UNIVERSITÀ DEGLI STUDI DI PADOVA
Dipartimento Territorio e Sistemi Agro-forestali
Department of Land, Environment Agriculture and Forestry

Corso di laurea magistrale/Second Cycle Degree (MSc) in Forest Science

Evaluation of water-related ecosystem services

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Abbreviations and acronyms

ARPAV	Agenzia Regionale per la Prevenzione e Protezione Ambientale del Veneto
CAP	Common Agricultural Policy
CE	Choice Experiment
CICES	Common International Classification of Ecosystem Services
CM	Choice Modeling
CS	Compensating Surplus
CV	Contingent Valuation
DCE	Discrete Choice Experiment
EU	European Union
ES	Ecosystem Services
HP	Hedonic Pricing
IPBES	Intergovernmental Science–Policy Platform on Biodiversity and Ecosystem Services
MA	Millennium Ecosystem Assessment
MNL	Multinomial Logit
OP	Observed Preference
PES	Payments for Ecosystem Services
PMT	Protection Motivation Theory
RP	Revealed Preference
RUM	Random Utility Model
SP	Stated Preference
SDT	Self-Determination Theory
SCI	Sites of Community Importance
SPA	Special Protection Areas
TC	Travel Cost
TEEB	The Economics of Ecosystem and Biodiversity
TEV	Total Economic Value
TPB	Theory of Planned Behaviour
UNSD	United Nations Statistical Division
VBN	Value–Belief–Norm
WTA	Willingness to Accept
WTP	Willingness to Pay

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Summary

Water-related ecosystems, such as wetlands, rivers, and spring-fed systems, provide essential ecological and socio-economic benefits, including biodiversity maintenance, water purification, flood regulation, and cultural values. Despite their global importance, small and groundwater-dependent wetlands remain underrepresented in environmental economic research. This thesis aims to bridge this gap by conducting a systematic review of scientific studies employing Discrete Choice Experiments (DCEs) to value water-related ecosystem services (ES), with a particular focus on how motivational and psychological factors influence individuals' willingness to pay (WTP) for conservation. A total of 169 peer-reviewed studies published between 1999 and 2025 across 35 countries (mainly China, Italy, the USA, and Australia) were reviewed and analyzed to identify recurrent environmental attributes, methodological patterns, and gaps in the literature. Results show a notable rise in DCE applications after 2018, reflecting growing global interest in valuing water and wetland services. From these studies, 543 main characteristics (attributes) were identified and grouped into 12 categories, primarily related to water services, environmental quality, landscape and recreation, and biodiversity, which together account for 67% of all attributes. The study also examined how motivational theories, including the Value–Belief–Norm (VBN), Theory of Planned Behaviour (TPB), and Self-Determination Theory (SDT), have been integrated into environmental valuation, enhancing understanding of the drivers behind pro-environmental behavior. Findings reveal that while DCEs have become a central tool for quantifying preferences for non-market environmental goods, most applications remain grounded in economic rationality and neglect intrinsic motivations and relational values that underpin conservation attitudes. By synthesizing ecological and behavioral dimensions, this work establishes a reference framework for the design of a future DCE survey in the Risorgive del Bacchiglione (Veneto, Italy) area, a Natura 2000 spring-fed wetland system of high ecological and cultural significance. The framework supports the inclusion of motivational scales in survey design, linking respondents' WTP with their environmental values, identity, and perceived efficacy. This integrative approach contributes to the development of context-sensitive, socially legitimate, and behaviorally informed conservation strategies. Ultimately, the thesis highlights the importance of combining economic valuation with insights from psychology, sociology, and environmental ethics to advance ecosystem service assessment. Recognizing and quantifying how people relate to and value water ecosystems can strengthen conservation policies, improve resource allocation, and foster sustainable environmental governance. The results provide suggestions for interdisciplinary research and practical decision-making aimed at protecting vulnerable freshwater ecosystems.

Keywords: Discrete Choice Experiment; Ecosystem Services; Water Ecosystems; Motivational Theories; Willingness to Pay; Behavioral Economics; Risorgive del Bacchiglione; Wetlands Valuation.

1 Introduction

Water-related ecosystems, including wetlands, rivers, springs, lakes, and groundwater-dependent systems, represent some of the most ecologically productive and socio-economically valuable environments on Earth. They sustain biodiversity, regulate hydrological and biogeochemical cycles, buffer against floods and droughts, and provide cultural and aesthetic values that enhance human well-being (Bongaarts, 2019; Vollmer *et al*., 2022). These ecosystems deliver a portfolio of ecosystem services (ES) that underpin nutrient cycling, water purification, climate regulation, recreation, and food production (Costanza *et al*., 2017; Raymond *et al*., 2023). Recent global syntheses emphasize that maintaining freshwater and wetland ecosystems is indispensable for meeting the United Nations Sustainable Development Goals on clean water, climate action, and biodiversity (Baste & Watson, 2021).

Despite their crucial importance, wetlands and groundwater-dependent ecosystems remain among the most threatened biomes worldwide. The Global Wetland Outlook: Special Edition 2021 of the Ramsar Convention reported that roughly 35 per cent of natural wetlands have disappeared since 1970, primarily because of urbanization, agricultural drainage, and pollution (Convention on Wetlands, 2021). Such losses erode biodiversity, nutrient retention, and flood-regulating functions, while also diminishing cultural and recreational benefits (Wood *et al*., 2024). The ecological decline of wetlands, therefore, carries cascading consequences for climate resilience and human livelihoods. According to Ramsar (2021), continued degradation contributes to both greenhouse gas emissions and reduced adaptive capacity to extreme hydrological events.

The Intergovernmental Science–Policy Platform on Biodiversity and Ecosystem Services (Bongaarts, 2019; IPBES, 2022) identifies freshwater degradation as a leading driver of biodiversity loss and urges adoption of “plural valuation” approaches that capture economic, cultural, and relational values of nature. International frameworks such as the Kunming–Montreal Global Biodiversity Framework (2022) and the EU Biodiversity Strategy for 2030 reinforce this paradigm shift, promoting ecosystem restoration and the mobilization of innovative finance, including Payments for Ecosystem Services (PES), to reward conservation behavior (European Comission, 2024). In Europe, the Water Framework Directive (2000/60/EC) established river-basin management plans to achieve “good ecological status” for all waters (The European Parliament and the Council of the European Union, 2000). The Habitats Directive (92/43/EEC) and Birds Directive (2009/147/EC) created the Natura 2000 network of protected sites, while the new EU Nature Restoration Law (2024) sets legally binding restoration targets, including re-wetting of drained peatlands and enhancement of wetland connectivity (European Comission, 2024). Yet significant implementation gaps persist, as small or spring-fed wetlands often lie outside designated conservation areas and receive limited financial or policy attention (Vollmer *et al*., 2022).

Within Italy, spring-fed wetlands, known as risorgive, are emblematic of the Veneto Plain, where groundwater resurfaces at the boundary between Alpine aquifers and lowland alluvial soils. These ecosystems provide essential hydrological services such as groundwater recharge, nutrient filtration, and base-flow stabilization while hosting unique biodiversity (Rossetti *et al*., 2020). However, they are increasingly endangered by groundwater over-extraction, agricultural intensification, and infrastructure expansion (Condon & Maxwell, 2014; Sishodia *et al*., 2017). The Risorgive del Bacchiglione, located in the province of Vicenza,

exemplify this multifunctionality by combining ecological, recreational, and cultural values deeply rooted in local identity. Hydrogeological analyses reveal declining groundwater levels in the Bacchiglione basin, highlighting the urgency of integrative management (Meggiorin *et al*., 2021). Although the Veneto Region's Piano Territoriale Regionale di Coordinamento (PTRC) recognizes risorgive as heritage landscapes, their ecosystem services remain poorly quantified and undervalued in regional decision-making.

In response to the global need for sustainable resource allocation, economic valuation of ecosystem services has emerged as a key approach to linking ecological functions with human welfare. The Total Economic Value (TEV) framework distinguishes use, option, and non-use values, providing a structured means to evaluate benefits that markets fail to capture (Costanza *et al*., 2017). Among non-market valuation techniques, Discrete Choice Experiments (DCEs) have become particularly influential because they estimate how individuals trade off multiple attributes of environmental goods to infer their willingness to pay (WTP) (Hanley & Czajkowski, 2019; Mariel *et al*., 2021). DCEs grounded in random-utility theory simulate hypothetical markets and quantify preferences for ecological improvements, such as cleaner rivers or restored wetlands (Johnston *et al*., 2017). Recent applications illustrate the method's versatility: Dai *et al*. (2021) evaluated urban river restoration in China; (Ruberto *et al*., 2022) measured irrigation-canal ecosystem services in Lombardy, Italy; and Udugama *et al*. (2024) valued blue-ecosystem services in Sri Lankan natural pools. Technological advances, including random-parameter and latent-class logit models, have increased behavioral realism by capturing preference heterogeneity and information effects (Mariel *et al*., 2021). Nonetheless, conventional DCEs largely focus on monetary trade-offs and overlook deeper psychological and motivational determinants of environmental choices.

Interdisciplinary evidence shows that individuals' valuation of nature is shaped not only by economic rationality but also by intrinsic motivations such as moral obligation, biospheric concern, altruism, and identity (Gkargkavouzi *et al*., 2019b; Steg *et al*., 2014). Theories including the Value–Belief–Norm (VBN) model, the Theory of Planned Behaviour (TPB), and Self-Determination Theory (SDT) explain how internalised values and perceived control influence pro-environmental intentions. Integrating these frameworks into economic valuation enhances explanatory power and reveals why people support conservation beyond financial considerations. More recently, scholars aligned with IPBES have advanced the concept of relational values, the meaningful relationships and responsibilities connecting people and nature, which motivate stewardship and collective action (Pratson *et al*., 2023; Raymond *et al*., 2023). Empirical studies demonstrate that incorporating relational or motivational indicators into stated-preference models significantly improves predictive accuracy and policy relevance (Lengieza & Aviste, 2025). Individuals with strong place attachment or environmental identity tend to exhibit higher, more consistent WTP for ecosystem improvements, underscoring that valuation must account for affective and moral dimensions alongside utility maximization.

For spring-fed wetlands such as the Risorgive del Bacchiglione, this integrative approach is particularly relevant. These systems deliver tangible hydrological and ecological services while embodying intangible cultural, historical, and aesthetic meanings. Their conservation depends on voluntary participation, community engagement, and cross-sector collaboration among farmers, municipalities, and environmental agencies. Applying DCEs that explicitly include

motivational and relational attributes can generate richer insights into community preferences and identify leverage points for behaviorally informed policy instruments. Potential applications include the design of localized PES schemes, eco-tourism incentives, and participatory restoration programs that align with citizens' intrinsic values. In doing so, valuation becomes not merely a measure of willingness to pay but a means of articulating how people connect with and care for their environments.

Ultimately, while the theoretical and methodological foundations of ES valuation are well established, major gaps persist in bridging economic, psychological, and cultural perspectives. Addressing these gaps is crucial for developing socially legitimate and ecologically sound wetland-management strategies. The present study responds to this need by synthesizing contemporary evidence on DCE-based valuation of water-related ES and integrating motivational and relational value frameworks to inform sustainable management of Italy's Risorgive del Bacchiglione.

1.1 Problem Statement

Despite their ecological and socio-cultural importance, small and spring-fed wetlands remain underrepresented in ES valuation literature relative to large or coastal systems in Europe. Many studies prioritize monetary indicators without adequately considering psychological and motivational drivers behind public preferences. This limits the interpretability and transferability of results for policy design, particularly where voluntary support, local legitimacy, and behavioral engagement are necessary for conservation success.

There is thus a clear need for an integrative framework that (i) applies state-of-the-art DCEs to water-related ES, (ii) explicitly incorporates motivational theories, and (iii) is context-sensitive to the hydro-ecological and cultural specificities of Risorgive del Bacchiglione.

1.2 Research Questions

i. How has existing research used DCEs to value water-related ecosystem services?

ii. What are the most common environmental characteristics (known as attributes) identified in these valuation studies?

iii. In what ways have motivational theories been incorporated into the valuation of water ecosystem service?

iv. What are the key gaps and future directions for research on valuing water-related ecosystem services through DCEs?

1.3 General Objective

To review and synthesize the existing scientific literature on the valuation of water-related ecosystem services using DCEs, with attention to how motivational and behavioral dimensions are considered in studies related to the conservation and restoration of aquatic and wetland ecosystems.

1.4 Specific Objectives

i. To provide an overview of how DCE methods have been applied to assess water-related ecosystem services across different ecological contexts, such as wetlands, rivers, and springs.
ii. To identify key environmental attributes relevant to water ecosystem valuation and integrate them into a DCE future survey in Risorgive del Bacchiglione.
iii. To analyze the influence of psychological, motivational or behavioral variables (e.g., environmental concern, altruism, perceived behavioral control) on respondents' choices regarding water ecosystem valuation and integrate them into a DCE future survey in Risorgive del Bacchiglione.
iv. To highlight gaps, challenges, and emerging trends in literature that can inform future research and policy related to sustainable water ecosystems management and decision-making.

1.5 Thesis Structure and Organization

This thesis is structured into six chapters.

Chapter 1 introduces the background and context of the study, outlines the research problem, defines the research questions, and establishes the general and specific objectives.

Chapter 2 presents the framework on which the thesis is based. The first section discusses the most recognized definitions of ecosystem services (ES), the evolution of the concept, and the importance of integrating ES into decision-making processes, with a particular focus on water-related ecosystem services. The second section explores water ecosystems, highlighting their ecological and socio-economic importance. The third section addresses the economic valuation of water ecosystems, presenting the Total Economic Value (TEV) framework and the main valuation methods. The fourth section focuses on the Discrete Choice Experiment (DCE) approach, discussing its conceptual foundations, choice experimental design, econometric aspects, welfare measures, and applications of DCE in wetlands. Finally, the fifth section examines motivational theories relevant to understanding environmental attitudes and behaviors.

Chapter 3 describes the materials and methods used to obtain the results, including the characterization of the study site (Risorgive del Bacchiglione), the data collection procedures, and the data analysis approach. It provides sufficient detail to enable a competent professional to replicate the study and reproduce the results without difficulty.

Chapter 4 analyzes the data obtained through the methodology to address the research objectives. Presents the results of the literature review, organized into three sections. The first section provides an overview of how DCE methods have been applied to the valuation of water-related ecosystem services, followed by the identification of the most recurrent attributes to be considered in a future field study in Risorgive del Bacchiglione. The third and final section contributes to the development of the future survey, by compiling the main motivational theories identified in the literature on water ecosystem valuation, with the aim of integrating socio-psychological aspects into the questionnaire.

Chapter 5 revisits the objectives of the study, providing a critical analysis that validates the conclusions drawn from the results and discusses the relevance of the main findings. Develops the choice experiment design for future research, focusing

on the use of entrance and parking fees as tools to manage visitor flow and capture visitors' WTPmfor water-related ecosystem services (ES). Also, includes the theoretical, managerial, and policy discussions derived from the results, highlights the limitations identified throughout the research, and proposes recommendations for public policies and future studies.

Chapter 6 presents the main conclusions derived from the results and discussions, highlighting the key findings, policy implications, and recommendations for future research.

Finally, the last part of the document includes the list of references used throughout the study.

Moreover, Annex 1 includes additional supporting materials, that is, the list of attributes identified in the literature review.

2 Theoretical Background

2.1 Ecosystem Services (ES)

2.1.1 Definitions of the Concept and Origins of the Modern History of Ecosystem Services (ES)

In ecology, the term *ecosystem function* has traditionally referred to the set of processes occurring within an ecological system, regardless of whether these processes are beneficial to humans. Over time, however, several contributions began to highlight how specific “functions of nature” serve human societies. It was in this context, during the late 1970s, that the concept of *ecosystem services* emerged (Gómez-Baggethun *et al*., 2010). The idea was initially framed from a utilitarian perspective, presenting beneficial ecological functions as services to society, with the aim of increasing public interest in biodiversity conservation. Throughout the 1970s and 1980s, a growing number of scholars began expressing ecological concerns in economic terms to underscore the dependence of human society on natural ecosystems and to raise awareness about biodiversity loss (Gómez-Baggethun *et al*., 2010). The core objective was to show how the disappearance of biodiversity undermines ecosystem functions that are critical for human well-being. However, at this early stage, the notion of ecosystem services was used mainly as a communication tool and had not yet been integrated into economic processes of valuation, appropriation, or exchange.

The debate gained momentum in the 1990s, when the term “ecosystem services” became firmly established in scientific literature, accompanied by growing interest in methods to estimate their economic value (Braat & de Groot, 2012). The *Millennium Ecosystem Assessment* (MA, 2005) played a key role in placing ecosystem services at the center of environmental policy discussions, and since its publication, the literature on the subject has expanded rapidly. While emphasizing an anthropocentric approach, the MA framework also underscored human reliance not only on ecosystem services but on the ecological functions that underpin them, helping to highlight the essential role of biodiversity and ecological processes in supporting human well-being. Monetary valuation of ecosystems saw significant growth in the 1990s, as more natural scientists recognized the strategic value of framing ecological concerns in economic terms to influence policy and decision-making (Gómez-Baggethun *et al*., 2010).

2.1.2 Classification of ES: Provisioning, Regulating, Cultural, and Supporting Services

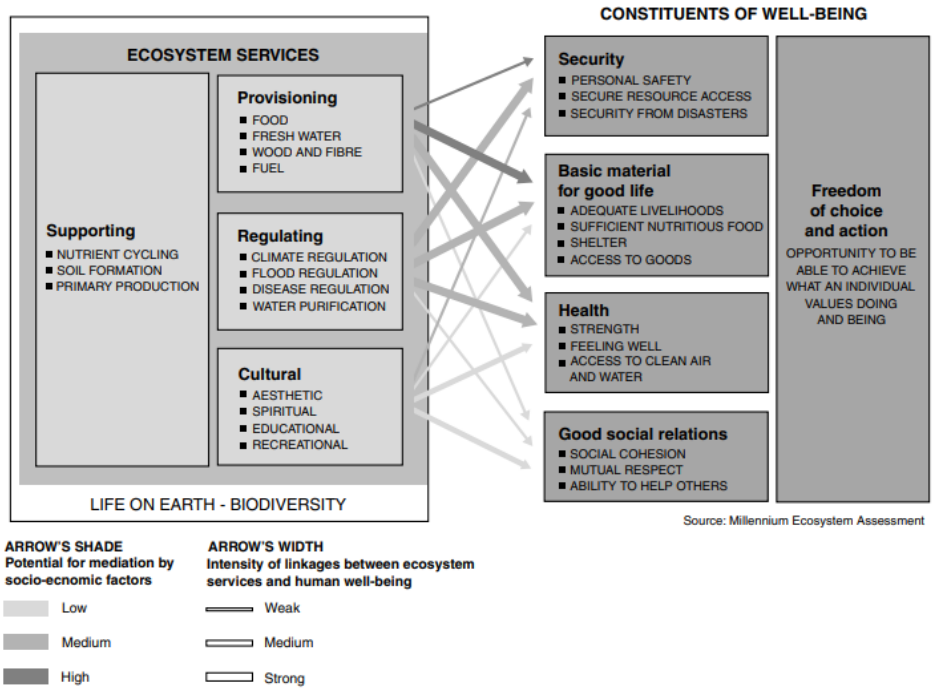
The concept of ecosystem services presents significant challenges when attempting to measure them or develop a classification system that allows for clear and standardized reporting of results (Potschin & Haines-Young, 2011).

A comprehensive assessment of the condition and importance of ecological systems to human well-being was carried out under the United Nations Environment Programme (UNEP) through the Millennium Ecosystem Assessment (MA), published in 2005. The MA proposed a typology that categorized ecosystem services into four broad groups: provisioning services (e.g., food, water, raw materials), regulating services (e.g., climate regulation, water purification), cultural

services (e.g., spiritual, recreational, and aesthetic values), and supporting services, which sustain the other three categories by maintaining ecological processes such as nutrient cycling and soil formation.

This conceptual model aims to emphasize the benefits or usefulness of ecosystem services to humans and to inform policy and planning decisions. It visually represents a flow from the natural science domain (on the left) to the social, human, and economic domains (on the right), indicating the transition from ecological structures and processes to human well-being (Figure 1). The diagram highlights key ecosystem components and dimensions of well-being, with arrow widths suggesting the relative importance of the connections between them.

Figure 1. Ecosystem Services as defined by MA and their impact on human well-being



Source: MA (2005)

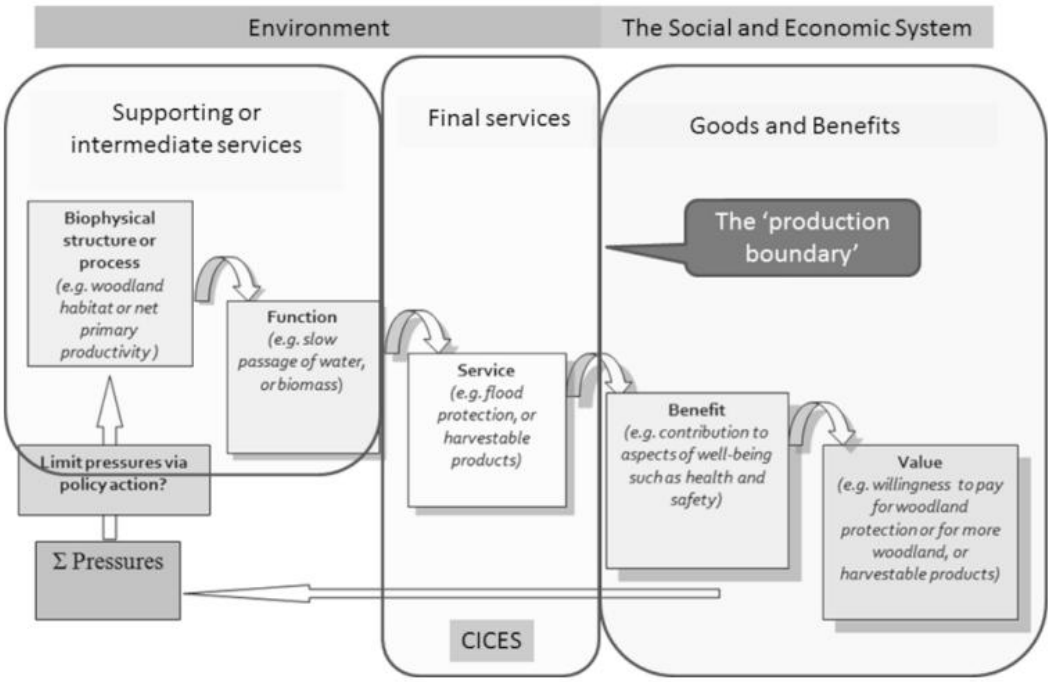
However, a key critique of the MA typology is that it blurs the distinction between means and ends, that is, between the ecological mechanisms that generate services (often referred to as ecosystem functions) and the benefits experienced by people. According to critics such as Wallace (2007), a true ecosystem service should refer only to what is directly consumed or experienced by people. Supporting services, therefore, differ from the other categories, as they do not provide direct benefits but rather enable the delivery of other services. This conceptual ambiguity has led to challenges in clearly defining and applying the MA framework.

Building on the MA, UNEP launched a second major initiative in 2007: The Economics of Ecosystems and Biodiversity (TEEB). This effort aimed to integrate the values of biodiversity and ecosystem services into public and private decision-making. TEEB promotes a structured valuation approach that helps decision-makers recognize, demonstrate, and, when appropriate, incorporate ecosystem values, particularly in economic terms, into policy and planning processes (TEEB

Foundations, 2010). The TEEB report received significant media attention, helping to broaden the reach of the ecosystem services concept to a wider audience.

The TEEB conceptual framework adopts a similar diagrammatic structure to the MA model, with ecosystem services flowing from left (natural systems) to right (human and economic systems). However, it makes important refinements: ecosystem services are placed explicitly at the interface between natural and human systems, and a clear distinction is drawn between services, benefits, and values. TEEB also more clearly illustrates that services arise from ecological structures, processes, and functions. Its diagram is an evolution of the cascade model introduced by Haines-Young and Potschin (2009), which emphasizes the stepwise transition from ecosystem properties to human values (Figure 2).

Figure 2. The cascade model adapted from Potschin and Haines-Young (2011)



Source: Potschin, M. and R. Haines-Young (2011).

Several authors have pointed out the challenges in precisely defining what constitutes an ecosystem service (Haines-Young & Potschin-Young, 2010). Despite differences in their approaches, there is broad agreement that a pathway exists linking ecosystems to human well-being, a process that begins with ecological structures and processes and culminates in tangible benefits for people.

In this context, the cascade model emerges as a valuable conceptual tool to represent these connections. It outlines a logical sequence that starts with ecosystem structures and processes, progresses through functions (the capacities of ecosystems), leads to ecosystem services (final outputs useful to people), which translate into benefits (direct impacts on human well-being), and ultimately into the values that society attributes to those benefits (Haines-Young & Potschin-Young, 2010).

In the version of the cascade model presented in Figure 2, a clear distinction is made between benefits and values, based on the idea that if benefits are

understood as gains in human well-being provided by ecosystems, then it is reasonable to assume that different groups may perceive and value these gains differently, depending on time, location, and social context (Potschin & Haines-Young, 2011). Despite this refinement, the core principle of the ecosystem services paradigm remains unchanged: a service is only considered as such if a human beneficiary can be clearly identified, and it is essential to distinguish between “final services”, which directly contribute to human well-being, and “intermediate ecological structures and functions” that support their provision (Potschin & Haines-Young, 2011).

The distinction between intermediate and final services is considered essential by authors such as Boyd and Banzhaf (2007) and Fisher *et al*. (2009), as it helps avoid double counting in valuation exercises. According to these authors, valuation should be limited to elements directly consumed or used by the beneficiary, since the value of the ecological structures and processes that enable this use is already embedded in the estimate (Potschin & Haines-Young, 2011). In other words, because a single ecological structure or function can support multiple services, it should be accounted for only once in valuation processes.

In this model, *function* refers to the ecosystem's capacity to do something potentially beneficial for humans. This interpretation is widely adopted by authors such as De Groot (1992), De Groot *et al*. (2002), Brown *et al*. (2007), Costanza *et al*. (1997), and Daily (1997), who use it as a foundation for their conceptual approaches to ecosystem services. For example, forests and wetlands in a watershed may have the function of slowing surface water runoff, which can reduce the severity of floods. While this is not an intrinsic property of the ecosystem, it is perceived as useful by humans, thus justifying its classification as a function. Whether it is considered an ecosystem service, however, depends on whether flood control is perceived as a valued benefit.

Ecosystem services occupy a central position, defined as the final outcomes of ecosystem functioning that, although connected to ecological structures and processes, directly generate goods or conditions valued by people.

Goods (or product) and benefits are understood as positive contributions to human well-being. These benefits gain importance as they are assigned values, which may be monetary (e.g., market value) or non-monetary (e.g., moral, aesthetic, or spiritual). These values guide societal and individual decisions on whether and how to manage or preserve ecosystems and the services they provide.

Despite its strengths, the cascade model has limitations. One major criticism is that it suggests a linear relationship between ecological components and human outcomes, while these interactions are complex, dynamic, and interconnected. Nevertheless, the model offers useful vocabulary for analyzing and communicating relationships, making it a valuable framework for understanding and conveying the links between nature and society in a structured and accessible way.

Due to the lack of consensus regarding the terminology of ecosystem services and the absence of alignment among service categories across different classification systems, the Common International Classification of Ecosystem Services (CICES) was developed within the revision process of the System of Environmental-Economic Accounting (SEEA), led by the United Nations Statistical Division (UNSD) (Haines-Young & Potschin, 2013). The main goal of CICES is to support the identification of final ecosystem services and to facilitate navigation across various existing typologies, promoting standardization in the classification, measurement, and reporting of results.

This classification system is the result of an ongoing refinement process. From the outset in 2009, a key guiding principle was to ensure, whenever possible, compatibility with other widely used classification systems, particularly regarding terminology (Potschin & Haines-Young, 2011). Thus, the basic structure of CICES shows correspondences with the categories used in the typologies of the Millennium Ecosystem Assessment (MA) and The Economics of Ecosystems and Biodiversity (TEEB). Accordingly, CICES adopted the typology proposed by the Millennium Ecosystem Assessment (MA, 2005) as its starting point, which was then refined to meet the needs for standardization and practical applicability.

The most widely used version of CICES was published in 2013 and revised in 2017 (CICES, 2025). At the highest level of its hierarchy, the system retains the three main categories established by the Millennium Ecosystem Assessment (MA, 2005): provisioning, regulation and maintenance, and cultural services (Table 1).

Table 1. Definition of the three Sections

Section	Description
Provisioning	Covers all nutritional, non-nutritional material and energetic outputs from living systems as well as abiotic outputs (including water)
Regulation and Maintenance	All the ways in which living organisms can mediate or moderate the ambient environment that affects human health, safety or comfort, together with abiotic equivalents
Cultural	All the non-material, and normally non-rival and non-consumptive, outputs of ecosystems (biotic and abiotic) that affect physical and mental states of people

Source: CICES (2025)

The CICES framework is structured into five progressively specific levels: Section (e.g., Provisioning), Division (e.g., Biomass), Group (e.g., Cultivated terrestrial plants for nutrition, materials, or energy), Class (e.g., Cultivated terrestrial plants, including fungi and algae, grown for nutritional purposes), and Class Type (e.g., Cereals, the ecological contribution to the growth of land-based crops that can be harvested and used as raw material for food production).This hierarchical organization allows users to select the level of detail most appropriate for their application, while also enabling the aggregation and generalization of data for broader comparisons and reporting.

As one moves from Section to Division, Group, Class, and finally Class Type, the ecosystem services become increasingly specific, with detailed service types nested within broader categories. The system is built on a hierarchical dependency, where characteristics used to define services at lower levels are inherited from higher-level categories. Additionally, it has a taxonomic structure, meaning that elements within the same Group or Class are conceptually more like each other, in terms of how they are used by people, than they are to services in other parts of the classification. At every level of the hierarchy, the categories are designed to be mutually exclusive and non-overlapping, ensuring that CICES functions as a true classification system rather than a loose or arbitrary nomenclature (Potschin & Haines-Young, 2011).

A key distinction between CICES and the typology proposed by the MA is that CICES does not include "supporting services", as shown in Table 1. This exclusion does not imply that such services are considered unimportant; rather, it reflects a methodological decision to focus on identifying final ecosystem services, that is, the

outcomes directly perceived and valued by people, which can serve as the basis for valuation and assessment (Potschin & Haines-Young, 2011). As illustrated in Figure 2, CICES aims to classify services that lie at the interface between the environmental and socio-economic components of integrated socio-ecological systems.

In any ecosystem assessment, once the most relevant final services are identified, discussions around sustainability and appropriate management strategies should then shift toward the underlying ecological structures, processes, and functions that support them (Potschin & Haines-Young, 2011). In this context, final services serve as entry points for deeper analysis.

One of the most complex issues highlighted by the development of CICES is the distinction between ecosystem services and benefits. For those who view ecosystem services as the actual benefits provided by nature, this distinction poses little difficulty. However, for those who advocate a clearer conceptual separation, where *services* are defined as *ecosystem functions or processes that generate benefits*, and *benefits* as *the various ways in which human well-being is enhanced through those services*, the challenge becomes primarily terminological.

This distinction becomes particularly problematic in the classification of cultural services (Potschin & Haines-Young, 2011). Several authors suggest that cultural services should be understood as ecosystem contributions to non-material benefits, such as experiences, capabilities, or symbolic meanings, which arise from the interactions between people and nature (Potschin & Haines-Young, 2011).

2.1.3 Water-Related Ecosystem Services (ES)

Water-related ecosystem services encompass the multifaceted benefits derived from ecosystems through their influence on the quantity, quality, and distribution of water across landscapes (Brauman *et al*., 2016; Grizzetti *et al*., 2019). These services sustain the hydrological balance essential for agriculture, industry, domestic use, and ecological integrity. They underpin almost every aspect of human life, linking natural systems with social and economic well-being.

Freshwater ecosystems, including rivers, lakes, wetlands, aquifers, and springs, provide a diverse array of ecosystem services. Provisioning services supply water for drinking, irrigation, and energy generation; regulating services moderate floods, recharge aquifers, and purify water; and cultural services contribute to recreation, aesthetic enjoyment, spiritual enrichment, and heritage value (Raymond *et al*., 2023). These services operate synergistically: a healthy riverine or wetland system simultaneously filters pollutants, supports biodiversity, and enhances local identity and recreation opportunities.

Globally, the demand for freshwater ecosystem services continues to grow while their supply is declining due to population increase, land-use change, industrial pollution, and climate variability (UNEP, 2022). The United Nations warns that approximately one-third of the world's freshwater ecosystems have deteriorated significantly over the past four decades, leading to reduced resilience and diminished human welfare (Convention on Wetlands, 2021). In Europe, the Water Framework Directive (2000/60/EC) established a pioneering model for integrated water resource management, aiming to restore good ecological status by connecting hydrological integrity with socio-economic valuation and stakeholder participation (Grizzetti *et al*., 2019).

Beyond policy, research increasingly highlights that valuing water-related ecosystem services is not only about assigning monetary worth but also about

recognising the ecological interdependencies and cultural meanings embedded in water systems (Pratson *et al*., 2023). For example, Balvanera *et al*. (2022) demonstrated that improved watershed management enhances both water security and biodiversity, while (Ruberto *et al*., 2022) found that valuing irrigation canals and spring systems in Lombardy fosters greater appreciation of multifunctional landscapes. These studies reinforce that economic valuation tools such as DCEs, when combined with ecological data and participatory insight, can bridge science and governance by translating ecosystem processes into policy-relevant evidence.

In nearly all classification systems currently in use, water is typically categorized as a provisioning service (Potschin & Haines-Young, 2011). Although biological processes influence its quantity and quality, the MA and TEEB consider water a provisioning service, even though it originates from abiotic hydrophysical processes.

According to Tognetti *et al*. (2006), watershed services result from ecosystem functions or processes that provide various direct and indirect benefits for humans, which can be grouped into the following categories (Table 2):

Table 2. Watershed services categories

Provision of Freshwater	○ Consumptive uses: drinking, domestic, agricultural and industrial ○ Non-consumptive uses: hydropower generation, cooling water and navigation
Flow Regulation and Filtration – ensuring the control and stability of water systems, these processes involve the control of mean surface runoff, peak or flood flows, base or dry season flow, and erosion and sediment load, as well as recharge of groundwater and soil moisture	○ Water storage in soils, wetlands and floodplains, supporting water availability ○ Erosion and sediment control, reducing soil loss and maintaining land productivity ○ Maintenance of river channels, wetlands, riparian habitats, fisheries, and other wildlife habitat ○ Preservation of coastal ecosystems, including mangroves, estuaries, and associated ○ Control of the level of groundwater tables ○ Water quality protection, minimizing impacts from pollutants like nutrients and organic matter, pathogens, pesticides and heavy metals, while managing changes in salinity and thermal regimes
Supporting Services	○ Maintenance of natural flow and disturbance regimes, that supports ecosystem resilience ○ Support for cultural values, including aesthetic qualities that enhance

	tourism and recreational uses, and support for ways of life

Source: Tognetti *et al*. (2006)

In the context of water systems, these services are deeply interconnected. These different types of services are interdependent, as there is a trade-off between the provision of freshwater for direct uses and the regulatory and supporting services that ensure its long-term availability (Tognetti *et al*., 2006). Therefore, it is essential to seek an appropriate balance between these uses.

The three categories mentioned represent only the potential benefits that a watershed can offer. However, these benefits are only considered ecosystem services when they have economic value (Tognetti *et al*., 2006). This highlights the importance of conducting site-specific assessments to identify which benefits are present in each context and at what scale they can be observed.

Without clear, local, and transparent assessments, payment schemes are often based on myths or assumptions about watershed services and their value (Tognetti *et al*., 2006). This can lead to ineffective actions, misallocation of resources, and even injustices, such as blaming remote rural communities for issues of scarcity or degradation, which can further worsen existing problems (Tognetti *et al*., 2006).

A central idea of the cascade model is that ecosystem services are always connected to human needs. To identify something as a service, it must be linked to a specific benefit or to the beneficiary. According to Banzhaf and Boyd (2005), this identification depends on context, since services are “contingent” on human activities or desires. This complexity, illustrated in Figure 3, shows how water quality can play different roles depending on the analysis of ecosystem services and social benefits.

In the model proposed by Banzhaf and Boyd (2005), there is a clear distinction between ecosystem services and benefits. Benefits are defined as the direct impacts on human well-being, such as increased or improved access to drinking water, or a more enjoyable recreational fishing experience. Unlike the definition presented by the MA, services are not the benefits themselves, but rather the factors that contribute to generating those benefits and enhancing human well-being.

In Figure 3, water quality plays a key role in the service supply chain that leads to benefits such as recreational angling and drinking water provision. However, only in the case of direct water consumption, such as for household supply, is water quality considered a final ecosystem service. Elements like wetlands and natural riparian vegetation are important in delivering that service, but according to Banzhaf and Boyd (2007), they are not services themselves, but intermediate components that enable it. In contrast, in the context of recreational fishing, water quality is no longer the final service. Here, the elements directly used are the fish population (e.g., bass) and environmental features such as surrounding vegetation, which influence the angling experience. In this case, water quality functions as an ecosystem capacity or function, necessary to produce the service (the presence of fish), but it is not the service itself.

Figure 3. Ecosystem services for recreational angling vs. drinking water

Benefit	Final Services	Intermediate Components
Recreational Angling	The water body	**The water body's quality**
	The bass population	
	The riparian forest	
Drinking Water	**The water body's quality**	Wetlands, natural riparian land cover

Source: Banzhaf and Boyd (2007).

Moreover, according to Banzhaf and Boyd (2007), wetlands provide another example of how ecosystem services are defined based on the specific benefit in question, as illustrated in Figure 4. For instance, wetlands have the capacity to absorb and slow down flood pulses, functioning as a form of natural capital that can substitute traditional infrastructure investments for flood prevention, such as dikes, dams, and levees. In this context, wetlands are considered ecosystem services associated with flood damage mitigation. However, they are not classified as an ecosystem service related to drinking water provision, not because they are unimportant to water quality, but because, in this case, the water quality itself embodies the value of the wetland and is recognized as the final service.

Figure 4. Ecosystem services for wetlands

Benefit	Final Services	Intermediate Components
Flood damage mitigation	**Wetlands**	
	Forest	
Drinking water availability		**Wetlands**
		Forest

Source: Banzhaf and Boyd (2007).

2.2 Water Ecosystems

2.2.1 Definition and Types of Natural Springs, Wetlands and Spring-Fed Wetlands

Wetlands are complex ecosystems that perform important ecological, biological, and hydrological functions valued by society. As scientific understanding of their roles in maintaining environmental balance has advanced, public recognition of their importance has increased. As a result, society increasingly prioritizes the conservation of wetlands over their conversion to private economic uses (Heimlich *et al*., 1998).

Wetlands provide goods or products through the direct use of one or more of their characteristics, while their ecosystem processes offer hydrological and ecological services that support or protect human activities and properties without being directly used (Brouwer *et al*., 2003). The stock of wetlands represents a multifunctional resource with significant economic value. It is estimated that

approximately 15% of the global value of ecosystem services and natural capital is generated by wetlands (Costanza *et al*., 1997).

There is still no universally accepted and precise definition of what constitutes a wetland (Barbier, Acreman, & Knowler, 1997; Heimlich, Wiebe, Claassen, Gadsby, & House, 1998; Brander, Florax, & Vermaat, 2006). However, the Convention on Wetlands, an intergovernmental treaty under the auspices of UNESCO, adopted in 1971 in the Iranian city of Ramsar, commonly known as the Ramsar Convention, provides a widely recognized and comprehensive definition, adopted by around 100 countries and applicable to a broad range of ecosystem types.

According to Article 1.1 of the Convention, wetlands are defined as: “areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres.”

Article 2.1 further clarifies that wetlands: “may incorporate riparian and coastal zones adjacent to the wetlands, and islands or bodies of marine water deeper than six metres at low tide lying within the wetlands.”

Based on these provisions, the scope of the Convention extends to a wide variety of habitats, including rivers, shallow coastal waters, and even coral reefs, although it does not cover deep-sea areas.

Based on the broad definition established by the Ramsar Convention, Scott (1989) proposed a classification comprising 30 types of natural wetlands and nine artificial ones. For illustrative purposes, however, wetlands can be grouped into five main categories (Barbier *et al*., 1997):

- Estuarine – areas where rivers flow into the sea and salinity is intermediate between freshwater and saltwater (e.g., deltas, mudflats, salt marshes);
- Marine – coastal areas not influenced by river flows (e.g., coral reefs, shorelines);
- Riverine – land areas that are periodically flooded due to river overflow (e.g., water meadows, flooded forests, oxbow lakes);
- Palustrine – wetlands with more or less permanent water and characteristic wetland vegetation (e.g., papyrus swamps, marshes, fens);
- Lacustrine – permanent bodies of water with little or no flow, usually of natural origin (e.g., ponds, kettle lakes, volcanic crater lakes).

Within this broad category, spring-fed wetlands represent a particularly distinctive form of groundwater-dependent ecosystem. They originate from natural springs, where groundwater discharges at the surface, often forming stable aquatic habitats characterised by constant temperature and mineral composition (Cantonati *et al*., 2012). When this groundwater interacts with surface geomorphology and vegetation, it creates wetland environments nourished by persistent subsurface inflow rather than seasonal rainfall (Rossetti *et al*., 2020). This steady water input gives rise to ecotonal habitats that connect terrestrial and aquatic systems, providing both hydrological stability and biological diversity.

Hydrogeologically, natural springs and the wetlands they feed are vital in maintaining connectivity between aquifers and surface waters, ensuring the continuity of streamflow and sustaining riparian vegetation (Land & Peters, 2023; Mustaq Shaikh & Farjana Birajdar, 2024). Their predictable discharge patterns act as a buffer against drought and temperature extremes, supporting cold-water biota and mitigating the effects of hydrological variability. Because of these characteristics, they are often termed “sentinel ecosystems,” serving as sensitive indicators of groundwater quality and catchment health (Cantonati *et al*., 2012).

Ecologically, spring-fed wetlands are hotspots of biodiversity. Their constant water supply supports unique assemblages of invertebrates, bryophytes, amphibians, and macrophytes, many of which are rare or endemic to groundwater-fed habitats (Rossetti *et al*., 2020). They also provide ecosystem functions such as nutrient retention, sediment trapping, and carbon storage. Through these processes, spring-fed wetlands maintain water quality and contribute to climate regulation, forming integral components of regional hydrological systems.

From an ecosystem service perspective, both general wetlands and their spring-fed counterparts deliver multiple benefits that span ecological, economic, and cultural dimensions. Their regulating services include water purification, groundwater recharge, and flood moderation; supporting services encompass habitat provision and nutrient cycling; and cultural services provide recreation, education, and spiritual enrichment. In many landscapes, springs and wetlands are integral to local heritage, representing long-standing relationships between people, land, and water.

2.2.2 Importance for the Ecosystem

Over the past 25 years, public recognition of the value of wetlands has grown significantly. Today, scientists and environmental organizations acknowledge the wide range of species and essential functions that depend on these ecosystems and actively work to raise public awareness of their importance for both the natural order and society (Heimlich *et al*., 1998).

The growing number of valuation studies on this environmental resource reflects the increasing recognition of its importance. Studies by Heimlich *et al*. (1998), Kazmierczak (2001), and Boyer & Polasky (2004) provide comprehensive reviews of wetland valuation literature, employing a range of methods such as contingent valuation (CV), hedonic pricing, replacement cost, avoided damage, and production value.

Wetlands are among the most productive ecosystems on the planet, providing a wide range of essential ecological functions and services, including flood and flow regulation, groundwater recharge and discharge, water quality maintenance, biodiversity conservation, carbon sequestration, and other life-supporting functions (Birol, Karousakis, & Koundouri, 2006).

Regarding their ecological functions, wetlands play a crucial role in maintaining water quality by acting as natural filters that remove nutrients and sediments from both surface and groundwater. This purification occurs through nutrient uptake by plants, adsorption into sediments, deposition of organic matter, and chemical precipitation. Moreover, the vegetation and flat topography of wetlands slow down water flow, promoting sediment deposition within the wetland itself and thereby reducing sedimentation in rivers, lakes, and streams (Heimlich *et al*., 1998).

Regarding their biological functions, wetlands are among the most biologically productive ecosystems in temperate regions, rivaling tropical rainforests in species richness and biological activity. Their high productivity stems from their ability to efficiently recycle nutrients and energy. These ecosystems provide essential habitat for a wide range of species. Some species spend their entire lives in wetlands, while others use them temporarily for feeding or breeding. Most freshwater fish depend on wetlands, as do many marine fish, which feed in wetlands or on organisms that originate there. Wetlands also serve as nurseries for various species, such as *alewife* and *blueback herring*, whose young seek shelter in these habitats. Many important sport fish, including pike, pickerel, muskellunge,

largemouth bass, and striped bass, spawn in or near wetlands. Amphibians and reptiles also rely on wetlands and are especially sensitive to habitat quality. About one-third of all bird species in North America use wetlands as stopover sites during migration, as well as for feeding, breeding, or protection from predators. In addition, many fur-bearing mammals, such as muskrat, beaver, otter, mink, and raccoon, prefer wetlands as their primary habitat. Unsurprisingly, wetlands are critical to the survival of numerous threatened and endangered species, largely due to habitat loss (Heimlich *et al*., 1998).

Regarding their hydrological functions, wetlands often form in areas where the water table is close to the surface, resulting in variable groundwater recharge and discharge. They play a crucial role in protecting shorelines by reducing wave impact and coastal erosion. Wetlands also store and convey floodwaters, as their interconnected root systems stabilize the soil along water margins, promote sediment accumulation, and reduce erosion by damping wave energy and slowing water flow. These ecosystems function like natural sponges, temporarily retaining floodwaters and releasing them gradually, which helps to lower flood peaks and protect downstream properties from damage. They also act as natural drainage channels, directing water from higher elevations to receiving water bodies and helping to mitigate extreme flood events (Heimlich *et al*., 1998).

Spring-fed wetlands contribute distinct ecosystem functions arising from their steady groundwater inputs. They provide thermal and hydrological refugia during droughts, stabilise baseflows to rivers, and enhance nutrient retention and water purification due to consistent redox conditions and long residence times (Cantonati *et al*., 2012). Their hydrochemical stability supports specialised assemblages, cold-water macroinvertebrates, bryophytes, and amphibians, making them biodiversity hotspots and sentinel ecosystems of groundwater quality (Mustaq Shaikh & Farjana Birajdar, 2024).

2.2.3 Importance for Human Well-Being

Wetlands perform essential functions such as preserving water quality, protecting against erosion and floods, providing habitat for aquatic and wildlife species, and offering recreational and scenic spaces. Although highly valued by society, private landowners rarely obtain direct economic benefits, except when these areas are converted to other uses, such as agriculture or urban development (Heimlich *et al*., 1998).

These ecological functions also translate into economic benefits, including flood protection, water supply and purification, recreational and commercial fishing and hunting, and mitigation of global climate change (Birol, Karousakis & Koundouri, 2006; Barbier *et al*., 1997; Woodward & Wui, 2001; Brouwer *et al*., 2003; Brander *et al*., 2004).

Depending in part on the type of wetland, these ecosystems can provide a wide range of goods and services that are highly valuable to humans (Barbier, 1991). While these services are derived from the ecological and physical functions of wetlands, they should not be confused with the functions themselves (Brander *et al*., 2006).

Although society does not always recognize the value of all the functions performed by wetlands, growing scientific understanding of their ecological roles has led to increased appreciation. The valuable functions wetlands provide, along with the open and pleasant spaces they create, contribute to enhanced quality of life. Various groups benefit from these services, including fishers, hunters, boaters,

downstream landowners, public water supply and flood control authorities, and recreational users, among others (Heimlich *et al*., 1998).

Wetlands directly support a large number of people and provide ecological processes that enable the extraction of goods and services in the form of natural resources such as water, fish and other edible animals, timber, and energy. Their soils are used for agriculture, their waters supply fish for consumption, their trees provide wood and fuel, and their reeds are harvested for making mats and roofing. Direct use also includes recreational activities such as birdwatching and boating, as well as scientific research. The wide range of services provided by wetlands is partly linked to geophysical processes, such as sediment retention and the mitigation of floods and storms. However, their benefits extend further, encompassing climatological, biological, and socio-cultural functions, including local and global climate regulation, biodiversity conservation, and the provision of environmental amenities that contribute to human well-being (Barbier *et al*., 1997; Brander *et al*., 2006).

In addition to direct uses, people also benefit from the ecological functions provided by wetlands. For instance, during floods, water that spreads across floodplains is temporarily stored, which helps reduce peak river levels and delay their crest, offering protection to downstream communities. Mangroves help dissipate wave energy, safeguarding coastal areas, while wetlands that recycle nitrogen contribute to improved water quality. These are examples of indirect uses, where people benefit from natural processes without directly utilizing them. While such functions can be replaced by engineered solutions, such as dams, levees, or water treatment plants, these alternatives are typically more costly than the services naturally provided by wetlands (Barbier *et al*., 1997).

However, not all wetlands perform these hydrological functions equally, or even at all. In some cases, they may have adverse effects, such as riparian wetlands that act as runoff-generating areas, increasing flood risks downstream. Therefore, it is crucial to assess and quantify the specific functions of a wetland before assigning it economic value (Barbier *et al*., 1997).

Moreover, the mere existence of wetlands can hold deep cultural and emotional significance for many individuals. For example, people who grew up in wetland areas but later moved to urban settings may still attribute high value to these ecosystems as part of their cultural heritage, even if they never visit them again (Barbier *et al*., 1997).

In addition to providing a range of valuable social benefits, such as biodiversity conservation, recreational opportunities, and landscape enhancement, studies by Gren (1993) and Byström (1999) demonstrate that wetlands represent a cost-effective alternative for nitrogen reduction, due to their efficient and low-cost nutrient absorption capacity.

Spring-fed wetlands also enhance human well-being by providing reliable baseflows, cleaner water, and aesthetic and cultural values associated with clarity and constancy of flow. They often serve as symbols of purity and local identity, offering educational and recreational opportunities while sustaining downstream water supplies. Because these benefits are non-market and widely shared, they are easily undervalued in conventional economic frameworks (Costanza *et al*., 2017; Wood *et al*., 2024; Vollmer *et al*., 2022) (Costanza *et al*., 2017; Vollmer *et al*., 2022; Wood *et al*., 2024).

2.2.4 Wetlands Loss

Protecting and enhancing the quality of remaining wetlands is essential, as fully functioning wetlands provide services that are highly valued by society. However, changes related to soil erosion, irrigation, deforestation, and urbanization in watersheds with significant wetland presence indicate a concerning trend: 75% of these watersheds show degradation in at least three out of the four quality indicators. While more than 60% report improvements in water-induced soil erosion and 22% show reductions in irrigation, 87% have experienced a decline in forest cover, and 96% have undergone increased urbanization (Heimlich *et al*., 1998).

Wetlands are constantly changing environments, shaped by natural processes such as droughts, land subsidence, sea-level rise, and sediment accumulation. As a result, many wetlands are temporary features of the landscape and are expected to eventually disappear, while new ones may form elsewhere. Human activity, both direct and indirect, has significantly accelerated these changes. Although some artificial wetlands have been created, such as reservoirs and canals, the overall loss of wetlands has far exceeded the gains (Barbier *et al*., 1997).

Historically, many wetlands have been regarded as unproductive areas, largely due to a lack of understanding of the value of the goods and ecosystem services they provide (Barbier *et al*., 1997). This perception has led to their conversion for intensive agricultural, industrial, and residential uses. In many cases, wetland drainage and water diversion for irrigation, particularly for upstream rice cultivation, have been justified by the belief that water in wetlands is wasted or less valuable (Barbier *et al*., 1997).

Moreover, wetlands have been severely affected by pollution from agricultural and industrial runoff, improper waste disposal, mining, and excessive groundwater extraction (Barbier *et al*., 1997). Other factors that degrade wetlands include poverty, economic inequality, population growth, immigration, mass tourism, and sociocultural conflicts (Skourtos *et al*., 2003). Often, communities that lose the ecosystem services provided by wetlands due to its conversion face consequences such as increased flooding, water scarcity, and declining water quality. Hydrological and landscape differences make certain areas more critical for maintaining ecosystem services, and their conversion can lead to abrupt and significant losses in ecosystem functions and services, the so-called 'threshold effects'. This can happen, for example, with flood storage or nutrient filtering, which rely on minimum discharge levels or specific habitat size, shape, and connectivity to support certain species (Heimlich *et al*., 1998).

The extent of wetland loss is difficult to measure, as the total global area of wetlands remains uncertain. However, data from specific countries provides an indication of the scale of the problem. The United States has lost approximately 54% of its original wetlands, primarily due to agricultural expansion (Barbier *et al*., 1997). Data on wetland loss in six European countries is presented in Table 3 (CEC, 1995). In Europe, between 50% and 60% of wetlands have been lost over the past century.

Table 3. Wetland loss in Europe

Country	Period	% Loss of Wetlands
Netherlands	1950-1985	55
France	1900-1993	67
Germany	1950-1985	57
Spain	1948-1990	60
Italy	1938-1984	66

Greece	1920-1991	63

Source: CEC, 1995

In response to growing concerns in the 1960s about the sharp decline in waterfowl populations, especially ducks, the Convention on Wetlands of International Importance, better known as the Ramsar Convention, was established in 1971 (Barbier *et al*., 1997). It was the first modern global intergovernmental treaty focused on the conservation and sustainable use of natural resources. The Convention came into force in 1975 and currently has 100 Contracting Parties (Ramsar, 1996). By 2004, the Ramsar Convention had designated 1,369 wetland sites across 139 countries worldwide, with a notable concentration in Western Europe (http://ramsar.org/sitelist.pdf). Collectively, these sites encompassed more than 120.5 million hectares of wetlands.

Its mission, as stated in Ramsar (1996), is “the conservation and wise use of wetlands by national action and international cooperation as a means to achieving sustainable development throughout the world.” The Convention provides a framework for implementing policies and actions aimed at the conservation and sustainable use of wetlands and their resources at both national and international levels.

In addition to international efforts, the European Union (EU) has also adopted policies aimed at preventing further loss or degradation of wetlands. Article 1(a) of the EU Water Framework Directive (2000/60/EC) explicitly identifies the protection, restoration, and enhancement of the water needs of wetlands as one of its core objectives, reinforcing the EU’s active role in safeguarding these ecosystems and its commitment to strategic policy development for their conservation. This commitment is further supported by other EU directives, such as the Birds Directive (79/409/EEC) and the Habitats Directive (92/43/EEC), which aim to preserve the key ecological functions provided by wetlands.

For instance, the conversion of wetlands to other uses in the United States has shown a gradual decline over time. From early settlement through 1954, more than 800,000 acres were converted annually. However, between 1982 and 1992, this figure dropped to less than 80,000 acres per year. During the same period, the share of wetlands converted for agricultural purposes also decreased significantly, from over 80% between 1954 and 1974 to just 20% between 1982 and 1992 (Heimlich *et al*., 1998).

Groundwater-dependent systems such as spring-fed wetlands face particularly acute pressures. Over-abstraction of aquifers, drainage and channelisation, and climate-induced changes in recharge reduce discharge volumes and alter water chemistry, often without immediate visible signs (Cantonati *et al*., 2012; Land & Peters, 2023). These processes lead to gradual ecosystem degradation and loss of regulating and cultural services long before surface indicators reveal damage. The Ramsar Global Wetland Outlook (2021) and the EU Nature Restoration Law (2024) highlight the urgency of restoring such hydrologically dependent wetlands.

2.2.5 Wetlands Challenges

The protection of wetlands is recognized as a public interest, but private landowners rarely obtain financial returns from the ecosystem functions these areas provide, since the benefits generated are diffuse and cannot be restricted to specific

beneficiaries. This can lead to divergences between public and private incentives regarding wetland protection (Heimlich *et al*., 1998).

The undervaluation of wetland resources and functions is one of the main factors leading to their misallocation, often resulting in their conversion to uses that prioritize immediate gains and short-term commercial returns (Barbier, Acreman, & Knowler, 1997).

Wetlands are highly multifunctional ecosystems. In addition to providing a wide range of essential products, they perform numerous ecological functions that support various economic activities. However, many of these services are not traded in markets, that is, they are neither bought nor sold, because their contribution to the economy is indirect and, as a result, often goes unrecognized (Barbier, Acreman, & Knowler, 1997).

The challenge in managing many ecosystem services lies in their nature as common-pool resources, resources from which it is difficult to exclude potential users. While users may or may not interfere with one another, these services are generally hard to value accurately and are often overlooked in decision-making processes (Haines-Young & Potschin, 2010).

Many uses associated with wetlands exhibit characteristics of public goods, making their commercialization highly limited or even unfeasible. Public goods are defined as those whose consumption by one individual does not diminish their availability to others and from which no one can be excluded, even if they have not contributed financially to their provision (Haines-Young & Potschin, 2010). This contrasts with private goods, which are both rivalrous, meaning one person's use reduces availability for others, and excludable, as access can typically be restricted through pricing or ownership (Haines-Young & Potschin, 2010).

These concepts can be better understood by considering two key criteria (Haines-Young & Potschin, 2010).:

- Excludability: whether access to the resource can be restricted;
- Rivalry: whether one person's use diminishes availability for others.

Furthermore, Table 4 presents ecosystem services classified according to their degree of excludability and rivalry, as proposed by Costanza (2008):

Table 4. Ecosystem services classified according to their degree of excludability and rivalry

	Excludable	**Non-excludable**
Rival	Rival market goods and services (most provisioning services)	Open access resources (some provisioning services)
Non-rival	Non-rival club goods (some recreation services)	Public goods and services (most regulatory and cultural services)

Source: Costanza, 2008

Market goods, such as food or timber, are typically both excludable and rivalrous, since producers control access and consumption reduces the supply (Barbier, Acreman, & Knowler, 1997). In contrast, many wetland uses are non-excludable but rivalrous, open to everyone, yet vulnerable to depletion as usage increases. A classic example is fisheries, which are accessible to all but prone to overexploitation (Barbier, Acreman, & Knowler, 1997).

On the other hand, some wetland services are non-rivalrous and non-excludable, the defining traits of pure public goods, such as biodiversity and non-use values. For instance, the conservation of biodiversity in a wetland benefits everyone without preventing others from enjoying the same benefit, making it

virtually impossible to charge for access. This characteristic contributes significantly to the undervaluation of such services (Barbier, Acreman, & Knowler, 1997).

This undervaluation would be less concerning if all wetland benefits could be enjoyed simultaneously, without conflict between uses. In an ideal scenario of harmonious multiple use, the combined value of different uses could easily justify conserving wetlands in their natural or semi-natural state (Barbier, Acreman, & Knowler, 1997).

However, in practice, there are inevitable conflicts and trade-offs between wetland uses, even when the ecosystem remains largely intact. For example, it may be unfeasible to manage a wetland simultaneously for recreation, commercial fishing, and wastewater treatment. Although wastewater treatment may offer greater economic value, its non-market and public good nature makes it less likely to be prioritized in market-driven decisions (Barbier, Acreman, & Knowler, 1997).

As Hardin (1968) warned, common-pool resources often face progressive degradation or depletion, as seen in marine fisheries, which are non-excludable and rivalrous. In contrast, services like flood protection are generally non-rivalrous and open-access, highlighting the difficulty of incorporating them into traditional market mechanisms, and thus into economic and policy decisions.

If decisions regarding wetland use are guided solely by market forces, it is unlikely that essential ecological functions, such as wastewater treatment, will be prioritized (Barbier *et al*., 1997). Since these services do not generate direct revenue or marketable outputs, they tend to be undervalued, often leading to the conversion of wetlands into uses that offer immediate economic returns but are ecologically inappropriate.

According to Barbier *et al*. (1997), another factor contributing to the misallocation of wetlands is the property rights regime that governs access and use of these resources. In some cases, wetlands are under open access, with no formal rules in place, leading to overuse and uncontrolled exploitation. In other situations, their use is regulated by traditional or informal systems, managed as common property resources, or owned publicly or privately (Bromley, 1989). Each regime results in different patterns of use and control. Open-access resources tend to be overexploited and undervalued. When economic valuation does not take the institutional context into account, the resulting estimates may be misleading, especially in times of transition, such as sudden privatization, nationalization, or the revival of traditional collective management systems.

Undervaluation becomes even more pronounced when it involves the permanent conversion of wetlands (Barbier *et al*., 1997). While activities such as agriculture, aquaculture, or real estate development generate marketable products and tax revenues, maintaining wetlands in their natural state preserves non-marketed goods and services, which are rarely accounted for in economic decision-making. As a result, conversion is often perceived as the more advantageous option, especially in regions under pressure for economic growth and revenue generation.

Even when direct revenue is not the primary goal, activities like farming and construction are commonly associated with regional development, as they generate employment and stimulate other productive sectors (Barbier *et al*., 1997). This gives strong political and economic appeal to wetland transformation, at the expense of their ecological functions, which often provide limited visible benefits and may even replace job-generating sectors, such as flood control or water purification. Moreover, these ecological services often require additional public investments, such as infrastructure for tourism or access roads.

In addition, some wetlands may be linked to more visible negative impacts, such as the proliferation of disease vectors, like malaria-carrying mosquitoes (Barbier *et al*., 1997). These effects tend to be more readily acknowledged, while the indirect and positive ecological functions of wetlands continue to be overlooked in both political and economic decisions.

The wide variation in estimated values reflects not only the intrinsic differences among wetlands but also the social and economic contexts in which they are situated. It is expected that coastal marshes, riparian wetlands, and prairie potholes receive different valuations based on the distinct functions and services they offer. However, less intuitively, socioeconomic factors, such as population density in surrounding areas, residents' usage patterns, education levels, age, income, and the availability of alternatives, can have an even greater impact on the estimated values than the wetlands' physical attributes themselves. Which means that identical ecosystems providing the same services may be less valued in remote, isolated areas surrounded by similar wetlands than in densely populated regions with few remaining sites (Heimlich *et al*., 1998).

A challenge in non-use valuation is that respondents may, either consciously or unconsciously, assign value to wetlands in general rather than to a specific site. While such estimates can serve as useful relative indicators of public support for environmental services, they are less reliable as absolute measures of benefit. This is especially true for generic or abundant wetlands, as opposed to specific sites facing development threats, where such values may have limited practical relevance. Moreover, comparisons between non-use and market-based values may be inappropriate, and the hypothetical nature of survey-based methods makes it difficult to distinguish between 'willingness to pay' and 'ability to pay' (Freeman, 1979; Anderson & Rockel, 1991; Barbier *et al*., 1997; Heimlich *et al*., 1998).

Economic valuation can provide critical information to decision-makers by enabling a comparative analysis of the costs and benefits of different wetland use options, information that would otherwise likely be overlooked in planning and policy-making processes.

For spring-fed wetlands, undervaluation is compounded by hydrological invisibility, their regulating functions occur below ground and benefits accrue diffusely downstream. Because these services are non-excludable and non-rival, they receive little policy or financial recognition despite their contribution to water security and biodiversity conservation. Integrating ecosystem-service valuation and motivational frameworks into groundwater and wetland management can help correct this imbalance (Raymond *et al*., 2023; Vollmer *et al*., 2022).

2.2.6 Gaps in Literature

The ecosystem service values of wetlands are often unknown and, consequently, tend to be excluded from decisions regarding their use and conservation (Brander, Florax, & Vermaat, 2006). To address this gap, an increasing number of studies have sought to estimate the economic value of various wetland areas. Alongside this growing body of research, there has been a notable diversification in both the valuation methodologies applied and the geographic locations assessed. This expansion is likely influenced, at least in part, by the enhanced accessibility of recent studies through electronic publications (Brander, Florax, & Vermaat, 2006).

2.2.7 Current Situation in North-Eastern Italy

The international literature on the economic valuation of wetland and groundwater preservation is extensive, employing different methodologies depending on the benefits assessed. The most used approaches are Contingent Valuation (CV) and averting expenditures, although Choice Experiments (CEs) have become increasingly prominent in recent years. In Italy, however, studies remain scarce and, with few exceptions, are mainly focused on coastal wetlands and the northern Adriatic lagoons (Tempesta & Vecchiato, 2013).

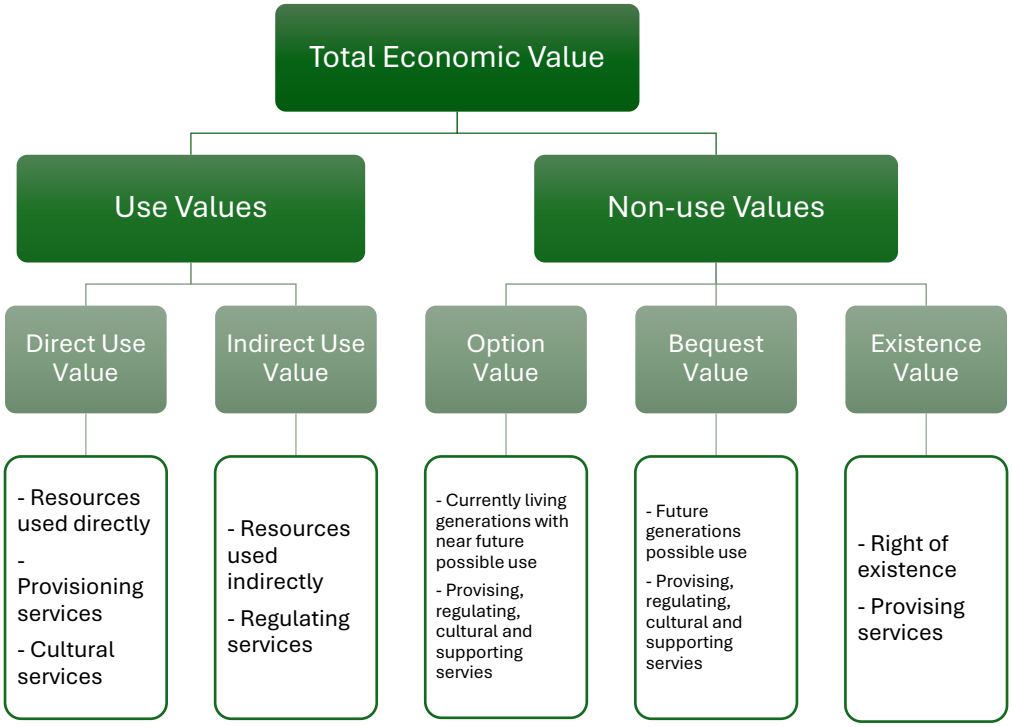
In Italy, information gaps are particularly significant given that the country is one of the most biodiverse in Europe. The network of Sites of Community Importance (SCI) and Special Protection Areas (SPA) covers about one-fifth of the national territory. Nevertheless, there is strong anthropogenic pressure on natural resources and local ecosystems, on which human well-being heavily depends (Masiero *et al*., 2019).

2.3 Economic Valuation of Water Ecosystems

2.3.1 Total Economic Value (TEV)

The economic value of environmental resources stems from all their attributes, which may or may not be associated with actual use. The Total Economic Value (TEV) of a resource represents a comprehensive measure, encompassing both use values (comprising direct and indirect use values) and non-use values (including option, bequest, and existence values), as shown in Figure 5.

Figure 5. Total Economic Value framework



Source: Own elaboration (2025)

Use values refer to the benefits derived from direct or indirect interaction with the resource. The Direct Use Value relates to the current utilization of a resource, such as extraction, recreation, tourism, or other productive and consumptive activities. These include both consumptive and commercial uses (e.g., timber harvesting), and non-consumptive uses (e.g., recreation), where the resource is used *in situ* without being depleted. The Indirect Use Value refers to the benefits people receive from the resources derived from ecosystem services, such as soil stabilization, water purification, and climate regulation, resulting from forest preservation.

Non-use values, on the other hand, represent the intrinsic worth attributed to environmental resources, regardless of any current use by humans or future use potential. The Existence Value is the satisfaction individuals derive from simply knowing that a particular species or ecosystem exists, often tied to ethical, moral, cultural, or altruistic motivations (Marre *et al*., 2015). The Option Value refers to the value placed on retaining the possibility of future use, either direct or indirect, particularly in contexts where the resource's availability is uncertain or threatened (Laird *et al*., 2009). Furthermore, the Option Value has been the subject of ongoing debate regarding its inclusion within the TEV framework, specifically, whether it should be considered part of TEV at all, and if so, whether it constitutes a use value or a non-use value. The Bequest Value reflects the benefit individuals perceive from preserving environmental assets for future generations, ensuring that they too can enjoy the direct and indirect benefits those resources provide.

2.3.2 Classification of TEV for Wetlands

To ensure that the values associated with wetland use are incorporated into policymaking, it is essential to have a conceptual framework that enables their identification and classification. The concept of Total Economic Value (TEV) provides such a framework and is widely recognized as the most appropriate approach (Barbier *et al*., 1997). Accordingly, the goods and services provided by wetlands can be categorized into different components of TEV based on their type of use (Barbier *et al*., 1997; Brander *et al*., 2006).

In simple terms, TEV distinguishes between use and non-use values (Barbier *et al*., 1997; Brander *et al*., 2006). Use values involve some form of human interaction with the environmental resource, whether direct, indirect, or related to future use options. Non-use values, on the other hand, refer to the value placed on the mere existence of the resource, either current or future, regardless of any direct utilization (Barbier *et al*., 1997). The TEV framework, as applied to wetlands, is illustrated in Table 5.

Table 5. Classification of TEV for wetlands

USE VALUES			**NON-USE VALUES**
Direct Use Value	*Indirect Use Value*	*Option and Quasi-Option Value*	*Existence Value*
○ Fish ○ Agriculture ○ Fuelwood ○ Recreation ○ Transport ○ Wildlife harvesting	○ Nutrient retention ○ Flood control ○ Storm protection	○ Potential future uses (as per direct and indirect uses)	○ Biodiversity ○ Culture, heritage ○ Bequest values

○ Peat/Energy	○ Groundwater recharge ○ External ecosystem support ○ Micro-climatic stabilisation ○ Shoreline stabilisation, etc	○ Future value of information	

Source: Barbier *et al*. (1997)

Use values can be categorized as direct or indirect. Direct use values arise from the immediate utilization of resources and services provided by wetlands, such as timber for energy or construction, water for irrigation, fishing, fuelwood collection, or recreational activities. These uses may occur in both commercial and non-commercial contexts (Barbier *et al*., 1997; Brander *et al*., 2006). According to Barbier *et al*. (1997), in developing countries, non-commercial uses of wetlands are often essential for the subsistence of local communities, whereas in developed countries they are more commonly associated with recreational activities. Commercial uses, in turn, can be significant for both domestic and international markets. In general, wetland goods and services that are traded in markets have more easily measurable values, while direct subsistence or informal uses are often overlooked in development decisions due to their lack of visibility in market transactions (Barbier *et al*., 1997).

In contrast, indirect use values are linked to ecosystem services provided by the natural functions of wetlands, particularly those that support or protect economic activities with measurable value (Barbier *et al*., 1997; Brander *et al*., 2006). These values arise from the positive influence of wetland functions on the production or consumption of goods and services. However, since such contributions are not traded in markets and do not generate direct financial returns, they are difficult to quantify and are often overlooked in decision-making processes.

According to Barbier *et al*. (1997), many regulatory ecological functions of wetlands generate indirect use values by reducing damage or supporting economic activities. Functions such as nutrient retention, storm protection, and shoreline stabilization help prevent material losses, yet coastal and riparian wetlands are still being drained for real estate development. Mangrove systems, crucial nursery habitats for fish and shrimp, essential to marine fisheries, are rapidly being converted into aquaculture ponds. Likewise, natural floodplains that recharge aquifers used in agriculture, livestock, and industry are increasingly threatened by dams and upstream water diversions for irrigation.

Another important type of value is the option value, which relates to uncertainty about future demand for a resource or its future availability. It can be estimated by comparing the value assigned before and after the resource is used (the difference between e *ex ante* and *ex post* valuation). When there is a belief that the resource may have high future value or that its conversion would be irreversible, a quasi-option value may arise, reflecting the benefit of delaying its exploitation. This is not considered a separate category of benefit but rather a way to account for the

value of information gained by postponing the conversion decision (Barbier *et al*., 1997).

In contrast, non-use values are not related to any direct, indirect, or future use, but instead reflect the economic value attributed to the mere existence of wetlands (Brander *et al*., 2006). Some individuals, even without utilizing these ecosystems, support their preservation due to their intrinsic worth, known as existence value. This type of value is particularly difficult to measure, as it involves subjective judgments that are unrelated to any actual or potential use, whether personal or by others (Barbier *et al*., 1997). An important subset of non-use values is bequest values, which reflect the importance placed on conserving tropical wetlands for future generations. These values tend to be especially high among local communities that rely on these ecosystems and wish to preserve their way of life for their descendants (Barbier *et al*., 1997).

Often, the various components of total economic value are not captured by the wetland owner, leading to the neglect of significant values in decisions regarding conversion to alternative uses. While some goods and services can be traded directly in well-established markets, with observable marginal values, many important services remain unmarketed, either directly or indirectly, due to market failures such as poorly defined property rights or the (quasi-)public nature of certain ecosystem services (Brander *et al*., 2006).

Some ecosystem services can be indirectly valued through markets, such as amenities that increase property values near wetlands or the provision of water for agriculture. Other services, like bequest and existence values, are typically not captured even indirectly (Brander *et al*., 2006).

According to Brander *et al*. (2006), when the economic values of wetland ecosystem services are not reflected in efficient markets, they can be estimated using a variety of non-market valuation techniques.

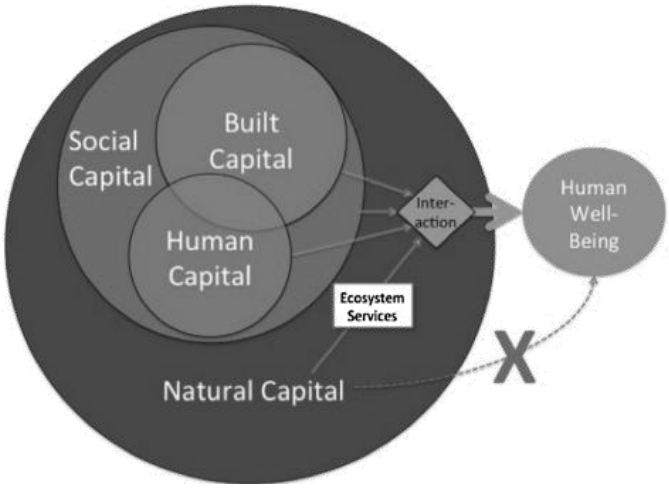
2.3.3 Economic Valuation and the Importance of Valuation of Wetlands

To understand the significance of economic valuation for wetland management and policy, it is first necessary to examine how valuation influences decisions regarding the use of environmental resources in general, and wetlands in particular (Barbier *et al*., 1997).

Valuation is the process of assessing choices that involve trade-offs to achieve a goal. Every decision that involves a trade-off includes some form of valuation, either directly or indirectly. To do this, it is important to clearly understand the goal. Ecosystem services are the benefits people receive from nature, and their value reflects the extent to which ecosystems contribute to achieving that goal (Costanza *et al*., 2014).

The benefits provided by ecosystems depend on the interaction between natural, human, social, and built capital. Human and built capital, which together form the economy, are embedded within society, which in turn is embedded within nature. Ecosystem services do not flow directly from natural capital to human well-being; it is only through interaction with the other three forms of capital that natural capital generates benefits. Thus, ecosystem services represent the relative contribution of natural capital to human well-being. This interaction is illustrated in Figure 6. The main challenge in valuing ecosystem services lies in assessing the relative contribution of natural capital within this interaction and balancing our assets to promote sustainable human well-being (Costanza *et al*., 2014).

Figure 6. Interaction between built, social, human, and natural capital required to produce human well-being



Source: Costanza *et al*. (2014)

The concepts of ecosystem service flows and natural capital stocks have become increasingly useful in highlighting, measuring, and valuing the interdependence between humans and nature. While complementary to other conservation strategies, this approach provides unique conceptual and empirical tools and enables engagement with diverse audiences for different purposes (Costanza *et al*., 2014).

The contribution of ecosystem services can be represented using different units of measurement. Since built capital in the economy is typically assessed in monetary terms, a widely understood language, it is often used as a reference to compare other forms of capital, including natural capital. However, other units such as land area, energy, or time can also be used, depending on which best convey the values to the target audience in a given decision-making context (Costanza *et al*., 2014).

Thus, economic valuation seeks to assign quantitative values to the goods and services provided by environmental resources, even in the absence of market prices to guide the process. However, this definition is somewhat limited. It is important to understand what economists mean by 'value.' In general, the economic value of a good or service is measured by people’s willingness to pay for it, minus the costs of providing it. When an environmental resource simply exists and provides goods and services at no cost, its value is determined solely by individuals’ willingness to pay for its benefits, even if no financial transaction takes place (Barbier *et al*., 1997).

It is incorrect to assume that assigning monetary values to ecosystem services necessarily implies privatizing them or turning them into tradable commodities in private markets. Most of these services are public goods, characterized by non-rivalry and non-excludability, or common-pool resources, which are rival but non-excludable. Under such conditions, privatization and conventional markets tend to be ineffective or unsuitable for their management. Moreover, the values estimated through non-market valuation methods often reflect use or non-use values rather than exchange values (Costanza *et al*., 2014).

Still, understanding the value of ecosystem services is essential for more effective management. It is also important to recognize that valuation is unavoidable: every decision involving trade-offs related to ecosystems already

implies, even implicitly, a form of value attribution. Making this process clearer and more transparent, even with its limitations, contributes to more informed decision-making (Costanza *et al*., 2014).

It is important to value environmental resources because, although their importance is often intuitively recognized, this is not always enough to ensure their proper use. Many of these resources are complex and multifunctional, making it difficult to understand how the variety of goods and services they provide contribute to human well-being. In some cases, exploitation may be justifiable; in others, conservation is essential. Economic valuation, therefore, emerges as a valuable tool to guide complex decisions, offering support to planners and policymakers in the growing use of this approach to inform decision-making processes (Barbier *et al*., 1997).

The concept of ecosystem services makes it clear that the idea of a trade-off between 'environment' and 'economy' is a false dichotomy. If nature contributes significantly to human well-being, it must be considered an integral part of the real economy. The real challenge, therefore, lies in managing all our assets, both natural and built, more efficiently and sustainably (Costanza *et al*., 1997; Costanza *et al*., 2000; Costanza *et al*., 2014).

The loss of environmental resources represents an economic problem, as it involves the disappearance of important (potentially irreversible in some cases) values. Every decision regarding the use of an environmental resource, whether to preserve it, allow its degradation, or convert it to another use, entails trade-offs between gains and losses in value. Assessing whether a current or proposed use is appropriate requires careful evaluation of these trade-offs. Therefore, it is essential to thoroughly consider all the values associated with each possible use alternative (Barbier *et al*., 1997).

Preserving an area in its natural state, for example, involves direct costs related to the establishment of conservation units, which in developing countries may include paying park rangers, maintaining the area, and implementing buffer zones for surrounding communities. Choosing preservation also means forgoing development opportunities, which translates into additional costs from unrealized benefits. These costs are generally easy to identify, as they are tied to marketable activities such as fishing or subsistence farming, especially in areas like wetlands (Barbier *et al*., 1997).

Governments and funding agencies often consider both the direct costs and the foregone development benefits when deciding whether to preserve an environmental resource or manage it sustainably. However, the same level of consideration should apply when evaluating its conversion to another use. In such cases, it is essential to assess not only the direct costs of conversion but also the lost values that the original resource would no longer provide. These may include the loss of key ecological functions and, in complex systems like wetlands, the loss of valuable biodiversity as well as recreational and cultural values (Barbier *et al*., 1997).

Many of the values associated with natural or managed resources are not traded in markets and are therefore often overlooked in both public and private development decisions. The market value generated by converting an environmental resource for commercial purposes often fails to reflect the environmental benefits that are lost. As a result, development decisions tend to favor uses with visible market returns. The failure to properly account for the economic costs of converting or degrading environmental resources is one of the main

reasons behind misguided development policies, leading to overconversion and overexploitation of natural resources. This issue is particularly common in both public and private decision-making, especially in ecosystems such as wetlands, highlighting the need for more rigorous evaluation of the net economic benefits associated with different resource uses (Barbier *et al*., 1997).

Economic valuation is just one of the tools available to improve the management of environmental resources such as wetlands. However, decision-makers must also balance multiple conflicting interests when determining the best use of these areas. Valuation can support this process, but only if its purpose and limitations are clearly understood by those managing the resources (Barbier *et al*., 1997).

The valuation of ecological services provided by wetlands is a relatively recent approach. In the past, these ecosystems were often regarded as unproductive areas, whose value depended on their drainage and conversion to other uses (Woodward & Wui, 2001). Today, although there is widespread recognition that wetlands offer important ecological services, significant debates remain regarding the economic efficiency of their current use and the appropriate level of public and private investment in their protection and restoration. In this context, the need to quantify the value of the services provided by these ecosystems is becoming increasingly important (Woodward & Wui, 2001).

It is also incorrect to claim that protecting ecosystem services underestimates human ingenuity. On the contrary, recognizing and measuring natural capital and ecosystem services demonstrates our capacity for innovation. Studies in this field have highlighted the limitations and high costs of relying solely on heavy engineering solutions, which can often be more effectively replaced by natural systems. For example, showing that coastal wetlands provide better and more cost-effective protection against hurricanes than built levees, and that they also store carbon, proves that restoring or recreating these areas is a smart use of our knowledge and capabilities (Costanza *et al*., 2014).

Although some wetland values can be determined through market transactions or from the income generated by production factors used to produce marketable goods, most of the economic benefits associated with wetlands must be estimated using non-market valuation methods (Heimlich *et al*., 1998). Wetlands provide a wide range of services, including habitat for species, flood protection, water purification, as well as amenities and recreational opportunities. Since these services typically lack market prices, their valuation relies on non-market methods.

Numerous studies have attempted to estimate these values, yielding highly varied results (Woodward & Wui, 2001). A review by Heimlich *et al*. (1998), for instance, identified 33 studies conducted over a span of 26 years, with per-acre value estimates ranging from US$0.06 to US$22,050. Understanding the value of wetland services is important for several reasons. The main one is that it improves benefit transfer efforts, which can help estimate approximate values of resources and support ongoing projects. Even if it’s not suitable for precise calculations, this approach is useful for guiding decisions. It can also be used to estimate the total value of similar ecosystems at national or global scales, helping set priorities and inform public policies (Woodward & Wui, 2001). An example is the study by Costanza *et al*. (1997), which estimated the value of the world’s ecosystems. Although ambitious, such data helps draw the attention of policymakers and the public.

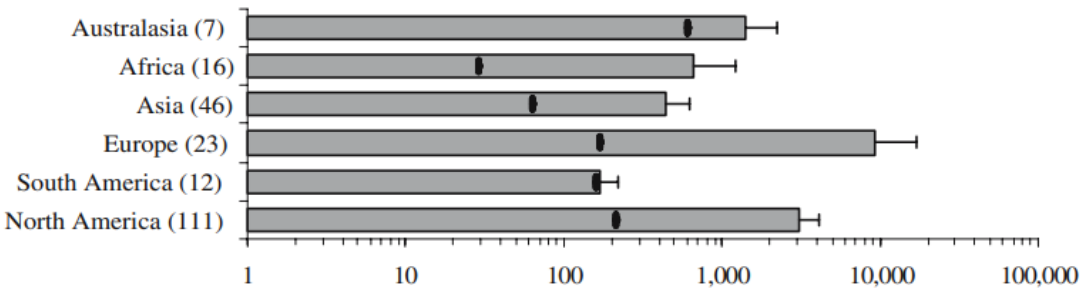
Public recognition of the importance of wetlands has increased significantly over the past two and a half decades. Today, scientists and environmental organizations acknowledge the strong dependence of many species and ecological functions on these ecosystems and have actively worked to raise public awareness about their relevance for both nature and human well-being. However, this growing recognition has not yet translated into economic values that can be directly captured by landowners through market mechanisms. Many of the benefits currently attributed to wetlands, such as water quality improvement and biodiversity conservation, are considered non-marketed goods, meaning they are not traded in formal markets (Heimlich *et al*., 1998).

Although these ecosystem services are vital to society, they are often undervalued when compared to the perceived economic benefits of converting wetlands to alternative land uses. To address this issue, economists have developed non-market valuation techniques aimed at estimating the monetary value of the services provided by wetlands. However, the resulting estimates can vary widely due to differences in the methods applied, the biophysical characteristics of the ecosystems, the position of wetlands in the landscape, and the surrounding socioeconomic context (Heimlich *et al*., 1998).

It is often argued that the excessive degradation and conversion of wetlands stem largely from the underestimation of their non-market environmental values in development planning (Barbier *et al*., 1997). In this context, economic valuation emerges as a strategic tool, capable of measuring and comparing the multiple benefits provided by wetlands, thus contributing significantly to their sustainable use and effective management at the global level. The economic value of resources such as wetlands is equal to the benefits (net of costs) that these systems provide to humans (Freeman, 1993).

Brander, Florax, and Vermaat (2006) conducted a comprehensive meta-analysis of the economic valuation of wetlands. Given that the studies reviewed reported values using different metrics, currencies, and reference years, the authors standardized all values to 1995 US dollars per hectare per year, as shown in Figure 7. In doing so, they faced the challenge of distinguishing between average and marginal values, both commonly expressed as monetary amounts per hectare. While most valuation studies provide estimates of total or average values, a significant number also report marginal values. The authors emphasize that small changes in wetland areas should be assessed using marginal values, whereas average values are more appropriate for comparing the overall value of wetlands relative to their total size.

Figure 7. Wetland value by continent



Source: Brander, Florax, and Vermaat (2006)

Figure 7 presents both mean and median wetland values, which, as expected, vary significantly across continents. While this graphical representation does not account for variation explained by other important factors, it provides an initial overview of the key drivers influencing the differences in wetland values reported in the literature. The figure shows that average wetland values are highest in Europe, followed by North America, Australasia, Africa, Asia, and lastly South America.

According to Barbier *et al*. (1997), the main goal of economic valuation in guiding wetland management is to identify which uses offer the greatest economic efficiency. This is based on the idea that wetland resources should be allocated to uses that generate the highest net benefit to society, that is, the difference between the economic benefits and costs of each alternative. However, valuation focuses on efficiency and does not consider who gains or loses from a given use. As a result, a wetland use that delivers a high net benefit may be considered efficient, even if the main beneficiaries are not those bearing the costs. Therefore, it is important that wetland policies and investments are assessed not only for their efficiency, but also for their impacts on the distribution of benefits and costs.

In many cases, the decision to preserve or convert a wetland has already been made, and valuation is used merely to justify that decision. In such situations, it adds little to the decision-making process. In other cases, more is expected from valuation than it can realistically deliver. One of the main challenges in valuing complex environmental systems, such as wetlands, is the lack of accurate information about the ecological and hydrological processes that underpin the various values these ecosystems provide. When data are lacking, which is common, especially for environmental values without market prices, analysts must be realistic about the limitations of valuation to estimate the benefits provided by these ecosystems. Likewise, decision-makers need to understand that without further scientific and economic research, valuation cannot provide reliable estimates (Barbier *et al*., 1997).

Economic valuation guides the use of wetland resources in ways that enhance human well-being. To achieve this, the various environmental benefits of these areas are measured based on their contribution to the production of goods and services valued by society. However, some argue that wetlands and the living beings within them have intrinsic value, beyond their usefulness to humans. From this perspective, conserving these ecosystems is a moral duty. Furthermore, decisions about wetland management can also be influenced by political factors. Therefore, economic values represent only one part of the decision-making process, which should also consider other important dimensions (Barbier *et al*., 1997).

According to Heimlich *et al*. (1998), in the United States, wetlands in typical agricultural areas are rarely evaluated economically, while coastal and urban wetlands have received more attention from economists. This is likely because the benefits and functions of coastal wetlands are more visible, as they are near large populations or perceived as more threatened, even though agricultural conversion rates may be higher.

Marketable goods from wetlands, such as fish, shellfish, and fur-bearing animals, generally have lower economic value than non-marketable goods. If the opposite were true, wetland owners could capture more of the value directly, for example, through fishing activities.

Heimlich *et al*. (1998) also point out that per-hectare values assigned by people who do not directly use wetlands are often higher than those given by direct

users. This apparent paradox exists because non-users are willing to pay less individually, but there are many more of them overall. Evidence shows that willingness to pay tends to decrease with distance from the wetland. However, this pattern is complicated by nationally or internationally known wetlands, which may have non-use values assigned by people thousands of kilometers away. Additionally, respondents may be assigning value to wetlands in general, rather than to a specific one.

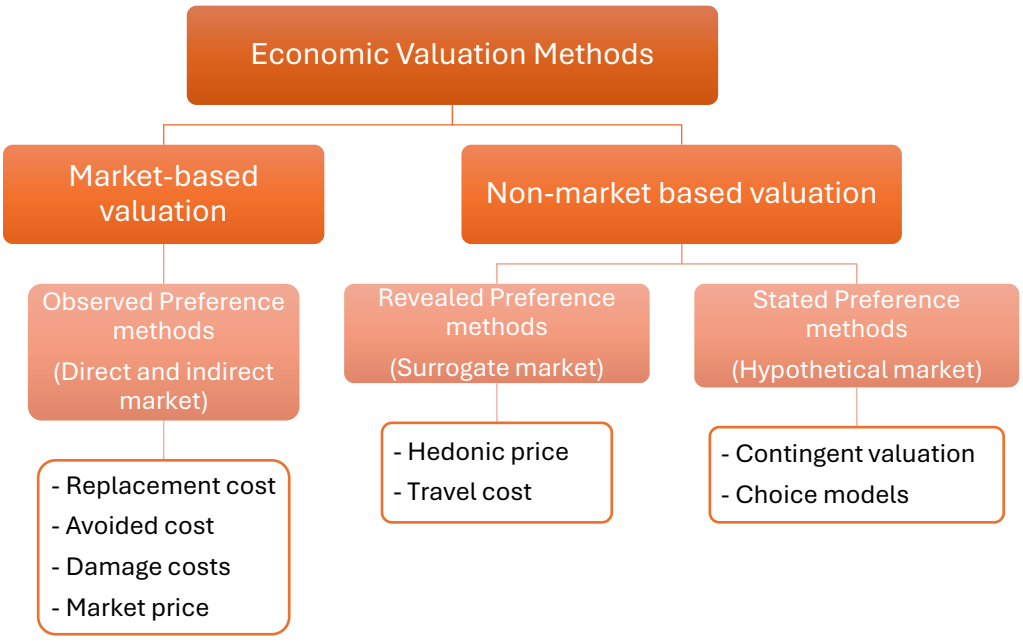
2.3.4 Environmental Economic Valuation Methods

Valuation seeks to assign economic value to goods and services that are not traded in conventional markets. To estimate the value of a non-marketable good or service, such as an environmental asset, it is essential to identify the various components that constitute its TEV.

Environmental economic valuation methods are designed to measure individuals' preferences for environmental resources or services. Accordingly, what is monetarily valued are not the resources themselves, but rather people's preferences concerning changes in the quality or quantity of the environmental good or service provided (May, Lustosa & Vinha, 2003).

Economic valuation methods are generally classified into two categories: Production function approaches (market-based valuation), and Demand function approaches (non-market-based valuation), as illustrated in Figure 8 (Motta, 1998).

Figure 8. Economic valuation methods



Source: Own elaboration (2025)

Production function methods (or market-based valuation), classified as Observed Preference (OP) approaches, infer the economic value of environmental resources from their role as inputs in the production of private goods or services. Although changes in resource availability do not directly affect market prices, it is still possible to estimate their environmental benefits or costs by examining their effects on production. Individuals' preferences for already marketed ecosystem

services are revealed through market transactions and are assessed using approaches such as market prices, marginal productivity, and substitute-goods methods (replacement cost, avoided costs, and damage costs).

Demand function methods (or non-market based valuation) are categorized into Revealed Preference (RP) methods (indirect approaches) and Stated Preference (SP) methods (direct approaches). These methods are based on the premise that changes in the availability or quality of an environmental good or service influence individuals' WTP for its preservation or willingness to accept (WTA) compensation for its loss, including impacts on complementary private goods.

Changes in individual well-being are estimated by analyzing variations in consumer surplus, the additional satisfaction or benefit a consumer receives when the price paid is lower than the maximum amount they are willing to pay. These variations in consumer surplus reflect the economic value of changes in environmental resource availability.

Estimate economic values from demand function either:

- indirectly, through demand curves derived from existing complementary markets (e.g., travel cost method, hedonic pricing) for private goods or services to the environmental resource, or
- directly, via hypothetical markets constructed through surveys (e.g., contingent valuation or choice models), specifically designed to elicit preferences for the environmental resource under evaluation.

Revealed Preference (RP) methods estimate environmental values based on consumers' actual behavior in related markets. They observe real transactions to infer individuals' willingness to pay for environmental quality through surrogate markets, where the environmental resource is indirectly reflected in the price of associated goods or services. The two main RP approaches are the Travel Cost (TC) method and Hedonic Pricing (HP).

Stated Preference (SP) methods, on the other hand, are based on individuals' or users' preferences for natural resources. They directly measure individual's values for environmental quality by asking respondents about their preferences and how much they are willing to pay (WTP) for certain environmental goods or services. These data are typically collected through questionnaires, surveys, and hypothetical markets, as in the case of the Contingent Valuation (CV) method and Choice Modeling (CM) approaches.

The main difference between the two methods is that RP methods are limited to measuring use values, as they rely on observable market transactions, whereas SP methods can capture both use and non-use values, expanding their applicability in contexts where no direct market exists for the environmental resource in question, as shown in Table 6. Thus, while RP methods use behavioral data to estimate *ex post* willingness to pay, SP methods rely on hypothetical data to estimate *ex ante* willingness to pay for goods and services meaning they can assess preferences for changes that have not yet occurred.

Benefit transfer is also a non-market based valuation, that is not a valuation method itself, but a technique that applies economic estimates from previous studies of similar environmental quality changes to value an environmental change at a different policy site, the context where information is needed, based on the outcomes of the original research context.

Table 6. Values captured by non-market based valuation methods

Non-market based valuation		
	Valuation methods	**Values captured**
Revealed Preference	Travel cost	Use values
	Hedonic price	Use values
Stated Preference	Contingent valuation	Use values; Non-use values
	Choice models	Use values; Non-use values
Benefit transfer		Use values; Non-use values

2.3.5 Justification for the Use of SP Methods for Water Services

Despite the clear benefits provided by ecosystem services, their economic valuation remains a challenge, especially because they are special types of goods that are not traded in conventional markets. This prevents the direct inference of marginal impacts from changes in their supply or relative prices on individual utility, as is typically done in economic analyses based on consumer behavior (Hernandez & da Costa, 2022).

Due to the difficulties in capturing the full range of values linked to water-related services, economic valuation has emerged as a key instrument for informing decision-making and policy development, particularly in situations where market prices do not account for environmental and social benefits.

Revealed preference methods are less advantageous than stated preference methods, whose main strength lies in their flexibility to be applied across a wide range of contexts (Bashir, Equbal & Raina, 2022).

The main advantage of revealed preference methods is that they rely on actual market transactions, where prices are effectively paid rather than hypothetical. However, these methods have important limitations, for example, they require the existence of private goods that are closely linked to and can reflect the value of the public good. Real estate is commonly used in this context, as property prices capture not only the characteristics of the home itself but also those of the surrounding environment, including environmental quality (Hasler, Lundhede, Martinsen, Neye, & Schou, 2005).

Nonetheless, there are no traded goods that adequately reflect the value of water quality. Moreover, non-use values, such as the value of a resource's mere existence or its preservation for future generations, cannot be captured by revealed preference methods. These approaches only reflect the preferences of individuals who directly use the associated good, such as homeowners. As a result, they fail to estimate the total economic value of a public good, which includes both use and non-use values (Hasler, Lundhede, Martinsen, Neye, & Schou, 2005).

Therefore, the valuation of water-related ecosystem services requires the application of stated preference methods, as these allow for the assessment of both use and non-use values, offering a more comprehensive and policy-relevant estimate. There is a relatively extensive body of stated preference studies addressing the non-market aspects of water quality and river ecosystems (Huenchuleo, Barkmann, & Marggraf, 2016).

One of the main advantages of stated preference methods is their flexibility, allowing for the construction of realistic scenarios even for policies that have not yet been implemented. Often, hypothetical choices are the only way to obtain relevant information for public policy design. Moreover, these methods, particularly contingent valuation and conjoint analysis, are capable of capturing non-use values, which cannot be measured using revealed preference approaches that rely solely on observed market behavior. For this reason, stated preference methods are

particularly well-suited for valuing ecosystem services, as they can account for both use and non-use values (Whitehead, Pattanayak, Van Houtven, & Gelso, 2005; Zaman & Ali, 2023; Udugama *et al*., 2024).

Although various non-market valuation techniques have been employed to estimate the monetary value of ecosystem services, stated preference approaches have gained increasing prominence, as they are based on individuals' self-reported preferences or values (Udugama *et al*., 2024). Moreover, these methods offer adaptability to different contexts and can be applied to estimate WTP for both current and future ecosystem services (Hernandez & da Costa, 2022). So, SP methods are widely applied to elicit public preferences and to estimate individuals' WTP for changes in environmental quality and ecosystem services (Hatton MacDonald, Rose, Johnston, Bark, & Pritchard, 2019).

Stated preference methods are widely used to estimate the welfare effects of non-market goods through hypothetical choice scenarios. The most prominent approaches are the contingent valuation method and choice experiments, both based on the construction of a hypothetical market presented to a representative sample through a carefully designed questionnaire (Haipeng & Xuxuan, 2012; Chaikaew *et al*., 2016; Hernandez & da Costa, 2022).

The first stated preference studies employed the CV method, in which respondents are simply asked whether they would be willing to support, financially and, in the case of public goods, also politically, a proposed change in the provision of certain goods or services (Hanley & Barbier, 2009).

An alternative and more structured approach emerged in the early 1980s with the development of DCEs, introduced through the works of Louviere and Hensher (1983) and Louviere and Woodworth (1983). In contrast to CV, DCE involve the selection of multiple attributes, each with varying levels of provision, related to the good or service to be improved, defined during the study's design phase. Respondents are then asked to choose their preferred scenario among a set of alternatives, compared to a baseline. These alternatives differ based on the levels of the specified attributes, and a hypothetical cost is included among them. This setup allows respondents to express their preferences through trade-offs between the attributes presented.

One of the key strengths of this method is its ability to significantly reduce the cognitive effort required from respondents, as directly expressing preferences for specific attributes can be challenging. Moreover, a well-designed DCE allows participants to complete multiple-choice tasks, effectively increasing the sample size and enhancing the robustness and reliability of the results (Hernandez & da Costa, 2022).

In the context of ecosystem services, valuations derived from stated preference methods can support broader public policy analyses and environmental damage assessments. These valuations contribute to informed decision-making by public and private institutions, judicial rulings in environmental damage cases, and advocacy efforts by NGOs (Hernandez & da Costa, 2022). Therefore, stated preference methods, particularly DCE, offer valuable and policy-relevant insights that can effectively inform sustainable landscape management strategies (Castillo-Eguskitza, Hoyos, Onaindia, & Czajkowski, 2019).

However, the validity of the results obtained through these methods remains a topic of debate, primarily due to the hypothetical nature of the scenarios presented, the main limitation of these methods. This characteristic may lead respondents to overstate their actual willingness to pay, resulting in what is known

as hypothetical bias (Hernandez & da Costa, 2022). Respondents are often placed in unfamiliar contexts with limited information. In such situations, even sincere answers may be imprecise or constrained. In more critical cases, the hypothetical setting can lead to trivial or inconsistent responses (Whitehead et al., 2005).

To mitigate this issue, it is essential that the baseline conditions, proposed changes, and their underlying mechanisms are presented clearly, realistically, and coherently. The information provided must be easily understood, accepted as plausible, and perceived as credible by respondents to enhance the reliability of their answers (Johnston *et al*., 2017; Hernandez & da Costa, 2022).

2.4 Choice Experiments (CE)

2.4.1 Definition and Theoretical Foundations

Among stated preference methods, Choice Experiments (CEs) stand out for their widespread application in valuing various environmental goods affected by market failures. This approach has proven effective in addressing some of the methodological limitations associated with the CV method. Although the hypothetical nature of stated preference surveys presents certain methodological challenges, CEs have been successfully applied to the valuation of non-marketed goods, enabling the reliable estimation of consumer preferences (Bashir *et al.,* 2022).

Several valuation methods have been developed to capture individuals' preferences, estimating their Willingness to Pay (WTP) for improvements in environmental services or their Willingness to Accept (WTA) compensation for losses in these services. CV, CEs, and Contingent Ranking are among the most widely used valuation methods. Within the Choice Modelling (CM) approach, four main methodologies are distinguished: Choice Experiments, Contingent Ranking, Paired Comparisons, and Contingent Classification. These techniques differ primarily in how they elicit consumer preferences regarding various changes, either by selecting the preferred environmental good or service, or by ranking or classifying the available alternatives. Over the past decade, CEs have become one of the most widely applied techniques in environmental valuation (Bashir *et al.,* 2022).

The Discrete Choice Experiment (DCE) technique is a stated preference method initially developed by Louviere and Hensher (1982) and Louviere and Woodworth (1983), based on advances in various disciplines such as psychology, marketing, and notably transport economics. In these fields, stated preference data proved especially useful in understanding individuals' behavior when faced with different alternatives. Although DCEs have been more recently adopted in environmental economics, their use was already well established in areas such as marketing and transportation (Bashir *et al*., 2022; Whitehead *et al.,* 2005).

Although the first application of a DCE in the context of environmental resources was carried out by Adamowicz *et al*. (1994), most studies focusing on water resource valuation using this method emerged after 2000. Notable contributions include EFTEC (2002) on bathing water quality, Willis *et al*. (2002) on water services, and Carlsson *et al*. (2003) on wetlands (Hoyos, 2010).

Over the past decade, applications of DCEs have grown significantly, establishing them as a widely used stated preference method for environmental

valuation, particularly in contexts involving non-use values (Hoyos, 2010; Bashir *et al*., 2022; Thiene & Scarpa, 2009).

Most microeconomic models of consumer behavior are based on utility maximization under a budget constraint. In the case of DCEs, their theoretical foundation lies in Lancaster's characteristics theory of demand (1966), consumer theory, and welfare economics. According to this approach, individuals derive utility not from the good itself, but from its attributes. Therefore, changes in prices can lead to discrete shifts between bundles of goods, depending on which combination of attributes offers the best cost-benefit efficiency. The main feature of DCEs is that they allow respondents to make hypothetical economic choices based on goods described in terms of their attributes and the importance assigned to each one (Hoyos, 2010; Bashir *et al*., 2022).

The approach of DCEs to preference elicitation is like the choice-based consumer theory, as it explicitly assumes that respondents' observed choices reveal their underlying preferences. Thus, DCEs combine Lancaster's attribute-based theory of value with the consumer demand models developed by Hanemann (1984). According to Hanemann, consumer decisions can be decomposed into two stages: a discrete choice regarding which good to select, and a continuous choice regarding how much of the selected good to consume. In the context of a choice experiment, the structure is designed to isolate the discrete choice. Given the public nature of many non-marketed environmental goods, meaning that their supply is fixed and shared among all individuals, each respondent selects only one alternative from the choice set, considering both the cost and the attributes presented (Hoyos, 2010).

The conceptual foundation of CEs is that any good can be described by a set of attributes and the different levels those attributes may take. Respondents are presented with various combinations of these attributes in hypothetical scenarios and asked to choose the option they prefer among alternative uses of the resource. Each group of options presented in a question is called a *choice set*, which typically includes a reference scenario representing the current condition (Bashir *et al*., 2022).

The dichotomous version of the CV method and CEs share the same conceptual foundation, random utility theory, and follow similar processes in questionnaire design. The main difference between them lies in the structure of the choice: while CV presents a single specific trade-off between the current scenario and an alternative, CEs use multiple combinations of attributes across repeated hypothetical scenarios. Both methods aim to estimate consumer surplus when moving from the current situation to improved service condition (Bashir *et al*., 2022).

However, CEs offer additional advantages, such as greater flexibility in modeling choices and the ability to estimate values assigned to individual attributes relative to price. Moreover, they tend to provide increased realism, better suitability for multidimensional issues, greater adaptability, and richer information compared to CV (Bashir *et al*., 2022).

Discrete choice models are generally based on random utility theory. In RP methods, respondents are faced with many alternatives that have different characteristics. In contrast, SP methods, such as CEs using conjoint analysis, present participants with a smaller set of alternatives, typically two or three, whose characteristics are defined through an experimental design that eliminates the multicollinearity commonly found in RP data (Whitehead *et al*., 2005).

Thus, the Random Utility Model (RUM) (McFadden, 1974) serves as the conceptual foundation of CEs, where respondents are presented with sequences of

alternatives and asked to select their preferred option, thereby revealing their preferences (Bashir *et al*., 2022). A common application in environmental economics is the use of the random utility model to analyze recreational site choice. In such cases, site attributes may include fish catch rates, landscape features, water characteristics, and cost (Whitehead *et al*., 2005).

2.4.2 Choice Experiment Design

Designing a DCE requires an appropriate research framework, following the same guidelines applied to Contingent Valuation studies (Hoyos, 2010). However, special attention must be given to the conceptualization of the choice process, ensuring:

a. Incentive compatibility, so that respondents reveal their true preferences
b. Realism of the decision context, so that choices reflect real-life situations

In addition, a status quo option should be included, as it is essential for accurately estimating welfare measures.

The design of a DCE is a cyclical process that involves six fundamental stages: (1) describing choice problem, (2) definition of attributes and levels, (3) cost attribute/ payment vehicle, (4) experimental design, (5) questionnaire development, and (6) sampling strategy.

Different approaches can be applied at various stages of the design process or during pilot testing. These qualitative methods include focus groups and interviews, which assist in defining attributes, levels, and the questionnaire structure. Additionally, think-aloud exercises, open-ended feedback, and debriefing interviews are often used to identify misunderstandings and to explore respondents' experiences when completing the choice tasks (Kløjgaard *et al*., 2012).

The selected approach, or combination of approaches, should align with the objectives of the qualitative work and the specific context in which the experiment will be conducted (Kløjgaard *et al*., 2012).

2.4.2.1 Describing Choice Problem

The first step in developing a DCE is to clearly define the problem and the status quo, assessing whether this method is the most appropriate or if CV would be more suitable. The problem should be described using focus groups, literature reviews, and expert consultations, considering (Bashir *et al*., 2022):

a. Physical and temporal scope
b. Characteristics associated with the changes analyzed

It is also necessary to define who will be affected and how, by the changes in the attributes of the policy under analysis. Finally, the researcher must evaluate whether the policy can be represented through attributes, and whether marginal attributes are needed for policy analysis. The DCE is more appropriate for assessing management or policy-related issues, whereas CV is better suited for estimating the value of specific experiences (Bashir *et al*., 2022).

2.4.2.2 Definition of Environmental Attributes and Levels of Provision

The construction of choice sets in a DCE requires a clear definition of the change to be evaluated, as well as of the environmental attributes and their provision levels to be used (Hoyos, 2010). These attributes, which represent variations resulting from different policy scenarios, must be relevant to both

individuals and decision-makers, as they are the only source of information on preferences (Bashir *et al*., 2022; Hensher, 2007).

Although the context and objectives of experiments may vary and there is no single standard for defining attributes (Louviere *et al*., 2000), there is some consensus on key principles. Attributes should be quantifiable, policy-relevant, and demand-oriented (Bashir *et al*., 2022), considering that an environmental attribute is relevant if its exclusion alters the conclusions about consumer preferences (Lancaster, 1991).

Since DCEs rarely include all important attributes, it is crucial to select those that are most significant to most respondents, avoiding omitted variable bias or validity issues (Kløjgaard *et al*., 2012; Hoyos, 2010). This selection should be tested through pilot surveys (Lancsar & Louviere, 2006). The set of attributes must clearly represent the choice context and allow realistic trade-offs, consistent with the economic framework of compensatory decision-making, while reflecting respondents' true motivations (Kløjgaard *et al*., 2012; Lancsar & Louviere, 2008).

Although policy considerations may influence which attributes are included, attributes that are politically relevant but not meaningful to respondents tend to be ignored and become statistically insignificant. Nonetheless, attributes of marginal relevance to respondents but important for policy may still be included (Bashir *et al*., 2022).

Properly defining attributes to describe environmental outcomes of alternative resource management strategies (e.g., water management) is one of the most critical steps in any choice modeling application. If the environmental good being valued cannot be decomposed into a coherent and meaningful set of characteristics, both validity and value estimates may be compromised (Bashir *et al*., 2022).

The appropriate number of attributes and levels is context-specific and should balance the number of relevant attributes with the overall complexity of the experiment (Louviere *et al*., 2000). To achieve this, researchers often use focus groups, interviews, literature reviews, and expert consultations to identify representative environmental attributes and potential causal relationships among them (Bashir *et al*., 2022; Hoyos, 2010; Kløjgaard *et al*., 2012).

This balance entails a trade-off between the risk of omitted variable bias and the cognitive burden placed on respondents, of which can be assessed through focus groups and pilot tests. These sessions can also help identify potential interaction effects between attributes. Furthermore, the complexity of the choice task can be examined using verbal protocols, and providing "warm-up" tasks may help ensure respondents fully understand the decision-making process (Hoyos, 2010).

Finally, iterative pilot tests are essential for developing a DCE questionnaire. These tests should assess respondents' understanding of the context and the choice task, the appropriateness of the selected attributes and levels, and operational factors such as the duration and response time (Hoyos, 2010).

Attributes may be quantitative or qualitative, generic (with the same levels across all alternatives) or alternative-specific (with different levels or attributes between alternatives), but they should remain limited and manageable. It is also necessary to decide whether each attribute will be analyzed individually or in combination with others (Bashir *et al*., 2022; Hoyos, 2010; Kløjgaard *et al*., 2012).

During attribute selection, researchers must pay attention to causal and interdependent relationships among attributes, as these can influence respondents'

behavior and distort utility estimates (Blamey & Bennett, 2002). This issue can be addressed by excluding or combining correlated attributes, provided such decisions are clearly tested and communicated to respondents, even if this entails some loss of information (Kløjgaard *et al*., 2012).

The second stage in designing a DCE involves defining the appropriate levels for each attribute (Ryan, 1999; Lancsar & Louviere, 2008). These levels should be relevant, easy to understand, and have sufficient variation to allow trade-offs between attributes without becoming extreme or unrealistic (Green & Srinivasan, 1978).

The range of levels directly influences the estimates obtained: intervals that are too narrow make the differences insignificant, while overly wide intervals may create dominant levels, reducing respondents' willingness to make trade-offs (Kløjgaard *et al*., 2012).

Moreover, the number and distribution of levels affect the type of effects that can be estimated: two levels allow only linear effects, whereas multiple levels make it possible to detect non-linear effects. Evenly spaced levels also facilitate interpretation (Lancsar & Louviere, 2008).

Finally, it is important to avoid distortions in the perceived importance of attributes due to differences in the number of levels, since attributes with more levels tend to be seen as more relevant, by maintaining, whenever possible, a balanced number of levels across attributes (Ratcliffe & Longworth, 2002).

In most contexts where DCEs are applied, it is essential that the researcher has a deep understanding of the respondents, their evaluation system, and the decision-making context in which they are situated. Despite the extensive theoretical literature on the subject, there is still limited practical guidance on how to obtain the prior knowledge necessary to select attributes and levels that ensure face validity, that is, the extent to which the DCE includes the truly significant factors and therefore reveals the real utility of an attribute (Hall *et al*., 2004; Bateman *et al*., 2002).

Studies show a significant increase in the use of qualitative methods to enhance face validity, particularly in applications within the health sector (De Bekker-Grob *et al*., 2012). Between 1990 and 2000, only 18% of studies reported using qualitative approaches to identify attributes, a proportion that rose to 69% between 2001 and 2008. Similarly, the use of such methods to define levels increased from 18% to 33% during the same period. However, the use of qualitative pre-testing of full questionnaires declined from 47% to 38%, indicating progress in the exploratory phase but less emphasis on the final validation of instruments (Kløjgaard *et al*., 2012).

2.4.2.3 Cost Attribute/ Payment Vehicle

In CEs, the cost or monetary attribute plays a central role. Each alternative within a choice set is defined by a set of attributes, usually including a financial component, which allows respondents to make implicit trade-offs between environmental and monetary factors when expressing their preferences (Bashir *et al*., 2022).

The inclusion of a monetary or cost attribute in the choice set, combined with the random utility theory framework, enables the estimation of welfare measures such as WTP or WTA compensation for changes in the attributes levels of an environmental good or service, revealing the trade-offs among different options (Bashir *et al*., 2022; Bateman *et al*., 2002).

Cost levels should be plausible and relevant, though not necessarily reflective of current conditions. Focus groups and pilot tests can help identify appropriate cost ranges. The payment vehicle and time frame should be consistent with the valued good or service and credible, to avoid protest responses. If the proposed cost is too low, respondents may not take it seriously and may focus only on the environmental attributes when making their preferred choice (Bashir *et al*., 2022; Hoyos, 2010).

In environmental economics, such data are valuable for evaluating both policies that induce behavioral change (Phaneuf & Smith, 2005) and policies that do not directly affect behavior (Carson & Hanemann, 2005).

2.4.2.4 Experimental Design

The experimental design is a central and complex component of stated choice studies, responsible for the structured creation of choice sets, that is, the way attributes and levels are combined into alternative profiles and grouped into decision sets (Bashir *et al*., 2022; Hoyos, 2010).

It defines how attributes and their levels are organized to generate the alternatives presented to respondents, ensuring that there is no correlation among attributes and allowing for the accurate estimation of parameters (Bashir *et al*., 2022).

The way in which attributes and attribute levels are allocated to form the choice sets can significantly influence the results of the experiment (Bliemer *et al*., 2009)

In a typical DCE, respondents are presented with several hypothetical scenarios (choice sets), each containing alternatives that differ in certain attributes and levels. The choices made in each scenario are then used to estimate the parameter weights associated with each attribute, allowing the analyst to calculate marginal rates of substitution between them (Hoyos, 2010).

A well-designed experimental structure increases statistical efficiency and reduces implementation costs, and recent advances based on multinomial logit models have improved the development of optimal designs for CEs. However, beyond statistical optimality, practical factors, such as task complexity, response heuristics, and the inclusion of a *status quo* option, must also be considered. Although the *status quo* may reduce efficiency, it enhances realism and consistency with consumer theory. Overly complex designs can generate inconsistent responses, requiring greater caution in planning. Thus, evidence indicates that DCEs should balance optimality and plausibility, ensuring realism without compromising statistical efficiency.

Specialized software such as SAS, SPSS, JMP, and more recently the widely used NGENE, have been employed to generate experimental designs. Additional online resources include a library of orthogonal arrays and a webpage developed by Devorah Street and Leonie Burgess, dedicated to the construction and validation of experimental designs (Bashir *et al*., 2022; Hoyos, 2010).

2.4.3 Advantages of CE for Environmental Services (ES) Valuation

SP methods require individuals to assign economic value to non-marketed goods and services through hypothetical scenarios, such as those related to aquifer protection and drinking water purification. Among these methods, Contingent Valuation (CV) and Choice Experiments (CE) stand out, as they use questionnaires

to simulate hypothetical markets and estimate respondents' willingness to pay (WTP) (Hasler et al., 2005).

In CV, a hypothetical market is constructed in which participants receive information about the environmental problem, the proposed policy, the environmental conditions before and after the intervention, and the payment mechanism. They are then asked to choose between maintaining the current situation or paying for environmental improvement (Mitchell & Carson, 1989; Whitehead et al., 2005).

The contingent behavior approach also relies on hypothetical scenarios but, instead of directly asking for willingness to pay, focuses on the alternative behaviors individuals might adopt. For example, respondents may be asked about their frequency of visits to recreational areas with or without improvements in water quality. DCE, another choice-based approach, presents respondents with sets of alternatives containing different combinations of attributes, allowing researchers to identify their preferences and the trade-offs involved, particularly in recreational and environmental contexts (Whitehead et al., 2005).

Thus, both methods rely on hypothetical scenarios to estimate and reveal individuals' WTP. Although CV is widely used, CE is a more recent approach that has gained prominence and has been applied in numerous studies (Hasler, Lundhede, Martinsen, Neye & Schou, 2005).

Despite the overall effectiveness of stated preference methods in estimating economic values, CV presents several well-known limitations, particularly regarding the valuation of passive-use values. For this reason, when assessing non-market environmental goods, economists have increasingly adopted alternative approaches such as Choice Modelling, considered a more robust substitute for CV. The growing interest in this approach stems from the weaknesses identified in the CV literature and the expectation that CM can reduce some of its inherent biases (Bashir, Equbal & Raina, 2022).

In the CV method, the focus is on the direct description of a change in the provision of an environmental good, with respondents being presented with a specific improvement proposal. Therefore, CV is considered a direct valuation approach, as it explicitly asks individuals how much they would be willing to pay for an improvement in the quality or quantity of a public good, such as drinking water or surface water. In contrast, DCE invites respondents to choose between different alternative scenarios. The method is based on discrete choices, and the environmental good and its changes are described in terms of attributes or characteristics. The choices made reveal the trade-offs individuals are willing to make among the alternatives presented in each choice set (Hasler *et al*., 2005).

In other words, while respondents in CV directly express their willingness to pay by answering hypothetical questions, those in DCE reveal this willingness through their choices among different hypothetical alternatives (Hasler *et al*., 2005). According to Bateman *et al*. (2002) and Navrud (2000), CV is more appropriate when the goal is to estimate the total value of an environmental good or service. In contrast, DCE is better suited for assessing the relative values attributed to specific attributes or characteristics of a public good. However, aggregating the individual values of attributes may result in an overestimation of the total value of the good (Foster & Mourato, 1999).

While CV presents the good in a single, static format, CM describes it as a combination of attributes with varying levels, allowing respondents to assess trade-offs between different aspects of the good. Thus, instead of focusing on a single

attribute, typically the price or payment mechanism as in CV, CM captures preferences across multiple characteristics of the good (Bashir, Equbal & Raina, 2022).

Therefore, despite its widespread use, the CV method does not allow for the individual estimation of each attribute's value when the good being evaluated has multiple characteristics. In such cases, the CE is considered more appropriate, as it enables the estimation of WTP for each attribute individually in goods with multiple components (Bashir, Equbal & Raina, 2022).

Policy makers are generally more interested in improving the quantity or quality of specific attributes of an environmental good rather than assessing its total loss or acquisition. In this context, the DCE is the most suitable approach, as it allows for the analysis of marginal variations in environmental attributes and provides more accurate insights to support policy formulation (Bashir, Equbal & Raina, 2022).

Regarding hypothetical bias, one of the main concerns with the Contingent Valuation Method (CVM) is that, being based on hypothetical scenarios, respondents may overestimate their true values. In contrast, in CEs, the structured presentation of attributes and levels facilitates the decision-making process, making it more comparable to real market choices. Furthermore, the CE format reduces the bias of "yea-saying" (automatic affirmative responses) common in dichotomous CVM questions, as respondents face different payment levels, which encourages them to reveal more consistent and realistic preferences regarding the good being evaluated (Bashir, Equbal & Raina, 2022).

In CEs, respondents make trade-offs among sets of attributes, including non-use values, making this type of bias less prevalent compared to CVM surveys. CEs also tend to reduce strategic bias, as monetary values are predefined within the choice sets, thereby lowering the risk of over- or underestimating true values. Finally, CVM surveys, especially those involving large probabilistic samples and individual interviews, can be highly costly, and expenses increase even further when split sampling is required. In such cases, CEs can significantly reduce costs, since they allow for the estimation of multiple attribute values within a single questionnaire, providing more information than CVM studies based on dichotomous choices (Bashir, Equbal & Raina, 2022).

2.4.4 Applications of Choice Experiments in Wetlands

Discrete Choice Experiments, as survey-based stated preference methods, have been increasingly applied to explore public preferences and inform policy decisions in both developed and developing countries. In the context of wetland management, DCE studies used to determine what features people want for long-term conservation have revealed that people tend to favor conservation strategies that combine environmental protection with livelihood-supporting activities. In developing countries, DCEs have been particularly useful in assessing willingness to pay for improved environmental quality, thereby supporting sustainable resource management. Despite their effectiveness in resource-starved areas, successful application of DCEs requires specialized expertise in design and implementation (Kong, Mohamad, Rosland, & Lee, 2025).

According to Tentes and Damigos (2014), the available literature on water valuation using Choice Experiments can be grouped into six main categories:

1. Wetlands
2. Freshwater quality and health

3. Seawater and coastal water quality
4. Water Framework Directive 2000/60/EC and river basin management
5. Public water services and urban water
6. Groundwater

In addition to these categories, some studies consider water as an attribute within the valuation of other resources, such as land and soil use (Birol *et al*., 2008), while others apply modeling approaches for water-related action planning (Panagopoulos *et al*., 2013; Getzner, 2014).

Specifically, regarding wetlands, the focus of this study, despite the growing number of valuation studies on this environmental resource, the application of CE remains limited, highlighting an important gap in existing literature. Thus, the CE applied to the Risorgive del Bacchiglione in North-Eastern Italy represents a valuable contribution to this still scarce body of literature.

The main objective of wetland valuation studies has been to estimate the WTP for the conservation or improvement of wetland areas, particularly regarding the use and non-use values associated with environmental quality. In CEs, on the other hand, the focus is on assessing the value attributed to different wetland attributes, such as biodiversity, ecosystem services, and other specific environmental characteristics.

Several studies have applied the CE method to the valuation of wetlands across different regions. Notable examples include: Morrison, Bennett, and Blamey (1999) in the Macquarie Marshes, Australia; Carlsson, Frykblom, and Liljenstolpe (2003) in the Staffanstorp wetland, Sweden; Othman, Bennett, and Blamey (2004) in the Matang mangroves, Malaysia; Birol, Karousakis, and Koundouri (2006) in the Cheimaditida wetland, Greece; Christie and Rayment (2012) in the Sites of Special Scientific Interest (SSSI) in England and Wales; Tempesta and Vecchiato (2013) in the Serio River Park, Italy; Tentes and Damigos (2014) in the Asopos River, Greece; Kountouris, Godyn, and Sauer (2014) in Czestochowa, Poland; Dias and Belcher (2015) in the province of Saskatchewan, Canada; Chaikaew, Hodges, and Grunwald (2017) in the Suwannee River Basin, USA; Castillo-Eguskitza, Hoyos, Onaindia, and Czajkowski (2019) in the Urdaibai Biosphere Reserve, Spain; Hernandez and da Costa (2022) in the Taravo Wild River, Corsica, France; and Udugama *et al*. (2024) in natural pools in the Kandy district, Sri Lanka.

Based on consumer theory, the marginal rate of substitution (MRS) between two attributes is obtained by the ratio of the partial derivatives of the indirect utility function with respect to each attribute. When the utility function is linearly additive, the MRS corresponds to the ratio of the estimated parameters. If the second attribute is the cost attribute, this ratio represents the implicit price of the analyzed attribute (Hoyos, 2010):

$$MRS_{X_1,X_2} = \frac{\partial V/\partial X_1}{\partial V/\partial X_2}$$ *(Equation 1)*

The compensating surplus (CS), which represents the welfare change associated with a variation in attribute levels, can be calculated according to Hanemann (1984b) and Train (1988):

$$CS = -\frac{1}{\alpha}\left[\ln\left(\sum \exp\left(\beta X_y^0\right) - ln\left(\sum \exp\left(\beta X_y^1\right)\right)\right]\right.$$ *(Equation 2)*

Where: α= marginal utility of income (typically represented by the coefficient of the cost attribute) / X_y^0 = vector of environmental attributes at the initial level (status

quo) / X_y^1 = vector of environmental attributes after the change in environmental attributes

In practical terms, the CS reflects the expected change in utility resulting from a variation in the provision level of an environmental good, weighted by the marginal utility of income (Hoyos, 2010).

Simplifying the above equation, the marginal change in an attribute is estimated by the ratio between the attribute's coefficient and the cost coefficient, representing the monetary value assigned to the change in each attribute (Hoyos, 2010):

$$WTP = -\frac{\beta_{attribute}}{\beta_{cost}}$$ *(Equation 3)*

Where: WTP= marginal WTP for a change in the attribute / $\beta_{attribute}$= coefficient of the attribute being evaluated (represents how much the utility changes with a unit change in the specific attribute) / β_{cost} = cost coefficient (represents how utility decreases as the cost increases) / $-\frac{\beta_{attribute}}{\beta_{cost}} =$ implicit price (indicates the value individuals assign to changes in a specific attribute)

Thus, the willingness to pay for a marginal change in the provision level of an environmental attribute is obtained by the ratio between the attribute's coefficient and the cost attribute's coefficient, representing the marginal rate of substitution between income and the change in the attribute.

Because the equation captures the monetary trade-offs individuals are willing to make for changes in specific attributes of an environmental good or service, the result highlights the importance of each attribute and the economic value individuals assign to improvements, which is crucial for policy-making and resource management.

2.4.5 Methodological Issues in Choice Experiments

Internal validity tests verify whether individuals' preferences are stable, complete, monotonic, and transitive, with preference consistency observed when these axioms are satisfied in stated preference (SP) survey designs. Some studies have reported preference inconsistencies, especially when moving from CVM to DCEs, where greater complexity due to the number of attributes may lead to less consistent responses. However, more careful experimental designs and analyses can mitigate these issues. Rationality tests are used to identify inconsistent respondents, as irrational responses may bias estimates and increase variance; nonetheless, removing all such responses may cause sample selection bias and reduce the statistical efficiency of the model (Hoyos, 2010).

External validity refers to empirical evidence that the choices and utility estimates derived from DCEs reflect behavior observed in real markets, comparing actual and hypothetical behavior. The literature indicates that DCEs generally exhibit good external validity and less hypothetical bias than the CVM, which is expected since DCEs more closely resemble real consumer behavior. However, research has shown that there may be hypothetical bias in mean WTP, but not in marginal WTP, suggesting that DCEs accurately estimate marginal changes but perform poorly in predicting actual market shares (Hoyos, 2010).

Regarding welfare measures, both CVM and DCEs yield similar parameters for the marginal utility of income, resulting in economically equivalent values once variance heterogeneity is considered (Hoyos, 2010).

Finally, some biases typically associated with CVM have also been examined in DCEs. Regarding scope sensitivity, studies have shown that DCEs tend to exhibit greater sensitivity than CVM. However, research also indicates that DCEs may be susceptible to anchoring effects or starting point bias, common in CVM. As a generalization of CVM, the experimental design is equally crucial in DCEs, and that careful survey planning can help mitigate such biases. Therefore, is crucial to minimize framing and anchoring effects, ensuring that stated preferences in non-market valuation studies closely reflect the preferences individuals would reveal in real market situations. Nevertheless, for DCEs to become fully established methodologically, further testing and empirical validation of their properties are still required, as has been done with CVM (Hoyos, 2010).

a. Framing Effect

The framing effect, inherent to stated preference methods, refers to the way in which the decision context is presented, influencing respondents' choices. The information provided in a Choice Experiment questionnaire plays a substantial role in decision-making, for example, in the assessment of costs and benefits. Since many individuals do not have pre-formed preferences regarding the environmental goods being evaluated, their decisions may be shaped by the information and framing of the survey. Therefore, it is essential that the survey frame be clearly defined and carefully designed to ensure consistent and reliable results, as consumers' preferences may adjust to both the presentation format ("survey frame") and the informational content of the questionnaire (Bashir *et al*., 2022).

b. Anchoring Effects

Welfare estimates in CEs are sensitive to the study design, including the selection and presentation of attributes and their levels, which can influence marginal utilities and consumer surplus. Anchoring bias occurs when respondents are influenced by the way attributes are presented rather than by their true preferences, a phenomenon also known as the starting point bias in CVM. Anchoring bias occurs when respondents interpret the values presented as indications of what would be an appropriate answer, rather than basing their responses on their true preferences. Studies in Contingent Valuation have shown that starting bids can influence respondents' WTP, and similarly, cost levels in Discrete Choice Experiments may affect the implicit price of attributes, making them also susceptible to anchoring bias (Bashir *et al*., 2022).

2.5 Motivational Theories

Cultural ecosystem services arise from the interaction between the biophysical characteristics of a place and the psychological aspects of its visitors. Therefore, when assessing a natural area, such as through the attributes included in a DCE, it is essential to consider not only the site's physical features but also the values, motivations, and profiles of visitors or potential visitors. Understanding these motivations allows for the identification of groups with similar needs and expectations, which helps natural area managers design strategies tailored to each segment, thereby enhancing environmental conservation, increasing visitor satisfaction and willingness to pay, and contributing to crowd management and the reduction of user conflicts (Pellizzari, 2022).

In literature, motivation is defined as the set of psychological needs that influence human behavior. Over time, several theories have been developed to

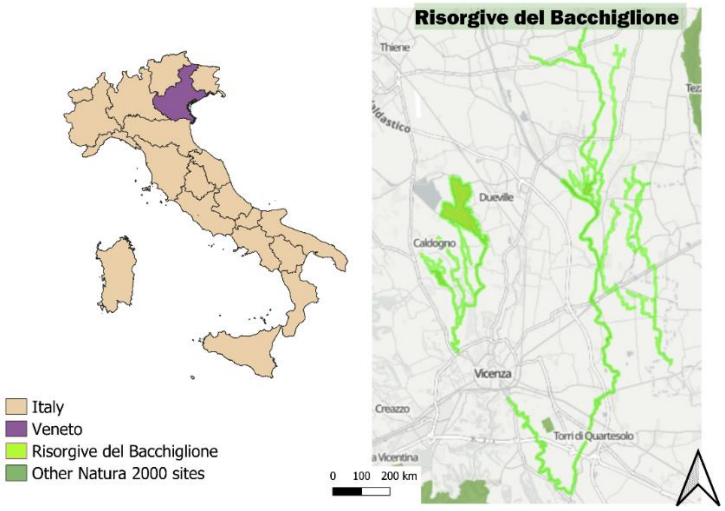
identify the motivational factors that affect the valuation of wetlands and groundwater. Since motivations shape these decisions and consequently influence individuals' WTP for the conservation of wetland areas, future studies assessing the introduction of a fee at *Risorgive del Bacchiglione* should investigate not only preferences for environmental attributes (through a DCE) but also motivational theories. This approach would enable a deeper understanding of the different motivational groups among individuals and how these motivations affect their WTP.

3 Material and Methods

3.1 Study Site

The Risorgive del Bacchiglione natural area, located in the Province of Vicenza (Northeastern Italy), lies in the upper part of the Veneto Plain, on the border between the municipalities of Dueville, Villaverla, and Caldogno, near the confluence of the Bacchiglioncello and Timonchio streams. Between latitude 45.5621 and longitude 11.5204, the area covers approximately 715 hectares, with an average elevation of 48 meters above sea level (Figure 9).

Figure 9. Location map of the Risorgive del Bacchiglione, Veneto, Italy (Official Natura 2000 map – Eurostat)



Source: Own thematic elaboration on QGIS 3.28.12 platform (2025)

The site lies along the “fascia delle risorgive” (spring line), a transition zone between the coarse gravel deposits from the Alps and the finer sand and clay sediments of the Po Valley. In the Veneto Plain, particularly around Dueville, groundwater flowing from the Pre-Alps is forced to the surface by thick impermeable clay layers, which block its underground flow and give rise to numerous natural springs (risorgivedelbacchiglione.it).

The area is composed of springs and channels that flow into progressively larger watercourses. To the south and east of Dueville, there is a complex network of springs forming a dense hydrographic system that crosses the plateau north of Vicenza and flows into the region’s main rivers. The lands along these waterways are intensively used for agriculture.

There are also non-irrigated agricultural areas, such as hay meadows and cornfields, as well as wet meadows dominated by Molinia, with Molinietum woodlands developing on moist and peaty soils along the margins of springs and channels. The landscape also features extensive hedgerows and wooded areas that contribute to ecological connectivity.

The most significant natural elements are mainly found along the banks of irrigation channels and ditches, where well-preserved riparian vegetation can still be observed, and within the riverbeds themselves, which are rich in aquatic vegetation and ecologically important fish species, such as the Po Valley lamprey.

These springs form the typical features of the region, small depressions where the constant outflow of water creates resurgent streams. A resurgence (or spring) is a natural freshwater source, sometimes artificially brought to the surface by humans, common in alluvial plains such as the 'Pianura Padano-Veneta'. The Bacchiglione River is one such spring-fed river (*fiume di risorgiva*), formed by the union of the Bacchiglioncello channel, which collects several spring outlets in the area, with the Timonchio torrent (risorgivedelbacchiglione.it).

Originating at the foot of the Vicenza Pre-Alps, the Bacchiglione flows across a wide portion of the Po Valley and has historically served as an important communication route, with its waters used since antiquity for agricultural, industrial, and commercial purposes (risorgivedelbacchiglione.it).

The constant groundwater flow gives rise to small pools (locally called *polle* or *olle*) which overflow into a network of streams and irrigation channels. These perennial springs have crystal-clear, cool waters that maintain a stable temperature between 10 and 13 °C throughout the year, creating a unique microclimate that supports rare plant and animal species within the surrounding agricultural landscape (risorgivedelbacchiglione.it).

The site forms a green oasis within the agricultural landscape and represents the core of an ecological corridor stretching north to south across the Vicenza plains. The continuous year-round water flow and diverse range of habitats make the Risorgive del Bacchiglione an essential natural asset for regional biodiversity and landscape connectivity (risorgivedelbacchiglione.it).

3.1.1 Historical Aspects of Risorgive del Bacchiglione

The Bacchiglione spring area, located within the Dueville Forest, holds great ecological significance, forming part of one of the largest aquifer systems in Europe and serving as the core of the ecological corridor that extends through the Province of Vicenza from north to south (risorgivedelbacchiglione.it).

In the 1960s, human interventions significantly altered the site: parts of the area were converted into artificial ponds and channels for commercial fish farming, changing the natural morphology of the springs, diverting most of the water flow, and confining the Bacchiglioncello to the outer branch along the eastern side of the area (risorgivedelbacchiglione.it).

In the 2000s, ecological restoration efforts began. In 2009, the Province of Vicenza acquired about 20 hectares of former fish farming areas and, with the support of the European Union, implemented the LIFE+ SOR.BA. project (09/NAT/IT000213) in collaboration with the Regional Forestry Service, the Peripheral Unit of Vicenza, Veneto Agricoltura, and the Alta Pianura Veneta Land Reclamation Consortium, aiming to restore the hydrology and natural habitats of the springs from the Bacchiglione River, one of the most important spring-fed river in Europe (risorgivedelbacchiglione.it).

Since 2017, at the request of the Province, the Risorgive del Bacchiglione have been managed as a nature reserve by Viacqua, a public company responsible for the integrated water service (supply, sewerage, and wastewater treatment) across 68 municipalities in the Province of Vicenza. In collaboration with local associations, environmental groups, and volunteers, Viacqua works to make the site accessible, educational, and welcoming, while promoting conservation and environmental awareness activities (risorgivedelbacchiglione.it).

The area is divided into three zones with different management uses (risorgivedelbacchiglione.it):

a) Public recreation zone, featuring nature trails and open to visitors;
b) Controlled-access zone, available only through guided tours;
c) Strict protection zone, closed to the public and reserved exclusively for scientific research.

3.1.2 Biodiversity and Ecological Aspects of Risorgive del Bacchiglione

The Risorgive del Bacchiglione exhibit unique environmental characteristics and are subject to important ecological protection measures. As part of one of the largest aquifer systems in Europe, the Alpine aquifer of the Veneto Plain, they represent a rare example of spring ecosystems that remain close to their natural state in the region. Owing to their high ecological value, the springs are included in the Natura 2000 network as a protected area, a European Union (EU) initiative aimed at ensuring the long-term preservation of natural habitats and of rare or threatened plant and animal species.

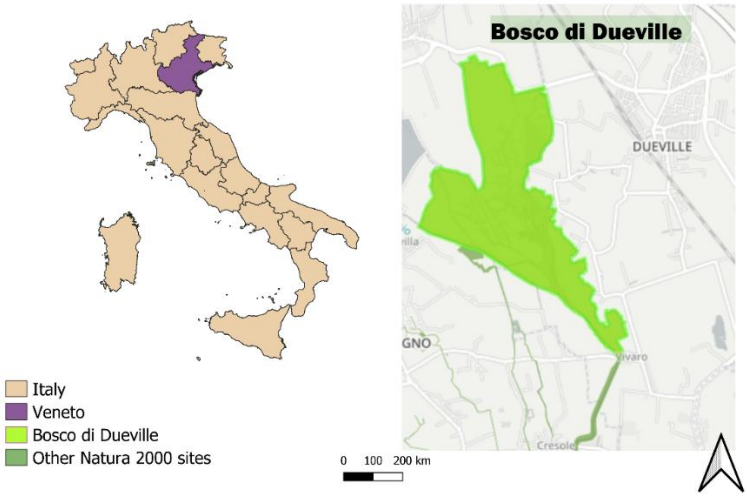
Natura 2000 is a European Union network of protected areas established under two key directives: the 1979 Birds Directive (79/409/EEC, later replaced by 2009/147/EC) and the 1992 Habitats Directive (92/43/EEC). Its purpose is to conserve Europe's most valuable and threatened species and habitats. The network comprises more than 27.000 terrestrial and marine sites, including Sites of Community Importance (SCI) under the Habitats Directive and Special Protection Areas (SPA) under the Birds Directive, together covering over 100 million hectares, approximately 18% of the EU's land area and 10% of its marine area. The Natura 2000 network serves as a cornerstone of the EU Biodiversity Strategy to 2020 (European Commission, 2017).

The habitat types designated as SCI represent outstanding examples of the typical characteristics of one or more of the following biogeographical regions: Alpine, Atlantic, Continental, Macaronesian, Mediterranean, Black Sea, Boreal, Pannonian, Steppic, Anatolian, and Arctic. The main biogeographical regions of Italy are Alpine, Continental, and Mediterranean.

The Risorgive del Bacchiglione cover a wide area and are simultaneously designated as SCI "Bosco di Dueville e risorgive limitrofe" (site code IT3220040) and as SPA "Bosco di Dueville" (site code IT3220013) for birds. These designations indicate that the site hosts habitats and species of conservation interest in Europe (risorgivedelbacchiglione.it).

The SPA IT3220013, established in 2003, covers an area of 319 hectares (Figure 10) (biodiversity.europa.eu).

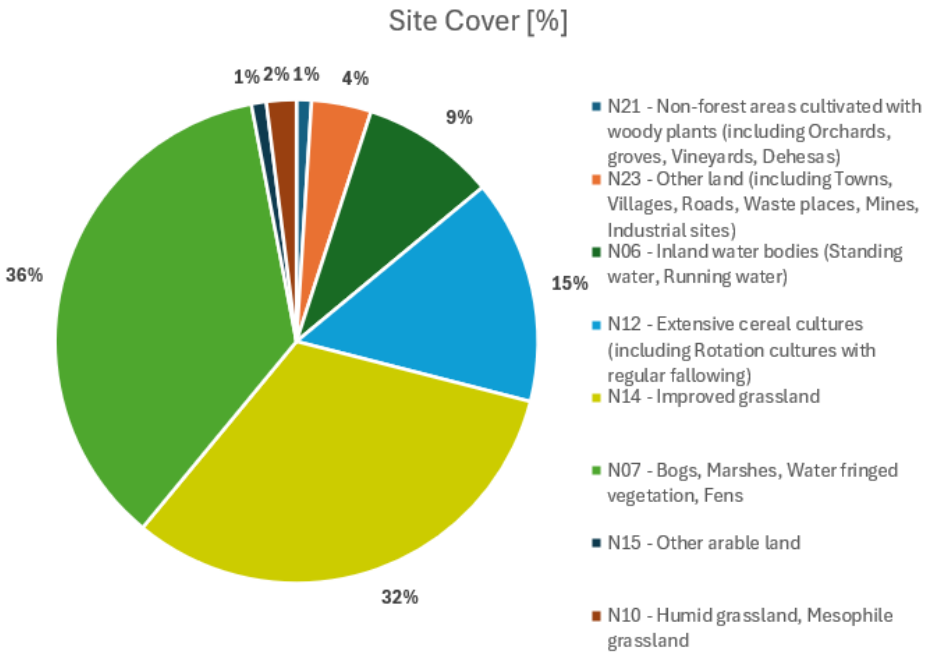
Figure 10. Location map of the Bosco di Dueville, Veneto, Italy (Official Natura 2000 map – Eurostat)



Source: Own thematic elaboration on QGIS 3.28.12 platform (2025)

The total coverage of the SPA site IT3220013 includes the following habitat classes, as shown in Figure 11:

Figure 11. Site characterization by % cover of habitat classes

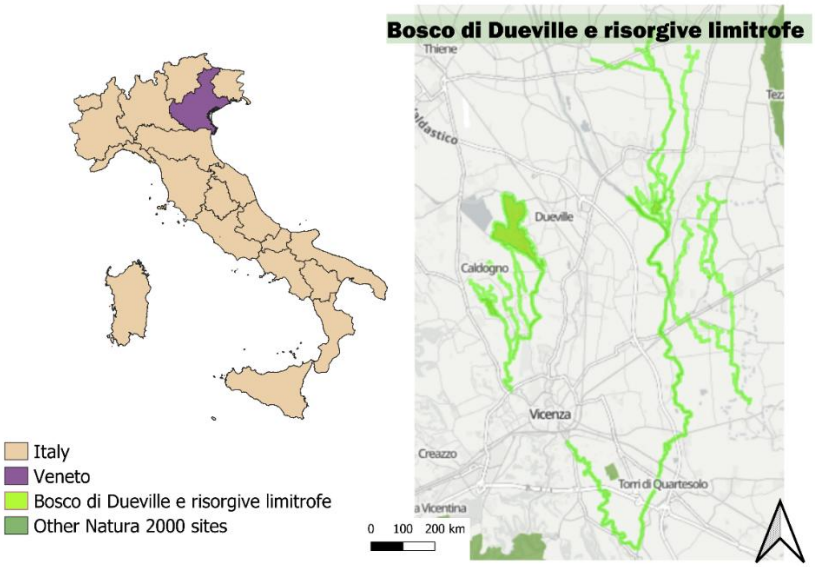


Source: Author's elaboration based on the Natura 2000 Standard Data Form (2025)

Located in a 100% Continental biogeographical region, it includes seven species protected under the EU Habitats and Birds Directives, all of which are birds: Kingfisher (*Alcedo atthis*), Barn Owl (*Tyto alba*), Long-eared Owl (*Asio otus*), Night Heron (*Nycticorax nycticorax*), Scops Owl (*Otus scops*), Water Rail (*Rallus aquaticus*), and Woodcock (*Scolopax rusticola*) (biodiversity.europa.eu).

The SCI IT3220040, established in 2006, covers 715 hectares, including the 319 ha of SPA IT3220013 (Figure 12) (biodiversity.europa.eu).

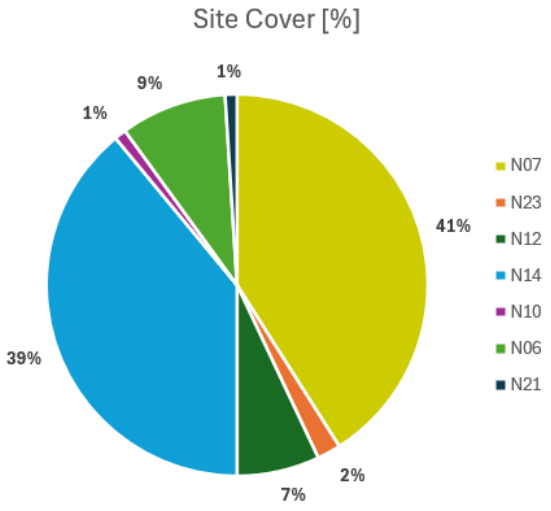
Figure 12. Location map of the Bosco di Dueville e risorgive limitrofe, Veneto, Italy (Official Natura 2000 map – Eurostat)



Source: Own thematic elaboration on QGIS 3.28.12 platform (2025)

The total coverage of the SCI site IT3220040 includes the following habitat classes, as shown in Figure 13:

Figure 13. Site characterization by % cover of habitat classes



Source: Author's elaboration based on the Natura 2000 Standard Data Form (2025)

Also located in a 100% Continental biogeographical region, it contains five protected habitats under the EU Habitats Directive and four protected species under the EU Habitats and Birds Directives (biodiversity.europa.eu).

The protected habitats include (biodiversity.europa.eu):

a) Forest: priority habitat 91E0 – Alluvial forests with Alnus glutinosa and Fraxinus excelsior (Alno-Padion, Alnion incanae, Salicion albae);
b) Freshwater: 3260 – Water courses of plain to montane levels with Ranunculion fluitantis and Callitricho-Batrachion vegetation;
c) Grasslands: 6430 – Hydrophilous tall herb fringe communities of plains and of the montane to alpine levels, 6510 – Lowland hay meadows (Alopecurus pratensis, Sanguisorba officinalis), and 6410 – Molinia meadows on calcareous, peaty or clayey-silt-laden soils (Molinion caeruleae).

The protected species include (biodiversity.europa.eu):

a) Amphibians: Italian agile frog (Rana latastei);
b) Fish: Freshwater sculpin (Cottus gobio), Lombardy brook lamprey (Lethenteron zanandreai), and Telestes muticellus.

The riparian forests of the Risorgive del Bacchiglione constitute a priority natural habitat type, meaning a habitat at risk of disappearing, for which the European Committee holds special responsibility for conservation due to the significant portion of its natural range occurring within this territory.

These forests include formations of ash (*Fraxinus excelsior*) and alder (*Alnus glutinosa*), typical of temperate and boreal Europe lowland and hill watercourses; riparian woods of *Alnus incana*, found along montane and submontane rivers of the Alps and northern Apennines; and tree galleries dominated by willows (*Salix alba, S. fragilis*) and poplars (*Populus nigra*), characteristic of lowland, hilly, and submontane rivers of Central Europe. These habitats occur on humid alluvial soils, periodically flooded and rich in herbaceous species and vernal geophytes such as *Filipendula ulmaria, Carex spp*., and *Anemone nemorosa*. Among the main subtypes, the most representative are the ash–alder woods of fast- or slow-flowing rivers (*Alno-Padion*), the montane and submontane grey alder galleries (*Alnion incanae*), and the white willow gallery forests (*Salicion albae*), all essential for the conservation of riparian vegetation and associated biodiversity (biodiversity.europa.eu).

The habitat type 3260 is characterized by watercourses of plains and mountains with submerged or floating vegetation, typical of the *Ranunculion fluitantis* and *Callitricho–Batrachion* alliances (during dry periods, with reduced water levels), or aquatic mosses. This habitat may be associated with communities of the *Butomus umbellatus* group and is considered highly relevant for the selection of conservation areas (biodiversity.europa.eu).

The habitat type 6430 is characterized by hygrophilous and nitrophilous perennial tall herb communities occurring along watercourses and forest edges, belonging to the orders *Glechometalia hederaceae* and *Convolvuletalia sepium*, and to the class *Betulo–Adenostyletea.* They are typical of montane and alpine environments, characterized by high humidity and nutrient-rich soils (biodiversity.europa.eu).

The habitat type 6510 is characterized by species-rich hay meadows developed on slightly to moderately fertilised soils, typical of plains and submontane areas, belonging to the *Arrhenatherion* and *Brachypodio–Centaureion nemoralis* alliances. These extensive grasslands, rich in wildflowers, are not mown before the flowering of grasses and are managed only once or twice a year (biodiversity.europa.eu).

The habitat type 6410 is characterized by molinia meadows, found from lowlands to montane areas, occur on wet, nutrient-poor soils, resulting from

extensive management with late annual mowing or from degraded drained peatlands.

They may develop on neutral to calcareous soils, rich in species (*Eu-molinion*), or on acidic soils associated with the *Junco-Molinion* (*Juncion acutiflori*) alliance, which are typically species-poor or degraded and peaty (biodiversity.europa.eu).

From an ecological perspective, the Risorgive del Bacchiglione represents a mosaic of wetland and riparian habitats embedded within an intensively cultivated landscape, functioning as an ecological refuge due to the high habitat diversity within a relatively small area, where much of the natural cover has been lost (risorgivedelbacchiglione.it).

The landscape, strongly shaped by the presence of water, consists of spring-fed channels, ponds, and a high groundwater table, elements that directly influence local vegetation and land use. The area is characterized by small linear woodlands along ditches and canals, wet meadows dominated by rushes and sedges, and streams with aquatic vegetation typical of spring and slow-flowing waters, bordered by dense riparian vegetation. It also includes hay meadows, extensive agricultural and grazing lands, reedbeds (canneti), sedge marshes (cariceti), Molinia meadows on peaty soils, and fragments of alluvial forest, such as the Bosco di Dueville, forming a natural-like wetland environment within a highly anthropized, predominantly agricultural context (risorgivedelbacchiglione.it).

This diversity of habitats supports high biodiversity, including plant and animal species of conservation interest. The wet Molinia meadows on peat soils and sedge marshes stand out as rare and increasingly scarce habitats in the heavily drained Po Plain, hosting diverse plant and invertebrate communities, as well as crucial breeding areas for amphibians (risorgivedelbacchiglione.it).

The area serves as an important feeding and breeding site for aquatic birdlife, such as the night heron (*Nycticorax nycticorax*), and hosts rare plant and animal species associated with wetland ecosystems. It is notable for the relict presence of rare hygrophilous and microthermal plant species, as well as a very rare endemic plant association of high ecological value (*Plantagini altissimae – Molinietum caeruleae*). Rare or declining animal species have also been recorded in the area (risorgivedelbacchiglione.it).

The wetlands provide a permanent water supply, even during dry periods, supporting robust populations of aquatic and semi-aquatic fauna and functioning as important resting and feeding grounds for migratory birds crossing the Po Valley during seasonal migrations (risorgivedelbacchiglione.it).

Among the species of community interest found in the Risorgive del Bacchiglione are amphibians and reptiles such as the Italian agile frog (*Rana latastei*), endemic to the Po Valley lowlands, and the European pond turtle (*Emys orbicularis*), as well as the dice snake (*Natrix tessellata*), which thrives in the spring-fed channels. The permanent presence of water and abundant aquatic vegetation also support a diverse fish community, including native species indicative of good water quality, such as the weather loach (*Cobitis sp.),* the European bullhead (*Cottus gobio*), and the Italian vairone (*Telestes muticellus*) (risorgivedelbacchiglione.it).

The birdlife is equally remarkable: the mosaic of open areas, wetlands, and woodlands provides habitat and nesting sites for a variety of species. The kingfisher (*Alcedo atthis*) is often seen along the spring streams, while the wetlands serve as feeding and breeding grounds for species such as the marsh harrier (*Circus*

aeruginosus), the hen harrier (*Circus cyaneus*), the night heron (*Nycticorax nycticorax*), and the little egret (*Egretta garzetta*), among other waterbirds (risorgivedelbacchiglione.it).

In addition to their rich biodiversity, the springs perform essential ecological functions. They act as natural water reservoirs, releasing steady discharges that maintain stream flows even during dry periods. This regular water supply has been crucial for the historical development of local agriculture and continues to support downstream ecosystems, contributing to the regulation of hydrological flows (risorgivedelbacchiglione.it).

The dense wetland vegetation, composed of reeds, cattails, and peat mosses, promotes natural water purification: as groundwater emerges and flows through the marsh, biological filtration and nutrient uptake occur. A healthy spring system, with abundant vegetation and slow-flowing waters, has a strong self-purification capacity, reducing organic and inorganic contaminants (risorgivedelbacchiglione.it).

Wetlands also play a key role in local climate regulation. Their saturated soils and high plant biomass enhance carbon sequestration, as these habitats accumulate peat and organic matter over time, thereby removing carbon from the atmosphere and helping to mitigate climate change. Moreover, the stable water temperature contributes to microclimate moderation, preventing extreme temperature fluctuations in the surrounding areas (risorgivedelbacchiglione.it).

3.1.3 Ecosystem Services of Risorgive del Bacchiglione

Using the conceptual framework of the Millennium Ecosystem Assessment, the ecosystem services provided by the Risorgive del Bacchiglione can be categorized into four main groups: Provisioning, Regulating, Supporting, and Cultural Services. The following sections describe each category and the specific ecosystem services that are particularly relevant to this spring system.

3.1.3.1 Provisioning Services

Provisioning services are the tangible goods and resources provided by the ecosystem. The main provisioning service of the springs is the supply of freshwater, which is essential for local water availability. Historically used for agricultural irrigation, the Risorgive still recharge irrigation canals and maintain soil moisture in surrounding farmlands. The steady flow of high-quality water can also be used for drinking water supply after minimal treatment, as the local water utility manages the Risorgive del Bacchiglione, highlighting their importance for regional water resources.

Another provisioning service is the supply of biological resources from the wetlands. The springs provide habitat for various plant and animal species that serve as food and natural materials, such as fish, crayfish, reeds, and willows, indirectly supplying food resources (fish, game, wild herbs) to local communities. Although harvesting is now limited due to conservation efforts, these wetlands still represent a potential or symbolic provisioning value, reflecting the nutritional and cultural benefits of a healthy spring ecosystem.

3.1.3.2 Regulating Services

Regulating services represent the benefits arising from the regulation of natural processes within the ecosystem. Hydrological regulation stands out,

ensuring the maintenance of base flows and groundwater levels. The Risorgive act as natural “safety valves”, gradually releasing groundwater into surface streams, which stabilizes river flow regimes, such as those of the Bacchiglione and its tributaries, thus reducing the risk of floods and droughts. During dry periods, the springs sustain flows vital to downstream ecosystems and communities, while in wet seasons, the peat soils act like sponges, absorbing and slowing runoff, thereby contributing to the hydrological balance of the landscape.

Another fundamental service is the natural purification of water. As groundwater flows through pools and wetland areas, it is filtered by soil and plant roots, which remove or transform pollutants such as excess nutrients, organic matter, and heavy metals. Springs in good ecological condition have a high self-purification capacity, which significantly improves downstream water quality, benefiting human water supply and the maintenance of aquatic ecosystems.

The springs also play an important role in climate regulation at both local and regional scales. Wetlands sequester carbon, storing atmospheric CO_2 in peat and biomass, and cool the microclimate through evapotranspiration. Over time, the carbon accumulated in peat soils contributes to climate change mitigation.

Moreover, the dense vegetation acts as a natural filter, trapping dust and air pollutants and serving as a windbreak, which improves air quality and helps stabilize the local climate.

Finally, springs support erosion control: vegetation stabilizes the soil, and the gradual release of water prevents sudden runoff and erosion, thereby protecting soils and nearby waterways.

3.1.3.3 Supporting Services

Supporting services underpins all others by maintaining the fundamental ecological functions of the system. The main supporting service of the Risorgive is the provision of habitat for biodiversity. The springs create a unique ecological niche, characterized by constant cool water, wetland vegetation, and transitional zones between aquatic and terrestrial environments, which sustain the life cycles of many species. They serve as breeding grounds for amphibians and fish, feeding areas for birds, and refuges for species that have disappeared from the surrounding drained lands.

MEA classification includes in supporting services primary production and nutrient cycling. In the Risorgive, abundant water and nutrient-rich soils promote high primary productivity, the generation of biomass through photosynthesis, supporting food webs of insects, birds, and other wildlife. Nutrient cycling is enhanced by the wetlands’ ability to retain and transform nutrients; for instance, nitrogen and phosphorus from upstream sources are absorbed by plants or denitrified in waterlogged soils.

The springs also play a crucial role in the regional hydrological cycle, acting as groundwater discharge points that feed larger rivers and even the Adriatic Sea, thereby connecting aquifers and surface waters.

Supporting services provide indirect benefits to people, often perceived only in the long term. For instance, by serving as habitats, the springs sustain pollinators and natural pest predators, which benefit local agriculture, even if this relationship is not immediately visible.

In summary, by providing essential habitats, preserving genetic diversity, and sustaining key ecological processes, the Risorgive forms the foundation for the provision of all other ecosystem services in the region.

3.1.3.4 Cultural Services

Cultural services are the intangible benefits people derive from ecosystems, such as recreation, education, aesthetics, spiritual values, and cultural heritage, all of which contribute significantly to human well-being, even though they have no market value. The Risorgive del Bacchiglione provide a wide range of cultural services that strengthen the connection between local communities, visitors, nature, and the region's historical heritage.

The Bacchiglione springs possess remarkable landscape and cultural value, forming an integral part of the traditional landscape of the Venetian plains and shaping settlement patterns and land use over the centuries. Once regarded as agricultural obstacles for creating swampy and less arable lands, the natural springs later came to be valued for their irrigation potential and livestock use, giving rise to a cultural landscape where wet meadows and drained fields coexist with spring woodlands.

The site represents an important cultural heritage, shaped by the cultural identity formed around these springs because of the historical relationship between local communities and spring waters. This interaction has left visible traces on the landscape, such as ditches, small dams, traditional place names, and legends associated with specific springs, forming a tangible and intangible legacy. Traditional practices, such as irrigation techniques and the use of water for livestock, together with historical events like land reclamation and community water management, have created a collective memory linked to the springs. These springs are an integral part of the regional identity of the Venetian plain and are recognized as a "cultural heritage associated with historically valuable landscapes." Preserving them means safeguarding the cultural landscape and the historic irrigation structures that shaped local society. The Natura 2000 designation ensures the preservation of the site's ecological value, as well as its historical and landscape character.

Nowadays, the Risorgive del Bacchiglione is recognized as a natural and cultural heritage site, where people can reconnect with the region's environmental history. The combination of legal protection, unique hydrology, rare habitats, and high biodiversity highlights the area's exceptional value and the importance of its conservation and sustainable management.

From an aesthetic perspective, the springs and their surrounding landscape evoke inspiration and a deep sense of place. The clear bubbling waters, lush wetlands, and light filtering through the trees create a widely admired scenic beauty, which inspired the establishment of the reserve, its inclusion in ecotourism routes, and the appreciation of nearby properties. Many people choose to live in or visit the area precisely for the beauty and tranquility it provides.

Moreover, the area serves as an important site for recreation and ecotourism. The reserve is open to the public on weekends during the warmer months, offering guided tours, hiking trails, birdwatching, and educational activities. Visitors experience enjoyment, relaxation, and well-being while spending time in this peaceful natural environment. The recreational use of the springs is supported by studies showing that natural landscapes strongly influence people's choice of leisure destinations, and the growing number of visitors, including schools and families, reflects the increasing appreciation of the site.

Closely related to this is the educational and scientific value of springs, which serve as an open-air classroom and research site. Local schools conduct field trips to study hydrogeology, wetland ecology, and biodiversity conservation, while

researchers monitor the area through initiatives such as the LIFE+ project and various biological surveys, making it a living laboratory for science and environmental monitoring.

3.2 Data Collection of Journal Publications

This assessment draws on a meta-analysis of the scientific literature on choice experiment applications for evaluating water ecosystem services, conducted between July and August 2025, in other words a qualitative systematic literature review. The studies included in this assessment were identified through a keyword search in the Scopus database that have been published between 1999 and 2025 in English, using the search strings listed in Table 7. Scopus was selected as it is widely recognized as a consolidated reference for literature reviews of published scientific articles.

Table 7. Summary of search strings

Search string
Search string (TITLE-ABS-KEY((‘discrete choice experiment’ OR ‘choice model’) AND (‘water ecosystem services’ OR ‘river catchments’ OR ‘water resource’ OR ‘wetland attributes’ OR ‘wetland values’ OR ‘water quality’) AND (‘willingness to pay’ OR ‘environmental valuation’ OR ‘stated preference’ OR ‘non-market valuation’ OR ‘indirect use values’ OR ‘non-use values’) AND (‘random parameter logit’ OR ‘multinomial logit model’ OR ‘welfare estimates’))
Search string (TITLE-ABS-KEY(""discrete choice" AND "spring ecosystem services" AND "attributes" AND "freshwater")
Search string (TITLE-ABS-KEY(""stated preference" AND "river ecosystem services" AND "freshwater" AND "attributes")
Search string (TITLE-ABS-KEY(""choice modelling" AND "spring ecosystem services" AND "attributes" AND "freshwater")
Search string (TITLE-ABS-KEY(""choice experiment" AND "ecosystem services" AND ("river" OR "spring") AND "freshwater")

Source: Own elaboration (2025)

Finally, to investigate the most used approaches for understanding people's motivations and behavior, a traditional literature review was conducted during the same period. This type of review, characterized by a simpler and non-systematic approach, was selected because the objective was not to cover all existing motivational theories, but rather to obtain an overview of the main theories discussed in literature and to provide a basis for developing a theoretical framework in the context of water ecosystem valuation.

3.3 Data Analysis of Journal Publications

Initially, the relevance of the articles identified through the Scopus search was assessed by examining titles, keywords, and abstracts (or, in case of doubt, the introduction and conclusion) to decide on their inclusion in the analysis. For each article, the following aspects were considered: valuation method employed (e.g., choice experiment, contingent valuation, travel cost), study area (at the country level), year of publication, journal, sample size, objective, attributes used, and main

results. This provided an overview of how residents' WTP for water services is addressed in the existing literature. In addition, a complementary analysis was conducted to identify which attributes were employed and could therefore be considered in the future DCE to be developed in Risorgive del Bacchiglione. At this stage, studies deemed not relevant, studies in which the valuation method was not CE, duplicates, and inaccessible publications were excluded. This step served as input for defining potential attributes to be incorporated into the future DCE to be developed in the study site.

Moreover, after analyzing the motivational theories, those that best meet the needs of the Risorgive del Bacchiglione case study were selected.

3.4 Methodological Limitations

Several limitations must be acknowledged. First, the meta-analysis of 169 studies required the consideration of research conducted independently by different authors, at different times, with varied approaches to assessing causality and without a unified research design.

Second, the study relied exclusively on searches in the Scopus database, thereby reproducing its inherent limitations, which focuses on journal publications. While Scopus covers more than 39.000 journals, it includes only 1.628 book series, 514 conference proceedings, and no books published outside established series. This generates a bias in favor of scientific communities that publish mainly in journals, to the detriment of those whose output is concentrated in books or edited volumes.

Third, despite the growing body of research on the ecosystem service values of wetlands, data availability remains limited, particularly at the local level and in less developed countries.

Finally, the review relied primarily on publications and data available in English, which tends to underrepresent findings from regions where English is not the dominant working language.

4 Results

4.1 Analysis of the Selected Journal Publications

According to the guidelines established in Section 3.2, the literature search conducted in the Scopus database identified 169 published articles, reviewed and analyzed by the author together with a contributing scholar, which were reduced to 147 after removing duplicates:

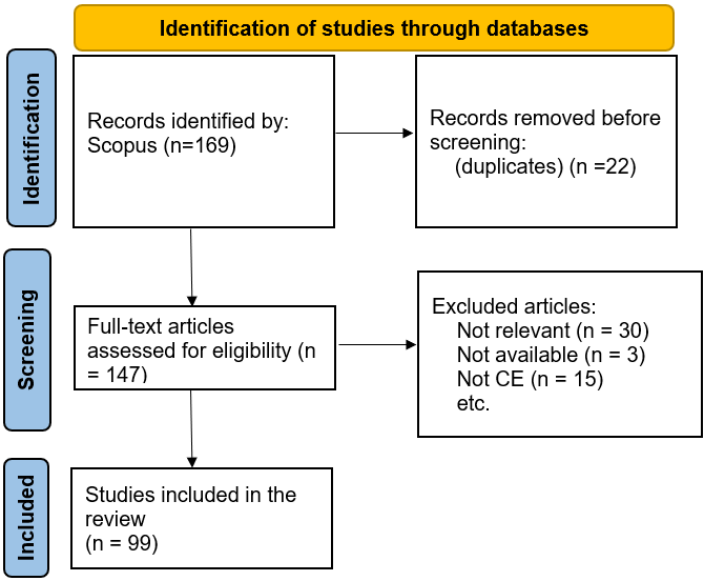
Table 8. Summary of search strings and number of articles reviewed

Search string	Articles identified through Scopus research	Articles selected without duplicates	Articles used in analysis
Search string (TITLE-ABS-KEY((‘discrete choice experiment’ OR ‘choice model’) AND (‘water ecosystem services’ OR ‘river catchments’ OR ‘water resource’ OR ‘wetland attributes’ OR ‘wetland values’ OR ‘water quality’) AND (‘willingness to pay’ OR ‘environmental valuation’ OR ‘stated preference’ OR ‘non-market valuation’ OR ‘indirect use values’ OR ‘non-use values’) AND (‘random parameter logit’ OR ‘multinomial logit model’ OR ‘welfare estimates’))	55	55	46
Search string (TITLE-ABS-KEY(""discrete choice" AND "spring ecosystem services" AND "attributes" AND "freshwater")	31	25	12
Search string (TITLE-ABS-KEY(""stated preference" AND "river ecosystem services" AND "freshwater" AND "attributes")	24	21	12
Search string (TITLE-ABS-KEY(""choice modelling" AND "spring ecosystem services" AND "attributes" AND "freshwater")	30	24	15
Search string (TITLE-ABS-KEY(""choice experiment" AND "ecosystem services" AND ("river" OR "spring") AND "freshwater")	29	22	14
Total	169	147	99

Source: Own elaboration (2025)

According to the guidelines established in Section 3.3, among these, 48 were excluded from the analysis due to lack of relevance (30 articles), not using the choice experiment method (15 articles), and unavailability (3 articles). Thus, 99 articles were selected for analysis. The sequence of this process is illustrated in Figure 14.

Figure 14. PRISMA diagram

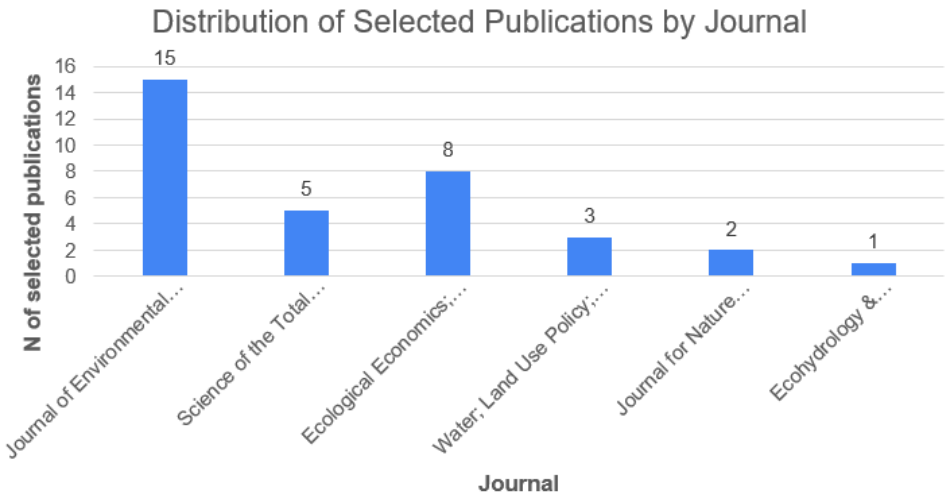


Source: Own elaboration (2025)

The following section presents the main results derived from this set of publications.

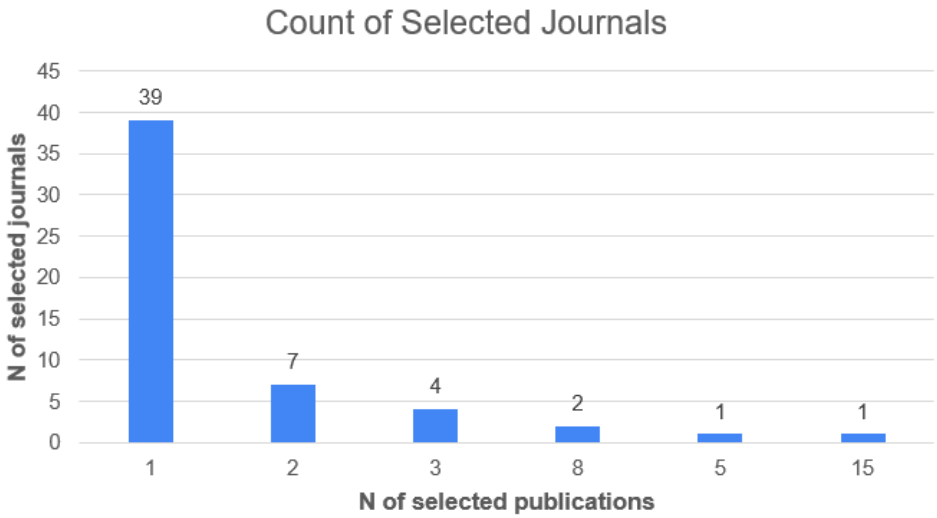
Of the 99 publications selected for analysis (Figures 15 and 16), 39 journals contributed with one paper each (*Ecohydrology & Hydrobiology; Habitat International; Marine and Freshwater Research; Knowledge and Management of Aquatic Ecosystems; Italian Review of Agricultural Economics; La Revue Française d'Economie et de Gestion; Ecohydrology; Ocean and Coastal Management; Environmental Economics and Policy Studies; Urban Forestry & Urban Greening; Heliyon; Journal of Investment and Management; Ciencia e Investigación Agraria; Applied Ecology and Environmental Research; Environmental Science & Policy; Hydrology and Earth System Sciences; Journal of Water Resources Planning and Management; Water Resources Research; Environmental and Resource Economics; Sustainability; Journal of Outdoor Recreation and Tourism; Biological Conservation; Journal of Environmental Economics and Policy; Regional Environmental Change; Journal of Arid Environments; Acta Ecologica Sinica; Marine Resource Economics; Tecnologia y Ciencias del Agua; Water Economics and Policy; Society and Natural Resources; PLoS One; Desalination and Water Treatment; International Business Management; Land; Environmental Resources Research; Landscape and Urban Planning; Marine Policy; Sustainable Development; Economic Research-Ekonomska Istrazivanja*). In addition, seven journals contributed with two papers (*Journal for Nature Conservation; Wetlands; Global Ecology and Conservation; Ecological Engineering; Environment and Development Economics; Journal of Cleaner Production*), four journals contributed with three papers (*Water; Land Use Policy; Ecological Indicators; Environmental Science and Pollution Research*), and two journals contributed with eight papers (*Ecological Economics; Ecosystem Services)*. The journal *Science of the Total Environment* contributed with five papers, and the journal with the highest number of selected articles was the *Journal of Environmental Management*, with 15 papers.

Figure 15. Distribution of selected studies by journal



Source: Own elaboration (2025)

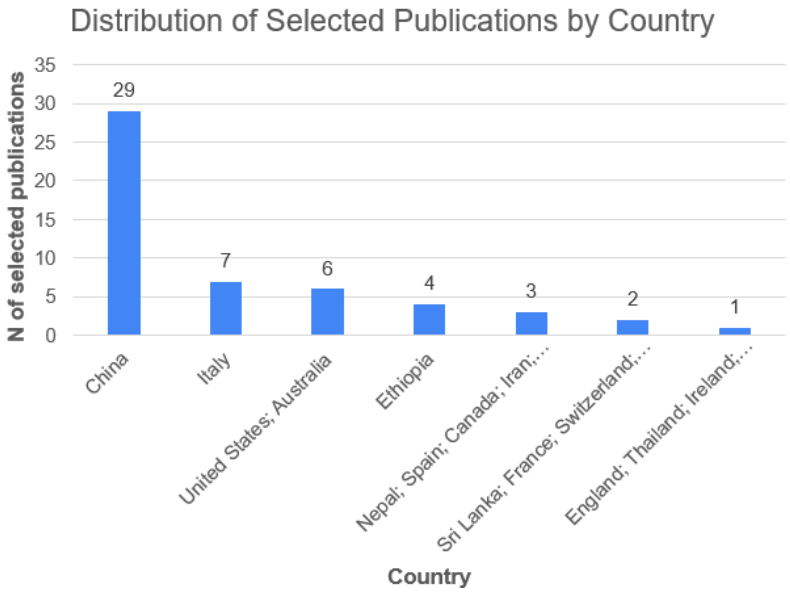
Figure 16. Frequency of journals by number of included articles



Source: Own elaboration (2025)

Regarding the study areas of the selected publications, 35 countries were identified (Figure 17). Most studies were conducted in China, accounting for 29 publications, followed by Italy with seven, United States and Australia with six each, and Ethiopia with four. Nepal, Spain, Canada, Iran, Greece, and India contributed with three publications each, while Sri Lanka, France, Switzerland, Vietnam, Austria, Chile, and Malaysia contributed with two each. Finally, England, Thailand, Ireland, Mongolia, United Kingdom, Brazil, Mexico, Poland, Tanzania, Romania, Uganda, Scotland, Indonesia, Wales, Finland, Sweden, and Belgium were represented by a single publication each. Although 99 publications were analyzed, the total number of country cases reached 101, as two studies covered more than one country.

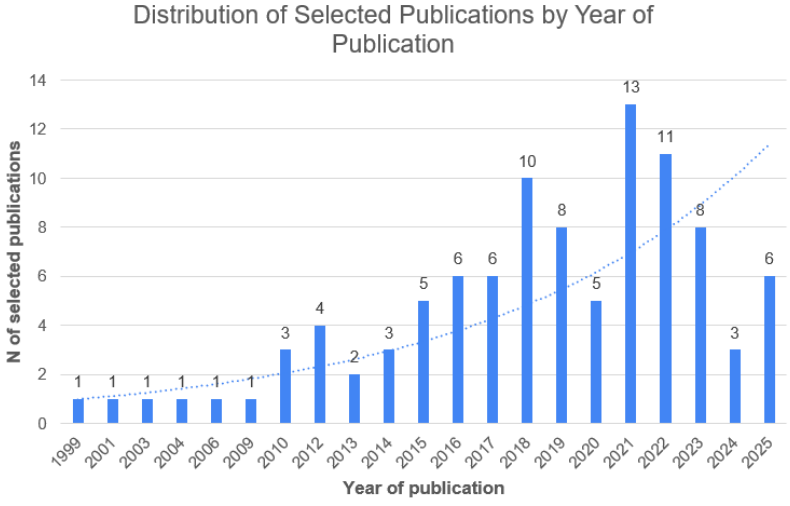
Figure 17. Distribution of selected publications by country



Source: Own elaboration (2025)

Regarding the year of publication, the selected articles span from 1999 to 2025 (Figure 18). A clear increase is observed in recent years, particularly from 2018 onwards, with 10 publications, reaching a peak in 2021 with 13 publications. Moreover, according to the trend line, the number of economic valuation studies of various wetland areas is expected to continue growing in the coming years, with an exponential growing. The year 2020 stands out with only five publications, which may be explained by the COVID-19 pandemic. Although a decline is noted after 2021, the number of articles published remains higher than in the years prior to 2018. This trend is consistent with the literature, which highlights a growing number of studies in the economic valuation of water ecosystem services. Alongside this growth, there has also been a notable diversification in both the valuation methodologies applied and the geographical areas assessed, a development likely facilitated by the increased accessibility of recent studies through electronic publications.

Figure 18. Distribution of selected publications by year of publication



Source: Own elaboration (2025)

4.2 Attributes Used in the Choice Experiment of the Selected Journal Publications

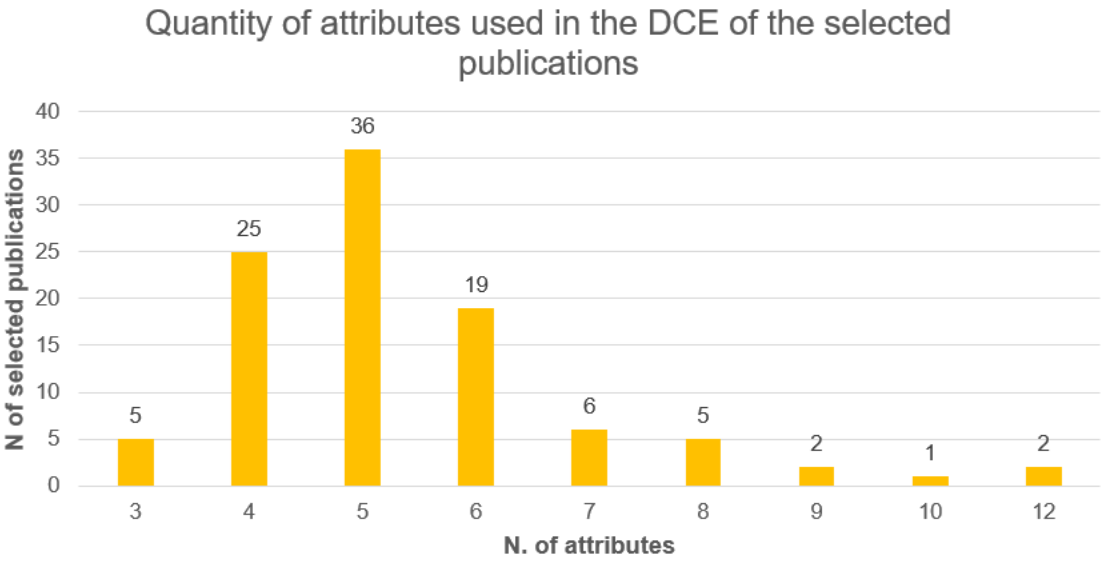
In the article by Kløjgaard *et al*. (2012), the authors emphasize that attributes in DCEs can be quantitative or qualitative, and are defined based on interviews, group discussions, literature reviews, and expert opinions. Since the context and objectives of each experiment can vary considerably, there is no single standard for attribute definition. Nevertheless, literature points to some common aspects that should guide this process.

In general, it is not feasible to include all relevant attributes in a DCE, but it is essential to select those that are most significant for most respondents. Otherwise, respondents may assume the existence of omitted attributes, which would compromise the validity of the results. Attribute selection should prioritize those that reflect the true motivations of participants, ensuring that they are willing to make trade-offs among them, in line with the theoretical framework of compensatory decision-making. Moreover, it is recommended to work with a limited number of attributes, which can be either generic or specific, and they should always be formulated in a clear and concise manner.

According to Kløjgaard *et al*. (2012), quantitative attributes are those that can be expressed in numerical and objective measures, such as waiting time to start treatment, number of weekly exercise hours, or risk of relapse expressed as a percentage. This type of attribute is more straightforward, as it presents clearly measurable values, facilitating comparison and the trade-offs made by respondents. Qualitative attributes, on the other hand, are related to descriptive characteristics, experiences, or perceptions that cannot be directly translated into numbers, but rather into categories or conditions. Examples include treatment modality (surgical or non-surgical), physical, mental, and social capacity after treatment, the presence of symptoms, or the choice of hospital. These attributes require greater care in wording and formulation to ensure that respondents understand them and that they adequately reflect the decision-making situations. In summary, while quantitative attributes provide objective and numerical measures, qualitative attributes involve more subjective and contextual aspects, both of which are essential to broadly capture individual preferences in a DCE.

Regarding the number of attributes used in the 99 articles analyzed, Figure 19 shows that the range varied from 3 to 12, totaling 543 attributes in total. The highest concentration was found in studies that employed 5 attributes (36 publications), followed by those with 4 attributes (25 publications) and 6 attributes (19 publications). More extreme cases were rare: only one article included 10 attributes, while two studies used 9 attributes and another two studies considered 12 attributes.

Figure 19. Quantity of attributes used in the DCE of the selected publications



Source: Own elaboration (2025)

The literature highlights the need to consider possible causal or interdependent relationships between attributes, as they can influence respondents' choices and compromise utility measures. To address this issue, different strategies can be adopted, such as excluding one of the related attributes, refining its definition during pre-tests, or even combining them, although this latter option may entail some loss of information (Blamey and Bennett, 2002).

Regarding the 543 attributes identified across the 99 publications analyzed, the attributes can be grouped into 12 main categories of similar aspects, as illustrated in Figure 20. Although the total number of attributes may seem large, every DCE survey is required to include a monetary attribute. Consequently, all the reviewed articles contained at least one monetary attribute; however, the total number of monetary attributes amounted to 101 instead of 99, since two studies applied two different surveys. When the monetary attribute is excluded, the attributes can be classified into 11 main categories, totaling 442.

Figure 20. Main attribute groups identified in the selected studies

Category	Attributes	Count of Attributes	%
Water-related Ecosystem Services	Aquaculture Infrastructure;	87	20%
Environmental and Water Quality	Wetland environmental qua	72	16%
Landscape & Recreation and Tourism	Landscape aesthetics; Nur	71	16%
Biodiversity & Wildlife & Natural Resources	Bird populations; Biodivers	68	15%
Agriculture & Land-use	Land area; Agricultural lanc	26	6%
Risks & Damages	Human health risk (lead); V	24	5%
Ecosystem Services and Functions	Regulation ecosystem serv	24	5%
Environmental Damage Reduction & Conservation Measures	Reduction of phumdi; The i	20	5%
Forest Area, Structure and Composition	Mangrove forest coverage;	18	4%
Social and Economic Impacts	Community income increas	16	4%
Others	Number of years until the b	16	4%
Total		**442**	100%
Costs & Payments (Monetary Attribute)	Financial cost; Cost; Costs	101	

Source: Own elaboration (2025)

As shown in Figure 20, the 11 categories are: Water-related Ecosystem Services; Landscape, Recreation and Tourism; Biodiversity, Wildlife and Natural Resources; Environmental and Water Quality; Risks and Damages; Agriculture and Land Use; Ecosystem Services and Functions; Environmental Damage Reduction and Conservation Measures; Forest Area, Structure and Composition; Social and Economic Impacts; and Others.

The analysis of the 442 attributes identified across the 99 publications reveals a strong concentration in certain categories, highlighting the main aspects that have been prioritized in DCE applications related to ecosystem services.

Costs & Payments (Monetary Attribute), with 101 attributes, reflects the obligatory inclusion of a monetary attribute in every DCE survey.

Water-related Ecosystem Services is the category with the largest number of attributes, accounting for 87 attributes, which represents 20% of the total, emphasizing the importance of water provision, regulation, and related services in valuation studies.

Other prominent categories include *Landscape, Recreation and Tourism* (71 attributes) and *Biodiversity, Wildlife and Natural Resources* (68 attributes), which together underscore the relevance of cultural and ecological dimensions in respondents' preferences. Environmental concerns are also highly represented: *Environmental and Water Quality* (72 attributes) and *Risks & Damages* (24 attributes) capture issues related to pollution, health risks, and environmental hazards.

Attributes related to *Agriculture & Land Use* (26), *Ecosystem Services and Functions* (24), and *Environmental Damage Reduction & Conservation Measures* (20) reflect the connection between land management, ecological functioning, and conservation policies. Forest-specific aspects are considered in *Forest Area, Structure and Composition* (18 attributes), while socioeconomic aspects are represented by *Social and Economic Impacts* (16 attributes). Finally, the categories *Others* (16) consolidate residual or case-specific attributes.

Overall, the distribution shows that DCE studies on ecosystem services tend to balance ecological, social, and economic dimensions, with particular emphasis on water-related services, landscape, biodiversity, and monetary valuation.

Considering the first four groups of attributes (Water-related Ecosystem Services; Landscape & Recreation and Tourism; Environmental and Water Quality; and Biodiversity & Wildlife & Natural Resources) and excluding the mandatory monetary attribute present in all DCE surveys, these categories together account for 67% of the attributes identified in the selected publications, representing the majority.

The first group of attributes, which accounts for the largest share, is Water-related Ecosystem Services. This group was addressed in a variety of ways across the reviewed studies, reflecting the broad relevance of water services. The main examples include (Table 9):

Table 9. Classification of Water-related Ecosystem Services and Examples

Water-related Ecosystem Services	Examples of Attributes Found
Fishery resources and aquatic biodiversity	native fish abundance, number of fish species, fish catch, dolphin observability, fisheries management, and presence of specific species (e.g., Cyprinids, crayfish).
Water quantity and availability	per capita water volume, water availability, water storage, drinking water provision, and minimum instream flow.
Wetlands and riparian areas	wetland biodiversity, wetland area, floodplain area in good environmental condition, mangrove conservation, and healthy aquatic plants.
Water regulation and risk management	flood regulation, flood reduction, flood return period, sediment control, riverbank protection, and water regulation services.
Infrastructure and restoration	aquaculture, pisciculture, wetland popular science education, river-restoration activities, channel widening, restored river length, and riverbank protection.
Regional and site-specific aspects	ecological water distribution across upper, middle, and lower basins; the area of East Juyan Lake and Ejinaqi Oasis; and hydropower production (percentage change).

Source: Own elaboration (2025)

The second most frequent group is *Environmental and Water Quality*, which was addressed in different ways across the reviewed studies. The attributes can be organized into the following aspects (Table 10):

Table 10. Classification of Environmental and Water Quality and Examples

Environmental and Water Quality	Examples of Attributes Found
General water quality indicators	overall river water quality, drinking water quality, and level of water quality (high, medium, low).
Physical characteristics of water	water visibility, clarity, odor/smell, and swimming-ban length as a proxy for pollution.
Chemical and ecological conditions	amount of nitrates in groundwater, aquatic ecosystem health, blue–green algae bloom, healthy aquatic plants, and water purification capacity.
Wetlands and floodplains	wetland environmental quality, area of healthy wetlands, and floodplain area in good environmental condition.

Recreational and cultural aspects	quality of rivers for recreational fishing, and conditions of waterholes important to Indigenous communities.
Shoreline and landscape conditions	condition of riverbanks/shorelines and canal landscape condition.

Source: Own elaboration (2025)

The third most frequent group is *Landscape & Recreation and Tourism*, which was addressed in various ways across the reviewed studies. In summary, *Landscape & Recreation and Tourism* attributes capture aspects of landscape aesthetics, recreational infrastructure, tourist dynamics, and ecotourism experiences, highlighting both the physical environment and service-related dimensions. The attributes can be grouped into the following aspects (Table 11):

Table 11. Classification of Landscape & Recreation and Tourism and Examples

Landscape & Recreation and Tourism	Examples of Attributes Found
Landscape quality and aesthetics	landscape aesthetics, scenic beauty, natural scenery, proportion of natural landscape, and winter landscape appearance.
Recreational infrastructure and facilities	marked hiking and biking trails, recreational paths, picnic areas (tables, bins, signs), general recreational facilities, and eco-tourism/park conditions.
Recreational activities and services	number of recreational activities, swimming in the river, rafting period (months), barbecuing on the riverbank, guided boat trips, and access to recreational opportunities.
Tourism services and amenities	tourism information availability, tourist amenities, leisure and entertainment areas, and expected number of visitors.
Tourist dynamics	visitation rates, tourist group size, and tourist flow rate.
Ecotourism and nature-based experiences	visiting eco-tourism sites, walking along rivers, and nature-based recreational services.

Source: Own elaboration (2025)

The fourth most frequent group is *Biodiversity & Wildlife & Natural Resources*, which was addressed in multiple ways across the reviewed studies. The attributes range from general measures of diversity and habitat to specific species, vegetation, and ecological integrity indicators, reflecting the complexity of biodiversity in valuation studies. The attributes can be organized into the following aspects (Table 12):

Table 12. Classification of Biodiversity & Wildlife & Natural Resources and Examples

Biodiversity & Wildlife & Natural Resources	Examples of Attributes Found
Species diversity and richness	number of species (e.g., rare birds, migratory birds, threatened species), plant and animal diversity, endangered and protected species, and overall species richness.
Specific species and taxa	bird populations, waterbird breeding, Sarus crane populations, charismatic vs. non-charismatic species, and presence of macrophytes.
Habitats and ecosystems	wildlife habitat availability, flora and fauna conservation, surrounding vegetation, hedgerows, natural coastlines, and green corridors.
Vegetation characteristics	grass coverage area, xerophytes, surrounding vegetation, and area of healthy vegetation.

Ecological condition and integrity	ecological state, ecological security, environmental conditions, morphological features of ecosystems, and Index of Biotic Integrity.

Source: Own elaboration (2025)

In contrast, the categories with lower representativeness account for 33% of the attributes identified in the selected publications. These categories include: Agriculture & Land-use; Risks & Damages; Ecosystem Services and Functions; Environmental Damage Reduction & Conservation Measures; Forest Area, Structure and Composition; Social and Economic Impacts; and Others.

The fifth most frequent group is *Agriculture & Land-use*, which was addressed in different ways across the reviewed studies. The attributes can be organized into the following aspects (Table 13):

Table 13. Classification of Agriculture & Land-use and Examples

Agriculture & Land-use	Examples of Attributes Found
Agricultural area and land use	total land area, cultivated land production, cropping area, and agricultural land use.
Irrigation and water management	irrigated area (in hectares), farmland irrigation rate, irrigation water availability, and guarantee rate of farmland irrigation.
Agricultural practices and production quality	organic farming, traditional agriculture, and quality of agricultural products.
Environmental impacts	pesticide use and presence of wind farms in agricultural landscapes.

Source: Own elaboration (2025)

The sixth most frequent group is *Risks & Damages*, which was addressed in different ways across the reviewed studies. In summary, *Risks & Damages* attributes encompass environmental degradation, health and ecological risks, and human-induced pressures, highlighting how DCE studies capture the vulnerability of ecosystems to both natural and anthropogenic disturbances. The attributes can be organized into the following aspects (Table 14):

Table 14. Classification of Risks & Damages and Examples

Risks & Damages	Examples of Attributes Found
Human and wildlife health risks	human health risk from lead exposure and wildlife lead poisoning risk.
Soil degradation	soil erosion intensity, erosion area, and occurrence of landslides.
Sandstorms and desertification	sandstorm frequency and severity, and soil loss.
Greenhouse gas emissions and climate effects	greenhouse effect and GHG emissions.
Waste and pollution	amount of waste and its impact on local environments.
Ecological and land-use impacts	water loss, and presence or absence of woods and hedges as protective barriers.
Hunting-related pressures	hunting activity, and number of ducks hunted,

Source: Own elaboration (2025)

The seventh most frequent group is *Ecosystem Services and Functions*, which was addressed in different ways across the reviewed studies. The attributes can be organized into the following aspects (Table 15):

Table 15. Classification of Ecosystem Services and Functions and Examples

Ecosystem Services and Functions	Examples of Attributes Found
Main ecosystem service categories	provisioning, regulating, supporting, and cultural services.
Ecological and regulating functions	carbon sequestration, climate regulation, and slope stability.
Habitat and ecological integrity	habitat-related ecosystem services and general ecological functions.
Cultural and educational values	environmental education facilities, heritage-enhancement actions, folk culture preservation, and opportunities for research and education.
Social and cultural well-being	promotion of Tibetan culture and social harmony as expressions of cultural ecosystem services.

Source: Own elaboration (2025)

The eighth most frequent group is *Environmental Damage Reduction & Conservation Measures*, which was addressed in different ways across the reviewed studies. In summary, these attributes reflect preventive and restorative approaches, addressing soil and water conservation, emission control, ecosystem recovery, and sustainable resource management. The attributes can be organized into the following aspects (Table 16):

Table 16. Classification of Environmental Damage Reduction & Conservation Measures and Examples

Environmental Damage Reduction & Conservation Measures	Examples of Attributes Found
Soil and water protection	controlled soil and water erosion area, reduction of fertilizer use, and mitigation of hydrological risk.
Climate and emission control	reduction of greenhouse gas effects and other environmental damage mitigation measures.
Vegetation and ecosystem restoration	riparian vegetation protection programs, hydrological restoration scenarios, and species restoration actions.
Resource use management	reduction of groundwater allocation for agriculture, monitoring of extraction behaviors, and penalties for over-extraction.
Conservation and sustainable practices	biodiversity protection, forest litter and fuelwood collection control, and general conservation actions.
Cultural and landscape restoration	restoration of temples and managed realignment as part of broader environmental recovery efforts.

Source: Own elaboration (2025)

The ninth most frequent group is *Forest Area, Structure and Composition*, which was addressed in different ways across the reviewed studies. The attributes can be organized into the following aspects (Table 17):

Table 17. Classification of Forest Area, Structure and Composition and Examples

Forest Area, Structure and Composition	Examples of Attributes Found
Forest cover and extent	forest coverage area, forest cover ratio, and total forested area.

Forest type and composition	tree species diversity and presence of native forests.
Protected and managed forest areas	size of protected areas and designated environmental forest areas.
Riparian and mangrove ecosystems	mangrove forest coverage, mangrove area, and percentage of riverbank areas planted with trees.
Forest condition and health	area of healthy forest remnants and general forest condition indicators.

Source: Own elaboration (2025)

The tenth most frequent group is *Social and Economic Impacts*, which was addressed in different ways across the reviewed studies. In summary, these attributes highlight the interlinkages between environmental management, livelihoods, and local economies, addressing both the benefits and trade-offs associated with ecosystem-based interventions.The attributes can be organized into the following aspects (Table 18):

Table 18. Classification of Social and Economic Impacts and Examples

Social and Economic Impacts	Examples of Attributes Found
Income and economic benefits	community income increase from recreational activities and income from irrigated agriculture.
Employment and labor dynamics	number of people employed, labor contribution (hours per visit), and irrigation-related employment.
Economic losses and compensation	financial compensation for affected irrigators and economic losses from extreme weather events (e.g., storm level 10).
Agricultural transitions and social adaptation	re-training of farmers, paddy farmers' support schemes, and number of farmers leaving agricultural activities.
Broader socio-economic conditions	overall socio-economic attributes and availability of alternative employment opportunities.

Source: Own elaboration (2025)

The last group, *Others*, includes attributes that do not fit into the previous categories and represent a variety of distinct aspects. In summary, attributes classified as *Others* encompass miscellaneous temporal, spatial, administrative, and perceptual factors that complement the main analytical categories, capturing diverse elements considered in DCE studies.The main examples can be organized as follows (Table 19):

Table 19. Classification of Others and Examples

Others	Examples of Attributes Found
Temporal aspects	number of years until benefits are experienced, time to improvement, time delay, contract length, and duration.
Spatial and contextual factors	distance and location of the area to be managed.
Administrative and regulatory elements	program administration and regulatory mechanisms for implementing measures (e.g., groundwater reduction rules).
Economic and behavioral aspects	beach cleanup expenditure and different types of program uses.
Perceptual and experiential dimensions	sense of experience and residential neighborhood characteristics.

Source: Own elaboration (2025)

Thus, these represent the potential attributes to be used as inputs for discussions with stakeholders, considering that, after the identification of possible attributes for inclusion in a Discrete Choice Experiment in a future field study at the Risorgive del Bacchiglione, the final selection should be carried out through a qualitative process involving focus groups and interviews.

4.3 Motivational Theories for Water Ecosystem Valuation

Considering that site characteristics are not the only factors influencing the decision to value water-related ecosystem services, and that individual motivations also play a fundamental role in this process, the main theories used to study psychological and motivational aspects in the context of water valuation were compiled, with the aim of supporting a future study to be conducted in in Risorgive del Bacchiglione.

Individual motivations play a significant role in economic valuation, complementing the influence of site attributes. To better understand the factors shaping visitors' motivations and behavior, several theoretical frameworks have been developed over time. The literature review identified five key motivational theories, summarized in Table 20, whose application to the context of water ecosystem valuation is discussed in the following table.

Table 20. Motivational theories used in water ecosystem valuation

Theory	Description and applications in environmental research	Main references
Self-Determination Theory (SDT)	Explains motivation through satisfaction of three psychological needs, autonomy, competence, and relatedness, along a continuum from extrinsic to intrinsic motivation (Deci & Ryan 1985, 2000). SDT has been applied to pro-environmental behaviour, conservation volunteering, and energy use (Pelletier *et al*. 1998; Kasser 2009; Baxter & Pelletier 2020; Yang *et al*. 2022). It helps interpret whether restoration support arises from intrinsic enjoyment of nature, relational connection, or instrumental concern.	Deci & Ryan (1985, 2000); Pelletier *et al*. (1998); Kasser (2009); Baxter & Pelletier (2020); Yang *et al*. (2022)
Protection Motivation Theory (PMT)	Describes protective actions as outcomes of two cognitive appraisals: threat (perceived severity & vulnerability) and coping (response efficacy, self-efficacy, response cost) (Rogers 1975; Floyd *et al*. 2000). PMT is widely used in studies of flood-risk perception and climate adaptation (Grothmann & Reusswig 2006; Babcicky & Seebauer 2019). It helps assess whether perceived environmental threats and confidence in collective efficacy motivate restoration support.	Rogers (1975); Floyd *et al*. (2000); Grothmann & Reusswig (2006); Babcicky & Seebauer (2019)
Theory of Planned Behaviour (TPB)	Posits that attitude, subjective norm, and perceived behavioural control (PBC) determine intention, which predicts behaviour (Ajzen 1991). Extensively applied to WTP and participation in environmental programmes (Kaiser & Gutscher 2003; Fishbein & Ajzen 2010). Empirical examples include ecotourism intention (Lee & Jan 2018), WTP for suburban parks (López-Mosquera & Sánchez 2012), and green product purchasing.	Ajzen (1991); Kaiser & Gutscher (2003); Fishbein & Ajzen (2010); López-Mosquera & Sánchez (2012); Lee & Jan (2018)

Value–Belief–Norm (VBN) Theory	Links personal values (egoistic, altruistic, biospheric) → ecological beliefs (NEP) → moral norms → behaviour (Stern *et al*. 1999). VBN explains value-based moral obligation toward environmental protection. Applied to energy policy acceptance (Steg *et al*. 2005), park conservation (López-Mosquera & Sánchez 2012), and marine protected areas (Kenter *et al*. 2016). Combined TPB–VBN models improve prediction of environmental support (Lee & Jan 2018).	Stern *et al*. (1999); Steg *et al*. (2005); López-Mosquera & Sánchez (2012); Kenter *et al*. (2016); Lee & Jan (2018)
Impure Altruism / Warm-Glow Theory	Suggests individuals gain utility from both the public good and the moral satisfaction ("warm glow") of contributing (Andreoni 1990, 1993). Explains positive WTP beyond personal benefit (Kahneman & Knetsch 1992). Environmental valuation studies consistently detect warm-glow motives in payments for restoration and conservation (Nunes *et al*. 2003; Czajkowski *et al*. 2017).	Andreoni (1990, 1993); Kahneman & Knetsch (1992); Nunes *et al*. (2003); Czajkowski *et al*. (2017)

Source: Own elaboration (2025)

Moreover, compiling the main motivational theories found in the literature on water ecosystems contributes to the development of the future survey in Risorgive del Bacchiglione, by enabling the incorporation of socio-psychological aspects into its structure.

5 Questionnaire Development and Discussion

5.1 Choice Experimental Design for Future Research

Building upon the insights from the literature, future research should focus on developing a structured questionnaire tailored to the valuation of spring-fed and groundwater-dependent wetlands.

The survey should begin with a brief introduction and an information section outlining the project objectives, contact details, the importance of honest responses, and the principles of confidentiality and consent. Participants are informed that there are no “right” or “wrong” answers, as all responses are valid as long as they reflect their personal opinions and help characterize the sample. Participation would be entirely voluntary, and responses would be anonymous, being reported only in aggregate form and used exclusively for research purposes.

The Choice Experiment (CE) is a non-market valuation method that identifies respondents’ preferences through hypothetical scenarios combining different attributes and their levels in a choice set. The questionnaire must be designed to investigate public knowledge, attitudes, and preferences regarding the ecosystem services provided by the Bacchiglione spring-fed wetlands, aiming to understand how people perceive and value these ecosystems.

The first step in designing a CE is to identify the ecological attributes and their corresponding levels to be evaluated. According to the MA, ecosystems perform four fundamental functions: supporting, provisioning, regulating, and cultural. Based on a literature review and the specific characteristics of the Risorgive del Bacchiglione, and considering these four categories of MA ecosystem services, future studies must define one payment attribute and an appropriate number of ecological attributes with different levels, that is, balance the number of relevant attributes with the overall complexity of the experiment.

The questionnaire should be structured into seven main sections: (1) Knowledge and Perception of Spring Ecosystems; (2) Environmental Awareness and Challenges; (3) Perceived Benefits of Restoration and Management Preferences; (4) Choice Experiment; (5) Reflection on the Choice Experiment; (6) Affirmations Based on Motivational Theories; (7) Socio-demographic Characteristics. Each section was designed to explore public knowledge, perceptions, and attitudes toward he Bacchiglione springs and the ecosystem services they provide.

The Knowledge and Perception of Spring Ecosystems section assesses respondents’ familiarity with the concept of risorgive, their awareness of the Bacchiglione springs, and the benefits they associate with spring-fed wetlands, such as clean water, biodiversity habitat, flood protection, recreation, and education. It also examines geographical proximity, and the activities people engage in or wish to pursue (e.g., walking, cycling, birdwatching, educational visits), revealing the degree of connection individuals feel with these ecosystems.

The Environmental Awareness and Challenges section focuses on public perceptions of the current ecological condition of the springs and the main threats they face, including agricultural pollution, excessive water extraction, urbanization, biodiversity loss, invasive species, and climate change. A brief explanatory text provides scientific background on existing pressures, ensuring that responses are based on an informed understanding. This section aims to measure public concern and identify which environmental problems are perceived as most urgent.

In the Perceived Benefits of Restoration and Management Preferences section, participants evaluate potential restoration and renaturalization projects in the Bacchiglione area, considering their effects on water quality, biodiversity, flood control, and recreation. Respondents are also asked about their support for conservation measures that may restrict access to certain areas and to indicate who they believe should be primarily responsible for managing the springs, local or regional governments, NGOs, or citizens. The responses will help determine levels of public support for different management strategies.

The Choice Experiment section will present hypothetical management scenarios with varying costs, environmental outcomes, and access rules, allowing researchers to assess how participants value different ecosystem services and trade-offs.

The Reflection on the Choice Experiment invites participants to evaluate their experience with the choice tasks, including how easy or difficult they were, their confidence in their answers, and the factors influencing their decisions. This reflection helps interpret not only what choices were made, but why.

The Affirmations Based on Motivational Theories section includes behavioural and motivational statements aimed at exploring the psychological and value-based dimensions of environmental behaviour. Respondents will rate their level of agreement on a scale, providing insights into the motivational drivers behind conservation attitudes.

Finally, to avoid response bias in the questionnaire, as suggested by Malhotra (2004), the Socio-demographic Characteristics section characterizes respondents based on socioeconomic variables such as residence (province and municipality), age, gender, education level, occupation, and income. These data are crucial to contextualise responses and analyse how demographic factors influence awareness, attitudes, and preferences regarding the Bacchiglione springs, supporting the development of targeted communication strategies and policy recommendations.

Entrance and parking fees in natural areas serve multiple purposes: beyond generating additional revenue and reducing dependence on government funding, they enable improvements in park management, enhancement of visitor facilities, protection of the environment from overcrowding, and the promotion of social objectives such as environmental education (Pellizzari, 2022).

However, there is still a lack of information to guide the proper determination of fee levels, which creates challenges for managers responsible for their implementation. Given the scarcity of comparative studies on this topic, there is a clear need for benchmarks and reference data that could support the establishment of fair and effective pricing strategies across different protected area contexts (Pellizzari, 2022).

A possible explanation for the lack of data lies in the nature of recreational use of forests, which are traditionally considered public goods. This characteristic is particularly evident in Northern Europe, where all natural areas, whether public or private, are freely accessible. Finland explicitly states that no entrance fees are charged in its national parks, and the same applies to Sweden (Pellizzari, 2022).

However, as one moves further south in Europe, discussions on the potential introduction of fees have gained momentum. In France, the topic was historically regarded as “completely taboo”, but since 2020, debates over implementing entrance fees in national parks have intensified, driven mainly by the decline in public funding for conservation (Pellizzari, 2022).

Another relevant aspect concerns the relationship between transparency and public acceptance. Studies show that when visitors clearly understand that the fees collected are allocated to environmental conservation and the improvement of park infrastructure, there is a higher level of acceptance of the “user-pays” principle, as well as an increase in their WTP (Pellizzari, 2022).

In the case of the Risorgive del Bacchiglione, it might be considered to use an entrance fee for the payment attribute, given that public access to the natural area (Area A) is currently free (risorgivedelbacchiglione.it). This fee would be allocated to the maintenance, conservation, and environmental improvement of the site, contributing to its sustainability and effective management.

Unlike taxes, which are compulsory contributions used to fund public expenditures, an entrance fee is charged for the specific use of a service, in this case, access to the park. Because it is uniform for all visitors, it raises issues of equity and efficiency: it may be considered fair in terms of equality, as it treats everyone the same, but also unfair, as it places a proportionally greater burden on lower-income individuals. From an efficiency perspective, it is a simple mechanism to administer, although it may be ineffective if it does not account for differences in payment capacity. In some contexts, such a fee can resemble a tax, and progressive pricing systems, which adjust the fee according to income, can better balance fairness and efficiency.

Moreover, an appropriate sampling strategy must consider the target population of the study. The sample size should be large enough to ensure statistically reliable estimates, but it is inevitably constrained by budgetary and operational limitations. Determining the optimal sample size in nonlinear choice models is challenging, as it depends on the true values of the unknown parameters to be estimated (Hoyos, 2010).

According to Pearmain *et al*. (1990), samples with more than 100 respondents are sufficient to provide a robust basis for preference modeling in DCE designs. Previous studies (Birol *et al*., 2006; Carlsson *et al*., 2003; Morrison *et al*., 1999; Nthambi *et al*., 2021) employed random sampling techniques, resulting in sample sizes ranging from 300 to 1200 respondents. Bunyangha *et al*. (2022) employed a multistage sampling approach to select 400 respondents for the interviews.

5.2 Theoretical Implications

a. Advancing the Valuation of Ecosystem Services

From a theoretical perspective, this research contributes to bridging ecological economics and environmental psychology by emphasizing the role of motivations in shaping individuals’ preferences and WTP for wetland conservation. It reinforces the notion that valuation should go beyond monetary estimates, encompassing behavioral and value-based dimensions that influence decision-making in nature conservation.

This study confirms that water-related ecosystem services are among the most valued environmental goods, especially concerning water quality, biodiversity, and flood regulation. These results reinforce the conceptual basis of the Total Economic Value (TEV) framework, which captures both use and non-use benefits (Costanza *et al*., 2017).

However, this review also highlights an ongoing theoretical shift from purely economic valuation toward integrated socio-ecological approaches, aligning with the post-2020 Ecosystem Services Science Agenda (Reyers & Selig, 2020;

Saarikoski *et al*., 2016). DCEs are increasingly used not only to elicit WTP but also to examine trade-offs between ecological, cultural, and social values. This evolution reflects a broader trend from instrumental valuation (focusing on utility) to relational valuation (focusing on meaning and identity) (Chan *et al*., 2016).

b. The Role of Motivational Theories in Environmental Valuation

The integration of motivational frameworks, such as the Value–Belief–Norm (VBN) theory, Theory of Planned Behaviour (TPB), and Self-Determination Theory (SDT), significantly enhances the explanatory capacity of DCE models. Findings show that individuals driven by intrinsic motivations (biospheric concern, altruism, moral norms) exhibit higher and more stable WTP values compared to those motivated extrinsically (Gkargkavouzi *et al*., 2019a; Steg *et al*., 2014).

This outcome confirms that environmental decision-making cannot be fully understood through economic rationality alone. Instead, it reflects a combination of cognitive, affective, and moral processes, where values shape beliefs, and beliefs drive behavioural intentions (Stern P.C., 2000). Incorporating motivational constructs in DCEs helps address preference heterogeneity and captures latent psychological drivers that conventional economic models overlook.

The hybrid choice modelling framework represents methodological advancement by integrating attitudinal variables directly into utility functions, creating a bridge between environmental psychology and economics (Kenter *et al*., 2019; Mariel *et al*., 2021).

c. Implications for Spring-Fed Wetland Valuation

For the Risorgive del Bacchiglione, these theoretical insights are particularly relevant. Spring-fed wetlands combine tangible ecological services (e.g., groundwater filtration) with intangible cultural and aesthetic values tied to local identity. Recognising these multiple dimensions requires multidisciplinary valuation frameworks capable of linking ecological indicators, public preferences, and motivational orientations.

This research suggests that including cultural heritage and sense of place as attributes in future DCEs could better capture the emotional and symbolic relationships local communities have with the Risorgive. In doing so, economic valuation becomes not merely a tool for cost–benefit comparison but a mechanism for articulating shared values and fostering participatory conservation (Raymond *et al*., 2023).

5.3 Managerial Implications

a. Enhancing Local Wetland Management

From a managerial standpoint, the study highlights that conservation strategies in water-related ecosystems should consider both ecological attributes and visitors' motivations.

Understanding which attributes people value most, such as water quality, biodiversity, and recreational access, can guide resource allocation and restoration priorities. For the Risorgive del Bacchiglione, management strategies should emphasise ecological restoration combined with public accessibility, aligning with both conservation and community engagement goals.

DCE-derived valuation results can help local authorities and consortia (e.g., Consorzio di Bonifica Bacchiglione) quantify public support for different conservation interventions. For instance, higher WTP values for biodiversity protection can justify

investment in habitat restoration projects, while moderate WTP for recreation could support the development of eco-tourism infrastructure with minimal ecological impact.

b. Integrating Social and Motivational Dimensions

The findings also highlight the need for managers to consider psychological and social factors influencing conservational behaviours. Recognising intrinsic motivations such as environmental stewardship and moral responsibility can improve communication strategies and foster collective ownership of conservation outcomes.

Environmental education programs, community involvement in restoration activities, and participatory planning processes can enhance intrinsic motivations, shifting public perception from “passive beneficiaries” to active custodians of wetlands (Kenter *et al*., 2019).

Understanding motivational profiles can help managers of natural areas design more effective communication campaigns, environmental education programs, and funding mechanisms that reflect the values and expectations of different visitor segments.

Incorporating motivational factors in stakeholder engagement can also reduce resistance to payment schemes or conservation fees, improving policy uptake.

c. Application to Monitoring and Evaluation

Valuation results, particularly WTP estimates, can be incorporated into cost-effectiveness and benefit–cost analyses for wetland restoration. However, managers should interpret these figures not as absolute market values but as indicators of social preference intensity. Integrating motivational data, such as willingness to volunteer or participate in monitoring, can complement monetary estimates, offering a broader perspective on public engagement.

5.4 Policy Implications

a. Alignment with European Policy Frameworks

The results align closely with the objectives of the EU Water Framework Directive (2000/60/EC), the EU Biodiversity Strategy for 2030, and the Green Deal, which all emphasise ecosystem-based management and public participation.

By identifying attributes that citizens value most, DCE outcomes can inform environmental policy design, particularly regarding prioritisation of funding under the Common Agricultural Policy (CAP) and the LIFE Programme. For instance, WTP estimates for improved water quality and biodiversity can serve as input for Payment for Ecosystem Services (PES) schemes, guiding compensation levels for farmers who adopt sustainable practices.

b. National and Regional Implications

In Italy, integrating ecosystem service valuation into decision-making remains a growing priority (Masiero *et al*., 2021). The Risorgive del Bacchiglione can serve as a pilot case for implementing valuation-based management. Regional agencies such as ARPAV (Agenzia Regionale per la Prevenzione e Protezione Ambientale del Veneto) can utilise valuation results to justify the inclusion of Risorgive within regional protected areas or ecological corridors under the Rete Natura 2000 network.

Additionally, incorporating motivational insights into communication and outreach can support more effective environmental governance, especially when engaging local communities and landowners.

c. Toward Participatory and Relational Valuation

A significant policy implication of this study is the move toward participatory valuation, which integrates both economic and relational values. Policymakers are increasingly recognising that public acceptance of conservation measures depends not only on perceived benefits but also on value congruence, the alignment between policies and citizens' moral and cultural values (Chan *et al*., 2016).

Future DCE-based policy tools should thus embed deliberative elements, such as focus groups, workshops, or citizen assemblies, to co-create valuation scenarios that reflect shared meanings of wetlands, rather than treating preferences as isolated economic decisions.

5.5 Limitations

a. Methodological Constraints

This study presents some limitations that should be acknowledged. First, the heterogeneity of methods, contexts, and assumptions across the selected literature may have introduced variability in the results. Additionally, heterogeneity in survey design and context-specific attributes limits direct comparability across cases.

As a literature-based analysis, the study relies on secondary data and published results, which may introduce publication bias. Studies reporting insignificant or low WTP values are less likely to be published, potentially inflating mean valuation outcomes.

Moreover, the study acknowledges limitations inherent to survey-based research and valuation methods, including potential measurement errors, omitted variables, and sampling or instrumental biases that may affect model accuracy. Given the scarcity of recent studies in this field, further research is recommended to strengthen and validate the findings.

Future research could address these issues through meta-analysis and cross-country comparative DCEs, incorporating standardized attribute frameworks and experimental designs.

b. Integration of Motivational Constructs

Another limitation lies in the conceptual and methodological challenges of linking motivational theories with economic valuation. While this study proposes integrating psychological and motivational variables (such as environmental concern, altruism, and perceived efficacy) into DCEs, the practical implementation of this integration remains hypothetical. Future empirical research is needed to test these interactions using primary data.

Although recent advances in hybrid choice modelling are promising, the measurement of psychological constructs remains a challenge. Many studies use short, non-validated scales for complex motivations (e.g., altruism or biospheric values), which may compromise reliability. Future work should integrate validated psychometric instruments and structural equation modelling (SEM) to better capture causal relationships between values, attitudes, and preferences (Gkargkavouzi *et al*., 2019a).

c. Contextual Application to the Risorgive del Bacchiglione

Finally, the study focuses on a specific case study in Northern Italy, which may limit the generalizability of the findings to other geographical or socio-economic contexts.

The application of these insights to the Risorgive del Bacchiglione requires empirical testing. Conducting a pilot DCE survey in the region would provide real data on local preferences and motivations. Collaboration with local authorities, universities, and NGOs could facilitate participatory research, ensuring cultural relevance and policy applicability.

5.6 Recommendations for Future Research and Policy

The literature review confirmed that Stated Preference Methods (SPM), particularly the DCE, have gained widespread popularity in the valuation of environmental resources worldwide. The findings also informed the selection of attributes to be discussed in future DCE to be conducted in Risorgive del Bacchiglione.

Future research should empirically test the integration of motivational variables within DCE models, allowing for a better understanding of how psychological factors influence economic preferences. Conducting a primary survey at the Risorgive del Bacchiglione would provide valuable insights into local perceptions, behaviors, and willingness to pay, enhancing the robustness of future models.

It is also recommended that future studies explore cross-site comparisons between different types of wetlands and groundwater-dependent ecosystems in Italy and other regions, to identify common patterns and context-specific differences.

For policymakers, this study suggests that the joint incorporation of economic and motivational dimensions can lead to more comprehensive and socially informed conservation policies. Integrating ecosystem service valuation into regional planning and public awareness initiatives may foster a more inclusive approach to ecosystem management, aligning environmental objectives with community values and long-term sustainability goals.

6 Conclusion

Wetlands generate substantial ecological, social, and economic benefits, many of which are not captured by traditional market mechanisms. This study has explored the valuation of water-related ecosystem services (ES) with a particular focus on natural springs and spring-fed wetlands, ecosystems that play a fundamental role in maintaining hydrological stability, biodiversity, and human well-being.

Through a systematic literature review, the study synthesized current applications of DCEs in assessing public preferences and willingness to pay for the conservation and restoration of aquatic ecosystems. It further examined how motivational and psychological theories contribute to understanding the deeper drivers behind individuals' environmental choices.

By providing an overview of the main attributes used in DCEs addressing aspects related to water ecosystems, along with the most common theories on visitor motivations, it was possible to establish a reference framework specifically applicable to the Risorgive del Bacchiglione case study, in terms of site attributes, motivational factors, and their possible ways of conceptualization.

The results represent the first step of a future field study in Risorgive del Bacchiglione, aimed at assessing visitors' WTP for water conservation. Through the application of a DCE and the integration of motivational theories, the research demonstrates that visitors' willingness to pay for conservation is not determined solely by environmental attributes, but also by underlying psychological motivations and values.

The Risorgive del Bacchiglione represent a case where ecological relevance is combined with a strong cultural and symbolic connection, reinforcing the importance of participatory and value-based management strategies. Recognizing and incorporating motivational aspects into ecosystem valuation can enhance public support for conservation policies and improve the design of economic instruments for sustainable financing.

The findings reveal that DCEs have become a central method in ecosystem service valuation, offering a structured and flexible framework to quantify preferences for complex, non-market environmental goods. However, the review also highlights a significant imbalance in research focus: while rivers, lakes, and large wetland systems have been widely studied, smaller and groundwater-dependent ecosystems, such as natural springs and spring-fed wetlands, remain underrepresented in the economic valuation literature. These systems are ecologically unique and culturally significant, yet their benefits are often overlooked in both science and policy.

Moreover, the review underscores that most DCE studies remain rooted in economic rationality, emphasising trade-offs and monetary values but often neglecting the role of intrinsic motivations, moral norms, and identity-based connections to nature. Integrating behavioural and motivational theories, such as the Value-Belief-Norm and Self-Determination frameworks, can enhance the explanatory power of DCEs and yield richer insights into the social foundations of conservation preferences. This integration is essential for capturing not only what people are willing to pay, but also why they value and support environmental protection.

The study, therefore, calls for a more interdisciplinary approach to valuation, one that combines economic modelling with insights from psychology, sociology, and environmental ethics. It suggests the development of a context-specific

questionnaire that incorporates motivational scales alongside DCE design, enabling future research to link willingness to pay with pro-environmental values and behavioural intentions. Such an approach will allow policymakers to design conservation programs that resonate with both economic and ethical dimensions of human–nature relationships.

Ultimately, this research contributes to bridging the gap between ecological economics and behavioral sciences, offering insights that can inform decision-making processes aimed at preserving the integrity and multifunctionality of water-related ecosystems in Italy and in other similar contexts.

In conclusion, valuing water-related ecosystem services is not only an economic exercise but also a reflection of how societies perceive and prioritise the natural systems that sustain life. Recognising and quantifying these values, particularly for vulnerable and understudied ecosystems like spring-fed wetlands, will contribute to more inclusive, effective, and sustainable environmental governance. By integrating ecological science, economic reasoning, and human motivation, future valuation studies can better inform policy decisions that align environmental protection with societal well-being and intergenerational responsibility.

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Annex 1. Attributes found in the literature review

Authors	Att1	Att2	Att3	Att4	Att5	Att6	Att7	Att8	Att9	Att10	Att11	Att12
(Brock, Cromie, Newth & Wood, 2025)	Wetland environmental quality	Bird populations	Human health risk (lead)	Wildlife lead poisoning risk	Financial cost							
(Shore, Jaleta, Zeleke & Beyene, 2025)	Water quality	Native fish abundance	Landscape aesthetics	Cost								
(Tonin, 2025)	Regulation ecosystem services	Habitat ES	Cultural ES	Number of years until the benefits are experienced	Costs							
(Amirnejad, Hosseini & Azadi, 2025)	Regulating Services	Supporting Services	Provisioning Services	Cultural Services	Price (USD)							
(Bigirwa et al., 2025)	Water clarity	Water quality	Programs for protecting the riparian vegetation	Payment for each 20 L of water sourced from water resources in LRSB								

(Khan et al., 2025)	Guaranteed rate of irrigating farmland	Area of east Juyan lake	River water Quality level	Sand storm frequency	Leisure and entertainment conditions	Your family is willing to pay for this (every year)
(Udugama et al., 2024)	Environmental damage Reduction	Number of recreational activities	Community income increase from recreational values	User labour contribution (hours per visit)	Fund per adult per visit (LKR)	
(Lan et al., 2024)	Reducing groundwater allocation for agriculture	Regulation on how to implement the reduction of groundwater	Financial compensation for affected irrigators	Monitoring extraction behaviours	Penalty of over-extraction (based on the current water price, $2.66/kL)	Cost to you (one-off)
(Ureta et al., 2024)	Richness	Fish catch	Index of Biotic Integrity	Flow quality	Public Access	Residents' WTP for water quality improvement
(Nie et al., 2023)	Biodiversity	Mangrove forest coverage	Water quality	Landscape aesthetics	Payment (yuan/ year/person)	
(Liu, Li & Wang, 2023)	Contract length	Land area	Fertilizer reduction	Annual cash subsidy		
(Jamnia, Eidouzahi & Ghaleno, 2023)	Dolphin Observability	Natural landscape Watching	Facilities	Tourism Information Availability	Aquaculture Infrastructure	Payment for Ecosystem conservation trust fund (per entry per person)

(Sreeraman, Singh & Leichombam, 2023)	Fish catch	Recreational sites	Reduction of phumdi	Alternative employment opportunities	Payment							
(Oh & Gramig, 2023)	Aquatic life	Recreational activities	The size of the dead zone	Winter landscape appearance	Annual cost to household							
(Shi, Li & Zhao, 2023)	Water Quality	Water Quantity	Fish species	Eco-tourism, leisure, and entertainment area	Cost							
(Ge et al., 2023)	Forest coverage	Water visibility	Tourist flow rate	Amount of waste	Willingness to Pay							
(Ali et al., 2023)	Cultivated land yield	Quality of agricultural products	Oasis	Biodiversity	Greenhouse effect	landscape	Water quality	Farmland irrigation	Sandstorms	Area of east Juyan Lake	Leisure & entertainment	WTP
(Hernandez & da Costa, 2022)	Water quality (swimming-ban length)	River-restoration activities	Heritage-enhancement actions	Local tax								
(Xu & He, 2022)	Number of species of rare birds	Water visibility	Open park area	Wetland popular science education	Folk culture display and cultural inheritance	Tourists' WTP						
(Ruberto, Branca, Troiano &	Canal landscape condition	Animal biodiversity (canal)	Macrophyte presence (canal)	Recreation /fish (canal)	Water availability (fontanili)	Water quality	Payment					

Zucaro, 2022)									
(Amiri, Abildtrup, Garcia & Montagné-Huck, 2022)	Equipment (picnic tables, bins, information signs)	Marked hiking and biking trails	Water stream	Pesticides	Distance	Hedgerows and biodiversity	Agricultural land use	Tree species	Payment
(Bunyangha et al., 2022)	Wetland area protected for biodiversity	Education and research	Recreation and tourism	Pisciculture	Paddy farmers' schemes	Payment/donation			
(Khan et al., 2022)	Forest cover ratio	Level of water quality	Per capita water volume in the Wei River basin	Erosion area	Intensity of soil erosion	Natural landscape	Ecotourism and forest park conditions	Willing to pay	
(Bhat, Ahmad, Sofi & Kumar, 2022)	Wetland Biodiversity	Wetland Area	Research & Education	Cost					
(Arellanos, Guzman & García, 2022)	Water regulation	Sediment control	Water yield	Quality maintenance	Price				
(Nguyen & Simioni, 2022)	mangrove coverage	economic losses from storm level 10	Biodiversity	Payment					

(Khan et al., 2022)	Cultivated Land Production	Agricultural products quality	Size of Ejinaqi Oasis	Biodiversity	GHG	Landscape	Willingness to Pay
(Ureta et al., 2021)	water quality	wildlife habitat	water supply	Additional amount in the water bill per month			
(Nesha Dushani, Aanesen & Vondolia, 2021)	Guided boat trips	Beach cleanup expenditure	Tourist group size	wetland management fund			
(Dechasa, Senbeta & Guta, 2021)	Biodiversity	Water availability	Recreation services	Payment (environmental fund)			
(Ureta, Motallebi, Vassalos, Alhassan & Ureta, 2021)	Flood reduction	Water quality	Wildlife habitat	Recreation	Scenic beauty	stormwater fee	
(Wang, Li & Xu, 2021)	Fishery resources	Natural coastline	Water quality	Biodiversity	Payment		
(Jue, 2021)	Water purification	Biodiversity	Cultural services/ facilities	Information-related facilities	Fee		
(Getnet & Taw, 2021)	Water quality	Irrigation	Fishing	Recreation	Monetary cost		

(Cheng et al., 2021)	Proportion of natural landscape	Tourism index	Forest coverage	Water quality	Biodiversity	Water Volume	Willingness to pay	
(Nijhum et al., 2021)	Residential neighborhood	Green corridor	Wetlands	Recreation	Property tax			
(McGinnis, Atallah & Huang, 2021)	Water Quality	Water Regulation Services	Program Administration	Eligible Land	Additional Fee			
(Sinclair et al., 2021)	Water quality management	Fisheries management	Mangroves management	Payment				
(Khan, Lei, Muhammad & Zhao, 2021)	River Water Quality	Guarantee rate of farmland irrigation	Sand storm Weather	Area of east Juyan Sea	Recreational conditions (Area)	Annual payment (WTP)		
(Hua et al., 2021)	Biodiversity	Morphological features	Water Quality	Recreational Facilities	Cost			
(Strazzera et al., 2021)	Water ecosystem improvement	Water service improvement	Jobs lost as a result of the renaturation of river belts	Mitigation of hydrological risk	Improvement of recreational activities	Increase in local taxes		
(Khan, Hayat, Xia, Liu &	Forest cover ratio	Water quality level	Water quantity per capita	The mount of controlled soil and	Water loss and soil erosion intensity	Natural landscape	Visiting eco-tourism & park	WTP of household per

Zhao, 2020)				water erosion area				year (Yuan)				
(Xu et al., 2020)	Biodiversity	Environmental education facilities	Expected number of visitors	Entrance fee: for adult visitors only								
(Ali et al., 2020)	biodiversity (BD)	level of river water quality	reduction in greenhouse gases effect	Cost								
(Kunwar, Bohara & Thacher, 2020)	River water quality	Riverbank protection	The percentage of riverbank area planted with trees	The provision of a river monitoring and educational program	The regulatory mechanism	WTP						
(Khan et al., 2020)	Cultivated land yield	Quality of agricultural products	Oasis	Biodiversity	Greenhouse effect	Landscape	Water quality	Farmland irrigation	Sandstorms	Area of east Juyan Lake	Leisure & entertainment	WTP
(Castillo-Eguskitza, Hoyos, Onaindia & Czajkowski, 2019)	Organic farming	Biodiversity protection	Quality of water bodies	Native forest	Recreation	Cost						
(Feyisa & Bersisa, 2019)	Fish stock	Biodiversity	Water quality	Water availability	Annual payment (Birr)							
(Khan et al., 2019)	River water quality	Guarantee rate of farmland irrigation	Sand storm frequency	Area of east Juyan Lake	Biodiversity	Farmland Landscape	Leisure and entertain	Willing to pay				

							ment conditions	
(Khan & Zhao, 2019) .	Average water quantity (100 million m3)	Upper basin ecological water distributio n	Middle basin	Lower basin	Water quality of Hongya reservoir	Payment (RMB per year)		
(Khan et al., 2019)	River water quality level	Guarantee rate of farmland irrigation	Sandstorm frequency	Area of the east Juyan Lake	Leisure and entertainm ent conditions	Your family is willing to pay for this		
(Logar, Brouwer & Pailex, 2019)	Length of the river section that will be restored	Walking along the river	Swimming in the river	Barbecuin g on the river bank	High biodiversity	Medium Biodiversity	Price	
(Liski, Koetse & Metzger, 2019)	Managed realignme nt	Conservat ion actions	Recreational paths	monthly donation				
(Khan et al., 2019)	Forest cover ratio	Level of water quality	Amount of water per capita	Amount of controlled soil and water loss area	Erosion intensity	Natural landscape	Condition s for eco-tourism and parks	Costs per househ old each year
(Tan et al., 2018)	Mangrove area	Water quality	Biodiversity	Payment				

(Masiero et al., 2018)	Slope Stability (Adamello RP)	Flora Conservation (Adamello RP)	Fauna (Adamello RP)	Recreation (Adamello RP)	Landscape (Adamello RP)	Tax (Adamello RP)			
* (Masiero et al., 2018)	Carbon sequestration (Ticino RP)	Water quality (Ticino RP)	Biodiversity (Ticino RP)	Landscape (Ticino RP)	Recreation (Ticino RP)	Tax (Ticino RP)			
(Tan et al., 2018)	Mangrove area	Biodiversity	Water quality	Payment					
(Chen et al., 2018)	Improvement of morphological feature	Improvement of water quality	Biodiversity enhancement at riverbanks and in rivers	Recreational facilities	Cost (per household/ year)				
(Rai, Shyamsundar, Nepal & Bhatta, 2018)	Drinking Water Quality	Drinking Water Quantity	Soil Erosion and Landslides	Water fee					
(Khan et al., 2018)	Natural landscape	Tourist amenity	Sandstorm frequency	Forest coverage area	Grass coverage area	Xerophytes	Average water quantity	Water quality	Voluntary payment
(Seeteram, Engel & Mozumder, 2018)	Hydrological restoration scenario	Species restoration scenario	Socio-economic attributes	annual cost per household					
(Huenchuleo & de	Flora and fauna	Water quality	Water availability	Water storage	Payment				

Kartzow, 2018)										
(Khan et al., 2018)	Natural landscape	Tourist amenity	Sandstorm frequency	Forest coverage area	Grass coverage area	Xerophytes	Average water quantity	Water quality	Ecological security	Voluntary payment
(Aguilar et al., 2018)	Average water quantity	Water quality of Hongya reservoir	Voluntary payment							
(Chaikaew, Hodges & Grunwald, 2017)	Type of ecosystem services to be managed	Program administration	Location of area to be managed	Annual program cost (WTP)						
(McGinnis, Atallah & Huang, 2021)	Water Quality	Water Regulation Services	Program Administration	Eligible Land	Additional Fee					
(He, Dupras & Poder, 2017)	Biodiversity habitat	Flood protection	Water quality	Climate regulation	Annual cost					
(Mao et al., 2017)	water conservation	wetland area	biodiversity	natural landscape	WTP					
(Suziana, 2017)	Environmental conditions	Biodiversity	Recreational services	Flood control	Conservation cost per year and household					

(Ángel & Martínez-Paz, 2017)	Biodiversity	Water provisioning	Hunting	Fishing	Public access	Economic contribution to a conservation fund
(Kahn, Vásquez & de Rezende, 2017)	Restoration	Time	Cost			
(Huenchuleo, Barkmann & Marggraf, 2016)	River pollution risk	Number of threatened species	Local fishery yields	Payment		
(Vollmer, Ryffel, Djaja & Grêt-Regamey, 2016)	Channel widening	Protected forest	River banks	Maintenance fee		
(Brouwer et al., 2016)	Flood return period	Water quality	Price			
(Baniasadi, 2016)	Beauty and landscape	Plant and animal diversity	Soil and water resources conservation	Price		
(Yu, Xu & Wang, 2016)	Natural environment	Social harmony	Tibetan culture	Ecological resettlement cost		

(Andreopoulos et al., 2016)	Irrigated area (in ha)	Rafting period (in months)	Hydroelectricity production (% decrease)	Ecological state	Monthly payment for 10 years		
(Dias & Belcher, 2015)	Riparian zone	Wildlife habitat	Water quality	Price			
(Andreopoulos, Damigos, Comiti & Fischer, 2015)	Irrigated area (in hectares)	Rafting period (in months)	Hydroelectricity production (% decrease)	Ecological state	Monthly payment for 10 years		
(Rai, Shyamsundar, Nepal & Bhatta, 2015)	Drinking water	Irrigation Water	Forest litter collection	Fuelwood collection	Labor contribution	Cost	
(Andreopoulos et al., 2015)	Irrigated area (in ha)	Rafting period (in months)	Hydroelectricity production (% decrease)	Ecological state	Monthly payment for 10 years		
(Chaikumbung, Scarborough & Doucouliagos, 2015)	Cropping area	Fish catch	Healthy aquatic plants	Bird species	Temples restored	Time delay	Cost
(Kountouris, Godyn & Sauer, 2014)	Nitrate pollution	Time to improvement	Additional water charge				

(Doherty, Murphy, Hynes & Buckley, 2014)	Access to recreational activities	Aquatic ecosystem health	Water clarity and smell	Water body type targeted	Condition of banks or shoreline	Annual household income tax		
(Tentes & Damigos, 2014)	Uses	Ecosystem	Duration	Cost				
(Tempesta & Vecchiato, 2013)	Minimum instream flow	Amount of nitrates in groundwater	Presence of woods and hedges	Cost				
(Zander et al., 2013)	Area of foodplain is in good condition	Quality of river for recreational fishing	Quality of waterholes importatnt to aboriginal people	Income from irrigated agriculture	Cost of management plan (in AUS$)			
(Christie & Rayment, 2012)	Nature's gifts	Research and education	Climate regulation	Water regulation	Sense of experience	Charismatic species	Non-charismatic species	Cost per household (per year for 10 years)
(Bliem, Getzner & Rodiga-Laßnig, 2012)	Flood return period	Water quality	Cost price (Euro/household/year)					
(Kaffashi et al., 2012)	Natural Scenery	Water quality	Biodiversity	Ecological functions	Conservation value			

(García-Llorente et al., 2012)	Traditional agriculture	Ecotourism	Wind farms	protected area (PA) size	River quality	Cost
(Kosenia, 2010)	Water clarity (WAT)	Number of Cyprinids (ROA)	Bladder wrack (BLW)	Blue–green algae bloom (BGA)	Tax for household (PRICE)	
(Zander, Garnett & Straton, 2010)	Area of floodplain in good environmental condition	Quality of the river for recreational fishing	Conditions of waterholes important to Aboriginal people	Income from irrigated agriculture	Cost of management plan	
(Zander & Straton, 2010)	Area of floodplain in good environmental condition	Quality of the river for recreational fishing	Conditions of waterholes important to Aboriginal people	Income from irrigated agriculture	Costs of strategy (in AUS$) as a one-off payment	
(Do & Bennett, 2009)	Area of healthy vegetation	Number of Sarus cranes	Number of fish species	Number of local households affected	Cost options	
(Birol, Karousakis & Koundouri, 2006)	Biodiversity	Open water surface area	Research and education	Re-training of farmers	Payment	
(Othman, Bennett &	Environmental	Visitation rates	Number of migratory bird species	Number employed	contribution to a trust fund	

Blamey, 2004)	forest area						
(Carlsson, Frykblom & Liljenstolpe, 2003)	Surrounding vegetation	Biodiversity	Fish species	Fenced waterline	Crayfish	Walking facilities	Total cost
(Whitten & Bennett, 2001)	Area of healthy wetlands (USE)	Area of healthy remnants (USE)	Threatened species that will benefit (USE)	Number of ducks hunted (USE)	Cost to the respondent (USE)		
*(Whitten & Bennett, 2001)	Area of healthy wetlands (MRF)	Population of native water and woodland birds (MRF)	Population of native fish (MRF)	Number of farmers leaving (MRF)	Cost to the respondent (MRF)		
(Morrison, Bennett & Blamey, 1999)	irrigation related employment	wetlands area	Waterbirds breeding	Endangered and protected specie	water rates (one-off increase)		

Note: *Article repeated, as the attributes differ between studies, which were conducted independently