; Article 9 requires the assessment of the level of recovery of costs for water services; and Article 11 and Annex III state that the cost-effectiveness of PoMs should be assessed.

Overall, the issue of the effectiveness of adopted policy interventions is central to the WFD and directly related to economic principles. However, as at the time when the WFD was introduced only a few Member States had experience with using economic approaches in environmental management, the initial reports of the Member States were not able to fulfil the economic analysis requirements for assessing effectiveness (Kanakoudis and Tsitsifli, 2010). To assist Member States with these aspects of the WFD, the European Commission published several guidance documents. The first of its kind was the WATECO document (European Commission, 2003) that was developed under the Common Implementation Strategy (CIS) process, which aimed to foster the harmonisation of economic knowledge in the field of water economics throughout the Member States. In 2006, a Cost-Effectiveness Analysis (CEA) document (CEA Drafting Group, 2006) was drafted aiming to prepare the Member States to undertake a more integrated approach to decision-making. After that, a few more studies were developed to provide a standard approach to assessing the effectiveness of PoMs and provide guidance on how to account for their costs and benefits (European Commission, 2010; Nocker et al., 2007).

Generally, though the WFD does not explicitly require the implementation of specific methods, the supporting documents have focused on two approaches, namely CEA and Cost-Benefit Analysis (CBA) for the evaluation of PoMs. One of the main differences between the two methods is that CEA compares costs and physical benefits, whereas CBA compares monetarily valued social, environmental, and economic costs and benefits[4]. Over the years, the European Commission has been supporting cost-benefit assessments, in relation to the costs of PoMs, the benefits of reaching good water status as well as the costs of not achieving the WFD objectives among others (Boeuf et al., 2016). Additionally, England, Scotland, France

[4] Methodological and practical issues of the two methods are further presented in the last section of this chapter.

(Seine, Normandy), the Netherlands and Denmark have shown a strong preference for CBA instead of CEA. However, the lack of a unified framework has often resulted in confusion about what costs and benefits should be accounted for in relevant analyses (Greenhalgh et al., 2017), resulting in weak assessments according to the 5th Implementation Report (European Commission, 2019).

In 2009, in a CIS document (European Commission, 2009) that concerned exemptions under Article 4, the Commission made the first explicit reference to ecosystem services in the context of WFD, signalling the potential added value of integrating this concept into economic analysis to improve its outputs (Bouwma et al., 2018a). Ever since, supporting documents related to the economics of the WFD have been trying to provide a clear link between ecosystem services and the evaluation of policies (e.g. European Commission, 2016) recognizing that they constitute a concept that can accommodate qualitative, quantitative, and monetary assessments (Ozdemiroglu et al., 2010). In addition, there are several studies that demonstrate how ecosystem services can be operationalized for the implementation of the WFD (Heink et al., 2016; Martin-Ortega, 2012; Pistocchi et al., 2016). From a systems point of view, the status of a water body could be considered as an indicator of its overall health, which affects its capacity to generate ecosystem services. PoMs, by managing the pressures in a way to improve water status, affect the functioning of the aquatic system and the consequent benefits it provides to humans. Ecosystem services can be used to frame these benefits more holistically and improve the quantification of the extent to which natural resources contribute to human well-being (Hails and Ormerod, 2013).

The use of ecosystem services in economic valuation exercises has been a growing trend in the academic literature in the last decades (Birol et al., 2006; Doherty et al., 2014; Eftec, 2005; Liu et al., 2010; Pavanelli and Voulvoulis, 2019; Voulvoulis, 2015). In these studies, different WFD status class categories are defined by the quantity of ecosystem services they provide. However, these studies do not provide an actual framework on how to integrate their results into decision-making, thus they have been rarely utilized (Laurans et al., 2013). Consequently, interdisciplinary methodologies to integrate social as well as environmental benefits that give a practical explanation of how they can be applied in real cases have yet to be developed.

5.3. An ecosystem services framework for assessing the cost-effectiveness of PoMs

The WFD requires Member States to undertake economic analysis in order to make judgements about the best combination of measures that will achieve the Directive's objectives. Considering the costs of the selected PoMs and their impacts on water body status through the elimination of pressures expressed as changes in the delivery of ecosystem services (Grizzetti et al., 2019), this section proposes evaluating PoMs effectiveness as the ratio of the value created from these changes elicited through relevant stakeholders' preferences to the implementation costs. Taking into account the lack of harmonised approaches and that many EU Member States have not yet developed CEA methodologies; the proposed framework (Figure 5.1) can help clarify the national approaches and enable their comparability.

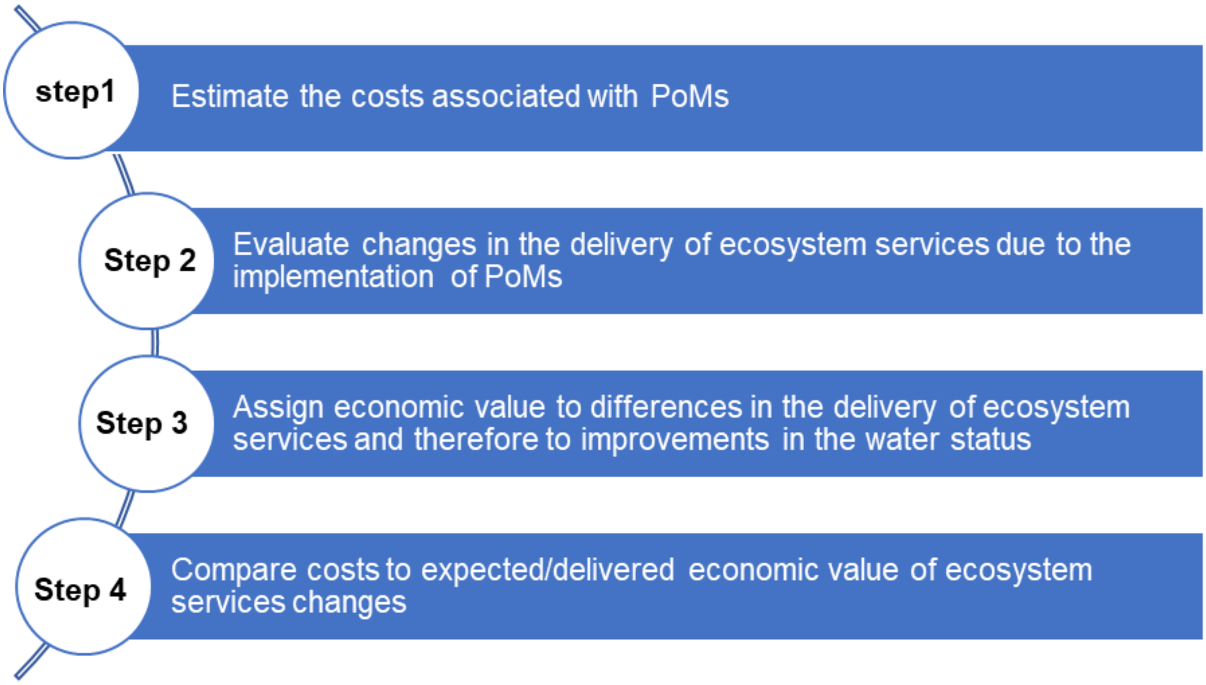

Figure 5.1 Framework for assessing the effectiveness of selected or implemented PoMs

Step 1: Estimation of the costs of PoMs

To assess the effectiveness of implemented PoMs, the first step includes the evaluation of all negative impacts of these measures in economic terms. The magnitude of costs may be affected by the type of measures, their duration as well as the area they target. This should be straightforward, as most competent authorities include in the River Basin Management Plans the cost estimates of the associated basic and supplementary PoMs. Such costs are capital/ investment and operational costs, as well as any other negative impacts that generate welfare losses (for example, a policy intervention may increase consumer prices or decrease production output). To put that into perspective, the potential costs of measures to achieve the water body and protected areas objectives in the Anglian river basin that are included in the second River Basin Management Plans (2015-2021) are estimated to be £ 5,050 million and £ 4,740 million (undiscounted) in the Thames river basin (Defra and Environment Agency, 2015a, 2015b).

Step 2: Evaluation of improvements in the provision of ecosystem services due to the implementation of PoMs.

The second step aims to evaluate the effectiveness of the PoMs in improving the provision of ecosystem services through the elimination of pressures on water bodies. It involves understanding the level and type of delivery of ecosystem services before and after any policy intervention (expected or delivered). Although there is no standardized way to identify and assess ecosystem services (Malinga et al., 2013), such information might be drawn from assessments undertaken by the managing authorities or local knowledge. The aim of this step is to evaluate the effectiveness of the PoMs as changes in the provision of ecosystem services before and after their implementation as a result of reducing pressures on water bodies. In obtaining such information, the following cases are recognized (Figure 5.2):

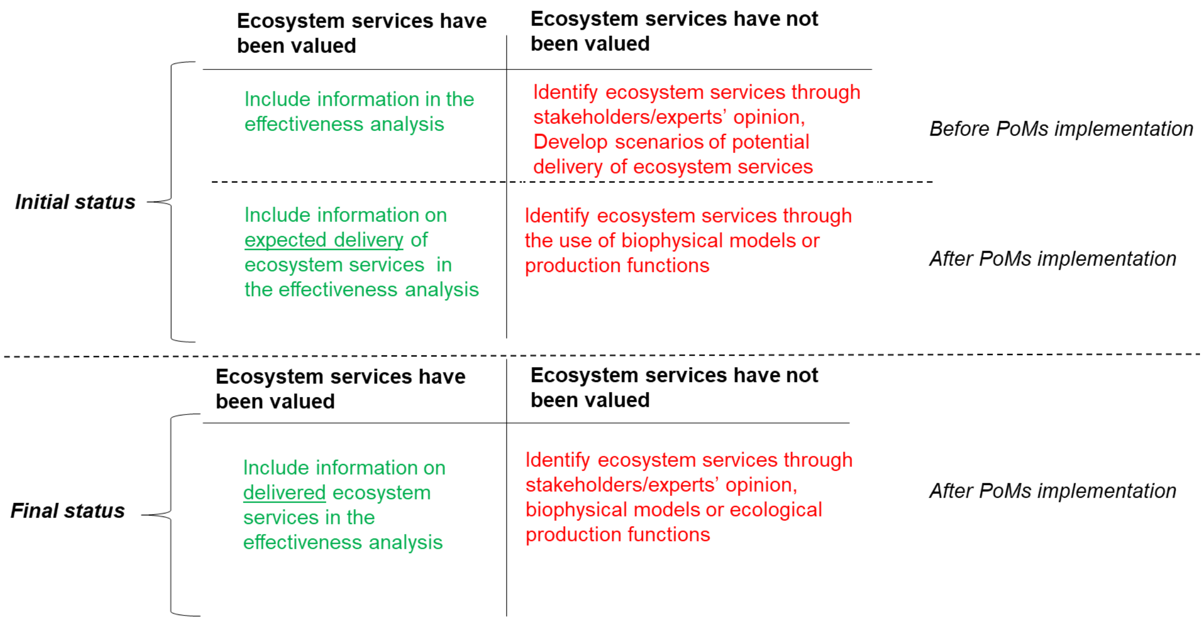

Figure 5.2 Assessment of changes in ecosystem services due to PoMs implementation

i) *Ecosystem services provision has been evaluated both before and after the implementation of PoMs.*

In the case where identification and quantification of ecosystem services delivery have taken place, the analysis should proceed to the next step, whether an ex-ante or ex-post assessment of the effectiveness of PoMs is concerned.

ii) *Ecosystem services provision has not been assessed or has been assessed for either before or after the implementation of PoMs.*

When PoMs have been developed without any consideration of ecosystem services, identification of ecosystem services can be realised through consultation with stakeholders and/or experts or the deployment of models that are able to assess them. In the case of an ex-ante assessment of the effectiveness of PoMs, the expected delivery of ecosystem services can be characterised based on alternative land use scenarios (Brauman et al., 2011; Egoh et al., 2012; Maes et al., 2013). Other types of analysis such as ecosystem structure and habitat data (Raffaelli, 2006) or functional traits of plants (Lavorel and Grigulis, 2012) have been used in the past and could be operationalised for ex-post assessments. Additionally, catchment stakeholders could engage in this process through participatory workshops (García-Nieto et al., 2015), where they express their knowledge and opinion on the effects of specific PoMs on ecosystem services.

Step 3: Assigning economic value to improvements in the delivery of ecosystem services that have resulted from the implementation of the PoMs.

The changes in the provision of ecosystem services established in the previous step, need now to be evaluated in monetary terms (Saarikoski et al., 2016). Several techniques exist for valuing the use (related to the actual use of a service) and non-use values (related to passive use of a service) of ecosystems. A strand of the literature has focused on revealed preference methods, such as hedonic pricing (Day, 2001), travel cost (Loomis, 2003) and averting behaviour; whereas another strand has been concerned with applying stated preference methods, such as the contingent valuation method (Pinto et al., 2016) and choice modelling (Andreopoulos et al., 2015). The choice of the most suitable technique depends on the ecosystem services to be valued (see, for example, Reynaud and Lanzanova, 2015). Additionally, benefit transfer methods could be cost-effective alternatives. These methods consist of procedures for transferring estimated economic benefit values from a study to a policy site (Plummer, 2009). Given that a substantial number of valuation studies have been performed, benefit transfers started becoming a standard approach in the 1990s (Boutwell and Westra, 2013). Although such methods exhibit several shortcomings (Boutwell and Westra, 2013), one of the main advantages is the low cost of applying them, since a low volume of site-specific data is not necessary to be gathered.

Step 4: Comparison of the results of the expected or delivered value of ecosystem services to the costs of PoMs (efficiency).

The assessment of the effectiveness of PoMs can now be undertaken through the comparison of the economic value of ecosystem services resulting from the implementation of specific PoMs (Step 3) with the costs of the implemented measures (Step 1). As costs and benefits may be distributed over a number of years, their values need to be turned into current values using a discount rate. This process can evaluate the effectiveness of implemented measures not just in terms of improving water status classification through eliminating significant pressures, but also delivering welfare benefits when the economic value of eliminating pressures thus improving water status outweighs the costs of the implemented measures.

5.4. Materials and Methods

The Broadland River catchment, in the UK, was selected as a case study for the application of the ecosystem services framework for assessing the effectiveness of PoMs, to demonstrate how it could be operationalised and utilised by water managers at different river basins. This was based on the availability of information dwelling from the background documents used for the development of the River Basin Management Plans.

It should be noted that the following application is only an example that showcases how the methodology could be used to evaluate the effectiveness of selected PoMs. A condition for its application is that the PoMs have been appropriately designed to target catchment pressures to levels that are compatible with the achievement of the ecological objectives introduced by the WFD, or in other words, they have been designed properly to deliver water status improvements by minimizing pressures and not just targeting improvements in elements classification as shown to be often the case (Giakoumis and Voulvoulis, 2019).

5.4.1. The Broadland Rivers catchment, UK

Broadland Rivers is a catchment in the Anglian River Basin District studied by the GLOBAQUA project[5]. The Broadland Rivers catchment covers an area of 3200 km^2 and it is mostly rural (Figure 5.3). The catchment includes 94 river water bodies with the four main sub-catchments being the Bure, Wensum, Yare and Waveney and 19 lake water bodies (Environment Agency, 2014).

In 2014, the Broadland River Catchment Partnership developed a strategic plan for managing key issues in the catchment such as water quality, water quantity, wildlife habitat and recreation. The catchment plan included 7 goals and 19 actions related to the management of land, water, wastewater, flood risk and sustainable drainage, river and floodplain, recreation, and investments to increase funding for the projects (Broadland Catchment Partnership, 2014). To achieve these goals and also meet the objectives of the WFD, 84 measures (51 basic and 33 supplementary) were selected during the 1st implementation cycle (Environment Agency, 2010, 2015). However, at the end of 2014, the number of water bodies below good status increased to 108 out of 111, compared to 102 in 2009, indicating that the adopted PoMs were not effective. Giakoumis and Voulvoulis (2019) claim that the reason for this was that

[5] GLOBAQUA: Managing the effects of multiple stressors on aquatic ecosystems under water scarcity. Funded by the European Union's Seventh Programme for research, technological development, and demonstration under grant agreement No 603629.

managing authorities focused on managing quality elements rather than catchment pressures, treating the symptoms but not the causes.

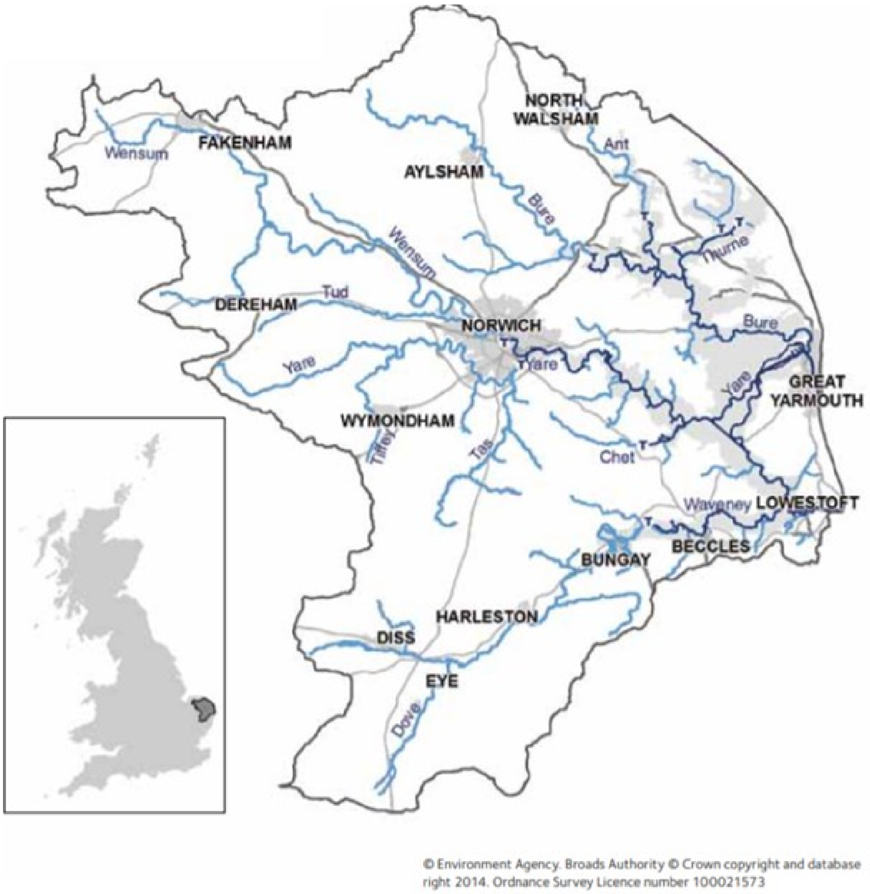

Figure 5.3 The Broadland Rivers catchment

The completion of the 1st implementation cycle plans was followed by the adoption of measures included in the 2nd management cycle plans. The new and updated measures concerned improving modified physical habitats, and managing pollution from wastewater, urban sources and transport, and from rural areas. Overall, 71 policy interventions were considered (Table A.1, Table A.2, and Table A.3 in **Appendix A**).

5.4.2. Collection of data

For the application of the assessment framework for PoMs in the Broadland Rivers catchment, I collected data (e.g., types and costs of measures, types and importance of ecosystem services, value of ecosystem services etc.) from various sources. More specifically, information on costs was taken directly from the economic appraisals for the second River

Basin Management Plan of the Anglian Region provided by the Environment Agency[6] upon request. Data on stakeholders' perceptions of the connection between local pressures and ecosystem services was gathered by a team of Imperial College London researchers during a participative ecosystem service valuation workshop through dynamic group and individual activities (GLOBAQUA, 2018). The agenda of the workshop included a presentation on existing work on ecosystem services in the catchment area; the identification of ecosystem services; as well as the application of a methodology on pressure prioritisation by Giakoumis and Voulvoulis (2019), results of which I use in this chapter to establish the connection between ecosystem services and PoMs. Participants (33 in total) of a Broadland Rivers Catchment Steering Group workshop formed groups and identified 37 ecosystem services as relevant in their catchment. Each stakeholder group was introduced to the concept of ecosystem services as per TEEB classification and was asked to place each service in the following groups: provisioning, regulating, cultural, and supporting services. Information on the possible change in ecosystem services due to the implementation of PoMs was obtained from the catchment summary report (Environment Agency, 2014).

5.4.3. Values obtained for the implementation of the benefit transfer approach

To estimate the economic value of benefits resulting from the elimination of pressures due to the application of PoMs, I used the benefit transfer approach, which was based on transferring values from previously conducted economic studies to the Broadland Rivers catchment. The unit transfer method that I use assumes that the marginal value to the average individuals is similar between two sites (Navrud and Ready, 2007). However, to minimise transfer errors, after a thorough and detailed review of the economic valuation literature, I selected several studies based on their relevance to the ecosystem services identified for the Broadland Rivers catchment. To achieve that, I used the Environmental Valuation Reference Inventory[7] (www.evri.ca) that consists one of the largest sources of economic valuation studies, containing 5305 economic studies published from 1971 onwards. More specifically, in January 2021, I accessed the database and used specific keywords ("ecosystem services", "water quality" "recreation", "soil erosion", "drinking water", "irrigation" "species", "wildlife", "food") to obtain a list of 371 primary studies conducted in the past. By focusing on studies in Europe whenever possible; cases with similar ecosystems (i.e., different types of inland waters); areas

[6] Public sector information licensed under the Open Government Licence v3.0
[7] The Environmental Valuation Reference Inventory is a databank that contains empirical studies on the economic value of environmental assets and human health.

with similar socio-economic characteristics; as well as areas that describe similar changes in the status of water resources, I selected a number of 18 studies that I used for the economic valuation exercise. While a detailed list of the literature considered for the estimation of the socioeconomic value of ecosystem services can be found in Appendix A Table A.4, in summary, developing countries with heavily polluted ecosystems were excluded, with most of the studies selected sharing the same policy framework with the Broadland Rivers catchment. Furthermore, given differences in socioeconomic characteristics as well as the time when the primary studies were conducted, I adjusted the obtained values for income[8] and time differences, using data on GDP per capita, PPP (Purchasing Power Parity) and CPI (Consumer Price Index) from the World Bank database (https://data.worldbank.org/). Therefore, I performed the following adjustments:

Adjustment for differences in income levels

$$WTP_{policy\ site} = WTP_{study\ site}\left({Y_{policy\ site}}/{Y_{study\ site}}\right)^{E}$$

where WTP denotes the willingness to pay estimate for the policy and study site; Y indicates the income per capita; and E the income elasticity of willingness to pay that describes how responsive is the willingness to pay to changes in the level of income.

Adjustment for the year of value and general price levels

$$WTP_{policy\ site} = WTP_{study\ site}\left({GDPd_{policy\ site}}/{GDPd_{study\ site}}\right)$$

where WTP denotes the willingness to pay estimate for the policy and study site; and $GDPd$ indicates the GDP deflator index for the year of the policy site assessment and the study site valuation.

Adjustment for purchasing power and currency

$$WTP_{policy\ site} = WTP_{study\ site} * PPP$$

where WTP denotes the willingness to pay estimate for the policy and study site; and PPP indicates the purchasing power parity adjusted exchange rate between the policy and the study site currencies.

Having obtained the adjusted WTP for different ecosystem services, I then estimated the average value per ecosystem service category based on the relevance of each service for each subcatchment.

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[8] Following the study of Koundouri et al. (2014), an income elasticity of demand for the environmental good in question of 0.5 is used in this valuation exercise.

5.5. Results

The application of the framework in the Broadland Rivers catchment allowed for the assessment of the effectiveness of PoMs selected by the managing authorities to be implemented from 2015 onwards to improve water status by 2021. Results are presented in the following sections that correspond to the steps of the framework for the four sub-catchments of the study area.

5.5.1. Cost of PoMs implemented in the Broadland catchment area

For the estimation of the PoMs costs, I compiled data from the WFD background documents of the Environment Agency that were provided to me under the Freedom of Information Act 2000 and Environmental Information Regulations 2004 According to these documents, investment costs, administrative and operational costs, and resource costs, were obtained through the use of hydro-economic modelling (Brouwer et al., 2009) and environmental costs were estimated with the use of market and non-market valuation techniques. For each of the four sub-catchments of the Broadland Rivers catchment, a description of measures and the associated investment and operational costs are presented in Table 5.1 (more detailed information on the measures can be found in Tables Table A.1, Table A.2 and Table A.3 in Appendix A).

Table 5.1 Capital and operational costs of the PoMs in each sub catchment

Operational catchment	Measures related to	Total length of the river (km)	Capital costs	Operational costs (per year)
Yare	Catchment sensitive farming (arable and farming, nutrients); nutrient reduction- phosphate stripping; surface run-off and drainage; enabling fish passage; increasing channel morphological diversity; upgrading existing private sewage systems; channel maintenance strategies; removing obsolete structures;	158.14	£6,282,006	£144,555

Operational catchment	Measures related to	Total length of the river (km)	Capital costs	Operational costs (per year)
	and improving sustainable drainage.			
Bure	Catchment sensitive farming (pesticide management); improving in-field grass buffer strips on tillage land and improving riparian buffer strips; enabling fish passage; increasing channel morphological diversity; planting trees; controlling and eradicating of selected high-risk species; supporting established local fora by providing advice and guidance; sharing best practice; increasing awareness of the 'preventative approach'; improving rural sustainable drainage system within fields, tracks and rural road system; upgrading existing private sewage systems; channel maintenance strategies; and removing obsolete structures.	156.03	£6,115,410	£104,075
Waveney	Catchment sensitive farming (arable, farming, livestock, pesticide management, nutrients); nutrient reduction-phosphate stripping; improving in-field grass buffer strips on tillage land and riparian buffer strips; improving rural sustainable drainage system within fields, tracks, and rural	209.26	£9,396,387	£306,032

Operational catchment	Measures related to	Total length of the river (km)	Capital costs	Operational costs (per year)
	road system; enabling fish passage; increasing channel morphological diversity; and planting trees.			
Wensum	Catchment sensitive farming (arable, farming, livestock, pesticide management, nutrients); nutrient reduction-phosphate stripping; improving in-field grass buffer strips on tillage land and riparian buffer strips; improving rural sustainable drainage system in highway, road, site and housing estate drainage, as well as within fields, tracks and rural road system; enable fish passage; increasing channel morphological diversity; supporting established local fora; share best practice; increase awareness of the 'preventative approach' channel maintenance strategies; remove and/or modify obsolete structures; eradication and control of invasive non-native species at selected sites of special scientific interest (SSSI) and Natura 2000 sites; and the Wensum restoration strategy.	170.53	£10,796,810	£205,887

The cost estimates differ across sub-catchment based on the length of they are implemented on and/or their type. For example, the costs of measures in Yare and Bure do not vary significantly given that the length that is covered and the types of measures are similar. On the other hand, though measures in Wensum concern a similar length of the catchment, a greater number of measures increase the capital and operational costs.

5.5.2. Changes in the provision of ecosystem services in the catchment area as a result of the PoMs

In order to establish the connection between selected PoMs and ecosystem services, I made use of the responses of stakeholders[9] during a workshop that took place under the GLOBAQUA project[10]. Stakeholders identified the pressures and ecosystem services that are relevant to the Broadland Rivers area. In a recent paper, Giakoumis and Voulvoulis (2018a) analyzed the responses and provided the connection between the identified pressures and ecosystem services. Based on their findings, in the current study, I associated the selected PoMs with the identified ecosystem services through the types of pressures they had been designed to address. Overall, stakeholders neglected supporting services, arguing that such types of services do not provide direct benefits to humans (Haines-Young and Potschin, 2012). Therefore, for the purposes of the undertaken analysis, PoMs are associated with these three types of services (Table 5.2). Additionally, since four groups of stakeholders participated in the workshop, the current study considers the links between pressures and ecosystem services that were recognized by at least three out of the four groups.

Table 5.2 Association between PoMs addressing identified pressures and ecosystem services

Type of pressure addressed by PoMs	**Pressure relevant to catchment**	**Ecosystem services categories**	**Identified ecosystem services sub-categories**
Diffuse source pollution	All catchments	Provisioning services	Intensive farming (poultry, pigs)
			Water quality drinking water
			Water for industry
			Water for irrigation
			Water for breweries

[9] Among the participants were representatives of the Broadland Catchment Partnership, the Broads Authority, the Anglia Ruskin University, the Anglian Water Services Limited, the Country Land & Business Association Limited, the Department for Environment, Food & Rural Affairs, the Environment Agency, the Northumbrian Water Ltd, the National Farmers Union, the National Trust, the Natural England body, the Norfolk County Council, the Norfolk Wildlife Trust, the River Waveney Trust, the Rivers Trust, the Suffolk Wildlife Trust, the University of East Anglia, the Society for the Protection of Birds and the Water Management Alliance.

[10] Methodology and results of ecosystem services co-definition and participative valuation workshop are described on GLOBAQUA (2017).

			Arable ponds (wildlife)
			Timber (fuel wood), coppice
			Wind energy, solar energy, biomass
		Regulating services	Natural hazards regulation (flooding)
			Soil erosion
			Carbon sequestration
			Air quality (woodland)
			Natural water purification
			Drought protection
		Cultural services	Health and wellbeing
			Tourism coasting holidays
			Local recreation (angling, bird watching, boating)
			Archaeology (built buried)
			Walking, Cycling
			Sense of place, uniqueness
			Landscape beauty: Big skies, wilderness, tranquillity
			Dark skies
Point source pollution	Yare, Wensum, Waveney	Provisioning services	Water quality drinking water
		Regulating	Soil erosion
Modified habitat	All catchments	Provisioning services	Water quality drinking water
			Arable ponds (wildlife)
		Regulating services	Natural hazards regulation (flooding)
			Attenuation of sea level change
		Cultural services	Health and wellbeing
			Tourism coasting holidays
			Local recreation (angling, bird watching, boating)
			Archaeology (built buried)
			Sense of place, uniqueness
			Landscape beauty: Big skies, wilderness, tranquillity
			Dark skies
Other anthropogenic	Wensum, Waveney	Provisioning	Grazing marsh
			Water for irrigation
			Reed and sedge
		Regulating	Natural hazards regulation (flooding)
			Attenuation of sea level change

To evaluate changes in ecosystem services from the implementation of the selected PoMs, I used data from a catchment summary report, published by the Environment Agency (Environment Agency 2014). For each of the main sub catchments, Table 5.3 presents the direction (positive/negative) as well as the magnitude of projected impacts of ecosystem services by 2021 as a result of the implementation of PoMs.

Table 5.3 Impact of ecosystem services in the Broadland Rivers catchment accruing from measures implementation. Pluses and minuses express the magnitude of the effect on each ecosystem service (Environment Agency 2014).

Type of ecosystem services	Sub catchment			
	Bure	Waveney	Wensum	Yare
Freshwater	+	++	+	+
Food	+			-
Climate regulation	+	++	+	+
Erosion regulation	+	++		+
Water regulation		+	+	+
Water purification and waste treatment		++	+	+
Nutrient cycling		++	+	+
Provision of habitat	+	+	++	+
Aesthetic value		+		
Recreation and tourism	+	++	+	+
Existence values	+			
Cultural heritage		+	-	-

The expected changes differ in magnitude across sub-catchment areas, but most of them are positive. In Waveney, the provision of ecosystem services is expected to be increased the highest, whereas, in the cases of Wensum and Yare, some services will be negatively impacted. Overall, the information presented above is in line with recent findings concerning the provision of ecosystem services at different water statuses (Grizzetti et al., 2019).

5.5.3. Value of expected changes in the delivery of ecosystem services

For estimating the expected value of changes in ecosystem services in the Broadland Rivers catchment, as described above, I used the benefit transfer approach that utilized WTP estimates from pre-existing primary studies at similar sites to estimate the relevant values for the study site. Using this method, the following results were obtained (Table 5.4).

Table 5.4 Type of services and range of values obtained from the benefit transfer application.

Type of service	Range of values (£/person/year)
Provisioning	1.96-15.54
Regulating	1.23-72.88
Cultural	0.48 -23.20

The variation in the values can be attributed to the fact that the selected studies included both finite and infinite time horizons. To reduce uncertainty[11] in monetary estimates, several studies per ecosystem service were used and the time horizon of each selected study was considered. The final estimates included in the analysis were average values of all estimates per selected study. More specifically, after adjusting the value of each ecosystem service sub-category obtained from each study, I averaged these values to obtain the adjusted average value per ecosystem service. Supplementary material regarding the minimum, maximum and average value per sub-category of ecosystem services is included in Table A.5 in Appendix A. I then calculated the aggregated value of ecosystem services by considering the population in each sub catchment or the area of each sub catchment in cases where the unit of value was pounds per hectare. This approach does not account for preference heterogeneity that among others may be related to the distance of the stakeholders from the resource (Bateman et al., 2006), therefore, the obtained values may be overestimated due to transfer errors (Johnston and Rosenberger, 2009; Ready and Navrud, 2006). For this reason, a transfer error of 70% was considered, which is within the transfer error estimates recognized by relevant studies (Stellin and Candido, 2006) (Table 5.5). Following the assumptions for the estimations of costs and benefits by the managing authorities, as well as the recommendations of the Green Treasury Book (H.M. Treasury, 2022), I used a discount rate of 3.5%.

[11] The term relates to the uncertainty embodied in the economic estimates of relevant economic studies.

Table 5.5 Value of ecosystem services in the four operational catchments (£ in 2015)

Category	**Bure**	**Waveney**	**Wensum**	**Yare**	**Discounted Total (40-year period, 3.5% discount rate)**
Provisioning (e.g., consumption of water for domestic and agricultural use)	50,231,898	122,391,854	50,725,269	54,212,506	277,561,526
Regulating (e.g., flood and erosion control)	104,673,371	353,405,720	162,345,138	166,842,335	787,266,564
Cultural (e.g., recreation, landscape beauty, sense of place)	38,089,579	140,149,944	59,075,751	60,712,235	298,027,509

5.5.4. Comparison of expected costs and benefits due to the implementation of PoMs

The values obtained in step 4 were aggregated given the number of inhabitants estimated using the area of each river catchment and the population density obtained from the Office for National Statistics published data (Office for National Statistics, 2011). The following formula was used to estimate the net present value of cost:

$$\sum_{t,i}^{T,N} \frac{w_i \hat{B}_{i,t} - \hat{C}_t}{(1+r)^t}$$

where $\hat{B}$ and $\hat{C}$ are the estimated benefits and costs, w is the probability of obtaining the estimated benefits and r the discount rate. The time horizon used for the estimation of benefits is 40 years and the success rate of PoMs was 70%, similar to that included in the economic appraisal of measures of the Environment Agency (Table 5.6).

Table 5.6 Net present value of benefits and costs (£ in 2015) incorporating the value of ecosystem services in the Broadland Rivers basin sub-catchments

Sub-catchment	Present value of costs	Present Value of Benefits	Net present value	Benefit-Cost ratio
Discount rate: 6%				
Yare	8,457,026	139,718,116	131,261,091	16.5
Bure	7,681,365	95,652,582	87,971,217	12.5
Waveney	14,001,037	163,461,551	149,460,514	11.7
Wensum	13,894,652	134,953,270	121,058,618	9.7
Catchment total	**£44,034,079**	**£533,785,519**	**£489,751,440**	**12.1**
Discount rate: 4.5%				
Yare	8,942,050	170,279,403	161,337,353	19.0
Bure	8,030,568	116,611,701	108,581,133	14.5
Waveney	15,027,863	199,222,156	184,194,293	13.3
Wensum	14,585,463	164,467,755	149,882,292	11.3
Catchment total	**£46,585,944**	**£650,581,014**	**£603,995,071**	**14.0**
Discount rate: 3.5%				
Yare	9,368,991	197,236,953	187,867,962	21.1
Bure	8,337,955	135,096,394	126,758,439	16.2
Waveney	15,931,725	230,765,460	214,833,735	14.5
Wensum	15,193,549	190,502,311	175,308,762	12.5
Catchment total	**£48,832,219**	**£753,601,117**	**£704,768,898**	**15.4**

As the temporal distribution of benefits and costs is not known, a sensitivity analysis was undertaken (for more information, see for example Pearce et al., 2006; Moore et al., 2004) to investigate how these would respond to different discount rates (6%, 4.5% and 3.5%). The higher the discount rate the less value is placed on benefits/costs the further they are in the future. Under every scenario, the benefits of implementing the selected water measures were found to be higher than the relevant costs.

Benefits and costs were not found to be distributed uniformly across the four sub catchments. Regulating services seem to obtain the highest value in every sub catchment. This is in line with earlier results of similar estimation exercises (e.g., Koundouri et al., 2015; Costanza et al., 2006). Cultural services are less valued, which contradicts the results of other studies (Ghermandi et al., 2010), however, this should be attributed to the magnitude of change in each area, as well as negative impacts on sights of significant cultural and aesthetic importance due to the implementations of PoMs. Another reason associated with this might be that though river landscapes incorporate high aesthetic and cultural value (Thiele et al., 2019), it is difficult to quantify such non-material values and associate them with alterations in ecosystems (Verbrugge et al., 2019). Secondly, the total value of benefits is influenced by the population in each area, as well as the expected magnitude of change in the provision of ecosystem services. The ecosystem services are expected to be affected the most in Waveney, where according to estimates the majority of the population is located. As a consequence, the highest values of benefits are associated with Waveney under each scenario.

Besides that, the benefit-cost ratios seem significantly higher than those included in the catchment summary report, which presents benefit cost ratios ranging from 1.24 to 4.9. This can be attributed to two factors. The first is that though supporting services were excluded from the analysis included in this chapter, the specific subcategories of ecosystem services associated with significant pressures in the area reported by the stakeholders are broader than that included in the catchment summary report that mostly was the result of expert opinions. Therefore, if the values presented in this study are not subject to transfer error higher than 70%, stakeholders' participation may result in capturing a wider range of benefits. This means that had the PoMs been designed to address pressures and not improved the elements classification, the net benefit resulting from water status enhancement would be higher than that reported in the catchment report. Secondly, the cost estimates also differ. As cost data were taken from official sources, it is not clear why these values are different from those reported in the catchment report, nor what kind of cost elements (economic, environmental, resource) are included in each case. Nevertheless, the NPV estimates sustain a positive sign regardless of the discount rate, which is in line with the previous results included in the catchment summary report.

Finally, the costs are higher in sub catchments with a higher number of water bodies, where more interventions take place. Additionally, the costs of PoMs for Bure, Wensum and Waveney that include a mix of technical and non-technical measures (e.g., sharing best practices) are relatively lower than the PoMs for Yare as they consist mainly of technical interventions. In terms of financing, this might mean that when capital is a significant constraint, and the ecological status of water is not heavily impacted, non-technical measures that focus on the changing of stances in relation to water resources might be an alternative cost-effective route.

5.6. Discussion

Achieving the water status classification objectives of the WFD requires in-depth understanding of the interactions between the natural and social systems (Voulvoulis et al., 2017). Managing authorities require knowledge of the sub-systems embodied in each catchment and developing management tools that are able to influence how these subsystems interact with one another. In support of this, the WFD adopted the Drivers-Pressures-State-Impacts-Responses (DPSIR) framework (Commission, 2003). As a result, when implementing the Directive, Member States need to assess the gap between the current status and optimal environmental conditions defined as a status, where pressures are absent or unable to affect water quality (Voulvoulis et al., 2017). Consequently, the main purpose of PoMs is to alleviate identified pressures and their effects on waters. Additionally, effective PoMs, need to be able to achieve an equilibrium among various often conflicting objectives related to these subsystems, ultimately reducing the gap to good water status.

Deciding on the most suitable measures should be based on information on their ability to tackle pressures as well as on their costs. The WFD necessitates the use of economic principles and techniques to assess their effectiveness. In many instances, CEA has been used for this procedure, however, the literature recognizes several issues (Martin-Ortega, 2012; Messner, 2006). First, CEA might neglect social aspects of water status improvements, which might impact the actual implementation of policies. Secondly, the costs of measures may exhibit nonlinearities and may be space, time, and scale specific, which makes comparisons of CEA results problematic. Additionally, measures may have indirect side effects on “separate spheres” (Brock, 2003), which may be beyond the scope of the environmental problem they aim to tackle. Another method that has also been suggested by the European Commission and overcomes some of the significant flaws of CEA is CBA, but its application requires caution as it is conditional to the appropriate design of PoMs. CBA assigns monetary values to direct and indirect costs and benefits of policy intervention and can be used for assessing the economic efficiency of environmental policies both ex-ante and ex-post. At the core of CBA is the assessment of whether an environmental policy results in achieving the desired objective, while improving social wellbeing by generating use and non-use benefits (Hanley et al., 2006; Hanley and Black, 2006). As CEA, CBA has also received criticism. According to Sunstein (2005), WTP is not always an appropriate measure, as through it, citizens may express their judgements instead of their preferences. Furthermore, given biases embodied in the valuation methods used to elicit economic value, philosophical discussions about CBA focus on the difficulty of assigning economic value to what is conceived as invaluable (Hansson, 2007), but the use of ecosystem services alleviates this challenge to an extent.

The 5th Implementation Report states that 11 out of the total number of EU Member States developed some kind of CBA (European Commission, 2019) and though steps were made in performing economic analysis, significant gaps still exist in translating the results of economic analysis in measures, thus follow more integrated water management approaches. Member States have been focusing more on complying with the requirements of the Directive, than harvesting the true potential of the WFD (Behagel, 2012; Giakoumis and Voulvoulis, 2018b; Petersen et al., 2009). This fact has been expressed through the insignificant increase in the number of water bodies whose status improved after the implementation of measures. In line with this, the current study presents a case where although the economic analysis of selected measures was sophisticated and took into account environmental aspects, the results in terms of enhancing waters have been discouraging, demonstrating the importance of developing appropriate PoMs. Unless the environmental problems are framed properly in terms of pressures and impacts, there is little hope in evaluating the effectiveness of measures when their application does not deliver overall status improvements.

Recognizing this, as well as the results of a previous study (Giakoumis and Voulvoulis, 2018a), the methodology presented in this chapter describes a way of assessing PoMs by connecting pressures to measures and ecosystem services. At the heart of the methodology lies the idea that improvement in the water status results from mitigating pressures on water. Consequently, following the spirit of Forrest et al. (2020), the developed methodology requires the selected combination of measures to enhance water status. Making use of the principles of CBA for assessing the effectiveness of measures, the study incorporates ecosystem services into the analysis to achieve a straightforward connection between the costs and benefits (social and environmental) of policy measures and the impacts on the wellbeing of relevant stakeholders. The use of ecosystem services in the assessment of policies facilitates a connection between the environmental and social systems (Maes et al., 2018) and comprehensive communication of the benefits of effective implementation of the WFD, thus has the potential to promote commitment to policy decisions (Howarth, 2009). Additionally, it provides a systemic view of the nature-society relationship (Voulvoulis, 2012) and decreases the risk of adopting traditional standardised practices that are not related to the catchment (Sabatier et al., 2005). Lastly, it enables managers to address multiple goals (Everard, 2014) which could have added benefits for the European Union, where environmental management practices are defined by extensive legislation for the different aspects of environmental systems (Beunen et al. 2009; Bouwma et al. 2018a; Jordan and Lenschow 2010; Schleyer et al., 2015). The empirical analysis demonstrates how the methodology for assessing the effectiveness of PoMs can be applied prior to the implementation of measures. Its application is information-intensive, as several types of data are needed for fulfilling its steps. The cost of PoMs is an essential part of the analysis, therefore such estimates should either be collected through WFD documents or be estimated. In addition to that, in order to harvest the benefits of this methodology, data on ecosystem services provision is essential. However, as the adoption of this concept is a growing trend in several countries (e.g., the UK National Ecosystem Assessment (Watson et al., 2011), the Spanish National Ecosystem Assessment (Fundación Biodiversidad, 2014), the Portuguese Millennium Ecosystem Assessment (Pereira et al., 2009)), as well as in scientific projects funded by the EU (e.g., GLOBAQUA, MARS, OpenNESS), such information might already be available in several EU countries. Concerning the estimation of the value of ecosystem services, in the empirical example presented in this chapter, a Benefit Transfer method was applied. Due to possible transfer errors (Boutwell and Westra, 2013; Johnston et al., 2018; Kaul et al., 2013), primary studies (e.g., hedonic pricing and choice experiments) should be used in cases where capital and time constraints are less strict. Furthermore, in the study, I used a constant discount rate to estimate the net present value of costs and benefits. Using such a discount rate, the present value of future costs and benefits becomes less and less important the further the distance from the present, thus impacts far in the future are irrelevant to decisions made today (Groom et al., 2005). However, fairness considerations of sustainable development require future generations to be taken into account, a problem which recent research has tried to solve by employing time-declining discount rates (Pearce et al., 2003; Sáez and Requena, 2007; Koundouri, 2009). Such discount rates increase the weight placed on future values compared with conventional constant rates (Guo et al., 2006). The use of such discount rates is justified by evidence concerning individual’s discount rates that decline with time (Loewenstein and Prelec, 1992; Henderson and Bateman, 1995; Frederick, Loewenstein and O’Donoghue, 2002; Karp, 2005); uncertainties about future economic conditions (Weitzman, 1998, 2001; Gollier, 2002b, 2002a); considerations of intergenerational equity that require increased weight to be placed on future generations (Chichilnisky and Heal, 1997; Heal, 2000, 2005; Li and Löfgren, 2000; Chichilnisky, 2017); and heterogeneous time preferences (Qiang and Ogaki, 2000; Pleeter and Warner, 2001; Gollier and Zeckhauser, 2021). If the time horizon of implemented measures spanned over several hundred years, a declining rate might be a more appropriate choice for discounting future values (Groom et al., 2005). In the case of this study, while the net present value estimated with a declining discount rate might have been higher than that estimated with the constant discount rate[12], given the short period considered it would not potentially differ significantly. Nevertheless, the unexpected high benefit cost ratios obtained, reveal the importance of developing the measures appropriately before applying the economic analysis. In other words, there is a clear risk in evaluating measures that do not mitigate pressures but return high benefit cost ratios in the assessment. It should be highlighted therefore, that the methodology does not assess whether or not adopted measures are able to ensure water status improvements but compares measures that have been designed to address pressures and deliver status improvements in terms of benefit-cost ratios. Furthermore, the application in the Broadland Rivers catchment demonstrated how the methodology can also accommodate stakeholders' participation in the assessment of environmental policies and through them, reveal the impacts of improvements of the status of natural resources.

[12] Such evidence is provided by Birol, Koundouri and Kountouris (2010) who estimated benefit and cost values for a 200-year horizon with varying discount rates.

5.7. Conclusions

The methodology I propose provides a holistic way for water managers to assess the effectiveness of PoMs by utilizing the opinions of stakeholders on the connection between pressures and local ecosystem services, and through that, select measures that do not yield disproportionally high costs in relation to benefits from improving water status. The proposed assessment framework could benefit water management practitioners to frame environmental problems more accurately and assess the effects of their practices in a more systemic manner. It can be used either in the initial process of selecting cost-effective measures to provide insight into their socioeconomic impact, or after the implementation of measures to validate whether they have been economically beneficial or not. Additionally, it could be used after the conclusion of a management cycle, to assess whether implemented actions have been effective or not.

Finally, the study included in this chapter presents a possible way to integrate different kinds of knowledge (e.g., biology, sociology, economics, ecology, etc.) into a common framework. Economists or ecologists would most likely fail to understand the mechanics of the suggested methodology as well as obtain sound results if they only focused on the assumptions, methods, and research practices of their own discipline. Therefore, the implementation of the suggested methodology requires both collaboration among experts of various fields as well as understanding of how different disciplines distinguish themselves.

6. Natural Capital Accounting informing water management policies in Europe

6.1. Introduction

The natural environment is consistently undervalued in decision-making. However, besides the inherent value of natural resources to human wellbeing, a wide range of government policies including investments in infrastructure and economic growth are influenced by the value of natural resources and their availability (Ruijs et al., 2019a). Indeed, it is now increasingly recognized that environmental degradation diminishes the capacity of the planet to sustain economic development (Jouanjean, Tucker and Willem, 2014; Lu et al., 2017; Hasan et al., 2020).

The presence of human pressures on water resources coupled with ineffective and unstainable management practices deeply affect the ability of ecosystems to deliver services. Ecosystem services are the source of benefits which people gain from natural ecosystems, and natural capital is the stock of natural ecosystems from which these benefits flow (Costanza and Daly, 1992). Reduction in the delivery or loss of ecosystem services results in economic losses, which, given the current monitoring schemes in Europe, are hardly considered by national economic policies. However, maintaining natural capital, i.e., ecosystems and their services, is fundamental to human welfare and development. Given the pressures and threats on European ecosystems, Europe risks losing natural capital without valuing what is being lost (European Environment Agency, 2019). Methods of monitoring and assessing the importance of such services to a society and its economy have increasingly gained the interest of governments in the last two decades (Balvanera et al., 2017), making the case for environmental protection.

The publication of the Millennium Ecosystem Assessment (MEA, 2005) ignited a broad discussion on the interaction of humans and the environment and influenced the development of assessment methods (Barbier, 2007; Fisher and Turner, 2008; Koundouri et al., 2015; Liu et al., 2010; NRC, 2005; Turner et al., 2010; Wallace, 2007), providing the conditions for the development of approaches of natural capital accounting and assessments, a promising avenue for improving the status of ecosystems, while supporting policymaking. Natural capital and ecosystem services are both definitions included in the ecosystem approach (Robinson, Hockley and Reynolds, 2016). Natural capital, a term introduced by Pearce et al. (1989), comprises the ecosystem and abiotic assets of the earth that provide ecosystem services such as food, climate regulation, and recreation (European Environment Agency, 2018b), or, as Costanza et al. (1997) put it, natural capital can be described as "the stock of materials or information" contained within an ecosystem. Natural capital as a stock provides flows of materials, energy, and information in the form of ecosystem services that, when combined with other forms of capital (social, human and or built capital), contribute to human welfare (Costanza, 2020). In other words, ecosystem services are the results of the interaction of biotic and abiotic components of natural capital (Smith et al., 2017) (Figure 6.1).

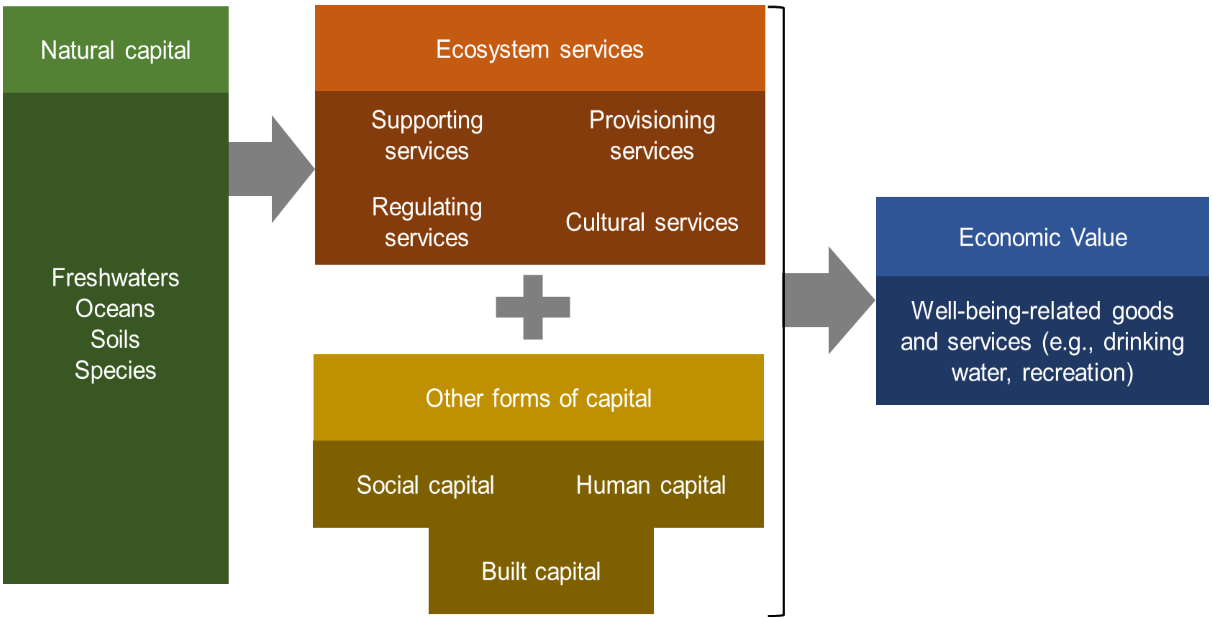

Figure 6.1 Natural capital assets, ecosystem services, and well-being

Policymaking is fundamentally concerned with choosing among various options (or combinations of different types of capital) to obtain the most valuable outcome. Consequently, valuation is an integrated part of designing and implementing policies. The study presented in this chapter considers that the value of natural capital assets acts as an integrated indicator of the condition of the overall system (social and environmental). Increases in their value may also indicate an enhancement of the condition of natural capital or an increase in the marginal value of benefits provided to humans by the environment. For the purpose of monitoring the contribution of nature to welfare, natural capital accounting potentially has multifaceted roles in policymaking. Estimating the quantity and assessing the quality of natural capital assets systematically, as well as the benefits they provide to the economy and society, reveals how the use of resources influences economic development, thus providing opportunities for increasing the efficient use of natural resources, as well as for their protection (Badura et al., 2017; Bateman and Mace, 2020). The identification of pressures and possible risks provides the basis for an evaluation of the effectiveness of policy instruments and fosters the adoption of practices that promote sustainability (Mace et al., 2015). Furthermore, assessing how natural capital is affected by different industries of the economy has the potential to minimize emerging risks faced by businesses (Capitals Coalition, 2020). This is particularly important, as environmental issues take top spots in the World Economic Forum's Global Risks Report (The World Economic Forum, 2021).

In this respect, a major advancement has been the development of Environmental-Economic Accounting (United Nations- Statistics Division, 2013), led by the UN Statistical Commission with the involvement of international organizations such as the European Commission, the World Banks, and hundreds of scientists and nongovernmental organizations. Since then, 24 countries, some of them in Europe (e.g., the United Kingdom, the Netherlands, Norway, Italy, Spain, Australia, and Canada) have compiled such accounts (Hein, Bagstad, et al., 2020). The accounting of natural capital aims at establishing consistent approaches of identifying, assessing, and monitoring the flow of goods and services and, consequently, the benefits generated by nature (DeWitt et al., 2020). Overall, the natural capital principles and methodologies provide several important tools for managing authorities (Russell et al., 2020). The use of a commonly accepted classification of ecosystem services and the identification, as well as recognition, of benefited stakeholders, helps to organize information and frame each given management problem in a concise way. Following a standardized methodology to assess the value of ecosystem services and natural stocks also assists in keeping track of changes that occur over time.

In the European Union, the concept of natural capital accounting has been recognized by the EU Biodiversity Strategy to 2020 (European Commission, 2011) and the Seventh Environment Action Programme of the EU (European Union, 2013), which highlights the importance of developing standardized natural capital accounting practices as a means to protect and enhance natural capital (La Notte et al., 2017; European Commission, 2020a). Additionally, the Eighth Environment Action Programme of the EU (European Commission, 2020a), which is to be adopted in 2021, prioritizes among other things the development and application of ecosystem-based management practices, including natural capital accounting and nature-based solutions (European Commission, 2020a). Mainstreaming natural capital accounting for the implementation of environmental policy has therefore been an issue of increasing interest in the European Union, as it informs policymaking and fosters the implementation of nature-based solutions, which have the potential to provide higher socioeconomic benefits at lower costs compared to traditional approaches (European Commission, 2019b). However, the concept is in its infancy; therefore, when the European Environment Agency implemented pilot projects (Capriolo et al., 2020), such as a project in the Warnow Basin in Germany, where different accounting applications were performed using WFD reporting data, it was concluded that the data sets that were contemporaneously available in Europe did not match the requirements of ecosystem accounting (European Environment Agency, 2018b).

The WFD has been the main driver for the collection of data since its adoption in 2000. According to its provisions, Member States are required to develop River Basin Management Plans, which include an abundance of information (Carvalho et al., 2019) ranging from biological to socioeconomic data at the catchment level, aiming to assess the pressures on, and status of, inland waters, and to develop programmes of measures to improve the overall health of such ecosystems (Santos et al., 2021). The lack of common definitions and objectives (Moss, 2008; Josefsson and Baaner, 2011) as well as the knowledge deficit of Member States in applying integrated methodologies has resulted in overall underperformance of the Directive (Berbel and Expósito, 2018; Zingraff-Hamed et al., 2020). In other words, the implemented programmes of measures did not provide the desired results, leading to a questioning of the effectiveness of the Directive (Moss et al., 2020). This, as discussed in Chapter 5, gave rise to the exploration of how approaches based on ecosystem services can be applied to foster a higher degree of integration between pressures, impacts, and programmes of measures to improve water status overall.

Acknowledging the importance of the WFD and developments in natural capital accounting, the aim of the chapter is to explore its potential to inform the selection of programmes of measures and to provide a concise way of assessing how implemented measures impact the use and value of natural resources through changes in their overall water status. After a brief discussion on the connection between the WFD and natural capital, possible steps are presented that could be followed to assess how policy interventions affect the value of natural capital, both through a theoretical and a practical approach. Finally, natural capital accounting is applied for the estimation of the asset value of two of the ecosystem services provided by water bodies in two case studies in Europe that have not yet used the information of such accounts for the development of River Basin Management Plans and the assessment of programmes of measures. In the current chapter, I take a step back from ways of assessing pre-selected programmes of measures (Chapter 5) to identifying areas of possible intervention, which are denoted by fluctuations in the natural capital accounting tables. A common element in both chapters is the estimation of the value of ecosystem services either as flows (Chapter 5) of benefits or stocks (Chapter 6). Therefore, the goal of this chapter is to test how the natural capital approach can complement the implementation of the WFD to manage water resources as well as contribute to the relevant empirical literature.

6.2. Natural capital and the Water Framework Directive

The adoption of the WFD has been a decisive turn in water management in Europe (Zingraff-Hamed et al., 2020). Acting as an overarching legal document, the Directive aimed at embracing all fragmented pieces of water law in Europe, with the ultimate goals of preventing the deterioration of the quality of waters and achieving good water status by managing water resources effectively (Bone et al., 2011). More importantly, the WFD introduced a new paradigm in water management by promoting integrated river basin management and stakeholders' participation, focusing on enhancing the overall health of the system instead of just the chemical status of water, and by including economic principles and tools as key features of its implementation. Furthermore, the Directive took up a systemic approach by introducing river basins as the main governance unit, therefore recognizing that each river basin constitutes an interconnected system (Giakoumis and Voulvoulis, 2018b, 2018c). Instead of managing specific elements in isolation from the broader system they are traced, the WFD took a decisive step away from the command-and-control practices introduced by traditional water management policies (Giakoumis and Voulvoulis, 2019). Additionally, compared to previous environmental Directives, the WFD set a specific date for achieving its objectives and requires the introduction of specific policy interventions, considering their cost-effectiveness (Kochskämper and Newig, 2021). Finally, the WFD requires the interventions designed and implemented by the Member States in each river basin, as well as detailed information on the status of water resources, the types of pressures, water uses socioeconomic characteristics, etc., to be included in the River Basin Management Plans (RBMP), which should be updated in fixed intervals (management cycles). Though promising, the implementation of the Directive has faced significant obstacles, leading to growing concern that many EU Member States will be far from achieving the objective of good status by 2027 (Carvalho et al., 2019). According to the WFD fitness check (European Commission, 2019b) published in 2019, there had not been substantial improvement in the status of water in the first two cycles. Potentially, this is due to delays in the implementation of the Directive, the high number of deadline extensions that were granted to 40% of all surface water bodies and 11% of groundwater bodies (Boeuf and Fritsch, 2016), as well as misunderstandings about the definition of ecological status (Voulvoulis, Arpon and Giakoumis, 2017).

Deepening the understanding of the relationship between the environment and society, two components of the same system, through the identification and assessment of the value of nature to humans, has been considered to improve catchment management (Everard et al., 2009). Ecosystem services, the benefits that the environment provides to humans, have been suggested as a possible tool to shed light on the interaction between the two components of the socio-environmental system and to promote the protection and restoration of ecosystems (Guerry et al., 2015). As far as this paradigm is concerned, ecosystem services are the nexus between the condition of ecosystems and well-being, as the status affects the delivery of ecosystem services (Maes et al., 2018). Several authors have used ecosystem services to demonstrate their suitability for the implementation of the WFD, for the purposes of economic analysis, design and implementation of programmes of measures, assessment of pressures, and stakeholders' participation (Vlachopoulou et al., 2014; Borrego-Marín, Gutiérrez-Martín and Berbel, 2015; Koundouri et al., 2015; Grizzetti et al., 2016; Grizzetti, Liquete, et al., 2016; Pistocchi et al., 2017; Giakoumis and Voulvoulis, 2018a; Pacetti et al., 2020; Souliotis and Voulvoulis, 2021a).

The WFD does not refer explicitly to natural capital. Its purpose is to protect waters (inland surface waters, transitional waters, coastal waters, and groundwater), enhance their status, and promote their sustainable use through implementing PoMs that eliminate pressures and recover costs of water services (Borrego-Marín, Gutiérrez-Martín and Berbel, 2015). However, if it is not technically feasible for a Member State to achieve a good status within the set timeframe, or if natural conditions do not allow for the achieving of a good status, or if costs are disproportionate to the benefits of improving water statues, an extension of the deadline for reaching good ecological status or setting lower targets may be allowed (Macháč, Brabec and Vojáček, 2020). The disproportionality principles apply when the financial ability of Member States is such that does not allow for the implementation of programmes of measure, or when the undertaking costs of implementing measures are significantly higher than the benefits of improving water status (Martin-Ortega, 2012; Martin-Ortega et al., 2014). In economics, however, disproportionate cost is not a standard concept (Brouwer, 2008), and there are no standards on which a benefit–cost ratio should be considered prohibitive. Moreover, the WATECO Guidance Document (European Commission, 2003d) suggests the use of economic tools to assess the disproportionality of costs, however, it states that decisions on the need for derogation remain political. From an economic perspective, Cost–Benefit Analysis (CBA) is the obvious tool used to assess the disproportionality of costs. CBA considers the welfare value of benefits accruing from a change in the circumstances and compares it to the cost of policy options. On the contrary, natural capital accounting considers the exchange prices of ecosystem services based on current pricing mechanisms and market conditions (Hein, Bagstad, *et al.*, 2020). In cases where exchange prices cannot be obtained, it might be feasible to use welfare values, assumed as exchange values (Obst, Hein and Edens, 2016).

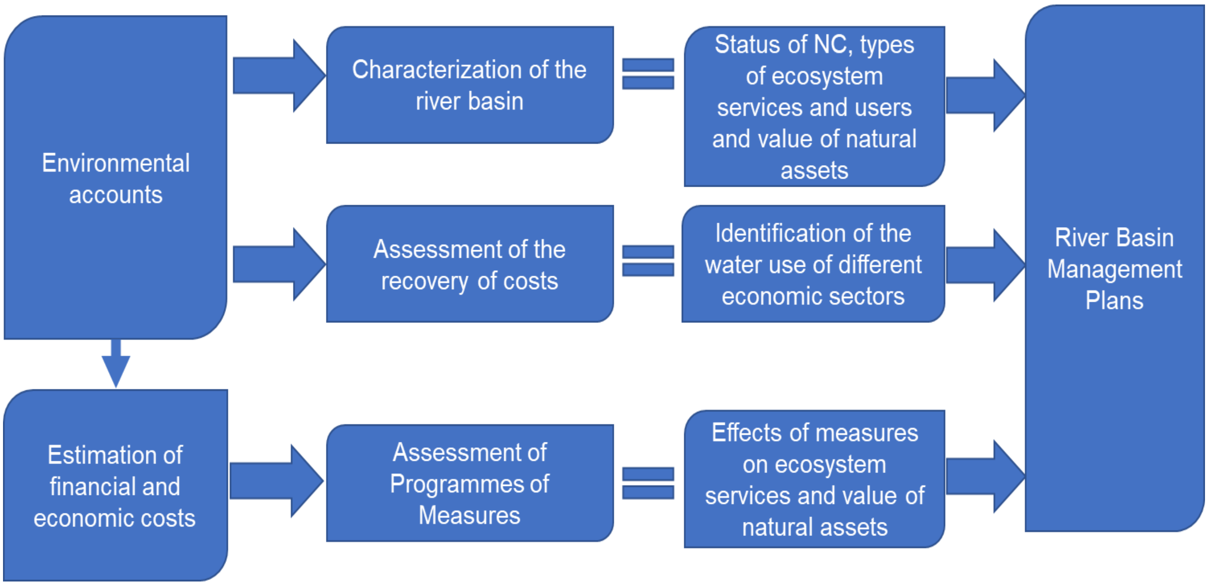

Figure 6.2 Use of natural capital accounts information for the implementation of the WFD

Natural capital accounting provides information on the condition of the ecosystem, the physical and monetary flow of ecosystem services, and the monetary value of ecosystem assets; therefore, it constitutes a tool to measure the changes in the stock of natural capital. The process of designing programmes of measures and consequently assessing their cost-effectiveness and proportionality can be informed by natural capital accounts in the following ways (Figure 6.2):

1. By identifying the users and uses of water resources within each catchment area that will be impacted the most by policy interventions;

2. By assessing the trade-offs between different ecosystem uses;

3. By establishing a common currency to allow for a comparison of changes within and between each asset of each ecosystem;

4. By incorporating information from a natural capital assessment into a CBA or other appraisal techniques.

6.3. Assessing the value of natural assets in line with the WFD

Assessing the effectiveness of programmes of measures has been a troublesome experience for most of the EU Member States. However, from the First Implementation Report published in 2007 to the Fifth published in 2019, Member States have made significant progress concerning the development, assessment, and implementation of PoMs, although significant gaps still remain in translating the results of the economic analysis into concrete measures (European Commision, 2019). Meeting the targets of the WFD requires an increased investment in technical and non-technical measures, which will require sophisticated economic justification to facilitate water-related decisions. Estimating the stock value of the flow of services according to natural capital principles can be aligned with the required economic underpinning to better serve the needs of the Water Framework Directive. Taking that into account, this section describes the steps (Figure 6.3) that need to be taken to obtain information on the status and contribution of the ecosystem and how this can be fed into assessments of policies.

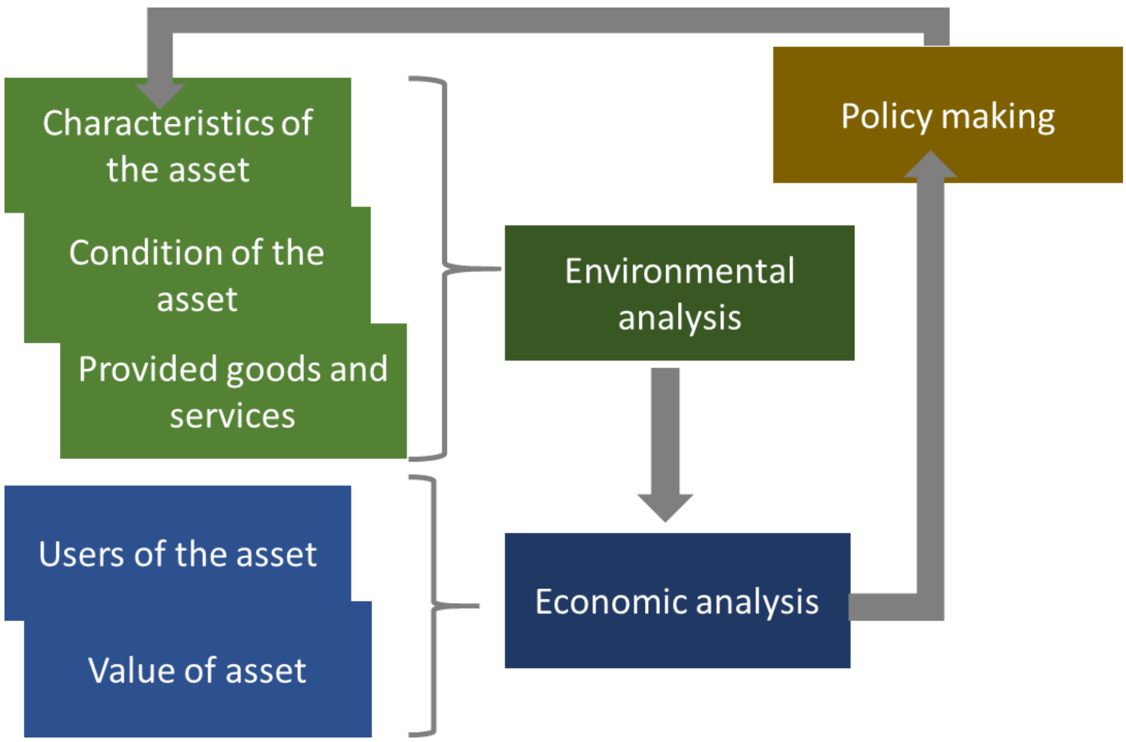

Figure 6.3 Steps for assessing the value of natural assets

Step 1: Characteristics of the water body.

The initial step is to understand the components of the broader system encompassing the natural resource of interest which may influence policy outcomes. Therefore, general characteristics of the natural resource (asset) and the wider system help in constructing a baseline that considers land cover the classes or the type of the ecosystem and their extent. Such data may be spatial information, land use data, climate information, as well as information on the socioeconomic characteristics of the wider area that can provide an indication of current and future pressures. For this purpose, classifications of ecosystem assets may be used, such as the UK Broad habitat types (Natural Capital Committee, 2014). Besides that, Article 5, and Annex VII of the WFD require policymakers to undertake an analysis of the characteristics of each River Basin District, a review of the impact of human activity on the status of water and an analysis of its uses for the drafting of the River Basin Management Plans.

Step 2: Condition of the asset.

Assessing the condition of the asset requires to consider the physical, chemical, and biological aspects of the resource. In Europe, the WFD provides detailed consideration to the meaning of good ecological status. More specifically, Annex V (European Commission, 2000) set out a list of biological, hydromoprhological and physicochemical quality elements (Everard, 2012). In addition to that, understanding the relationship between the characteristics of the aquatic ecosystem and human pressures helps to design targeted measures to improve the ecological status of water systems (Grizzetti et al., 2016).

Step 3: Types of goods and services the asset provides to water users.

Aquatic ecosystems provide a wide range of critical ecosystem services that can be categorized into provisioning (e.g., water provisioning and fish production), regulating, supporting and cultural services (e.g., recreation) (Costanza et al., 1997; De Groot, Wilson and Boumans, 2002; Millennium Ecosystem Assessment, 2005; Daily, 2013b). Identifying the specific ecosystem services provided by the natural capital, their flow and the users that benefit from these services is an essential part for obtaining a preliminary indication of the importance of natural capital. Additionally, it helps determine the direct and indirect benefits to users, option value and the related non-use values (existence and bequest) (Pearce et al., 2006; United Nations Statistics Division, 2012).

Step 4: Value of the provided goods and services.

Economic value for a stream of services relates to the contribution of ecosystem services to human welfare and broadly speaking is measured based on each individual's own preference and assessment of their wellbeing (Freeman III, 2010). Costanza (2020) proposes three different paradigms for assessing the value of natural capital. The first relates to "homo economicus", where value is obtained through individuals' stated or revealed willingness to pay; the second relates to "homo communicus", where the community rather than the individual (Wilson and Howarth, 2002) define the value of natural capital; lastly the third is associated with "homo naturalis". According to it, individuals are integrated components of the system, therefore value encompasses social, biophysical, and economic dimensions of the ecosystem services (Fontaine et al., 2013; Fontaine et al., 2014). Nevertheless, in the natural capital context, emphasis is given to the value of past, current, and future flow of benefits of ecosystem services. The flow of benefits is discounted to present values to estimate the total benefit of an environmental asset (Dickie and Neupauer, 2019). At the EU level, the methods used for valuing ecosystem services depend on the goal served by each particular account (Badura et al., 2017). Furthermore, the SEEA EEA classifies the valuation methods into three broad categories: market-based or cost-based methods (e.g., unit resource rent, production function, replacement cost, defensive expenditure, averting behaviour), revealed preference methods (e.g., hedonic pricing and marginal values from travel cost demand functions) and stated preference methods (e.g., contingent valuation and choice experiment).

As policymakers need to evaluate the effects of policy changes, the value of flows of benefits could play an important role in the assessment of management options. An advantage of the natural capital accounts is that they include not only the economic value of ecosystem services, but also physical data on the natural capital stock. This is particularly important when policy makers from various organizations, need to implement integrated methodologies, such as that proposed by the WFD. The prime focus of natural capital accounts is to reveal the ecosystems' contribution to the economy (Hein et al., 2020). Additionally, natural capital accounts can be used either for backward-looking or forward-looking assessments. For instance, assessment and monitoring of environmental-economic macro-indicators, reviews on implemented projects concerning expenditures and benefits and sustainable development monitoring, national development plans, and land use strategic planning to name a few (Ruijs et al., 2019a). Vardon et al. (2016) explain that information dwelling from natural capital accounts can inform decision makers at any stage in the policy cycle (agenda setting, policy implementation and evaluation and measuring success). Natural capital accounting can be used in parallel to other economic methods, such as Cost–Benefit Analysis (CBA) suggested by WFD supporting documents (e.g., WATECO 2003). While CBA considers the flows of services and their benefits, natural capital accounting considers the stocks of natural resources, and thus incorporates sustainability considerations that cannot be captured by CBA (Bateman and Mace, 2020).

6.4. Materials and methods

6.4.1. Description of the case studies

Two case studies in Europe, one in Greece and one in the UK were selected for applying the natural capital approach. Both areas are operational catchments within a River Basin District and were studied by the GLOBAQUA (Grant agreement no. 603629-ENV-2013-6.2.1-Globaqua) project (Navarro-Ortega et al., 2015). To my knowledge, natural capital accounts were not used in the development of River Basin Management Plants in either of the two countries, indicating that if not at all, only to a limited extent the stock value of ecosystem services affected policy decisions. The UK has undertaken a national ecosystem assessment (UK NEA, 2011) and since 2013 the Office of National Statistics has been publishing annual environmental and ecosystem accounts (Maes et al., 2020). Data on natural capital accounts are available through the Department for Environment, Food and Rural Affairs (Defra) and the UK Environment Agency and the Office for National Statistics and can be easily accessed by the public. On the contrary, Greece has not yet compiled and published natural capital accounts. The main portal for all environmental information is that of the Ministry of Environment and Energy, however, datasets on EU environmental legislation are not available (European Commission, 2019d). Data on aspects of the WFD can only be extracted from the River Basin Management Plans that have already been submitted to the European Commission and no background documents are accessible. Therefore, the selection of these case studies helps to explore the difficulty in using ecosystem-based approaches in more and less methodologically advanced countries. The aim of this chapter is to promote the development of accounts at minimum for some ecosystem services based on information that is already available from WFD reporting. The section starts with a general description of the areas, including the status of water resources, present pressures, and socioeconomic characteristics.

The first, the Evrotas river basin (RB) is located in the Eastern Peloponnese River Basin District in Greece in the Prefectures of Laconia and Arcadia (Figure 6.4). The catchment area occupies the biggest share of the basin, with a length of 93 km and a total catchment area of 2,410 km^2 (Marinou et al., 2017). The main tributaries are the Oinountes, Magoulitsa, Gerakaris, Kakaris, Rasina, Mariorema, and Xerias (Querner et al., 2016). Overall, there are 44 river systems in the Evrotas RB. The climate of the area is Mediterranean with high levels of precipitation, however, the low ratio of mean annual precipitation to potential evaporation characterizes the area as semiarid (Karaouzas et al., 2018). Furthermore, in the last 35 years decreasing trends in rainfall and discharge have been observed (Skoulikidis et al., 2011).

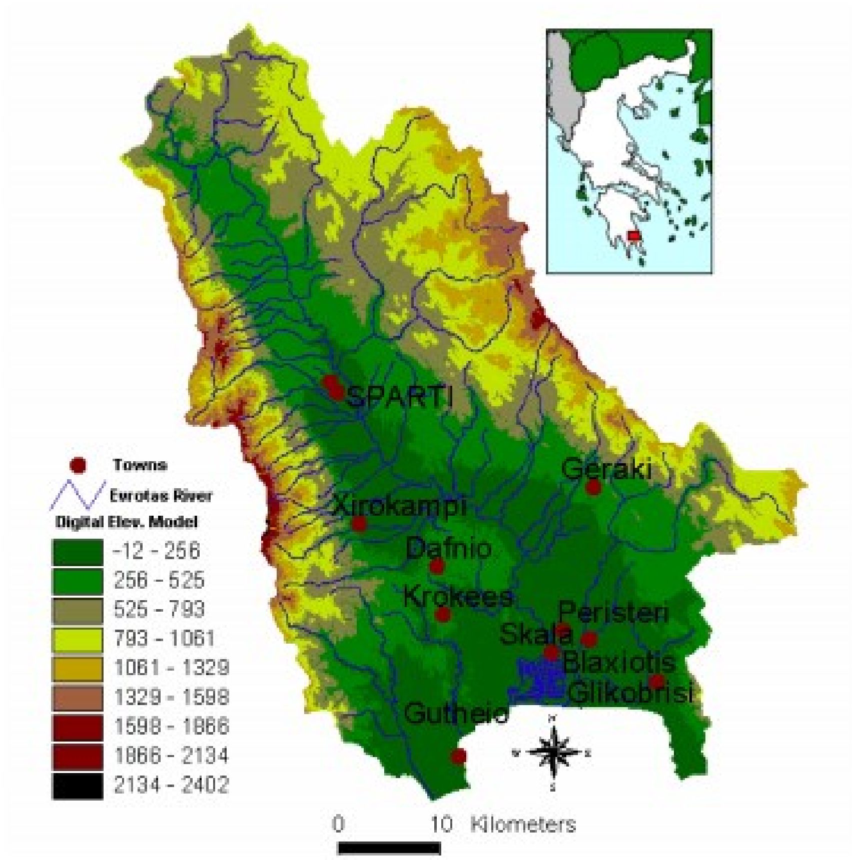

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Figure 6.4 The Evrotas river basin (Skoulikidis et al., 2011)

According to the River Basin Management Plan of Eastern Peloponnese, the main pressures in the Evrotas catchment are related to water quantity, water abstraction for irrigation and droughts. Additionally, pressures on the quality of the water relate to agricultural activities (e.g., use of pesticides), aquaculture/fish farming, urban waste, septic tanks, and mining. Humans intervene in the area by removing natural vegetation, constructing embankments and by removing riverbed material, leading to morphological pressures.

The second, the Broadland Rivers catchment covers an area of 3200 km^2 and it is mostly rural. The catchment includes 94 river water bodies with the four main (sub-catchments) being the Bure, Wensum, Yare and Waveney and 19 lake water bodies (Environment Agency, 2014) (Figure 6.5).

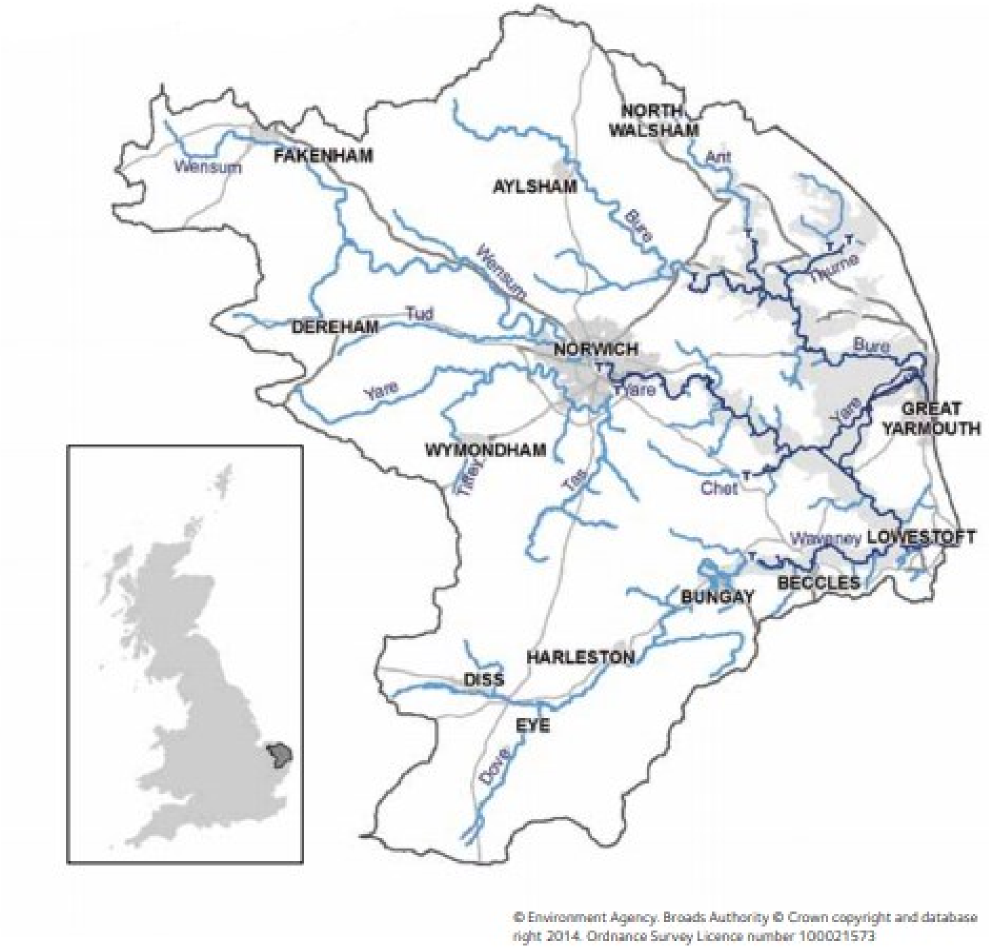

Figure 6.5 The Broadland Rivers catchment. Source: Environment Agency (Environment Agency, 2014)

The largest settlements within the catchment area are the city of Norwich and the seaside towns of Great Yarmouth and Lowestoft. Additionally, the catchment encloses the Broads Executive area, which has the management status of a national park(Environment Agency, 2014). The vast majority of the area (approximately 87%) is used for agricultural purposes (non-irrigated arable land and pastures). Urban areas (including parks, industrial, commercial, transport units, mines, dump, and construction sites) account for 7.56% of the total area, while the remaining 3.92% is covered by forest and other nature units (Giakoumis and Voulvoulis, 2019).

As far as the water status in the case studies is concerned (Figure 6.6), 70% of the rivers in the Evrotas catchment achieved good status in 2017 compared to 32% in 2013 (Ministry of Environment and Energy of Greece, 2016). This could be attributed to the implemented PoMs the majority of which were in the implementation phase at the time when the RBMP was published. In line with this, Apostolaki et al. (2019) claimed that the regulatory measures were in place, but stressed the importance of progressing with the implementation of technical measures, which might explain why the status of one of the water bodies is indicated as bad. On the contrary, in the Broadland Rivers catchment, the status of water bodies has progressively deteriorated. More specifically, while two rivers were at good status in 2015, none of them maintained the same status in 2019, where the majority of rivers were classified as moderate and the remaining as poor (EA, 2018). Giakoumis and Voulvoulis (2019) claim that this happened because the programme of measures developed by the management authorities focused on managing specific quality elements, thus neglecting the connection between the pressures and the overall health of the system.

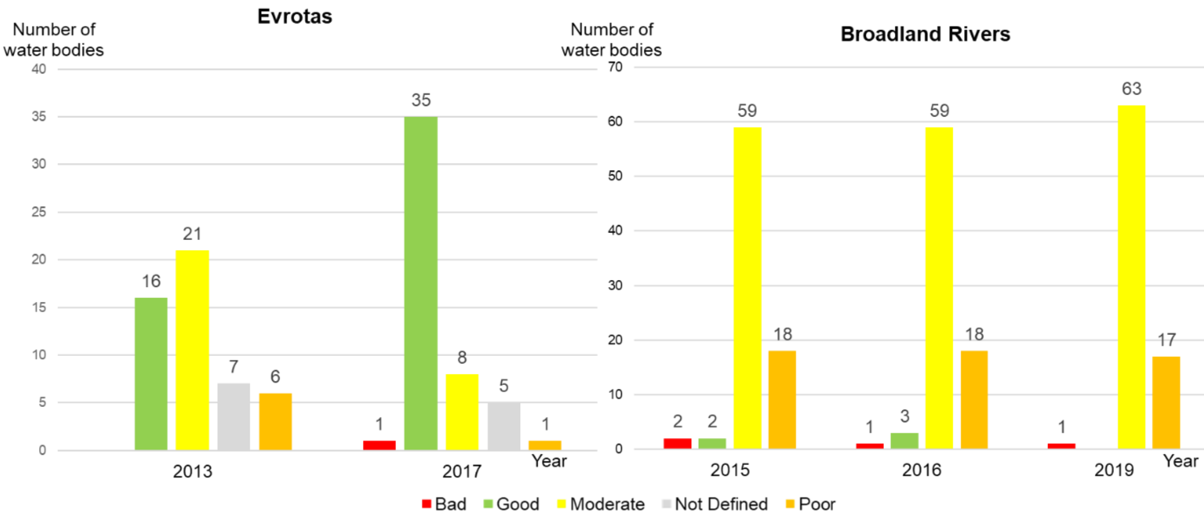

Figure 6.6 Water Status of Evrotas catchment in 2014 and 2017 and Broadland Rivers catchment between 2015 and 2019 (Information was obtained from the River Basin Management Plans and background documents, where available. The status of river bodies in the Evrotas catchment in 2017 consists of projections as reported to the European Commission)

In relation to pressures in the two areas, the most dominant in the Evrotas catchment relate to agricultural activities and concern mainly overexploitation of water resources for irrigation (GLOBAQUA, 2018a). Overextraction results in partial desiccation in late summer-early autumn (Díaz-Cruz et al., 2019). The catchment also faces diffuse agrochemical pollution and pollution from the wastewater treatment plant of Sparta (Lampou et al., 2016). Similarly, agricultural pressures exist in Broadland Rivers. Giakoumis and Voulvoulis (Giakoumis and

Voulvoulis, 2018a) developed a framework that allows for the ecosystem approach to be operationalized for assessing and ranking pressures based on stakeholders' perceptions. They concluded that the most prominent pressures in the Broadland Rivers catchment are intensive nutrient or pesticide use, activities related to agricultural enhancement, pollution from urban areas and abstractions for potable supply.

6.4.2. Collection of data

For the estimation of the asset value of ecosystem services, I used a variety of data sources. Where possible, I gathered data directly from the River Basin Management Plans. In cases where additional information was needed, I utilized other official national data sources. Overall, information on abstractions and water uses is included, as well as data on visits to the catchment area for recreational purposes. Additionally, using data from the Statistics Offices of each respective country, I adjusted the obtained value to account for factors such as inflation, income etc. To overcome the issue of missing data and in order to present comparable values, I also used proxy variables that are described below. For example, for the estimation of the value of recreation, the number of overnight visits and the number of trips were considered in the Greek and the UK case, respectively. The lack of adequate information resulted in a limited assessment of the benefits of the flow of ecosystem services. Overall, data processed for the different types of analysis is believed to be reliable as taken directly from official bases. Nevertheless, the following sections include a detailed description of the data sources.

6.5. Estimation of the value of natural capital

In the subsequent sections, the asset value of water for residential purposes and the value of recreation of Evrotas and Broadland Rivers basin are estimated.

6.5.1. Water for residential use—Evrotas

Regional economic activity includes mainly agricultural, livestock and industrial activities. Agriculture is the main user of water. Around 3,500 public and private wells are estimated to be used to cover the needs of the sector (Huber García et al., 2018), however, there is a great number of illegal surface abstractions of surface water (Skoulikidis et al., 2011). For this reason, only the residential water supply was considered. Water pricing in the river basin area is differentiated into eight categories based on the type of user (public supply, agriculture, industry etc.).

In terms of the local population, the Evrotas catchment area hosts approximately 70,000 permanent residents (Ministry of Environment and Energy of Greece, 2016). Data concerning the population in the catchment area was obtained from the first River Basin Management Plan (Ministry of Environment and Energy of Greece, 2013). The document includes data for 2011 and projections for 2015 and 2021. Based on that the population for the missing years was estimated. The estimation of water use per year was based on assumptions included in the River Basin Management Plan[13]. More specifically, in line with the River Basin Management Plan, I assumed that each person consumes 250 litres per day. Based on that and the unit value of output taken directly from the River Basin Management Plan the total value of output was calculated. Additionally, by considering the amount of water abstracted by both the water and sewage companies and the municipal utilities operating in the area and the capital and operating costs of these suppliers, I estimated the unit cost per m^3 of supplied water and consequently the total cost of supplied water. By subtracting the cost from the total value of output, resulted in the value of water for residential use for each consecutive year (Table 6.1).

Table 6.1 Value of water for residential use (£ million, 2019 prices)

Year	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**
Flow (Mm^3)	6.3	6.3	6.4	6.5	6.5	6.6	6.7
Value	2.6	2.8	2.7	2.6	2.5	2.5	2.6

6.5.2. Water for residential use- Broadland Rivers

Estimating the annual value of water for residential use was based on information on the number of licenses and the maximum permitted volume per license derived from an Environment Agency database[14]. Using GIS data provided by the Environment Agency

[13] The River Basin Management Plan reports a unit value of output of 0,68 €/ m^3. By taking into consideration costs related to compensation of employees and depreciation of capital, the unit cost of abstracting water was estimated to be approximately 0.218 €/ m^3. Taxes and subsidies on water extraction were not included in the valuation as they are not relevant for the water supply in Greece.

[14] Water_Abstractions_20150101.mdb

(Environment Agency, 2016), I constructed a layer to obtain the licenses that are relevant for the Broadland Rivers catchment. As in the case of UK natural capital accounts (ONS, 2021), only water abstracted for public water supply was included in the estimation. Since the quantities associated with the water abstractions in Broadland Rivers considered the maximum abstracted quantity allowed, I calculated the volume of used water for each given year by computing the volume of used water as a percentage of the maximum abstracted quantity allowed included in the national water accounts for the England region (Table 6.2). Since input-output tables publicly available by the Office for National Statistics were referring to the national level, I used the annual values provided for England, which I then adjusted in order to calculate the value of flows of this service for the Broadland Rivers Catchment.

Table 6.2 Value of water for residential use (£ million, 2019 prices)

Year	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**
Annual flow (Mm^3)	71.10	71.41	72.30	76.27	74.97	77.83	148.67
Annual value	26.13	30.84	30.19	25.46	18.92	41.73	57.39

6.5.3. Recreation-Evrotas

The estimation of the annual value of recreation in the case of Evrotas was a troublesome task due to a lack of data. The revised RBMP includes information on the number of overnight stays from 2005 to 2009 and the estimated number of overnight stays for 2015 and 2021. Based on the annual percentage change of this variable, I estimated the missing values for 2011-2014 and 2017. Additionally, the number of trips was calculated based on the average duration of stay in days (7.35 overnight stays per trip) obtained from the Institute of the Association of Greek Tourist Enterprises (INSETE, 2019). Finally, I used the value transfer approach (Boutwell and Westra, 2013) to estimate the value per trip following a previous relevant study. More specifically, Latinopoulos (2014) applied the travel cost method to assess the demand for outdoor recreational services for protected riparian ecosystems in Northern Greece and it was considered the most relevant to the catchment of Evrotas. By adjusting for inflation, the results of that study show an average consumer surplus value of 217.7 (€, 2019 prices) per trip was obtained. Multiplying this value by the estimated number of trips yielded the total annual value of recreation (Table 6.3).

Table 6.3 Estimated value of recreation (£ million, 2019 prices)

Year	2011	2012	2013	2014	2015	2016	2017
Overnight stays	184,100	188,850	193,723	198,721	203,100	203,367	203,634
Estimated number of trips	25,048	25,694	26,357	27,037	27,633	27,669	27,705
Annual value	4.79	4.91	5.04	5.17	5.29	5.29	5.3

6.5.4. Recreation-Broadland Rivers

The value of recreational benefits related to the Broadland Rivers basin was estimated using the Travel Cost Method (TCM). The method which was suggested by Hotelling (1949) and fully developed by Clawson (Clawson, 1959), suggests that the recreational benefits at a specific site can be estimated based on a demand function that accounts for the number of trips/ number of visitors to the actual costs of a given visit (Whitehead, Haab and Huang, 2000; Navrud and Ready, 2002). Generally, the recreational demand function of a specific site can be defined as a function that relates the number of visits for a specific period of time (e.g., number of trips per year) to the travel expenses (e.g., bus fares, entrance fees, cost for food etc.); the socioeconomic characteristics of the visitors (e.g., age, level of income etc.); the characteristics of the site (e.g., water quality); and other factors that relate to the trip (e.g., type of activities that individuals engage in). Such a function reads as (Latinopoulos, 2014):

$$V_i = f_{(c_i, cv_i, sc \quad , q_i)},$$

where V_i is the number of trips that individual i takes to the site within a year; c_i is the cost related to the visit faced by individual i; cv_i the characteristics of the visitor; sc is a vector of characteristics that influence the choice of the site; and q_i a vector of other relevant factors.

Traditionally, on-site surveys are implemented followed by econometric analysis. For the purpose of the study, the main data source for the analysis was the Monitor of Engagement with Natural Environment (MENE) Survey (Natural England, 2019) dataset. MENE datasets include a wide range of information related to the visit location, travel and visit time, costs related to visits as well as the socioeconomic characteristics of the visitors and are considered an appropriate source for ecosystem accounts (Eftec, 2015). The study considers data for 2011 to 2019 for Suffolk and Norfolk counties, as the study area lies within these two locations. Furthermore, for the purposes of the study, I considered solely responses related to visits to rivers, lakes and canals (the datasets include visits to other locations such as mountains, woodland, beaches, and parks among others).

Given that the working status of individuals included people in the labor market, unemployed individuals, pensioners, and students, in order to estimate the opportunity cost of each visit, I considered the hourly paid wage of each year (Annual Survey of Hours and Earnings datasets for 2009 to 2019), which I then adjusted to 2019 prices using the current prices index (CPI). Concerning the cost of travelling, the respondents mentioned several modes of transportation (e.g., bike, car, train, bus, foot etc.). Besides those that travelled to the location by bicycle or on foot, the cost of travelling was estimated as follows. For those that travelled by car, I considered both the travel distance as well as the average cost per mile. The latter was taken from RAC reports (RAC, 2011, 2012, 2013) for 2011 to 2013. For the remaining years, I adjusted the values for inflation. Self-reported expenditure on bus and train fares was extracted from the MEME database. In cases where no cost was reported, I estimated the average cost per mile by the responses of other individuals for each given year. As far as the cost of travelling by taxi is concerned, data on tariffs was taken from annual Taxi Fares and Tariff consultation reports (Transport for London, 2014, 2015, 2017). Expenditures for visitors travelling on foot and by bicycle were considered negligible. Concerning the cost of travelling time, the average speed of a car in England (DfT, 2016), as well as the average speed of train (Railway Performance Society, 2021), were considered. Based on that as well as the self-reported distance from the starting point of each respondent's trip, I estimated the travelling time for different modes of transportation. Additionally, the time was multiplied by 75% of the average hourly wage, as suggested by Fezzi et al. (2014) to estimate the opportunity cost of time spent on travelling. Finally, expenditures on entrance fees and consumables goods on sites were obtained directly by the MEME dataset.

For the estimation of the value per trip, I considered two models, namely a Poisson regression model and a Negative Binomial model. This was due to the fact that the number of visits, which was the dependent variable in the model is a nonnegative integer and the frequency of small numbers of visits consisted a sizable fraction of the data set (Parsons, 2003). Dependent variables that were used for the estimation besides the cost per visit were the age of the respondent, the work status, whether they own a car or not and the size of the household, however only the cost per visit variable was significant. Therefore, the other variables were finally omitted. As shown below (Table 6.4), in both models the cost per visit coefficient as expected carries a negative sign. For the used models, the surplus of the individual *n* estimated as $S_n = \lambda_n / -\beta_{tc_r}$, where λ_n is the expected number of trips (here 1) and β_{tc_r}, the estimated coefficient of the cost per trip variable. The estimated values were £69.32 for the Negative Binomial model and £58.01 for the Poisson model.

Table 6.4 Estimates from the Negative Binomial and Poisson model

	Negative Binomial model	Poisson model
Variables		
Constant	1.347 ***	1.371 ***
	(0.0666)	(0.0411)
Cost per visit	–0.014 ***	–0.017 ***
	(0.0035)	(0.0028)
	AIC: 989.27	AIC: 1158.3
	'log Lik.' –491.636	'log Lik.' –577.1431

*** coefficient significant at P≤0.001.

In order to obtain the total value of recreation for visitors, I calculated the percentage of individuals travelling to a location with a river within the catchment area . By considering the number of tourists in England (excluding business trips) from 2011 to 2019 (Visit Britain, 2011, 2012, 2013, 2014, 2015, 2016, 2017, 2018; Great Britain Day Visitor Survey, 2020) and obtaining the number of tourists in the Broadland Rivers Catchment in 2018 (Day and Smith, 2016), I estimated the percentage of the total tourists in England visiting the area of interest. Estimating the individuals traveling to the catchment in 2018 was based on information obtained from the Outdoor Recreation Valuation Tool (ORVal: Version 2.0) developed by the Land, Environment, Economics and Policy Institute (LEEP) at The University of Exeter. The tool consists of an interactive map that allows the selection of land covers at various scales and provides the corresponding economic values related to recreational benefits. For the current study, I selected all Middle layer Super Output Areas (MSOA) that correspond to the study area and contain rivers canals and lakes, which allowed the estimation of the number of tourists making a visit to a water site. Keeping the above percentage constant, I estimated the number of tourists in Broadland Rivers for each year. Finally, by multiplying the number of tourists by the consumer surplus (S_n), the total value of recreation was estimated (Table 6.5).

Table 6.5 Estimated value of recreation 2011-2019 (£ million, 2019 prices)

Year	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Number of visitors (thousands)	954	725	900	700	583	1,053	690	594	940
Annual value (Poisson model)	55.3	42	52.2	40.6	33.8	61.1	40	34.4	54.5
Annual value (Negative Binomial model)	66.1	50.2	62.4	48.5	40.3	73	47.8	41.1	65.1

The final step included the estimation of the asset value of ecosystem services. This was done by estimating the net present value of future flows of the ecosystem services benefits (SEEA EEA, 2017; Turner, Badura and Ferrini, 2019). As there is not an expected life span for the two catchments, the flow of future value was estimated using a 100-year asset life as indicated by the UK Office of National Statistics (Dutton, 2020). Furthermore, concerning the discount rate, estimates assume a 3.5 % discount rate for up to 30 years, declining to 3.0% up until 75 years, and further declining to 2.5% thereafter which is in line with the UK natural capital accounts methodology (Dutton, 2020).

6.6. Results

By completing the procedure described above, I obtained the asset value of the two ecosystem services. The results indicate how past and current water management practices adopted to eliminate pressures and improve the status of water, influence the economic value of harvested ecosystem services. Accounts such as the following that integrate information on the economic consequences of interventions in a systematic and rigorous manner can be expected to provide support for assessing the effectiveness of programmes of measures and the overall effectiveness of the Directive.

Table **6.6** presents the asset value of the two assessed ecosystem services. As observed the values demonstrate fluctuations across years, which is expected given differences in the intensity of use and in the unit value from year to year. For example, water abstraction in Evrotas obtains the highest value in 2011, it declines in 2012 and finally starts increasing again in 2016. As the population and the flow of water increase from year to year, the stock value of water follows the trend of the unit price. Nevertheless, in both Evrotas and Broadland Rivers, there is clearly an increasing trend from 2015 onwards. A way of interpreting this is that from 2015 onwards the contribution of providing water to households becomes more dominant. If such a trend continued policymakers should be alarmed, as sudden events that may influence the availability of water may have a severe impact on the wellbeing of households. Comparing the two cases, it is noted that on average the annual change of the volume of water abstractions is higher in the Broadland Rivers than in Evrotas, which in the WFD context potentially signifies that the latter catchment faces milder pressure on water from residential consumption than the former. Concerning the asset value of water abstraction, as it increases steeper in the Broadland Rivers, it could be said that the dependence of the economy on this specific service is higher in the UK case study, which could mean that the pressures on water from residential consumption will most likely be more intense in the Broadland Rivers in the future.

Table 6.6 Estimated asset value (£ million, 2019 prices)

	Catchment	2011	2012	2013	2014	2015	2016	2017	2018	2019
Water for residential use	Evrotas	73	68	71	67	60	68	73		
	Broadland Rivers	763	817	829	807	765	840	937		
Recreation	Evrotas	136	128	135	130	118	134	144		
	Broadland Rivers (Poisson model)	1491	1347	1379	1322	1263	1341	1300	1267	1306
	Broadland Rivers (Negative Binomial model)	1866	1609	1647	1580	1510	1602	1554	1514	1561

Furthermore, in Broadland Rivers, as previously discussed water quality declined from 2015 to 2019. During these years, however, the asset value of water abstraction increased significantly leading to a peak in 2017 following a notable rise in the flow of abstracted water. These opposite effects may indicate that long-term household water consumption is unsustainable. This signals that the managing authorities may be required to adjust the WFD programme of measures or even develop new that will further disincentivize households from consuming excessively.

As far as the value of recreation is concerned, it also exhibits volatility from year to year. By construction, this variable measures the amount of time people spend outdoors, thus the changes can be attributed to that rather than the money people spend on recreation. In the Broadland Rivers case, the estimates obtained through the use of the Negative Binomial model demonstrate a similar trend to the values included in the UK Natural Capital Accounts (Dutton, 2020). Additionally, the asset value of the flow of recreation is higher than that of abstracted water in both cases, which indicates that the contribution of recreation to the economy is higher, which is in line with results from previous reports in the United Kingdom and the Netherlands (Horlings et al., 2019; ONS, 2020). For policy purposes, this proposes that programmes of measures developed in the context of WFD could be such that are effective at improving water status, while improving the sustenance and provision of recreational ecosystem services.

6.7. Discussion

Overall, the two case studies differ in terms of improvements in the status of the rivers within the catchments. The RBMP for Eastern Peloponnese shows that a high number of rivers in the Evrotas case reached good status in 2017. On the contrary, in the Broadland Rivers catchment, most of the water bodies reached moderate status in 2019, as the condition of some water bodies deteriorated.

To the author's knowledge, neither of the two catchments has used the natural capital methodology in the process of developing the RBMP. However, the tables presented above can be fed into policy analysis to further improve the implementation of the WFD. More specifically, besides improving the description of the case studies, accounting tables can be used to inform the development of programmes of measures and can be utilized in the assessment of the recovery of costs. For example, the Evrotas case study might be more susceptible to pressures related to tourism than to water abstractions in the future, as the number of overnight stays has been increasing, whereas the volume of water abstractions remained almost constant throughout the years. From a policy perspective, this could impose either an opportunity for or a threat to the sustainable use of water resources. On one hand, a higher number of tourists could mean that a higher share of the natural capital will be used for satisfying related needs. Consequently, the use of land in the area might change in the future, as there will be a higher demand for tangible and intangible amenities, such as public transportation and lodging facilities, accommodation facilities (e.g., hotels), parking places, and transportation facilities, bringing about the emergence of new pressures on water resources. On the other hand, policymakers could further improve the overall health of the ecosystem through the adoption of green measures, such as green streets, pocket parks, and tree planting (Mell, 2016; Iii, 2021) that play a critical role in protecting water resources and providing opportunities for recreation, among other things, and hence benefit society (Lovell and Taylor, 2013). As a result, natural capital accounting can assist in identifying areas for public investments that can simultaneously promote human development and the conservation or restoration of natural capital (Ouyang et al., 2016).

On the contrary, the Broadland Rivers catchment seems to be more susceptible to pressures related to water abstractions for residential purposes than those related to recreation, given that, from 2016 to 2017, there has been a major increase in the amount of abstracted water. Future population increases could pose a threat to the sustainable use of water resources if water consumption is not adequately controlled. Nevertheless, the Water Exploitation Index (Eurostat, 2021) for the two countries demonstrates that Greece has been facing increasing pressure on renewable freshwater resources from 2015 onwards, whereas the overall position of the UK has relatively improved compared to previous years. However, the information concerning this index provided by the European Environment Agency refers to the national rather than the catchment scale, therefore disregards regional and seasonal changing conditions. Besides that, as far as the value of recreation in the Broadland Rivers is concerned, it is observed that the average growth rate is negative, which might mean that management practices adopted in the area might have had a negative impact on society. Information on the potential effects of measures on ecosystem services provided by the Environment Agency (Environment Agency, 2014) verifies this result. More specifically, the cultural services of Waveney River (one of the main water bodies in the Broadland Rivers catchment) were expected to be negatively impacted by the proposed measures, as implemented measures could negatively influence areas and structures of cultural interest. Assuming that policy interventions that effectively target pressures are realized, the status of water should be expected to further improve in the Broadland Rivers case, which will provide further opportunities for harvesting ecosystem services in the future. This, on one hand, could increase the annual flow of ecosystem services (Grizzetti et al., 2019), however, reductionist planning that does not account for the effects of measures on other aspects of the resource might lead to changes in the overall functioning of the socioenvironmental system, therefore leading to lower-than-expected benefits. Taking this into account, this chapter suggests that the presented procedure can be of great use to managing authorities. By developing natural capital tables, managing authorities are enabled to obtain insight into the current use of natural resources, as well as the potential aspects that could be influenced in the future to design effective corrective measures.

In other words, such information can be used to monitor the development of the economic–environmental system and form the basis for evaluating the trajectory of future development and the effectiveness of programmes of measures. Natural capital accounts assess the stock value of natural capital and can signal whether PoMs contribute to sustainability. This is particularly important for appraisals of spending options, where considerations such as securing benefits for future generations need to be considered (Bateman and Mace, 2020). For the purposes of the WFD, such information can supplement cost–benefit analysis, which focuses on the flow of benefits from nature (Bright, Connors and Grice, 2019). Besides this, through natural capital accounting, policymakers can evaluate the impact of measures on specific ecosystem services, identify the stakeholders that are affected by water status changes, and assess the unintended consequences of policy responses (Bass et al., 2017). In addition to that, environmental indices were created to measure the interaction of society with environmental resources (Plummer et al., 2012) such as the water resource vulnerability index (Gunda, Benneyworth and Burchfield, 2015) can complement natural capital accounting by deepening our understanding of ecosystem changes (Hattam et al., 2015; McKenna et al., 2019).

Finally, as natural capital accounting methodologies are recent developments (Hein et al., 2016), and because several issues concerning the contribution of natural capital to the economy are still to be resolved (Barbier, 2019), caution should be taken when undertaking such an analysis and interpreting results. Some of the most dominant issues include uncertainty pertaining to our capacities to anticipate the future, the quality of gathered information, and a faulty understanding of the system of interest (Vardon et al., 2018). Furthermore, the development of natural capital accounting tables was based on the application of common economic valuation approaches. For instance, the resource rent method was applied to value water resources in the Netherlands (Edens and Graveland, 2014) However other authors favor the replacement cost method (Barbier, 2007; Remme et al., 2015; Horlings et al., 2019), to avoid undervaluation of water use benefits (Horváthová, 2022). Nevertheless, developing standards for natural capital accounting and further improving current methodologies can foster a better understanding of the complexities of the system, transforming them into manageable risks through the use of a single unit of measurement to express the condition, extent, and value of different aspects of nature (Maechler et al., 2019; Maechler and Graz, 2020), thus, improving water management.

6.8. Conclusions

The European Commission defines natural capital accounting as a tool with which to monitor changes in the stock and condition of natural capital at different scales and a means to integrate the value of ecosystem services into reporting systems (European Commision, 2015). As shown in earlier sections, developing natural capital accounts requires a great amount of data, such as detailed information on ecosystem services supply, assessment of the status of ecosystem assets, and identification of the uses of the ecosystem services as well as their value. In this study, I discuss the links between the steps of the implementation of the WFD and the development of natural capital accounts. Overall, monitoring annual changes in the state of an ecosystem is both a requirement of the WFD and a prerequisite for developing natural capital accounts (Dworak et al., 2005; SEEA EEA, 2017). Assessing trends in ecosystem services can increase our understanding of how the environment functions (Bennett, Peterson and Gordon, 2009) and shed light on the dynamics of the interactions between societies and the environment.

Environmental accounts and, in particular, water accounts, have had many applications around the globe, from preparing catchment management plans and assessing the level of cost recovery (Nagy, Peevor and Vardon, 2017; Oosterhuis, 2017) to monitoring progress towards sustainable development (Schenau, 2017), but still, there is a lack of studies due to limited data and unresolved methodological issues (Balasubramanian, 2020). Assessing how the economic value of the services of interest and the status of water change across the years provides useful insight for policymaking that can reveal the added value of investing in nature. As per the WFD, EU Member States are obliged to design and implement measures to prevent further deterioration of the quality of waters and improve their overall status. The measures implemented in the Broadland Rivers to some extent failed to achieve that (Giakoumis and Voulvoulis, 2019). Therefore, the question arises as to whether traditional measures besides being able to improve water classification can deliver benefits to society. Natural capital accounts have the potential to contribute to the answer to this question, as the obtained economic value incorporates information about the structure of the institutional setting, the intensity of ecosystem services harvesting, and the extent and condition of natural resources (Mullin et al., 2018; Fenichel and Hashida, 2019). They can provide information on trends across time and allow for comparability among river basins, measure the effects of policy interventions on water resources, and give an indication of the cost-efficiency of policies aiming to improve the health of the environment (Russi and Brick, 2013). Taking into account that there are still significant gaps in the assessment of PoMs (European Commission, 2019c), natural capital accounting has the potential to improve their cost-effective analysis to ameliorate the design of policy interventions that target pressures, thus improving water status and, at the same time, contributing towards increasing the benefits societies obtain from the environment (DeWitt et al., 2020).

This study involved the development of accounts of the asset value of two ecosystem services in two areas in Europe that are managed under the Water Framework Directive. To do this, the ecosystem services concept and the principles of the natural capital methodology were utilized. The analysis showed that the data included in the WFD River Basin Management Plans, combined with national statistics, could potentially be used to assess the value of the flow of benefits from efficiently managed water resources. The aim of the study was to explore the benefits of such an approach in a country that has institutionalized it and in a country that has not yet started the process of developing environmental accounts. The estimation of the stock value of ecosystem services in the UK case study was relatively easy, as national databases and databases containing background information on the River Basin Management Plans were publicly available. On the contrary, in the case of Evrotas, data besides that found in the River Basin Management Plan was limited. As a result, a more sophisticated technique was used to estimate the stock value of recreation in the Broadland Rivers case, which provides greater confidence in the obtained values.

While a discussion on the suitability of PoMs is out of the scope of this study and cannot be supported by processed data, this chapter suggests that nature-based solutions might be more appropriate for increasing the benefits obtained from the environment, while benefiting the environment at the same time. Green infrastructure (European Commission, 2013b), another name for nature-based solutions has the potential to make the implementation of overlapping policies and legislation more efficient (European Commission, 2019b) and also generate a high number of co-benefits to society (e.g., enhancement of riverbank vegetation for managing erosion also generates benefits in the form of carbon sequestration). Such policy interventions go beyond managing nature effectively by focusing on societal factors, such as human well-being and poverty alleviation and development while sustaining or improving environmental conditions. Eggermont et al. (2015) classify three types of nature-based solutions according to the degree of technical intervention: i) Better use of ecosystem through minimal interventions; ii) Approaches that relate to the development of sustainable and multifunctional ecosystems; iii) Creation and management of new ecosystems. Maes and Jacobs (2017) define nature-based solutions as “any transition to a use of ecosystem services with decreased input of non-renewable natural capital and increased investment in renewable natural processes”. For example, wetland and floodplain restoration are attractive options, as they offer a high degree of risk protection, have the potential to provide ecosystem services benefits beyond the scope of intervention, and are less costly compared to grey infrastructure alternatives (EEA, 2017). Assuming that such measures could achieve the primary objective of the WFD, nature-based solutions could assist in maximizing the benefits associated with better conditions of water resources, which could effectively increase the value of natural capital. Nevertheless, claims concerning the relationship between different types of PoMs and natural capital should be further investigated.

Finally, a shortcoming of the study is that it focused on two ecosystem services rather than the whole spectrum of benefits provided by the rivers in the two catchments. Data constraints were the primary reason for this choice. The current study is the first that shows how data from River Basin Management Plans can be used for assessing the value of natural capital, though further development of the national databases containing environmental information is needed to obtain better results. More specifically, casting light on the relationship between nature and society requires time series data on various social and economic aspects to be gathered in fixed intervals, for example, every one or two years. That is particularly relevant for the Greek case, where concise databases do not exist. As a result, further investing in the creation of such repositories of information is required, along with the establishment of common protocols for data collection. The WFD, along with other environmental Directives and EU policies, provide a solid base with which to define the collected data needed to support transdisciplinary management practices and the adoption of holistic frameworks.

7. Operationalising nature-based solutions for the design of water management interventions

7.1. Introduction

Across the globe, countries are facing a broad range of challenges, from depletion or even exhaustion of natural capital (i.e., ecosystems and abiotic assets) (Souliotis and Voulvoulis, 2021b) and degradation of its status to pressures on water security and human health. Climate change further exacerbates these phenomena, as global warming is likely to intensify the hydrological cycle leading to increased intensity of extreme events (Tabari, 2020). Additionally, as humans and nature constitute components of a socioenvironmental system that form complex and dynamic connections, any changes in societal and economic variables might serve to alter the quality, quantity, and form of natural capital. Drivers such as land use changes due to urbanization, intensive agriculture and population growth influence significantly biogeochemical cycles, biodiversity and consequently water quality (Teixeira et al., 2014).

The adoption of the Water Framework Directive in 2000 in the EU and similar catchment-based approaches in water management around the world, aimed for a paradigm shift in water management practices towards more holistic and integrated systems thinking (Voulvoulis, Arpon and Giakoumis, 2017). The Directive requires Member States to develop and select Programmes of Measures (PoMs) (Article 11) that improve the status of inland waters (Kallis and Butler, 2001; Souliotis and Voulvoulis, 2021a). However, investing in PoMs has not produced so far the expected results, as still a high percentage of inland water resources in Europe are under threat (European Environment Agency, 2018a). Reasons behind this are that the measures have not adequately addressed identified pressures (European Commission, 2015b; Voulvoulis, Arpon and Giakoumis, 2017), thus, have been unable to address issues such as point and diffuse source pollution holistically (Carvalho et al., 2019); and that Member States have been focusing primarily on fulfilling their compliance obligations and relied their management approach on easy technological fixes, thus did not accommodate the non-linear relationships among components of different socio-economic and environmental systems (Everard and Powell, 2002). In this context, interventions that aim to improve the overall water status rather than specific indicators have a greater potential for success, particularly when those aim to directly restore natural ecosystems by managing the pressures that affect them, often referred to as “nature-based”. This has also been recognised by the 5th WFD Implementation Report, which suggests the broad use of nature-based solutions (NbS) to assist in improving water status and climate proofing (European Commission, 2019c). Indeed, NbS have attracted significant attention from European academics and institutions after the launch of the EU Strategy (European Commission, 2013a).

NbS is a relatively new concept that evolved from principles related to sustainability, resilience, ecosystem management and ecosystem services (Lafortezza et al., 2018), which aims at addressing complex problems of the socioenvironmental system, integrating sectoral policies across different scales (Wright, 2011; Artmann et al., 2019) and advocates for the integration of land, water, and biotic resources (Faivre et al., 2017). The International Union for the Conservation of Nature defines NbS as "*actions to protect, sustainably manage, and restore natural or modified ecosystems, that address societal challenges effectively and adaptively, simultaneously providing human well-being and biodiversity benefits*" (IUCN, 2020). A key element of nature-based solutions is the operationalization of ecosystem services, the tangible and intangible goods and services generated through the functioning of natural ecosystems that support the economy and the wellbeing of humans (Turner, 2003; Koundouri et al., 2015; Lafortezza and Chen, 2016; Maes et al., 2016). NbS by considering the importance of nature to the wider system, focus on making use of interventions to the natural and manmade environment that mimic nature and have the potential of satisfying a range of policy objectives (e.g., ecosystem restoration, increasing human well-being, improvement of the overall status of water resources, sustainable use of resources, climate change adaptation, conservation of biodiversity). Therefore, the design of such solutions incorporates environmental as well as societal factors, such as human wellbeing that influence the overall health and performance of the system. Such measures that are commonly associated with ideas such as natural solutions, ecosystem-based approaches, green infrastructure, and ecological engineering (Benedict and McMahon, 2006; Cowan et al., 2010; Borsje et al., 2011; MacKinnon et al., 2011) are interdisciplinary interventions that have the potential to foster the successful implementation of environmental policies at a relatively lower cost than conventional measures (EEA, 2017) and to generate benefits for the environment and society.

In relation to water management, NbS can support the implementation of the Water Framework Directive and other policies related to water resources (European Commission, 2019b). Evidence suggests that they are particularly cost-effective alternatives to tackle diffuse water pollution (Cuttle et al., 2007; Newell Price et al., 2011; McGonigle et al., 2012). They can deliver multiple benefits (Wolf, 2003; Wise et al., 2010; Raymond, Pam, et al., 2017), such as climate mitigation and adaptation; advancing water management; coastal resilience;

conservation and enhancement of urban biodiversity; improvements in air quality; urban regeneration; increased participation of stakeholders; social cohesion; recreation (Kambites and Owen, 2006); and improvements in public health and well-being (Tzoulas et al., 2007). Their application has also been shown to deliver significant economic outcomes. For example, the restoration of Emscher Landscape Park in Germany resulted in the creation of approximately 86,000 jobs in 20 years (Portugal et al., 2020). However, it has been argued that differences among stakeholders and their perceptions affect the values to be attached to ecosystem services (Sanon et al., 2012; Small, Munday and Durance, 2017). This can lead to trade-offs among stakeholders when some stakeholders enjoy more of an ecosystem service at the expense of others (Howe et al., 2014), influenced by social norms and perceptions (McShane et al., 2011; Wam et al., 2016; Alves et al., 2020). Trade-offs among stakeholders are shown to be low in the short term and high in the long term (Giordano et al., 2020) following the changes in the provision of ecosystem services, thus changes in the types of generated benefits. Nevertheless, most of the evidence on benefits stemming from NbS comes from studies that assess their application in urban areas and only a small share in lakes, ponds, and rivers (Chatzimentor et al., 2020). Therefore, currently, a significant knowledge gap exists concerning how such interventions should be assessed both in terms of their effectiveness and the ecosystem services they produce.

Taking that into account, the current study evaluates the potential of NbS as interventions for the implementation of the Water Framework Directive. The following sections describe the types of such interventions and their potential benefits. Through a case study in the United Kingdom, where a constructed wetland was used by a private water company to reduce its negative impact on a local river and align with the targets set by the Directive, this chapter evaluates its potential as an intervention.

7.2. Natured-based solutions and their relevance to water management

NbS is an umbrella concept that encompasses a range of actions such as ecosystem-based adaptation and green infrastructure (Seddon et al., 2020) and could be regarded as the counterpart of “grey infrastructure”, which consists of the stock of engineering measures, facilities and installations used for complementing or substituting functions performed by ecosystems, such as water collection, purification, and storage, among others. To clarify the distinction between engineering measures and NbS, Eggermont et al. (2015) propose a typology to characterize NbS in terms of how much engineering interventions are utilized and the number of ecosystem services and stakeholder groups targeted by their design and implementation. They describe three different types of NbS: actions that include minimal or no interventions in ecosystem services; management approaches that influence the functionality of ecosystems and landscapes; and practices related to intrusive management of ecosystems or creation of new ecosystems aiming to increase the overall production of ecosystem services.

In relation to managing inland water resources, examples of NbS practices include the use of buffer strips to mitigate diffuse pollution, protect biodiversity, reduce erosion of the riverbanks and increase the aesthetic value of the landscape (Smith et al., 2014; Münch et al., 2016; Cole, Stockan and Helliwell, 2020); river restoration aiming to return water bodies to a status that provides a higher volume of ecosystem services (Blau et al., 2018); natural water retention measures to mitigate flood risk (Collentine and Futter, 2018); and artificial streams (Brown et al., 2018) among others (Table 7.1).

Table 7.1 Types of nature-based solutions and their benefits

Type of NbS	Description	Potential to mitigate WFD pressures	Non-exhaustive list of benefits	Indicative examples
Reconnecting rivers to floodplains	Removing barriers along the course of the river	Point and diffuse source pollution; water abstractions; physical modification	Water supply regulation; flood mitigation; water purification; erosion reduction; biodiversity; opportunities for recreation; nutrient replenishment; resilience to extreme climate events; educational opportunities; livelihood opportunities	(Pander, Mueller and Geist, 2015; Schindler et al., 2016; Funk et al., 2019)

Type of NbS	Description	Potential to mitigate WFD pressures	Non-exhaustive list of benefits	Indicative examples
Reforestation	Increase in the number of trees and other vegetation in the catchment	Point and diffuse source pollution; physical modification habitat loss; sediments	Water supply regulation; riverine flood mitigation, water purification; erosion control; biodiversity; recreation and tourism; carbon sequestration; climate regulation; livelihood opportunities	(Nisbet et al., 2011; Perni and Martínez-Paz, 2013)
Soils and vegetated land	Maintaining good soil structure and vegetation cover	Point and diffuse source pollution; physical modification; sediments	Improved soil structure; increased drainage; water quality improvement; increased crop production; resilience to extreme climate events	(Soana, Fano and Castaldelli, 2021)
Riparian buffers	Vegetated areas between water streams and terrestrial ecosystems	Point and diffuse source pollution; physical modification	Flood mitigation; water purification; erosion reduction; water temperature control (due to shade); biodiversity; opportunities for recreation; aesthetic value	(Volk, Liersch and Schmidt, 2009; Bergfur et al., 2012; Stutter, Chardon and Kronvang, 2012)
Wetlands	Construction of shallow vegetated water bodies	Point and diffuse source pollution; physical modification; habitat loss	Water supply regulation; flood mitigation; water purification; water temperature control; biodiversity;	(Karjalainen and Heikkinen, 2005; Harrington, O'Donovan

Type of NbS	Description	Potential to mitigate WFD pressures	Non-exhaustive list of benefits	Indicative examples
			opportunities for recreation; livelihood opportunities; resilience to extreme climate events; educational opportunities; aesthetic value; carbon sequestration	and McGrath, 2013)
River and wetlands restoration	Reinstating the natural processes of water bodies	Point and diffuse source pollution; physical modification; artificial flow pressures; habitat loss; sediments	Water supply regulation; flood mitigation; opportunities; aesthetic value; biodiversity; recreation and tourism	(Hoffmann, Kronvang and Audet, 2011; Gilvear, Spray and Casas-Mulet, 2013; Darwiche-Criado et al., 2017)
Sustainable drainage systems (SuDS)	Drainage systems that manage rainfall where it falls	Point and diffuse source pollution; physical modification; abstractions and other artificial flow pressures	Water supply; flood mitigation; groundwater recharge; habitat; water quality improvement; aesthetic value	(Phil Jones and Neil Macdonald, 2007)

In order to distinguish NbS from traditional management approaches, the International Union for the Conservation of Nature proposes eight principles (Cohen-Shacham et al., 2016) upon which NbS are based: 1) embrace nature conservation; 2) can either be implemented alone or in combination with other measures to tackle societal challenges; 3) are specific to a site and local cultural context; 4) produce benefits to all members of the society in an equitable way, promoting transparency and participation; 5) maintain biological and cultural diversity and enable ecosystems to evolve over time; 6) landscape is the scale of their application; 7) encompass the trade-offs between the delivery of short term economic benefits for development and future options for the production of a wider range of ecosystem services; and 8) are an integral part of policy design. Comparing these with the PoMs approach introduced by the WFD brings to light several common elements (Figure 7.1).

Embrace nature conservation	The WFD aims at preventing deterioration of water resources and enhance the status of aquatic ecosystems
Can either be implemented alone or in combination with other measures	Each PoM includes 'basic' and 'supplementary' measures
Are specific to a site and local cultural context	PoMs are implemented at the river basin level
Produce benefits to all member of the society in an equitable way, promoting transparency and participation	The WFD relies on the information, consultation and involvement of the public/users
Maintain biological and cultural diversity and enable ecosystems to evolve over time	The WFD recognizes the diverse conditions and needs across Member States that require specific solutions PoMs must be adjusted to regional and local conditions
Landscape is the scale of their application	PoMs are implemented at the river basin level
Encompass the trade-offs between the delivery of short-term economic benefits for development and future options for production of a wide range of ecosystem services	The WFD promotes sustainable water use based on long-term protection of available water resources
Are an integral part of policy design	PoMs are integral part of the River Basin Management Plans PoMs are reviewed every 6 years to inform policy decisions concerning the subsequent management cycle

Figure 7.1 Connection between the principles of NbS and the WFD

The Directive follows an integrated water management approach and defines catchments as the systems of interest. For each catchment, Member States are obliged to report programs of measures (Basic and Supplementary) that must accord to 25 Key Type of Measures, which are expected to deliver the majority of environmental improvement in each water body (European Commission, 2015a). At a minimum these measures must collectively: a) accommodate the provisions of water and other environmental legislation; b) implement Article 9 of the WFD concerning the recovery of the cost of water use; c) promote efficient and sustainable use of water resources; d) protect drinking water quality; e) control abstractions from surface and groundwater; f) control recharging of groundwater; g) control point source discharges; h) control inputs of diffuse pollutants; i) address other significant impacts on status; j) prohibit direct discharges to groundwater; k) eliminate or reduce pollution by Priority Substances; and l) prevent accidental pollution (CIS, 2016). Additionally, the development and implementation of PoMs must accommodate the engagement of relevant stakeholders (Pellegrini, Bortolini and Defrancesco, 2019). An innovative feature of the Directive that also relates to the notion of NbS is the inclusion in the list of PoMs of a type of measures described by the WFD Guidance documents (European Union, 2014) as Natural Water Retention Measures. Such actions have been associated with reducing the negative effects of floods and droughts (Environment Agency, 2010b; Linnerooth-Bayer et al., 2013) using natural means and processes, however, in the majority of Member States, natural water retention measures are not reported yet as part of the operational programmes to tackle significant pressures (European Commission, 2021a).

The Directive has overall introduced the utilization of the functions and processes observed in nature in water management. As a result, there is high correlation between the aims of the programmes of measures and the definition and principles of NbS, though as a concept was developed approximately ten years (International Union for Conservation of Nature (IUCN), 2009) after the adoption of the Directive. The nexus between ecosystem services and nature-based solutions (Babí Almenar et al., 2021) provides a direct way of comprehending the effects of interventions on the environmental and societal systems and enables managing authorities to address multiple goals (Everard, 2014). Furthermore, utilizing NbS in environmental management and promoting awareness through the effective communication of their ecosystem services benefits, has the propensity to foster the collaboration of different stakeholder groups and unlock private sector investment (IDB, 2019). To further provide evidence of the applicability of such measures and to accelerate their uptake in environmental management, the European Commission diverted a significant share of funding to Horizon 2020 framework programme (approximately €185 million) for research and innovation projects (e.g., NATURVATION[15]; NAIAD[16]; PEGASUS[17]; PHUSICOS[18]). Furthermore, the European Green Deal Investment Plan (European Commission, 2020c) developed to facilitate the transition to a climate-neutral inclusive economy is expected to further promote projects related to natural capital and nature-based solutions.

In light of the problems with its implementation, incorporating NbS in the designing of PoMs could reinforce a more systemic view of the WFD, according to which its objectives encapsulate the improvement of the overall health of the system (Voulvoulis, Arpon and Giakoumis, 2017). Following the principles of the integrated water resources management paradigm (Biswas, 2004), the WFD assigns PoMs with the goal of enhancing the status of water without disregarding consequential wider effects of achieving this, such as increasing human welfare, sustaining development, distributing costs and benefits equitably, mitigating the impacts of climate change amongst others. Therefore, designing measures that are holistic and can yield maximum benefit return requires relevant authorities to take into account how components of the wider system interact with each other and how by changing the parameters in one part of the system affects other parts. NbS either implemented alone or when appropriate, in conjunction with traditional approaches might be a solution to the problem of improving the status of water resources, without disproportionately increasing the welfare cost to society.

Nevertheless, although a range of case studies are available, few include economic information (Le Coent et al., 2021), which indicates a significant gap in the assessment of NbS performance. Such data is essential for increasing our understanding of the societal aspects of NbS, improving water management decisions, and increasing the uptake of such approaches by public and private organisations. Taking the above into account and aiming to foster the inclusion of NbS in PoMs, in the following sections a case study is presented where the development of a wetland was favoured over grey infrastructure alternatives by public and private stakeholders to remove nutrients discharged in a chalk river in the UK. The chapter includes information on the effectiveness of the NbS as well as its benefits and costs.

[15] https://naturvation.eu/home
[16] http://naiad2020.eu/
[17] http://pegasus.ieep.eu/
[18] https://phusicos.eu/

7.3. Materials and methods

7.3.1. Description of the case study

The River Ingol is a 10.3 km length chalk stream in North Norfolk, which runs from its source in Shernborne to Snettisham Nature Reserve. Ingol is one of the 200 chalk rivers in the world and one of the twelve found in Norfolk. The catchment covers an area of 35.3 km^2, which includes Sites of Special Scientific Interest (SSSI), Special Areas of Conservation (SAC) and areas of Ramsar status. In terms of the EU Water Framework Directive classification, the river has a failing status for invertebrates, fish, and phosphate concentrations (The Norfolk Rivers Trust, 2015). Ingol constitutes a heavily modified river, the status of which was classified as Poor in 2010 and moderate in 2019 (EA, 2021). Among other pressures, such as physical modifications, a Water Recycling Centre operating in the area was identified as a major source of nutrient concentrations, which affected the quality of the river.

Figure 7.2 The area by the Ingoldisthorpe Recycling Centre after the completion of the works. Credit: William Morfoot

In order to avoid deterioration of the status of the Ingol River, the Environment Agency required the adoption of additional measures to address a reduction in ammonia and phosphorus levels (1 *mg/l* NH_3*-N* and 4.5 *mg/l P*). Available methods to achieve this include electrolysis (Kim et al., 2013), micro- and ultra-filtration (Nir et al., 2009; Arnaldos and Pagilla, 2010), reverse osmosis (Kumar et al., 2007) and Membrane Bioreactors Technology (Daigger et al., 2005; Gupta et al., 2012; Iglesias et al., 2017; Voulvoulis, 2018), ion exchange (Guida et al., 2021), air stripping (Sengupta, Nawaz and Beaudry, 2015) to name a few. Indicatively, the unit cost (including capital and operating) of such alternatives ranges between 0.297 and 2.746 $/m^3 depending on the capacity of each facility (Bhojwani et al., 2019). After assessing the cost-effectiveness of Membrane Bioreactors Technology and diverting flows to a neighbouring recycling centre, the water company in collaboration with the local River Trust proposed a soft engineering solution, in the form of a constructed wetland to naturally filter water and improve the quality of water released into the Rivel Ingol.

Such an alternative was deemed suitable for removing nutrients from water at a significantly lower cost. More specifically, the company estimated that installing chemical phosphorus stripping at the Ingoldisthorpe Recycling Centre in order to reduce concentrations by 90% would incur a one-off capital cost of £1 million and recurring maintenance costs of £500 thousand per year. In addition, the drive to reduce greenhouse gas emissions, with the water industry committing to deliver net zero by 2030, two decades ahead of the UK Government's legally binding target of 2050, was a contributing factor. The solution put forward had a carbon footprint of approximately 179 $C0_2$ eT, 55% lower than traditional construction methods (Water Projects, 2019). Besides concerns about costs, the construction of a wetland was expected to provide additional benefits (The Norfolk Rivers Trust, 2015). More specifically, the literature identifies several benefits dwelling from integrated wetlands that positively influence society either directly or indirectly. Indicatively, water supply; water quality; flood abatement, carbon sequestration and management, opportunities for recreation and aesthetic value; flood abatement; and biodiversity support are goods and services that have been reported (Costanza et al., 1997; Zedler and Kercher, 2005; Yang et al., 2008; Zhou et al., 2009; Ghermandi et al., 2011; Reynaud et al., 2015; Merriman et al., 2018; Xu et al., 2020; Zhou, Wu and Gong, 2020). Therefore, if the constructed catchment had been proven to be unsuccessful to treat water, there would still have been some benefits from its application. Nevertheless, adopting an NbS was in line with the water industry national environment programme which encourages water companies in the UK to engage in investments that provide a thriving natural environment with increased environmental value through catchment-based approaches (Defra, Environment Agency and Ofwat, 2021).

In 2017, in an area covering approximately 1.08 ha, the construction of the wetland began. The project consisted of four interconnected shallow pools, close to the Water Recycling Centre in Ingoldisthorpe. The wetland was enriched with 25,000 native aquatic plants (e.g., *Iris pseudacorus*, *Cyperaceae sp.*, and *Typha latifolia*) to filter the water passing through the pools and 1,400 native tree species (e.g., *Acer campestre*, *Quercus robur* and *Carpinus betulus*) to assist in integrating the wetland into the wider ecosystem (Cooper et al., 2020). Operationally, the water flows from the Water recycling Centre into the wetland zone through an installed pipe. Then it flows through the four cells and upon reaching the fourth discharges into the River Ingol (Figure 7.2).

The total capital cost of the development of the wetland was £194.000 (Cooper et al., 2020), which was covered by the water company. The funding was part of the first Green Bond in Europe totalling £250 million, issued by the company. Additionally, the Norfolk River Trust has been actively involved in the implementation of the project by undertaking its feasibility study, designing, and constructing the wetland, and carrying out the maintenance of the wetland (WWF, 2020).

7.3.2. Collection of data

Assessing the impact of the introduction of the Ingol wetland was based on a previous study by Cooper et al. (2020), who assessed the environmental and economic efficacy of the wetland in mitigating eutrophication risk. Besides that, background documents and data published or provided either by the local River Trust, the Environment Agency or the water company were used to identify the type of ecosystem services that were relevant to the case study as well as the effects of the wetland on these types of services. Furthermore, communicating with the aforementioned organizations clarified a series of issues and influenced the extent of the assessment. A shortcoming of the study is that Ingol wetland has only been operational since mid-2018 and as COVID-19 hit the UK in 2019, field campaigns were halted in 2020. To address this issue, where data for specific services was not available, the study relied on results of other relevant studies.

7.3.3. Assessment of the value of benefits of the Ingol wetland

As far as the water quality is concerned, the wetland influenced the concentration of main pollutants (European Commission, 2000). Data from the Environment Agency that compares the concentration of Phosphorus (Figure 7.3) and Ammonia (Figure 7.4) in water exiting the

Water Recycling Centre and the Ingol wetland verify that the latter is efficient in removing both pollutants.

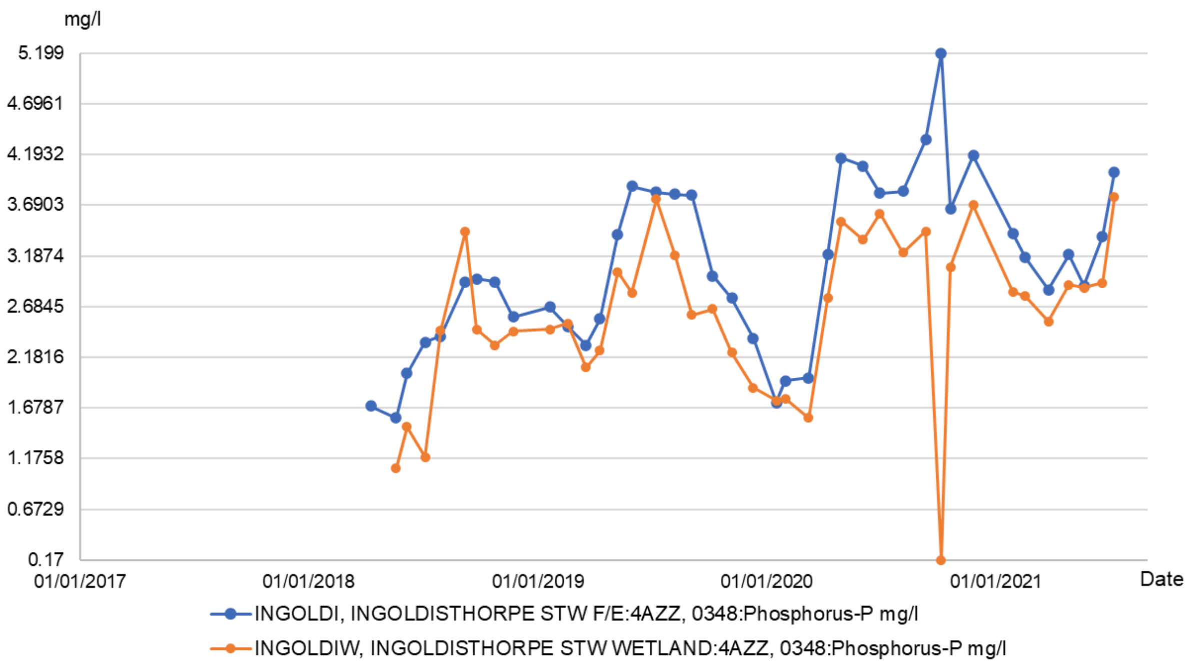

Figure 7.3 Phosphorus concentration in water entering and exiting the wetland, source: Environment Agency (2021)

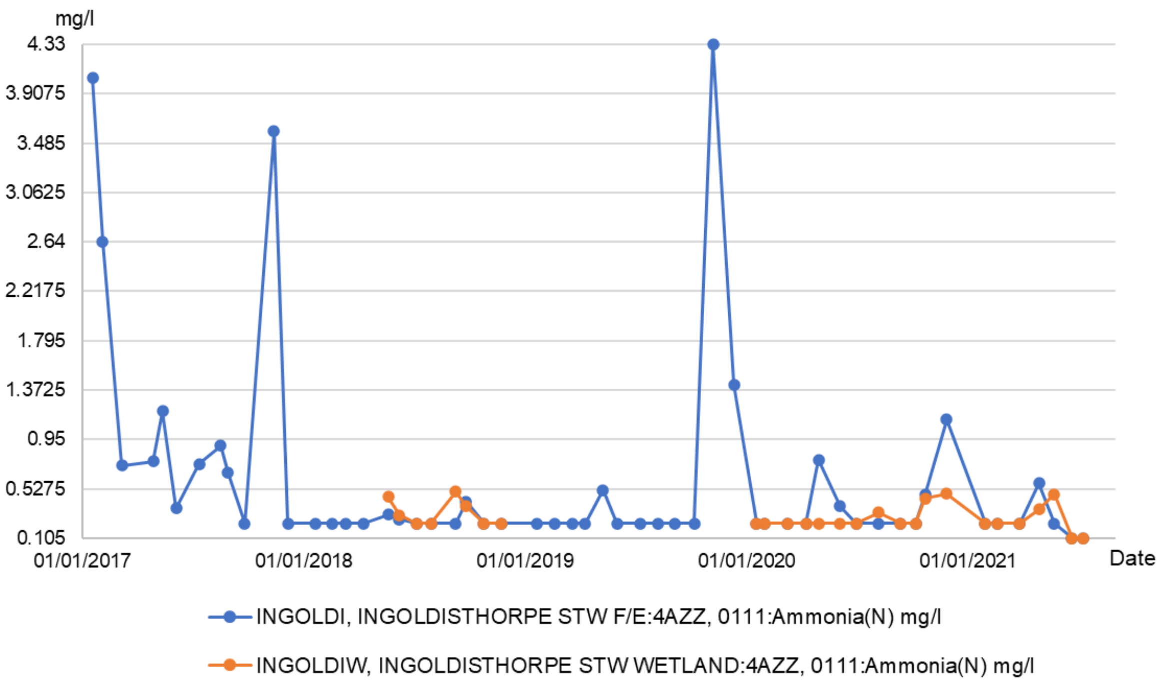

Figure 7.4 Ammonia concentration in water entering and exiting the wetland, source: Environment Agency (2021)

In line with this, the study of Cooper et al. (2020) demonstrates that the constructed wetland removed additional loads of nutrients in the discharges of the local Water Recycling Centre. More specifically, the wetland was estimated to receive on average 301.1 m^3 and discharge 128.7 m^3 of water into the river per day. Assessing the nutrient loads in the entry and exit points of the NbS showed reductions of 72% for nitrate and 69% for phosphate (Table 7.2). Furthermore, the project was able to reduce dissolved organic carbon (DOC) loads by 53.3%, thus acted as a sink of DOC. Additionally, the total carbon (TC) load discharging from the constructed wetland was 43.5% lower in the outlet compared to the inlet.

Table 7.2 Summary of nutrient loads at the intel and outlet of the Ingol catchment (Cooper et al., 2020)

	Inlet (kg/day)	**Outlet(kg/day)**	**Reduction (%)**
Nitrate	8.55	2.42	71.7
Phosphate	0.61	0.19	68.9
DOC	1.93	0.90	53.3
Total Carbon	8.88	5.02	43.5

The economic value of effluent polishing performed by the wetland was assessed through the use of the replacement cost method (Boyer and Polasky, 2004). This approach takes into account alternatives for obtaining the same result (i.e., water filtration) should the wetland no longer function properly or seize to exist. In other words, if the wetland had not been constructed, an obvious choice to achieve a reduction in nitrate and phosphate would have been to further invest in enhancing treatment at the Ingoldisthorpe recycling centre. Using unit cost values for the reduction of nitrate and phosphate found in relevant national studies (Ofwat, 2005; OXERA, 2006), it was estimated that the wetland provides water purification benefits that amount to approximately £28,000 per year (Table 7.3). This should be regarded as the minimum economic benefit obtained by the application of this intervention, as the unit value of treating nutrients does not account for the acquisition cost of more advanced equipment.

Table 7.3 Economic value of natural filtering of nutrients

	kg/day	**Cost**	**Value per year (£ in 2020)**
Nitrate-N	6.13	7.4	22,103.16
Phosphate-P	0.42	46	3,990.80
Total			27,938.33

Besides that, wetlands with a high variety of plant species have the capacity to sequester carbon (Du et al., 2018), thus, contribute to the mitigation of climate change effects (Vymazal, 2011). They either act either as a sink or a source for DOC and are considered vital carbon sinks despite covering a small percentage of our planet (Melton et al., 2013). Studies report that they store up to one-third of the organic soil carbon of the world (Villa and Bernal, 2018). Natural England (Natural England, 2019) provides a breakdown of wetland carbon storage by type ranging from 57.5 Mt C for raised bogs to 186.4 Mt C for lowland fens, which have been substantially degraded by agricultural conversion. Though small-scale constructed wetlands used for reducing effluent discharges may sequestrate small amounts of carbon, they are considered considerable carbon sinks due to the difference in energy consumption between the wetland and other alternatives, such as wastewater treatment plants (Ogden, 2013). However, management options that affect land uses in the area where the wetland is located impact the ability of the ecosystem to sequester carbon (Were et al., 2019). De Klein and Van der Werf (2014), assessed the CO2 equivalent balance of a constructed wetland which was estimated to range from 0.27 to 2.4 kg per m^2 per year. Another study by Lloyd (2006) that measured the carbon balance of Tadham Moor, a lowland wet grassland, found that during 2002 the carbon assimilated into the wetland was 169 gr per m^2 higher than the produced carbon. Taking into account the values of different types of wetlands (Table 7.4), an average value of 8.02 CO_2 equivalent per hectare per year was estimated. Using a price of £13.15 per ton of carbon equivalent (BEIS, 2019), the estimated total value of carbon sequestration was estimated to be £108.62 per hectare per year.

Table 7.4 Carbon sequestration in different types of wetlands

Type of resource	**tCO_2 equivalent per hectare per year**	**Source**
Floodplains	3.365	(Natural England, 2019)
Lakes	7.1	(Natural England, 2019)
Ponds	16.12	(Natural England, 2019)
Constructed wetland	13.35	(De Klein and Van der Werf, 2014)
Wetland meadow	0.169	(Lloyd, 2006)

Additionally, constructed wetlands have the potential to increase local biodiversity by providing food and breeding sites (Sebastián-González, Sánchez-Zapata and Botella, 2010; Semeraro et al., 2015). Van Biervliet et al. (2020) who monitored bird species richness in Mundesley Beck report that the post-commissioning number of avian species was 28 in 2015 and 26 in 2016 compared to 10 in 2014 prior to the installation of a wetland similar to that by the Ingol River. Several academic studies have assessed the economic value of such an improvement to users of wetlands. For instance, Perni and Martínez-Paz (2017) used the Choice Experiment method to assess the value of different levels of ecosystem service provisioning of the El Hondo wetland located in the Segura River Basin District in Spain. Their econometric analysis concluded that respondents would be willing to pay 0.18 euros per year for a 1% increase in biodiversity. Taking this into account as well as other studies (Stevens, Benin and Larson, 1995; MacDonald, Bergstrom and Houston, 1998; Reynaud et al., 2015; Rulleau, Dumax and Rozan, 2017; Aguilar, Obeng and Cai, 2018) that assessed the value of increasing flora and fauna species in wetland and by employing the benefit transfer approach (Plummer, 2009; Boutwell and Westra, 2013; Koundouri et al., 2015), it was estimated that the Ingol wetland could provide a value of £30.4 per user per year by supporting the increase in biodiversity and providing habitat for species.

7.4. Results

The Ingol constructed wetland was an application of a nature-based solution that was successful in reducing specific nutrients in the effluents of the Ingoldisthorpe Water Recycling Centre. In terms of its efficiency, data signifies that it has achieved the aim of its design by reducing pressures associated with eutrophication in the receiving waters. More specifically, past information collected through sampling that took place approximately 2km downstream from the works provided by the Environment Agency indicates the improvement of the phosphate classification in the River from the Poor/Bad boundary to Moderate/Poor (Figure 7.5). Thus, combining the NbS with the already established grey infrastructure showed to be a successful measure in terms of following the requirements of the WFD.

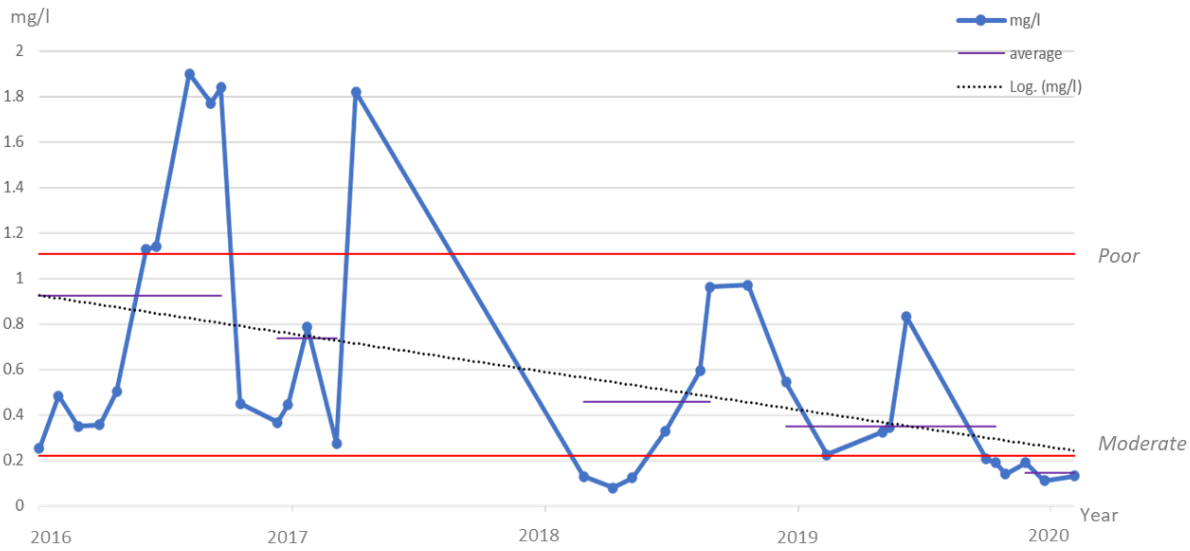

Figure 7.5 Phosphate (mg/l) concentration in the Ingol River, source: Environment Agency (2021)

Furthermore, the adoption of this NbS for improving water quality resulted in significant cost savings and provision of ecosystem services, i.e., effluent polishing (water purification), habitat for species and carbon sequestration. Concerning these benefits, effluent polishing obtains the highest value, followed by carbon sequestration and habitat for species (Table 7.5). Larger wetlands might result in higher sequestration of carbon, thus higher value per year. Considering the carbon emissions during construction, the construction of the wetland produced approximately half of the carbon emissions compared to adopting grey infrastructure alternatives (179 instead of 396 CO_2 eT) (Water Projects, 2019).

Table 7.5 Monetized socioeconomic benefits provided by the Ingol Wetland

Type of ecosystem service	Monetized benefit	Unit
Effluent polishing	£27,938.33	per year
Habitat for species	£30.4	per user per year
Carbon sequestration	£108.62	Per hectare per year

A small number of ecosystem services was assessed, due to the specific features of the wetland, and information provided by the Norfolk River Trust concerning the characteristics of the area and the users of the wetland. For instance, though such projects may have aesthetic value for local residents, such effects were not assessed. Aesthetic value is usually elicited through the influence of proximity to environmental amenities on housing prices (Sohn et al., 2020). The Ingol wetland is located in a relatively isolated location, and it is not likely to be viewed from properties in the area, therefore its effect on the prices of properties would most likely be negligible. Furthermore, the number of tourists or visitors in the area had not been recorded and Norfolk River Trust estimated that they would be less than 10 in a year. As a result, recreational benefits were not considered. However, constructed wetlands may yield significant recreational benefits. For example, Ghermandi and Fichtman (2015) estimated that mean recreational benefits of constructed wetlands to be 8,397 €/ha/year. Finally, since data on non-ecological parameters have not been collected yet (e.g., number of visitors), estimates were based on the literature and information provided either by the water company or the local River Trust. A thorough assessment of the ecosystem services might bring to light additional benefits and costs.

7.5. Discussion

Using the forces of natural processes is an underexplored opportunity for managing water resources. Soft engineering approaches, such as NbS either operating alone or in conjunction with traditional technical measures, if designed properly, have the capacity to deliver environmental objectives and provide additional economic benefits, thus constitute holistic interventions. The potential of operationalizing such approaches for the WFD has not been considered by EU Member States yet. It is an opportunity for designing cost-effective PoMs that correspond to anthropogenic pressures on water bodies and increase welfare beyond the level of traditional practices.

The implementation of the Directive comes with significant financial costs to the managing authorities, estimated at around 119.8 billion Euros according to an assessment of the first River Basin Management Plans of 15 Member States (De Paoli, Mattheiß and Strosser, 2012). Moreover, the application of measures can be limited by the available financial mechanisms and the lack of designated EU funding (Farmer, 2011; Grygoruk and Okruszko, 2015) for the WFD implementation (Zingraff-Hamed et al., 2020). Capital-intensive investments can therefore pose a risk to the delivery of environmental protection. NbS can offer an alternative, as they are less costly, thus may lead to a higher degree of compliance, as well as having the potential to attract funding from several sources given the multi-benefits nature of such projects (Pecharroman et al., 2021). From this perspective, NbS could offer a more sustainable solution as they are fiscally prudent and ensure a better return on investment not only for public authorities, but also for private companies through the provision of multiple ecosystem services (Wade and McLean, 2014; Hamann et al., 2020). Moreover, NbS provide opportunities for synergies between different types of entities, as well as opportunities for corporations to align with wider policy objectives such as the Sustainable Development Goals (Kim, 2021). For example, Seddon et al. (2021) present a list of projects funded by private companies some of which are developed with the engagement of NGOs and/or public institutions. Furthermore, the stakeholder participation aspects of the WFD could be realized through the designing and development of NbS, as such interventions require the collaboration of different stakeholders (e.g., national, and local authorities, non-governmental organizations, farmers, companies, and local citizens).

Regardless of the opportunities provided by such interventions, care must be taken to avoid their application as a cheap alternative to programme of measures required to reduce pressures on water systems. Despite the higher cost of grey infrastructures, their effectiveness may be easier to be assessed, given the currently available knowledge on the functioning of

NbS. Furthermore, PoMs must be adapted to the pressures and characteristics of each specific site. This might mean that despite the value of multiple benefits generated by NbS in the long term, the need for immediate action might necessitate the adoption of more traditional approaches to avoid further damage. NbS are not a panacea; favouring them over traditional approaches without carefully designing and implementing them can cause adverse effects on ecosystem services and local communities (Seddon et al., 2021).

The nature-based solution adopted by the water company in the case study presented was demonstrated to be a novel initiative, not only due to its technical but also due to its governance characteristics, being a privately funded project, which necessitated the involvement of the Environment Agency, the Norfolk River Trust, the private company, and the owner of the land on which the wetland was built. The wetland was designed to reduce water pressures as per the requirements of the WFD while generating additional environmental and socio-economic benefits. Thus, its overall value exceeds the value of the benefit of the water company becoming compliant with existing regulation and Environment Agency recommendations. As shown above, the wetland provides water purification benefits, as well as benefits to society that do not relate to the primary objective of its construction, and which could not have been delivered by conventional water treatment approaches. By considering the costs of alternative approaches to improve water quality assessed in the design phase of the project, it could be claimed that the developed NbS has been proven to be a cost-effective measure.

However, the wetland only treats a small amount of water and occupies a wide area compared to the traditional treatment systems. In line with this, Stefanakis (2015) provides a comparison between conventional treatment systems and constructed wetlands in terms of their ecological, technical, and financial characteristics: typical facilities consist of complex mechanical equipment that is not environmentally friendly, and whose maintenance is expensive as it is usually performed by specialized personnel. Furthermore, they generate by-products such as sludge and are associated with visual and olfactory disturbances and are characterised by high emissions. While constructed wetlands overcome such issues as they are easy to build, consume less energy, require low maintenance, and they do not produce by-products apart from plant biomass. The results in this study demonstrate that creating a hybrid system, consisting of a water recycling centre and a wetland, results in benefits that extend beyond those that would have been obtained if only conventional approaches had been followed.

Furthermore, when not implemented alone, NbS are proven to improve the performance of existing infrastructure by improving the functionality of the system and reducing costs

(Opperman and Warner, 2011). When compared with grey infrastructure measures, NbS solutions exhibit a higher degree of flexibility in modifying the performance of the system to correspond to radical changes imposed by climate change and economic development (Kapetas and Fenner, 2020). Mainstreaming the implementation of NbS is associated with challenges concerning design standards, regulation, socio-economic challenges, financing, and innovation (Zuniga-Teran et al., 2020). Additionally, NbS measures may require more space than traditional approaches (Galler, von Haaren and Albert, 2015; Vineyard et al., 2015) and have the weaknesses of being less tested than grey measures (Alves et al., 2018). Kumar et al. (Kumar et al., 2021) who reviewed the literature and presented state-of-the-art models and tools that are commonly used for the optimal allocation, design, and efficiency evaluation of NbS for hydro-meteorological risks, concluded that the literature that concerns their effectiveness, costs and benefits is still underdeveloped. Consequently, given the lack of standardized assessment methods of their performance, wide adoption of such approaches is still slow.

Interventions that incorporate NbS characteristics may face uncertainty about their performance due to insufficient technical knowledge and experience on NbS measures. For instance, though green spaces provide health benefits (Berg et al., 2015), the effects of NbS on health are still unknown due to a lack of relevant studies (Nieuwenhuijsen, 2020). Moreover, if not assessed, designed, and implemented in a systemic way, such measures may result in improvement in one part of the system and negatively impact other components (e.g., the introduction of trees and plants in cities may increase pollen particles in the air and trigger allergies (Ostrom, 2003). Therefore, institutions that design such kind of measures must account for the specific characteristics of the implementation area. For example, the efficiency of nutrients removal of constructed wetlands depends heavily on the type of wetlands, the introduced plant types, the season, the climatic conditions, and management practices (Jahangir et al., 2016). Additionally, lack of properly defined property rights might hamper the adoption of NbS. In general, NbS can potentially create a wider range of benefits than engineering solutions. From a policy perspective, such benefits are public goods, in the sense that they are non-excludable and non-rivalrous in consumption (Ostrom, 2003). For example, manmade wetlands provide habitat for pollinators (Harrington and McInnes, 2009), which could increase crop productivity, a benefit harvested by farmers in the vicinity. Such externalities (positive or negative) need to be internalised into the decision-making mechanism to enhance the cost-effectiveness assessment of the measures.

Finally, attention should be drawn to the fact that NbS should not be regarded as a one-size-fits-all measures as any potential benefits are site- and context-specific. In the case of the

Ingol wetland, designed parameters were such that could not substitute a pre-existing measure but rather complement its effectiveness. The application and specifications of nature-based solutions should correspond to the pressures on water resources, and to a number of other characteristics that may influence how well they perform.

7.6. Conclusions

In this chapter, the potential of NbS as interventions for catchment management in the context of the WFD was examined. A case study where their application was for the purpose of improving effluent quality of a water recycling centre, demonstrated that the effective design of nature-based solutions and their operation in tandem with traditional grey infrastructure can achieve water quality targets while offering added value by providing carbon sequestration and wider habitat benefits. Nature-based solutions have the potential to outperform the end-of-pipe solutions used to protect the environment while delivering multiple benefits.

Such approaches can promote a holistic view on the interaction between the environment and society, ensure additional socioeconomic benefits through the provision of ecosystem services and due to their low cost and increased benefits incentivize private sector uptake. In particular, the combination of a NbS with a traditional recycling centre can improve water quality at lower cost. The operationalization of NbS could therefore have far-reaching implications for water management. However, decisions on choosing such alternatives should not be based on the costs alone, but on their overall potential to improve the functioning of the socioenvironmental system.

Finally, the wide adoption of NbS currently faces certain barriers (Seddon et al., 2020) that need to be addressed both by academics and practitioners. NbS require coordination of actions of different stakeholders (Vogl et al., 2017; Albert et al., 2019) who see such measures in terms of their operation and remit, while disregarding the multiple facets of NbS. Consequently, issues for further investigation may include assessing the trade-offs among social, environmental and economic dimensions of nature-based solutions; identifying ways in which the perceptions of citizens can be incorporated into the designing and selection of nature-based solutions (Chatzimentor, Apostolopoulou and Mazaris, 2020); as well as analysing further the economic impact of such measures and identifying ways which would incentivise private companies to opt-in for investing in these types of practices.

8. Sustainability transitions: The role of economics in system transformations

8.1. Introduction

According to the latest Global Assessment on Biodiversity and Ecosystem Services by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) “nature is declining globally at rates unprecedented in human history – and the rate of species extinctions is accelerating, with grave impacts on people around the world” (IPBES, 2019). Deforestation, intensive agriculture, and overextraction of natural resources among other pressures considered by the UN Environment's Global Environment Outlook series reports (UNEP, 1997, 2000, 2002, 2007, 2012, 2019) have been producing cumulative negative effects on the environment. Coupled with climate change, extinction of species and loss of land and water biodiversity, create the conditions for ecosystem collapse, with catastrophic impacts to human development. Such collapse has been in the making for more than 100 years, with global population and economic output increasing by more than 368% and 7,172% respectively between 1900 and 2018 (Morgan and Fullbrook, 2019).

While there is increasing consensus that changes in the organization of human society and economy are needed to stop climate change and the degradation of the natural environment (Voulvoulis et al., 2022), and to avoid ecosystem collapse, based on two different normative ideals, economic growth and degrowth, the two main narratives put forward project opposing views of the relationship between economic growth and environmental protection (Sandberg, Klockars and Wilén, 2019). The interactions between economy and environment are extremely complex (Costanza et al., 1993; Rosser, 2001), and these two academic and political schools of thought (Raza, 2016) struggle to find common ground, looking at these interactions through different normative ideals and reference points. Indeed, the subject of economic growth is terribly polarizing, and a sterile debate between these communities, infused by austerity visions of degrowth (Davidson, 2000; Phillips, 2019) versus GDP-driven business as usual endless growth (Lietaert, 2010; O’Neill, 2012; Kallis et al., 2018), has led to the ongoing national and international political gridlock around sustainability issues. Moreover, there is an increasingly polarized political environment, with nationalist movements, socialism and climate activism rapidly growing in power worldwide (Dunlap, McCright and Yarosh, 2016; Conversi, 2020; Rekker, 2021).

"*Sustainability*", a term traced back to the 17th century (Estoque, 2020) as articulated by the Brundtland report (World Commission on Environment and Development, 1987) recognizes that human development is subject to the status of environmental systems and limited by finite resources utilized to satisfy current and future needs. Economic growth, expressed by an increase in real output has been empirically proven to negatively affect natural capital, through increased consumption of non-renewable resources, due to early-stage low technological progress (Dinda, 2004), higher levels of pollution, global warming through the production of greenhouse gases (Lapinskienė, Peleckis and Radavičius, 2015) and the potential loss of environmental habitats due to land-use changes and/or environmental pollution and degradation (Powers and Jetz, 2019; Tang et al., 2021). International scientific organizations such as the Intergovernmental Panel on Climate Change and the World Meteorological Organization agree that industrial, agricultural, and other human activities are the key drivers of climate change and environmental degradation (Pincheira and Zuniga, 2021), ultimately affecting human wellbeing and diminishing the capacity of the planet to sustain economic development. Moreover, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services report (IPBES, 2019) stresses that increasing anthropogenic pressures on ecosystems in the last 50 years have resulted in significant reductions in ecosystem services. To reverse these trends, the need for decisive action is dampened under the influence of multiple centres of power with own vested interests, while consumers' lack of awareness and limited knowledge of the impact of their actions and consequences of their decisions at different scales, creates resistance in even recognizing the extent of environmental degradation (Morgan and Fullbrook, 2019) and slows down departing from the status quo (Smith, Hargroves and Desha, 2010).

Considering that humanity has not reached a state of development with regards to life, mortality, and health but also standard of living, productivity, and poverty as well as education and freedom, that is neither desired nor enjoyed by everyone on the planet, the prevailing argument has been to sustain economic development while reducing its impact on the environment. Satisfying this is believed that can be achieved through environmental regulations, technology developments and increases in resource efficiency (Holdren, 2008; Conrad and Cassar, 2014; Fletcher and Rammelt, 2017), an idea extensively used by ecomodernists (Albert, 2020). "*Decoupling*" is a concept that implies that economic activities and their environmental impact can be separated, or that their link can be broken (OECD, 2002), or that their current relationship can be reversed (when economic activities restore nature). Among others, this concept has been adopted by several national and international institutions as a critical priority for sustainable development (Yu et al., 2017). For example, in the EU the Green Deal aims at creating a competitive, resource-efficient economy, where "economic growth is decoupled from resource use" (European Commission, 2019e). Additionally, the United Nations Environment Programme calls for "decoupling through maturation", meaning that the natural transition from an extraction to a service-oriented economy can reduce the intensity of its negative impact on the environment by means of increasing efficiency (von Weizsäcker et al., 2014). However, increasing resource efficiency might be a valid policy objective to decrease pressures on the environment but not sufficient to avoid collapse, as pressures imposed by economic drivers (e.g., economic development and population growth) can outweigh its effects. Parrique et al. (2019) argue that rising energy expenditures, rebound effects and inadequate comprehension of the system hamper the possibility of increasing growth without negative effects.

Poor progress in the practical application of sustainable development shows insufficient understanding of its challenges, stemming from a long tradition of pursuing solutions to complex issues either through a social or ecological perspective (Adetunji et al., 2005). On the one hand, reducing manmade pressures on the environment, through reducing growth may lead to severe (real or perceived) social consequences e.g., poverty increase or lowering people's standards of living. Economic growth is considered the most powerful instrument for reducing poverty and improving the quality of life in developing countries (DFID, 2007), with both cross-country research and country case studies providing overwhelming evidence that rapid and sustained growth is critical to most United Nations Sustainable Development Goals (SDGs) but particularly the eight first (poverty, hunger, health, education, equality, water, energy and decent work), as economic development can lead to higher income per capita (Adams Jr, 2013). Moreover, visions of sustainability as a "Simpler Way" society defined by low but sufficient material living standards (Trainer, 2010; Alexander, 2015), fail to inspire people to make the necessary changes for realising such visions, or even worse giving up altogether, if these are accepted as the only visions under which human civilization can operate viably on our finite planet (Trainer and Alexander, 2019). On the other hand, historical rates cannot justify the goal of sustaining economic growth and reducing its impact on the environment by means of resource efficiency improvements and environmental protection policies alone (Haberl et al., 2020; Hickel and Kallis, 2020), other than leading to chilling visions of a future where society becomes increasingly relied on technology, artificial intelligence systems or humanity moving to other planets.

To move the discussion beyond the current gridlock, the study assumes that there is another vision of sustainability, away from these two extreme visions of our future, that can realise with fundamental changes in the use of natural capital (Haberl et al., 2017) and the way humans interact with the environment. For this, the current chapter reviews the relationship between development and environmental degradation and revisits existing knowledge about them as derived from different disciplines (Soga and Gaston, 2021), looking at sustainability challenges from a systems perspective, and exploring the role of economics in the process. The purpose of this exercise is to reenergize discussions about sustainable development futures of increased prosperity that are desirable and can be delivered through sweeping environmental and economic radical changes and planetary-scale reforms.

8.2. Development and environmental impact

The systematic link between economic development and its effects on the environment has been receiving increasing attention in the last three decades (Dinda, 2004), aiming to elucidate how the different stages of economic development influence environmental quality. The Environmental Kuznets Curve (EKC) has been a common approach to investigating the link between environmental quality and economic growth. The EKC developed by Kuznets (1955) hypothesizes an inverted-U long-run relationship between pollution and economic development (Figure 8.1). Though some pollutants, such as carbon dioxide increase as per capita income increases due to for example higher use of vehicles and intensifying production, the majority of pollutants (suspended particulate matter, sulfur oxides, nitrogen oxides, and water pollutants) rise to a point as income increases and then they decline (Hill, Magnani and Hill, 1955). The main thrust of the EKC is that at the early stages of development, the intensification of industrialization leads to rapid growth at the expense of the environment and income equality. As the growth of income per capita continues it reaches an inflection point (Nkwatoh, 2022), beyond which people start valuing higher improved states of the environment, and through the adoption of regulatory instruments, increasing environmental awareness and improved technology, degradation slowly diminishes (Dasgupta et al., 2002). Grossman and Krueger (1991) identify three different effects that dominate the economic growth-environment relationship at the aggregate level. First, as economies move from agriculture to industrial and manufacturing processes, higher investments in manmade capital result in environmental damage due to increases in the use of natural resources. Second, economic growth is accompanied by a composition effect (Taylor and Copeland, 2004) that relates to structural changes in the economy as it moves from being heavily relied on agriculture, to gradually consisting of industries that produce tangible goods and finally reaching a stage where a great share of industries produce services. Finally, the technique effect relates to the progressive replacement of obsolete inefficient technologies and processes by cleaner technologies that reduce the effects of production on the environment.

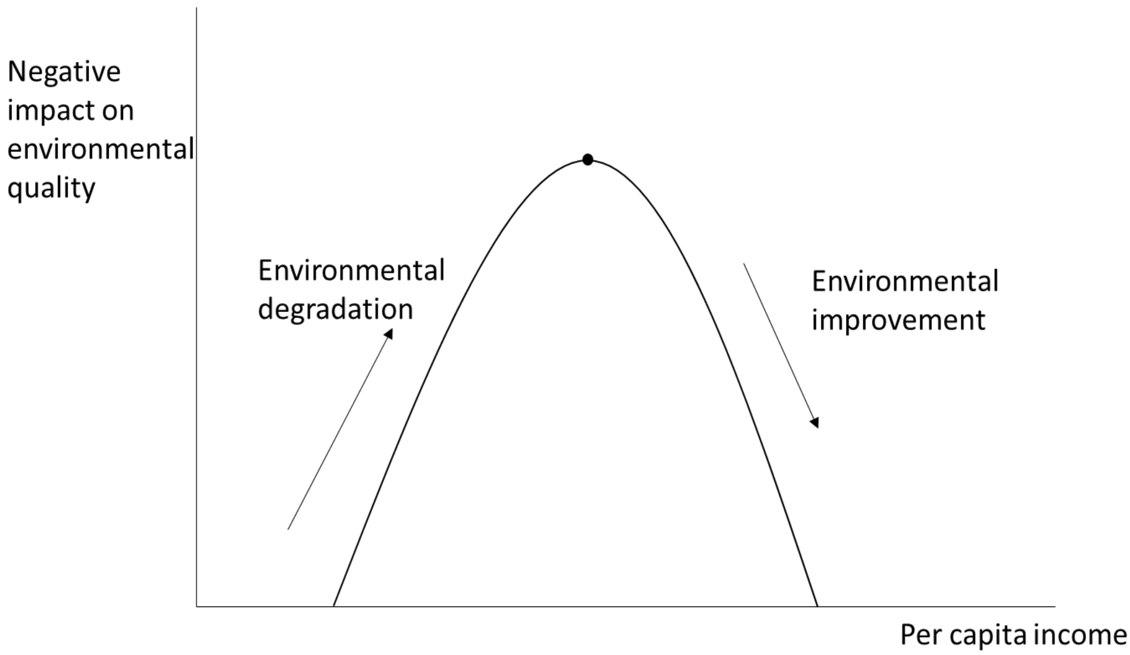

Figure 8.1 A typical representation of the EKC

Scholars have conducted various studies that among others use forest logging rate (Panayotou, 1994), suspended particulate matter (Selden and Song, 1994) and industrial water use (Gu, Zhang and Pan, 2017) as environmental variables to validate the EKC. In relation to water resources, the first study that validated EKC for water withdrawals was that of Rock (Rock, 1998) followed by others (Cole, 2004; Duarte, Pinilla and Serrano, 2013) that used cross-sectional data. In addition to that, the EKC has been found to represent the relationship between economic development and its effects on the environment for water withdrawal applications both in the industrial (Yang and Jia, 2005; Hemati, Mehrara and Sayehmiri, 2011) and agricultural sectors (Goklany, 2002; Bhattarai, 2004). Other studies that assess how water quality is influenced by development are that of Paudel et al. (2005) and Thompson (2014) that did find evidence of the relationship between the two, while Farzin and Grogan (2012) did not. Since then, and particularly with the development of the “natural capital” concept, and several studies investigating how it is impacted by economic activities, the EKC relationship has been challenged.

The term *Natural Capital* was introduced by David Pearce in 1988, and can broadly be defined as the quantity of natural resources and the ecological services they provided that when combined with manmade and financial capital result in the provision of marketed products and intangible benefits that satisfy human needs (Bateman and Mace, 2020). One such study is that of Wang et al. (2021) who by investigating the level of economic development and natural capital in China, found that the pattern of demand for the latter across regions varies significantly, depending among others on the industrial structure, population size and energy efficiency. Additionally, recent studies indicate that the curve may follow an N-Shape as in the study of Chuku (2011) about the income-environment relationship in Nigeria and in that of Brockwell et al. (2021) assessing the EKC relationship between income and water quality in twenty European countries, or an S-shape (Gangadharan and Valenzuela, 2001; Friedl and Getzner, 2003), while several studies do not find evidence of the EKC hypothesis (Stern and Common, 2001; Perman and Stern, 2003; ChienChiang, YiBin and ChiaHung, 2010).

Furthermore, the influence of several other factors, such as the intensity of foreign trade (Saidi and Mbarek, 2017), urbanization (Ozatac, Gokmenoglu and Taspinar, 2017), environmental patents (Cheng et al., 2019), institutional quality (Allard et al., 2018), finance (Nassani et al., 2017) and social variables such as social capital (Paudel and Schafer, 2009; Hao et al., 2020; Rahman and Alam, 2021) among others have been shown to determine environmental degradation. However, the effects of such variables are inconclusive, influenced by the heterogeneity across countries in terms of their level of development and income. For instance, Allard et al. (2018) conclude that trade increases $C0_2$ for lower-middle-income but not for high-income countries, while other studies conclude that trade openness reduced environmental pollution in countries such as China and India (Aydin and Turan, 2020). Other studies have been including the ecological footprint instead of pollutants as the explained variable, shown in some cases to be positively affected by economic growth (Alola, Bekun and Sarkodie, 2019; Destek and Sinha, 2020) (U-shaped curve), while in others the EKC hypothesis was verified (Ahmad et al., 2021). Furthermore, a body of research shows socio-political parameters playing a key role in determining the shape of EKC. For instance Farzin and Bond (2006) found a significant relationship between income inequality, age distribution, education, and CO2 emissions; Dutt (2009) included an index of socioeconomic conditions related to the levels of unemployment, consumer confidence and poverty, an index of education. In this study the high correlation of such variables with income did not show significant effects, however, their improvement could speed up improvement in environmental quality. Additionally, the human development indicator was included in the analyses of Farhani et al. (2014) and showed to positively influence CO_2 emissions.

The EKC has also been criticised for using aggregate data and disregarding microeconomic information stemming from non-market valuation (McConnell, 1997). Besides structural effects, behavioural factors that influence individuals' choices on environmental services influence the economic development-environment relationship (Panayotou, 2000). Models developed to study the micro foundations of the EKC show that low income and consumption in combination with increased environmental endowments lead to increasing environmental damage (Murty, 2003; Pfaff, Chaudhuri and Nye, 2004) at least for low incomes. Ma and Shi (2014) using a static model explain that at low-income levels, individuals perceive pollution caused by economic growth as acceptable as they are more concerned with wellbeing stemming from the consumption of produced goods. Additionally, given limited financial capital at such stages, investments for improving environmental quality are not favoured. Furthermore, at the micro level, the relationship between environmental degradation and economic growth has been studied, though not extensively, under the prism of the Environmental Engel Curve framework (Baudino, 2020) that incorporates socioeconomic characteristics, commonly hypothesised to affect household behaviour (Sager, 2019; Borghans et al., 2021). Critiques raised against the EKC extend to this framework and include among others the issue of neglected reverse causality, i.e., bi-directional causality between income/growth and pollution (Baudino, 2020). From a theoretical perspective, the greatest share of research disregards feedbacks from environmental degradation to economic output, by specifying the income variable as exogenous (Stern, 2003). Contrary to that, Barassi and Spagnolo (2012) examining the causal relationship of per capita $C0_2$ emissions and output growth for six countries (Canada, France, Italy, Japan, the United Kingdom, and the United States), found feedback between the two, revealing that economic growth may not only be the cause of pollution, but also the result of it.

While one might be tempted to question the validity of all these studies offering contradictory findings, the main problem seems to be with how the EKC has been applied among economists to model the connection between development and its environmental impacts, as a relationship that is fixed, predetermined and unconditional. The EKC simply models an essentially empirical phenomenon (Stern, 2004) and is only correct when it happens that the actions and policies on the ground support and deliver what it claims to predict. Concentrations of some local pollutants have clearly declined in developed countries when the right policies were introduced (Stern, 2017), but emissions of many pollutants have increased in the absence of such policies (Hoang et al., 2019). Studies of the relationship between per capita emissions and income find that per capita emissions of pollutants rise with increasing per capita income when other factors are held constant if these are not targeted. Moreover, even changes in these other factors may be sufficient to reduce pollution if these complex interactions are understood and managed appropriately. For example, in rapidly growing middle-income countries, the effect of growth overwhelms the effects of other factors (e.g., production efficiency, state of technology, input mix), while in wealthy countries, growth is slower, and pollution reduction efforts can overcome its effects (Stern, 2018) through reshaping the interactions among human capital, technology, production and consumption among others (Song et al., 2021). Such evidence reinforces the claims that environmental problems should not be expected to be eliminated through simply achieving higher economic growth, but through targeted interventions and policies (Arrow et al., 1996) able to take into account the complex relationships of a socio-ecological system.

8.3. Human nature interactions and sustainable development

Human-nature interactions, and particularly our relationship with nature have evolved over time. From the ancient Greek notions of cosmology that viewed the natural world as one unified organism (Furley, 1987), with humans being a factor contributing to the organism's overall functioning, to the Renaissance, when modern scientific thinking began to take shape, seeing the natural world as a machine (Oakley, 1961), and humans as being located outside of nature. While contemplating nature from the outside (Parisi, 2000), scientists believed that objective observation and controlled experiments could decode the workings of nature. Environmental management followed a similar trajectory, with passive strategies implemented in the beginning of the industrial revolution viewing the environment as being able to absorb wastes generated from production and consumption activities, with end-of-pipe technologies treating pollution at the end of production processes (Mengist, 2020). Despite evidence of low performance, such approaches are still being implemented today, not as a result of concrete analysis and data, but rather based on "how we are used of doing things" (Boeuf, Fritsch and Martin-Ortega, 2018). Similarly, when economic analysis is employed, it often mainly revolves around financial costs, disregarding environmental and resource costs and benefits (Souliotis and Voulvoulis, 2021a). Reductionist approaches, under the assumption of certainty and predictability, are shown to fail to address complex problems (Gorzeń-Mitka and Okręglicka, 2014). Sectoral planning that does not consider the wide range of effects of policy interventions, for example, often fails or results in short term improvements that do not improve wellbeing (Souliotis and Voulvoulis, 2021b). Indeed, most traditional approaches to economic and environmental management have been based on static, compartmentalized models that through mechanistic approaches often fail to understand the complex relationship between human societies and the natural world. Moreover, the study of social, economic, and ecological domains was traditionally performed within disciplinary boundaries the same way scientific knowledge in ecological and social sciences has been developing independently (Ostrom, 2009). This minimal interaction between natural and social sciences (Rosa and Dietz, 1998) has led to neglecting the importance of ecosystems as unities with different socioeconomic, environmental, biological, chemical, and other characteristics (Liu et al., 2007) and to tackling concurring interlinked challenges in silos (Haberl et al., 2019). A systemic understanding of nature-human interactions, on the contrary, offers a more integrative view of the one and same system, where humanity and nature constantly interact by exchanging energy, information, and materials (Rees, 2019). Consequently, defining goals requires understanding of how processes that take place in one part of the system affect the status of the entire system. Examining how changes occur in the system, for example, shifts from low pollution-low socioeconomic costs to high pollution-high socioeconomic costs stages, calls for integration of disciplines and the development of interpretative frameworks that focus on the interactions of components rather than the components as outcomes. Ecosystems and industrial systems are tightly coupled and dynamic systems, which often operate far from equilibrium and exhibit nonlinear and sometimes chaotic behaviour. Systems thinking recognizes that our economies are subsets of their environments, and instead of viewing the world as a collection of unconnected objects, allows us to see reality as a nested holarchy of interacting systems (Taylor, 2009). The linkages between natural and economic systems exhibit complex threshold effects (Folke et al., 2002), dangers of irreversible damages, and interactions between global changes and place-based, location-specific effects.

Complexity and nonlinear dynamics are areas of important recent innovations in the natural sciences, that pose a challenge to standard economic models (not yet been fully absorbed) (Farrell, 2019; Spangenberg and Polotzek, 2019). Systems-based views, values, social structures, technologies, and economic processes are rapidly emerging. They describe a different worldview, where humans and ecological systems interact, impacting one another and co-evolve over time (Quintas-Soriano et al., 2018). This systemic interpretation of the relationship between humans and nature is now becoming the cornerstone of integrated environmental management policies (Kelly et al., 2013). For instance, the leading policy instrument to manage water resources in the EU, the Water Framework Directive, by defining each river catchment as the system of interest (Voulvoulis, Arpon and Giakoumis, 2017) calls for understanding the interactions of society with water resources, the trade-offs between economic benefits and water status classifications and designing interventions that take into account socio-environmental sustainability considerations.

Another prominent issue that dominates the relationship between humans and nature relates to structural aspects and the mechanics of the socioecological system. Conceptualising nature as natural capital (Pearce, Markandya and Barbier, 1989) and the benefits humans obtain from interacting with the environment as ecosystem services has not only promoted the development of systemic socio-ecological approaches especially in the field of environmental and ecological economics (Sullivan, 2014), but has become synonymous with environmental care (Carver and Sullivan, 2014) by shedding light on the nexus between the satisfaction of human needs and protection of nature (Prugh, 1999; Daly, 2019). In line with this, the notion of critical natural capital (Turner, 1994; Pearce, 1997; Ayres, 2007) signifies the limits to utilizing nature for sustaining production and consumption. According to it, natural capital performs environmental functions that cannot be replaced by other types of capital (Dietz and Neumayer, 2007; DesRoches, 2019). Preserving some natural capital stock is vital for maintaining the provision of ecosystem services (Rudolf De Groot et al., 2003; O'Neill et al., 2018). Therefore, such a concept denotes the lower level of stock of natural capital, below which ecosystems malfunction and some ecosystem services seize to exist, with negative socio-cultural, ecological, sustainability, ethical, and economic consequences (Brand, 2009). For instance, as Ekins (Ekins, 2003) presents, the long-term heavily polluted state of the Trent River in the UK and modifications in its flows, made the water of the river unsuitable for human consumption and significantly reduced the wildlife and biodiversity previously supported by the river.

The popularization of the concept of ecosystem services, commonly attributed to the publication of the Millennium Ecosystem Assessment (Millenium Ecosystem Assessment, 2005) and its adoption by various disciplines has been generating integrated tools and approaches that enable policymakers to account for the reciprocal relationship between humans and nature in relevant decision-making processes. Ecosystem services have been used extensively among others in land management (e.g., Metzger et al. (2006); Bateman et al. (2013)), agricultural policy (e.g., Zhang et al. (2007); Bauer and Johnston (2013); Palm et al. (2014)) and climate change research (e.g., Mooney et al. (2009); Munang et al. (2013)). Additionally, economics has incorporated ecosystem services into valuation techniques (Ghermandi et al., 2010; Costanza et al., 2014; Koundouri, Giannouli and Souliotis, 2016; Koundouri et al., 2017), which have claimed to improve relevant analyses by considering a broader spectrum of positive and negative interactions within the socioecological system (Souliotis and Voulvoulis, 2021a).

Economics is often criticized for adopting a narrow definition of the economy as a system (Goodwin, 2019), leading to considering social, economic, and environmental impacts stemming from human activities as external effects (externalities), often disregarded in economic analysis, even when having a measurable footprint (Beaton and Maser, 2011; Unerman, Bebbington and O'dwyer, 2018). Monitoring natural capital through the development of use, extent and ecosystem services flow accounts aims to provide information on the status of the environment, the dependence of the economy and society on natural resources, promote their sustainable use and reveal the broad effects of policy interventions (Souliotis and Voulvoulis, 2021b). Natural capital accounting, a methodology promoted by the

United Nations (United Nations- Statistics Division, 2013) has been mobilized to reveal how economic activities and policy interventions influence nature and consequently the wealth of a nation. The metaphor of nature as natural capital and consequently its valuation may indeed shape development goals following the logic that extensive exploitation of natural capital resources beyond their critical levels, reduces welfare in the long-term (Ulgiati, Zucaro and Franzese, 2011) both by losing intangible benefits (e.g., recreational and health benefits) and inputs for sustaining production, leading to 'uneconomic' development. In other words, a systemic view of the human-nature relationship considers (external) effects inherent to the system (Vatn and Bromley, 1997), relating them to its structure which accommodates a specific configuration of interconnections, ultimately reframing the notion of sustainability and shaping its normative goals (Gibbons, 2020). The fundamental premise of policymaking is to intervene in the system (Meadows, 2009) in such a way that the flow of information and materials ensure "sustaining life-enhancing conditions" (Reed, 2007) rather than achieving specific targets in different domains (Du Plessis and Brandon, 2015; Robinson and Cole, 2015), that often do not work or produce unexpected outcomes.

Human-nature interactions may produce positive economic outcomes when natural resources are used at a smaller rate than their rate of self-replenishment (Bierkens and Wada, 2019; Bateman and Mace, 2020); negative in the opposite case; or positive socioecological effects when human activities result in further enhancing the ability of ecosystems to produce services (Blignaut, 2019). The latter demonstrates the potential of natural capital regeneration as a vessel for economic growth through decisions that influence the properties of the system towards thrivability (Du Plessis, 2012; Hes and du Plessis, 2014; Gibbons, 2019). This is promising particularly considering that most research concerned with the relationship between economic development and environmental degradation often disregards the regenerative ability of nature (Bertinelli, Strobl and Zou, 2008), which constitutes a significant aspect of the resilience of ecosystems as complex adaptive systems (Adger et al., 2005). Policy decisions directly or indirectly can influence ecosystems and their processes positively or negatively, ultimately affecting their regeneration ability (Seddon et al., 2016).

Loss of resilience in socio-ecological systems, and their operation near tipping points, where rapid shifts occur, has been observed to be followed by slow recovery from shock, as the effects of positive feedback loops are of higher magnitude than the stabilizing effect of negative ones (Bueno, 2012). Therefore, improving the regenerative ability in such systems, safeguards that the overall system will operate away from critical conditions that may lead to its destruction. Sustainable development that takes into account the regenerative aspects of socio-ecological systems, aims to create conditions for development through restoring the health of the system (Clegg, 2012). According to Du Plessis (2012), sustainability in this respect considers that nature and humans are one autopoietic system, which requires focusing on understanding how nature works and base development on that rather than on developing processes to control ecological functions. Consequently, policy decisions in the form of technical, economic, and legal interventions must support the health of the entire system (Gibbons, 2020), ultimately resulting in the protection and regeneration of natural capital to support human welfare.

8.4. Policies for sustainability transformation

Reversing the trend of environmental degradation and reaching sustainability requires intentional transformation of technology, social practices, societal norms, policy instruments and business models (Voulvoulis et al., 2022). Internationally, the SDGs and in Europe, the Green Deal and 2030 EU Biodiversity Strategy aim at addressing interlinked environmental, societal, and economic challenges. Sustainability transformations consist of “fundamental changes in structural, functional, relational, and cognitive aspects of socio-technical-ecological systems that lead to new patterns of interactions and outcomes” (Patterson et al., 2017), following a vision of a sustainable society and actions to realise it (Holmberg and Larsson, 2018).

Achieving that requires the adoption of structural, systemic, enabling approaches or a combination of the above (Scoones et al., 2020). For instance, degrowth theory prescribes structural changes that relate to changes in the production and consumption practices, whereas decoupling in a growing economy paradigm is associated with systemic approaches of transitioning to different system states through ‘niche’ innovations and transforming the rules that govern the interactions of different components (Geels, 2005; Smith and Raven, 2012; Späth and Rohracher, 2012; Köhler et al., 2019). From a systems perspective, these approaches could ultimately converge to a similar sustainable future. Additionally, enabling approaches focus more on factors that create the capacity of individuals to take action and collectively shift the system towards states that correspond to their values (Scoones et al., 2020).

Tackling environmental pressures through a systems thinking perspective requires managing authorities to take into account the environment at large (Jager et al., 2016), evaluating how each sector of the economy interacts with the environment, and assessing the various economic, aesthetic, cultural, emotional, and environmental dimensions of natural ecosystems

(Hellegers and Davidson, 2021). Therefore, while it could be claimed that such a tool could lead to “putting a price on nature” (Pavan Sukhdev, 2012), deepening our understanding of the functioning and outputs of nature as well as assessing how actions affect the environment and in turn, human welfare, provides opportunities for identifying sources of welfare and growth. In fact, it is claimed that natural capital and its services, directly and indirectly, generate $44 trillion of economic value each year (White et al., 2020). Reporting on stocks of natural capital, and not just flows of its services reinforces socio-ecological transformation by influencing spending options (Bateman and Mace, 2020) towards desired outcomes for humans and nature, that may reveal opportunities for transitioning from a high to a low environmental impact economy. Nevertheless, sustainability transitions will not occur simply by following a methodological approach to assess human-nature relationships, but through large-scale social, political, and behavioural changes that can ensure the reshaping of human-nature interactions and consequently minimize negative impacts. For instance, Klingert (1998) suggests that environmental improvements require radical dematerialisation. Others (Maxwell, Sheate and van der Vorst, 2006; Baines et al., 2007) claim that “servitization”, i.e., the integration by manufacturers of service elements into physical products (Szász and Seer, 2018), can contribute to the reduction of environmental impact. Consequently, shifting to a more sustainable paradigm requires a new vision of prosperity, which will require radical policy changes both at a micro and a macro scale.

Closing the gap between the current and a desired state where pressures are minimized, calls for reversing the downward trends of environmental quality through decoupling opportunities and increasing the regeneration capacity of ecological systems (Figure 8.2). Disentangling economic growth from resource use and negative environmental impact, a key component for increasing resilience, given lock-ins and rebound effects (York, 2006; Haberl et al., 2017) can only happen through systemic sustainability transformation on the premise that fundamental changes in production and consumption (regarding, for example, the type of inputs, technology, and followed processes) lead to rebalancing of socio-ecological systems (similar to the structural changes proposed for degrowth). Regeneration, on the other hand, proposes investing in policy measures that increase biodiversity and natural capital and through that increase socioeconomic benefits from ecosystem services, an option that has not been explored exhaustively yet. Indeed, the OECD estimates global biodiversity finance at USD78–91 billion per year from 2016-18 (OECD, 2020), whereas Seidl et al. (2020) estimate that the annual public biodiversity expenditure was 0.19–0.25% of global GDP over the past decade, noting that a higher volume of investment is needed for reducing pressures on biodiversity and promoting its conservation and sustainable use.

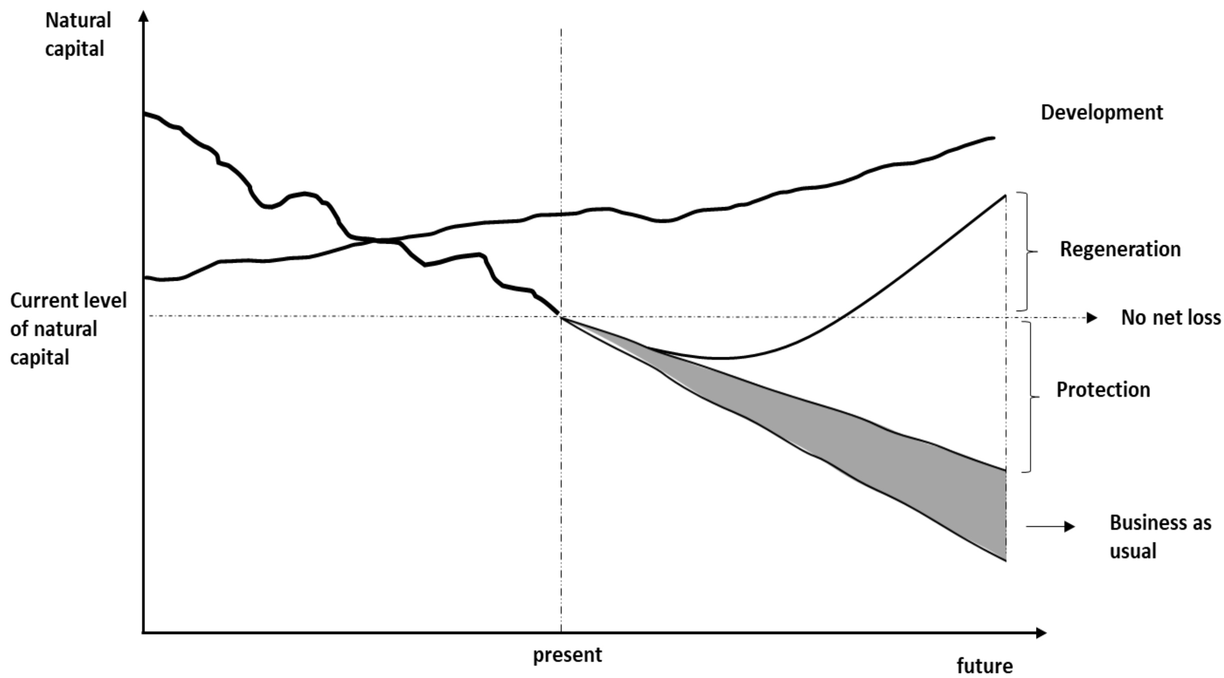

Figure 8.2 Schematic representation of the proposed objectives. Adjusted from Natural Capital Committee (2015)

Research suggests that (assisted) natural regeneration of degraded ecosystems is able to sequester significant amounts of CO_2 (Brown et al., 2011), protect against flooding (Kelly et al., 2016) and increase resilience against the effects of climate change among others (Chausson et al., 2020). Furthermore, natural capital can increase through deliberate investments in replenishing habitats for species and restoration of ecosystems (Segura and Boyce, 1994; Hinterberger, Luks and Schmidt-Bleek, 1997). Regeneration investments result in significant benefits. For instance, an investment of 1 million USD in aviation can result in the creation of 19 jobs, while the same amount can generate almost 40 jobs if invested in reforestation, land and watershed restoration and sustainable forest management (Edwards, Sutton-Grier and Coyle, 2013), that also deliver other health and welfare benefits through ecosystem services generation rarely accounted for, explaining conventional investment in grey infrastructure. In relation to this, decoupling studies have often been criticized for using GDP as the measure of the outcome of the economy, with scholars advocating for the use of welfare indicators instead (Beça and Santos, 2014; Bleys and Whitby, 2015; Menegaki and Tugcu, 2016). For instance, Kalimeris et al. (2020) note that using GDP as the index of economic welfare provides an optimistic vision of the dependence of economic development on the environment, given that empirical estimates show a higher degree of decoupling.

Mainstreaming investments in natural capital face barriers related to institutional failures in the sense that users reap benefits whereas policymakers face their costs (Turner and Daily, 2008); undervaluation of benefits that reinforces free riding; and lack of information regarding the distribution of benefits among users (Vogl et al., 2017). New approaches such as nature-based solutions (NbS) (Singhvi, Luijendijk and van Oudenhoven, 2022), have the potential to overcome such issues, designed to provide additional benefits besides those directly related to minimizing identified environmental pressures. NbS loosely defined as interventions that operationalize the functioning of nature to reduce pressures on the environment, while generating a wide range of socioeconomic benefits have been shown to be cost-effective (Souliotis and Voulvoulis, 2022) and have the potential to attract private investments (Loiseau et al., 2016; Sutton-Grier et al., 2018; Kok et al., 2021). These initiatives can involve protecting, restoring, and managing existing ecosystems or creating new ones to maintain biodiversity and its functioning and/or enhancement to alleviate negative impacts on the environment (Cohen-Shacham et al., 2016; Rodriguez-Gonzalez, Rico-Martinez and Rico-Ramirez, 2020) while addressing social and economic challenges (Faivre et al., 2017). The emergence and widespread recognition of the significance of NbS have been heavily influenced by the concept of ecosystem services (Hanson, Wickenberg and Alkan Olsson, 2020) and the theory of systems thinking (Keesstra et al., 2018). Positive and negative feedback loops inherent in natural systems, and their ability to adapt to their environment (Cropp and Gabric, 2002) are key elements in properly designing such activities. NbS are considered to provide multiple benefits, such as offsetting greenhouse gas emissions, removing water and air pollutants, as well as recreational and health benefits (Joscha, Michael and Axel, 2015; Liquete et al., 2016; Kabisch, van den Bosch and Lafortezza, 2017; Raymond, Frantzeskaki, et al., 2017). Additionally, NbS are strongly associated with benefitting biodiversity, either through increases in the diversity and/or populations of species, and the improvement of habitat quality and/or community composition (Chausson et al., 2020), a necessary condition for the resilience of the system (Seddon et al., 2021).

Shifting from reductionist to systems worldviews and thinking, not only shapes policy objectives, but also the means to achieve them. Instead of asking "What is the optimal level of growth that does not lead to environmental degradation?", managing complex environmental interactions through systems thinking would pose the question "What interventions could we undertake to influence the interactions among the society, economy and nature in such a way to reach a desired state?". In other words, under such a worldview, the emphasis is given to interactions between components, that give rise to properties. In the context of decision-making, integrated economic assessments that capture a wider range of social, health, environmental and economic costs and benefits can translate these interactions into a common currency, improving understanding of how different components of the system are interrelated. Transitions from the current to a desired state involve extensive changes that relate to a broad range of actors (Markard, Raven and Truffer, 2012), which inevitably reform the economy as a system[19]. Government agencies play a significant role in guiding visions of transitions (Späth and Rohracher, 2012) through policies, regulations, and funding of environmental programmes. Practical approaches, such as economic valuation and cost-benefit analysis inform such decisions. However, as it is often argued, neglecting the full spectrum of ecosystem services benefits leads to and low awareness of the importance of nature and consequently to mismanagement (Neill, O'donoghue and Stout, 2020). Systemic accounting tools that track information on the stock of natural capital, the ecosystem services it provides and their value to humans are essential for providing direction to systemic changes and assist in detecting signs of increased pressure in the system (Barnosky et al., 2012; Galli et al., 2012). Economic valuation thus serves the role to communicate the magnitude of interactions between components of the system in a common unit of value (Kemp-Benedic and Kartha, 2019). Additionally, unravelling how preferences transform within a system (Fischhoff, 1991), reveals patterns of behaviour and structures enabling us to move away from those that do not serve us well. Consequently, in this type of world, by accepting that interventions are not only associated with costs, but the reduction of pressures creates beneficial interactions with different components of the system, promotes diverting public investments towards generation of benefits[20], which might further favour the role of NbS for increasing regeneration in the system.

By putting humans back in nature and treating the human-nature interface as the one system where the fates of humanity and nature are intertwined, enables the emergence of a truly sustainable world. Systems thinking allows us to look far into the future, think beyond ourselves about the greater collective (born and unborn, human, and non-human), and look deeper below the surface to understand how things really work, and not just for avoiding ecosystem collapse but ultimately creating conditions of prosperity for all.

[19] I follow the argument of Norgaard (Norgaard, 2019) that describes the economy as a system consisted of values, knowledge, technology, social organization.

[20] Gomes and Barros (2022) explain that the return of private companies of investments in environmental technologies are less than the social benefits, which requires governments to provide the conditions to mitigate this issue.

8.5. Discussion

Several theories exist that mould the strategies societies must follow to achieve sustainable development. Green growth, degrowth and a-growth are mostly discussed in Europe (Lehmann, Delbard and Lange, 2022), whereas the steady-state economy proposed by Daly (1973) is a concept used more widely in North America (Martínez-Alier et al., 2010). Degrowth, at one end of the spectrum, treats natural capital and ecosystem services as un-substitutable by other forms of capital, with their intrinsic superior to their instrumental value (Gabriel and Bond, 2019). Proponents of degrowth claim that sufficiently reducing impacts to levels that can ensure ecological resilience and increased well-being cannot be accompanied by increasing economic growth (Kallis et al., 2018). Consequently, this means that enhancing ecological conditions requires downscaling of consumption and production (Schneider, Kallis and Martinez-Alier, 2010; Krpan and Basso, 2021), and as a result, a reduction of GDP. The theory of a-growth rests in the middle of the spectrum, largely influenced by the works of van den Bergh (Lehmann, Delbard and Lange, 2022), primarily aiming at developing effective policies for the protection of the environment that are socially acceptable, without necessarily attempting to achieve specific economic development objectives (Bergh, 2010, 2017). At the other end of the spectrum, proponents of economic growth advocate that growth remains essential for supporting continued improvements in factors that affect people's wellbeing, from health and employment to education and quality of life, and for helping governments deliver on a range of policy objectives, amongst them environmental ones (Everett et al., 2016), as well as investing in the development of more efficient technologies that are able to minimize the impact of production and consumption on the environment (Ekins, 2002).

Somewhere in between there is also a rather misunderstood concept referred to as "green growth" or more specifically the concept referred to as decoupling - decoupling of economic growth and ultimately of our prosperity from resources, pollution, waste, and carbon emissions. More of a victim of the rivalry between the two above extremes, opponents from the one side, claim that green growth cannot be achieved without jeopardizing economic growth (Fernandes et al., 2021); while the others argue that it is not possible to respect sustainability if intensive consumption of goods continues to foster economic growth. Still, decoupling is a foundational component of the UN 2030 agenda of Sustainable Development Goals, and specifically a target of SDG 8 on sustainable economic growth. Target 8,4 refers to the need to "improve global resource efficiency in consumption and production and decouple economic growth from environmental degradation. Indeed, dematerialisation, servitisation, collaborative consumption and a shift from ownership to access have the potential to restructure the economics of consumption, accelerate decoupling, and help us to envision and potentially create a circular economy that delivers social, economic, and environmental benefits for all (Voulvoulis, 2022).

Proponents of degrowth, such as Hickel and Kallis (Hickel and Kallis, 2020) claim that despite economies shifting from manufacturing to services and the development of possible technological innovations which will decrease the dependence of the economy on natural capital, absolute decoupling is not likely to occur. Furthermore, they advocate for limiting economic growth within sustainable ecological limits through structural changes in production and consumption while increasing human welfare (Latouche, 2009). The conditions though under which a state of degrowth can be achieved have not been adequately investigated and such a vision has not been proven practically possible (Sandberg, Klockars and Wilén, 2019). On the other hand, a counter argument on reducing growth could be that of van Krevel (2021) who claims that policies that may result in the depletion of natural capital for the purposes of economic growth promote sustainable development through the generation of manmade capital assets that increase the per capita Inclusive Wealth[21]. Decoupling finds itself between two extreme schools of thought and a current debate on sustainability challenges focusing on the optimal level of growth following a long empirical tradition of associating the development of socioeconomic variables to environmental impacts.

From a systems perspective, economic growth is not simply the result of intensified use of natural resources, but rather the result of a series of interactions that take place simultaneously (social, cultural, institutional etc.). Therefore, pursuing some sort of transformation either through technological progress to promote the efficient use of natural resources, or through reducing economic output to maintain environmental integrity are ill-thought visions that provide neither holistic objectives nor the means to achieve them (Jakob and Edenhofer, 2014) and disregard a large number of parameters that influence both growth and environmental integrity. Sustainability challenges are complex given the high number of agents, interactions, and feedbacks that socioecological systems encompass. Instead of reducing complex sustainability issues to manageable problems revolving around the level of growth, falsely leading to the belief that socioecological systems can be controlled, policymakers, academics, and the society as a whole need to focus on how to harness or influence (Mueller, 2020) such systems towards a vision of prosperity that goes beyond growth, as well as beyond efficiency and recycling, delivering prosperity sustainably.

[21] The study is based on the idea of weak sustainability (Solow, 1974, 1986, 1993; Hartwick, 1977, 1978, 1990) that does not account for ecological sustainability.

Individuals obtain benefits not only by directly consuming manufactured goods, expressed in GDP terms, but through a broad range of services provided by the environment, natural or manmade as infrastructure, green or grey, that need to last long to deliver those services. Changes in the production and/or provision of ecosystem services, either ignited by human activities or shocks affect human wellbeing. To elucidate that, a recent study by the River Trust finds that recreational fisheries in England's freshwater bodies alone provide economic benefits of more than £1.7 billion per year (The Rivers Trust, 2021). However, currently, 93% of principal salmon rivers in England are assessed as being at risk due to urban, industrial, and agricultural pollution (Environment Agency, 2020), which, if not reduced, might lead to diminishing market and non-market benefits. Consequently, a goal towards sustainable development would be to eradicate pressures that increase the risk of losing ecosystem services. Such an argument, however, requires caution. For example, agriculture constitutes a significant sector for the production of food, while it is identified as a leading driver of river eutrophication, land use changes, depleting water tables etc. (Stoate et al., 2009; Monaghan et al., 2013; van Vliet et al., 2015). Reducing agricultural production for the sake of the environment would potentially decrease food security. On the contrary, measures to mitigate pollution such as Nitrate Vulnerable Zones (NVZs), agri-environment schemes (Environmental Stewardship and Countryside Stewardship), and the Catchment Sensitive Farming (CSF) partnership implemented in the UK (Jones et al., 2017), might be proven effective in satisfying dietary needs at a lower environmental impact. From a policy perspective, resource efficiency may be improved through deliberate efforts, without the need to forego economic growth, though technological, institutional, and behavioural transformation may be required as the appearance of rebound effects (Shao and Rao, 2018; Joyce et al., 2019) may cancel out any benefits that may result from decoupling opportunities.

Policy decisions either directly or indirectly affect natural capital and the regenerative capacity of natural ecosystems, which in turn influence economic performance (Borucke et al., 2013). Increasing the capacity of the system to respond to disturbances necessitates radical societal changes (Olsson, Galaz and Boonstra, 2014). A transformation towards sustainable development requires both monitoring such parameters as well as incorporating such considerations into day-to-day decision making. Furthermore, the type of implemented investments is crucial as it determines the path that the system follows from the present to the future. Time delays in systems mean that an intervention may influence different long-run and short-run responses (Sterman, 2015). However, policy interventions are often myopic, prioritizing short-term benefits over long-term successes (Goodwin, 2019; Mayor et al., 2021; Toxopeus and Polzin, 2021), and follow reductionist approaches that promise to provide easy solutions to complex problems. Positive and negative effects that are generated by and unfold in the system are frequently seen as static and external (Carlaw and Lipsey, 2002; LeSage and Fischer, 2012; Sahdev, 2016) often being disregarded from relevant economic analyses reinforcing convictions of system equilibria. In that regard, economics plays the role of promoting understanding of how system properties emerge through concepts related to socioeconomic values (e.g., wellbeing, preferences, benefits, costs, natural capital accounting), and based on that assist in shaping policymakers' aspirations, contributing to moving away from the "mechanical application of generic rules" (Scott, 1998). In line with this, as Mueller (2020) notes, in order to decrease policy failure, we must opt for those that are "immune to specific problems" created by complexity, meaning actions that do not rely heavily on interventions from policymakers, their design emerges from the bottom up and are able to accommodate the preferences of stakeholders.

In recent years, cost-effective systemic solutions have been gaining increasing currency, currently forming a paradigm of 'working with nature' (European Commission, 2020e). Nature-based solutions, the leading example of such approaches, demonstrate a new norm of environmental management that aims to address economic and societal challenges, while tackling the global environmental crisis (Maes and Jacobs, 2017). In essence, using nature-based solutions entails a paradigm shift, as it requires abandoning the dichotomy between nature and humans, and generating evidence to increase trust in natural processes (Fernandes and Guiomar, 2018) and the potential of tailor-made approaches to tackle complex socio-environmental issues. An emerging body of research points out that such alternatives can be cheap to implement, with the accruing value of benefits significantly overshooting costs. For instance, Souliotis and Voulvoulis (2022) show that a constructed wetland was able to enhance the quality of water discharged from a recycling centre, creating, and supporting new habitats, at a cost 5 times lower than the installation of new filters to the treatment facilities considered as the alternative.

Still, to truly harvest the benefits of decoupling opportunities through regenerative investments a new paradigm of management is required. Currently, prices and not value determine the selection of policies (Adam, 2014). Therefore, there is an urgent need to broaden the spectrum of costs and benefits that feed into economic analyses, through quantification and mapping of ecosystem services (Egoh et al., 2008; Tallis and Polasky, 2009; Villa et al., 2009; De Groot et al., 2010; Willaarts, Volk and Aguilera, 2012) and the development of the associated natural capital accounts (Edens and Hein, 2013; Sumarga et al., 2015) to monitor the flows of goods and services of nature as well as their value (La Notte et al., 2017). Furthermore, assessments need to focus on specific contexts (environmental, cultural, socioeconomic) and scales (local, regional, global) to account for heterogeneity (Hasse and Krücken, 2012) between systems.

Besides that, selected policy objectives need to be in accordance with the specific characteristics of the system, its status, and the way it interacts with systems of lower or higher levels (Gunderson and Holling, 2002). Finally, understanding how decision outcomes are valued by stakeholders is a key issue in setting objectives and achieving sustainability (Rammel, Stagl and Wilfing, 2007). Participation may bring to light conflicts among heterogeneous groups of stakeholders, information on the natural environment and its history of changes, as well as promote the acceptance of policy prescriptions (Santos et al., 2006; Bijlsma et al., 2011; Beyers and Arras, 2021). Addressing socioenvironmental challenges that "sit between science and society" (Surridge and Harris, 2007) calls for structural changes and a transition towards integrated approaches (Macleod, Scholefield and Haygarth, 2007; Claudia Pahl-Wostl et al., 2008; Jager et al., 2016), based on a better understanding of human-nature interactions and a long-term vision of the socioecological system realised by strategies that promote its longevity and prosperity.

8.6. Conclusions

Despite significant advancements in a wide range of disciplines, the development of interdisciplinary frameworks, and increased awareness of the status of the environment, our society still lacks understanding of the processes that take place within socioecological systems and a vision for its future state of sustainability. Opposing schools of thought and heated discussions have created polarization concerning the means to achieve sustainability.

Through assessing dominant empirical methodologies and their results when looking at the relationship between economic growth and environmental degradation, the study argues that the mainstream way of analysing the connection between them is oversimplistic and disregards the complexity that is inherent in socioecological systems. Aiming to contribute to the discussion on sustainability challenges, the study raises the need for a new vision of sustainable development that views the world from a systems thinking lens and proposes radical transformation of the processes that take place within it through better understanding the mechanics of the system. To achieve this, economics can play a crucial role in changing society's perception about the relationships among components of the system, and guide decisions towards a new vision of prosperity.

9. Overall Discussion

9.1. Learning how to improve environmental management decisions from implementing the Water Framework Directive

More than 20 years after the introduction of the Water Framework Directive, and after the Member States invested considerable effort in managing freshwater resources sustainably, significant pressures and their subsequent effects on society are still present, posing a threat to achieving sustainable development. During that period, governance processes were redefined through the establishment of new management structures (i.e., the river basin district management authorities) at the catchment scale, responsible for coordinating and overseeing the implementation of the Directive. Among others, their responsibilities include monitoring natural and socioeconomic characteristics of each river catchment and gathering of relevant information to be used for the drafting of the river basin management plans.

The purpose of such a process has been to understand the initial conditions of the catchment, determine the desired state, identify the gap between the two and, based on the specific structure and functioning of the system, design and implement interventions that would ensure reaching the desired state. Furthermore, through a long-term iterative and experimental process the Directive aimed at generating knowledge (Hering et al., 2010) and addressing such environmental issues through a thorough investigation of the effects of multiple pressures on water resources; improving the assessment and monitoring of the ecological status of waters across countries (Birk et al., 2012; Voulvoulis et al., 2017) and the dynamics of hydrological and biochemical processes (Hamilton, 2012); and enhancing the identification of pressures (Vigiak et al., 2021) and participatory forms of water governance among others. The holistic spirit of the Directive and the work that the Member States undertook had a strong appeal to countries outside of the European Union and has led to the recognition of the Directive as a blueprint for Integrated Water Resources Management (Fritsch and Benson, 2020), pointing towards the adoption of system thinking perspectives (section 7.1).

Research (Mockler and Michael Bruen, 2018; Irvine, 2018; Gerend, 2019) however, argues that the implementation of the WFD deviated from considering the catchment as an integrated complex system, with defined systemic goals, the lack of which led to unsatisfactory outcomes. Indeed, transitioning from the old to the new regime, and consequently adopting a systems perspective has not been an easy path for the majority of Member States. European

Commission reports on the outcomes of each management cycle have been underlining the disconnection between pressures and measures, which has brought criticism on the effectiveness of the WFD as a policy tool (Hering et al., 2010), as measures implemented so far have not managed to achieve significant results. More specifically, it has been claimed that the focus has been on managing elements of water rather than fostering an understanding of the catchments as an open system and the broader context in which it functions (Irvine, 2018; Gerend, 2019). In line with these, scholars argue that the overall implementation of the WFD and its so far unsatisfactory results are products of applying reductionist approaches reminiscent of the pre-WFD era (Vlachopoulou et al., 2014; Giakoumis and Voulvoulis, 2019). Additionally, Carvalho et al. (2019) conclude that advancing WFD objectives demands to explore further the causes of status deterioration and providing evidence, as well as accommodate a dialogue among stakeholders for the selection of policy alternatives and improve policy integration in the planning and implementation stages of PoMs.

Building on the momentum that integrated environmental policies have been gaining since the 1970s, the spirit of the WFD encapsulates the idea that managing water resources cannot be separable from political, societal, and economic factors. However, the utilization of technical and scientific approaches to support the implementation has so far been skewed towards natural sciences for assessing key environmental parameters, and the development of monitoring programmes assessing pressures and their impacts, resulting in a reduction of the number of water bodies with unknown ecological status (European Environment Agency, 2018a). Social and economic aspects have been poorly developed and abstractly defined (Steyaert and Ollivier, 2007) influenced by conservative water management authorities that have been inflexible in fully adopting the new water management paradigm (Moss, 2008). Therefore, misunderstandings that have been recognized throughout the process of its implementation, stem from the gap between the letter and the spirit of the WFD that has been expressed by the lack of political will to opt out of traditional management regimes (Bouleau et al., 2020; Martínez-Fernández et al., 2020). The all-embracing though generic nature of the original text, being the result of political struggles during the drafting phase of the Directive (Kaika and Page, 2003), provided the freedom to the Member States to adjust the implementation to local conditions, and stressed the shortage of experience in systemic approaches that place the focus on the margin of scientific principles rather than their centre. Interpreting the WFD as another piece of environmental regulation, resulted in focusing on a small part of the socio-environmental system, insufficiently considering the role of rich, complex interactions between society and ecosystems that go beyond the identification and assessment of pressures and their impacts. As a result, as far as the economic analyses are concerned, stakeholders’ views are not taken into account in the estimation of either costs or benefits, which reinforces the argument that successful implementation of the Directive is commonly assessed in terms of timely achieving defined environmental objectives (Green and Fernández-Bilbao, 2006). In contrast to scholars that support further exploring systemic approaches to manage European waters, Linton and Krueger (2020) stress that conceptually the WFD promotes the view that nature and humans are separated, thus problems arise from the design and not the implementation of measures. The current thesis, in chapters 5 and 6, discussed in detail problems faced by the Member States concerning the economic analysis needed to be undertaken for assessing selected PoMs designed to improve water status, focusing on methodologies and approaches that could be utilized by policymakers.

In recent years, the concept of ecosystem services (Kumar, 2012), has been a valuable multidisciplinary tool for assessing the relationship between humans and nature (Busch et al., 2012), which could help adapt integrated water resources management to local contexts (Vollmer et al., 2022) and weaken its "top-down" implementation (Giordano and Shah, 2014). The level of ecosystem services provision and their quality depend on the capacity of the system to supply them (Haines-young and Marion Potschin, 2007; Angeler et al., 2015; Hein et al., 2015; Schröter et al., 2015). Therefore, policy actions that are successful in decreasing pressures on the environment, influence the provision of at least some ecosystem services (Grizzetti et al., 2019), ultimately affecting human wellbeing. Taking this into account, as well as the shortcomings in the assessment of PoMs by the majority of European countries in Chapter 5, I developed an integrated methodology for assessing the cost-effectiveness of PoMs. The methodology facilitates the inclusion of stakeholders' opinions concerning the relationship between pressures and ecosystem services, linking environmental improvement to various levels of benefits accruing from policy interventions. By applying this approach in a case study in the UK, I showed that the benefit-cost ratios of selected PoMs differ when non-market values are used and when the opinion of stakeholders is considered for the identification of relevant pressures, which might mean that when these are neglected, costs and not benefits drive policy decisions. The empirical part of the work aims at reducing the lack of studies in actual decision contexts, as noticed by Harrison-Atlas et al. (2016); analysing how ecosystem services valuation can be used by decisionmakers, an underexploited issue by the academic literature (Laurans et al., 2013); and making connections between research and practice (Marre et al., 2016; Vollmer et al., 2022). The findings have significant implications for policy making. First, incorporating ecosystem services in environmental management facilitates decisions to be taken at the junction of various fields, and has the potential to inform multi-sector decisions, as ecosystem services and their value is the common denominator of environmental change. Therefore, it enables policymakers to accurately measure the direct effects of specific measures and perceive the influence of other policies on the issue of interest. Secondly, assessing the effects of policies in terms of ecosystem services provision assists not only in defining their total costs but also reveals their benefits leading to better-informed decisions. Therefore, the integration of ecosystem services into decision-making through their incorporation in economic assessments, emphasizes the social and economic dimensions of improvements in the status of natural resources; justifies the costs of interventions; and provides opportunities for public engagement (Maltby et al., 2022).

Furthermore, actions taken to manage pressures in one sector may lead to unintended consequences in other fields. In Chapter 6, I argued that monitoring and evaluating the socioeconomic effects of any policy interventions on the system, using holistic structured approaches, are crucial for revealing potential necessary policy adjustments. Managing water holistically and effectively necessitates a focus on learning instead of assessing. It requires combining harmoniously the findings of natural sciences regarding the state, elements and functioning of the environment with deepening the knowledge of the dynamics and functioning of the socioeconomic system through establishing processes that enable the consideration of the spatial, cultural, social, and economic heterogeneity inherent in the socio-ecological system. Following a systems approach to define the state of a system, encourages understanding of how properties, and thus phenomena emerge, through non-linear interactions between individual elements. Consequently, defining the state of the catchment solely based on the ecological status of its river bodies would be as robust as defining it based solely on the GDP of the area. Natural capital accounting methodologies, though still in an experimental phase, constitute analytical systems approaches to examine the socioecological system (Bateman and Mace, 2020; HM Government, 2020; Dasgupta, 2021). Systematically collecting and organizing data related to ecosystem services, and their value can shed light on the effects of changes in the parameters of the system. Using this approach, in chapter 6, I showed how natural capital accounting can be used in the implementation of the WFD to facilitate assessing the status of a catchment holistically. Obtaining the value of natural capital requires collaboration between natural sciences to determine the types and extent of the resources, their conditions, and the flow of ecosystem services they provide; and economics to estimate the demand for natural capital and estimate its value (Vallecillo et al., 2019). Though natural capital accounting has been gaining currency with applications in different contexts (Ruijs et al., 2019b), its use in public policy decisions is limited (Recuero Virto et al., 2018). The work undertaken for the purposes of this thesis showcases how natural capital accounting can be used in a policy context. The developed accounts demonstrate that the asset value of recreation in both the Greek and the UK cases is higher than that related to water for residential use, which indicates that non-market benefits are also important for humans. From a policy perspective, natural capital accounts can capture the combined effects on natural resources and consequently on society of policies with different objectives. Therefore, while the suggested methodology aimed at assessing benefits generated by rivers, its applicability extends beyond that. Natural capital accounting methodologies can be used in multisectoral management contexts, such as those introduced by the Water-Energy-Food Nexus (Keairns, Darton and Irabien, 2016; Rodríguez-de-Francisco, Duarte-Abadía and Boelens, 2019), and the circular economy model (Cong and Thomsen, 2021). By developing measures comparable to GDP, natural capital accounts intend to assess the overall state of the system and point to priorities and issues that may arise in the future (Bateman and Mace, 2020). Omitting to capture the combined effects of sectoral policies on the state of natural resources, may lead to misguided conclusions about the overall effectiveness of proposed interventions, and lead to decisions based on a narrow view of the functioning of the socioecological system.

Traditionally, environmental management has been pursuing to minimize negative effects on natural resources in order to reduce ecological destruction (Browne, 2002; Robinson and Cole, 2015) and eliminate undesired outcomes (Waldron and Miller, 2013). Such an agenda promotes identifying and/or developing ways to reduce harm through, for example, advancements in technology, command and control mechanisms, and incentive-based tools (e.g., pricing). However, it might be the case that such approaches treat symptoms than the root causes of socioenvironmental problems. As Fournier (2008) puts it: “whilst there is a growing recognition of environmental degradation, the policies of sustainable development, or ecological modernization offered by national governments and international institutions seem to do little more than ‘sustain the unsustainable’ ”. Natural resource degradation can be seen from a governance (Bouckaert et al., 2018), economic (Quiggin, 2001), and ecological perspective (Feld et al., 2011) among others. However, managing the network of interactions between humans and natural systems that occur in complex systems, requires interdisciplinary research and integration of knowledge from various fields (Voulvoulis, 2012). Framing problems at the intersection of different disciplines is a key element of systems thinking, which enables blending multiple skills and viewpoints, thus accepting the complexity of environmental issues. For instance, the culvert in the River Wandle, a tributary of the River Thames in South London, was lodged with the aim of improving the health safety of local residents jeopardized due to the release of waste in the water. Despite achieving that, this measure increased flood risk and contributed to poor water quality (Webb et al., 2020). A broad view of the dimensions of any given environmental problem and integrated knowledge are prerequisites for proving how integrated interventions can be employed to influence complex social, economic, and environmental systems, as well as for minimizing unintended feedbacks (Larrosa, Carrasco and Milner-Gulland, 2016), and identifying unaccounted externalities.

Despite significant advancements in various scientific fields and progress in bringing different disciplines together to create more holistic approaches, it still seems hard to abandon traditional approaches that give the impression that by looking at a fraction of the whole we are able to manage complexity. In Chapter 8, I made the claim that only by adopting a new way of studying and understanding the world can we change the way we generate, process, and utilize knowledge for the benefit of society. In other words, as Boulding (1956) stated, only if we move science away from "its tendency to shut the door on problems and subject matters which do not fit easily into simple mechanical schemes", we will be able to enhance decision-making. By doing that, the idea that humans can control nature is rejected, which gives rise to interventions that focus on working with nature, embrace the complex web of interactions between humans and the environment and accommodate changes occurring in ecosystems rather than counteract them (Rutten, Cinderby and Barron, 2020). Consequently, accepting that humans and nature are inextricably coupled, provides the basis for setting more holistic objectives that go beyond the optimal level of growth, to understanding the multiple effects of measures, and based on that selecting those that move the system closer to a desired state.

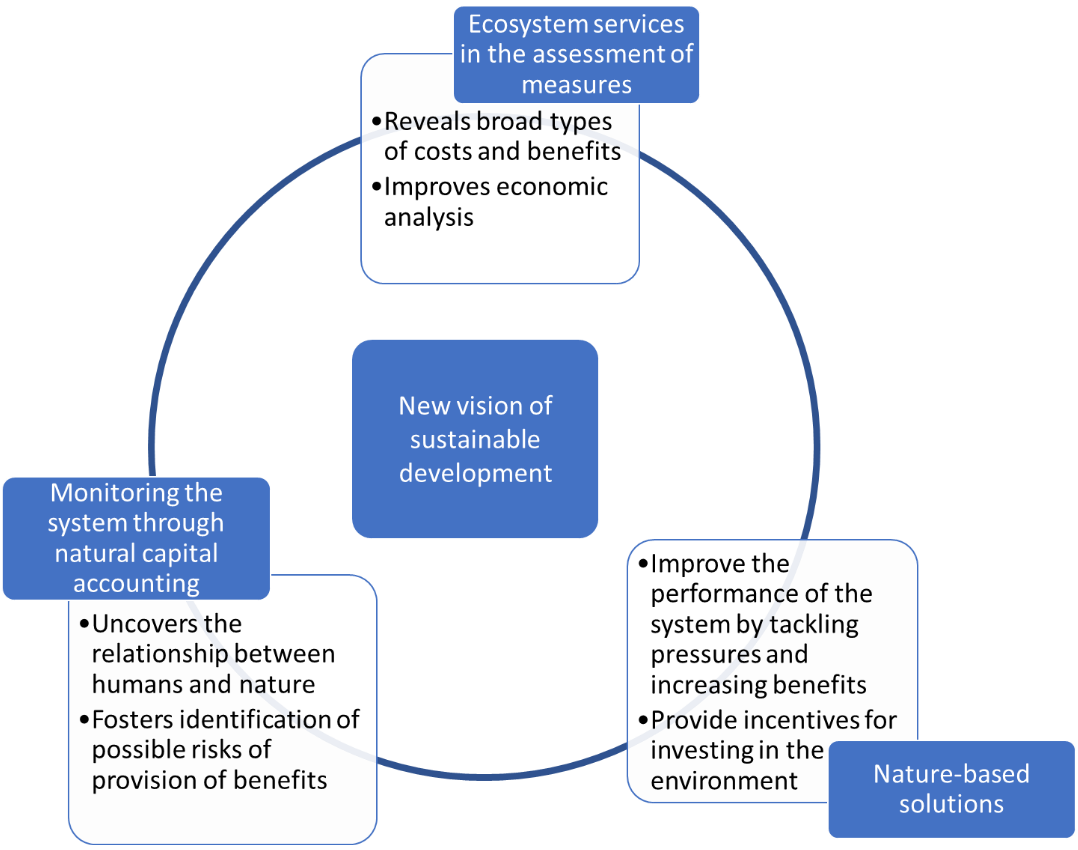

Figure 9.1 Schematic presentation of conceptual links between Chapters 5 to 8

A practical example of the “working with nature” idea is that of nature-based interventions, that involve managing human-nature interactions in such a way that enables ecosystems to correct human-produced negative effects and to minimize socio-environmental challenges. The aim of such actions is dual: improving the capacity of ecosystems to absorb pressures while providing benefits to society. When designed properly, NbS can lead to greater outcomes than traditional technical interventions that rely heavily on infrastructure (Seddon et al., 2021). The major body of NbS literature recognizes the multiple ecosystem services benefits they provide. For example, as shown in Chapter 7, besides benefits stemming from polishing nutrients, a constructed wetland was associated with regulating and supporting services. However, based on their design and the characteristics of their wider environment, NbS are able to provide significant recreational benefits (Haase, 2021) at relatively lower costs compared to traditional alternatives. Furthermore, an important aspect of NbS relates to what Cárdenas et al. (2021) describe as “indirect benefits”, meaning the positive influence that participation in the different stages of NbS projects (i.e., design, implementation, monitoring) exerts on individuals, ultimately motivating them to adopt pro-environmental attitudes. Nevertheless, such interventions entail changes in land uses, thus calling for attention to property rights, equity, and responsibilities (Paavola and Primmer, 2019), which promote a paradigm shift in environmental management (Bark, Martin-Ortega and Waylen, 2021). Supported by the European Union Green Deal Strategy (European Commission, 2019e) and the Biodiversity Strategy to 2030 (European Commission, 2019a), the role of such interventions will be further reinforced in the coming years. These strategies and the accompanying funding, create opportunities for showcasing how NbS can tackle complex problems. However, given their multifaceted effects, a reductionist implementation that focuses solely on sectoral outcomes, could lead to distortion across sectors and generate scepticism about their effectiveness. Similarly, the effective implementation of NbS should be based on assessing social, economic, and environmental synergies and trade-offs, as well as accommodating affected stakeholders’ views and preferences (Sowińska-Świerkosz and García, 2021).

9.2. Future avenues of research and potential application of the work

9.2.1. Generating a more integrated definition of the desired state

A systemic approach as prescribed by the spirit of the WFD, puts emphasis on dynamic processes that take place in the catchment and aims at identifying the systemic causes of a problem rather than the observed outcomes, accepting a non-linear connection between outcomes and intuitively expected causes (Lezak and Thibodeau, 2016). While a great deal of knowledge has been acquired since 2000, turning it into action entails incorporating in decision-making the economic, social, cultural, and political elements; understanding the trade-offs between anthropocentric and non-anthropocentric benefits and the way networks of interactions are related (Desjardins, 2019); while also realizing that people shape the system, and the system influences their decisions (Best and Holmes, 2010).

Fulfilling WFD requirements as an integrated management tool necessitates transforming the catchment into a scientific laboratory, within which, the complexity of socio-environmental networks and interactions can be studied. In this process of increasing information and knowledge, acknowledging the social and economic dimensions of the system is key. Participation aims at enhancing management decisions, by accommodating the stances and information of those closest to the problem, thus improving the acceptance of interventions (Owens, 2000; Newig, Pahl-Wostl and Sig, 2005b; Reed, 2008). The WFD encourages public participation in facilitating the process of identifying benefits and costs to justify potential exemptions (European Commission, 2009); in the initial analysis of the characterisation of the river basins, and the economic analysis of water uses; as well as the design and assessment of programmes of measures (Albrecht, 2016). Involving stakeholders in the stages of WFD implementation should be expected to influence policy decisions, through the reciprocal relationship between individuals and the properties of the system. Two aspects are of high relevance to the decision-making process to manage the complex links between changes in water status and society.

The first is that the perceptions and preferences of involved stakeholders play a crucial role in the success of achieving policy goals, as they determine the level of society’s support (Flávio et al., 2017). For example, the perception of farmers of the advantages and disadvantages is a crucial factor for determining their willingness to participate in environmental protection programmes (Söderqvist, 2003). Additionally, as observed by Heldt et al. (2016), if stakeholders deem that a project generates more private and public disadvantages than advantages, it is more likely that they will not endorse it. Furthermore, stakeholders belonging to distinct groups have different preferences and attitudes towards the same management outcomes (Giannoccaro, Pedraza and Berbel, 2013; Christen et al., 2015). Brown et al. (2010) report that government authorities rank avoiding extreme events and securing water supply higher compared to environmental non-government organisations that prioritize recovering freshwater ecosystem services. Managing authorities may profit from involving stakeholders in management decisions by obtaining local knowledge about the conditions of the ecological system, understanding the perception of users of natural resources, gaining insights into the socio-economic system and eliciting information about the acceptance of proposed interventions (Newig, Pahl-Wostl and Sigel, 2005). Although Chapter 5 provides a framework that considers stakeholders' opinions about how potential measures may influence pressures and thus ecosystem services, and in different parts of the work the importance of engaging stakeholders in environmental management decisions is discussed; the undertaken work is more focused on explaining the architectural elements of new management tools and approaches. Regardless of the process selected (e.g., meetings, workshops, surveys, interviews, etc.) for engaging stakeholders (Reed et al., 2009), incorporating their opinions in economic valuation exercises should be expected to shape the obtained economic values. For instance, the creation of a new habitat for species, as a result of the development of a NbS might be perceived as a positive development for farmers, due to increased pollinators abundance (Catarino et al., 2019); however, NbS in other contexts, such as the development of urban forests, might be perceived as being associated with adverse effects (e.g., damage to infrastructure, gentrification and increase of criminality) (Portugal Del Pino, Borelli and Pauleit, 2020). Consequently, I consider that future research could benefit from engaging stakeholders in exercises concerning the assessment of the extent and benefits of natural capital (chapter 6), different types of measures designed to improve environmental resources while increasing societal benefits, construction of scenarios of possible future developments to manage uncertainty and overall to influence and orient decision-making processes.

The second aspect relates to the visualisation of good water status as a desired system state, in terms of its social and economic characteristics. From an ecological perspective, specific levels of biological and chemical parameters and the hydrological characteristics of each water body are used to determine the deviation from reference conditions (Nõges et al., 2009), thus, the distance from good water status. Observing the development of such parameters has further promoted natural scientists' understanding of how the environment functions, and resulted in better capturing the impact of human pressures on the environment, as well as the synergistic, antagonistic, and additive interactions among pressures (Brown et al., 2013; Jackson et al., 2016; Schinegger et al., 2016; Orr, Rillig and Jackson, 2022). However, the definition of an undisturbed state may not accurately be defined by the absolute lack of pressures, as the interaction between humans and nature is ever-evolving. In fact, the end state described by the WFD is subjective and can be perceived differently by the public, the scientific community, and the managing authorities (Valinia et al., 2012). Interpreting the definition of good water status in a broader sense that considers the status of the socioecological system as a whole, may assist in better explaining the reciprocal relationship between human populations and the environment, and through that define the level of pressures that would lead to ecological improvement of water ecosystems, while securing benefits associated with the existence of pressures.

Economics provides not merely the tools (e.g., prices, quotas etc.) for managing water uses, but also frameworks for understanding interactions in the systems, as presented in chapter 7 and discussed in chapter 8, that can foster the realisation of the full potential of the Directive, by providing a vision of an end state that considers the health of the environment and its capacity to generate ecosystem services as well as the welfare of the society. At the core of economics is the generation of knowledge on how 'the various pathways through which millions of decisions made by individual human beings can give rise to emergent features of communities and societies' (Dasgupta, 1997). Individual decisions are influenced by the emergent features of the system, which may be the result of individual decisions (Gibson, Ostrom and Ahn, 2000). The theory of Planned Behaviour (Ajzen, 1991), one of the most widely used theories of behaviour in psychology, argues that attitudes, beliefs, and values can adequately explain behavioural intentions, though there may be discrepancies between intended and actual behaviour. Empirical evidence suggests that individual preferences predict a wide range of behaviour (Stern and Dietz, 1994; Schleich et al., 2019; Fuhrmann-Riebel, D'Exelle and Verschoor, 2021). Studies have investigated how different variables determine the demand for environmental goods and services, such as socio-demographic characteristics (e.g., the level of income and education, age, gender, number of children etc.) (Campbell, 2007); and the relationships between stakeholders and the good under investigation (Hoyos et al., 2012). Additionally, variables related to environmental literacy, consciousness, social psychology, accessibility, spiritual messaging, ethics etc. have been verified to contribute to preference heterogeneity (Spash et al., 2009; Chew et al., 2019; Li et al., 2022). However, usually, policy implications of preference heterogeneity are not discussed. In future research, such issues could be incorporated into the methodology developed in chapter 5 to improve the assessment of the effects of policy interventions that aim at tackling pressures and increasing human welfare; or assist in developing new policies that focus on the interactions between humans and nature rather than nature itself.

Furthermore, central to this study is the concept of socioeconomic value. Extensively studied through different scholarly disciplines, this concept has attracted significant attention, especially in the field of economics generating voluminous results that can form the informational base for policy formulation. Economic valuation, a tool distinct in all chapters of this thesis, assists in elucidating the connection between humans and the flow of benefits generated by ecosystems. Therefore, by adopting a systems thinking perspective the current study claims that economics in the context of freshwater and environmental management, in general, can provide more information than just the costs of different alternatives. However, ecological functions are site-specific and defined by the whole ecological system (Harrison et al., 2014). Consequently, the relationship between nature and society depends both on the preferences and attitudes but also on the benefits harvested by the environment resulting in differentiated human-nature interactions across catchments that management practices need to consider. For instance, if the magnitude of regulating and provisioning services differs between two areas, adopting the same management practices for improving water status will result in different socioeconomic effects on residents in these two areas. In other words, economics and its tools can reveal the magnitude of the connection between the social and the natural systems and add to our knowledge how altering such connections or even the functioning of the system would impact its overall “health”. Following this argument, economic valuation may be more appropriate for assessing the flow of goods and services stemming from marginal changes in the status of a resource above critical thresholds beyond which the ecosystem may reach an undesirable state. Therefore, it has been proposed that the insurance value of ecosystems is also considered in management decisions (Mäler, 2008; Walker et al., 2010; Baumgärtner and Strunz, 2014), i.e., the value of ecosystems sustaining their capacity to maintain their functioning and provisions of benefits despite any disturbances and changes. Knowledge about the demand for insurance services, currently limited (Wolff, Schulp and Verburg, 2015), could inform investment decisions that influence ecosystem resilience.

Additionally, economic information plays another crucial role, besides explaining the state of the environment. Experiments in economics and behavioural studies demonstrate that people choose in ways that do not agree with the assumptions governing “homo economicus” behaviour. Individuals are not selfish, but they care about the benefit of others (Pollitt and Shaorshadze, 2013). Their choices are not rational, they face difficulty in choosing among a wide range of alternatives (Iyengar and Lepper, 2000), and use heuristics for updating probabilities of future events as a response to new information (Colin and George, 2004). Information about which actions affect the system and how is important for influencing behaviour. For instance, tailored information on energy use may influence the behaviour of households towards reducing their consumption (Abrahamse et al., 2007; Steg and Vlek, 2009). Therefore, the description of the status of the socioenvironmental systems and the effects of individual and policy actions through quantification of benefits promotes awareness about the importance of flows of services to sustain life and human wellbeing (Oliver, 2019), which has the potential to alter individual behaviour. Relaxing dominant assumptions and expanding the scope of economic analysis through the inclusion of social and natural science aspects promotes a transition from linear models of production and consumption to more holistic ones. For instance, the current market economy paradigm based to a considerable extent on neoliberalism (Ghisellini, Passaro and Ulgiati, 2021) is not able to account for positive or negative effects (externalities) due to the current structure of the market as well as the frameworks used to describe the processes that take place within it. Moving towards a more integrated vision of prosperity as presented in Chapter 8 is associated with the need to analyse the flows and stocks of natural capital and assess how institutions shape the interactions between the state of ecosystems and human behaviour.

Natural capital accounting (Chapter 6) helps to connect nature to the national economic statistic systems, highlighting how economic progress depends on the natural ecosystem, thus revealing the real risk of what may be lost and what may be earned by investing in nature. Natural capital accounting has gained significant traction in recent years. For instance, its significance has been stressed by the EU Biodiversity Strategy to 2020 and the Seventh Environment Action Programme of the EU and the European Green Deal which includes natural capital accounting in a range of activities to promote green finance and investment. The current thesis included an empirical example of the use of such an account in assessing the state of the system to reinforce the selection of holistic interventions and assist in identifying potentially emerging disturbances, in the context of WFD implementation. Structured monitoring protocols and methodologies are necessary to deepen our knowledge of how socioecological changes occur. Placing such knowledge at the core of decision-making is necessary to effect social norms and large-scale transformation in policies and practices (Guerry et al., 2015; Kurniawan, Sugiawan and Managi, 2021) to manage water resources sustainably and integrate environmental and welfare aspects into the dominant notion of economic growth. At the current state of development several challenges exist (Bagstad et al., 2021; Brandon et al., 2021) for the implementation of natural capital methodologies that relate to the complexity of asset-benefits relationships (Mace, 2019); the relationships between biodiversity and ecosystem services (Lefcheck et al., 2015); uncertainty about how ecosystem respond to disturbances; lack of data required for valuation exercises; methodological challenges; and coordination between different institutions to name a few. Future works in the area may try to resolve such issues to further promote the integration of principles under such a transdisciplinary methodology.

Finally, experimental, and behavioural studies in natural resources management have not yet been integrated sufficiently to produce a large number of results and elaborate procedures. Economic research in the current era of complexity[22], such as behavioural and experimental studies suggests that individuals behave less rationally than is traditionally assumed (Holt et al., 2011), due to exhibiting myopic behaviour (Cooper and Kovacic, 2012), the fact that they value loss greater than gains (Kahneman and Tversky, 2019), and biased behaviour (Oechssler, Roider and Schmitz, 2009; Puri and Robinson, 2011; Gigerenzer, 2018). Furthermore, the properties of the economy as a system emerge from non-linear market and non-market interactions of a large number of elements, therefore a micro foundation approach is over-simplistic (Gaffeo et al., 2008). Adopting a systems thinking approach to manage water resources should be expected to be ineffective if it is not accompanied by an effort to expand our understanding of what actually constitutes the whole. Identifying the components of the system, their interactions, and the potential feedback loops (Holland John, 2006; Levin et al., 2013) enables describing phenomena that occur within the system to build up general models to describe them (Boulding, 1956). Therefore, given recent advancements in interdisciplinary economics investigating the microeconomic foundation of macroeconomic policies might lead to increased opportunities for environmental management.

9.2.2. Further use of tools beyond the implementation of the WFD

Water ecosystems, in general, are affected by the uses of water, climate change, legislation, policies, and initiatives that alter ecosystems adjacent to them. At the same time, international policy developments, such as the Sustainable Development Goals (SDGs) of the United Nations instruct countries to increase their effort to manage environmental resources sustainably. Four out of the 17 SDGs are environment-related and two of them refer explicitly to water-related issues: SDG 6: Clean Water and Sanitation and SDG14: Life below Water. The environmental SDGs along with other two groups that concern the economy (e.g., SDG 8: Decent Work and Economic Development) and society (e.g., SDG 3: Good Health and Wellbeing) form a net of general goals and targets that require integrative policy interventions in different domains (e.g., food, energy, water, labour market etc.). The Green Deal (European Commission, 2019e) constitutes a key element of the EU strategy to implement the UN SDGs.

[22] The term 'era of complexity' is used as in (Holt, 2011) , to denote current advances in heterodox economics.

Furthermore, the Farm to Fork strategy (European Commission, 2020d) of the Green Deal aims to achieve reductions in pesticide and antibiotics use, and nutrient losses in agricultural production in order to reduce the footprint of the sector on the climate and the environment. Additionally, the revised Common Agricultural Policy (CAP) that will be introduced in 2023, is envisaged to contribute to the delivery of the Green Deal agriculture-related objectives (European Commission, 2020b). Such international commitments call for a structured system of monitoring socioeconomic changes across time and space (Crossman et al., 2013). Natural capital accounting can be particularly relevant to facilitate such a task, however, current practices implemented in a number of countries vary in scope, types of ecosystems and spatial detail (Hein, Bagstad, et al., 2020). As the adoption of this methodology is currently spreading, there is an increasing need in deciding what needs to be measured and how in order to assist natural capital accounts in accurately describing the state of the system to facilitate the prioritisation of investments. Besides a number of methodological issues, another area that needs further elaboration is the valuation of certain types of ecosystem services. For example, while cultural and provisioning services obtain high values, regulating services are undervalued, despite their high significance for sustaining the functioning of ecosystems. In addition to that, further investigation of certain types of ecosystems such as intermittent rivers (Koundouri et al., 2017) is required, in order to define how their specific characteristics can be incorporated into relevant analyses.

Therefore, as discussed in Chapter 5, though ecosystem services are highly praised for explaining the human-nature relationship in a straightforward manner, there is still a significant lack of studies on how they can be operationalized in environmental management. The current thesis provides such frameworks and examples; however, more effort could be allocated to assessing how their magnitude fluctuates across different locations and contexts, and how different levels of environmental quality affect welfare benefits. A relevant example of such work, is that of Grizzetti et al. (2019) that quantify ecosystem services provided by freshwater and connected ecosystems, though their assessment concerns a limited number of ecosystem services and refers to a continental scale. Further research is required to investigate how the characteristics of the ecosystems affect the provision of multiple ecosystem services. In relation to economics and human wellbeing, environmental management and policy-making more often than not make use of aggregate assessments of ecosystem services, which though useful for demonstrating the importance of nature to humans (Costanza et al., 2014), do not count for the distribution of ecosystem services benefits and costs across individuals (Cord et al., 2017). Such an issue is particularly relevant to holistic interventions, such as nature-based solutions, in cases where land uses are required to change and new ecosystems are created (e.g., through the introduction of a forest to improve air quality), and the positive and negative effects they generate may vary across diverse types of stakeholders. Additionally, given recent advancements in sustainability science, integrated interventions need to be further analysed through the prism of explaining how structural changes can transform systems to satisfy human needs while safeguarding the quality of natural resources. Chapter 8 aimings to contribute to current sustainability discussions, put forward a new vision for sustainable development that includes increasing efficiency in the use of natural resources, and enhancing the regenerative capacity of natural capital for sustaining the provision of ecosystem services. Moreover, investigating incentives and policy mixes to overcome lock-ins (Simoens, Fuenfschilling and Leipold, 2022) inertia and path dependence (Söderbaum, 2014) are needed to improve the overall conditions of socioecological systems.

10. Summary of findings- Conclusions

The thesis considered the role of economics in the assessment, design, and selection of environmental management interventions, as well as the potential of economic concepts and methodologies in reinforcing systemic changes. The analysis revolved around the implementation of the Water Framework Directive, the first systematic attempt to apply systems thinking in policy development and implementation and the resulting mismatch between the prescriptions of the Directive and its implementation across Europe.

The use of economic tools and principles was reviewed to identify why interventions designed and implemented to address pressures on water ecosystems have not managed to improve the overall health of the system, which has led to a considerable number of water bodies in lower than good water status. Findings demonstrated limited understanding of how societal and economic information can be integrated into environmental policies to produce outcomes that improve both the status of ecosystems and social and economic conditions. Significant ambiguity revolving around the definitions of the economic aspects of the Directive and lack of clarity regarding the methodologies that should be followed, have resulted in oversimplistic economic assessments that disregarded a great amount of information, necessary to achieve the goals of an integrated environmental policy. Economic information included in the RBMPs has been limited and economic analysis has not been wide enough to bring to light the interactions between the different elements of the system.

Implementing environmental policies without a clear understanding of the network of interactions between different elements within the catchment, has resulted in disconnection among pressures, measures and water status and overlooked the importance of interventions to relevant stakeholders. Investigating the incorporation of ecosystem services in the assessment of programmes of measures in the river of the Broadland catchment was shown to significantly influence the value estimates of resulting socioeconomic effects. More specifically, taking into account the opinion of local stakeholders in order to connect planned interventions with reducing pressures and the consequent effects on ecosystem services provision, was proven to yield a higher value of benefits for the selected measures than previously estimated. This implies that in the worst-case scenario where decisions have been based solely on financial costs, implemented measures might have been accompanied by hidden environmental and social costs. Focusing on a specific aspect of a given problem rather than on its multiple dimensions hinders a complete and well-rounded understanding, and does not create the conditions for a socially, economically, and environmentally desirable state to emerge.

Policy choices influence the system beyond the specific sector or issue they target. A holistic understanding of the conditions of the system requires policymakers to go further than merely observing data, to understand patterns of behaviour, enabling them to comprehend the underlying structure that drives specific outcomes. Natural capital accounting, an interdisciplinary methodology provides a structured way of assembling information to reveal socioecological changes that take place over time. Applying this methodology in two cases demonstrated its usefulness in assessing how water management practices influence the provision of ecosystem services and their economic value. In addition to the importance of natural capital accounting as a monitoring tool, its premise that there is a reciprocal and complex relationship between humans and nature provides a new perspective on how to intervene in the system and manage interactions to reach the desired state.

In other words, a systemic view of socio-environmental issues brings to light measures that go beyond controlling the system. Nature-based solutions discussed in the thesis, constitute a new type of action to tackle socioecological challenges. Contrary to traditional technical measures whose unintended consequences are neglected, nature-based solutions constitute actions designed in a way that their application, transforms multiple processes of the system, changing how elements relate to each other and giving rise to new properties. The assessment of a constructed wetland presented in this thesis demonstrated that beyond its primary goal to polish effluents from a recycling centre to improve water status, the implemented solution stimulated the generation of additional ecosystem services, such as carbon sequestration and habitat for species. Nature-based solutions either implemented alone or in combination with grey-infrastructure turn the focus on the wider improvement of the habitat quality at a scale that considers specific resources (e.g., rivers) and their broader environment that consists of land ecosystems, humans, and businesses. Therefore, their adoption is concerned with reducing environmental pressure, while improving the resilience of the whole.

However, currently, despite wide recognition of the deteriorating status of natural ecosystems accruing from a long tradition of exploitation for satisfying human needs; humanity is far from being on a sustainable path. Even worse, although command-and-control practices, end-of-pipe solutions and reductionist models have proven ineffective to improve living conditions on Earth, the existing theories for achieving sustainable development are far from offering concise alternative frameworks. Therefore, while the fact that the WFD introduced a novel integrated water resources management paradigm in Europe is broadly acknowledged, its implementation has been marred by the deployment of obsolete ideas. Through investigating the dominant theories of sustainable development, the thesis puts forward the importance of having a clear vision of prosperity. A prerequisite for achieving such prosperity is the realisation that humans are not external observers of nature, but components of the same socioecological system. Such a viewpoint encourages evaluating how human activities affect nature and how nature influences humans, thus acknowledging the interdependencies between humans and nature. Therefore, it promotes the design of policy objectives that consider the impact that different levels of ecosystems' capacity to generate services have on wellbeing. In this kind of policymaking processes, economics can foster an understanding of how socioeconomic value is determined, assist in revealing points of intervention to alter system behaviour and contribute to defining desired states in terms of both environmental and socioeconomic characteristics.

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Appendix A

Table A.1, Table A.2 and Table A.3 contain the list of measures selected to improve water status either by 2021 or beyond 2027 in the Broadland Rivers catchment. The information was obtained by the Environment Agency (Environment Agency, 2015a).

The measures are described in section 3.3 Part 1 of the Anglian River Basin Management Plan and include measures that will be realised by 2021 and achieve environmental outcomes, but there is not enough confidence (in location or scale of improvement) to predict specific outcomes.

Table A.1 List of measures with low confidence that will happen by 2021

No. of measure	Operational Catchment	Measure category	Description of measure	Estimated start date	Sector of lead organisation	Key Type of Measure
1	Wensum	To improve modified habitat	Habitat improvement - Wensum tributaries	1/7/2015	Environment, Farming, Rural	KTM6 - Improving hydromorphological conditions of water bodies other than longitudinal continuity
2	Waveney	To improve modified habitat	Habitat restoration - Waveney habitat project	1/7/2015	Environment, Farming, Rural	KTM6 - Improving hydromorphological conditions of water bodies other than longitudinal continuity
3	Bure	To control or manage diffuse source inputs	Reduce diffuse pollution pathways - "Broadland Slow the Flow" project	1/7/2015	Environment, Farming, Rural	KTM2 - Reduce nutrient pollution from agriculture
4	Waveney	To control or manage point source inputs	Additional treatment to reduce concentrations of nutrients 5from Pulham St Mary STW	31/3/2020	Wastewater treatment	KTM1 - Construction or upgrades of wastewater treatment plants
5	Waveney	To control or manage point source inputs	Additional treatment to reduce concentrations of phosphate from Hoxne sewage treatment works.	31/3/2020	Wastewater treatment	KTM1 - Construction or upgrades of wastewater treatment plants
6	Waveney	To control or manage abstraction	Change in abs lic condtn(s) to address	26/6/2015	Environment, Farming, Rural	KTM6 - Improving hydromorphological conditions of water

			potential serious damage at full license, Dickleburgh Stream			bodies other than longitudinal continuity
7	Wensum	To control or manage abstraction	Change in location of abstraction	1/7/2015	Environment, Farming, Rural	KTM6 - Improving hydromorphological conditions of water bodies other than longitudinal continuity

The measures listed below were used to produce the summary programmes of measures in table 22 in section 3.5 of Part 1 of the river basin management plan.

Table A.2 A summary of the additional measures needed to achieve objectives beyond 2021

No. of measure	Operational catchment	Bundle	Measure category 1	Measure category 2	Measure category 3
8	Bure	G1 (to good status bundle, cost beneficial)	To improve modified habitat	Removal or easement of barriers to fish migration	Enable fish passage (e.g., fish pass)
9	Bure	G1 (to good status bundle, cost beneficial)	To improve modified habitat	Improvement to condition of channel/bed and/or banks/shoreline	Increase in-channel morphological diversity
10	Bure	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
11	Bure	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
12	Bure	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Arable soils
13	Bure	G1 (to good status bundle,	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Livestock

		cost beneficial)			
14	Bure	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
15	Bure	G1 (to good status bundle, cost beneficial)	To control or manage non-native invasive/alien species	Building awareness and understanding (to slow the spread)	Implement Individual Species Action Plans for priority species.
16	Bure	G1 (to good status bundle, cost beneficial)	To control or manage non-native invasive/alien species	Mitigation, control, and eradication (to reduce extent)	Share best practice on partnership working
17	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage point source inputs	Mitigate/Remediate point source impacts on receptor	Install nutrient reduction
18	Waveney	A (alternative objective bundle, cost beneficial)	To improve modified habitat	Removal or easement of barriers to fish migration	Enable fish passage (e.g., fish pass)
19	Waveney	A (alternative objective bundle, cost beneficial)	To improve modified habitat	Improvement to condition of channel/bed and/or banks/shoreline	Increase in-channel morphological diversity
20	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
21	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
22	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Nutrients
23	Waveney	A (alternative objective bundle,	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry	Surface run-off & drainage management

		cost beneficial)		to water environment)	
24	Waveney	A (alternative objective bundle, cost beneficial)	To improve modified habitat	Vegetation management	Plant new vegetation
25	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Arable soils
26	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Livestock
27	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Pesticide management
28	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage non-native invasive/alien species	Early detection, monitoring and rapid response (to reduce the risk of establishment)	Control and eradication of selected high-risk species
29	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage non-native invasive/alien species	Mitigation, control, and eradication (to reduce extent)	Share best practice on partnership working
30	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage non-native invasive/alien species	Building awareness and understanding (to slow the spread)	Implement Individual Species Action Plans for priority species.
31	Waveney	A (alternative objective bundle, cost beneficial)	To control or manage non-native invasive/alien species	Mitigation, control, and eradication (to reduce extent)	Support established local fora by providing advice and guidance
32	Wensum	G1 (to good status bundle, cost beneficial)	To improve modified habitat	Improvement to condition of channel/bed and/or banks/shoreline	Increase in-channel morphological diversity
33	Wensum	G1 (to good status bundle,	To improve modified habitat	Removal or easement of barriers to fish migration	Enable fish passage (e.g., fish pass)

		cost beneficial)			
34	Wensum	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
35	Wensum	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
36	Wensum	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Nutrients
37	Wensum	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Arable soils
38	Wensum	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
39	Wensum	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
40	Wensum	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Pesticide management
41	Wensum	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Livestock
42	Wensum	G1 (to good status bundle, cost beneficial)	To control or manage non-native invasive/alien species	Mitigation, control, and eradication (to reduce extent)	Share best practices on partnership working
43	Wensum	G1 (to good status bundle,	To control or manage non-native	Building awareness and understanding (to slow the spread)	Implement Individual Species Action Plans for priority species.

		cost beneficial)	invasive/alien species		
44	Yare	G1 (to good status bundle, cost beneficial)	To improve modified habitat	Removal or easement of barriers to fish migration	Enable fish passage (e.g., fish pass)
45	Yare	G1 (to good status bundle, cost beneficial)	To improve modified habitat	Improvement to condition of channel/bed and/or banks/shoreline	Increase in-channel morphological diversity
46	Yare	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
47	Yare	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
48	Yare	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Nutrients
49	Yare	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Livestock
50	Yare	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution at source	Field & Crop - Arable soils
51	Yare	G1 (to good status bundle, cost beneficial)	To improve modified habitat	Removal or modification of engineering structure	Remove structures
52	Yare	G1 (to good status bundle, cost beneficial)	To control or manage diffuse source inputs	Reduce diffuse pollution pathways (i.e., control entry to water environment)	Surface run-off & drainage management
53	Yare	G1 (to good status bundle,	To control or manage point source inputs	Mitigate/Remediate point source impacts on receptor	Install new private STW

		cost beneficial)			
54	Yare	G1 (to good status bundle, cost beneficial)	To control or manage point source inputs	Mitigate/Remediate point source impacts on receptor	Upgrade existing private STW

The estimated date of most of the following programmes described in section 3.3 of Part 1 of the river basin management plan is set to be 2015.

Table A.3 Summary of the programmes of measures that will improve the water environment by 2021

No of measure	Water body ID/ name	Measure category 1	Description of outcome	Key Type of Measure
55	GB105034055882	To improve modified habitat	To prevent eels and elvers from being entrained (sucked into) river abstractions and prevented from returning upstream by obstructions, the Eels Regulations require appropriate screening to be fitted to abstractions and obstructions to be removed or by-passed. These measures should not only prevent entrainment of eels, but also other fish species.	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity
56	GB105034055882	To improve modified habitat	To prevent eels and elvers from being entrained (sucked into) river abstractions and prevented from returning upstream by obstructions, the Eels Regulations require appropriate screening to be fitted to abstractions and obstructions to be removed or by-passed. These measures should not only prevent entrainment of eels, but also other fish species.	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity
57	GB105034051281	To improve modified habitat	To prevent eels and elvers from being entrained (sucked into) river abstractions and prevented from returning upstream by obstructions, the Eels Regulations require appropriate screening to be fitted to abstractions and obstructions to be removed or by-passed. These measures should not only prevent	KTM6 - Improving hydro morphological conditions of water bodies

			entrainment of eels, but also other fish species.	other than longitudinal continuity
58	GB30536989 GB30547009 GB105034055730 GB105034055881	To control or manage diffuse source inputs	prevent deterioration, or contribute to the achievement of protected area objectives, reduce the impact of diffuse pollution that arises from rural land use	KTM23 - Natural water retention measures
59		To improve modified habitat	River restoration works to improve hydro morphology and diffuse pollution on the urban Clipstone Brook	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity
60	Catchment wide	To improve modified habitat	Broadland Catchment Partnership will provide multiple benefits, joining the resources of a range of organisations towards delivery of WFD objectives. This project will allow the delivery of an action plan published in 2014 and is supported financially by a range of partners from public, private and third sectors. The Broadland Catchment Plan identifies seven main goals, including reducing flood risk and promoting sustainable drainage. The Partnership will work with Norfolk County Council using the mapping of surface water flooding risk across the county. It will work with landowners and highways to reduce flood risk. Several actions in the Plan are already underway, in particular those for land management. By continuing to contribute to the Broadland Catchment Partnership we can help ensure these actions and others are completed. Specific focus this year will be given to setting up demonstration projects for rural drainage in high run off areas and exploring potential locations for constructed wetlands to reduce phosphorus downstream of urban areas.	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity

61	Catchment wide	To improve modified habitat	This is partnership project with Rochford District Council and the Essex Wildlife Trust will restore a Heavily Modified Water Body to a more natural state and address water quality issues from both urban and rural areas. The section of Nobles Green Ditch where this project will be undertaken encompasses a Country Park downstream of a large sewage works. As well as being impacted by urban influences and road runoff this water course is also impacted by the surrounding land. The project will entail in-channel and riparian habitat improvements as well as removing a barrier currently impassable to fish. This project will lead to an improved ecological status for this water body.	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity
62	Catchment wide	To control or manage diffuse source inputs	The catchment of the Yare is largely agricultural, but also passes through Wymondham and southern suburbs of Norwich in its lower reaches. Both towns can cause urban pollution from Combined Sewer Overflows and misconnections. Road drainage in parts of the catchment acts as a conduit for sediment borne pollution, a particular problem on narrow rural roads where verges are undermined and sediment is lost from field gates. Many protected areas are fed by these rivers and receive nutrient enrichment. Parts of the catchment is designated under the Habitats Directive and subject to Diffuse Water Pollution Plans to improve the status of these designated sites. Measures identified in these plans include the reduction of run off from highways. We will use the output from a project that identified sediment pathways to the rivers from highways sources to identify highway drainage improvements. This will include sediment trapping in suitable locations where landowners are willing. We will also work with Norfolk County Council as they develop their flood risk strategy and surface water management plans to identify mutually agreeable options for attenuating flood water from urban areas and highways sources.	KTM21 - Measures to prevent or control the input of pollution from urban areas, transport and built infrastructure
63	Area wide	To improve modified habitat	Several of our water bodies are failing for fish, plants, and invertebrates. Tree planting, particularly in the headwaters, will provide shading; cooling the water temperature which will benefit fish spawning,	KTM6 - Improving hydro morphological

			invertebrates and reduce macrophyte growth. Working with a number of landowners, Catchment Partnerships and other partners across the catchments of our three counties we will undertake several tree planting projects which will achieve these benefits as well as reducing agricultural run-off from entering the water courses.	conditions of water bodies other than longitudinal continuity
64		To improve modified habitat	This project will implement the mitigation measures improving habitats required under WFD, alongside other locally identified opportunities.	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity
65		To improve modified habitat	Wetland creation to reduce nutrient input and levels in the River Glaven upstream of recent river restoration works.	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity
66		To improve modified habitat	Urban river morphology project involving local community, volunteers, Ipswich Borough Council and Suffolk County Council. A detailed plan has been produced by Environment Agency in collaboration with partners	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity

67	GB104028053340, GB104028042490, GB104028042501, GB104028042502, GB104028042510, GB104028042520, GB104028046430, GB104028042640, GB104028053310, GB104028053380, GB104028042400, GB104028046680, GB104028046840, GB104028047030, GB104028053110, GB104028053250, GB104028064290, GB109054044140, GB109054044520, GB109054044660, GB109054049144, GB70410266, GB70410508, GB70410537, GB70910519, GB71210541	To improve modified habitat	Habitat restoration in headwaters of River Waveney	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity
68		To improve modified habitat	Improvements to riparian habitat, tree planting, reconnecting flood plain, reconnecting old river channel and bank re-profiling. Working with Chelmsford City Council Parks It is hoped to re-designate the park as a Local Nature Reserve and provide an on-going maintenance regime to enhance its biodiversity value.	KTM6 - Improving hydro morphological conditions of water bodies other than longitudinal continuity
70		To control or manage diffuse source inputs	Sediment laden run-off from land informally used by 4x4 vehicles is causing water quality issues and habitat degradation at two locations in the Chelmer catchment, one of which is affecting a local wildlife site. Rural Sustainable Urban Drainage systems are to be used to trap sediment and improve habitat, whilst land damaged by the 4x4 activity will be restored and security improved to help prevent un-authorised vehicle access.	KTM17 - Measures to reduce sediment from soil erosion and surface run-off

71		To control or manage diffuse source inputs	The project is led by the Game and Wildlife Conservation Trust and the Freshwater Habitats Trust. The Water Friendly Farming project in the upper Welland takes a rigorous approach to implementation of resource protection measures, with two 'treatment' catchments and a comparable control catchment. The project focuses heavily on soil and land use management, and delivers a balance between research and development, and practical implementation of a wide variety of best practice land management techniques and technology trials. The project aims to reduce sediment loss, and associated diffuse pollution to watercourses, which impacts the ecology, and amenity use, and often requires removal of sediment by use of public funds in the lower sections of the catchment. The Water Friendly Farming project will help us further understand how to achieve effective integrated catchment management; providing both environmental and flood risk benefits, with results and methodology that can be applied to other catchments where necessary and appropriate. The project is ongoing with annual investment since at least 2011 from landowners, academic institutes, Anglian Water, and private sector agricultural organisations.	KTM17 - Measures to reduce sediment from soil erosion and surface run-off

Table A.4 Studies used for the estimation of the ecosystem services benefits from improvements in the Broadland Rivers catchment

Study	Country	Type of services
Adamowicz et al. (1995)	UK	Cultural
Ahtiainen et al. (2014)	Finland	Regulating
Alcon et al. (2013)	Spain	Regulating
Barrio and Loureiro (2013)	Spain	Cultural
Bateman et al. (2006)	UK	Provisioning
Birol et al. (2013)	Poland	Regulating
Brouwer and Bateman (2005)	UK	Regulating
Castro et al. (2016)	USA	Cultural, regulating, provisioning
Doherty et al. (2014)	Ireland	Cultural, regulating
Bouscasse et al. (2011)	France	Provisioning
Genius et al. (2008)	Greece	Provisioning
Genius et al. (2012)	Greece	Provisioning
He et al. (2017)	Canada	Regulating
Hein (2011)	Netherlands	Provisioning
Koundouri et al. (2014)	Greece	Provisioning
Markantonis et al. (2013)	Greece	Regulating
Polyzou et al. (2011)	Greece	Provisioning
Stithou et al. (2012)	Ireland	Cultural, regulating

Table A.5 Minimum, maximum and average value per ecosystem service sub-category estimated from relevant studies

Identified ecosystem service sub-categories	Min	Average	Max	unit
		Provisioning services		
Water quality drinking water	1.06	1.96	3.47	Per person
Water for irrigation	1.61	15.54	16.87	Per person
Timber (fuel wood), coppice	-	6235.19	-	Per hectare
		Regulating services		
Natural hazards regulation (flooding)	31.64	72.88	105.14	Per person
Soil erosion	8.25	10.51	12.78	Per person
Carbon sequestration	-	11.41	-	Per person
Air quality (woodland)	-	1.23	-	Per person
Natural water purification	0.97	59.99	87.88	Per person
		Cultural services		
Local recreation (angling, bird watching, boating)	1.68	5.59	8.87	Per person
Archaeology (built buried)	-	0.48	-	Per person
Sense of place, uniqueness	21.89	23.30	24.51	Per person
Landscape beauty: Big skies, wilderness, tranquillity	4.28	7	9.72	Per person

Appendix B

The R code used for the estimation of the Negative Binomial Regression and Poisson models in Chapter 6 is the following:

```
                                   #### set directory
getwd()
setwd("C:/Users/isoul/OneDrive/Documents/R")
                                    ### read data
data<-read.table("BR2.csv", header= TRUE, sep=",")
summary(data)
                                   #### load libraries
library(mgcv)
library(lmtest)
rm(list=ls())
library(foreign)
library(MASS)
library(nlme)
     ########## Negative Binomial Regression analysis###############
m1<-glm.nb(q1~ cost19, data= data)
summary(m1)
wtp1<--1/coef(summary(m1))
wtp1
logLik(m1)
             ############## Poisson model ###############
m2<- glm(q1 ~ cost19, family="poisson", data= data)
summary(m2)
wtp2<--1/coef(summary(m2))
wtp2
logLik(m2)
```

Data collected from the Monitor of Engagement with Natural Environment Survey (Natural England 2019) were used for the assessment of recreational services in Chapter 6. These are presented below. Concerning the meaning of variables: *q3* refers to the total visit duration in minutes; *q8* refers to the distance to visit destination; *age* refers to the age of the respondent; car and dog obtain the value 1 if the respondent owns a car or a dog; *q1* indicates the number of visits taken by the respondent over the previous seven days; *sex* denotes the gender of the respondent; *segall* denotes the social grade; *q11* refers to the form of transport used on the visit described by column *q112; cost19* refers to the total expenses as stated by the respondents.

Table C.1 Data used for the estimation application of the travel cost method.

Respondent ID	Year	q3	q8	age	car	dog	q1	sex	physical	q11	segall	q112	cost19
236671144	2011	150	0.5	55-64	0	0	1	1	2	0	E	On foot/ walking	0
263651129	2011	240	15.5	25-34	1	0	1	1	2	1	D	Car/van	29.58
1087341128	2011	180	1.5	35-44	1	1	1	1	2	1	D	Car/van	2.86
1137771128	2011	360	1.5	55-64	1	0	2	0	0	1	C1	Car/van	2.86
1195241205	2012	120	8	45-54	1	1	5	0	6	1	A	Car/van	15.41
1668091116	2011	285	4	65+	1	0	1	0	2	1	C1	Car/van	3.4
1669011116	2011	600	30.5	35-44	1	0	1	0	6	1	C2	Car/van	58.2
2175221131	2011	240	15.5	45-54	1	0	1	1	0	1	C1	Car/van	29.58
2259451118	2011	240	0.5	25-34	1	1	5	1	7	0	E	On foot/ walking	0
2538851121	2011	120	1.5	55-64	1	0	2	1	0	0	C1	On foot/ walking	0
2739931144	2011	210	15.5	55-64	1	0	1	0	2	1	C1	Car/van	29.58
3051721133	2011	900	120	65+	1	0	1	0	0	1	C2	Car/van	102.01
3581511132	2011	180	4	25-34	1	0	2	0	4	1	C1	Car/van	7.63
4759431122	2011	270	8	55-64	1	0	1	0	3	1	B	Car/van	15.27
5110581147	2011	30	0.5	65+	0	0	6	1	7	0	C1	On foot/ walking	0
5421271126	2011	180	15.5	35-44	1	0	1	0	3	1	C1	Car/van	29.58
5508081140	2011	75	1.5	65+	1	0	3	1	0	1	C2	Car/van	1.28
8642551126	2011	240	0.5	65+	1	0	5	1	4	0	B	Boat (sail or motor)	0
8673021204	2012	300	4	25-34	1	0	1	0	1	1	C2	Car/van	11.1

8704751133	2011	540	30.5	65+	1	0	1	1	0	1	C1	Car/van	25.93
8866591117	2011	480	50.5	35-44	1	0	1	1	2	1	B	Car/van	125.17
146781240	2012	180	4	16-24	1	0	12	1	0	1	C2	Car/van	7.7
961981234	2012	360	15.5	16-24	0	1	1	1	4	1	C1	Car/van	29.86
1558181220	2012	360	120	65+	1	0	5	1	3	1	E	Car/van	106.22
2819081248	2012	160	8	55-64	1	1	7	0	7	0	A	On foot/ walking	0
3552671235	2012	525	0.5	55-64	1	0	4	1	0	0	E	Boat (sail or motor)	0
3579151237	2012	60	0.5	55-64	1	1	2	1	7	0	C2	On foot/ walking	0
3874431212	2012	120	0.5	45-54	0	0	1	0	0	0	E	On foot/ walking	0
6918211240	2012	180	8	16-24	1	0	4	1	0	1	D	Car/van	15.41
7397461213	2012	120	4	35-44	1	0	1	0	7	0	D	On foot/ walking	11.33
7397761213	2012	75	1.5	65+	1	0	1	1	2	0	D	On foot/ walking	0
8154491308	2013	70	4	65+	1	0	4	1	6	1	B	Car/van	3.55
8364701209	2012	180	1.5	65+	0	0	1	1	0	1	B	Car/van	1.33
8685021233	2012	900	30.5	55-64	1	1	2	0	0	1	B	Car/van	58.75
9260251238	2012	180	4	35-44	1	1	3	0	7	1	C1	Car/van	7.7
642891316	2013	420	8	65+	1	0	2	1	2	1	C2	Car/van	7.1
1670511311	2013	120	8	45-54	1	0	1	0	1	1	B	Car/van	15.48
2537041314	2013	45	1.5	45-54	0	1	7	1	7	0	D	On foot/ walking	0
2679961407	2014	60	0.5	65+	1	0	2	1	3	0	D	On foot/ walking	0
3006891317	2013	120	4	55-64	1	1	1	1	0	0	D	On foot/ walking	0
4796871321	2013	30	30.5	65+	1	0	1	1	2	1	B	Car/van	27.05
5838621322	2013	180	15.5	65+	1	0	1	0	7	0	E	On foot/ walking	3.9
6407401336	2013	315	30.5	45-54	1	0	1	1	6	1	D	Car/van	59.03
6410261408	2014	60	0.5	65+	0	0	3	1	7	0	B	On foot/ walking	0
6415781408	2014	300	15.5	45-54	1	0	1	0	1	1	C2	Car/van	29.7
6441061404	2014	120	15.5	35-44	1	0	2	1	1	1	A	Car/van	29.7
6863601310	2013	60	1.5	65+	1	0	1	1	6	0	C1	On foot/ walking	0
7681871348	2013	60	1.5	16-24	1	1	1	1	2	0	C1	On foot/ walking	0

8153071318	2013	480	15.5	25-34	1	0	1	1	4	1	B	Car/van	30
853751410	2014	420	30.5	35-44	1	1	2	1	7	1	C2	Car/van	58.45
1285171436	2014	240	15.5	35-44	1	0	2	0	4	1	C1	Car/van	29.7
1452281437	2014	480	15.5	16-24	1	0	2	1	7	1	C2	Car/van	29.7
2309691413	2014	180	0.5	25-34	1	1	7	0	5	0	C2	On foot/ walking	0
2312591413	2014	60	0.5	65+	1	1	3	1	5	0	D	On foot/ walking	0
4071631505	2015	30	1.5	65+	1	0	2	0	2	0	C2	On foot/ walking	0
7125761412	2014	420	15.5	65+	0	0	1	1	2	1	E	Car/van	13.49
7661941503	2015	120	4	25-34	1	0	3	1	3	1	B	Car/van	7.65
8180301439	2014	480	4	55-64	1	0	1	0	0	1	A	Car/van	14.43
8739471441	2014	90	1.5	55-64	0	0	1	1	0	0	C1	On foot/ walking	0
8779361440	2014	180	4	16-24	1	0	4	0	0	1	C1	Car/van	7.67
9553211504	2015	30	0.5	25-34	0	0	1	1	3	0	C1	On foot/ walking	0
532641511	2015	180	4	65+	1	0	2	0	7	1	B	Car/van	3.5
775411552	2016	120	30.5	65+	1	0	1	0	0	1	B	Motorcycle/ scooter	26.79
896041603	2016	120	8	45-54	1	1	7	0	7	0	C2	On foot/ walking	0
1427941515	2015	60	4	65+	1	0	21	0	7	1	B	Car/van	3.5
2133901541	2015	180	1.5	35-44	1	0	1	0	2	0	B	On foot/ walking	0
2456161545	2015	240	8	55-64	1	0	1	0	0	0	D	On foot/ walking	0
3081981604	2016	120	4	35-44	1	0	6	1	4	1	C1	Car/van	10.59
3122981536	2015	720	30.5	55-64	1	1	1	0	0	1	B	Car/van	58.35
4377831511	2015	480	30.5	65+	1	1	2	0	4	1	C1	Car/van	58.35
4424511608	2016	60	4	25-34	0	0	1	0	4	1	E	Car/van	7.61
4523061606	2016	30	1.5	16-24	0	0	2	1	5	2	C1	Bicycle/ mountain bike	0
4589101517	2015	120	8	45-54	1	1	7	1	7	0	D	On foot/ walking	0
4595261527	2015	60	0.5	45-54	1	1	7	1	7	0	E	On foot/ walking	0
5122111518	2015	60	4	55-64	1	1	7	0	7	0	C1	On foot/ walking	0
5489291526	2015	420	8	65+	1	0	3	0	2	1	C1	Car/van	15.3
6161871525	2015	60	1.5	55-64	1	1	7	0	7	0	C1	On foot/ walking	0

6162111525	2015	90	1.5	65+	1	0	3	0	7	0	C2	On foot/ walking	0
6448341513	2015	180	8	45-54	1	0	1	1	4	2	C2	Bicycle/ mountain bike	0
6759721535	2015	30	1.5	35-44	1	1	4	0	4	0	B	On foot/ walking	0
6773161519	2015	240	1.5	55-64	1	1	7	1	7	1	B	Car/van	1.31
7170701540	2015	300	0.5	45-54	1	0	7	0	7	0	D	On foot/ walking	0
8791871513	2015	60	0.5	55-64	1	0	4	0	7	0	B	On foot/ walking	0
8913271536	2015	150	15.5	65+	1	0	1	1	3	1	D	Car/van	29.65
9734681524	2015	120	1.5	65+	1	0	5	0	7	1	C2	Car/van	1.31
9952261606	2016	150	4	65+	1	0	1	0	0	1	E	Car/van	7.61
8310421612	2016	180	4	16-24	1	1	1	0		1	D	Car/van	7.61
227455271630	2016	30	0.5	55-64			7	0		0	C2	On foot/ walking	0
236867701643	2016	210	4	55-64			5	0		1	C1	Car/van	7.61
256206091713	2017	300	1.5	55-64			7	0		0	E	On foot/ walking	0
260307331717	2017	60	1.5	35-44			3	1		0	A	On foot/ walking	0
273070431735	2017	120	0.5	45-54			1	1		0	C2	On foot/ walking	0
281690321747	2017	180	4	16-24			7	0		2	C1	Bicycle/ mountain bike	0
293478641813	2018	600	15.5	35-44			1	1		4	C2	Train (includes tube/underground)	11.3
302494711826	2018	60	0.5	65+			7	1		0	B	On foot/ walking	0
302496511826	2018	120	1.5	65+			1	1		0	C2	On foot/ walking	0
302795521826	2018	120	4	25-34			4	1		0	C1	On foot/ walking	0
307975711835	2018	180	4	65+			5	0		0	B	On foot/ walking	0
311682991839	2018	300	15.5	65+			1	0		1	C1	Car/van	29.39
325867871909	2019	360	30.5	35-44			1	1		1	B	Car/van	105.27
1595381052	2011	480	30.5	55-64	1	0	1	0	1	1	B	Car/van	25.93
5085631102	2011	120	15.5	55-64	1	0	1	0	2	1	B	Car/van	13.18
6300041104	2011	360	30.5	65+	1	0	1	0	0	1	D	Car/van	30.54
7254371108	2011	420	8	65+	1	0	3	0	2	1	B	Car/van	14.86
657121119	2011	60	50.5	65+	1	1	9	0	7	1	C1	Car/van	42.93
1195951128	2011	30	0.5	16-24	1	0	4	1	4	0	C2	On foot/ walking	0

1196321128	2011	180	4	55-64	1	0	1	0	6	1	B	Car/van	3.4
1284351152	2011	60	1.5	55-64	1	0	3	1	0	0	D	On foot/ walking	0
1346651207	2012	150	1.5	16-24	0	0	6	0	5	2	E	Bicycle/ mountain bike	11.33
1586081201	2012	60	4	25-34	1	0	1	0	0	1	C2	Car/van	3.54
1671941116	2011	180	4	35-44	1	0	1	0	6	1	D	Car/van	3.4
1869171127	2011	240	120	35-44	1	0	1	1	0	1	C1	Car/van	102.01
1878811208	2012	60	0.5	45-54	0	0	7	0	6	0	E	On foot/ walking	0
1889161208	2012	90	0.5	65+	1	0	1	1	0	0	C1	On foot/ walking	0
1957961135	2011	510	8	65+	1	0	1	0	7	0	C1	On foot/ walking	0
2266361118	2011	90	4	45-54	1	1	6	1	7	0	C1	On foot/ walking	0
2592741111	2011	120	0.5	35-44	1	0	2	1	3	0	D	On foot/ walking	0
2745161111	2011	60	0.5	25-34	1	0	18	0	3	0	C1	On foot/ walking	0
3544411113	2011	90	1.5	65+	1	1	7	1	7	0	E	On foot/ walking	0
3587271132	2011	90	1.5	45-54	1	1	21	0	7	0	C1	On foot/ walking	0
3759571142	2011	60	0.5	45-54	1	1	7	0	7	0	C1	On foot/ walking	0
3759831142	2011	60	0.5	65+	1	0	2	0	2	0	B	On foot/ walking	0
3760171142	2011	30	0.5	45-54	1	1	1	1	1	0	C1	On foot/ walking	0
3999561147	2011	45	0.5	55-64	1	0	5	1	2	0	B	On foot/ walking	0
4000071134	2011	480	30.5	55-64	1	1	6	0	7	1	C2	Car/van	33.88
4000331147	2011	120	0.5	16-24	0	1	5	1	0	0	D	On foot/ walking	0
4055851137	2011	180	0.5	65+	1	0	1	0	4	0	C2	On foot/ walking	0
4414501129	2011	360	50.5	16-24	1	0	2	0	7	1	C1	Car/van	42.93
4724601207	2012	300	30.5	25-34	1	0	1	1	3	1	D	Car/van	44
5032511145	2011	180	1.5	65+	0	0	1	1	0	0	E	On foot/ walking	0
5423321126	2011	180	15.5	45-54	1	1	1	0	0	1	C1	Car/van	39.67
5535111130	2011	40	0.5	35-44	0	1	6	0	7	0	E	On foot/ walking	0
7767711117	2011	480	8	16-24	1	0	2	1	0	0	E	Boat (sail or motor)	17.28
7961031115	2011	45	0.5	65+	1	0	1	1	1	0	C1	On foot/ walking	0
8723021150	2011	1440	50.5	45-54	1	0	1	1	0	1	C1	Car/van	89.01

8761061117	2011	240	30.5	65+	1	1	2	1	7	1	D	Car/van	30.52
1727321302	2013	240	4	55-64	1	0	3	1	5	1	A	Car/van	3.55
1848831234	2012	120	8	65+	1	0	1	1	0	1	E	Car/van	7.08
2385181236	2012	420	15.5	45-54	1	1	4	0	4	1	A	Car/van	13.72
2431911236	2012	240	4	35-44	1	1	13	1	7	1	C1	Car/van	3.54
2431971236	2012	180	1.5	65+	1	0	1	0	1	1	C1	Car/van	1.33
2637101301	2013	90	0.5	45-54	1	0	1	1	0	1	C1	Car/van	0.44
2674121301	2013	120	4	65+	1	1	1	0	7	0	C1	On foot/ walking	0
2740251240	2012	120	4	45-54	1	0	1	0	0	1	B	Car/van	3.54
2770951214	2012	300	4	65+	1	1	2	0	7	1	C2	Car/van	3.54
2823631236	2012	480	50.5	65+	1	0	2	0	7	1	A	Car/van	44.7
3033871224	2012	90	0.5	35-44	1	1	7	1	7	0	C2	On foot/ walking	0
3049331224	2012	30	0.5	65+	1	0	2	1	2	0	C1	On foot/ walking	0
4114431235	2012	330	15.5	45-54	1	1	1	1	1	1	B	Car/van	21.65
4802081308	2013	120	4	65+	0	0	2	0	3	0	D	On foot/ walking	0
5228021305	2013	210	15.5	25-34	1	0	2	1	4	1	C2	Car/van	13.75
6180961242	2012	180	15.5	45-54	1	0	2	0	3	1	C1	Car/van	13.72
6507081229	2012	180	0.5	25-34	1	0	1	0	1	0	D	On foot/ walking	0
7756251301	2013	120	0.5	16-24	1	0	3	0	4	0	C2	On foot/ walking	0
8392741304	2013	360	4	35-44	0	0	1	0	5	0	C1	On foot/ walking	0
8724341226	2012	330	1.5	55-64	1	0	2	0	7	1	D	Car/van	1.33
8727561226	2012	20	1.5	65+	1	1	3	1	0	0	C2	On foot/ walking	0
9088641238	2012	150	15.5	25-34	1	0	1	1	0	1	B	Car/van	13.72
890991409	2014	120	4	25-34	0	0	1	1	5	0	E	On foot/ walking	0
1687591341	2013	240	0.5	45-54	0	0	3	0	7	0	C2	On foot/ walking	0
2094521332	2013	240	1.5	35-44	0	0	1	0	3	0	E	On foot/ walking	0
2193441343	2013	60	1.5	65+	1	0	7	0	7	0	B	On foot/ walking	0
2492021328	2013	60	0.5	16-24	0	1	13	1	7	0	C1	On foot/ walking	0
2550441405	2014	45	1.5	65+	1	0	14	0	7	0	C2	On foot/ walking	0

2553091405	2014	60	1.5	65+	1	1	7	1	0	0	A	On foot/ walking	0
2561181342	2013	65	0.5	55-64	0	1	7	1	7	0	E	On foot/ walking	0
2591621337	2013	90	4	65+	1	1	1	0	7	1	E	Car/van	3.55
2810731327	2013	60	1.5	65+	0	1	8	1	7	0	C2	On foot/ walking	0
2817201327	2013	180	15.5	55-64	1	0	6	0	2	1	B	Car/van	13.75
2933141316	2013	360	8	25-34	1	1	2	0	7	1	D	Car/van	7.1
3018161317	2013	120	0.5	25-34	1	1	3	1	7	0	D	On foot/ walking	5.01
3024511317	2013	120	1.5	45-54	0	1	13	0	7	0	D	On foot/ walking	0
5211871338	2013	240	15.5	45-54	1	0	1	0	3	1	D	Car/van	13.75
5375381338	2013	120	15.5	65+	1	0	1	1	7	1	B	Car/van	13.75
5755001320	2013	120	4	25-34	0	0	1	1	5	0	D	On foot/ walking	0
5937151337	2013	150	8	45-54	1	0	2	0	2	1	C1	Car/van	7.1
6674991406	2014	20	0.5	55-64	1	1	2	1	4	0	E	On foot/ walking	0
6926191311	2013	270	1.5	35-44	1	0	1	0	5	1	D	Car/van	1.33
7172691351	2013	60	0.5	65+	1	0	1	0	0	0	B	On foot/ walking	5.57
7658071348	2013	180	8	45-54	0	1	1	0	7	1	C2	Car/van	7.1
7733111402	2014	25	0.5	65+	1	0	4	1	0	0	B	On foot/ walking	0
8114151343	2013	60	0.5	55-64	1	1	7	0	2	0	B	On foot/ walking	0
8374031340	2013	330	8	65+	1	0	4	1	5	1	B	Car/van	7.1
8792521339	2013	720	1.5	35-44	1	0	1	0	0	1	C1	Car/van	1.33
9138441326	2013	90	15.5	65+	1	0	3	0	7	1	B	Car/van	15.75
140381418	2014	360	8	55-64	1	0	3	0	4	1	A	Car/van	6.96
142461508	2015	90	0.5	16-24	0	1	2	0	4	0	D	On foot/ walking	0
161561436	2014	240	15.5	65+	0	0	1	1	0	1	D	Car/van	13.49
360851444	2014	180	1.5	35-44	1	0	1	1	1	0	B	On foot/ walking	5.47
521501423	2014	270	8	65+	1	0	1	0	0	3	C1	Coach trip/ private coach	6.96
587891442	2014	420	70.5	65+	1	1	1	1	7	1	D	Car/van	61.35
2807021503	2015	240	4	55-64	1	0	2	1	3	0	A	On foot/ walking	0
3017641421	2014	150	15.5	65+	1	1	1	0	1	1	D	Car/van	24.43

3277351434	2014	240	15.5	16-24	1	0	1	0	5	2	D	Bicycle/ mountain bike	4.38
3304691429	2014	60	0.5	16-24	1	1	2	1	0	0	C2	On foot/ walking	0
3653371412	2014	30	0.5	65+	1	1	14	0	7	0	B	On foot/ walking	0
4274181507	2015	180	8	16-24	1	1	3	0	0	1	D	Car/van	15.3
7040281427	2014	60	1.5	25-34	1	1	7	0	7	0	C2	On foot/ walking	0
7295441415	2014	120	1.5	25-34	1	0	1	1	4	0	B	On foot/ walking	0
7601421415	2014	240	0.5	25-34	0	0	2	1	4	0	C2	On foot/ walking	0
7636381438	2014	45	0.5	55-64	1	1	5	0	3	0	A	On foot/ walking	0
9316781440	2014	36	0.5	35-44	1	1	7	0	7	0	C1	On foot/ walking	0
869821540	2015	90	0.5	45-54	0	0	1	0	5	0	E	On foot/ walking	0
895421603	2016	240	8	25-34	1	1	2	1	7	0	C1	On foot/ walking	0
4345751603	2016	240	1.5	16-24	0	0	2	0	3	3	C1	Public bus or coach (scheduled service)	5.96
4377601511	2015	120	30.5	55-64	1	0	3	1	7	1	B	Car/van	26.65
4810641521	2015	120	1.5	55-64	1	1	4	1	3	0	C1	On foot/ walking	0
5823751523	2015	210	70.5	16-24	0	0	1	1	3	0	B	On foot/ walking	0
6933941519	2015	60	1.5	65+	1	1	21	1	7	0	B	On foot/ walking	0
8025991534	2015	180	0.5	45-54	0	1	7	1	0	0	C1	On foot/ walking	0
8369571524	2015	420	4	65+	1	0	1	0	7	1	D	Car/van	3.5
2962571621	2016	240	4	25-34			7	0		1	C1	Car/van	7.61
3637861609	2016	60	4	55-64	1	0	1	0		0	D	On foot/ walking	0
233597951639	2016	90	0.5	55-64			7	0		0	B	On foot/ walking	0
244220831650	2016	60	4	65+			2	1		1	E	Car/van	3.51
256082921713	2017	300	8	55-64			1	0		1	C1	Car/van	15.29
281694491747	2017	45	0.5	65+			7	0		0	C2	On foot/ walking	0
289120211808	2018	480	4	45-54			4	0		1	E	Car/van	28.01