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September 2009

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Colophon

SIXTH FRAMEWORK PROGRAMME

This report is part of the EU funded project AquaMoney, Development and Testing of Practical Guidelines for the Assessment of Environmental and Resource Costs and Benefits in the WFD, Contract n° SSPI-022723.

General				
Deliverable	D23 Economic Valuation of Environmental and Resource Costs and Benefits in the Water Framework Directive: Technical Guidelines for Practitioners			
Deadline	Month 42			
Complete reference	Brouwer, R., Barton, D.N., Bateman, I.J., Brander, L., Georgiou, S., Martín-Ortega, J., Pulido-Velazquez, M., Schaafsma, M. and Wagtendonk, A. (2009). Economic Valuation of Environmental and Resource Costs and Benefits in the Water Framework Directive: Technical Guidelines for Practitioners. Institute for Environmental Studies, VU University Amsterdam, the Netherlands.			
Status	Author(s)		Date	
Approved / Released				
Reviewed by				
Pending for Review				
Third draft				
Second draft				
First draft for Comments				
Under Preparation				
Confidentiality				
Public				X
Restricted to other programme participants (including the Commission Service)				
Restricted to a group specified by the consortium (including the Advisory Board)				
Confidential, only for members of the consortium				
Accessibility				
Workspace				
Internet				X
Paper				X

Content

1. Introduction

1.1 *Objective and scope of the guidelines*

These Guidelines aim to provide guidance on key issues in economic valuation related to the implementation of the Water Framework Directive (WFD). Practical guidance on undertaking valuation studies that are readily applicable for water policy purposes is necessary given the economic value of water, the important role of water economics in underpinning the WFD, and the limited economic information and expertise in many river basin authorities. The Guidelines are targeted at expert practitioners and economic specialists who carry out valuation studies, rather than policy advisors or policy/decision-makers. The latter are targeted through the AquaMoney policy briefs (see www.aquamoney.org).

The Guidelines provide guidance on how to address specific key issues in economic valuation studies of water resources and how, given a variety of difficulties encountered, these values can be aggregated to determine a water resource's *Total Economic Value*. The Guidelines look at how these values can be aggregated from the level of an individual water body to river basin level, whilst accounting for changes in substitution possibilities regarding water use as one moves upstream or downstream or across separate water bodies in a river basin. Included here is the use of *value transfer* and the estimation of value functions that include factors relevant for such aggregation through Geographical Information Systems (GIS). The Guidelines are not about how to apply a particular valuation methodology (about which there are already many existing guidelines – see Annex 1 for an overview), but rather about how to address the specific problem of valuing water resources in the context of particularly Article 9 and 4 in the WFD.

The WFD has been criticized for ill-defined wording (Grimaud, 2004), in particular the legal interpretation of cost recovery in Article 9 (Unnerstall, 2007). These Guidelines aim to clarify, insofar possible, the economic meaning of the terms used in Article 9. However, because the Guidelines were written by economic experts, not lawyers or official representatives from European Member States implementing the WFD, they lack legal status to provide enforceable definitions and instructions regarding the appropriate cost accounting methodology. They advise practitioners on the economic interpretation of important terms and concepts, such as cost recovery, where necessary with references to relevant international peer reviewed literature.

Contrary to the Wateco guidelines (EC, 2002), most reviews of Article 9 in the WFD (e.g. Moran and Dann, 2008) and also these Guidelines, interpret cost recovery as 'full' cost recovery. This is also what is implicitly suggested by the fact that cost recovery includes 'environmental and resource costs'. Article 9 introduces environmental and resource costs in paragraph 1, stating that '*member states shall take account of the principle of recovery of the costs of water services, including environmental and resource costs,… , and in accordance in particular with the polluter pays principle*'. Cost recovery is a well-known economic pricing principle, implying that the total private and social production costs of a good or service are recovered (e.g. Renzetti and Kushner, 2004).

Environmental and resource costs are not defined in the WFD, but clearly relate in most cases to the social costs of activities that employ a natural resource such as water. Water has important social value, but often no price, or if water is priced, the price often does not reflect its real value to society. The issue here is one of translating well-established theory into practice and the availability of reliable data and information about the social costs and benefits of water resources.

The Guidelines focus on the economic valuation of this social, non-market value in order to support the WFD objective of achieving good ecological and chemical status of water bodies in river basin districts across European Member States. The WFD is expected to generate substantial non-market values (Bateman et al., 2006; Brouwer, 2008). Notwithstanding some remaining relevant key issues in the calculation of private costs such as the use of a standard accounting system to facilitate cross-country comparisons of financial cost recovery, this is where most methodological issues in water resource valuation are found, including environmental and resource costs. Environmental and resource costs will be defined in more detail after this Introduction in Chapter 2.

The WFD environmental objectives are defined in Article 4. If these objectives are not met, Article 11 requires that water managers identify a cost-effective program of measures to reach them. If the costs of these measures are considered disproportionate, paragraphs 3-7 in Article 4 provide water managers with a legal instrument to lower their environmental objectives or delay their achievement in time.

The relevant valuation questions for policy and decision-making here are:

1) What is the total economic value of the environmental and resource costs associated with water services in reaching the WFD objective of good chemical and ecological status? (arising from Article 9, paragraph 1)

2) What is the total economic value of reaching the WFD objective of good chemical and ecological status compared to the WFD implementation costs? (arising from Article 4, paragraph 3-7).

The economic valuation of the environmental and resource costs associated with specific water services and (not) reaching the WFD objective of good chemical and ecological status is a necessary precondition to be able to assess and (re)consider the level of cost recovery of water services, including environmental and resource costs, through existing or new water pricing instruments. The economic valuation of the benefits of reaching the WFD objective of good chemical and ecological status - or the benefits foregone if the environmental WFD objectives will not be achieved - is a necessary precondition to be able to assess and (re)consider to what extent the identified program of WFD measures is disproportionate from an economic point of view. Even though the WFD does not explicitly ask for such an assessment of economic benefits (it only states that the measures needed to reach good chemical and ecological status have to be cost-effective), insight in the return value of the foreseen large investment programs in water quality improvements across European water bodies helps to prioritize limited available budgets for WFD implementation across water bodies within and between river basins in the next decades.

In both cases it is important to point out that the economic valuation exercise may be - and in many cases will be - insufficient to conclude whether all costs are recovered, including environmental and resource costs, or whether the WFD objectives are disproportionately costly. There often exist important reasons, for instance related to equal access to water services for all, why not all costs are fully recovered and/or (cross-) subsidized. From the same token, if the total economic costs of WFD implementation exceed their total economic benefits, this too does not imply that the Directive is disproportionate. This requires consensus within and between European Member States and legislation that clearly defines disproportionate costs as costs exceeding benefits, which is not the case in the WFD or any other European Directive. Furthermore, the Guidelines are not about how to evaluate appropriate levels of cost recovery, whether water pricing policies provide adequate incentives for users to use water efficiently, whether existing water prices include environmental and resource costs, or the definition of disproportionate costs. The Guidelines advise practitioners how to assess the total economic value of the social (non-market) environmental and resource costs related to reaching the WFD objective of good chemical and ecological status to inform and support policy and decision-making regarding current or future levels of cost recovery including environmental and resource costs.

In both cases the same economic valuation methods may apply. Hence, the specific policy question in Article 4 or 9 in the WFD does not in that sense influence the choice for a specific valuation method or methodology. The choice for a valuation method and the key issues surrounding the economic valuation exercise are primarily driven by the environmental goods and services provided by water quality and water flow improvements under the WFD, i.e. the definition of the water goods and services provided under good chemical and ecological status. Here, one of the most difficult, but important issues faced by both economists and water managers is to provide a description and assessment of the links between the functions of aquatic ecosystems and the goods and services that are of benefit to humans and from which they derive economic value. Economists are unable to provide any guidance on how to value water services or estimate the economic benefits of water quality and flow improvements without at least some qualitative description and assessment related to current (baseline) conditions and good chemical and ecological status and the implications of reaching good chemical and ecological status for different types of water goods and services.

Much of the time and effort in the practical testing of the Guidelines went into the development of common, applicable water quality and environmental flow ladders across EU Member States, together with water scientists, translating the WFD objectives into goods and services and explaining what good water status means for different groups of water users and nonusers. A clear distinction has to be made between the definition of water services in the WFD (Article 2, paragraph 38) as *abstraction, impoundment, storage, treatment, and distribution of water, wastewater collection and treatment* *facilities which discharge in surface water*, and the broader definition of water *ecosystem* goods and services used in these Guidelines.

In the former case, a public or private agent provides a (partly) paid or unpaid service to households, a public institution or an economic activity. Based on available market transactions, the financial costs of service provision (labour and capital costs) can be compared with the financial revenues (market prices, taxes and charges) paid by those who use the service involved (e.g. household water supply and wastewater collection and treatment). This then allows the assessment of the level of financial cost recovery, for example for drinking water supply or wastewater collection and treatment.
In the latter case, we refer to the aquatic ecosystem providing a variety of goods and services to society at large, including households, public institutions and economic activities, based on the ecosystem paradigm underlying contemporary science, including environmental economics. The Guidelines address the economic valuation of the intangible, non-market goods and services provided by the water system under good chemical and ecological status as defined by the WFD and affected by the water services defined in paragraph 38 in Article 2. Their economic valuation starts from the welfare implications of the change in water service provision for different water users and nonusers, not merely the financial costs of their provision through the water services defined in Article 2.

The economic valuation of the environmental and resource costs related to the goods and services provided by the WFD includes more than a financial analysis of the costs of water service provision as defined in paragraph 38 in Article 2.

Related to this is the ‘no deterioration rule’ in the WFD (Article 4, paragraph 4.4). This means that water status is not allowed to deteriorate, only to improve from some current (baseline) condition to the WFD objective of reaching good chemical and ecological water status. The environmental and resource costs are in that case equal to the benefits foregone of not reaching the WFD objectives.

The Guidelines cover an extensive list of water *ecosystem* goods and services, including market (such as drinking water, commercial fishing, reeds, industrial process water, etc) and non-market goods and services (such as recreation, biodiversity, natural sink, etc). These are valued within a standard economic value framework that measures their *Total Economic Value*. The *Total Economic Value* framework is linked to the specific water functions, goods and services mentioned above. Following on from this, the Guidelines consider the selection of appropriate economic valuation methods and key methodological issues in their implementation in the particular context of the WFD, adding to the existing amount of guidelines listed in Annex 1.

1.2 *How to use the guidelines*

The Guidelines have been designed to help expert practitioners and economic specialists involved in implementing economic valuation studies understand and take account of the key methodological and practical issues related to specific implementation of the WFD. To be able to use the Guidelines, a basic understanding and experience of applied environmental economics and economic valuation, including statistics and econometrics, is necessary. Advanced skills should not be required. The aim is to provide a practical, but theoretically sound handbook to guide valuation implementation. The Guidelines can be used alongside other economic valuation manuals such as those listed in Annex 1.

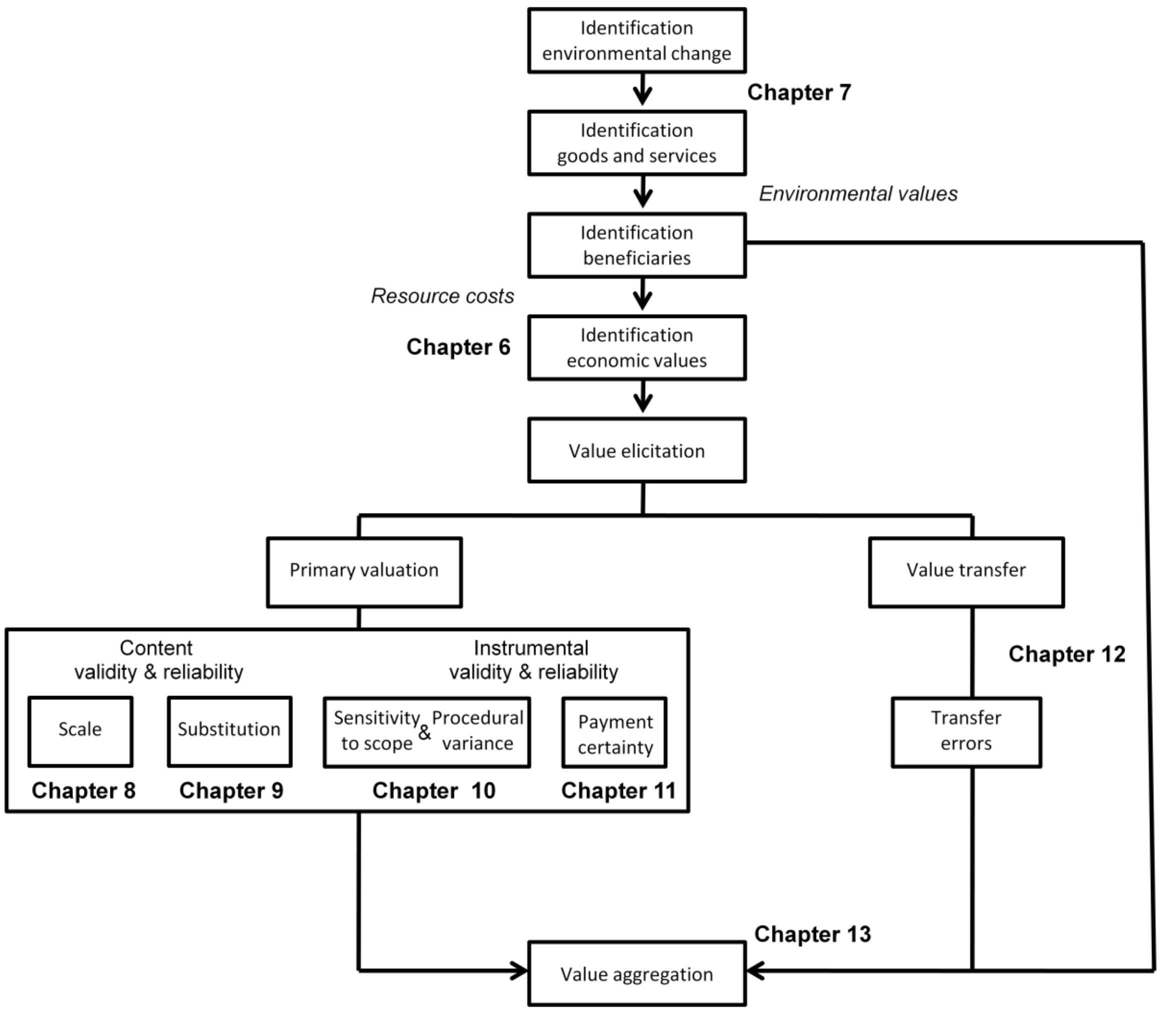
Producing conceptually valid and empirically accurate estimates of environmental and resource costs and benefits is a demanding task, requiring more time, resources and technical skills than is generally appreciated. The Guidelines cannot begin to reflect all of the issues that can possibly be brought to bear on assessing environmental and resource costs and benefits in the context of the WFD. Nevertheless, the intended scope is to detail the key factors and requirements. Compromises will inevitably have to be made, taking account of the limitations and restrictions imposed by time and resources. Because of the need to cater for the different levels of knowledge and experience in applying valuation studies, specialists may find that they can skip sections which detail their own areas of expertise. Nevertheless, all readers applying the Guidelines should be familiar with the conceptual basis of the water valuation framework as it provides the logic and justification for the practical valuation procedures outlined later.

The Guidelines are organised as follows. In **Chapter 2**, to set the stage, the reader is introduced to the concept of water as an economic good and the water valuation framework used to assess environmental and resource impacts. The analytical framework for water valuation assessment is outlined, focusing on the concept of *Total Economic Value* and the *Functional Approach* to valuation. Included here is a brief discussion of why the framework is relevant to the valuation requirements of WFD implementation, and the definition of environmental and resource costs. The link between the functional ecosystem approach and TEV is further detailed in **Chapter 3**, including a list of aquatic ecosystem goods and services. This is followed in **Chapter 4** by an overview of the various valuation methods and techniques used in water resource valuation to assess *Total Economic Value.* Recommendations over the type of valuation method chosen to assess the different water uses and services are given in this chapter.

The Guidelines then look at the application of these valuation methods and techniques to value water resources in the specific context of the WFD and the practical case studies carried out in AquaMoney. The focus here is on looking at those key issues of importance to the specific needs and circumstances related to WFD implementation. How these issues are incorporated into the design and implementation of a valuation study is outlined, accompanied by a discussion of validity and reliability issues. The focus here is on taking account of the issues and complications such that the values estimated are valid and suitable for use in WFD policy implementation. A separate section is also included here focusing on the role of hydro-economic modelling in assessing the shadow price of water in the context of water scarcity and resource opportunity costs.

The steps taken in an economic valuation study and the key issues addressed in these Guidelines are presented in Figure 1.1.

Figure 1.1: General steps in economic valuation and key issues addressed in different chapters throughout the Guidelines



The first steps in Figure 1.1 are fairly straightforward and can be found back in several general valuation handbooks and manuals (see Annex 1), except for the identification of the goods and services generated by the WFD amenable to robust economic valuation.

Chapter 5 starts with an overview of existing non-market values for different water services based on meta-analysis. More than 150 stated preference studies were collected worldwide and screened for the values they produce. An attempt is made to explain the variation in the value estimates based on differences in (i) the characteristics of the water services valued in existing studies, (ii) the characteristics of the beneficiaries of these water services and (iii) the characteristics of the methodological approaches followed in the existing studies. This is followed by an illustration of the use of a meta-value function for the purpose of water value mapping based on existing pan-European spatial data.

Chapter 6 addresses the important issue of water scarcity, which is especially a problem in South European Member States. We demonstrate the role of hydro-economic in assessing shadow prices for non-priced water resources based on water resource scarcity rents and the use of non-market valuation to assess the economic use and non-use values of water resource conservation for domestic and environmental purposes.

Chapter 7 starts with the translation of the WFD objectives into goods and services through the development of a WFD-specific water quality ladder in order to be able to assess the economic values underlying the physical environmental changes due to WFD implementation and facilitate their transfer across different river basin contexts. Chapter 7 discusses the use and usefulness of water quality ladders and presents an example of such a ladder developed in AquaMoney. Special attention will be paid to public perception and understanding of the water quality ladder.

When aiming to estimate the monetary economic value of an environmental change for policy appraisal purposes, two different approaches exist. Either an original (primary) valuation study can be commissioned by the responsible authority or one relies upon the values elicited in existing valuation studies addressing a similar policy problem in the past (value transfer). The latter approach is very attractive as it is much cheaper and quicker than conducting original valuation research. However, its use is not without difficulty, and comes at a cost in terms of the error involved when using existing values in policy and project appraisal. Applying existing values will never be as precise and accurate as carrying out original valuation results. The trade-off here is one between the extra cost of conducting a primary valuation study and the cost of getting it wrong when using existing values in policy and project appraisal and corresponding decision-making[1]. The magnitude of the errors involved are investigated in **Chapter 12**. The aforementioned water quality ladder was tested in five different European Member States using an identical valuation design to facilitate value transfer. Chapter 12 focuses in particular on the lessons drawn for economic value transfer.

When conducting a primary valuation study to assess the non-market value of the environmental and resource costs and benefits of WFD implementation, a number of key issues have to be addressed when using so-called stated preference valuation methods. These are the only methods able to capture so-called non-use values, which are expected to be of utmost importance in the economic analysis of the WFD. These key issues can be categorized as relating to the content and instrumental validity and reliability of water resource valuation. The former refers to the question whether we empirically value *what* we want to value theoretically in the context of the WFD, while the latter refers to the particular value elicitation procedure, i.e. *how* we elicit the values.

[1] A good example of 'getting it wrong' was the Axford public inquiry in the UK, where the Environment Agency water abstraction application in the river Kennet was rejected due to a flawed economic analysis of the non-market values based on value transfer (Moran, 1999).

Stated preference methods apply social survey formats to elicit the values held by those affected by the environmental change, in this case the water quality improvements in the context of the WFD. These values may differ depending on how questions are phrased in surveys or how the surveys are implemented. This is called procedural variance and a prominent and a well-documented case of procedural variance relates to whether values for an environmental good are elicited in isolation (part) or in a specific context (whole). **Chapter 10** illustrates this case by examining differences found when valuing part of a river basin like a water body first and the whole river basin second or the other way around. Another key issue when using stated preference methods is that of the error associated with values elicited under simulated, hypothetical conditions instead of deriving economic values from actually observed market conditions and market data. The uncertainty introduced through the use of stated preference methods is addressed in **Chapter 11** in an international river basin context.

Turning to content validity and reliability, a key feature of the WFD is its focus on water bodies at the scale of the river basin district. This poses interesting challenges for valuation in terms of *what* should be valued: the individual water bodies separately or the river basin as a whole? **Chapter 8** presents a new approach developed in AquaMoney to value water quality improvements at the level of the river basin district whilst accounting for the particular water quality baseline conditions and changes herein across the different sub-basins making up the whole basin district using GIS-based maps.

A related issue is that of substitution effects when improving water quality of water bodies and the geographical distribution of the population of beneficiaries around these water bodies in a river basin context. Substitution effects have long been neglected in the non-market valuation literature. How to properly account for substitution effects when estimating the non-market value of simultaneous ecological quality improvements across water bodies in catchments with multiple water bodies that provide a wide variety of ecosystem services is addressed in **Chapter 9**.

Sensitivity to scope, one of the acid tests in non-market valuation since the publication of the NOAA Blue Ribbon Panel recommendations (Arrow et al., 1993), can refer to the size of the water body or the basin where the WFD is implemented, but also the size of the water quality change. The WFD distinguishes different water quality levels from poor to very good and requires that the economic analysis of water quality improvement is carried out both at river basin and water body scale. An example of sensitivity to scope is presented in **Chapter 10**.

All these issues have important implications for value aggregation procedures to calculate a *Total Economic Value* for the river basin as a whole addressed in **Chapter 13**. However, also in the case of value transfer, the aggregation of the values found in the existing literature requires careful thought. Policy makers and engineers prefer economic values expressed in physical units like a hectare of land (€/ha) or a cubic meter of water (€/m^3) as this usually facilitates direct transfer of the values across the relevant policy area (in hectares) or abstraction level (in m^3) they had in mind and hence the calculation of a *Total Economic Value*. However, it is often forgotten that economics is a social science and especially economic values elicited through non-market valuation methods are based on revealed or expressed values from real people. Hence the explicit link in Figure 1.1 from the identification of beneficiaries to the aggregation of a *Total Economic Value*. Besides errors in primary or secondary value elicitation methods, the aggregation procedure is also an important source of error. One of the main messages from AquaMoney is the prescription of spatially explicit value functions instead of simple unit average values in value transfer and aggregation. Chapter 13 illustrates the effect of using value functions in the aggregation procedure on *Total Economic Value* instead of simple average values (common practice) with the help of another international river basin case study focusing on ecological restoration of heavily modified water bodies.

The Guidelines conclude in **Chapter 14** with a section with best practice recommendations based on the practical experiences and lessons learned in the AquaMoney case studies.

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2. Water valuation framework

2.1 *Water as an economic good*

Water resources provide important commodity and environmental benefits to society. As such, the management of water is an economic, social and political issue encompassing all sectors of an economy, and involving trade-offs between competing users, as well as between additional economic growth and aquatic ecosystem protection and further water resource depletion and degradation. Any particular use of water will be associated with opportunity costs, which are the benefits foregone from possible alternative uses of the resource. Decision-makers are faced with balancing, for example, water demands from agricultural irrigation for food production with the desire to protect rivers for fish and wildlife habitat. Striking a balance between the complementarity and the trade-off that exists between economic growth and water resource degradation and depletion is what defines the context that underlies the basic question of how we decide on the allocation of water resources. Economic valuation contributes towards improved allocations by informing decision-makers of the full social costs of water use and the full social benefits of the goods and services that water provides. The economic perspective on water portrays it as a natural asset that provides a flow of goods (e.g. drinking water, irrigation water) and services (e.g. hydroelectricity generation, recreation and amenity) that are utilised by agriculture, industry and households. Provision of many of these goods and services is inter-related, determined by the quantity and quality of available water. Management and allocation of water as an economic good entails consideration of its unique characteristics as a resource, which are discussed in brief below.

In terms of its more general characteristics, water is usually a scarce resource that is generally non-substitutable (although at the limit there is an almost infinite supply of seawater, which can be converted into fresh water at a cost of energy and some pollution). It has the ability to be used repeatedly or simultaneously for different uses and hence competition and complementarity are important to its consideration. Water also faces rising overall demand and use intensification.

Due to its physical nature, water tends to be a "high-exclusion cost" resource meaning that use of water is not easily controlled or prevented. Many uses of water involve the withdrawal of water ('at-site') from the hydrological system. Typically, only a small proportion of the water that is withdrawn is consumed. Consumption of water is exclusive (or rival) in its use: consumed water is retained in plants, animals, or industrial products, so is not available for other use. However, the majority of water that is withdrawn is not consumed and is returned to the water system for re-use at a later time and a different location. Water in return flows, can re-enter the surface water system further downstream, can percolate into aquifers, or evaporate, returning to the hydrological system in gaseous form. Water withdrawals are not, therefore, exclusive within a broad perspective on water use, but only within a narrow location- and time-specific context. Water can also be used without removal ('at-source') from the hydrological system (e.g. in hydroelectric power generation or boating). Such uses generally entail little or no consumption of water and hence are non-rival, though they may impact on the location and time at which water is available for consumption by other uses.

Different uses of water will have different characteristics of demand related to location, quality, quantity and timing:

- *Location* is important since water is a “bulky” resource, which means that its economic value per volume unit tends to be relatively low. The conveyance of water, therefore, entails a high cost per unit volume and often this is not economically viable. This can create values for water that are location (site) specific and hence has implications for comparing at-site and at-source uses.

- *Quality* of water is an important aspect since the quality of water (i.e. the nature and concentrations of pollutants) can exclude certain uses (e.g. drinking water for household use), but have no impact on others (e.g. hydroelectric power generation).

- *Quantity* of supply is a characteristic of water which cannot be readily specified. There is a distinction between quantity and use, as well as between withdrawals and consumption. The quantity of water available is variable and can be unreliable. This can preclude certain uses of water (e.g. the development of water-dependent industries) and can affect the value of water in some uses (e.g. irrigation).

- *Timing* related to use is important since different uses will have particular characteristics. Peak demand for water may not coincide with peak flows of surface water, creating the need for storage capacity which may be provided by naturally occurring water bodies (lakes, wetlands or aquifers) or manmade structures (dams).

Given these different characteristics of demand, any consideration of water as an economic good, needs to ensure its commensurability in terms of a common denominator of place, form and time.

Given its fundamental preoccupation with scarcity, economics defines the conditions required to secure the most efficient allocation of scarce water resources in a variety of contexts. The economic definition of an efficient allocation is one in which no reallocation can make anyone better off without making at least one person worse off, a condition that is described as ‘Pareto optimal’. Such an allocation maximises the value of water across all sectors of the economy. This is achieved through the allocation of water to uses that are of high value to society and away from uses with low value. Efficient allocation occurs in a competitive, freely functioning market when supply is in equilibrium with demand. Under these conditions, the marginal cost of the supply of water (the cost of supplying an additional unit) is equal to the marginal benefit of the use of water (i.e. the benefit of goods and services provided by an additional unit of water). Ideally, the marginal benefit and marginal cost are the same across all uses and equate with the market price. The relative efficiency of alternative allocations can be analysed with respect to this, i.e. in terms of whether they provide a ‘Pareto improvement’. The criterion of potential Pareto improvement forms the basis of cost-benefit analysis, which is used to analyse the relative economic efficiency of alternative courses of action (e.g. water allocations, new irrigation schemes, reduction of water pollution etc).

The main problem when considering economic choices related to water is that a competitive, freely functioning market does not exist for many water related uses. The reasons for this, which stem from some of the characteristics considered in the previous section, include:

- water is an essential commodity such that the value for a basic survival amount is infinite;
- water has natural monopoly characteristics;
- property rights for water resources are often absent and difficult to define;
- water is a ‘bulky’ commodity, thereby restricting the development of markets beyond the local area.

2.2 *Economic value*

Given the absence of a functioning market mechanism for many water uses and services, and in line with the increasing water conflicts and need for more efficient allocation, it is necessary to have knowledge and information of the marginal value or benefits of the resource in its alternative uses. The economic definition of value is defined in terms of economic behaviour in the context of supply and demand. Put simply, it is the maximum amount of goods or service – or money income that an individual is willing to forego (willingness to pay or WTP) in order to obtain some outcome that increases his welfare. If the outcome reduces welfare then this utility loss is measured by the minimum amount of money that the individual would require in compensation (willingness to accept or WTA) in order to suffer the changes (see Box A). These sums of money are demonstrated or implied by the choices people make, and thus reflect individuals’ preferences for the change in question. It should be noted that the WTP measure of the impact on social welfare does not consider inequalities in the distribution of gains and loses among individuals. WTP is constrained by individuals’ ability to pay.

Aggregated across those who benefit from a good or service and hence who will be affected by any change in their provision level, the aggregate WTP or WTA amount provides an indicator of their Total Economic Value (see Box B). As we shall discuss in section 2.5, economists have introduced a taxonomy of this TEV which captures the variety of values emanating from the different uses of water resources.

Box A. Economic Welfare Measures

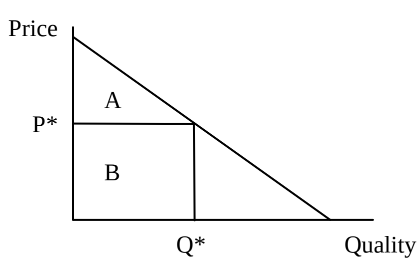
A distinction can be made between two types of welfare measures based on two different points of reference: the ‘compensating surplus’ (CS) and the ‘equivalent surplus’ (ES). The former equals the money income adjustment necessary tot keep an individual at his initial welfare level before the change in the provision level of the good, while the latter equals the money income adjustment necessary to maintain an individual at his new welfare level after the change in the provision level of the good. Four relevant welfare measures associated with welfare gains and losses can thus be distinguished:

1. WTP to secure a welfare gain (CS_{WTP})
2. WTA to forego a welfare gain (ES_{WTA})
3. WTP to prevent a welfare loss (ES_{WTP})
4. WTA to tolerate a welfare loss (CS_{WTA})

The WTP measures have become the most frequently applied in valuation studies and have been given peer review endorsement, especially because they are constrained by income whereas WTA is not.

In the context of the ‘no deterioration rule’ in the WFD (Article 4, paragraph 4.4), water status is not allowed to deteriorate, only to improve from some current (baseline) condition to the WFD objective of reaching good chemical and ecological water status. This implies that only the first two welfare measures (CS_{WTP} and ES_{WTA}) are relevant when valuing water resources in the WFD.

Box B. Demand Curve for Water: Economic Value and Price



The idea of willingness to pay and economic value can be discussed in terms of the demand curve for a product. This shows how much an individual is willing to pay for each extra unit of the product (i.e. the marginal value or benefit). The Total Economic Value to society of a good or service is determined as the aggregate of all individuals’ willingness to pay. This total willingness to pay for the product is given by the area under its demand curve (for the quality Q* this is A+B). The price of the product, P*, represents the marginal value of a unit of water quality at Q* and is the amount paid in the marketplace though in the case of water, markets often either do not exist or are highly imperfect. Some individuals are willing to pay more than this price and so receive an additional benefit over and above the amount paid. This additional benefit is known as the consumer surplus (given by area A). The average value of a unit of water quality can also be represented. When Q* units of water quality are used, the average value is equal to (A+B)/Q* and is larger than the marginal value (by the amount A/Q*). The figure illustrates how the price of a good or service and its economic value are distinct and can differ greatly: water can have a very high value, but a very low price or no price at all.

The aim of economic valuation is to secure efficient water resource allocation including pollution by providing the same level of value information that would normally be afforded by prices for a market good. Water valuation would ideally be considered under a general equilibrium framework, though this is in practice extremely difficult, such that a sector approach is usually taken. Such a sector approach means that there may be different ways of calculating values, which may result in fundamentally different definitions of value that differ between the uses being considered. In particular values can defined in terms of:

- Average or marginal value concepts, which as discussed earlier are quite different concepts. Marginal values are stressed for the purposes of efficient allocation. In economic terms, the allocation of water to alternative uses should be based on the marginal value of water in those uses.
- Short run and long run value contexts. In the case of same site and production processes, long run values will be smaller than short run values since in the short run the costs of fixed inputs can be ignored as sunk costs, whereas in the long run such input costs must be covered. Similarly, in the case of water demand, since responses and adjustments are more constrained in the short run than in the long run, then short run values may again be higher than values in the long run. Since public water policy decisions typically involve situations where the long run context is more appropriate then care must be taken in not attributing too high a value to some water uses.

2.3 *Economic cost*

The cost of a productive activity consists, in theory, of the opportunity costs of the necessary inputs. The opportunity cost of employing an input is the highest net benefit generated had it been employed elsewhere. This is the cost to society of use of the resource. Opportunity costs seek to measure the full societal cost of an activity or service that employs a natural resource such as water. It is considered in terms of a change at the margin, i.e. marginal opportunity cost, because management decisions usually entail relatively small changes in resource use.

Opportunity costs consist of the following main components (e.g. Pearce and Markandya, 1989)[2]:

- Direct economic user cost of water, such as for example the costs of labor, equipment and materials used for abstraction or wastewater treatment. Such costs

[2] They distinguish also a third cost component, namely *scarcity rent* due to resource exploitation resulting in its non-availability for future use. The scarcity rent relates to the value of the opportunity foregone by exploiting and using a water resource in the present period rather than at some time in the future. It also incorporates increases in the costs of future resource use and exploitation that occur as a consequence of current use and exploitation. For example increases in costs of future groundwater pumping in alternative, less easy accessible places. We exclude this category here to keep things simple and to be able to simply distinguish between private and social costs. The scarcity rent basically is another form of external cost imposed on future generations instead of current water users.

require adjustment for any subsidies, taxation and market imperfections in order to reflect true opportunity costs (shadow pricing).

- External cost that arises from water use including abstraction and pollution. This is the net value of any losses and gains in welfare that water use imposes on others than the direct users. External costs arise because changes in one component of the natural resource base affect other components and the efficiency with which other activities can be conducted.

If economic markets exist and function well, opportunity costs are reflected in the market prices paid for the inputs. For example, the price of the necessary labour, equipment or electricity to produce a good or service. However, in some cases, inputs are used or outputs produced for which no markets exist and hence also no market prices are available to reflect their opportunity costs or which are traded in distorted markets, which do not reflect the real opportunity costs of the inputs used or outputs produced. Examples include natural purification processes or a water resource's function as a disposal sink for emitted pollutants. Here, environmental damages caused by water use (e.g. water abstraction or the emission of pollutants) often do not result in a private cost to individual agents, but in a social cost to society. Failure to set water charges for irrigation on the basis of opportunity costs has, for example, been argued to be a classic cause of inefficiency in the agricultural sector.

Following the distinction above, **environmental and resource costs** are defined here as the total economic value of the environmental damage as a result of the gap between the current and good chemical and ecological status of water bodies, <u>including</u> the economic value of the opportunities foregone under scarcity conditions across different water uses and users due to existing water allocation and distribution rules. Part of these costs may be internalized through direct user costs (referred to as private costs in the Introduction), part of these costs will be external costs (referred to as social costs in the Introduction). The Guidelines primarily focus on the economic valuation of the external costs not observable through market prices as this is where most methodological issues in water resource valuation remain.

A much misunderstood, but important issue related to economic valuation entails the use of costs as determinants of economic value. As discussed above, the correct measure of economic value is determined based on net benefits, as indicated by the area under the demand curve. However, costs are often used as a proxy for benefits. This is based on the misplaced assumption that costs are necessarily a reasonable approximation of benefits and that the benefits are at least as great as the costs involved in repairing, avoiding or compensating for damage. Such cost based measures of value are derived from the supply of goods and services and should not be confused with demand-based measures. Whereas demand based valuation techniques attempt to derive recognised measures of economic value, such as consumer surplus or the measures of welfare shown in Box B, cost based approaches do not. As such they do not correspond with the notion of total economic value and measures of willingness to pay. Instead, these approaches provide a proxy of value by considering supply-side aspects.

Box C. Environmental and Resource Costs

In Wateco (2002) and various working groups under the WFD Common Implementation Strategy (CIS), environmental and resource costs have been split up in a separate environmental cost and resource cost category in order to account for the different water management issues playing a role across EU Member States, especially in South Europe where many water management problems are related to its limited availability. In the latter case water scarcity and corresponding inefficiencies in water allocation across user groups (the costs of foregone opportunities) are referred to as resource costs (e.g. Pulido Velázquez et al., 2008). Environmental costs would in that case only refer to the total economic costs (welfare loss) of the physical environmental damage to a water system (water body or river basin) as a result of the chemical and/or ecological state of the water system.

The distinction between the two concepts based on the characteristics of the water management problem (and the depletable nature of unsustainable water resource use) may be essential for policy purposes. They usually require different valuation approaches. In many cases, resource costs are computed using hydro-economic models focusing on the efficient allocation of water across different water users based on, for instance, daily water uptake in agriculture or industry and by domestic households. The shadow price of water is derived through its marginal contribution to production and consumption. However, in many of these models, environmental demand of water is mainly included through the use of environmental standards imposed as restrictions on economic production and consumption activities (Brouwer and Hofkes, 2008). Demand for environmental quality can be estimated through the non-market valuation methods presented in these Guidelines. In a subsequent step the marginal non-market value of environmental flow levels can be included in a hydro-economic model.

In the Guidelines, the most important point of reference for water resource valuation, including the estimation of environmental and resource costs, is provided by the WFD objective of reaching good chemical and ecological status. The environmental change valued consists of the difference between the baseline state of a water body or river basin and its good chemical and ecological status as defined in the WFD. Water flow levels are, in principle, an integral part of this environmental change as water quantity and water quality are intrinsically interwoven. By definition, the total economic value of the environmental and resource costs include the possible benefits foregone (opportunity costs), such as loss of the economic benefits derived from, for example, recreational opportunities, wildlife habitat and biodiversity conservation, i.e. the (indirect) welfare impacts of the environmental change on various relevant water uses and users.

Cost based estimates are widely applied due to the relative ease in their use and availability of data. In addition they integrate well with the other types of economic analysis used in the WFD, such as cost-effectiveness analysis. However, it is important to be aware of the limitations in terms of the information they convey. Since costs do not really measure total economic value, application of cost based approaches risks the under-valuation of water resources and hence their under-provision. Furthermore cost based estimates can be an inconsistent estimator of value. This is because the higher the level of water quality, the higher the incremental costs of further improvements usually are. Benefit estimates based on costs will thus incorrectly justify further improvements from higher levels of water quality compared to lower levels of water quality. Despite these problems, use of cost based approaches can sometimes be useful where it is possible to argue that any remedial work must take place because of some other constraint such as an existing water quality standard. Under such a situation the costs of achieving that standard are a proxy for the benefits of reaching the standard, since society can be assumed as having sanctioned the cost by setting the standard. However, if the remedial cost is a measure of damage then the cost-benefit ratio of undertaking the remedial work will always be unitary. That is to say remedial costs are being used to measure remedial benefits. In fact, use of cost based estimates as a proxy of benefits means that there is nothing to compare costs within a cost-benefit analysis. To say that the remedial work must be done implies that benefits exceed costs. Costs are then a lower bound of the true value of benefits. As such, the use of costs as a proxy of benefits would not provide us with the necessary information to make any judgements regarding disproportionate costs in the WFD implementation. Given the lack of welfare interpretation and the inconsistency of costs as an indicator for the value of environmental and resource costs (the higher the water quality, the higher the incremental costs), the emphasis in Aquamoney is on the welfare based (benefit) approaches to valuation.

2.4 *Water valuation framework*

As part of the WFD river basin planning process, a number of steps can be distinguished in trying to estimate the environmental and resource costs and benefits associated with water use and services. This starts off with a physical characterisation of the environmental damage caused, which provides the basis of the subsequent economic assessment and valuation of this damage. The environmental impact assessment determines the causes of aquatic ecosystem degradation using existing Impact-Pathway methodology frameworks such as the Driving Forces-Pressure-State-Impact-Response (DPSIR) framework. This involves the input (knowledge, expertise and information) of environmental experts and requires the analyst to:

1. Identify the significant pressure, which causes a water body to change and not reach the set environmental WFD objective(s). In principle, if there is no significant pressure (reduction), there will also be no environmental or resource cost (benefit).

2. Assess the full impact of this pressure (reduction) on the water environment, in relevant chemical and/or ecological terms. This requires the selection and description of the relevant attributes or characteristics of the water system for the economic valuation exercise.

3. Identify and, if possible, quantify the nature and extent of the damage involved, both on the water environment and other water users (i.e. human welfare significance). Damage is defined here as the difference between some reference and target situation and the corresponding effect on the provision and quality of the goods and services involved. Essentially a dose-response function is needed for this in order to determine how changes

in the composition of the water environment lead to changes in the water system functions affected and hence to the provision of benefits of these functions in terms of goods and services.

4. Asses the human welfare significance of the damage in monetary terms. This economic assessment, which follows the previous environmental impact assessment steps, is based on the establishment of market and non-market relationships regarding the changes in provision and quality level of water system goods and services.

The above steps can be visualized in terms of the overall Water Valuation Framework shown in Figure 2.1. Figure 2.1 shows how the economic valuation of environmental damage costs and (avoided) benefits, which is the subject of the Guidelines, is positioned clearly within the overall assessment framework and preceded by an environmental impact assessment. The Guidelines' focus is on the economic valuation issues, not on the construction and estimation of the physical dose-response relationships underpinning the environmental impact assessment.

Figure 2.1: Water valuation framework

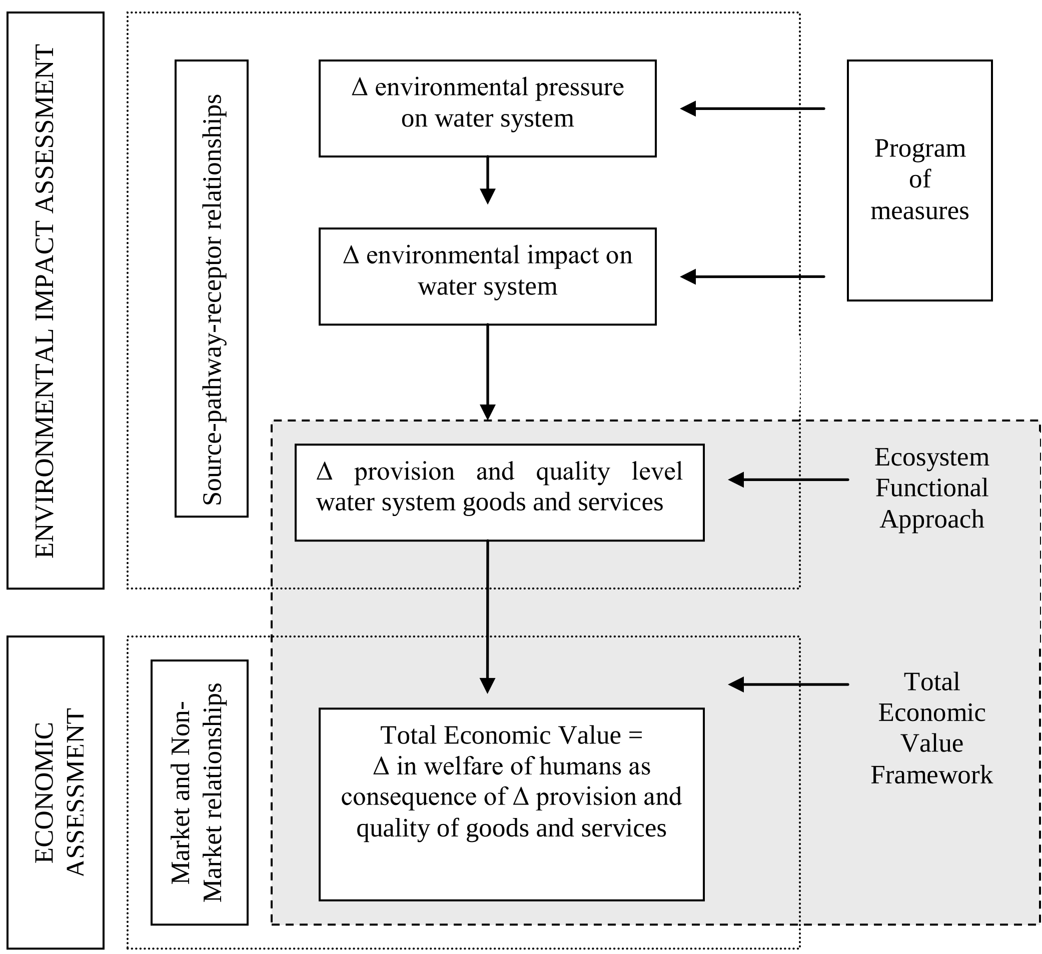
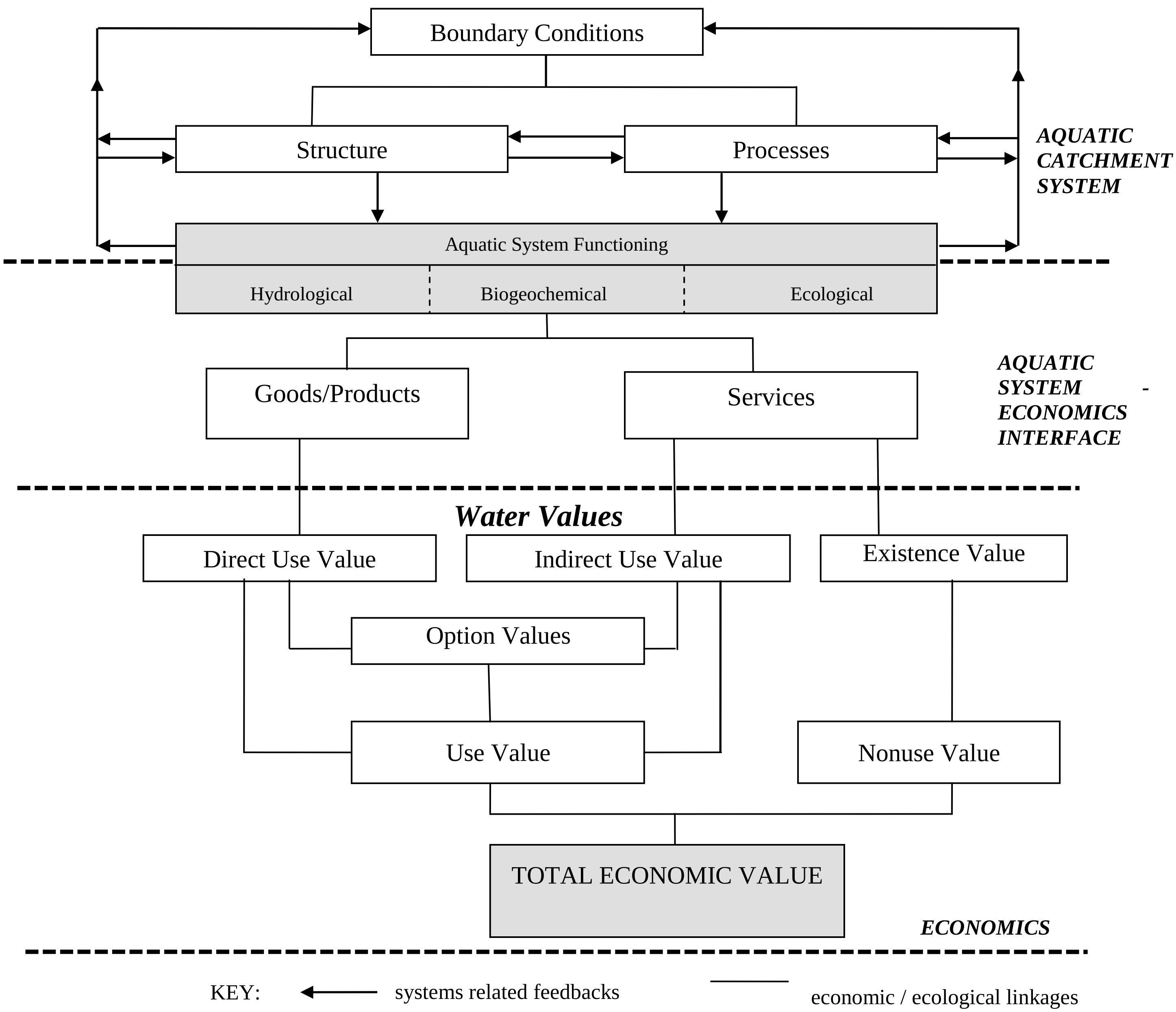


Figure 2.2: Framework for water resource valuation based on an aquatic ecosystem functional approach and the concept of Total Economic Value



Source: Turner et al. (2004).

The analytical framework and the methods and techniques detailed in these *Guidelines* are based on an approach that considers all water resources as functional components of a set of interrelated aquatic systems encompassed within catchment boundaries. The valuation framework is guided by the concept of *Total Economic Value* and the aquatic ecosystem *Functional Approach* to natural resource management. This interdisciplinary framework examines the Total Economic Value of water resources via the linkage between aquatic ecosystem structures and processes and the outcomes of the functioning of such systems in terms of the beneficial goods and services provided to society. As such the framework is able to appreciate the full functioning of the hydro-ecological system and the total range of functional outputs of goods and services that are provided. Figure 2.2 attempts to depict this Framework for Water Valuation.

The analytical framework can also be viewed in terms of a number of stages though which any analysis proceeds:

STAGE 1		STAGE 2		STAGE 3		STAGE 4
Classification/ Characterisation of Water/Aquatic System	→	Water/Aquatic System Functions	→	Functioning Outcomes [goods/service]	→	Valuation [methods/techniques]

The initial stage requires a working classification of the relevant aquatic system. This will depend on the types of catchment systems and characterisations relevant to the particular policy area. Once this classification is established the set of functions provided by the particular aquatic system needs to be identified. The third stage involved the matching of functions and/or combinations of functions with outcomes in terms of tangible and intangible goods and services utilised or appreciated (i.e. welfare affecting) by human society. The final stage is concerned with the valuation of the goods and services provided by the aquatic system, through a variety of economic valuation methods and techniques.

2.5 *References*

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3. Aquatic ecosystem functions and Total Economic Value

3.1 *Aquatic ecosystem functions*

Aquatic ecosystems provide a wide range of goods and services of significant value to society. Valuation of aquatic resources requires an understanding of water ecosystem functioning that can be described through the hydrological and physical components and their associated interactions. 'Valuing' the aquatic ecosystem can be conceived as essentially valuing the characteristics of the system, and capturing these values in an economic value framework. One of the major challenges for such a framework is integrating the hydrological and physical components of aquatic ecosystems into an economic valuation scheme. The aquatic ecosystem functional approach, along with the Total Economic Value concept, provides the foundations for doing this.

The interactions among an aquatic systems hydrology and physical components determine the general characteristics and the significance of the processes that occur in any given aquatic system. Processes refer to the dynamics of transformation of matter or energy. These processes enable the development and maintenance of the aquatic system structure. Structure is defined as the tangible items such as the water stocks and flows, plants, animals, etc, of which the aquatic system is composed. The structure is key to the continuing provision of human use outcomes that involve some direct utilisation of one or more characteristics of the aquatic system. The processes in an aquatic ecosystem, on the other hand, provide ecologically related services to humans, so that some aspect of an ecosystem supports or protects a human activity or human property without necessarily being used directly. Ecosystem structures and processes in themselves are therefore not necessarily of economic value; such values derive from the satisfaction of human needs and wants by the goods and services provided by the functions that are contingent on the structures and processes. Aquatic system functions are the result of interactions among characteristics, structure and processes. They include such actions as water recharge and discharge, flood control, nutrient retention, etc. The concept of aquatic system functions and functioning provides the link between the aquatic ecosystem (functioning of) structures and processes and the provision of goods and services that are of value to society. It thereby creates an interdependent perspective of the aquatic and economic systems. Although multiple definitions of functions exist in the literature, they have in common that they all reflect an anthropocentric perspective on ecosystem functioning, where aquatic system characteristics, structures and processes contribute to human welfare.

Aquatic system processes, both hydrological and ecological, will encompass changes that transcend the local scale and the short time period. In these linked hydrological-ecological systems, catchments provide a wide range of benefits and services that can often be ignored or under-valued. Indeed, the extent or degree to which different goods and services are deemed valuable is conditioned by a diversity of catchment level contextual factors. The human recipients of the aquatic system goods and services will also be distributed across different spatial and temporal scales. Aquatic systems should thus be seen within a catchment or river basin scale context. Use of a catchment scale with the functional approach allows us to more effectively consider water not just in terms of the reallocation of water supplies amongst water using sectors, but also with regard to other dimensions, including water quality and supply reliability. Furthermore, use of this approach allows better evaluation of all competing uses, in particular with respect to return flows, and the implications of withdrawals or depletion in considering values of water resources.

Implementation of the functional approach requires the practical coupling of economic, hydrological and ecological models. The first step is to compile a complete list of all the relevant characteristics that describe the system in the simplest and most objective terms possible. They are a combination of generic and site–specific features. A general list would include such characteristics as the hydrological, biological, chemical and physical features that describe an aquatic ecosystem. Hydrological information for example, includes factors such as rainfall, run-off, infiltration, water balance data, depth to ground water data, flow rates and direction, etc. Nevertheless, in principle the list is endless and site-specific. Inherent in understanding the physical nature of the aquatic system is the need to understand water quality as well. These characteristics singly or in combination, give rise via aquatic system structure and processes to a particular set of functions.

Many of these functions can be categorised according to whether they are hydrological, biogeochemical or ecological functions:

- *Hydrological functions* refer to an aquatic systems ability to store floodwaters, the interactions between ground and surface waters and the storage of sediments.
- *Biogeochemical functions* refer to the export and storage of naturally occurring chemical compounds that can have significant effects on the quality of the environment.
- *Ecological functions* relate primarily to the maintenance of habitats within which organisms live.

Interactions between the functions will affect the quantity and quality of water in the overall system at any time. The functions are often highly interrelated and these interrelationships need to be accounted for when considering the linkages between changes in the aquatic system and changes in economic values. While the interrelationships between functions can be complex, some of this complexity can be reduced where more aggregative impact/welfare effects are determined and then valued.

Nevertheless, the valuation of the functions of an aquatic system implies full knowledge of the use outcomes of the functioning of that system. What counts is whether human welfare is increased or decreased due to the provision of aquatic system derived water use outcomes. It is often the case that a combination of functions within an aquatic system will combine to produce an outcome that is valued by humans. From an economic valuation standpoint it is thus important to recognise that some relationships between functional outcomes are complementary, others competitive or even mutually exclusive.

The question is then whether all the benefits from aquatic ecosystems can be classified in terms of use outcomes of goods and services? It is evident that there are strong linkages between the types of benefits. For example, nutrient retention is necessary to ensure clean water. Nevertheless each of these benefits provides a distinct positive value to the overall system, although the need to ensure against double counting cannot be overstated. An assessment of the complete range of benefits using a standard classification of use outcomes is an essential step in deriving their value. In compiling a comprehensive list of water use outcomes, there exist a number of ways these can be classified. This can be according to a variety of spatial and sectoral criteria. Table 3.1 shows typical classifications found in the water valuation literature that can be used to help compile a comprehensive listing of water use outcomes.

Based on a review of the use outcomes associated with these classifications and avoiding overlaps, Table 3.2 shows a listing of use outcomes in terms of the goods and services that impinge on human welfare (across the top) and links these to some of the hydrological, biogeochemical and ecological functions/combination of functions (left hand column) typically found in many aquatic ecosystems, from which they are generated.

Table 3.1: Classifications of water use outcomes

• In-stream uses – those uses occurring in a watercourse and dependent on flow • Off stream uses – those uses making use of water away from the watercourse
• Intermediate (producers') uses – those uses that are employed to make final products (to be eventually used by consumers) • Final consumption uses – those uses that provide consumptive human satisfactions • Environmental service uses – those uses related to recreation and waste assimilation
• In-situ uses – those uses of water that occur or exist as a consequence of water remaining in place within the aquatic system • Extractive uses –those uses involving removal of water from the aquatic system
• Domestic uses – those uses for domestic or household purposes • Agricultural uses – those uses for increasing the productivity of agricultural land • Commercial uses – those uses for commercial and institutional purposes • Industrial uses – those uses related to water as a factor of production • Power generation uses – those uses related to the production of power • Ecosystem service uses –those uses related to natural processes that contribute to the productivity of the economy and the welfare of households • Non –consumptive uses – those uses related to recreation and more intrinsic uses that are passive
• Consumptive uses – those uses that cause diminishment of the source at the point of appropriation such that it is not immediately available for another use. • Non-consumptive uses - those uses where there is no diversion from the water source or diminishment of the source
• Abstraction uses – those uses for irrigation and other agicultural uses, domestic water supply and water for industrial production. • Fisheries – those uses for commercial fish and shell fisheries, non-commercial 'heritage' and recreational fisheries. • Recreation – those uses for recreation (canoeing, sailing, bathing, walking, picnicking, bird watching, etc). • Biodiversity and related landscape conservation – those uses for conservation of species and related natural process.

Table 3.2: Aquatic system functions and their use outcomes in terms of goods and services

	Goods							Services																				
Function \ Outcomes	Potable water for household use	Water for landscape maintenance and peat soil	Water for crop irrigation	Water for livestock consumption	Water for food processing	Water for other manufacturing processes	Cooling water for power plants	Water transport	Prevention of saline intrusion	Water/soil support for prevention of land subsidence	Natural erosion, flood and storm protection	Shoreline stabilization	Sediment removal	Transport, treatment, medium for wastes and other by-products of human activities	Improved air quality through the support of living organisms	Biological diversity provision	Recreational swimming, boating, fishing hunting, trapping and plant gathering	Commercial fishing, hunting, trapping and plant gathering	Energy production	On and off site observation and study for leisure, education and scientific purposes.	Micro-climate regulation	Macro-climate regulation	Toxicant removal	Toxicant export	Cultural value provision	Historical value provision	Aesthetic value provision	Wilderness value provision
Water discharge	■	■	■	■	■	■	■	■						■	■	■	■	■		■	■			■				
Water recharge	■	■	■	■	■	■	■		■	■					■	■	■			■	■							
Flood mitigation										■	■					■				■								
Sediment retention	■							■		■	■	■	■		■	■	■	■		■			■					
Nutrient retention														■	■	■	■	■		■							■	
Nutrient export														■	■	■	■			■								
Trace element storage																■	■	■		■			■					
Trace element export																■				■				■				
Carbon sequestration																■	■		■	■		■						
Biodiversity maintenance															■	■	■	■	■	■	■						■	■
Culture/heritage																	■			■					■	■	■	

It seems clear that all aquatic ecosystems to a greater or lesser extent provide some valued goods/services but clearly different systems will yield different mixes of goods/services. Nevertheless, similar systems may provide quite different mixes of ‘valued’ goods/services depending on their locational context. Locational factors serve to emphasise the importance of assessing aquatic resources within their relevant catchments or river basins. Relevant catchment-level contextual factors that require consideration are shown in Box D.

Box D. Catchment-Level Contextual Factors Requiring Consideration in Valuation of Water Resources

- The size and characteristics of human settlements and their proximity to the aquatic ecosystem and the outcomes of the functions it performs.
- Accessibility of the ecosystem to humans.
- Land use in the vicinity of and downstream from the aquatic ecosystem.
- Configuration of downstream resources and local industries such as farming, manufacturing, mining, tourism, forestry - not limited to the catchment, but the local region.
- Size and distinctiveness of the aquatic ecosystem in comparison with other nearby ecosystems.
- Likely changes in human factors in the catchment in the foreseeable future, such as urban expansion, change in farming practices, increasing aquatic management, road building, river management etc.
- Substitution possibilities at the regional scale and beyond
- The existence of aquatic management schemes within or nearby the ecosystem.
- Known or historical problems with the water environment, such as pollution of the river, flooding episodes, etc (including problems upstream and downstream)

We can now bring together in the following section the ecological and economic dimensions in order to make explicit the links between the outcomes in terms of goods and services provision and their reflected Total Economic Value in society.

3.2 *Total Economic Value*

Aquatic ecosystems are natural assets that create flows of goods and services over time. As outlined in the previous section, the key to their valuation is to establish the functions that they provide and link this to the use outcomes that are valued by society. If that link can be established, then the concept of derived demand can be applied. The value of a change in the functions provided can be derived from the change in the value of the stream of benefits. Given the multi-faceted nature of benefits associated with aquatic systems there is a need for a useable typology of the associated values. As already discussed, the focus here is on economic values, which depend on human preferences, i.e. what people perceive as the impact on their welfare. Economic values are relative in the sense that they are expressed in terms of something else that is given up (the opportunity cost). We need to consider how and to what extent the concept of economic value captures the variety of aquatic ecosystem values.

Although a number of classification systems exist to describe the different types of environmental and resource values associated with the goods and services provided by aquatic resources, economists have generally settled for a taxonomy based on the concept of Total Economic Value (TEV). TEV is determined as the sum of the components in Figure 3.1.

The key distinction made is between use values and a remainder called non-use value:

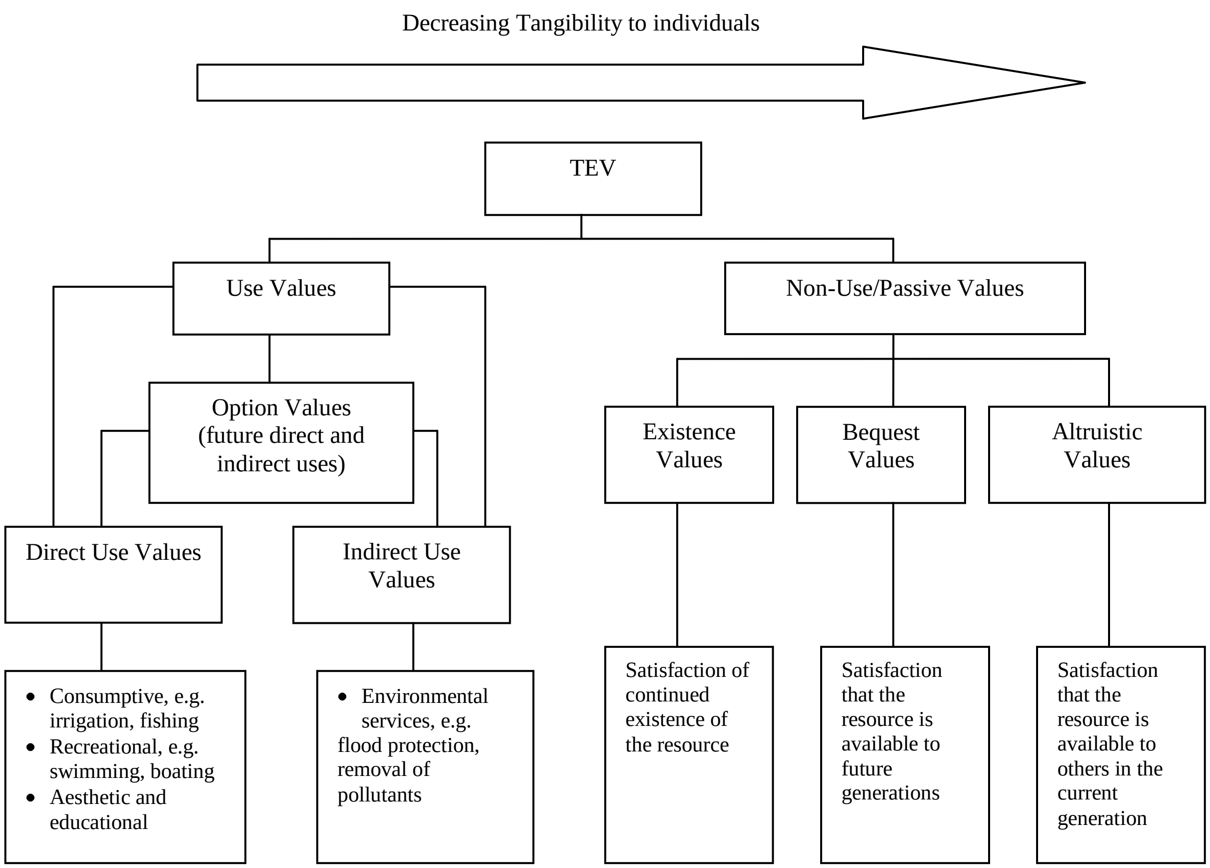
Use Values

- Direct use values arise from direct interaction with aquatic resources. They may be consumptive, such as use of water for irrigation, the harvesting of fish or they may be non-consumptive such as recreational swimming, or the aesthetic value of enjoying a view. It is also possible that 'distant use' value can be derived through the media (e.g. television and magazines), although the extent to which this can be attributed to a specific site, and the extent to which it is actually a use value, are unclear.
- Indirect use values are associated with services that are provided by aquatic resources but that do not entail direct interaction. They are derived, for example, from flood protection provided by aquatic ecosystems or the removal of pollutants by aquifer recharge.

There is a further type of value that is related to *future* direct and indirect uses. This is option value:

- Option value is the satisfaction that an individual derives from ensuring that an aquatic resource is available for the future given that the future availability of the resource is uncertain. It can be regarded as insurance for possible future demand for the resource.

Figure 3.1: Components of the Total Economic Value of water resources



Non-Use Values
Non-use value reflects value in addition to that which arises from usage and are derived from the knowledge that an aquatic resource is maintained. By definition, they are not associated with tangible benefits that can be derived from it (though resource users may derive non-use values). Thus individuals may have little or no-use for a given aquatic asset or attribute but would nevertheless feel a 'loss' if such things were to disappear. Non-use values are linked to ethical concerns and altruistic preferences, though it can be argued that these ultimately stem from self-interest. However the boundaries of the non-use category are not clear cut and some human motivations which may underlie the position that the asset should be conserved 'in its own right', and labelled existence value, are arguably outside the scope of conventional economic thought. In practice, what is at issue here is whether it is meaningful to say that individuals can assign a quantified value to the environmental asset, reflecting what they consider to be intrinsic value.

Non-use values can be divided into three types of value (which can be overlapping): existence value, bequest value and altruistic value:

- Existence value is the satisfaction derived from an aquatic resource continuing to exist, regardless of whether or not it might be of benefit to others. Motivations here could vary and might include having a feeling of concern for the asset itself (*e.g.* a threatened species) or a "stewardship" motive whereby the "valuer" feels some responsibility for the asset.
- Bequest value is the satisfaction derived from ensuring that an aquatic resource will be passed on to future generations so that they will have the opportunity to enjoy it in the future.
- Altruistic value is the satisfaction derived from ensuring that an aquatic resource is available to contemporaries in the current generation.

Economic values, given by TEV, can now be combined with the aquatic ecosystem functional (and related goods and service outputs) approach to provide a comprehensive economic assessment framework for water valuation. It is important to note that what is being valued is not the aquatic ecosystem per se, but rather independent elements of goods and services provided by aquatic ecosystems. The aggregation of all function based values provided by a given water ecosystem yields the TEV of that ecosystem. Table 3 shows the comprehensive list of aquatic ecosystem functional outcomes in terms of goods and service and links them to their respective TEV components. As such it provides the final stage in integrating the physical components of an aquatic ecosystem and the TEV of the goods and services provided by the ecosystem.

Table 3.3 also indicates whether the various categories of outcomes and TEV components are commonly expressed as either stock (asset) values or flow values, or as both. It is important to distinguish between stocks and flows if one is to avoid the problem of double counting or other errors. Flows of value, such as those from extractive uses, are those that recur over time, as opposed to stock values, which are the value of an asset that yields flows of services (and hence values) over time. Flow values and stock values are linked because a stream of values can be converted into an asset value by calculating the present discounted value of the flow. In instances where values are commonly expressed as either stocks or flows, it is important to specify whether the value is a flow or a stock.

It should be remembered that Total Economic Value does not, however, provide an exhaustive assessment of the value of water resources to society. It measures the extent to which goods and services provided by water touch on the welfare of society, as direct determinants of individuals' wellbeing or via production processes. It represents two fundamental sets of values: individual values and production values. Individual values include recreational and amenity values, as well as non-use values (existence, bequest and philanthropic values) of goods and services provided by water. Production values occur through the influence of water on the production and cost functions of other marketed goods and services (such as use of water as an intermediate good in irrigated crop production). The effects of this influence on the prices of other inputs and marketed goods and services translate into changes in individuals' welfare.

Table 3.3: Aquatic ecosystem functional outcomes, TEV and accounting values

Function Outcome Terminology (Goods and Services)	Total Economic Value terminology		Accounting terminology	
	Use Values[1]	Non-use Values	Stocks	Flows
Water for residential use	*			*
Water for landscape and turf irrigation	*			*
Water for agricultural crop irrigation	*			*
Water for livestock watering	*			*
Water for food product possessing	*			*
Water for other manufacturing processes	*			*
Water for power plants	*			*
Water transport	*			*
Prevention of saline intrusion	*		*	*
Water/soil support for prevention of land subsidence	*		*	*
Natural erosion, flood and storm protection	*		*	*
Shoreline stabilization	*		*	*
Sediment removal	*		*	*
Transport, treatment and medium for wastes and other by-products of human activities	*		*	*
Improved air quality through the support of living organisms	*		*	*
Biological diversity provision (incl. habitat and nursery provision)	*		*	*
Recreational swimming, boating, fishing hunting, trapping and plant gathering	*			*
Commercial fishing, hunting, trapping and plant gathering	*			*
Bio-energy production	*		*	*
On and off site observation and study for leisure, education and scientific purposes.	*		*	*
Micro-climate regulation	*		*	*
Macro-climate regulation	*		*	*
Toxicant removal	*		*	*
Toxicant export	*		*	*
Cultural value provision		*		*
Historical value provision		*		*
Aesthetic value provision	*			*
Wilderness value provision		*		*
Other non-use service provision		*		

[1] Includes option values (since these are concerned with future direct and indirect uses).

In practical terms, assessment of TEV is limited to those components that it is feasible to quantify. The ability to value aquatic ecosystem goods and services is constrained by the complexity of the aquatic ecosystem itself. The "production function" of aquatic ecosystems is so complex, and little understood in many instances, that reliable estimates of all outcomes cannot be made. An aspect of this complexity is that joint products are inherent in most water ecosystem processes. Accounting for value must recognise all these joint product values. Recognising the TEV of water even when one cannot develop quantitative separations of the various components is still important since delineation of what can and cannot be quantified can still provide insights for policy and management decisions.

Applying the framework terminology shown in Table 3.2 and Table 3.3 on a catchment or river basin basis also provides a consistent methodology for an aggregate accounting of water resources. By estimating the TEV of the goods and services generated by aquatic ecosystems on a catchment or river basin basis, an aggregate measure of the TEV can be developed, accounting for all aquatic resources. Changes in the use of water within the catchment or river basin, for example, leading to changes in the quality or quantity of water in the catchment or river basin, can be assessed using the change in TEV specific to that catchment or river basin.

The valuation framework must still take into account how time, institutions, water quality and quantity, hydrological, ecological and biogeochemical factors interact to affect the value of aquatic resources. There must also be recognition of any distorting impacts of government policies and regulations (e.g. agricultural subsidies inflating the agricultural sector's demand for water. Some of these issues will be considered in more detail in the following sections.

In conclusion, a typology of values based on the TEV concept is an appropriate way to represent the multi-faceted nature of the benefits associated with aquatic ecosystems; and despite some grey areas around the precise demarcation of use and non-use value categories, such a distinction is meaningful and practical. We now look at the methods and techniques used to assess the TEV of water resources

4. Economic valuation methods

4.1 *Water resource valuation methods*

There are numerous difficulties and problems in any attempt to estimate the various values associated with water resources. Renzetti (2002) outlines the following five particular problems:

- The paucity of market transactions in water. Even where markets do exist prices may not reflect the social marginal costs of supply.
- The variety of regulations and legal institutions related to water will influence the allocation of water and, thus, distort its value, e.g. trade barriers and subsidies.
- In general the value assigned to the use of water is a function of its quality. Whilst the impact of changes in water quality on its value are relatively well understood, there are many uses of water where such knowledge is incomplete, e.g. between water quality and its many ecological functions.
- Unwanted excess supplies of water can result in WTP for water that is actually negative, e.g. water related to flooding or erosion problems.
- Difficulties related to the application of neo-classical valuation techniques due to the cultural and religious roles of water that are emphasized by some societies.

Despite these difficulties a range of valuation methods and techniques exist and have been applied to estimate the value of water resources. The various techniques presented here include the estimation of demand curves and the area beneath them, analysis of market like transactions, use of production approaches that consider the contribution of water resources to the production process, estimation of the costs of providing alternative sources of water, as well other techniques used to estimate environmental resources more generally. The methods and techniques reflect the extent to which the good and services provided by aquatic ecosystems touch on the welfare of society either as direct determinants of individuals' well-being (e.g. as consumer goods) or via production processes (e.g. as intermediate goods).

However in applying these techniques to estimate water values, Young (2005) notes a number of general difficulties as follows:

- Data related to water use are often not available and expensive to collect.
- Values for water are usually site specific, especially since water is a "bulky" commodity which can exhibit high conveyance costs.
- Methods and assumptions across water resources are not standardised
- The time frame of water use is important since it relates to the fixity of certain inputs and hence differences in long run and short run values.
- The measure of water value has to be commensurable in terms of a common denominator of place, form and time.
- Quantity and quality measures for water are not standardised. e.g. withdrawals versus depletion or consumption for off-stream uses. But neither of these relevant to in-stream uses. need to account for form, timing and location
- Uncertainty (scientific and economic) related to water use may be quite high. Benefit estimation requires forecasting the behaviour of a number of economic, technological and social variables over a long time period. Highly unpredictable factors can be influential.

The aim here is to provide an overview of all the techniques before focussing on the practical application of some of those techniques that are of particular relevance to Aquamoney. Further details of the underlying theory and application of all the techniques is provided in general texts including, Braden and Kolstad (1991), Freeman (2003), Bateman et al (2002), Mitchell and Carson (1989), Champ, Boyle and Brown (2003), Kanninen (2007), Bockstael and McConnell (2006). In addition, Young (2005) and Renzetti (2002) provide a more detailed survey of the application of valuation techniques to water resources. A useful summary of all of the various methods and techniques used to value water resources is provided in Annex 2.

An important distinction to make is between market-based valuation techniques and non-market based valuation techniques. Market valuation means that existing market behaviour and market transactions are used as the basis of the valuation exercise. Economic values are derived from existing market prices for inputs (production values) or outputs (consumption values), through more or less complex econometric modelling of dose-response and/or damage functions. Examples include the economic value of fish, which is sold on a fish market (market analysis), the costs of replacing impaired environmental riparian functions such as nutrient retention and export through the installation of a wastewater treatment plant (replacement costs) or the costs of a water filter on tap water (avertive behaviour/defensive expenditures). The economic value of water use and exploitation such as surface or groundwater abstraction for agriculture, the food or paper industry, the energy sector or for drinking water purposes can also be measured directly through existing market prices for intermediate or final water products (e.g. drinking water price). Here, the market price is multiplied by the quantity of water consumed to yield the total market value. The market price may have to be adjusted to provide social or shadow prices, but otherwise they are likely to provide a relatively simple means of assessing economic value. In some cases, human resource use may also include recreational activities, which depend on water availability. Examples are fishing for fish species with specific water depth requirements, surfing, or white water rafting, but also the recreational activity swimming can be considered a function of water availability. In some cases recreational values can be derived from existing fish permits or entrance fees.

When the water resource is depletable (non-renewable), and current use limits future use opportunities resource accounting methods like the net price method may be used, where a ‘scarcity rent’ is calculated as the difference between the market or shadow price of the water resource and its marginal extraction costs.

Many water resources are not traded in markets and therefore remain un-priced. It is then necessary to assess the economic value of any environmental damage (avoided with the help of existing pollution abatement and mitigation measures) with the help of *direct* and *indirect* non-market valuation methods. Non-market valuation means deriving economic values in cases where such markets are non-existent or distorted. Direct methods (also called stated preference methods) refer to contingent valuation (CV), discrete choice experiments (CE), and contingent ranking (CR) techniques, where individuals are asked directly, in a social survey format, for their WTP for a pre-specified environmental change. WTP can also be measured indirectly by assuming that this value is reflected in the costs incurred to travel to specific sites, such as with recreational visits (travel cost studies), or prices paid to live in specific neighbourhoods (hedonic pricing studies) (also called revealed preference methods). The latter two approaches are based on preferences being ‘revealed’ through observable behaviour, and are restricted in their application to where a functioning market exists. CV, CE and CR, being based on surveys that elicit ‘stated preferences’, have the potential to value benefits in all situations, including non-use or passive use benefits that are not associated with any observable behaviour. The legitimacy of these methods and results is still contested, especially in the context of non-use values, and conducting surveys can sometimes be a lengthy and resource-intensive exercise. Of these methods, CV is probably the most widely applied method in contemporary valuation research (Carson *et al*., 1995; Bateman et al, 2002).

Water use and exploitation can likewise be estimated when no water market exists by, for example, measuring the value water adds as an essential input factor in production processes. In this case, the economic value of water is derived, indirectly, through more or less complex production function approaches, where output is a function of labour, man-made capital, intermediate input and natural capital including the water resource. Regressing output on these input factors yields a marginal value for water use. This marginal value can be used to estimate the total economic value of water use simply by multiplying it with the quantity of water use. More complex integrated water modelling approaches also exist, where limited water availability and other key water system characteristics (e.g. water balance, water flows, seasonal variation etc.) are coupled with and integrated in economic production and consumption functions, yielding shadow prices and opportunity costs of different existing and/or future water use patterns.

Sometimes, investments by public (especially government) bodies in water resources may represent a surrogate for aggregated individual willingness to pay and hence social value. These ‘public prices’ paid for resources have been used to approximate the value society places upon them, as for instance the costs of designating a wetland ecosystem as a nature reserve. For a variety of reasons, these are unlikely to accurately reflect aggregated individual values, although techniques exist for attributing economic value based on such ‘collective choice’ decisions.

Following on from the discussion of valuation techniques in the first part of this chapter, we now look at the applicability of the various techniques to value water resources from aquatic ecosystems. All valuation techniques have strengths and weaknesses (as shown in the summary tables in Annex 2), and the decision on which technique to use for a particular application requires experience and judgment on the part of the analyst. As with most methods and techniques in economics or any other field, it is important to apply the techniques in their appropriate context and to be aware of the pitfalls involved.

Table 4.1 summarises the valuation techniques commonly used to value water resources and provides a brief description of which technique is most useful for what type of water use. In addition, table 5 shows the various aquatic ecosystem use outcomes of goods and services along with the change in welfare that they generate. Table 4.2 shows for the general categories of water functions/services the applicable valuation techniques used to evaluate changes in welfare. Annex 4 provides a more detailed discussion of the application of the possible methods and techniques for the valuation of outcomes according to the various classes of aquatic ecosystem functions.

Table 4.1: Valuation methodologies relating to water

Valuation Method	Description	Value Measure	Method useful for which type of water use
Market based transactions	Observed prices from transactions for leases or sales of water rights. May require shadow pricing	Depending on study: Marginal value based on price; not Max WTP; sometimes average value;	At source or at site WTP for agricultural, industrial, municipal and environmental uses
Derived demand functions	Econometric procedures used to derive value from household's or firm's inverse demand function based on observations on water use behaviour.	Marginal value, net average value and gross average value depending on study	At site demands for municipal uses
Production and cost functions	Water treated as an input into the production of other marketed goods. Econometric analysis used to relate output or cost of production of marketed good to water inputs.	Estimates Producers surplus which can be converted to net average value	At site values for agricultural and industrial production uses
Residual imputation/ change in net rents/ value added	Makes use of budget analysis of value added measure from input-output models to estimate net producers' income or rents attributable to water/increment of water. Water treated as one input into the production of other goods. Calculate the total returns and subtract off all non-water related expenses.	Estimates Producers surplus which can be converted to net average value; Marginal value also.	At site or at source valuation of offstream intermediate agricultural and industrial uses
Mathematical programming and optimisation	Constructed residual models for deriving net producers' rents or marginal costs attributable to water via fixed price optimisation models	Calculate shadow price or marginal values for all constraints including water; Optimisation models estimate marginal values based on 'optimal' allocation of water.	At site or at source valuation of offstream intermediate agricultural and industrial uses
Choice Modelling	Construction of a hypothetical market by direct surveying of sample of individuals to make choices among alternative proposed policies	Average, Marginal or Total value can be estimated depending on purpose of study	At source valuation of environmental uses and at site valuation of residential water uses
Hedonic Price Method (HPM)	Derive an implicit price for an environmental good from analysis of goods for which markets exist and which incorporate particular environmental characteristics.	Marginal value if second stage of analysis undertaken, otherwise average value of water derived	At source demands for change in water quality/quantity revealed by transactions in residential/farm properties.

Travel Cost Method (TCM)	Costs incurred in reaching a recreation site as a proxy for the value of recreation. Expenses differ between sites (or for the same site over time) with different environmental attributes.	Usually consumer surplus based and hence average value of water; sometimes marginal value	Valuation of recreational uses and derived at source valuations for changes in water supply
Contingent Valuation (CVM)	Construction of a hypothetical market by direct surveying of a sample of individuals to state WTP amount for proposed policy	Average, Marginal or Total value can be estimated depending on purpose of study	At source valuation of environmental uses and at site valuation of residential water uses
Defensive Expenditures/ averting behaviour	Costs incurred in mitigating or avoiding the effects of incurring an external cost. Frequently represents only a lower bound measure of benefits of policies from reducing the externality.	Frequently a lower bound estimate of WTP; Marginal or Average value depending on nature of study	Valuation of reduced water pollution from biological or chemical contaminants
Replacement/ Restoration Costs/cost savings	Potential expenditures incurred in replacing the good/service that is lost; for instance by the use of next best alternative; Costs of returning the degraded asset to its original state.	Net average value based on market price of replacement; Can be used as proxy for Marginal value	At site or at source valuation of intermediate agricultural and industrial goods and instream services such as hydropower and transportation. Also for household uses

Adapted from Turner et al. (2004) and Young (2005)

Table 4.2: Impacts of aquatic ecosystem use outcomes of goods and services on human welfare and applicable valuation techniques

Outcomes of Goods and Services Provided	Effect on Economic Value
Potable water for household use	Change in welfare from change in availability of potable water. Change in human health or health risks.
Water for landscape maintenance and peat soil	Change in cost of maintaining public or private property.
Water for crop irrigation	Change in value of crops or production costs. Change in human health or health risks.
Water for livestock consumption	Change in value of livestock products or production costs. Change in human health or health risks.
Water for food processing	Change in value of food products or production costs. Change in human health or health risks.
Water for other manufacturing processes	Change in value of manufactured goods or production costs.
Cooling water for power plants	Change in cost of electricity generation.
Water transport	Change in economic output.
Prevention of saline intrusion	Change in cost of maintaining public or private property
Water/soil support for prevention of land subsidence	Change in cost of maintaining public or private property
Natural erosion, flood and storm protection	Change in cost of maintaining public or private property
Shoreline stabilization	Change in cost of maintaining public or private property
Sediment removal	Change in human health or health risks. Change in animal health or health risks. Change in economic output or production costs.
Transport, treatment and medium for wastes and other by-products of human activities	Change in human health or health risks. Change in animal health or health risks. Change in economic output or production costs.
Improved air quality through the support of living organisms	Change in human health or health risks. Change in animal health or health risks.
Biological diversity provision	Change in quantity or quality of recreational activities. Change in human health or health risks.
Recreational swimming, boating, fishing hunting, trapping and plant gathering	Change in quantity or quality of recreational activities. Change in human health or health risks.
Commercial fishing, hunting, trapping and plant gathering	Change in value of commercial harvest or costs. Change in human health or health risks.
Energy production	Change in economic output

On and off site observation and study for leisure, education and scientific purposes.	Change in quantity or quality of on/off site observation or study activities
Micro-climate regulation, Macro-climate regulation	Change in human health or health risks. Change in animal health or health risks. Change in economic output or production costs.
Toxicant removal,	Change in human health or health risks. Change in animal health or health risks.
Toxicant export	Change in human health or health risks. Change in animal health or health risks.
Cultural value provision	Change in personal utility or well-being
Historical value provision	Change in personal utility or well-being
Aesthetic value provision	Change in personal utility or well-being
Wilderness value provision	Change in personal utility or well-being
Other non-use provision	Change in personal utility or well-being

Table 4.3: Water uses and applicable valuation methods

Water function/service flow	Applicable valuation technique
Agricultural use	DD; PF; RI; RCS; OM; MT
Fisheries use	DD; PF; RI
Municipal use (Includes municipal domestic, rural domestic, commercial, and institutional water uses)	MT; DD,; ABC; RCS; ABC; CVE;
Industrial use	DD; PF; RI; RCS; OM
Commercial use	DD; PF; RI; RCS
Recreational use	TCM; HP; CVE
Ecological use	PF; RCS; ABC; CVE
Buffer use	OM; RCS; ABC; CVE
Subsidence Avoidance use	PF; HP; RCS; ABC; CVE
Existence values	CVE
Bequest values	CVE

Key: (MT = Market Transactions; DD= Derived Demand; OM = Optimisation Models; RI = Residual Imputation; PF= Production/Cost Function; RCS = Replacement Cost/Cost Saving; CVE = Contingent Valuation/Choice Experiments; TCM = Travel Cost/Random Utility Model; ABC = Averting Behaviour/Avoidance Costs; HP = Hedonic Price)

4.2 *Choosing between valuation methods*

Some of the general points to consider when choosing a valuation technique are shown in Annex 3. Nevertheless, the choice of technique when considering water resources will most frequently be related to the following issues:

- Type of water use outcome/good or service to be valued – some techniques are more appropriate to certain uses than other techniques (see Table 4.1 and 4.2).
- Type of values to be estimated – whilst use values are estimated by all of the various techniques, non-use values can only be estimated by stated preference methods.
- The purpose of the valuation - certain purposes will require valuation techniques based on the estimation of marginal values, whilst others will require values based on estimating total values as given by consumer (or producer) surplus. The various types of values (average, marginal, total, etc) generated by the techniques are shown in Table 4.1
- Data and information availability – existence of data from existing/secondary sources. In general, cost data will be more readily available than benefit data (though see the section on use of cost based approaches below).
- Accuracy of results required – the degree of uncertainty surrounding the outcome of different methods differs significantly and may be decisive when choosing a specific method for a specific purpose. For example, the acceptable level of uncertainty is much higher in a pre-feasibility cost benefit appraisal than when wanting to establish the 'correct' water price level based on current levels of cost recovery.
- Resources and time available to undertake the study – the various techniques can differ greatly in terms of the resource and time required to undertake a study.

All valuation techniques have strengths and weaknesses, and the decision on which to use for a particular application requires experience and judgment on the part of the analyst. Some further general points for the analyst to consider when making this choice are set out below.

First, it is often possible to use more than one valuation technique and compare the results. All methods involve some uncertainty; if the analyst has multiple estimates, he or she will have greater confidence in the value of the proposed change. Several of the valuation techniques typically use data from a household survey (for example contingent valuation, travel cost and hedonic property pricing methods). When a technique requires that primary data be collected with a household survey, it is often possible to design the survey to obtain the data necessary to undertake more than one valuation method. Household surveys are required for contingent valuation, opportunity cost and travel cost studies. Such surveys need to be designed with the goal of producing value estimates using multiple methods.

Second, different valuation techniques may measure different things. In this sense they should be considered as complimentary rather than competing tools. For example, the contingent valuation method and choice experiments are the only available technique for measuring non-use (or passive use) values. Suppose that estimates of use value of a national park and wildlife reserve were obtained using a travel cost model and estimates of non-use value were obtained from a contingent valuation survey. These value estimates are not substitutes for one another; both may be useful for policy makers. Similarly, revealed preference methods measure the perceived benefits to individuals; they do not capture the value of effects of which people are unaware. For example, if individuals do not know that a cancer-causing substance is in their drinking water, they obviously will not take action to avoid this risk. There will thus be no 'behavioural trail' that an analyst can follow to determine how much they would be willing to pay to avoid such a risk. However, using the damage function approach, an analyst could estimate the reduced cancer deaths that would result if the carcinogenic substance were to be removed from the water supply.

Third, it is important to consider the needs of the user(s) of valuation studies. In some cases clients have preferences for the use of one valuation technique over another. For example, estimates obtained from travel cost or hedonic property pricing methods may be considered too theoretical or too complex. A particular client may feel that contingent valuation estimates are too subjective and unreliable to support policy debate and discussion. The analyst carrying out policy work must be sensitive to such concerns.

Fourth, the analyst should consider not only the client's needs, but also the needs of the public. Information elicited on people's values for environmental improvement is often of great interest to a wide variety of groups in society. In choosing a valuation technique, thought should be given to how the information obtained will be received by the public and interested parties other than the immediate client. Information from valuation studies could be used in a 'top-down' hierarchical planning process or it could contribute to democratic dialogue or a participatory political process. A technique such as contingent valuation bears a resemblance to a referendum or voting process. Whereas the final decision on a policy or project may not be determined by an election, the process of eliciting information on people's preferences involves a certain degree of participation in decision-making. Analysts need to be aware of the consultative nature of the valuation task and sensitive to the political implications. They should choose techniques that inform and facilitate public debate. One useful step is to hold public hearings or meetings with local community leaders to explain the findings of valuation studies.

Fifth, the cost of carrying out a valuation study or set of studies must be weighed against the value of the information in helping to make a better policy or project decision. Clearly more money could be spent on a valuation study than a policy decision warrants. But it is also important to keep in mind that many policies and projects have large-scale environmental implications that extend far into the future. In this case there is a substantial risk that too little money will be spent on the use of valuation techniques.

4.3 *Value transfer*

An alternative to carrying out a new original valuation study is to use existing economic value estimates from previous studies – so called 'benefits transfer'. This technique applies the results of previous environmental valuation studies to new policy or decision-making contexts. In the literature, benefits transfer is commonly defined as the transposition of monetary environmental values estimated at one site (study site) to another site (policy site). The study site refers to the site where the original study took place, while the policy site is a new site where information is needed about the monetary value of similar benefits. The most important reason for using previous research results in new policy contexts is that it saves a lot of time and money. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to inform decision-making. The decision of whether to undertake an original study or to use existing value estimates can be considered in terms of the acceptability of errors produced by benefits transfer and the level of precision sought, i.e. the purpose of the study and when transfer errors may be too big for this purpose (see Box E).

Box E. Transfer Errors in the Value Transfer Literature

Although benefits transfer is used extensively in practice, relatively little published evidence exists about its validity and reliability. The table below gives an overview of water related studies, which tested the reliability of the transfer of WTP values. Although not complete, the table shows that most studies test the reliability of transferring contingent values. Three studies investigate the transferability of travel cost studies. The estimated benefits in these studies are related to different types of water use, such as recreational fishing, boating or other recreational water use. Also the study by Bergland et al. (1995) and Parsons and Kealy (1994) look at water quality improvements for recreational use. The last column presents the range of transfer errors found in these studies. So, a transfer error of 50% means that transferring the value from a study site to a policy site can be 50% higher or lower than the 'true' value at the policy site. The range of transfer errors refers to the transfer of average WTP values and WTP functions.

Study	Estimated benefits	Transfer errors (%)
Loomis (1992)	sport fishing benefits	5 – 40
Parsons and Kealy (1994)	water quality improvements	1 – 75
Loomis et al. (1995)	water based recreation	1 – 475
Bergland et al. (1995)	water quality improvements	18 – 45
Downing and Ozuna (1996)	saltwater fishing benefits	1 – 34
Kirchhoff et al. (1997)	white water rafting benefits	6 – 228
Morrison and Bennett (2000)	wetlands	4 – 191
Rosenberger and Loomis (2000)	water recreation	0 – 319
VandenBerg, Poe and Powell (2001)	water quality	0 – 298
Barton (2002)	beach bathing water quality	11-26
Barton and Mourato (2003)	beach bathing water quality	9-129
Brouwer and Bateman (2005)	flood control benefits	4 – 51

Table E : Errors found in water related economic valuation studies testing benefits transfer

From Table E it is difficult to say how large the errors can be expected to be when using existing economic value estimates in new decision-making contexts. In some cases they can be very low, in other cases they can be as high as almost five times the value, which would have been found if original valuation research was carried out. No distinct differences can be found when comparing transfer errors for contingent valuation and travel cost studies.

The errors reported in Table E have to be considered in the light of the purpose for which the user (policy or decision maker) wishes to use previous valuation results. In some cases a transfer error of 50 percent may be considered too high, in other cases such an error may be acceptable. The acceptability of the error will depend on subjective judgement by the user, the purpose and nature of the evaluation (e.g. CBA, pricing/cost recovery, environmental liability) and the phase of the policy cycle in which the evaluation is carried out.

Three important sources of error can be distinguished:

i) the error incurred when estimating the original unit value;
ii) the error incurred when transferring the original unit value to the new policy context;
iii) the error incurred when aggregating the transfer unit value to the whole population of beneficiaries and calculation of the TEV.

Source: adapted from Brouwer (2000) and Rosenberger and Loomis (2003).

Several necessary conditions should be met to perform effective and efficient benefit transfers (Desvousges et al., 1992).

First, the policy context should be thoroughly defined, including:
1. Identifying the extent, magnitude, and quantification of expected site or resource impacts from the proposed action in both quantity and quality terms.
2. Identifying the extent, spatial distribution and characteristics of the population that will be affected by the expected site or resource impacts.
3. Identifying the data needs of an assessment or analysis, including the type of measure (unit, average, marginal value), the kind of value (use, non-use, or total value), and the degree of certainty surrounding the transferred data (i.e., the accuracy and precision of the transferred data).

Second, the study site data should meet certain conditions for critical benefit transfers:
1. Studies transferred must be based on adequate data, sound economic method, and correct empirical technique.
2. The study contains information on the statistical relationship between benefits (costs) and socioeconomic characteristics of the affected population.
3. The study contains information on the statistical relationship between the benefits (costs) and physical/environmental characteristics of the study site.
4. An adequate number of individual studies on a recreation activity for similar sites have been conducted in order to enable credible statistical inferences concerning the applicability of the transferred value(s) to the policy site.

And third, the correspondence between the study site and the policy site should exhibit the following characteristics:
1. The environmental resource and the change in the quality and/or quantity of the resource at the study site and the resource and expected change in the resource at the policy site should be similar. This similarity includes the quantifiability of the change and possibly the source of that change.
2. The markets for the study site and the policy site are similar, unless there is enough usable information provided by the study on own and substitute prices. Other characteristics should be considered, including similarity of demographic profiles between the two populations and their cultural aspects.
3. The conditions and quality of the recreation activity experiences (e.g., intensity, duration, and skill requirements) are similar between the study site and the policy site.

Study quality is an important criterion, which can be assessed in a number of ways. Above all, one can look at the internal validity of the study results, that is, the extent to which findings correspond to what is theoretically expected. This internal validity has been extensively researched over the past three decades in valuation studies. Studies should contain sufficient information to assess the validity and reliability of their results. This refers, among others, to the adequate reporting of the estimated WTP function, including the applied statistical techniques and the definition of variables. Desvousges et al. (1998) suggest the following criteria for assessing the quality of a study site values for use in benefits transfer:

- *Scientific soundness* **-** data collection procedures, empirical methodology, consistency with scientific or economic theory, statistical techniques;
- *Relevance* **-** change in environmental quality, baseline environmental quality, affected services and commodities, site characteristics of affected commodity, duration and timing of effects, exposure path and nature of health risks, Socioeconomics characteristics of the affected population, property rights;
- *Richness in detail* **-** definition of variables and means, treatment of substitutes, cost of time (in Travel cost studies), participation rates ("extent of market" i.e. number of affected people)

Thus, while benefit transfer provides a quick and cheap alternative to original valuation research, some conditions must be met if it is to provide reliable results. Above all, the local circumstances and conditions in the new decision-making context need to be close enough to the ones prevailing in the original research. The risk of obtaining misleading results may be controlled and reduced by integrating more explanatory variables into the transfer. However this also increases the data requirements and the complexity of the analysis. Also, the possibilities of conducting a sound and reliable benefits transfer hinge on the number, quality and diversity of valuation studies available – the larger, the better and the more diverse the existing set of studies is, the more likely there will be a primary study that is close enough to the policy site for results to be transferable.

4.4 *Hydro-economic modelling*

The complexity of interactions between water and the economy can also be captured through formal, mathematical models linking relevant hydrological and biogeochemical structures and processes to economic 'laws' of supply and demand underlying the provision of scarce water services. Historically, these models have been developed by hydrologists and civil engineers, focusing on single and multiple objective decision-making and trade-offs (e.g. Dudley, 1972; Braat and Lierop, 1987; McKinney and Cai, 1996; Andreu et al., 1996; Rosegrant et al., 2000; Cai et al., 2003). Being the largest freshwater consumer in the world (FAO AquaStat, 2007), agriculture (irrigation) has been the prime focus of many of these models, going back to the 1960s when resource economists developed the first optimal control groundwater (demand management) models for irrigated agriculture (Burt, 1964, 1966). These models often include a detailed hydrological module - and in some cases also hydraulic and biogeochemical modules - to control for the hydro-geological heterogeneity in a basin area. Node networks are typically used as graphical delineations of water flows and stocks in a watershed or river basin into different water quantity and/or quality balance (monitoring) stations linked to specific water demand and supply. For each geo-referenced node, a water demand and supply function is estimated based on the geographical unit's hydro-geological and biogeochemical characteristics. In the case of agriculture, the demand and supply functions are for example based on an agronomic model, such as a crop yield function, which depends on factors like soil, crop acreage, rainfall, crop evapo-transpiration and irrigation system characteristics. Economic behavior is usually included through a profit maximization objective function, where fixed and variable production costs are subtracted from the yield benefits subject to the natural resource constraints such as land and water availability. The latter is obviously dependent on the hydro-geological conditions involved, including water supply and water quality constraints.

The key to integrated hydro-economic modelling is that water systems perform economic functions, they can be used as a source and a sink for socio-economic activity, and hence have economic value. Usually after some degree of transformation, water can be used as a source for economic consumption like drinking and recreation, and in economic production as an input factor in crop and food production, energy, paper, or metal production. At the same time, water is also used as a sink for the negative by-products of economic production and consumption processes such as the emission of polluting substances into surface and groundwater bodies. The interaction between the hydrological and economic realm works both ways: water is transformed for economic use and the impact of economic use on water availability and quality consequently has implications in both the short and long term for the transformation process to modify water for economic use.

In the literature, a distinction is made between two different approaches to integrated hydro-economic model development, i.e. (1) models which allow for an effective transfer of information from one component to the other: the compartment or modular approach and (2) the holistic approach based on one integrated model (Braat and Lierop, 1987). In the modular approach a connection is built between the hydrological and economic model, and output data from one module usually provides the necessary input for the other[3]. In principle, the modules operate independently of each other and systems of equations are solved in an exogenous way (input variables from one model into the other are exogenous). In holistic models, variables that are exogenous in a modular approach are solved endogenously in a system of equations (Cai and Wang, 2006).

Under the modular approach, a loose connection exists between the different hydrologic and economic components. The various sub-models can be very complex and the main problem is to find the right transformation of data and information between sub-models. In the holistic approach there is one single unit with both the hydrologic and economic component tightly interwoven in a consistent endogenous model. In order to be able to solve the complexity of simultaneous equations the different components have to be represented in a very simple way (McKinney et al., 1999). So, whereas information transfer between the various compartments or sub-models is one of the most important technical obstacles in the modular approach, the most important issue in the holistic approach is to find one single technique and denominator for the variable quantities and represent both the simplified hydrological and economic component in a meaningful way.

Linking hydrological and economic systems through holistic or modular models raises a number of important methodological and operational (programming-technical) issues and challenges. McKinney et al. (1999) identify the following limitations to integrated hydro-economic modelling:

- Hydrological models are often based on simulation techniques, whereas economic models usually use optimization techniques.
- Different spatial scales: water bodies, watersheds or basins usually are the geographical unit in hydrological models, while economic models often refer to administrative boundaries of a region (county, province, state) or a country as a whole.

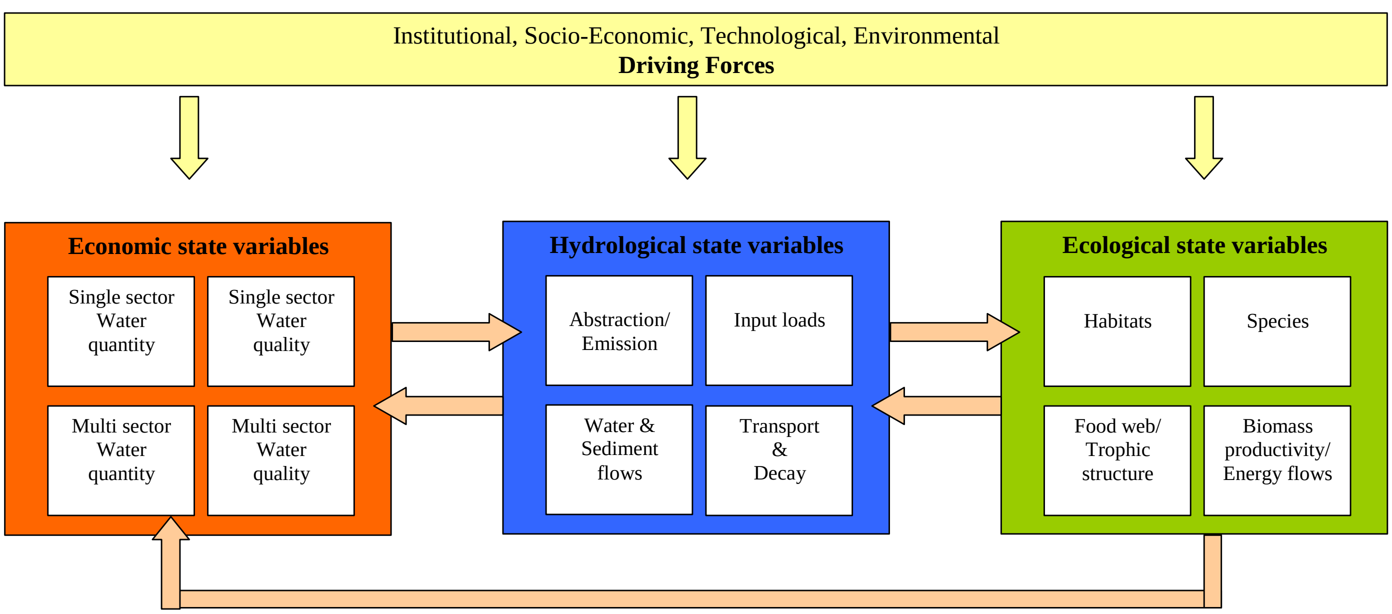
[3] The term hydrological is used here to mean both water quantity flow models and bio-geophysical water quality and water allocation models.

- Different time intervals and horizons: time scales in hydrological models often refer to days, months or seasons (summer and winter), while in economic models the time scales (intervals and horizon) are usually longer than that (years).

In practice, most hydro-economic models are based on a simple economic optimization algorithm subject to detailed surface and groundwater flow processes and their impact on one or multiple economic sectors, i.e. starting from the middle hydrology block in Figure 1 where the arrow points to the left to the economics block. Although they are driven by certain institutional and/or economic forces, their main focus is on the water system and the effect of for example water allocation problems on economic sectors. They are based on detailed node networks of water and substance balances throughout the river basin, linked to an economic activity through a demand function. This demand function often depends on fixed (exogenous) technical input-output parameters of the economic production process involved (e.g. irrigation demand from agriculture), and reflects at best a partial economic equilibrium system of demand and supply equations.

Few integrated hydro-economic models exist which are primarily driven by economic conditions and trade-offs and their impact on water system variables through water extraction and/or emissions to surface and groundwater bodies, i.e. starting from the left-hand side economics block in Figure 4.4. These models are based on economic demand (consumption) and supply (production) functions, which are related to different forms of water use where water is an essential input in consumption and production processes. Examples include input-output models of direct and indirect water use (e.g. Velázquez, 2006; Okadera et al., 2006).

Figure 4.4: Disciplinary dimensions underlying integrated hydro-economic modelling



Even less models exist which focus on the effect of changes in eco-hydrology state variables on economic starting point conditions and economic adaptation and mitigation processes, i.e. the feedback arrow from hydrology and/or ecology to the economic system in Figure 4.4. These models incorporate changes in the water system and their effect on the economic system. Most of these types of models are partial economic equilibrium models, based on production function approaches, where water is one of the input factors and changes in the availability or quality of water in the production process at hand is assessed through estimated physical dose-effect relationships, which are related to market prices in order to arrive at a marginal value of water use.

Finally, sometimes ‘meta-models’ are distinguished as a separate class of modelling tools. Generally, meta-models are used to construct and develop general frames around specific problems analyzed with the help of a variety of data, expert judgments and models. Meta-models integrate simulation results from sub-models (e.g. an economic optimisation and a water quality simulation model) in a cause-effect framework. They are called meta-models because they only include the results of underlying models, where underlying model response surfaces are summarised for example in conditional probability distributions. They lie somewhere between the holistic and modular approach. The bio-geophysical and economic components are linked, but the model is usually not solved or optimized simultaneously.

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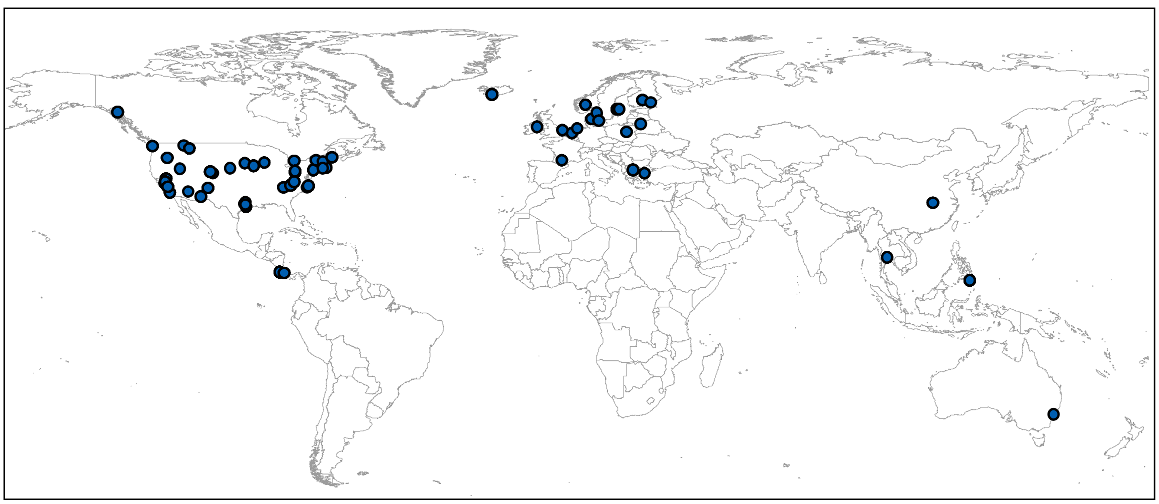
5. Meta-analysis of non-market values for water services

5.1 *Introduction*

Existing non-market valuation studies were collected and their results synthesized in a meta-analysis. The results presented here focus on contingent valuation studies of different ecosystem services provided by water resources, including the services which are part of the Resources For the Future (RFF) water quality ladder (Vaughan, 1986; Mitchell and Carson, 1989; Carson and Mitchell, 1993) such as boating, fishing, swimming and drinking (see Chapter 7 for a more detailed discussion of water quality ladders).

154 contingent valuation studies were collected and reviewed that estimate values for ecosystem services related to surface water quality. The studies were published between the years 1981 and 2006. The locations of study sites are largely found in North America and Europe with a small number in Central America, South East Asia, China and Australia. The geographic distribution of study sites is presented in Figure 5.1. There is clearly an absence of available value information for large areas of the world, particularly South America, Africa and most parts of Asia.

Figure 5.1: Locations of the water valuation study sites



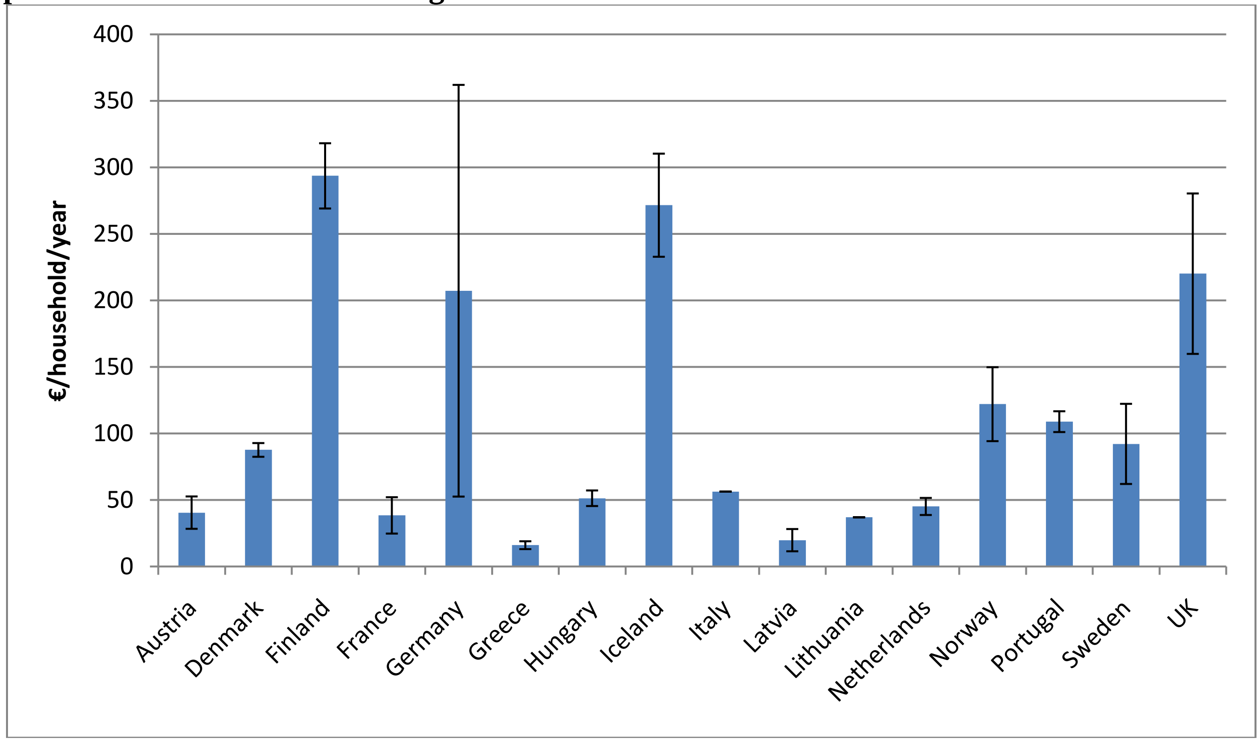
In terms of ecosystem services valued in this literature, both use and non-use values are well represented in the data. Table 5.1 presents the number of value observations for the set of ecosystem services included in the data. Many of the value observations are for multiple ecosystem services. The ecosystem service that is best represented in the data is non-use value related to preservation or improvement in water quality unrelated to any current or potential future use of the resource. Most contingent valuations comprise an element of non-use value in combination with use values, but it is difficult to assess the size of this component of total value. Direct use values related to water are also well represented in the studies. These are mainly related to recreational activities. Provisioning services such as drinking water and irrigation are less well represented. This arguably reflects the priorities for water use at the locations where valuation studies have been conducted.

Table 5.1: Water service categories valued in lakes and rivers studies.

Water service	Number of observations	Average value (2007 USD/household/year)	Standard deviation
Drinking water	17	382	287
Irrigation	3	185	111
Nature conservation	80	228	237
Fishing	151	89	100
Boating	128	76	66
Swimming	119	139	212
Walking	10	209	114
Other recreation	29	237	230
Health	4	513	283
Amenity	21	135	88
Non-use	275	129	155

As expected, there is considerable variation in mean values across ecosystem services, with services related to health and drinking water receiving the highest mean values and services related to boating and fishing receiving the lowest. The standard deviations of all mean value estimates are high, indicating considerable variation in values within each category of ecosystem service. This is especially the case for 'swimming', 'nonuse', and 'fishing' where the standard deviation is higher than the average value. For some ecosystem services the number of observations is very low (e.g. irrigation and health).

Figure 5.2: Average willingness to pay values across European countries for water protection and conservation in general

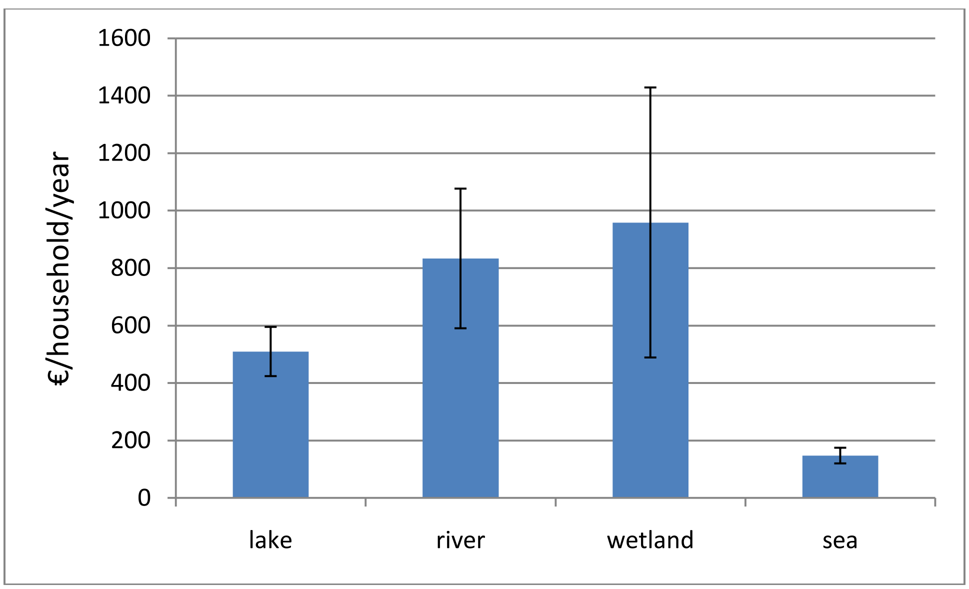


A third of the total number of observations relates to European valuation studies. On average and irrespective of the specific water service, a European household is willing to pay 105 Euros per household per year for additional water conservation. However, this value varies significantly across European countries, with the highest average values found in North Europe (Finland, Iceland and the UK), and the lowest average values in Greece and the Baltic States (Latvia and Lithuania) (Figure 5.2). The available WTP values display especially a relatively large variation in Germany as can be seen from the size of the standard error also shown in Figure 5.2.

WTP is expected to depend on a variety of factors, including (1) the characteristics of the water bodies the public were asked to value in the available studies, (2) the characteristics of the population of beneficiaries who were asked to value the change in water provision and quality, and (3) the characteristics of the method used to elicit the contingent values.

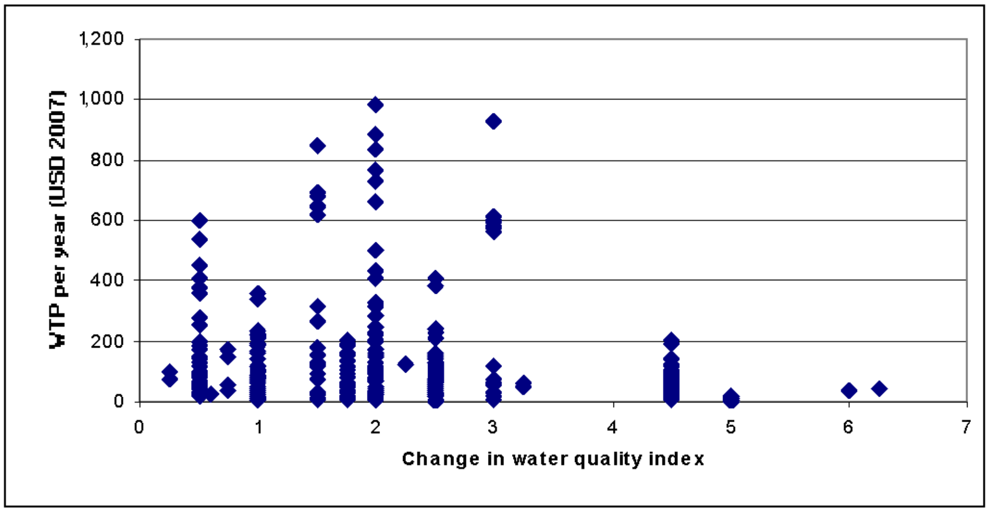
Average values differ significantly for the different water types in which the environmental changes were valued (Figure 5.3). Wetlands were valued significantly higher than rivers and rivers higher than lakes. However, the variability in wetland values was relatively high. A limited number of studies involved sea water quality changes (primarily changes in bathing water quality), which received the lowest average value.

Figure 5.3: Average willingness to pay values for different water body types



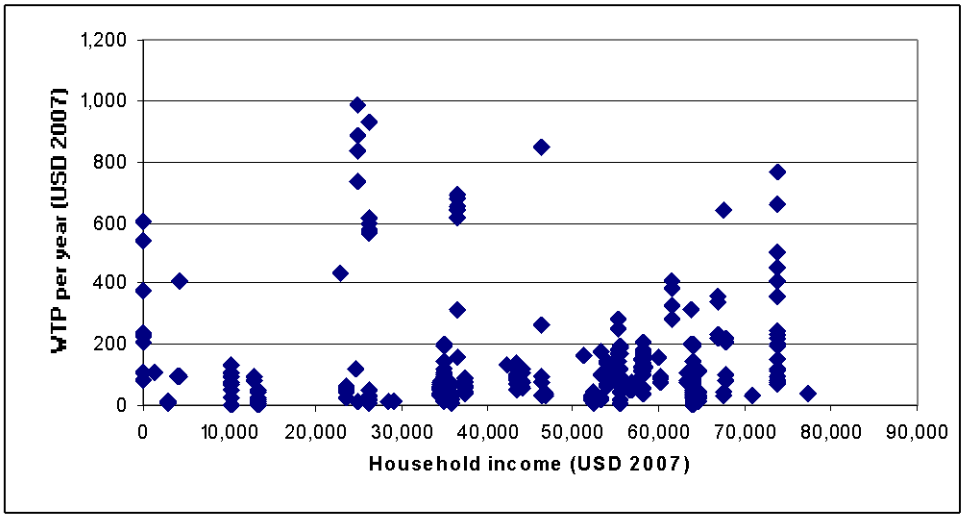
The studies screened here followed very different reporting formats for both the valuation design and the valuation results. In many studies, important information about specific study characteristics was missing. The most important missing information was a clear description of the change in water quality that could be translated into a standardised measure. The expectation is that the stated WTP for a change in water quality is partly dependent on the magnitude of the proposed change in quality. For those studies where information was available about the particular change in water quality away from the baseline level, this information was converted into the 10-point water quality index used by Van Houtven et al. (2007). This was only possible for 54 studies (35% of the identified total number of studies). The description of water quality in these studies was standardized to this 10 point index. The index is based on the water quality ladder developed by Vaughan (1986), which translates descriptions of water quality in terms of associated suitability for a number of activities into a 10 point index. For example, water quality that is considered "boatable" has a water quality index value of 2.5, "fishable" water has a value of 5.1, and "swimmable" water has an index value of 7.0. The mean value for a 1 unit change in the 10-point water quality index is 89 USD per household per year. The median value is considerably lower at 35 USD. Figure 5.4 presents WTP per year by the change in water quality. A clear relationship between the two is not evident.

Figure 5.4: WTP per year by change in water quality index



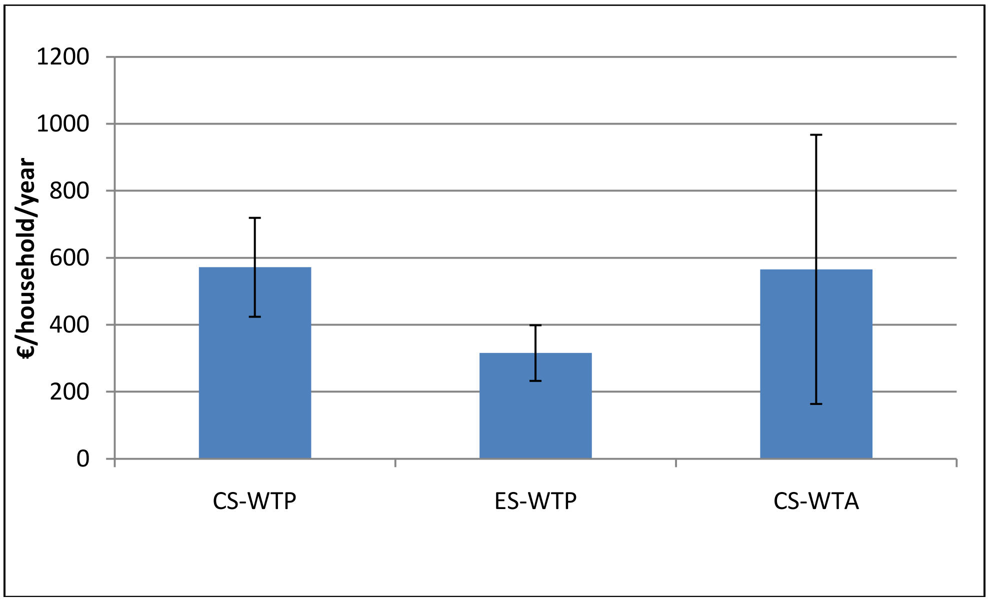
One of the most important population characteristics in WTP studies is ability to pay. What people say they are willing to pay can in principle not exceed what they are able to pay. The relationship between average WTP and average household income in the different studies is presented in Figure 5.5. As expected, higher WTP values seem to go hand in hand with higher income levels.

Figure 5.5: WTP per year by average household income



Another important finding is that the way non-market values are elicited has an important influence on the size of the values. WTP values can refer to a change that secures a future welfare gain or that prevent a future welfare loss. As expected, these two different WTP values are significantly different from each other. People are willing to pay almost twice as much for a welfare gain than to prevent a welfare loss. An important reason for this may be that people do not like to pay extra for something that they consider theirs already. On average across all studies investigated, public willingness to accept a welfare loss is the same as public willingness to pay to secure a welfare gain.

Figure 5.6: Average values for different welfare measures



Other relevant results include the effect of the survey method (Figure 5.7), elicitation format (Figure 5.8) and payment vehicle on stated WTP (Figure 5.9). Face-to-face interviews generate, on average, significantly higher WTP values than mail or telephone surveys. Open-ended (OE) and payment card (PC) CV studies generate on average significantly lower WTP values than the in the CV literature recommended dichotomous choice (DC) or iterative bidding (IB) procedures (Arrow et al., 2003). Finally, no significant differences can be detected between using the existing water bill or income taxation as the payment mode in existing valuation studies. However, using a voluntary fund results on average in a significantly higher stated WTP.

Figure 5.7: Average WTP values for different survey modes

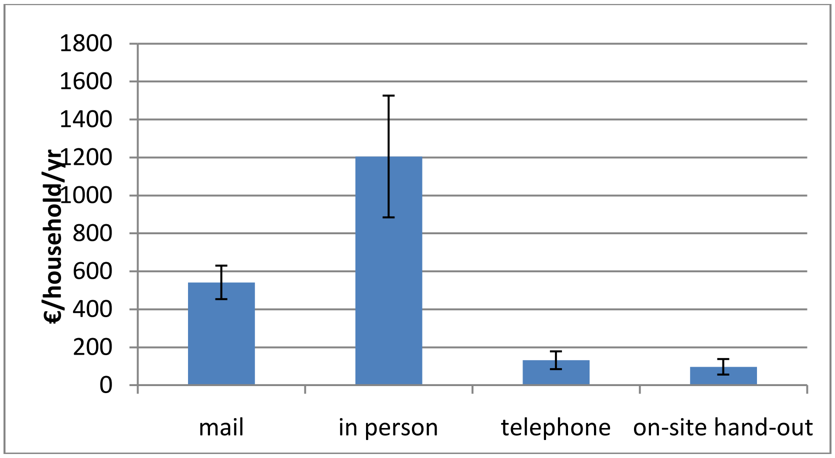


Figure 5.8: Average WTP values for different value elicitation procedures

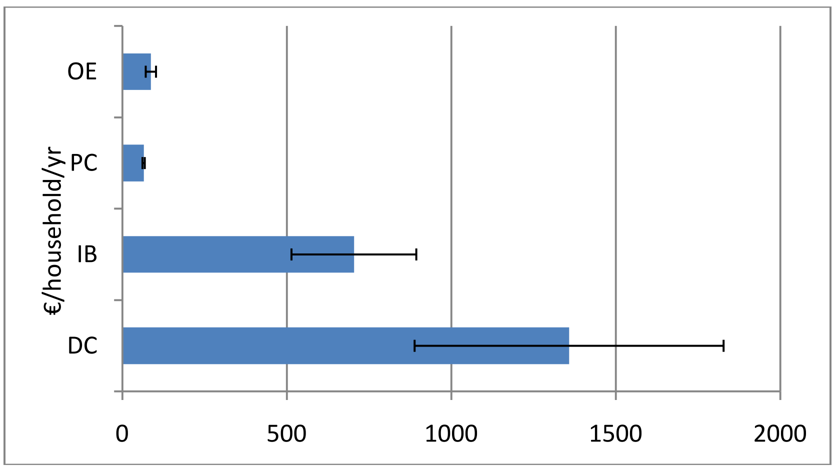
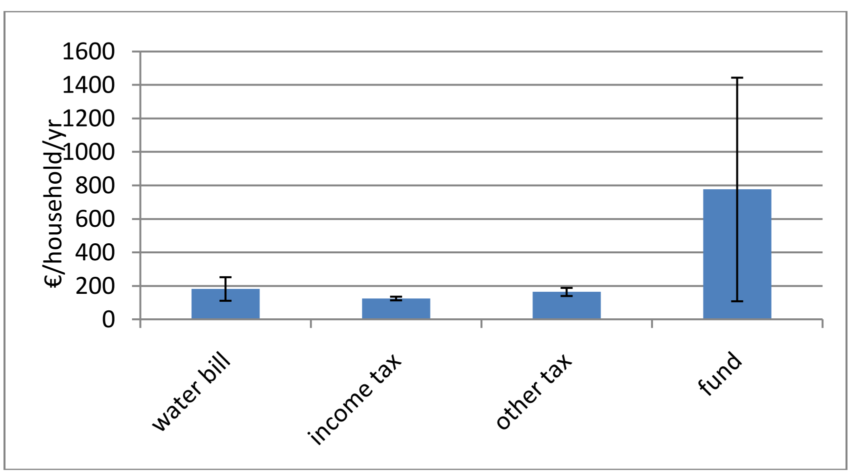


Figure 5.9: Average WTP values for different payment modes



5.2 *Meta-analysis results*

From the 54 selected studies that provide complete information on all the explanatory variables that we include in the statistical meta-analysis we are able to code 388 separate value observations. We therefore obtain multiple value observations from single studies, with an average of 7.2 observations per study. Separate value observations from a study were included if they represent different study sites, sample populations, ecosystem services, elicitation formats, or estimation methods. These characteristics of value observations are explicitly coded in the database and can be controlled for in the meta-analysis.

Value estimates from contingent valuation studies are reported in a wide variety of currencies, temporal units (e.g., per day, per visit, per month, present value etc.), and for different categories and groupings of beneficiaries (e.g., per person, per household, per visitor etc.). This value data was standardized to a common format, namely as willingness to pay (or accept) per household per year for a change in water quality in USD 2007 price levels. This is the dependent variable in the meta-analysis. Values that are originally reported as WTP per person have been multiplied by the average household size for the country in which the study is located. Values that are originally reported as WTP per other unit of time (e.g., day, month, lump sum) have been converted to annual values using appropriate multipliers or by calculating annualised values using time horizons and discount rates reported in the underlying study when available. Values reported in other currencies or for price levels other than 2007 have been converted to USD 2007 prices levels using PPP adjusted exchange rates and GDP deflators obtained from the World Development Indicators (World Bank 2010). Also value estimates of willingness to accept (WTA) compensation for negative changes in water quality are included.

The explanatory variables used in the estimation of the value function include two measures of water quality. These are the initial level of water quality and the change in water quality described in the contingent valuation scenario. The values for the majority of the explanatory variables are coded from information contained in the primary valuation studies themselves. In addition we used GIS to compute a number of spatially defined variables for each study site. These are the population within a 50km radius (capturing the number of potential beneficiaries within this vicinity), the area in hectares of lakes and rivers within a 50km radius (capturing the abundance of this ecosystem within the vicinity of the study site and therefore the scale of the good valued), and the length of roads within a 50km radius (capturing the accessibility of the study area). The source data for these variables are the Socio-Economic Data Center (SEDAC) Columbia University, Global lakes and wetlands database GLWD, and FAO - UN SDRN. The full set of explanatory variables used in the meta-regression are listed in Table 5.2 together with a short description and descriptive statistics for each variable.

The meta-regression model is estimated by ordinary least squares regression (OLS) using SPSS 16.0. The estimated model is presented in Table 5.2. The overall model fit is good. The adjusted R^2 value of 0.592 indicates that almost 60% of variation in the dependent variable is explained by variation in the explanatory variable included in the model.

The estimated coefficient on the water quality change variable is very small and positive but not statistically significant. One would expect the scale of change in water quality to positively influence willingness to pay for the change, but we do not find evidence of this. This may indicate insensitivity of respondents to the scope of change or a lack of understanding of the described change. The coefficient on the base water quality variable is negative and statistically significant at the 1% level, indicating that WTP for water quality improvement is lower when water quality is already high. The corollary is that the public is willing to pay more to improve bad water quality.

Table 5.2: Meta-regression results (dependent variable: WTP/household/year in 2007 USD)

	Dummy	Mean	Std. deviation	Coefficient estimate	Std. error	*p*<
Constant				-0.806	0.952	0.398
Water quality change	Abs value	2.288	1.261	0.003	0.047	0.949
Baseline quality level	Abs value	3.220	1.349	-0.349	0.042	0.001
Nonuse value	yes	0.707	0.456	0.528	0.103	0.001
Drinking water	yes	0.044	0.205	0.730	0.265	0.006
Irrigation water	yes	0.008	0.088	-1.207	0.480	0.012
Nature conservation	yes	0.206	0.405	0.222	0.156	0.156
Recreational fishing	yes	0.388	0.488	-0.282	0.125	0.025
Recreational boating	yes	0.329	0.470	-0.311	0.123	0.012
Recreational swimming	yes	0.306	0.461	0.154	0.112	0.170
Recreational walking	yes	0.026	0.158	-0.258	0.371	0.487
Amenity	yes	0.054	0.226	-0.811	0.271	0.003
Rivers	yes	0.470	0.500	-0.080	0.163	0.625
Lakes	yes	0.080	0.271	-1.154	0.225	0.001
Local scale	yes	0.126	0.332	-0.396	0.207	0.057
Regional scale	yes	0.522	0.500	-0.644	0.180	0.001
National scale	yes	0.229	0.421	-0.220	0.179	0.221
Welfare measure=CS	yes	0.740	0.439	0.283	0.142	0.046
In person interviews	yes	0.558	0.497	-0.231	0.233	0.322
Mail survey	yes	0.339	0.474	0.084	0.206	0.684
Telephone survey	yes	0.051	0.221	-0.299	0.327	0.361
Fund	yes	0.072	0.259	0.765	0.211	0.001
Income taxation	yes	0.136	0.343	-0.631	0.192	0.001
Municipal taxation	yes	0.129	0.335	0.514	0.198	0.010
Open-ended question	yes	0.265	0.442	-0.063	0.212	0.766
Dichotomous choice	yes	0.154	0.362	0.602	0.217	0.006
Iterative bidding	yes	0.375	0.485	0.226	0.208	0.279
Payment card	yes	0.152	0.359	0.425	0.232	0.067
Payment per day	yes	0.021	0.142	0.759	0.365	0.038
Payment per trip	yes	0.021	0.142	2.048	0.420	0.001
Payment per month	yes	0.136	0.343	1.252	0.199	0.001
Payment not specified	yes	0.023	0.151	-1.233	0.303	0.001
Household income	Nat log	10.651	0.596	0.504	0.087	0.001
Population density	Nat log	13.203	2.335	-0.049	0.035	0.161
Lake density	Nat log	4.005	1.761	0.057	0.034	0.091
Road density	Nat log	9.208	1.194	0.109	0.066	0.102
Adjusted R-square				0.592		
N				388		

Regarding the set of binary variables indicating the water services valued, we find that non-use values and drinking water uses tend to be higher valued than other services (the omitted category is other recreation activities). Irrigation, recreational fishing and boating, and amenity values have lower values.

The negative signs on the binary variables indicating lakes and rivers show that WTP for water quality improvement is lower for these types of water bodies than for other types (the omitted category is other surface water including sea and wetlands). The coefficient on the rivers variable, however, is not statistically significant.

The spatial scale of the valuation scenario described in the contingent valuation is found to influence estimated WTP. Local and regional scenarios are found to result in statistically significant lower WTP than international scenarios (the omitted category).

The estimated coefficient on the binary variable indicating the welfare measure compensating surplus is positive and statistically significant at the 5% level. The survey format used is found not to have a statistically significant influence on estimate WTP. The payment vehicle on the other hand does have a significant influence, with income tax resulting in lower WTP. Similarly the elicitation format has an influence on estimated WTP, with the dichotomous choice format producing higher values.

The temporal period for which payments are made (as described in the hypothetical payment scenario) has a strong influence on the WTP elicited. Shorter time periods (day, visit, month) are shown to result in statistically significant higher WTP than scenarios that used annual payments (the omitted category).

The positive sign on the income variable indicates that WTP increases with income. In other words, water quality is a normal good. Given the double log specification of the regression model, the coefficient in this case can be interpreted as the elasticity of WTP with respect to income. WTP for water quality improvement is found to be positively related but inelastic with respect to income.

The coefficient on the population variable is negative but not statistically significant. The variable measuring the area of lakes and rivers in the vicinity of each study site is positive and statistically significant at the 10% level, indicating that respondents are willing to pay more for water quality improvements in locations with a higher abundance of lakes and rivers. The estimated coefficient on the roads variable is positive, possibly indicating a positive accessibility effect, but is not statistically significant.

5.3 *GIS based value mapping*

In this section a methodology is described for spatial value transfer for water quality improvements using GIS and meta-analysis. The approach will be illustrated with an application to value water quality improvements in major European rivers. The results of this value transfer approach can be represented on maps in order to communicate the spatial distribution of benefits derived from improvements in water quality such as foreseen in the WFD.

There are a number of published meta-analyses of the economic valuation literature on water quality (Johnston et al., 2003; Johnston et al., 2005; Johnston et al., 2006; Van Houtven et al., 2007). These constitute the starting point for our analysis and provide a selection of specifications of water quality value functions. From the existing literature we selected to use the meta-analytic value function provided by van Houtven et al. (2007). This function provides a specification for explicitly valuing changes in water quality described using Vaughan’s (1986) common 10-point scale, which had, contrary to the results presented in the previous section, a significant effect on WTP. This function has been adapted to estimate values in Euros instead of dollars. We also amend the value function with a distance decay term taken from Bateman et al. (1999) in order to adjust values for the distance between beneficiaries and the water body under consideration in the analysis:

where WTP is annual WTP per household per year in 2003 Euros, WQB the initial (baseline) water quality measured in 10-point scale, WQC the change in water quality also measured in 10-point scale, Income average household income in thousands of 2003 Euros, and Km the kilometres distance from a water body.

In order to be able to apply the meta-function in GIS, spatial data are needed. The information sources for this GIS data are listed in Table 5.3, together with their basic characteristics and source information. In the next steps we describe how these source data were processed before it could be used in a spatial analysis that applies the value transfer function above. In these steps we also provide the most relevant details regarding the incorporation of the value transfer function in the GIS based spatial analysis. All GIS operations where carried out with ArcGIS software, version 9.3 from ESRI.

The main criteria for the spatial data selection have been:

- free of cost or within a reasonable budget
- pan-European extent and consistency
- availability of metadata/documentation describing data characteristics and quality aspects
- currency (less than 10 years old and limited age differences between datasets)
- scale and maximum scale differences (1:100.000 to 1:250.000)

These criteria where applied in a rather loose way as this study was in the first place aimed to finding a suitable method for value transfer to an European scale and to a lesser extent to directly obtaining reliable results. However, the biggest sources of error, uncertainties in spatial accuracy and attribute accuracy and so on in source data and methodology applied are (as far as possible) indicated and discussed.

Table 5.3: Spatial data sources

Nr	Data set	Download source/owner	GIS datamodel/ format	Coordinate system and extent	Scale/ resolution	Year
1	Administrative boundaries (NUTS 3)	ESRI maps & data 2008	Vector polygon / ESRI Shape	Geographic (WGS84) European Union	1:200 000	2005
2	Population density disaggregated with Corine land cover 2000 (inhab/km2)	European Commission - DG JRC, Ispra, 2007 http://dataservice.eea.europa.eu/dataservice/metadetails.asp?id=1018	Raster ESRI grid	Lambert Azimutal Equal Area (ETRS89) European Union	1:100 000 / 100 meter grid cells	2001
3	Europe NUTS 3 Gross domestic product (GDP) per capita in 2002	European Spatial Planning Observation Network (ESPON) http://www.espon.eu/mmp/online/website/content/tools/832/850/file_1525/04_wealth_and_production_nuts_3.xls	Tabel (MS Excel)	NUTS 3 administrative units European Union	NUTS 3 administrative units	2002
4	Large Rivers WISE, code REF1a, feature type a): Rivers with a catchment area > 50,000 km2)	WISE (Water Information System for Europe), European Environmental Agency (EEA) http://www.eea.europa.eu/data-and-maps/data/wise-large-rivers-and-large-lakes	Vector line / ESRI Shape	ETRS89 European Union	1:1.000.000	Date not provided
5	CCM - River and Catchment Database, v 2.1	European Joint Research Centre (JRC-EU) http://ccm.jrc.ec.europa.eu/php/index.php?action=view&id=24#	Vector line / ESRI Shape	ETRS89 European Union	1:250.000	2008
6	Waterbase rivers Measurement stations / quality	WISE (Water Information System for Europe), European Environmental Agency (EEA) http://www.eea.europa.eu/data-and-maps/data/waterbase-rivers-3/waterbase-rivers-quality	Personal Geodatabase, with XY coordinates of point locations measurement stations	ETRS89 European Union	1:250.000	1982-2007

Step 1: Selection of rivers

For the classification of large rivers in the member states of the European Union we used the Water Information System for Europe (WISE) from the European Environmental Agency (EEA)[4]. More specifically we used one of the WISE Reference GIS datasets that are based on data reported to the European Commission by Member States under Articles 3 and 5 of the Water Framework Directive. The specific layer we used is the feature layer 'Large Rivers' (code REF1a, feature type a): Rivers with a catchment area > 50,000 km2). Because this dataset has a low positional accuracy of 5000 meter (scale 1:10.000.000), mainly meant for cartographic background purposes, we used it in combination with the CCM2 database[5], which has a much higher positional accuracy. This database forms also the basis for the WISE river and lake data layers and is described in detail by Vogt et al. (2007a) and Vogt et al. (2007b). From this additional CCM database we selected the rivers corresponding with the previously selected large rivers in the WISE water database, using a table selection on the specific river names. We have controlled the completeness of this selection by overlaying both GIS layers and visually comparing them. Where river courses (e.g. for the Kemijoki river in Finland) or river naming was different we have chosen to select and combine the segments (using spatial selection, editing, merge and dissolve operations) and/or naming from the WISE water database, except for the cases where the courses didn't correspond with the rivers used in the WISE database with water quality measurements.

Table 5.4: Selected large European rivers and countries through which these rivers run

River	Country Code	River	Country Code
Danube	DE, AU, HU, HR, RS, BG, RO	Odra	CZ, PL, DE
Douro	ES, PT	Po	IT
Dvina	RUS, BY, LV	Rhine	CH, FR, DE, NL
Ebro	ES	Rhone	CH, FR
Elbe	CZ, DE	Sava	SLO, HR, BIH, CS
Guadalquivir	ES	Seine	FR
Guadiana	ES, PT	Tagus	PT
Kemijoki	FIN	Tisa	UA, RO, HU, CS
Loire	FR	Vistula	PL
Nemunas	BY, LT, RUS	Vuoksa	RUS, FIN

The resulting map layer with large European rivers is named 'CCMlrWISe' (note: this map layer does not contain the last Dutch part of the Rhine river, which was added later to the segmented

[4] WISE (Water Information System for Europe). See the WISE portal: http://water.europa.eu/en/welcome and http://www.eea.europa.eu/data-and-maps/data/wise-large-rivers-and-large-lakes. Definitions are from the WISE GIS guidance document (No. 22). Water Pattern Europe, scale 10 million, version 2, from EUROSTAT GISCO database; Water Framework Directive article 3 data on rivers and lakes from countries; Joint Research Centre catchment database CCM1

[5] CCM (Catchment Characterisation and Modelling) River and Catchment Database, version 2.1 (CCM2). The CCM2 database is produced by the European Joint Research Centre (JRC-EU) and covers the entire European continent, including the Atlantic islands, Iceland and Turkey. It includes a hierarchical set of river segments and catchments based on the Strahler order, a lake layer and structured hydrological feature codes based on the Pfafstetter system. CCM data are made freely available for non-commercial use. Website: http://ccm.jrc.ec.europa.eu/php/index.php?action=view&id=24#

river based on the water measurement stations, see sections below). The 20 large rivers included in the analysis with a catchment area bigger than 50.000 km2 are listed in Table 5.4. The Danube, Guadalquivir, Nemunas, Po and Rhine basins are also AquaMoney case study sites.

Step 2: Mapping of water quality

Water quality data for major European rivers were obtained from a database containing national monitoring stations available via WISE. The available water quality measurements at these stations are point measurements and the measures of water quality are in principle only representative for the river water directly around the monitoring station. Sound estimations of water quality based on this data of larger river segments depend on many factors, such as the number and spatial distribution of monitoring stations and characteristics of the river, its tributaries and the river catchment.

The data and information from the WISE database are obtained through the documented 'WISE-SoE data collection process' and are primarily used to compile indicator factsheets, associated with the EEA's Core Set Indicators, upon which EEA assessment reports are based. Rivers dataset include physical characteristics of the river monitoring stations, proxy pressures on the upstream catchments areas, as well as chemical quality data on nutrients and organic matter in rivers' (EEA, 2009b).

From the WISE Water portal a Personal Geodatabase (Waterbase_rivers_v9.mdb) can be downloaded containing 4 different tables. The table 'Locations' contains the geographic coordinates and the characteristics of a number water measurement stations, this table was exported to the shapefile 'Riverstations_waterbaseWISE_v9.shp' to select and plot the different river measurement stations. Parallel with the production of the large river dataset we selected from the shape file 'Riverstations_waterbaseWISE_v9.shp' all river measurement stations that are located in or directly connected to one of the large rivers. Because of possible errors and different naming conventions between the Large Rivers data set and the water measurement station locations, we used the field 'Rivername' in the river measurement stations map layer as the main indicator for the determination of the river for which the water quality measurements are valid. A visual check of the results was performed by overlaying the water measurement point stations with the Large Rivers map layer. Where measurement stations were located too far from the main river course (approximately more than 2 km from the main river course) they were removed from the database. These operations resulted in a selection of 329 water quality measurement stations (in map layer RS_WISE_LR_LEAproj.shp) that were located along the selected large European rivers, see Figure5.10.

As can be seen on the map in Figure 5.10, the distribution of the river measurement stations is not equal for all rivers even though 'the required numbers of river stations should be geographically spread across the country so that the catchment with the longest length of river has proportionally more stations' and 'the total station population should be based on total land area at a density of 1 river station per 1,000 km2' (EEA, 2009). E.g. rivers such as the Seine in France, the Odra in Germany, Poland and the Vistula in Poland have much lower number of river measurement stations than other large rivers in Europe.

From the 4 tables from the WISE Water portal Personal Geodatabase we also extracted the table Quality. The data in this table contains information on a number of different measures of water quality (e.g. total nitrogen, total Phosphorus, BOD, COD) for a number of years (1965-2007), and a variety of aggregation periods (annual, summer, winter). We choose to use a measure of water quality defined as the annual 2003 average of Phosphorus equivalents (mg/litre). Data on total Nitrogen levels are converted to Phosphorus equivalents using a conversion factor of 10 to 1 and added to measurements of Phosphorus to give a total measure of phosphorous equivalent. In a small number of cases, data on annual mean Nitrogen was not available. In such cases we approximated the annual mean from the minimum and maximum Nitrogen levels. In cases where data on either Nitrogen or Phosphorus for a monitoring station were missing, this monitoring station was omitted from the database to ensure consistency in the measurement of pollution levels.

Figure 5.10: WISE River measurement stations of European Large Rivers

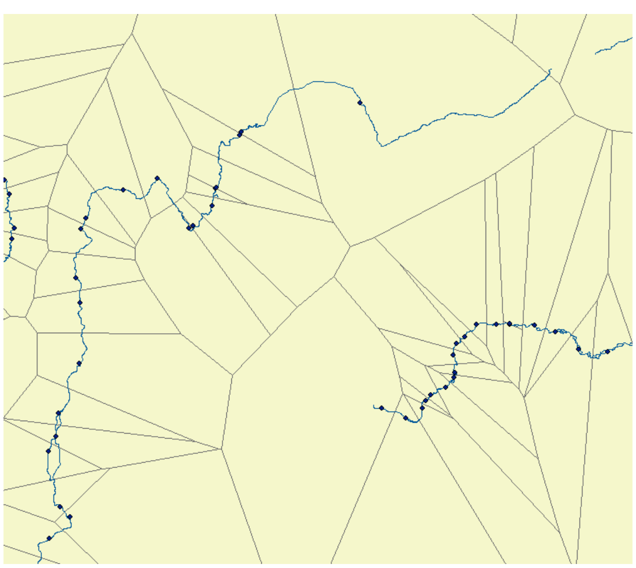
There are a small number of cases for which a single monitoring station has multiple water quality observations. In general, multiple observations of water quality were of similar magnitude. Multiple observations were filtered out of the database to ensure that each monitoring station has only one water quality measurement. The Phosphorus equivalent measure of water quality is then used to compute a 10-point water quality index (Vaughan 1986) using the categorisation in Table 5.. The derived river water quality index value per point measurement is then linked to different segments of the selected rivers.

Table 5.5: P-equivalent value ranges and corresponding water quality index (WQI) values

MIN	MAX	WQI	Water Quality Ladder
0.005	0.077	9.5	Drinkable
0.077	0.150	8.5	Drinkable
0.150	0.216	7.5	Swimmable
0.216	0.271	6.5	Swimmable
0.272	0.334	5.5	Swimmable
0.334	0.414	4.5	Fishable
0.414	0.511	3.5	Fishable
0.513	0.653	2.5	Fishable
0.653	0.905	1.5	Boatable
0.907	5.953	0.5	

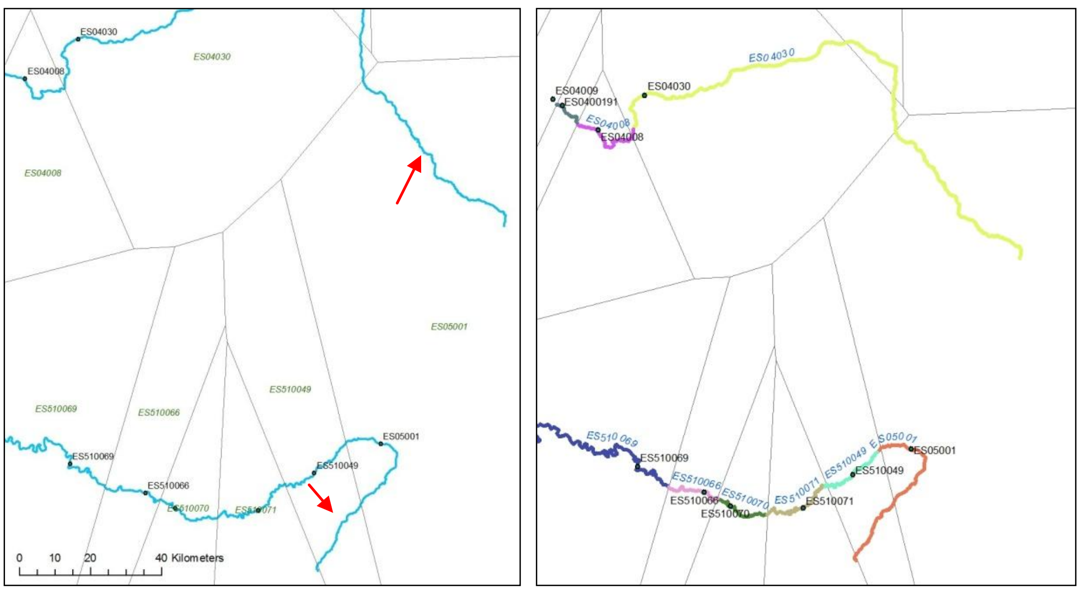
After the calculation of a general water quality index on the basis of the available measurements in the selected stations for the year 2003, this water quality index had to be linked to representative sections of the selected European large rivers. Obviously the water quality measurements cannot simply be interpolated along river courses between water quality measurement stations, as changes in water quality can both be gradual as radical, e.g. where the different water quality of a connected tributary river strongly influences the water quality in the main river course, or where the river enters a different catchment area. As there are no simple, formal and accepted ways to assign point based river quality estimates over an entire river network, we have chosen to draw borders between river quality levels halfway in between river quality measurements with the help of Thiessen polygons (see Figure 5.11), followed by intersection of rivers in different segment on the basis of the Thiessen polygon boundaries. Consequently a spatial join operation (point-line) was used to link the relevant attribute information of the different measurement stations to the spatially corresponding river segments.

Figure 5.11: Rivers segmented on the basis of location in Thiessen polygons of water quality measurement stations. Attributes of water quality measurement stations are assigned to corresponding river segments.



All automatically drawn Thiessen polygons were checked by hand and where Thiessen polygon intersection led to incorrect segmentation and naming of rivers, these river segments were corrected by hand. An example of this process is shown in Figure 5.12 where Thiessen polygons resulted in wrong segmentation and naming of two sections in two different rivers. From the colours of the river segments in Figure 5.12 (right), it can be seen that river segments are assigned to the correct water quality measurement stations after they were initially wrongly segmented by the Thiessen polygons. Also the spatial join operation between river measurement point stations and the created river line segments had to be checked and where necessary corrected manually, because in many cases the point measurement stations do not intersect exactly with the river segments. The resulting map layer with large European rivers segmented on the basis of water quality measurement stations is named ‘CCMlrWISe_WS4.shp’.

Figure 5.12: Left: wrong partition of river segments (indicated by red arrows) by Thiessen polygons. Right: corrected partition (by hand) of river segments



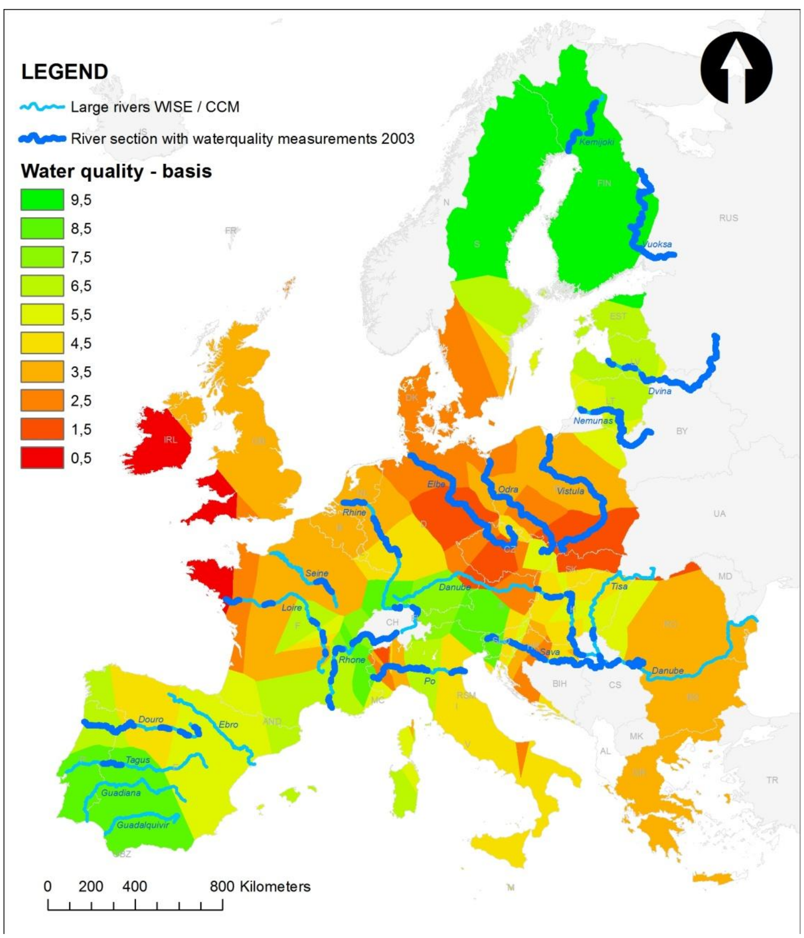
After this editing procedure river water quality information from the WISE data portal could be linked to the different river segments.

Because the map layer with large European rivers segmented on the basis of water quality (‘CCMlrWISe_WS4.shp’) contains two identification fields (WaterbaseID and National station ID) that correspond with the WISE water measurement stations, water quality information from these stations can be linked to the different river sections. In this way a new field could be added to the map layer ‘CCMlrWISe_WS4.shp’: WQ_base (labelled WQl_10) containing a water quality index figure (between one and zero) based on the combined parameters in the water quality table in the file Waterbase_rivers_v9.mdb. Next, another field was added in which the difference between the current water quality and the desired water quality (index value 0,9) is calculated. This new field is named WQchange. Records in which the resulting values were negative (meaning that already a water quality higher than the 0,9 index was present) were set to zero. Two more fields were added in which all values were multiplied with 10 as the next analysis step requires integer values.

Because for many of the measurement stations the necessary water quality measurements for estimating a general water quality index for 2003 were missing, the resulting map layer contained many records without a water quality index. As a consequence, only 133 measurement stations of European large rivers are suitable for calculating a general water quality index for the year 2003. Therefore the records with missing water quality information were removed from the attribute table after which the map layer was named 'CCMlrWISe_WQ.shp'. This map layer only contains large river sections with a corresponding water measurement station with a water quality index for the year 2003.

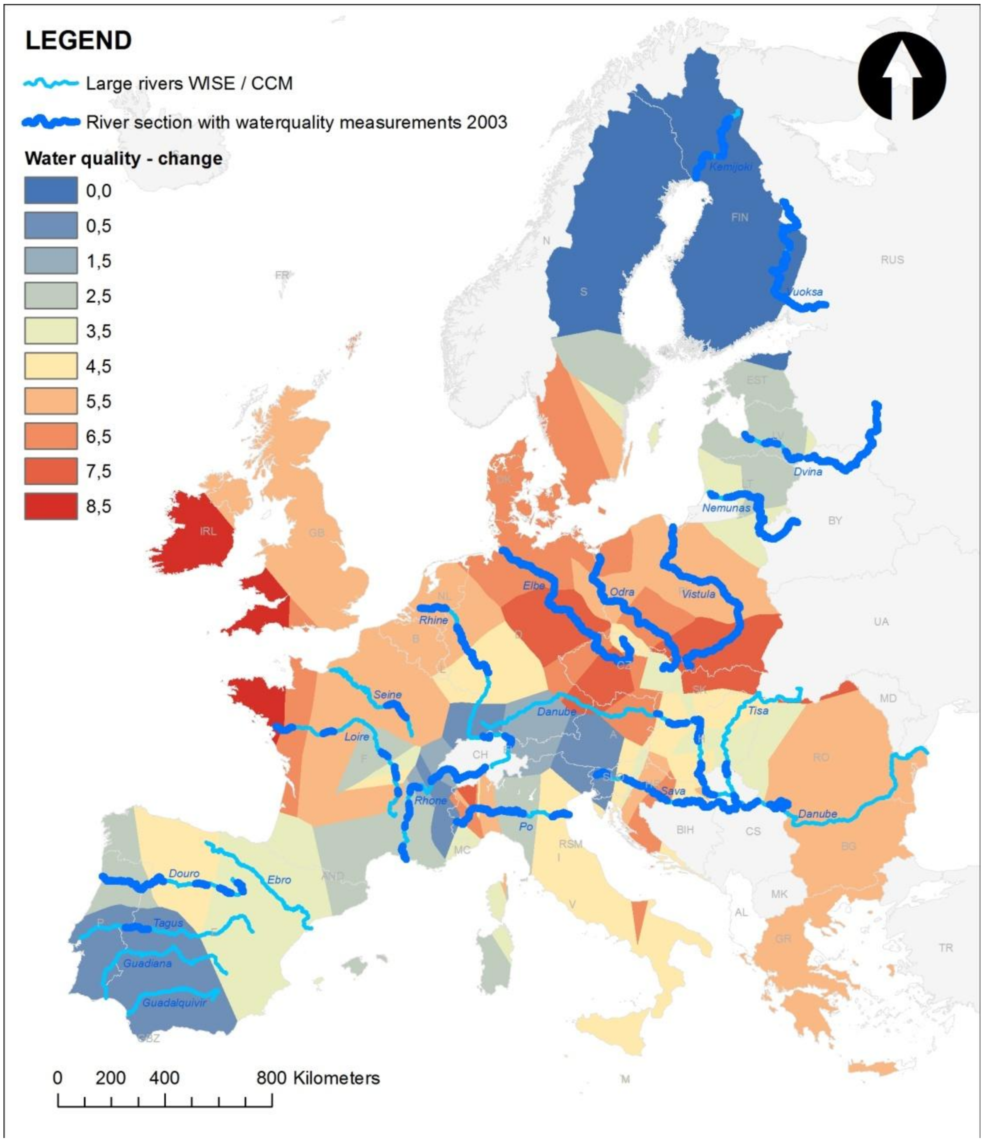
On the basis of the map layer CCMlrWISe_WQ.shp an Euclidean allocation procedure was carried out to calculate for each grid cell on the European land territory a water quality index value based on the quality of the most nearby river segment of a large river that contains a water quality index value. The attribute field 'Wqbasex10' was used as the source field for this operation (the field used to assign values to the source locations). Because not all river segments have a water quality value assigned, the consequence of this operation is that assigned values often do not concern the water quality of the most nearby river, but only of the most nearby segment of that river (or another river). Because no maximum distances are used another consequence is that also grid locations very far from large rivers and in other countries without selected large rivers get water quality index values assigned. The resulting map with allocated water quality index values was divided by 10, to get back the required index values between 0 and 1 named 'wq_base'. The resulting map is displayed in Figure 5.13.

Figure 5.13: Basis river water quality map in 2004



To derive the water quality change map, a similar procedure was used as for deriving the water quality baseline map. The only difference is that the water quality change map is calculated with a Euclidean allocation based on the attribute added field Wqchanx10, which indicates the difference between the current (baseline) water quality and the desired water quality with an index of 0.9. The resulting map with allocated water quality index values was divided by 10 to get back the required index values between 0 and 1 and named ‘wq_change’. The resulting map is displayed in Figure 5.14.

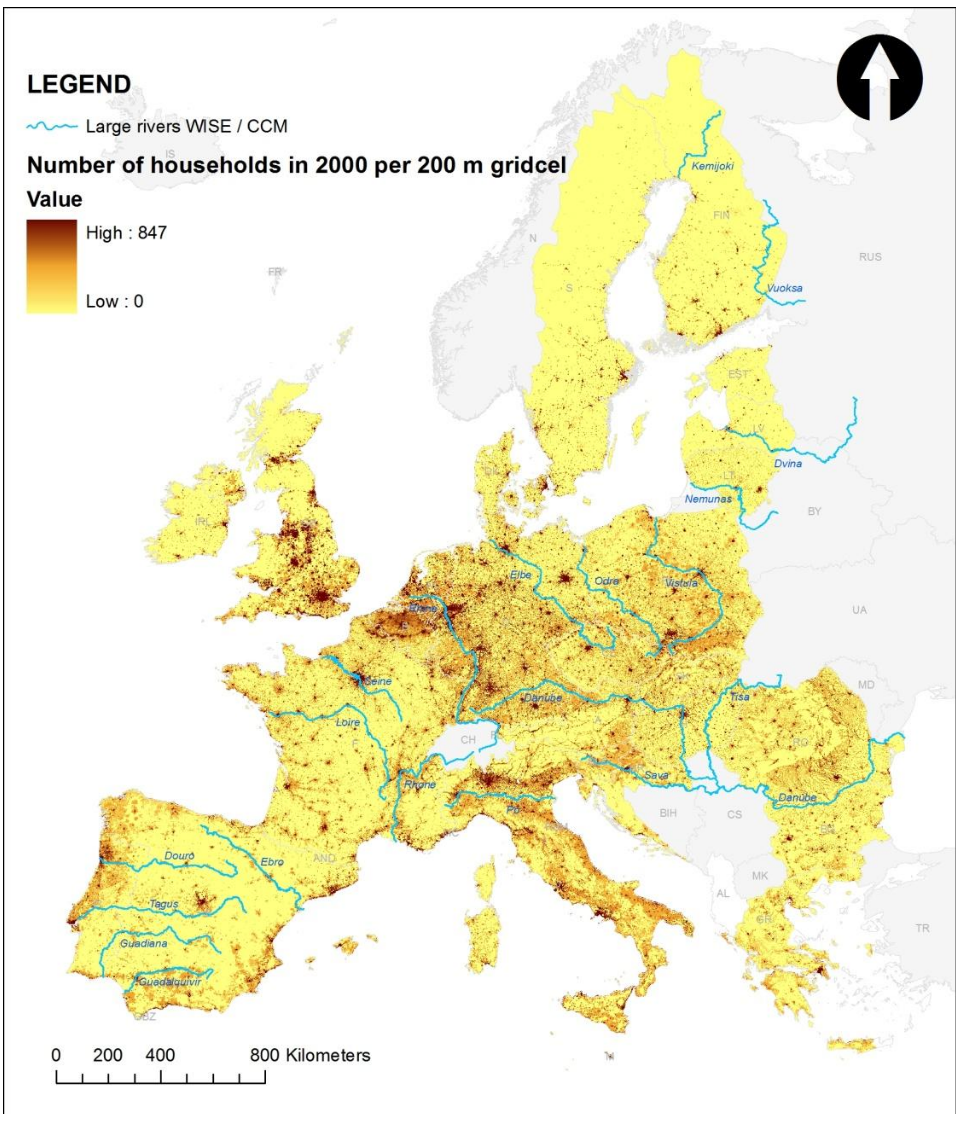
Figure 5.14: River water quality change from basis to 0.9 index scale.



Step 3: Socio-economic mapping

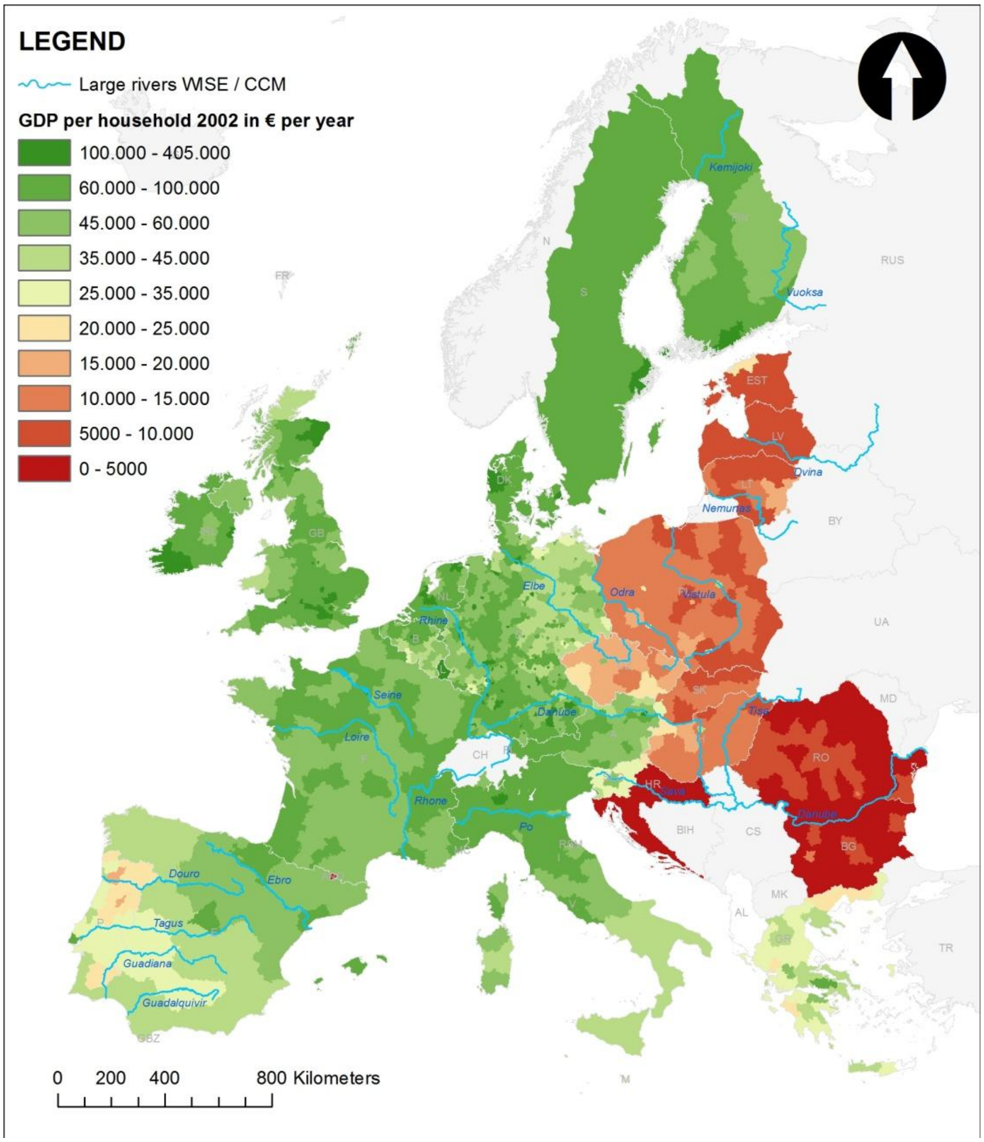
To produce a map with the distribution of households in Europe, a population density map is used as a basis. The JRC 2000 population density map was chosen and disaggregated from the 2000 European CORINE Land Cover database (inhabitants per km^2 in 100 x 100 m grid cells). The 100 meter population density grid of the EEA data service for the year 2001 was chosen because of its EU-wide extent in combination with the high horizontal resolution of 100x100 meter cells. The downscaling method that was used to produce this grid and conclusions on the accuracy of this dataset are described in an unpublished paper available on the EEA dataservice website (Gallego, 2008). As we chose to work with a resolution of 200 meter grid cells, we aggregated the 100 meter population density grid to a gridcell size of 200 meters. Next, we converted the population density to the number of inhabitants per 200x200m gridcel (divide by 25). To calculate the number of households per gridcel, we divided the number of persons by the average European household size, which is approximately three. The resulting raster layer is named ‘pophh’. The resulting map is displayed in Figure 5.15.

Figure 5.15: Household distribution in 2000



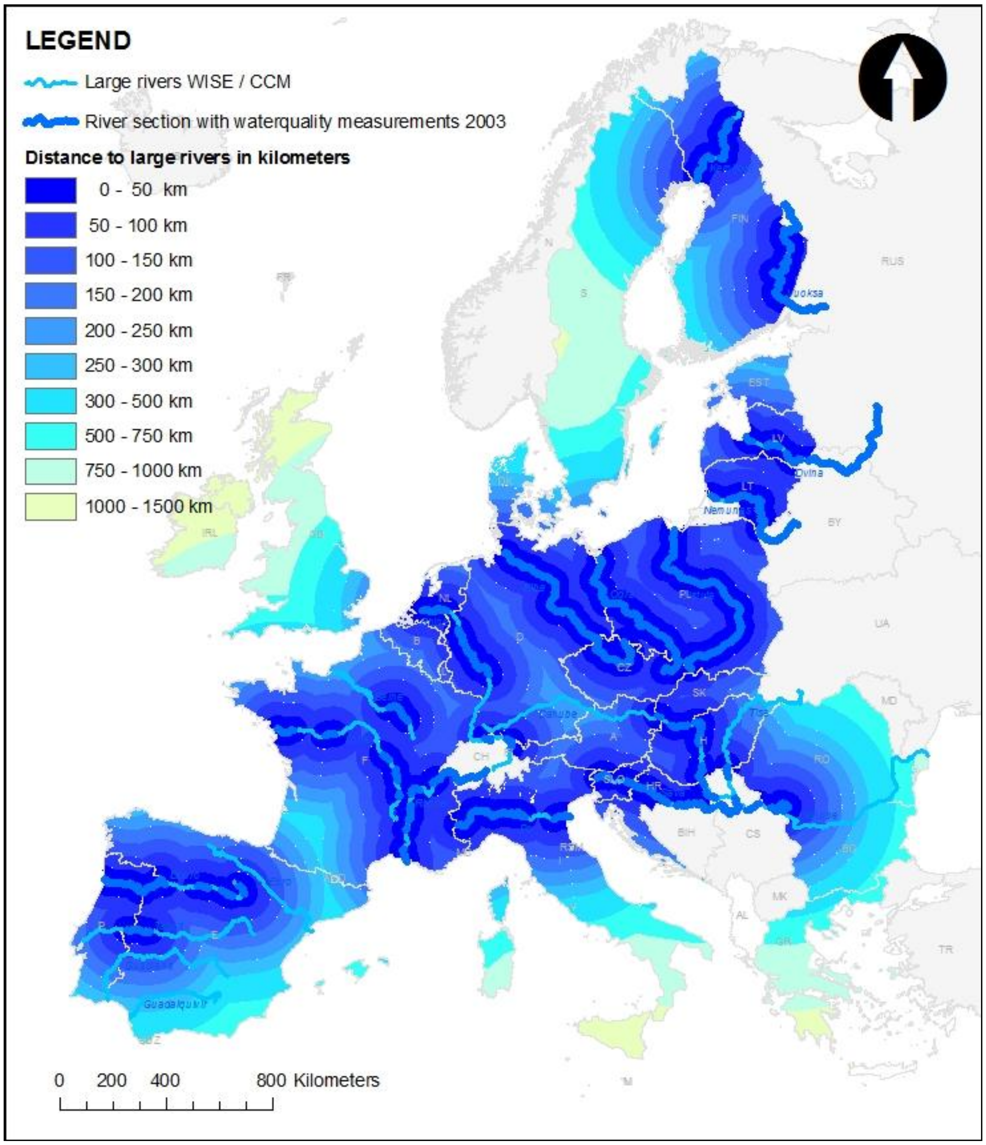
To produce a similar map with the distribution of GDP per household, the map with administrative boundaries of NUTS 3 areas (ESRI maps & data 2008) is used and linked to the table data from ESPON with Gross domestic product (GDP) per capita in 2002 per NUTS 3 region in Europe. This vector map grid was rasterized to 200 meter grid cells so each grid cell was assigned an average per capita income based on the specific geographical NUTS-3 area to which the specific grid cell belonged. As we need for each cell the average GDP per household instead of the average GDP per capita, all grid cells were multiplied by three (the approximate European average household size). To get values in thousands of Euro's the gridcells were also divided by 1000. The resulting raster layer is named 'gdphh02x1000' and the map is displayed in Figure 5.16.

Figure 5.16: GDP per household in 2002 (in € per year)



Finally, distances from each grid cell to nearby river segment were calculated. This map contains for each 200 meter grid cell the Euclidean distance to the most nearby river segment with a water quality value from a large river (i.e. an Eucledian distance operation based on ‘CCMlrWISe_WQ.shp’). The resulting kilometre distance map was named ‘wqkm1’ and is presented in Figure 5.17.

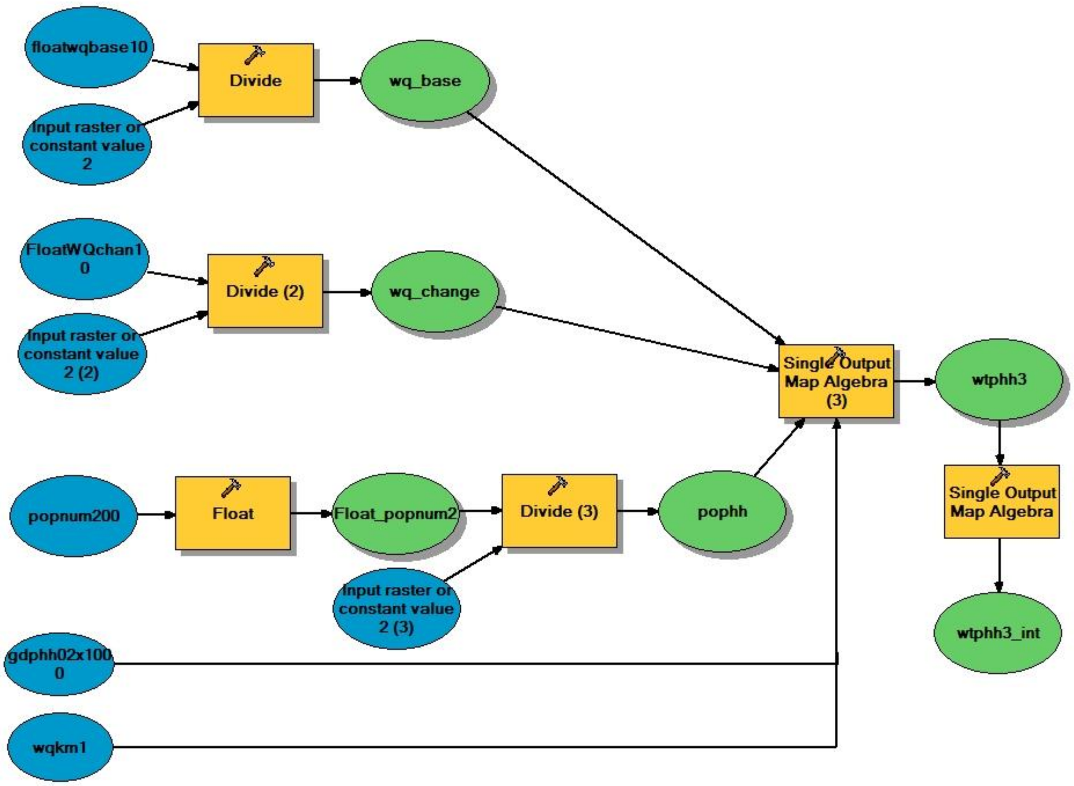
Figure 5.17: Distance to selected large rivers in Europe in kilometres



Step 4: Value mapping

ArcGIS modelbuilder, a spatial model building tool provided with the software, was used to implement the value transfer function based on the required input map layers. Name of the model constructed is ‘Valuetransfer-run1’ and is located in the Toolbox ‘Aquamoney_value.tbx’ (in ArcGIS project Run1.mxd). The applied model is presented in Figure 5.18.

Figure 5.18: Applied ArcGIS model for water quality value transfer



The operation "Single Output Map Algebra" contains the most important part of this model, i.e. the translation of the water quality transfer function presented at the start of this section into ArcGIS analysis terms. The resulting map layer is named 'wtphh3' (see Figures 5.19 and 5.20) and the aggregated WTP values per country derived from this layer are presented in Table 5.6. In the last step of this process this layer is converted into an integer value grid for cartographic purposes, named 'wtphh3_int'.

The maps in Figures 5.19 and 5.20 show a high correspondence between the density of beneficiaries (i.e., in cities) and the spatial distribution of values associated with water quality improvements. It also shows that values are in general higher when closer to the selected rivers such as the Rhine (Netherlands) and the Seine (France). To check for the validity of the results we selected a small number of sample cells at different random locations in Europe and used the values of the different input maps as input for the same value transfer function, but this time implemented as an Excel sheet function. After changing some details in the translated GIS function, the same input values generated exactly the same results (one decimal behind the comma) in both implementations of the function.

Figure 5.19: Value map for water quality improvements in Europe

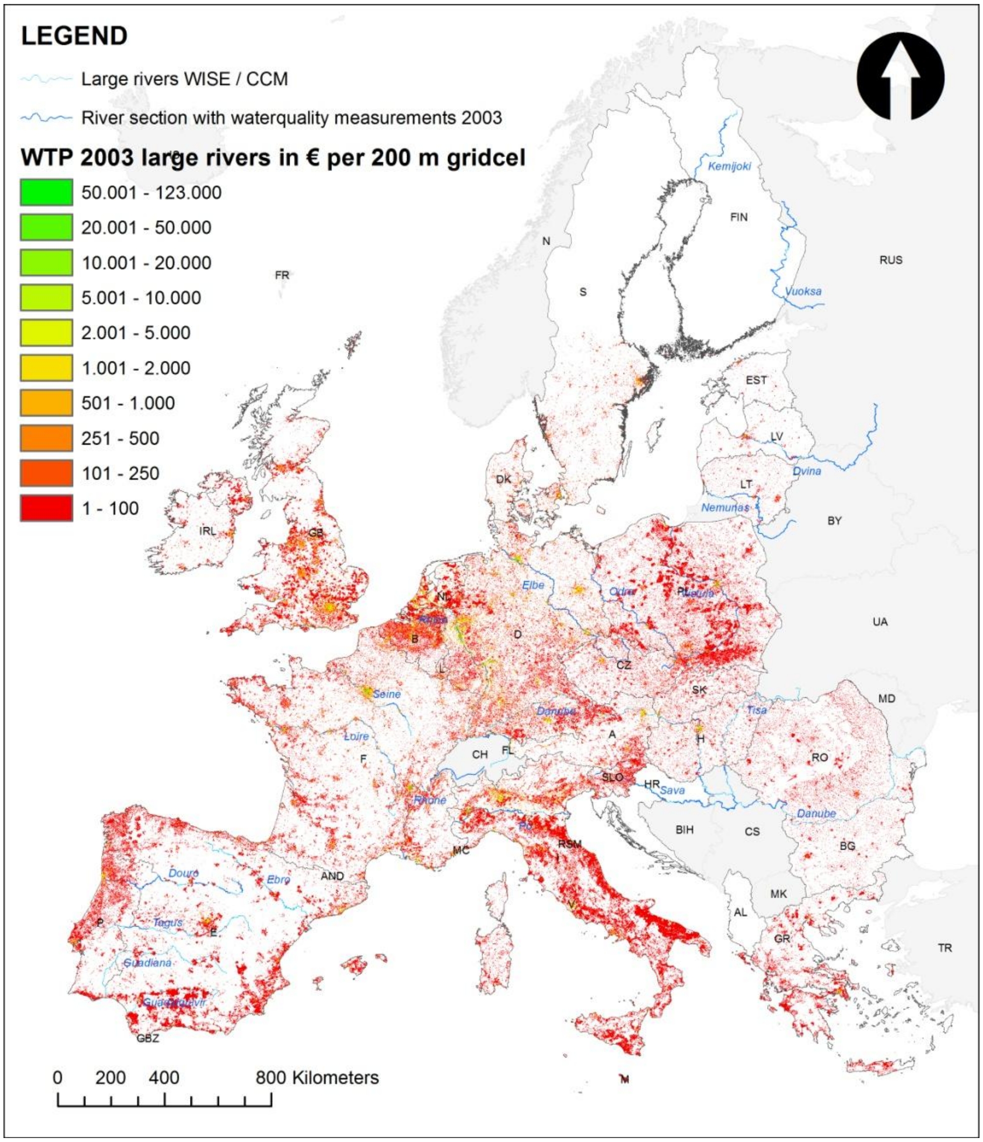
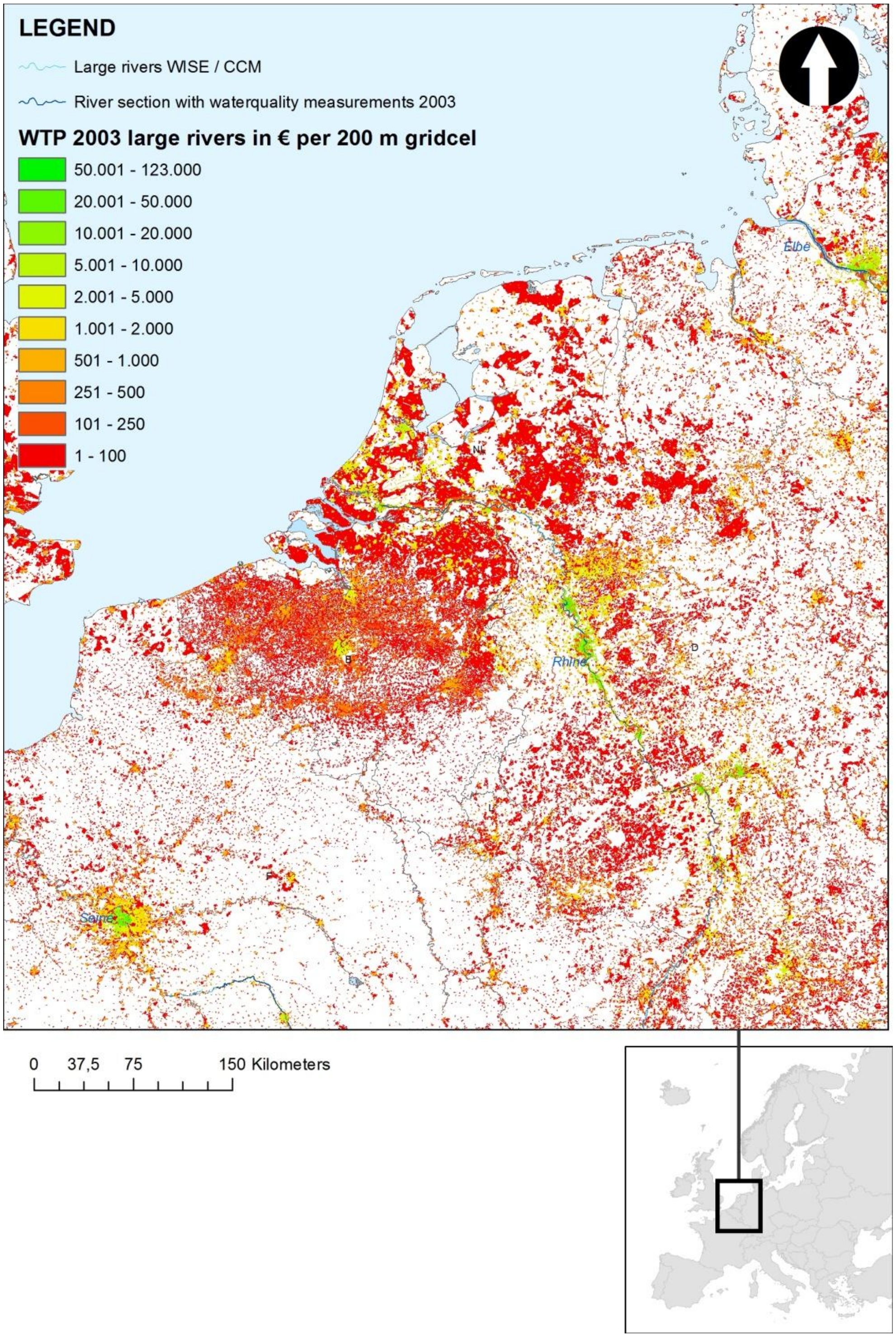


Figure 5.20: Value map for water quality improvements in Europe e (zoomed in to BeNeLux, plus parts of France and Germany)



To quantify and compare the results between the different European countries the estimated values were aggregated by country and are presented in Table 5.6 (extracted from wtp_cntrs.xls), which summarizes for each European country the standard statistics minimum, maximum, range, mean, standard deviation and sum of all grid cell values (ArcGIS function Zonal Statistics as Table).

Table 5.6: Aggregate WTP for water quality improvements to 'good ecological status'

Country	Segment length of known river quality in km	Total WTP
Greece	0	11,289,000
Netherlands	111	170,926,000
Belgium	0	58,117,700
France	896	406,266,000
Spain	680	103,173,000
Hungary	322	38,964,700
Italy	493	326,282,000
Romania	171	3,974,900
Austria	54	53,557,500
United Kingdom	0	197,578,000
Denmark	0	24,795,400
Sweden	0	24,340,600
Poland	1842	87,365,400
Germany	1606	792,553,000
Portugal	478	32,567,800
Luxembourg	0	4,920,980
Ireland	0	8,411,760
Malta	0	326,254
Cyprus	0	674,052
Finland	1273	134,484
Bulgaria	0	1,343,220
Lithuania	682	4,449,600
Latvia	838	5,306,970
Estonia	0	648,803
San Marino	0	2,010
Croatia	578	3,332
Slovenia	210	7,711,920
Czech Republic	586	28,825,100
Slovakia	66	8,608,560
Liechtenstein	0	8,971

It should be noted that the results in Table 5.6 have to be interpreted with the necessary care for a number of reasons. For instance, the following countries that are not listed contain river segments of large European rivers with river quality values: Belarus, Bosnia and Herzegovina, Russia, Serbia and Montenegro and Switzerland. Because of differences in either river distribution and data quality (total length of river segment per country with water quality measurements) comparisons between countries are subject to bias. This is illustrated by the second column which indicates for each of the countries the total length in kilometres of river segments of known quality. It should be mentioned that these river lengths can only be used for relative comparison between countries as it concerns measured river-segment lengths from a generalised database. Real river lengths, including all curves, should therefore be much longer. Comparing the countries this way, gives more predictable results. For example, comparing Germany and Poland, which are comparable in river segment length, but with a very different total value mainly as a result of income and partly of population numbers.

It will not come as surprise that scaling up an spatial analysis that usually is carried out for large scale areas such as water bodies confined in one river catchment to a small scale European dimension is subject to a large range of small to large quality issues. Here we will look in particular to the errors that have a spatial dimension. Even though the results of this analysis are based on probably the best ready available scientific data and methodology, the results may be seriously biased for the following reasons:

- Incompleteness and uneven distribution of water quality data. Some of the rivers completely lack water quality information or have a very uneven distribution of water quality measurements.
- Allocation of point quality data to river segments. This allocation should be based on specific knowledge concerning the selected rivers, the contributing river branches and catchment areas.
- The use of an average value transfer function for the whole of Europe. The question is whether this generic function is always applicable to any type of large river.
- Disregarding maximum distances of influence. Is it reasonable to include the whole United Kingdom in WTP calculation of rivers on the European mainland? Including more, that is also smaller, rivers will partly solve this problem, as e.g. the incorporation of the river Thames will make this river the most nearby river for the whole of the UK, which makes more sense than the current analysis setting.
- Not regarding substitution effects, as the analysis only considers the most nearby large river and not other rivers located slightly further.
- Use of outdated data (population 2001, GDP 2002, water quality 2003) even though the mutual time differences between are quite acceptable. However, these data can relatively easily be replaced by more recent data (e.g. LandScan population data 2008 (which has its own quality issues), GDP NUTS 3 data 2007 freely downloadable from EuroStat and more recent quality data (available until 2007, it should be checked how complete this is compared to the 2003 data).
- Scale differences of original data sources, especially the NUTS3 administrative scale of the GDP per capita data will negatively influence the accuracy of the results compared to the 1 km grid accuracy of population density and distance to rivers. It is however unlikely to get better GDP data on a more detailed scale for the whole of Europe.
- Uncertainties concerning positional accuracy and error in the original map layers. E.g. the used population density map does not differentiate population differences within municipality borders. Summarized values per municipality will however be consistent with census counts values, so the accuracy of this dataset will be as good or better than e.g. the used GDP dataset.
- Use of Euclidean distances instead of realistic travel times.
- No differentiation used in average number of households per European country, or better per NUTS3 region.

- Positional and attribute accuracy of large EU rivers. Considering however the other quality issues the CCM river dataset is considered more than adequate for this analysis. Carrying out visual comparison with other data sources of position of river courses and naming is however recommended.
- Error propagation and known geographical quality issues from GIScience. This concerns numerous quality issues of which some can have profound effects on the end results, some of the most serious ones are however already partly or completely covered by the quality issues mentioned here above.

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6. Water resource costs

6.1 *Introduction*

Water scarcity is a well-documented, worldwide environmental problem (UN, 2009). As a result of climate change, water scarcity is expected to increase further (IPCC, 2007). Since the 1960s, hydro-economic optimization models have been built by resource economists and water resource engineers to inform policy and decision-making about efficient levels of water allocation (Lund et al., 2006; Harou et al., 2009). Typically, these models are based on detailed descriptions of the hydrological system and include economic demand functions for agriculture, industry and domestic households (e.g. Draper et al., 2003; Pulido-Velázquez et al., 2004), often with a particular focus on the largest freshwater consumer worldwide, irrigated agriculture, using agronomic crop yield functions and farm-level microeconomic data (e.g. Moore et al., 1994; Cai et al., 2008). Optimization of water allocation across these main user groups results in the estimation of water shadow prices. Environmental water demand is usually imposed as a constraint in the water allocation optimization procedure (Kirby et al., 2006). The optimization procedure informs about the opportunity costs (benefits foregone) of restricted water use in agriculture, industry and domestic water supply using from the shadow prices of the environmental flow constraints (Medellín-Azuara et al., 2007; Pulido-Velazquez et al., 2005 and 2008). Other models use directly environmental and recreational economic value functions obtained using non-market valuation techniques, so that non-consumptive in-stream uses and consumptive uses compete for the allocation of water in the system (Ward and Lynch, 1996; Diaz et al., 2000). Applied general equilibrium models have also been used to better account for inter-sector linkages in economic systems (e.g. Berk et al., 1991; van Heerden et al., 2008; Brouwer et al., 2008). However, here too environmental concerns are primarily accounted for by imposing environmental constraints on inter-temporal welfare maximization.

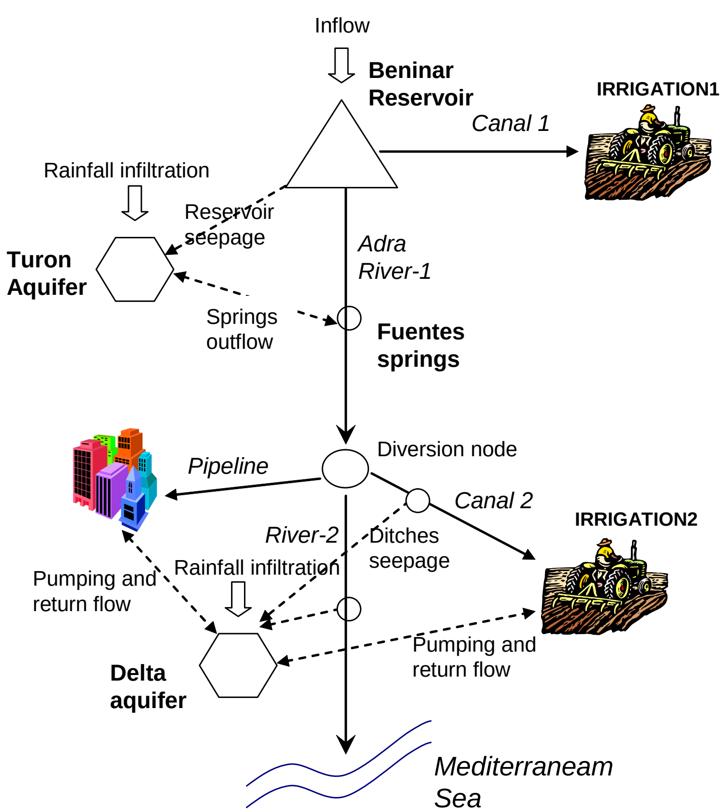
In this chapter, we present two case studies, one illustrating the usefulness of hydro-economic modelling to assess water resource opportunity (scarcity) costs and shadow prices of water use, the other illustrating the application of a non-market valuation approach to estimate demand for environmental water use.

6.2 *Case study illustration: hydro-economic modelling of water resource scarcity*

In the Campo de Dalias coastal plain in South-Eastern Spain, the high value of greenhouse crops has led to a spectacular increase of cultivated land (over 20,000 ha) that started four decades ago and has become the main factor of economic growth in the province. Associated to this quick development there has been an important increase of water demand, mostly supplied by groundwater from the Campo de Dalias' system of aquifers, which also provide the urban water supply for a population of about 250,000 inhabitants. The intense overexploitation has caused significant drawdowns in piezometric heads in the most productive limestone aquifers, provoking heads below the sea level and consequently, seawater intrusion problems and progressive water quality degradation in sectors of the eastern and especially western areas of the Campo (Pulido-Bosch, 2000). During the period 1987/88 this system began to receive water imported from the Beninar reservoir through the Beninar-Aguadulce channel as a means to reduce the overexploitation of the system of aquifers in the Campo.

A detailed simulation study of surface and groundwater conjunctive use in the Adra-Campo de Dalias system examined different management alternatives (Figure 6.1). A hydrologic-economic optimization model of the Adra's water resource system downstream the Beninar reservoir has also been developed. The area's water system was converted into a network flow scheme consisting of a reservoir and two aquifers that supply water to urban and agricultural demands. The *Irrigation1* demand, which corresponds to the portion of the Campo de Dalias demand supplied by imported water, receives water from the reservoir through *Canal1* (Benínar-Aguadulce canal). Downstream of the reservoir, the traditional irrigation districts of the Adra River basin, aggregated as *Irrigation2*, are supplied by streamflow diversions by canals (*Canal2*) and groundwater pumping from the alluvial aquifer of the Delta. The urban water demand (mainly the city of Adra, Almeria province) is conjunctively supplied by groundwater pumping in the Delta and streamflow diversions from a pipeline. The *Beninar reservoir*, in the middle basin of the Adra River, is the main element for flow regulation. It has significant water losses by seepage occur, part of which is collected by the carbonate *Turon aquifer*, which discharges back into the river through the *Fuentes de Marbella springs*. The aquifer's recharge comes from the reservoir seepage losses, seepage in the river downstream the reservoir, and rainfall percolation. Sources of recharge to the *Delta aquifer* include rainfall and runoff infiltration, seepage through irrigation ditches, seepage in the last river reach, and irrigation and urban return flows. Groundwater outflows include pumping to supply *City* and *Irrigation2* and sea-aquifer water exchange. Recharge from rainfall and runoff percolation is modeled in both aquifers by time series of infiltration obtained by rainfall-runoff modeling, while seepages from the Beninar reservoir, the last reach of the Adra River, and the irrigation ditches above the Delta are computed through empirical equations.

Figure 6.1: Schematic representation of the Adra system



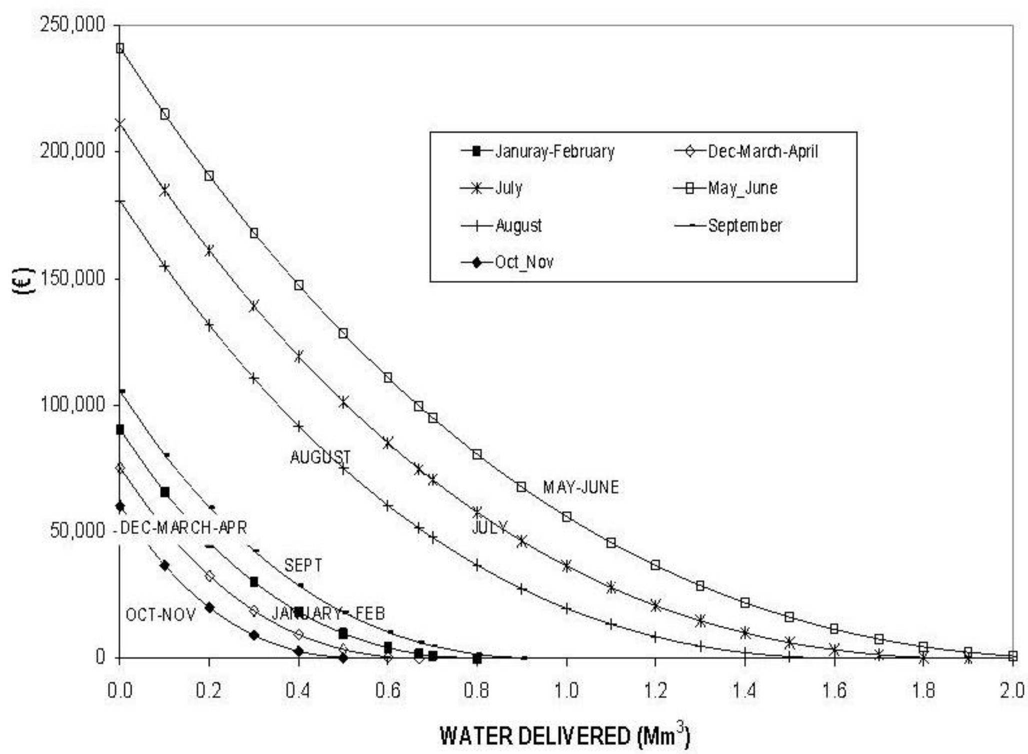
A holistic hydro-economic optimization model was developed that uses an implicit stochastic approach for long historic streamflow records. The non-linear model maximizes the net economic benefit from water use in the system over a ten-year long period with monthly intervals. The objective function is formulated as a cost-minimization problem: minimize the total cost of the system, including water resource scarcity costs and variable operating costs of water allocation and system operation:

$$z = \sum_t \left(\sum_i SC_{i,t} + \sum_c OC_{c,t} \right) \tag{1}$$

where $SC_{i,t}$ is the water resource scarcity cost in the supply to the demand i in time t, $OC_{i,t}$ is the variable operating cost in the element c and time t. Pumping cost in the Delta Aquifer is calculated dynamically as a function of pumping lift and pumping rate (Pulido-Velazquez et al., 2007). A constant unit operating cost for the urban supply represents the cost of water withdrawal, conveyance, treatment, distribution, and wastewater discharge.

Scarcity is the difference between actual deliveries and maximum beneficial water supply and water resource scarcity cost is defined here as the economic value of increasing deliveries to eliminate scarcity to different users. Scarcity costs are derived from economic demand curves for different water uses, relating the quantity of water demanded to its marginal value. Scarcity costs arise if demand is not fully met and can be found by integrating the area under the demand curve from the quantity demanded to the quantity actually supplied. In this case study, an annual quadratic demand function for irrigation based on Garrido (2000) was disaggregated into monthly irrigation demand curves according to actual monthly irrigation schedules (Figure 6.2).

Figure 6.2: Irrigation water delivered per month and associated costs



Urban demand is characterized by a constant-elasticity curve, calibrated from observed quantity-price relations and previous estimations of the price-elasticity of the curve, which is common practice when lacking more precise information (e.g., Jenkins et al., 2003).

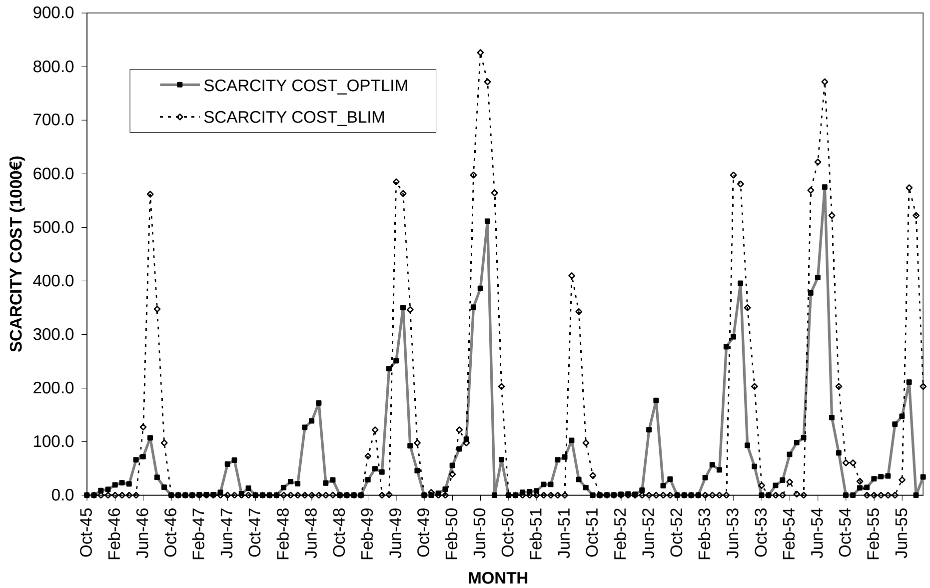
Environmental demand is represented as a monthly minimum streamflow constraint imposed at the river reach outlet for environmental reasons and to maintain seepage into the Delta aquifer. The shadow values associated to those constraints will provide information about the marginal opportunity cost of imposing these environmental requirements.

Two main scenarios were considered. In the *baseline scenario* water allocation is driven by current institutional priorities of water use among different demand units. The scenario is simulated by a module of AQUATOOL, with GAMS used as an economic postprocessor. Unlike the optimization model, the simulation model has no foresight of the future hydrology. Water allocation and operation decisions are made each monthly time-step according to demand targets, resource availability and predefined operating rules (priorities). The simulation model allocates water in the system to minimize at each monthly time step the sum of water shortages and deviations from minimum streamflow and hydroelectric production targets, weighted by pseudo-costs internally defined in terms of priorities (Andreu et al. 1996). Urban use has the highest priority, followed by *Irrigation2*, and then the interbasin transfer, *Irrigation1*. The simulation provides monthly storage and flow results that are post-processed to calculate the economic impact of current water allocation policies. Groundwater is used in months when there is not enough surface water to satisfy demand targets.

An economic *optimization scenario* optimizes system operation and water allocation using the GAMS model based solely on economic water consumptive use values. Constraints are based on physical limitations of the system and minimum streamflow in the last reach of the Adra river. If the aquifer exploitation is not constrained, results from both the baseline and economic optimization scenario show that the heads in the coastal cells will drop below sea level during a long period of time, giving rise to seawater intrusion. Comparison of the economic results for the simulation model that reproduces current water allocation rules with the results for the economically optimal strategy allows the assessment of the potential benefits of changing to more flexible conjunctive use water allocation strategies. The gain in aggregated benefit is due to both optimization of the system operation and water reallocation to the highest value uses. The problem is programmed in GAMS (Brooke et al. 1998), and solved using the MINOS nonlinear solver.

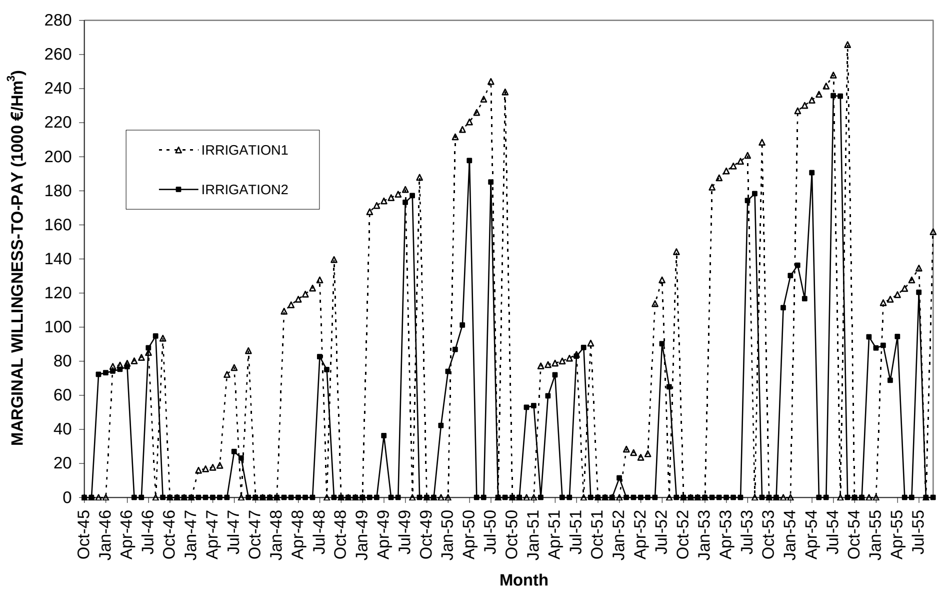
To avoid seawater intrusion and guarantee the sustainability of the resulting system operation, two more scenarios are considered: a new base simulation scenario, where pumping is limited to prevent coastal cell heads from falling beneath sea level (*scenario BLIM*), and an economic optimization scenario with similar head constraints (*scenario OPTLIM).* The model results provide monthly time series of scarcity and scarcity cost, aggregated for the whole system and disaggregated across water demands. The changes in system operation and water allocation under the optimal scenario, OPTLIM, significantly reduce water scarcity and scarcity cost (Figure 4), increasing the net aggregated benefit from water use for the system as a whole. Optimal water allocation results suggest changes in current system operation useful to improve the conjunctive use of the reservoir and aquifers.

Figure 6.3: Monthly aggregated scarcity costs for the BLIM and OPTLIM policy scenarios



Time series of the marginal shadow value of water scarcity shows the value of an additional unit of water for the users (Figure 6.4). These shadow values of water at a given demand location theoretically represent the prices at which users would just be willing to sell or buy an additional unit of water. In Figure 6.4 the graph clearly shows that marginal WTP is higher for Irrigation 1 than for Irrigation2. Marginal WTP for the city is in this case study zero, indicating there is no scarcity, often due to the fact that there are laws guaranteeing domestic water use.

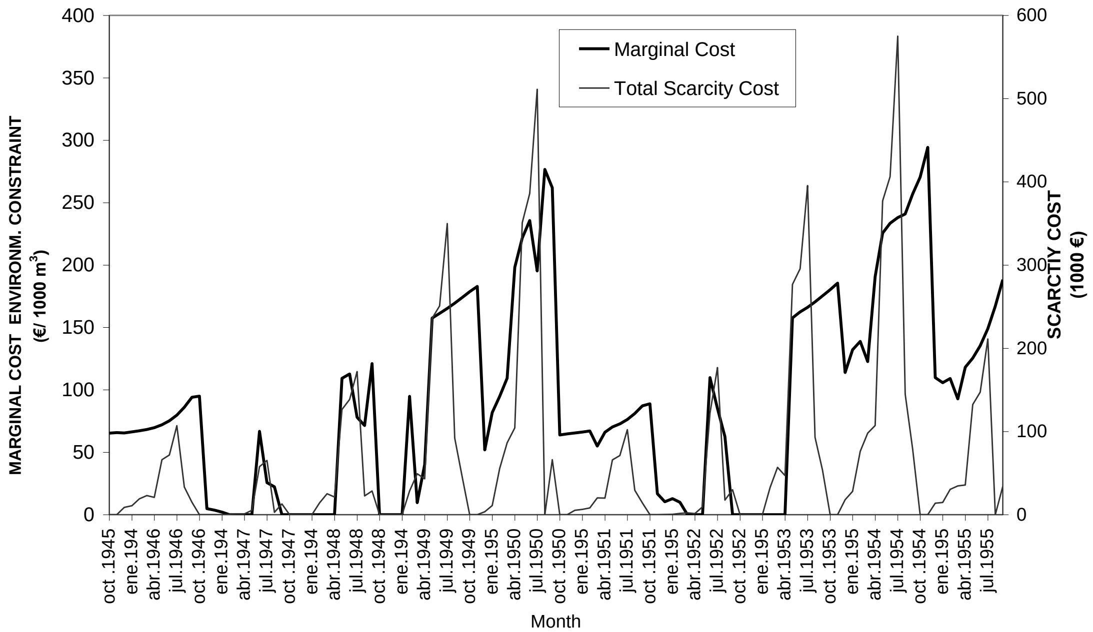
Figure 6.4: Marginal shadow values for water for two user groups under the OPTLIM policy scenario



Environmental constraints impose changes in the system operation and water allocation. Also if there is no direct financial implementation cost involved when taking additional measures to improve the ecological status of water bodies, if these changes imply a reduction in the supply to other uses in the basin, an opportunity cost exists. Integrated hydro-economic river basin models allow the assessment of the total and marginal opportunity cost of environmental flow or storage constraints, for example due to minimum environmental streamflow requirements.

The marginal opportunity cost of environmental requirements can be assessed as the cost for the system of increasing the environmental constraint by additional one unit. The shadow values accompanying the optimal solution include such opportunity costs. The marginal costs of maintaining the monthly minimum streamflow requirements downstream of the diversion node are given by the shadow price series of the corresponding constraint in the optimization model (Figure 6.5). The marginal opportunity cost becomes zero in periods in which streamflows exceed the required minimum and is obviously high (with a peak of €275/1000 m^3) if there is scarcity.

Figure 6.5: Marginal opportunity cost of a minimum streamflow at the last reach of the Adra river



In conclusion, hydro-economic river basin models can assist in the development and formulation of water policies that improve the economic efficiency of water use, taking into account available information on water values, supply costs, and environmental constraints to guarantee the sustainability of the resulting system operation. The shadow values of environmental constraints such as minimum streamflow threshold values show the system-wide marginal opportunity costs of maintaining those constraints. Allocation decisions of scarce water resources always imply an opportunity cost. The shadow values accompanying the solution of the maximum net benefit optimization model include such opportunity costs, considering system-wide allocation effects. Economically efficient water pricing will have to include the opportunity costs of water use, sending a signal to the water users of the economic value of water, which will vary in time and space depending on the scarcity of the resource. Shadow prices of water use related to environmental constraints are useful indictors for the economic analysis required by the European Water Framework Directive (Brouwer, 2004). Since opportunity costs depend on the alternative uses in the basin, an integrated basin-wide approach is needed to account for all major competing water uses, including environmental requirements. Integrated hydro-economic models are essential here as they aim to adequately simulate the physical behavior of the water system, including a realistic representation of the different water sources, including surface and groundwater, as well as their interaction, and the spatial and temporal variability of resource availability. Linking water flows to economic values of different water uses allows for the economic optimization of water resources in times of water supply constraints.

6.3 *Case study illustration: non-market valuation of water conservation*

In practice, the external social costs of water use and the water resource's possible scarcity rents, i.e. the opportunity cost of its unavailability in the future if abstracted until exhausted, are rarely accounted for (Koundouri, 2004). In the case of natural resources with public good values such as freshwater resources, an important question is to what extent their total economic value is fully captured through existing market prices and the opportunity costs of water use in traditional water dependent economic sectors. Non-market valuation techniques have been developed to measure a resource's use and non-use value. It is especially this non-use value which is expected to play an important role in the total economic value of improved water resources management in the context of the European WFD (Bateman et al., 2006; Brouwer, 2008). In this second example, the results are shown of a choice experiment applied in three South European member states facing frequent water supply constraints: Italy, Spain and Greece in order to capture the non-market use and non-use value of freshwater allocation for domestic and environmental purposes.

The developed common valuation design was kept as general and context-free as possible to enhance and facilitate comparison of results (e.g. Ready et al., 2004; Alberini et al., 2004). Using a context-free valuation design describing the same environmental change and related water services applying the same valuation method implies that the main source of variation to help explain differences in the elicited use and non-use values across the country case studies is found in the characteristics of the survey respondents and the specific scarcity context in each country. In each country a random selection of households was sampled.

The most important challenge was to keep the design generally applicable to all three countries. This meant compromising on the inclusion of case study specific detail while keeping the design at the same time sufficiently meaningful to policy makers and lay public in each country. Two alternative water allocations under increasing pressure and for which no or imperfect market prices exist were included in the choice experiment: water security for domestic household use and the environment. In the latter case, water availability has a direct impact on the ecological status of water resources and hence the feasibility of reaching improved levels of water status as defined in the WFD. In the former case, water security reduces the risk that households face water use restrictions in the future. These water use restrictions would primarily affect outdoor water use in terms of sprinkling gardens, washing cars, filling swimming pools and other secondary uses of water during certain hours of the day in the summer. Although water scarcity affected drinking water supply in some countries in the past, drinking water supply is considered a basic right by law in each of the three countries and would hence never be compromised by the imposed water use restrictions and could therefore also not be used as a variable in the design of the choice experiment.

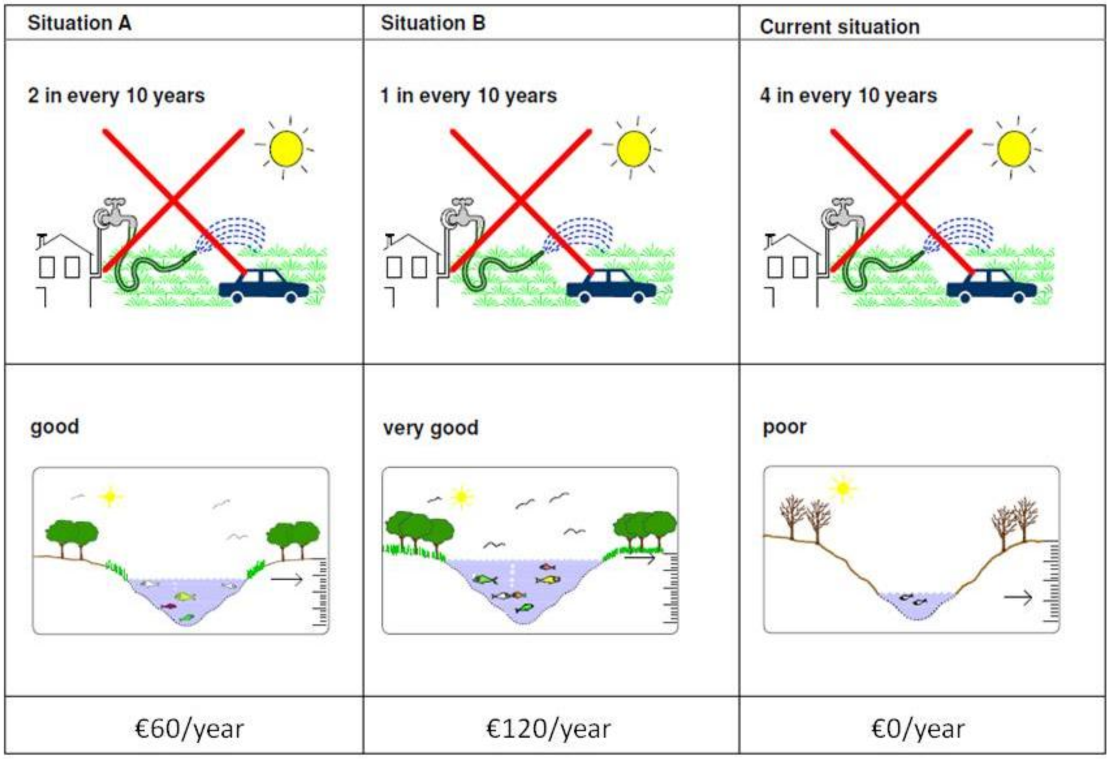
For both alternative uses, three possible levels of improvement were proposed. Following the classification used in the WFD, the ecological status of the freshwater resources in the region where the respondent lives can be improved from current low water levels and corresponding environmental quality to moderate, good and very good levels. Domestic water use security is represented as the likelihood of water use restrictions in the next 10 years based on time series analysis in the three Mediterranean countries and expert judgment. Households are expected to face future domestic water use restrictions in four of the next 10 years. This can be reduced to 3, 2 or 1 year every 10 years depending on additional measures taken to secure future water supply. It was explained to respondents that taking additional measures to secure future water supply for households and/or the environment will come at a cost. The more water will be made available for domestic and environmental use, the higher the (opportunity) cost. If households prefer improved environmental conditions of water resources and/or a reduction of the likelihood that they face domestic water use restrictions in the future, they would have to pay for this through an increase in their annual water bill. The trade-off hence is the price respondents pay as a household for the presented private (use related) water security benefit and the public (non-use related) water security benefit for the environment. Six price levels were used of equal increments of 20 Euros on top of a respondent's current annual water bill.

In the choice experiment, respondents were asked to choose between two possible policy alternatives in which water supply in the next 10 years is secured for domestic and/or environmental use compared to a baseline situation where water scarcity problems remain the same or aggravate. The baseline option describes the situation that currently exists with zero additional cost. The two alternatives in each choice task show possible improvements for either one or both water uses at the same time at an associated cost within a 10 year time frame. This design allows the derivation of compensating surplus welfare measures based on individual household WTP to secure domestic use and environmental non-use water security benefits. Respondents were given 4 of such choice tasks following the recommendation by Louviere et al. (2000) to limit the cognitive burden of the choice experiment. The alternatives and attributes were combined over 6 sets of 4 choice tasks. The allocation of the 6 sets across respondents was random. An example choice card is presented in Figure 6.6.

The choice experiment was part of a wider questionnaire survey. The survey was implemented in each country over a 2 month time period (July-August 2008) through face-to-face interviews by hired professional interviewers with a random selection of households. The number of interviews is lowest in Italy (n=241) and highest in Spain (n=394). In Italy the survey was conducted in the region Emilia-Romagna, in Spain in the region of Valencia region, and in Greece on one of the Aegean islands (Lesvos) (n=312). Except for the small share of women in the sample in Greece, the samples were representative of the regional populations from which they were drawn based on the limited available population statistics.

A majority of the respondents in Greece and Spain consider water scarcity a problem and a substantial share of the sample population experienced water use restrictions in the past. About a third all Italian respondents reported to suffer from domestic water use restrictions due to water scarcity and one in every fifth respondent in Spain. Greek respondents suffered significantly more frequently water use restrictions over the past 10 years than Italian respondents and Italian respondents more frequently than Spanish respondents. Almost a quarter of the Greek respondents are also affected professionally by water supply scarcity, partly because of the relatively higher share of farmers in the sample (Table 6.1).

Figure 6.6: Example choice card



When asked whether they believe they will face increasing water restrictions in the future due to climate change, a third of the Greek and Spanish respondents think this is likely to happen (Figure 6.7). This share is higher in the Italian sample. In Greece a quarter of the sample is 100 percent certain that they will face increasing water cuts in the future. In Italy and Spain this is less than 10 percent. A quarter of the respondents in the three countries do not know. On average across all three countries, less than 10 percent is not convinced that they will face increasing water cuts in the future and 20 percent thinks this will probably not the case.

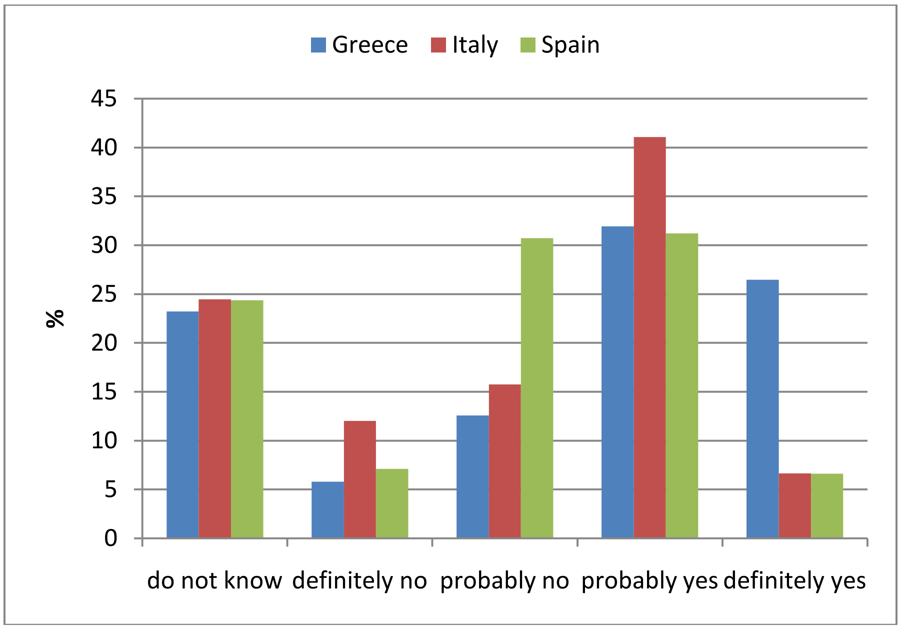
Although a small majority is of the opinion that water should be allocated to agriculture first when faced with water scarcity, between 30-45 percent believe that the environment should receive priority under these circumstances. Respondents are well aware of the negative impact of withdrawing water from natural resources and consequently their environmental conditions. A majority in each sample believes that water scarcity has a negative impact on the environment. On average across all three countries and after securing domestic water use, a small majority of respondents in all three samples is of the opinion that if there is water scarcity, the available water should first of all go to agriculture, followed by the environment (37%). On average less than 10 percent thinks the limited available water should be allocated to industry.

Table 6.1: Sample characteristics

Socio-demographic sample characteristics	**Greece**	**Italy**	**Spain**
Share female respondents (%)	25.6	40.7	49.5
Average age	42.7	40.8	46.0
(min-max)	(18-86)	(18-82)	(18-91)
Share of households with children (%)	37.5	26.9	33.8
Average net household income (€/year)	19,565	18,530	21,990
Share of respondents who is full-time farmer (%)	13.8	0.8	1.8
Perceptions, beliefs, attitudes			
Share perceiving water scarcity as a problem (%)	52.3	21.6	65.9
Share who experienced water use restrictions (%)	55.1	29.7	21.0
Average years restrictions over the past 10 years	5.5	3.7	1.6
Share professionally affected by water scarcity (%)	23.8	6.6	8.1
Share water scarcity affects the environment (%)	96.0	61.8	82.8
If scarce, water priority: - Agriculture (%)	53.7	54.4	52.8
- Environment (%)	45.6	31.1	35.0
- Industry (%)	0.6	14.5	12.2
Protest rate (%)	3.5	0.8	2.0

The number of respondents who consistently chose the opt-out during the four choice tasks for protest reasons such as lack of trust that the increase in their water bill will be spent on water saving measures is very low across the three countries due to thorough pretesting of the survey.

Figure 6.7: Respondent perception of the probability that they will face domestic water use restrictions in the next 10 years



The estimated results for the choice models are presented in Table 6.3. The same model is estimated in each country based on a limited set of theoretically expected explanatory variables to facilitate their comparison and transferability across countries (Brouwer and Bateman, 2005). The fact that answers of individual respondents across the sequence of choice tasks may be correlated is controlled for by allowing the random parameters to vary over individuals but not over the choice sequence. The outcome of the summary statistics of the estimated models suggest that the models fit the data well.

Table 6.2: Estimated choice models for Greece, Italy and Spain

Variables	**Greece**	**Italy**	**Spain**
Constant	-0.933^{**}	1.614^{***}	0.436^{ns}
	(0.441)	(0.365)	(0.294)
Attributes			
Domestic use restriction	-4.741^{***}	-5.289^{***}	-8.603^{***}
	(0.670)	(0.815)	(0.739)
Environmental quality	1.584^{***}	0.766^{***}	1.345^{***}
	(0.113)	(0.080)	(0.078)
Water price	-0.022^{***}	-0.037^{***}	-0.028^{***}
	(0.003)	(0.003)	(0.003)
Respondent characteristics			
Restriction experience	0.976^{***}	0.968^{***}	-0.499^{**}
	(0.301)	(0.255)	(0.250)
Environmental disposition	0.199^{ns}	1.438^{***}	1.303^{***}
	(0.314)	(0.271)	(0.236)
Household income	$1.289 \cdot 10^{-4***}$	$0.261 \cdot 10^{-4**}$	$0.598 \cdot 10^{-4***}$
	$(0.215 \cdot 10^{-4})$	$(0.117 \cdot 10^{-4})$	$(0.957 \cdot 10^{-5})$
St. dev. random parameters			
Domestic use restriction	4.741^{***}	5.289^{***}	8.603^{***}
	(0.670)	(0.815)	(0.739)
Environmental quality	1.584^{***}	0.766^{***}	1.345^{***}
	(0.113)	(0.080)	(0.078)
Model summary statistics			
Log Likelihood	-818.110	-796.097	-1270.270
Pseudo R-square	0.371	0.157	0.251
N	1204	956	1544

*** $p < 0.01$; ** $p < 0.05$; ^{ns} not significant

The attributes are all significant at the one percent level and have the expected signs. Utility for the policy alternatives increases if the frequency of domestic water use restrictions decreases and environmental conditions of the water resources improve. Price has a significant negative effect on choice probability. Random taste variation is detected for both the domestic use restriction and water status attributes (see the significant standard deviation of the random parameters).

Preference heterogeneity is furthermore controlled for through the inclusion of individual respondent characteristics interacting with the alternative specific constants in each model. These characteristics include respondent experience with water use restrictions (dummy with the value one if the household faced water use restrictions over the past 10 years), attitude towards environmental conservation (dummy with the value one if water should be allocated to the environment first in times of scarcity) and household income (measured in Euros per household per year).

Previous experiences with domestic water use restrictions and household income are statistically significant in all samples, environmental disposition only in Italy and Spain and not in Greece. Income and attitudinal disposition towards the environment have, as expected, a positive effect on choice probability (the higher the income level or a positive disposition, the more likely someone is willing to pay extra). Respondent experience with domestic water use restrictions has a positive effect on choice probability in Greece and Italy (if someone suffered water cuts in the past, (s)he will be more likely to choose one of the policy alternatives to reduce domestic water cuts and/or improve environmental water quality at a given price), but a negative effect in Spain. One possible explanation for this negative effect may be that the limited experiences of the respondents in the Spanish sample with domestic water use restrictions are not as negative as in Greece and Spain and they might therefore be less willing to pay extra for improved future water security. The Spanish respondents in the sample have less experience with domestic water use restrictions due to water scarcity (one in five) than Greek (one in two) and Italian (one in three) respondents and also suffered, on average, significantly less frequently water use restrictions.

Based on the estimated choice models, implicit prices reflecting marginal WTP (MWTP) and compensating surplus (mean WTP) welfare measures to allocate water for domestic and environmental water use can be calculated. The implicit prices are presented in Figures 6.8 and 6.9. Although measured in different units and hence not directly comparable, implicit prices are, as expected, much higher for securing domestic water supply than environmental water use.

Figure 6.8: WTP to reduce the likelihood of domestic water use restrictions over the next 10 years by one year

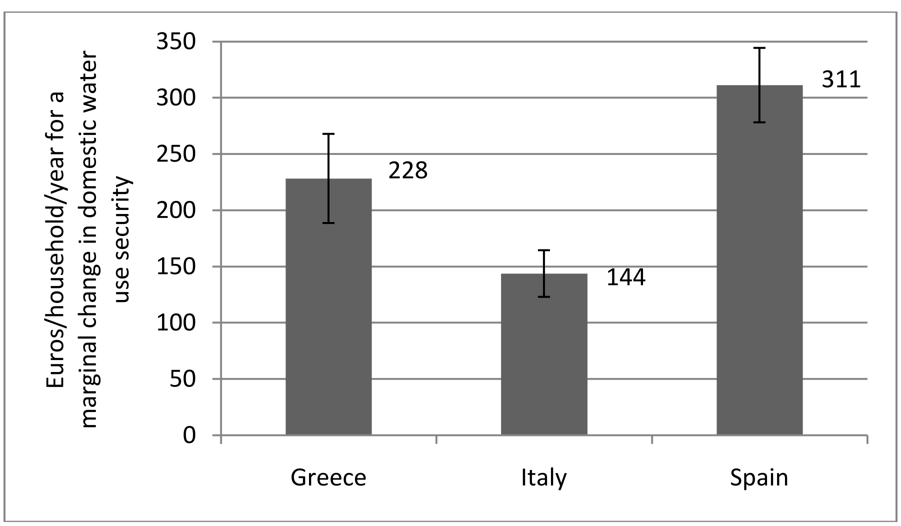
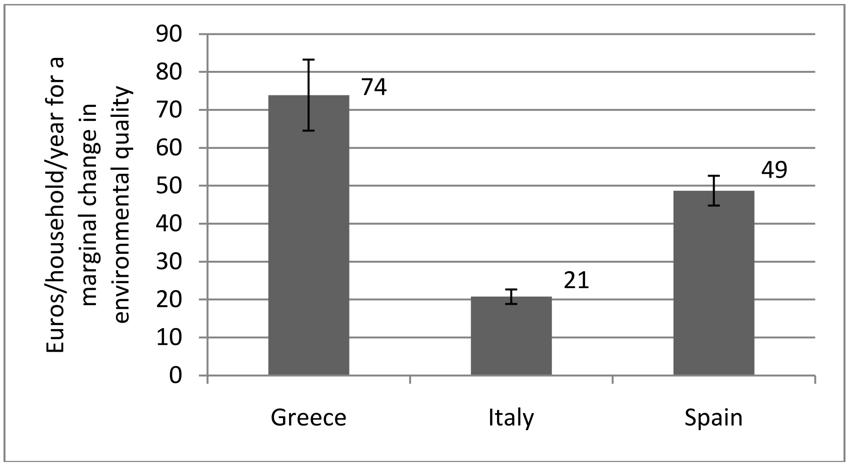


Figure 6.9: WTP to improve the ecological status of water resources over the next 10 years by one WFD quality category



In the case of domestic water use, the 95 percent confidence intervals around the implicit prices overlap in two of the three countries, indicating that securing domestic water use is valued more or less the same across these countries. On the other hand, preferences related to the allocation of limited freshwater resources to the environment differ in all three countries. Hence, significant differences exist between implicit prices for environmental water use across the three countries, but not for domestic water use. Implicit prices for securing domestic water use are not significantly different between Greece and Italy and Greece and Spain (at the 5% and 10% level respectively), only between Italy and Spain (at the 1% level). In the case of environmental water use, the implicit price is highest in Greece and lowest in Italy and differences are significant at the 1% level. Based on the high values found for domestic water use and public experiences with water supply cuts in the three countries, domestic indoor use values cannot entirely be ruled out.

Table 6.3: Estimated consumer surplus welfare measures for different policy scenarios (€/household/year)

Policy scenario	**Greece**	**Italy**	**Spain**
Improvement domestic water use security only			
1) Restrictions from 4 to 2 in 10 years	195.6	62.0	123.0
	(137.7 – 276.5)	(45.8 – 79.9)	(99.7 – 150.9)
2) Restrictions from 4 to 1 in 10 years	218.4	76.4	154.1
	(156.7 – 305.3)	(57.6 – 96.8)	(126.3 – 187.6)
Improvement ecological status water resources (non-use) only			
3) From poor to good status	297.7	74.7	158.1
	(224.8 – 402.6)	(58.7 – 92.5)	(132.4 – 189.1)
4) From poor to very good status	371.5	95.5	206.8
	(286.1 – 496.4)	(77.5 – 115.4)	(175.3 – 245.1)
Improvement domestic water use security and ecological status water resources			
5) Restrictions from 4 to 1 in 10 years	440.0	138.6	300.2
and poor to very good	(342.9 – 583.3)	(117.0 – 162.9)	(257.3 – 353.0)

95% confidence intervals between brackets

The compensating surplus welfare measures of five policy scenarios are also estimated (e.g. Bennett and Blamey, 2001), differing in the degree to which they capture domestic use values and environmental non-use values. Two policy scenarios only focus on an increase in domestic water use security, two policy scenarios on an improvement of environmental conditions of water resources (i.e. non-use) and one policy scenario on a combination of the two where both future domestic water use and environmental water use is secured (Table 6.3). In all scenarios, the improvement is a move away from the status quo or baseline conditions. As expected, the estimated CS increases when improving water use security (policy scenario 2 compared to policy scenario 1) and environmental water conditions (policy scenario 4 compared to policy scenario 3). The relative increase in CS for a one level improvement in environmental water conditions (from good to very good) is more or less the same across the three different countries (varying between 25 and 30%), whereas a further 50 percent decrease in the likelihood of domestic water use restrictions from once every five years to once every ten years results in two times higher increase of CS in Italy and Spain (24%) compared to Greece (12%). CS is highest for the fifth policy scenario where water is secured for both domestic use and environmental non-use.

6.4 *References*

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7. Water quality valuation: from WFD objectives to ecosystem goods and services

7.1 *Introduction*

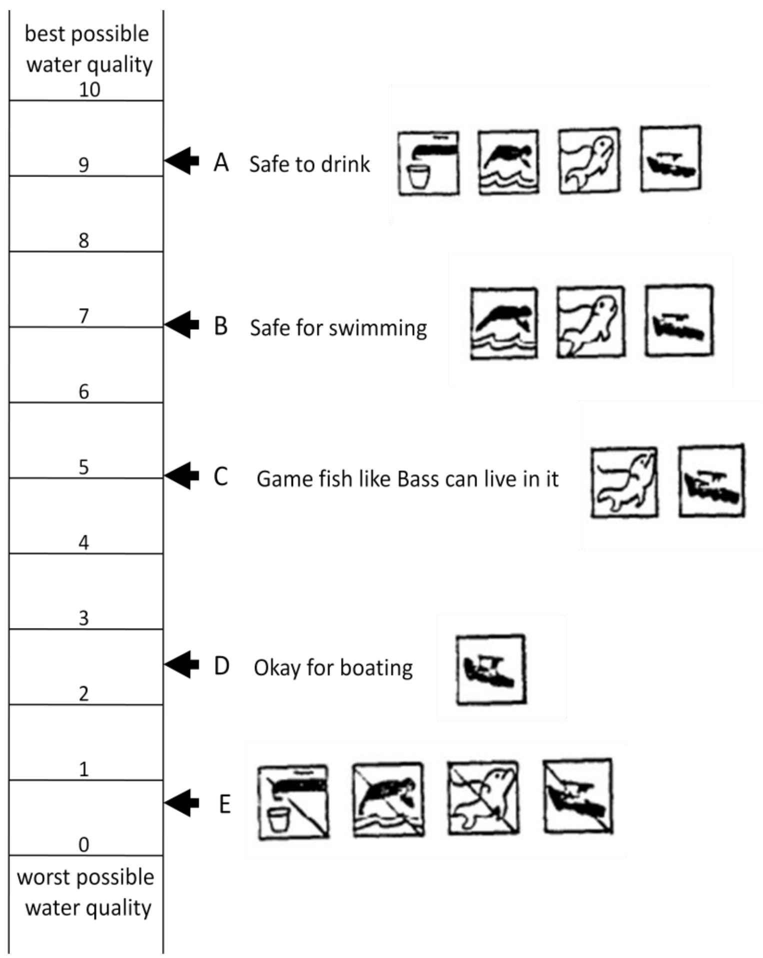
Any economic valuation exercise for a specific environmental change in water is based on a scientific understanding of the underlying environmental dose-response or damage functions. Ideally, reaching *Good Ecological Status* (GES) for natural water bodies and *Good Ecological Potential* (GEP) for heavily modified and artificial water bodies is defined in clear physical *state* (or impact) parameter values based on available knowledge and information about changes in (the pressures exerted on) the water system on chemical and ecological target values (e.g. concentration levels of priority substances or abundance of a specific species in the water column). In practice, even if these state parameter values can be determined based on ecological references, they are often surrounded by a lot of uncertainty as a result of limited knowledge about complex water systems. This uncertainty relates to the uncertainty in dose-pathway-effect relationships of reaching GES or GEP with the help of the proposed means or measures and the uncertainty about the environmental characteristics of GES or GEP and the stability of such an environmental system given ecosystem dynamics.

The relationship between economic values and these underlying physical environmental changes (via outcomes, functions, goods and services) is not always straightforward and has a number of important implications for the design of the valuation study and the analysis, interpretation and presentation of the results. Researchers applying stated preference methods like contingent valuation (CV) or choice experiments (CE) face an important 'information supply burden'. This means that they have to translate often complex and complicated scientific knowledge and information in lay public terms. Public participating in a stated preference survey should understand and be able to relate to the proposed changes in the water system in order to be able to derive meaningful answers and values. Scientific knowledge and information and the uncertainties surrounding this knowledge and information have to be translated into easy understandable terms in the valuation scenarios. The quality and quantity of information supplied in social survey research has important consequences for the outcome. Information effects have been tested extensively in the stated preference literature over the past decades. However, no specific, straightforward guidelines exist towards the optimal level and quality of information.

In this specific case, the question is how lay public can be best informed about existing and possible future water quality levels in an appealing, understandable and meaningful way. Related to this is also the question how knowledgeable the general public and/or specific water use groups are about the current water situation. Hence, the design of such information raises a number of empirical challenges. First and foremost, it must convey the nature of the proposed change in provision. To do so the information must initially define the current level of provision (e.g. the initial water quality) and the proposed future level of provision. This of itself can be complex where the technical information is challenging to the general public from whom the survey sample is drawn. Further complexity arises when natural variation means that in fact the current level of provision varies across locations (or indeed is liable to vary temporally due to ongoing changes). In such circumstances it is often common to find that the proposed final level of provision also varies spatially (and/or temporally). A second and often overlooked issue is to ensure that the information provided to respondents is such that the analyst can relate initial and proposed future quality states to existing environmental data to allow the extrapolation of derived values to real world locations and policy changes. So, for example, water quality states need to be related to existing data covering the extent of the desired study area.

A number of approaches have been used to convey information regarding water quality. Perhaps the most well known of these is the Resources For the Future (RFF) water quality ladder (Vaughan, 1986; Mitchell and Carson, 1989; Carson and Mitchell, 1993). This is a use-based measure, describing open water quality in an ascending scale from having no uses, to being suitable for boating, fishing and then swimming (Figure 5). These water functions were linked to the US EPA (environmental) water quality index.

Figure 7.1: RFF water quality ladder



7.2 *A WFD water quality ladder*

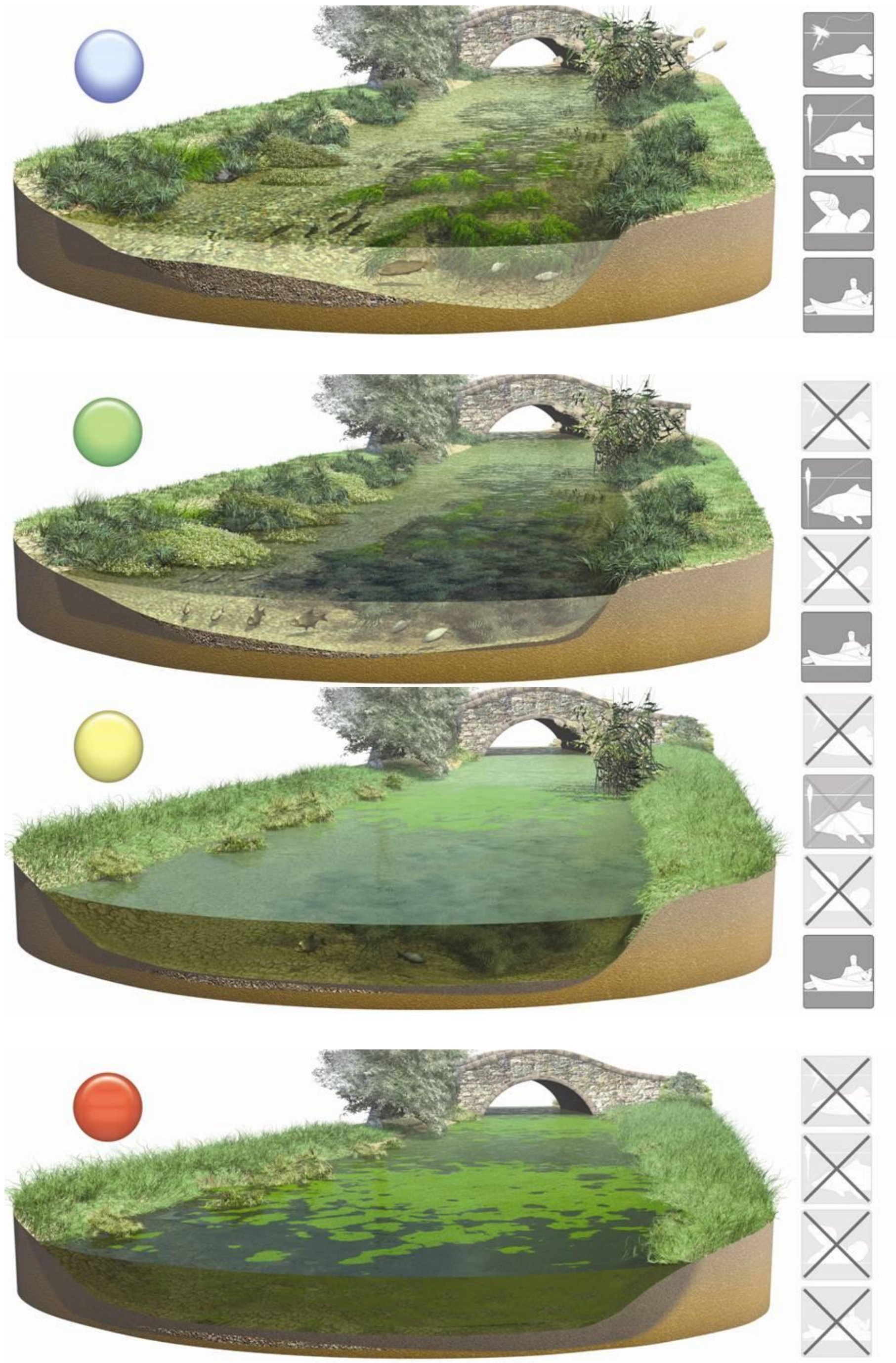
Variants of this approach have been used in a number of stated preference studies right up to the present day (e.g. Desvousges et al., 1987; Bateman et al., 2006; Bederli and Brouwer, 2007). While providing excellent service through the years, the categories used in the RFF are somewhat limited regarding the extent to which they convey the ecological changes (and associated use and, importantly, non-use benefits) implicit in movement up or down the ladder. Furthermore, the ladder focuses upon use categories which do not readily relate to national data on water quality (which to date typically tend to focus upon water chemistry measures). This limits the transferability of results. The limitations of the RFF water quality ladder have been thrown into sharper relief by the introduction of the EU WFD and the fundamental shift in the management of water quality in Europe to 'good ecological status'.

The WFD requires a substantial shift in assessment practice with the focus moving toward outcomes, in the form of ecological status, rather than chemical composition. However, this poses practical and methodological problems. To date there are few if any systematic assessments of ecological status in its entirety, where ecological status includes all features of the river environment such as aquatic plants, macroinvertebrates, bank-side vegetation and algae. Even the meaning of 'ecological status' remains the subject of a pan-European debate.

In order to address the problem of conveying the visible ecological response to changes in water quality of open-waters to samples drawn from the general public and to do so in a manner which ensures that resultant values can be transferred nationally and possibly internationally in the context of the WFD (see Chapter 11), a new water quality ladder was created (Hime et al., 2009), focusing on eutrophication (Figure 7.2). The process of eutrophication is a highly visible and well-documented phenomenon resulting in increases in algae and macrophyte biomass along with the decrease of macrophyte diversity, dissolved oxygen and water clarity with increases in pollution (e.g. Hilton et al., 2006). The process of eutrophication can be thought of in four key stages oligotrophic (described as having an aquatic plant community dominated by submerged macrophytes); meso-eutrophic (in which floating leaved macrophytes become dominant); eutrophic (where algae begin to out compete macrophyte communities); and hypereutrophic (where algae is dominant and macrophyte communities are replaced). The ladder eases survey respondents comprehension of these four stages by representing each with a separate colour: blue, green, yellow and red respectively.

The ladder has been designed to address not only the use value issues which have dominated previous such ladders, but also to capture ecological change (with its associated non-use benefits) and to relate all of this to available data concerning measures of water quality so as to enhance the transferability of value measures derived from using such a scale. Full details of the biophysical characteristics underlying the water quality ladder are found in Hime et al. (2009). A generic river was defined to control for various aspects that effect river classifications including the level of habitat modification, river flow rate, depth and width. The overall shape and depth of the generic river shown within the water quality ladder was specifically chosen to meet the requirements described in (River Restoration Centre, 2007) to support fish life and particular macrophyte species.

Figure 7.2: WFD water quality ladder related to eutrophication



Source: Hime et al. (2009). Copyright reserved.

Each ecological category within the water quality ladder was described by various attributes including fish life, aquatic vegetation, river bank vegetation, substrate composition and water clarity.

Box F. Economic Valuation of WFD Objectives, Not WFD Measures

It is important to point out that the economic valuation exercise is related to the end point or end situation of reaching GES or GEP, not the specific measures taken to reach GES or GEP. In other words, when using the valuation results in a cost-benefit analysis or for water pricing, the values should represent the welfare implications of the target situation (i.e. GES or GEP). So, when using revealed or stated preference methods, the public or stakeholders involved are not asked to value the measures and associated costs needed to reach GES or GEP, but the welfare implications associated with reaching GES or GEP (i.e. the individuals' experience of what that status confers in terms of their welfare).

To enable the water quality ladder to reflect the changes in water quality which implementation of the WFD might deliver, it was necessary to tie each ecological state to measurable chemical limits such as the expected BOD in each category of the generic water quality ladder. This relation allows for a translation from currently available measures of water quality to corresponding improvements in ecological status. In addition to the visualisations developed for each category, within the water quality ladder icons relating to the different recreational activities that can be carried out at rivers with certain categories of water quality (e.g. fish species expected and whether or not common birds were present) were added. Two additional icons relating to use i.e. swimming and boating were included and are affected most at extreme levels of eutrophication due to increases in algae (e.g. Pearson et al., 2001). Macro invertebrates and mammals were not overtly displayed within the generic water quality ladder illustrations. These would be hard to see at the scale of the illustration chosen, although separate information regarding such invertebrates could be presented. However, this was not included as the inclusion of rare, iconic species such as otters might unduly influence respondents into voting for particular ecological states despite the chances of viewing such species in the wild being relatively low.

It is best to demonstrate how current measures of water quality can be used in conjunction with the generic water quality ladder with a case study example. In order to display current water quality in terms of the categories defined within our generic water quality ladder several steps were taken. First, the mean current measures for BOD and Ammonia concentrations measured from 1986 to 1997 by the UK Environmental Agency (EA) were determined for each sampling station on three rivers Wharfe, Aire and Calder within the Humber catchment in North England, one of the case study areas in AquaMoney. These mean values were then converted into the appropriate generic water quality ladder measures, for instance where the mean BOD at sample point one was 5mgl-1, this was converted to the generic water quality ladder value of green. Finally, maps of the case study area were developed to show the mean generic water quality level of each sampling point taking into account either BOD (Figure 7.3) or Ammonia concentration (Figure 7.4).

Figure 7.3: The mean generic water quality category derived from BOD measures taken at sampling points along the rivers Wharfe, Aire and Calder

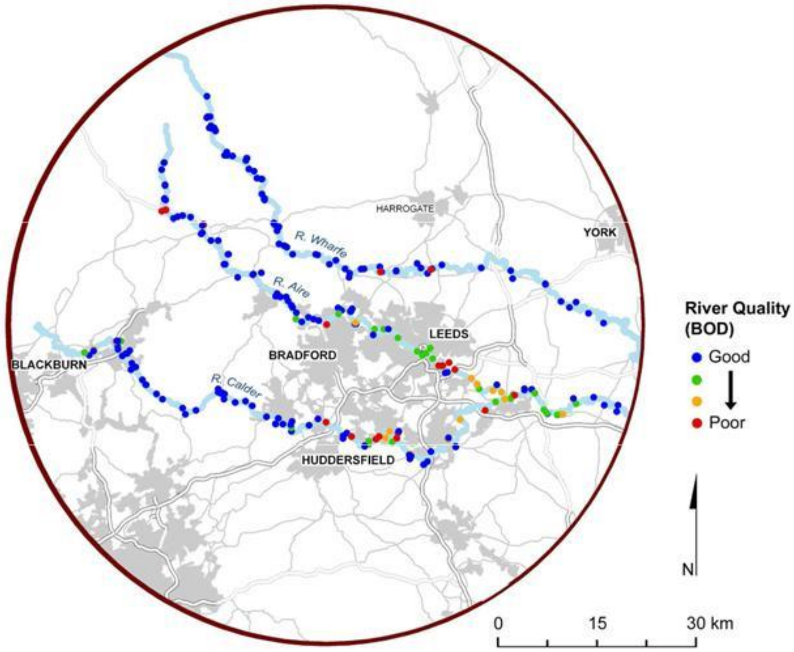
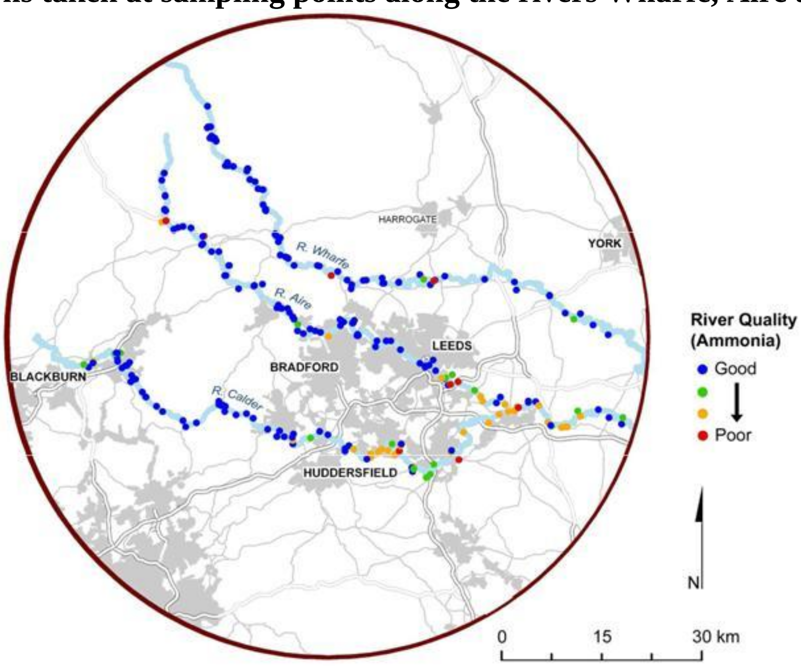
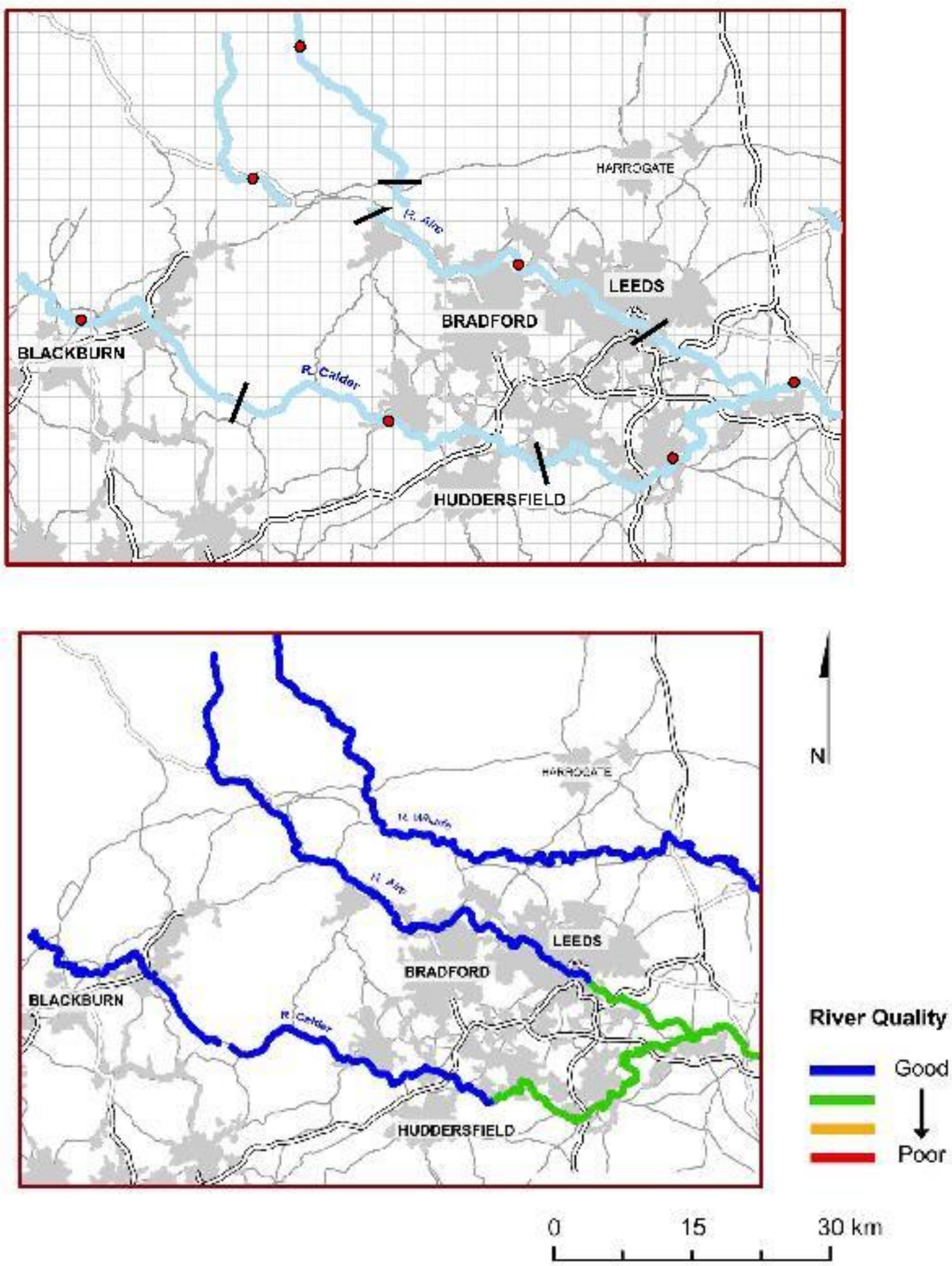


Figure 7.4: The mean generic water quality category derived from Ammonia concentrations taken at sampling points along the rivers Wharfe, Aire and Calder



In order to demonstrate how a benefits transfer approach might be applied in further valuation work these maps had to be simplified. To achieve this simplification the three case study rivers were split into nine stretches (see Figure 7.5). Each stretch was coloured to represent the water quality category within the generic water quality ladder. The appropriate category was calculated by determining the mean category for each stretch from the values of either BOD or Ammonia for each sample point within each stretch.

Figure 7.5: Current levels of generic water quality (derived from mean BOD measures and Ammonia concentrations) along the rivers Wharfe, Aire and Calder



The figures above describe the current status quo conditions of the Wharfe, Aire and Calder within the Humber region. However, maps depicting potential future leaves of water quality arising from the implementation of the WFD or other water quality policies can also be generated. Thus combinations of these maps allow valuation researchers to visually display in map form changes in the provision of water quality under alternative scenarios with reference to the generic water quality ladder. A comparison of current levels of water quality i.e. the status quo and various alternative schemes provide a potentially useful informational input into stated preference studies using contingent valuation, choice experiment and allied methods (see the next Chapters).

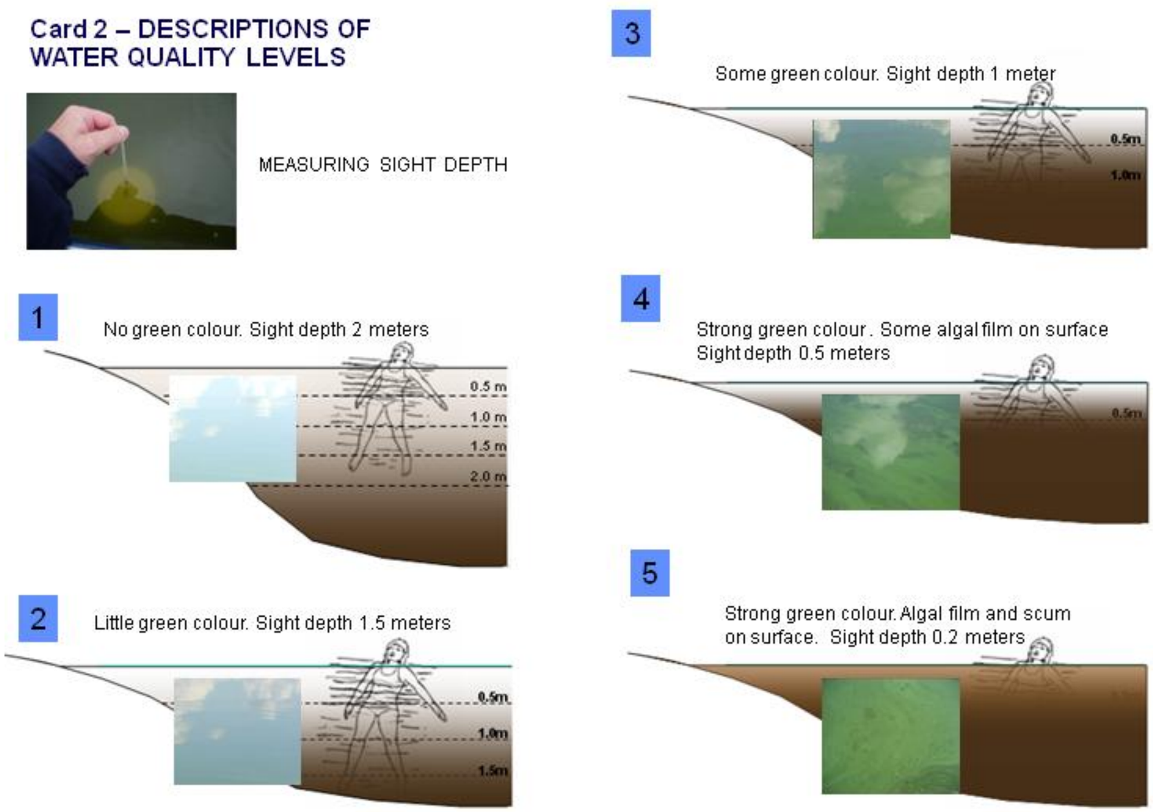
A key application of the generic water quality ladder is its relation to existing data on the spatial distribution of water quality along river stretches so as to generate water quality maps. The national coverage available for chemical measure data provides the requisite basis for the generation of a generally applicable water quality ladder and the estimation of the effects of improvements in those measures for any area of the country (providing the necessary science basis for the spatial transfer of associated benefit values). In addition, a significant advantage of basing valuation exercises upon water quality maps is the explicit incorporation of location within the valuation exercise. For example, Bateman et al. (2006) have shown that the location of improvements in relation to the individual can highly significantly affect resultant values. Through the use of a map based approach (linked to the spatial analytical capabilities of a geographic information system, GIS) the data necessary to parameterise a spatially explicit value function can be generated (see Chapter 12).

7.3 *Testing the water quality ladder: public perception and understanding*

The WFD water quality ladder improves on the RFF water quality ladder by including ecological components to the description of suitability for different water uses ("safe for drinking" etc.). The RFF water quality ladder – indeed any water quality focusing on suitability for water uses – makes the assumption that water users' perception of water quality are conform to the standards defined by authorities for suitability. In this section we show that when more information is included in the representation of water quality – including illustrations of esthetic qualities – the possibility of a 'standards-perception' divergence increases. This can happen when for example swimmers adapt to 'poor' water quality over time, they continue to swim in a lake or river perhaps because of other characteristics of the site not captured in the water quality ladder (proximity, lack of substitutes, landscape aesthetics), and as long as the water remains above their perception of personal use suitability level. This may be more likely for water bodies where natural conditions can be confounded with poor water quality. An example is humic lakes with high levels of natural organic materials, or rivers with large amounts of suspended sediment where water clarity measured as sight (Secchi) depth may be naturally lower.

In testing different approaches to water quality ladders, one version in AQUAMONEY focused on respondent perception of water colour and sight depth (figure 5.7). Respondents were asked to start by examining illustration number 1 and state 'whether you would bathe under such conditions'. If they answered yes they were asked the same question for number 2, and so on. The water quality descriptions were then used in a choice experiment alongside a bathing warning attribute ('months bathing warning') and a water and sewage fee increase attribute. The test was carried out in the Vansjø lakes in Norway.

Figure 5.7 – Description of water quality levels as sight depth and water colour

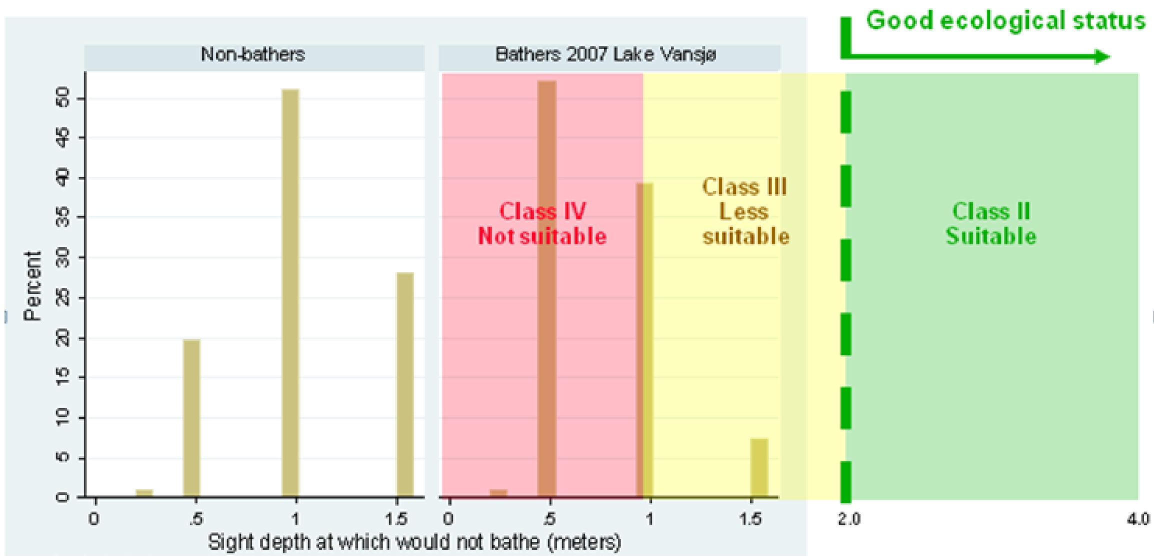


Source: Barton et al. (2009).

Figure 5.7 shows the percentage of 'bathers' and 'non-bathers' in the summer of 2007 who answered that they "would not bathe" at the different illustrated sight depths (0.2, 0.5, 1.0, 1.5 and 2.0 meters). For 'bathers' that summer the majority stated they would not bathe at 0.5 meters visibility, whereas for non-bathers that summer the majority said they would not bathe at 1 meter. The choice experiment showed respondents' utility for sight depth going from negative to positive between 1m and 1.5m. The choice experiment revealed that the threshold for positive utility was at higher sight depths (better water quality) than what was suggested by direct questions about suitable sight depths shown in Figure 5.7. This suggests that water quality improvements that make water quality merely 'suitable' for bathing may still not reveal positive willingness to pay by respondents in a choice experiment. In testing the WFD ladder, we found results that suggest that this may be due to a mismatch between improvements proposed and public expectations of water quality improvements.

In Figure 5.7 we then compare respondents' suitability thresholds to official Norwegian water use suitability standard and that of 'good ecological status' according to eutrophication criteria under the WFD for this lake type. According to the Norwegian authorities' guidance values for suitable water quality (SFT 1997), Class II is "suitable" for bathing Secchi depths of more than 2 meters. The definition of "good ecological status" for this lake type lies roughly at ChlA < 4 ug/l which corresponds to Secchi depth values 2.0-4.0 meters visibility. The question then is: "do the authorities' definitions of suitable water quality correspond to peoples' perceptions of what is suitable?"

Figure 5.7: Regulator versus user perceived recreational suitability

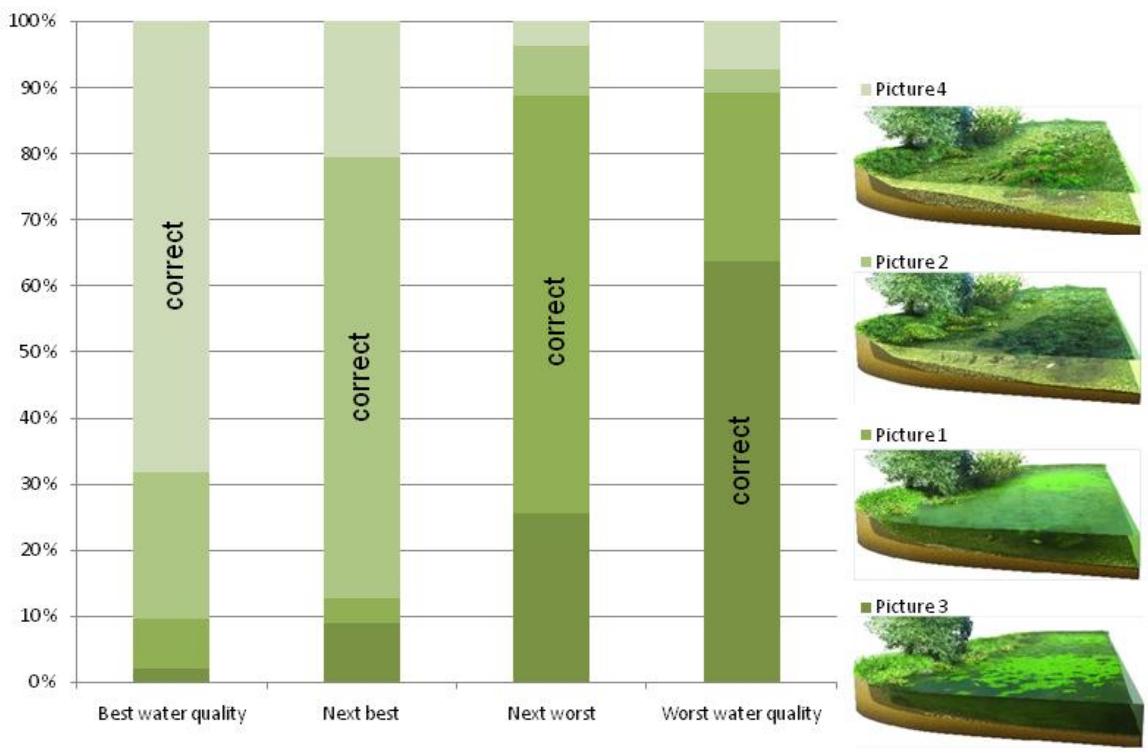


Source: Barton et al. (2009).

The comparison in Figure 5.7. shows that bathers had higher tolerance of eutrophication levels than the official standards. In both the 'bather' and 'non-bather' groups the definition of suitability was less strict than the Norwegian authorities' classification. Moreover, most respondents would bathe at water quality levels significantly lower than 'good ecological status'. Based on these results, Norwegian water quality guidelines are too 'precautionary' relative to water users perceptions and actual practices. This suggests that valuation studies using such official guidelines for suitability will tend to over-emphasise the benefits of water quality improvements on recreational uses such as bathing (which has the greatest contact with water). When we add other uses in the WFD water quality ladder we would expect the overestimation of benefits to be lower for recreational uses with less water contact. Figure 5.7 also suggests, at least for the local Norwegian population, that 'good ecological status' (GES) is more 'precautionary' than existing water use suitability standards, and more so than current perceptions of suitability suggest.

When we add ecological dimensions to the water quality description, we might expect some additional 'noise' due to some respondents interpreting the ordering of water quality levels perhaps differently from what researchers intended. In the Norwegian case study, respondent understanding of the WFD water quality ladder was tested by asking respondents to rank the water quality illustrations from best to worst. The results are shown in Figure 5.8.

Figure 5.8: Respondent ranking of water quality levels



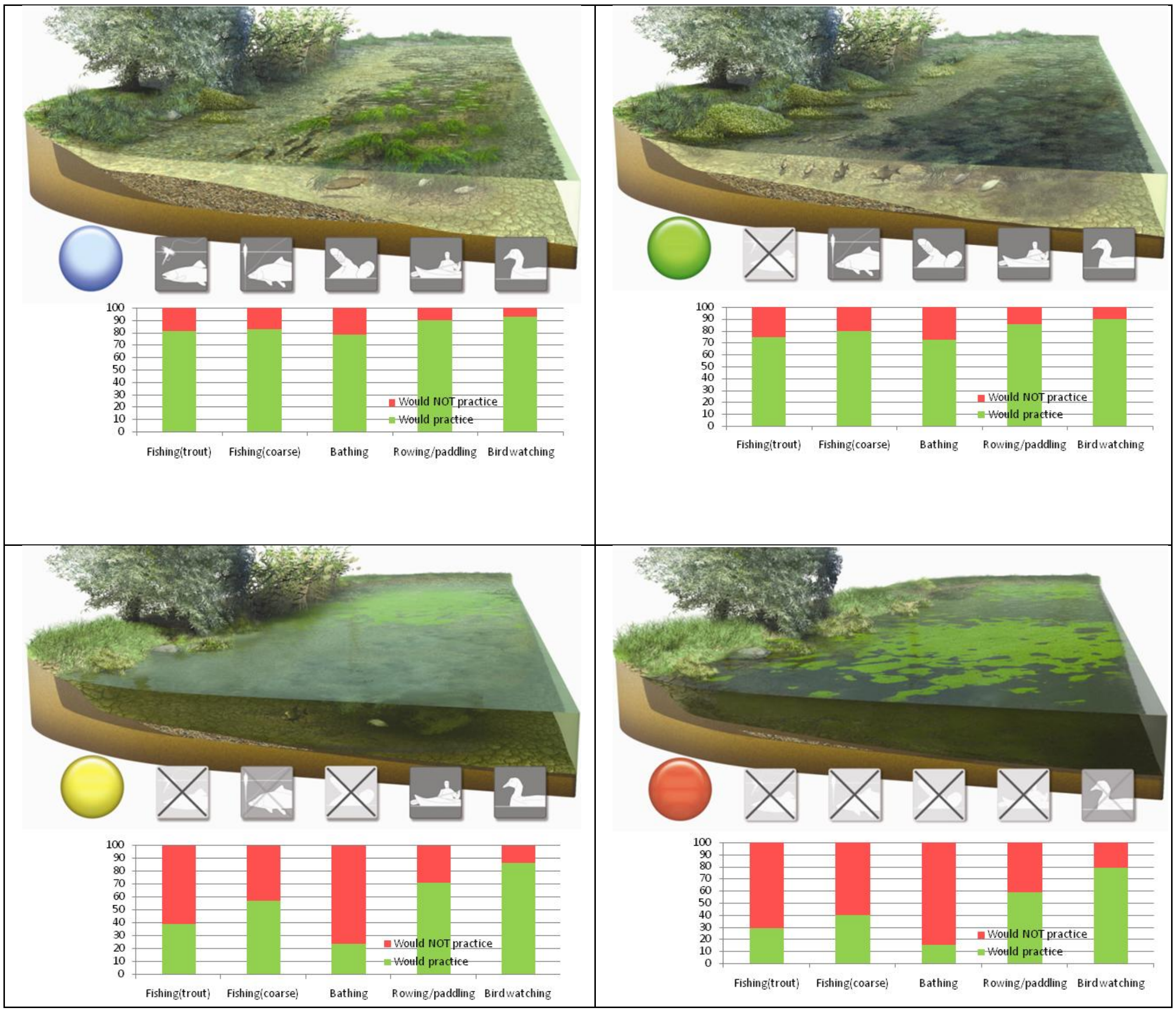
Note: the picture number shows the order in which they were numbered to respondents (the numbering gave no clues as to quality levels)

Source: Barton et al. (2009).

Between 65-70% of the sample ranked the water quality descriptions from best to worst as intended ("correct" section of the columns in Figure 5.8). About 20% confounded the best and second best levels relative to researchers' expectations; about 25% confounded the worst and second worst levels; and finally about 10% confounded the best or next best with the worst or next worst, and vice versa. The results in Figure 5.8 suggest one possible reason as to why some respondents' WTP may fail to show sensitivity to scope of changes in the Norwegian case study in particular, and in AQUAMONEY cases more generally – our results suggest lacking scope effects may be expected for about 10% of the sample even for increases of 2 water quality levels, and for up to about 25% of the sample for changes in 1 water quality level.

Finally, Figure 5.9 shows the results of the respondent answers to the question “which of the following activities would you not practice at this lake? Please click on the uses you would NOT practice if the water quality was as shown on the picture”.

Figure 5.9: Comparison of respondent perception of suitability of water uses with water quality ladder illustrations



Note: The water use icons show suitability at each water quality level as indented by researchers and presented in the valuation scenarios. They were not shown respondents until after asking respondent perceptions of suitability.

The responses in Figure 5.9 show that the pictures were broadly interpreted as intended in terms of monotonically increasing suitability, with highest use suitability across all uses in the "blue" level and lowest suitability in the "red" level. Also, uses which involved most contact with water (bathing) were deemed least suitable at each level, while uses with the least contact with water (bird watching) were most suitable at any given water quality level. This too suggests possible bias in benefit estimation when using official use suitability standards, which is expected to be higher for more water-contact intensive uses.

Large proportions of respondents had higher tolerance for eutrophication than what was deemed suitable by researchers. This was particularly the case for fishing and boating. As with the Secchi-depth water quality ladder, the WFD ladders' use suitability definition is too 'precautionary' relative to user perceptions. The results show a large change in perceived suitability between "green" and "yellow" quality levels. Notably, at the worst "red" quality level, 40% of the sample would still practice coarse fishing, and 60% boating, in stark contrast to suitability suggested by the water quality ladder. The water quality ladder also captures some unintended difference between "blue" and "green" quality level regarding swimming, possibly due to increased aquatic vegetation in the picture.

The results in Figure 5.9 provide two further reasons for the possible lack of scope effects for given water quality improvements:

1) small differences in user suitability perceptions between "red" and "yellow", and "green" and "blue" water quality levels
2) higher acceptability of eutrophication for certain uses such as fishing and rowing/ paddling than what was suggested by the water quality ladder.

These results suggest that the WFD water quality is expected to show non-linear WTP for consecutive increases in water quality, with the largest WTP expected for improvements to 'green', or 'good ecological status' (Hime et al. 2009).

In summary, the tests of the illustrated Secchi-depth and WFD water quality ladders suggest some cautions in interpreting the results of valuation studies:

- Respondent perception of use suitability are made more explicit and may diverge from what authorities deem is suitable for use.
- Use suitability thresholds may be significantly lower than good ecological status.
- Water quality improvements that attain use suitability do not necessarily have significant positive utility. In both the Secchi-depth and WFD water quality ladders respondents only had positive utility for improvements that went beyond the suitability threshold.
- Respondents may adapt their perception of suitability for use to current poorer quality, while maintaining expectations of a programme of measures to improve quality to a reference level (e.g. a regulatory standard such as GES, or restoration to a historically high level of water quality).

7.4 *References*

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8. Scale: from water body to river basin district

8.1 *Introduction*

The importance of space in economic valuation is nowhere better illustrated than in travel cost or hedonic price studies where the economic value is determined by the cost price of travelling to a specific water site or encapsulated in property prices. For instance, the further away someone lives, the higher the travel cost and hence – so it is assumed – the value attached to the site involved (measured through WTP). However, space may play various roles in economic valuation studies. In principle, the spatial dimensions underlying economic valuation of water resources and their functions relate to:

1) the spatial distribution of individual water bodies across the watershed or river basin and their physical characteristics relevant for economic valuation (e.g. shape, size, function, quality, accessibility, distance etc.);

2) the relationship between spatially interconnected (e.g. upstream-downstream) water bodies in terms of their functional complementarity or substitutability;

3) the spatial distribution of the population of beneficiaries or polluters inside and outside the river basin and their specific characteristics relevant for economic valuation (e.g. income levels, perception of quality, distance and accessibility etc.).

Space has traditionally been interpreted in terms of distance between objects. For example, the distance between an angler's place of residence and the place where (s)he fishes. Public perception of space and distance may differ from the traditional 'objective' measures. Travel cost studies have shown that public perception of distance may differ significantly from the 'objectively' measured travel distance and provide a stronger explanatory factor of travel behaviour. Subjective location specific factors (e.g. place of birth) may play an important role in someone's value attached to a specific place, including the presence of water in a landscape. The spatial distribution of water bodies and their users and non-users is expected to affect the economic value attached to the different water bodies and the river basin as a whole in a number of ways.

Often, economic valuation studies have attempted to use visual material in order to depict spatial dimensions. For example, maps have often been used to support stated preference surveys (see, for instance, Kramer and Mecer, 1997; Bateman and Langford, 1997). The use of maps entails two main advantages:

i) the possibility to asses spatial variability;
ii) the capacity to include more information in a more user-friendly and communicative way.

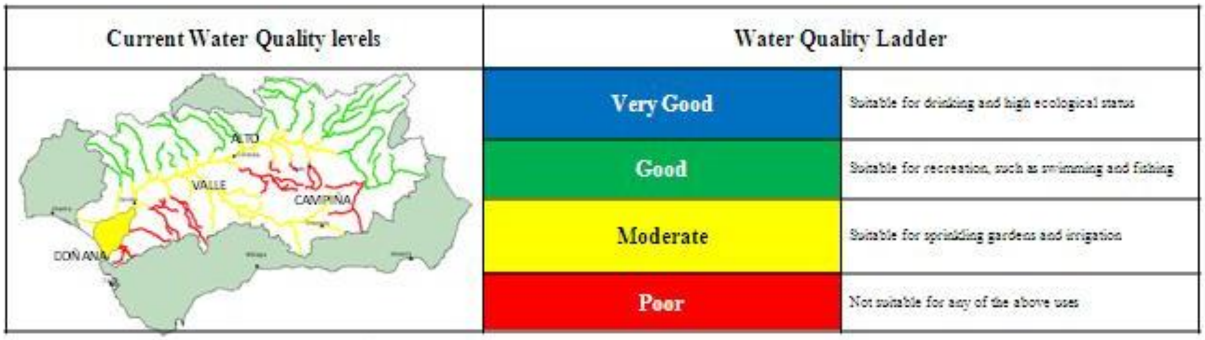
8.2 *Case study illustration*

Here an example is presented from one of the AquaMoney case studies carried out in the Guadalquivir River Basin (GRB) in South Spain. The Guadalquivir is the longest river in the south of Spain with a length of around 650 km. The GRB is an interesting case study, because it is perceived and referred to by the population living in the basin as a clearly defined geographical unit. The basin covers an area of 57,527 km^2 with a population of over 4 million people, and has been extensively studied in the context of the WFD. Water quality is a significant problem throughout the river basin. The main sources of pollution include urban and industrial wastewater discharge, erosion, and nutrient and pesticide runoff from agricultural land. Concentration levels of nitrogen, phosphorus, heavy metals and organic pollutants in surface and ground waters are expected to increase even further in the near future. More details are found in Martín-Ortega et al. (2008).

In the case study, the GRB is divided into four distinct sub-basins (Figure 8.1): *Sierra Morena and Alto Guadalquivir* in the north (*'Alto'*), *Valle del Guadalquivir* in the centre of the basin (*'Valle'), Campiña* in the south and the *Doñana National Park* at the mouth of the Guadalquivir river in the south-west (*'Doñana')*. The sub-basins differ in terms of water quality levels, but also with regards to the homogeneity of the landscape, land use and population characteristics. *Alto* can be characterized as a mountainous, low populated, extensive agricultural area. *Valle* is the valley through which the main Guadalquivir river stream flows, highly fertile with intensive agricultural land use and the highest population density, concentrated in the cities Sevilla, Cordoba and Granada, and some of the biggest industrial areas in the region. *Campiña* has a low-land agricultural landscape with a number of medium-sized cities. *Doñana* is home to a wide variety of rare protected species like the Iberian lynx and the Imperial eagle.

The current situation in the GRB is presented in Figure 8.1. The water quality levels (poor, moderate, good, very good) are described along the lines of the RFF water quality ladder (see Chapter 5) in terms of their consequences for different types of water use and environmental risk. This approach was understood best by lay public. During the pre-tests, a technical description of water quality levels based on physical indicators appeared too hard to understand for the general public, resulting in non-significance of the attributes as in Hanley et al. (2006). Moderate water quality is described as suitable for sprinkling gardens and irrigation, good water quality as suitable for recreation like swimming and fishing, and very good water quality as suitable for drinking water, reflecting at the same time a high ecological status of the water environment. As in the original RFF water quality ladder, each level includes the quality characteristics of the level below. The map is based on the WFD article 5 report for the GRB and was developed in collaboration with the Confederación Hidrográfica del Guadalquivir (River Basin Authority).

Figure 8.1: Current water quality levels in the Guadalquivir river basin

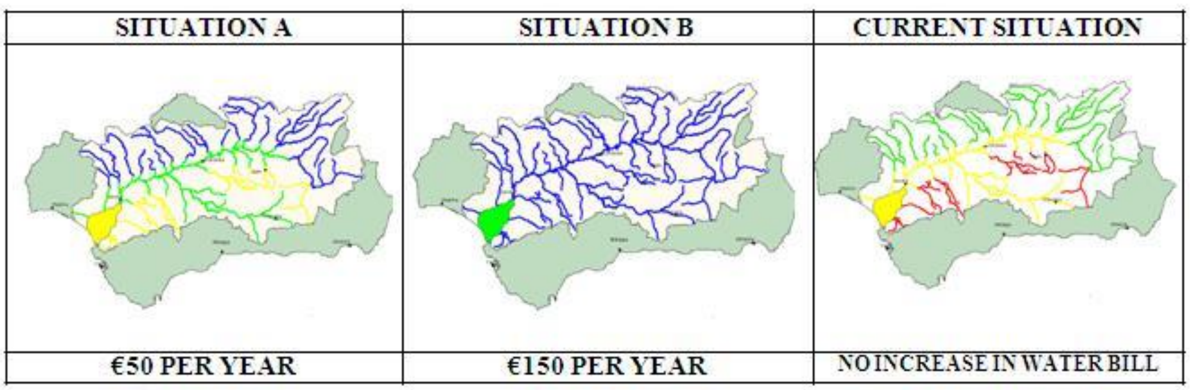


The focus of this case study is to value water quality improvements of different sub-river basins in a wider river basin context and account for the different values allocated by residents of different areas to the sub-basins and the river basin as a whole. The WFD distinguishes different water quality levels from poor to very good and requires that the economic analysis of water quality improvement is carried out both at river basin and water body scale. For this purpose, respondents were presented in a choice experiment different cards, each of them showing three maps of the GRB referring to policy alternatives defined in terms of water quality improvements in specific sub-basins throughout the river basin. The maps in the choice experiment were used as a visual aid to help respondents see where they live in relation to the different locations of the proposed water quality improvements. The water quality improvements can be realized at a certain price to be paid through the household water bill. People are asked to choose between the policy scenarios in which water quality of the sub-basins improves to different levels (situations A and B), compared to the current situation, which is expected to deteriorate if no additional measures are taken to improve current water quality.

Thus, the hydro-geographical units making up the river basin and the levels of water quality improvement are included as attributes in the choice experiment. Situation A and B always represent improvements with respect to the baseline situation, i.e. water quality in each sub-basin is equal to or better than the baseline situation. This results in a one level improvement for *Alto* (from good to very good), two levels of improvement in the case of *Valle* (from moderate to good and from moderate to very good), three levels of improvement for *Campiña* (from poor to moderate, from poor to good and from poor to very good), and two levels of improvement in the case of *Doñana* (from moderate to good and from moderate to very good). The water quality improvements in *Doñana* mainly benefit the environment and wildlife, and were also presented as such in the experiment. An example card is presented in Figure 8.2.

The study presented here differs from previous work focusing on part-whole bias in CV (e.g. Brown and Duffield, 1995) showing that WTP is sensitive to the information provided about the number of rivers protected. The approach allows us to account for the spatial distribution of the improvement of water services and their beneficiaries. Respondents living in different parts of the river basin are asked to value simultaneous water quality changes across different parts of the basin, enlarging the choice set in a spatially explicit way and hence enabling to more adequately capture possible substitution effects (see Chapter 9 for a more detailed discussion about substitution effects). The relevance of the work presented here is found in the design of appropriate rules of aggregation when aiming to estimate a total economic value for an environmental change such as water quality improvements, where both the environmental good in question, the population of beneficiaries and their preferences are expected to be asymmetrically distributed over space. This is particularly relevant for the implementation of the WFD where baseline conditions and the population of beneficiaries vary across European river basins. The WFD furthermore offers the opportunity to lower water quality objectives for all water bodies or delay their realization in time based on the concept of disproportionate costs. For this, a comparison of the expected costs and benefits and their distribution across water bodies and water quality levels is needed.

Figure 8.2: Example choice card



The approach presented here also differs from conventional testing of distance-decay effects in stated preference research (e.g. Bateman et al., 2006) in that variation is introduced not only in the spatial distribution of respondents who benefit from the change in environmental good provision in relation to one specific site or area (e.g. lake or river stretch), but in relation to multiple locations at the same time without relying upon unidirectional measurement units such as kilometers.

In order to obtain the necessary data to estimate the choice model, more than 600 face-to-face interviews were conducted in October 2006 throughout the GRB, targeting a random representative sample of the urban and rural population in 9 different municipalities equally distributed across the three populated sub-basins. The information provided in the questionnaire, including the maps displaying current water quality levels throughout the basin and the water quality ladder was developed and tested in collaboration with the water experts of the GRB Authority.

Water appears to be a very important issue for most sample respondents. Over 90 percent consider water issues one of the most important problems facing Spain. Forty percent perceives water quality in the GRB as poor. Less than 25 percent of all respondents believe that current water quality is good or very good. Only 10 percent of the sample population recreates in the Guadalquivir river and its tributaries, such as fishing and swimming. Forty percent is well aware of his or her current water bill, stating an average value for the current water bill (€250/household/year), which is not too far off the average household payments reported by the Spanish Ministry for the Environment (Ministerio de Medio Ambiente, 2007).

As we were interested in finding out to what extent it matters where respondents live as to what value they attach to water quality improvements across the basin and whether respondents value their own sub-basin more than the other sub-basins making up the GRB, two different models were estimated based on the collected data, one accounting for where people live throughout the basin and one not accounting for where people live in relation to where the water quality improvement takes place (Table 8.1). Full model details are found in Brouwer et al. (2009).

Table 8.1: Estimated WTP values (€/household/year)

				Choice model not accounting for where people live		**Choice model accounting for where people live**	
Policy scenario			**Basin**	**WTP**	**95%CI**	**WTP**	**95%CI**
Poor	→	Moderate	Campiña	48.9	37.8-60.5	45.2	29.1-62.5
Poor	→	Good	Campiña	59.5	46.4-73.1	58.6	41.7-76.9
Poor	→	Very good	Campiña	80.3	67.6-93.4	65.4	48.3-86.3
Moderate	→	Good	Valle	42.8	31.1-54.6	37.6	21.6-55.9
Moderate	→	Good	Doñana	9.6	-1.9-20.4	10.5	-7.4-26.9
Moderate	→	Very good	Valle	54.5	43.6-66.4	41.7	24.8-59.0
Moderate	→	Very good	Doñana	13.4	2.2-25.0	34.3	17.9-51.3
Good	→	Very good	Alto	10.5	1.3-19.0	13.9	-0.7-27.6
Current	→	Good	Whole	102.3	83.0-122.7	135.0	102.4-172.1
Current	→	Very good	Whole	158.7	134.5-184.7	211.9	169.0-257.3

The results presented in Table 8.1 show that WTP related to improvements in water quality across the sub-basins increase as the change in water quality increases, reflecting sensitivity to scope. The highest value is found for the change from a poor to a very good state in *Campiña* and the lowest value for the change from a good to a very good state in *Alto*. This is an interesting finding as it confirms that the biggest improvement in water quality (from poor to very good) has the highest value and the smallest improvement (from good to very good) the lowest value. An important observation from these findings is furthermore that the value of water quality improvements is spatially not uniformly distributed, partly due to differences in baseline conditions, but also because of significant preference heterogeneity. This is what the results related to the change from a moderate to a very good state in *Doñana* and *Valle* for example suggest. The baseline conditions are the same in these two sub-basins while the estimated values are significantly different.

An important second observation is that throughout the entire river basin the change to very good quality is valued in a significantly positive way in all sub-basins, but local residents place an additional value on this change in the specific sub-basin where they live compared to non-residents. Put the other way around: residents in one sub-basin value water quality improvements in another sub-basin, but no more than in their own sub-basin. For instance, local residents in the sub-basin *Valle* put an extra value on the change of water quality in their sub-basin from moderate to very good, and the same applies to the local residents in the sub-basin *Campiña* when they value the change from poor to very good in their sub-basin. Irrespective of the sub-basin where people live, the improvement of water quality to a *good* state is valued equally by local and non-local residents, but when improving water quality one level further to *very good* quality one has to account for the fact that local residents place an additional value on this improvement if it occurs in their own sub-basin. This has important implications for the welfare estimation procedure at river basin scale shown in the last two bottom rows of Table 8.1.

We find that when comparing mean WTP from the second model where we account for where people live with the value from the first model where we do not account for where people live, the latter is significantly lower than the former. Not accounting for how people value water quality changes throughout the river basin in relation to where they live results in an underestimation of the estimated economic value for improved water quality in the whole river basin. So, accounting for the finding that inhabitants in one sub-basin also hold values for water quality improvements in other sub-basins, these values can simply be added in order to obtain a total economic value for good water quality conditions throughout the river basin, but not in the case of the highest water quality level (very good), which is what the WFD ultimately aims for. In the latter case we have to include the additional value local residents attach to reaching very good water quality in their own sub-basin, yielding a significantly higher aggregate welfare measure across the individual sub-basins of 30 percent. This approach supports the derivation of economic welfare measures under circumstances where the physical characteristics of the ecosystem change (water quality changes in a river flow within a basin), the associated change in water services, and the population of beneficiaries are spatially distributed in non-uniform way.

Our interpretation of this result is that respondents have preferences for water quality improvements to acceptable levels throughout the entire river basin, but are not willing to pay extra to reach a more than good condition elsewhere, only in their own sub-basin. This is supported by the fact that we find no significant WTP by non-local residents for the improvement of water quality in *Alto*, the only sub-basin in the GRB where water quality levels are already in a good state. On the other hand, in the case of *Doñana* only very good water quality is considered worth paying for. We suspect that this is due to the high ecological value attached to the National Park, irrespective of the question whether respondents have visited the National Park before or intend to visit it in the future.

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9. Accounting for substitution effects

9.1 *Introduction*

Water resources provide different goods and services, some of which may be more or less replaceable through the use of other water resources located nearby (in the same basin) or further away (in the same basin or even outside the own basin). Different water bodies may provide different functions and goods and services. Lakes and ponds may provide, for example, different functions than rivers and creeks. The latter can be used for instance for recreational fishing on specific fish species not present in lakes or ponds, while the former may be more suitable for swimming and bathing. Hence, the substitutability of water bodies depends among others on the specific characteristics of the water system involved.

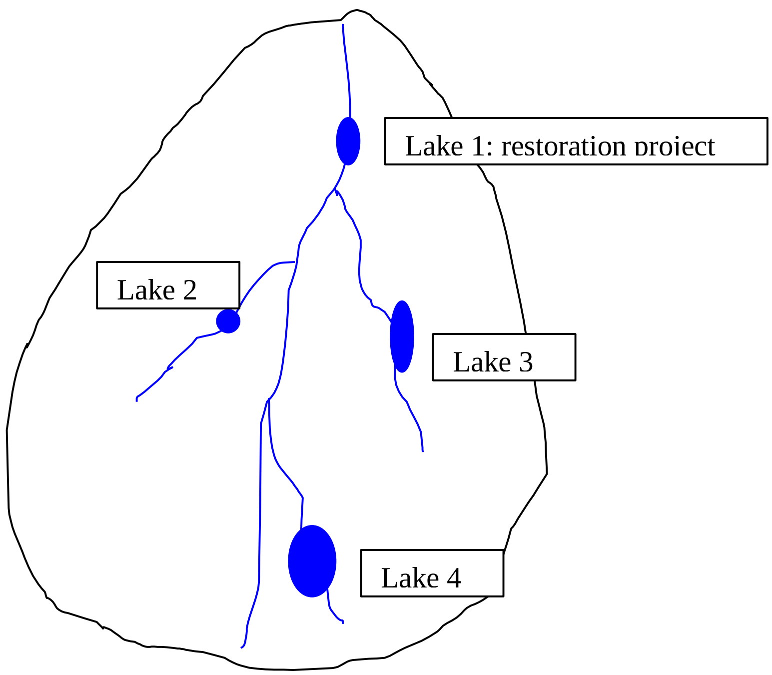
Certain water types may be (perfect) substitutes (e.g. irrigation versus domestic water use), while others are more complementary in terms of the provision of different functions (e.g. nature and recreation (not disturbing wildlife reproduction cycles, e.g. during nesting periods), and this may change as a result of interventions due to the implementation of the WFD. An example of complementarity is a system of interlinked lakes and ponds where the public is able to swim and sail from one lake to the other.

Different water use functions may be substitutable within the same water body, when space for recreational activities is limited, for example, due to congestion or overcrowding. A possible solution in those cases is spatial zoning, which assigns different functions to different parts of the water body or allows different functions at different points in time (e.g. boating from September until April, no access due to nestling of water birds during May and June and bathing in July and August).

Substitution possibilities are visualized in Figure 9.1. Figure 9.1 illustrates a number of important spatial aspects of water quality scenarios and valuation:

1) Restoration of the upstream located lake 1 or cleaning up pollution in lake 1 (e.g. elimination of point or non-point source pollution) may also influence water quality in the rest of the water system, including the other lakes due to the spatial (upstream-downstream) interdependencies of the water system.
2) The lakes may be either complements or substitutes, depending among others on the functions they perform. For example, are they all used and useable for recreational activities such as bathing, boating, fishing, bird watching and walking, or do they contain specific functional characteristics, making each of them relatively unique and less replaceable? The perception of recreational possibilities and uniqueness may vary across the population within the catchment.
3) The distance from the population residences and the accessibility of the water bodies will affect their substitutability.
4) Considering the effect that distance may have on the preferences for different water bodies, the population density around the lakes is important to keep in mind in the aggregation procedure of individual WTP to catchment level.

Figure 9.1: System of interdependent lakes and rivers



Although the average economic value for each water body (in this case lakes for example) may be the same throughout the entire basin, the total economic value (TEV) may differ significantly depending not only on the distribution and densities of the populations of beneficiaries involved, but also the substitution possibilities involved. This in turn depends on user group perception of the substitution possibilities involved and their determinants (e.g. distance, accessibility, etc.) In the case of the established relationship between WTP and distance (so called distance decay), substitutes introduce 'thresholds' where WTP falls more rapidly to zero in the presence of substitutes than without.

Despite the increasing use of Geographical Information Systems (GIS) to include a wide range of spatial variables and site-characteristics in valuation research (e.g. Bateman et al., 2002), only few studies address spatial substitution patterns. Substitution effects between sites have long been neglected in spatial choice models for non-market valuation. The latter focus on choices among sites as a function of site access and site-characteristics, for example, water quality changes. Substitution effects are expected to be especially relevant when estimating the non-market benefits of simultaneous ecological quality improvements across sites in a confined geographical area, for instance in catchments with multiple water bodies that provide a wide variety of use and non-use values. Many examples of travel cost studies of multiple-site recreation exist that estimate the impact of site-characteristics on site-utility (e.g. Parsons et al., 2003). However, there are surprisingly few stated preference (SP) studies in non-market valuation that account for substitution effects on WTP for environmental benefits that may arise when multiple changes take place simultaneously. Contingent Valuation (CV) studies usually focus on a single site, and only remind respondents of other expenditure options in the survey text (following the 1993 Blue Ribbon Panel recommendations). Because the CV method is limited in its ability to describe contextual changes, Choice Experiments (CE) have been argued to be able to better model substitution effects that arise if multiple sites are available (e.g. Rolfe et al., 2002).

In travel cost studies, the rate of substitution between sites is reflected by the correlation of the travel cost to one site with that of a substitute site (Freeman, 2003). The substitutability is in then directly related to distance. The main disadvantage of travel cost for environmental valuation is that they are based on revealed preference data and therefore do not capture non-use values. However, although SP studies can estimate both use- and non-use values for hypothetical quality improvements, the majority of studies focuses on a single site and do not account for changes in the availability or characteristics of nearby alternatives. Stated WTP estimates will be biased if respondents focus only on one particular location without fully considering the alternatives. The effectiveness of simply including a reminder of likely substitutes has been questioned (e.g. Loomis et al., 1994; Whitehead and Blomquist, 1999). Sequencing and scope-sensitivity effects found in CV studies are often attributed to substitution (Carson et al., 2001), but the results of these studies do not always give valid indicators of substitution effects, considering the lack of a priori information about the substitutability of the goods in question and the extraordinary magnitude of these scope effects (Bateman et al., 2004). A small number of CV studies include multi-programme scenarios in which different goods are valued simultaneously to test for substitution and complementarity effects (e.g. Hailu et al., 2000), but usually these studies are limited to the estimation of the effect of the availability (absence or presence) of alternative policy scenarios at different locations. They do not estimate substitution indicating to what extent the economic value of one alternative responds to changes in the characteristics of another alternative. This issue was addressed in the Dutch part of the Scheldt case study in AquaMoney.

9.2 *Case study illustration*

In the case study, differences in the substitutability across sites resulting from differences in site-characteristics were assessed alongside their effect on willingness to pay estimates. A site-selection CE was developed to estimate substitution effects across sites on WTP estimates for the use and non-use benefits of increases in ecological quality due to WFD implementation. Respondents were asked to choose the site they prefer to be improved based on different water management scenarios, presented as labelled options with site-specific attribute descriptions. Because the case-study sites differ in their provision of natural and recreational amenities, on-site improvements of ecological quality were expected to generate site-specific values. Moreover, substitution patterns between the sites were not expected to be conform the restrictions underlying standard statistical choice models[6]. Many studies often involve more than two alternatives of which some are perceived as being more similar than others.

Choice experiments have rarely been used to estimate substitution effects between sites in non-market valuation. A practical shortcoming of applying stated CEs in site-selection studies is that the number of alternatives that can be included in the design is limited, and may be substantially less than the real number of alternatives in the study area. Choosing among all alternatives may furthermore be hard, if not impossible, due to the cognitive burden this would impose on respondents. To demonstrate the limitations of single-site SP studies in areas with multiple alternatives, this case study assesses the impact of nearby substitute sites on WTP for an individual site by presenting respondents with similar ecological quality improvements at multiple sites simultaneously. A site-selection CE was developed with alternative specific attribute descriptions and labelled alternatives based on location names.

Three different locations were chosen after pre-testing to limit the cognitive burden of including too many different sites in the choice task. The three case-study sites are the beaches near Breskens, the Braakman-creek, and the “Verdronken Land van Saeftinghe” (see Figure 9.2). These sites are located in the south-western province Zeeland in the Netherlands in the Scheldt catchment and are popular for water recreation. The sites are subject to ecological quality improvements under the WFD. Quality levels at the three sites currently do not meet the WFD’s ecological standards. Reaching the WFD objectives implies different bio-physical outcomes for the three locations and the foreseen quality improvements are therefore explained by site-specific descriptions. Site characteristics that are not affected by the WFD, and hence not included explicitly in the choice task, but that are expected to affect choices, are captured by including the site names as labels.

The sites represent the most important water body types in the catchment, provide typical water-recreation functions, and are well-known among local residents. For all three sites expansion plans exist. Breskens is a popular beach site attracting local, national, and even international visitors. Braakman is a creek with brackish water, best known for family water recreation. Saeftinghe is an ecologically valuable tidal mud plain, which provides habitat for various protected species. This site attracts mostly nature enthusiasts, who can only visit the site accompanied by a guide. The three sites hence differ in the recreational functions they perform.

The WFD objectives were translated for each site into three attributes reflecting the WFD’s potential use and non-use benefits considered important by respondents and representing the site-characteristics subject to possible changes under the WFD: walking, bathing, and nature and wildlife amenities. Previous studies showed that attributes based on technical descriptions of water quality levels and bio-physical WFD indicators may be hard to understand for the general public, and result in insignificant parameter estimates

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[6] The Independence of Irrelevant Alternatives (IIA) property states that the relative probabilities of two options are unaffected by other alternatives and thereby imposes proportional substitution rates on alternatives (Kanninen, 2007).

(Hanley et al., 2006). The implications of the WFD objectives for the site characteristics were therefore translated in easily understandable and meaningful lay public terms.

Figure 9.2: Location of the three selected sites

Table 9.1 gives an overview of the attributes and their levels. Although the same attributes were used for the three locations, different photographs and textual descriptions were used to explain the attribute levels in order to accommodate the site-specific characteristics. For example, specific bird, fish and hydro-morphological indicators were used to represent the different attribute levels at the three sites. Additionally, restrictions were included in the CE to increase realism. For example, contrary to the current 'bad' quality levels of most attributes at all three sites, the current bathing water quality at Breskens is moderate, and the status quo at this site was therefore set at that level. Furthermore, the bathing water quality attribute was excluded at Saeftinghe, because there are and will be no bathing possibilities. The fourth attribute in the CE was a monetary attribute, expressed as an increase in annual water board taxes paid yearly by all local households.

Table 9.1: Overview of attributes and levels

WFD objective	Attribute	Levels	Description
Hydro-morphology	Walking	Good	Area expansion and natural banks/dunes
		Moderate	No area expansion, natural banks/dunes
		Bad	*Status quo*: stone (artificial) banks/dunes
Water quality	Bathing	Good	High clarity, no algal blooms or odour
		Moderate	Moderate clarity, occasional algal blooms and odour
		Bad	*Status quo*: low clarity, frequent algal blooms and odour problems
Ecology	Nature and wildlife amenities	Good	High number and diversity of fish and birds, including rare species
		Moderate	Moderate number and diversity of fish and birds, few rare species
		Bad	*Status quo*: low number and diversity, hardly any rare species

The survey was implemented through door-to-door interviews from July until September 2007. Trained interviewers conducted face-to-face interviews in 46 towns and villages throughout the catchment, following a geographical sampling strategy to maximise the variance in distance between the respondents and the locations in the experiment. Distances from respondents to the three sites ranged from 2 to 160 kilometres. Based on the pre-test results, local residents appeared to be sufficiently familiar and knowledgeable with the specific sites to be able to make well-informed choices. In total, 2,322 households were approached, of which 1,524 refused to participate, mostly because they did not have time or disliked surveys in general. This corresponds to a response rate of 34 percent. After data cleaning, the useable response was slightly reduced from 798 to 780 respondents.

Half of the respondents believe that current water quality is generally good throughout the catchment, but still find it important that water quality will be improved in the coming years. Another 40 percent finds the current water quality not good enough and thinks it should be improved further. Among the three sites, water quality at Breskens is perceived as good by most respondents. Water quality at Saeftinghe and Braakman is, on average, perceived as moderate, but the number of people who feel informed enough to evaluate water quality at these sites is substantially lower than at Breskens.

The survey results point out that 75 percent of all respondents also visit other sites besides the three included in the CE. This underlines the importance of accounting for possible substitution effects. Respondents reported substitution behavior in terms of sites and activities. If water quality at their most frequented site decreased to such a low level that their most preferred activity would no longer be possible at that site, two-fifth of the respondents would go to another, preferably nearby location. Thirty percent would keep going to their preferred site, but half of them would switch to another activity at that site. One-fifth of the respondents would no longer engage in any water recreational activities at all.

The economic valuation results are presented in Table 9.2 and 9.3. Table 9.2 presents the WTP values for the attributes expressed in Euros per household per year. Three different models were estimated. Model I is a standard choice including an alternative specific constant for each alternative location to control for location characteristics not covered by the attributes. The attribute parameters are generic and assume that changes in the amenities will be valued the same at all sites. Model II is also a standard choice model, but includes alternative-specific attribute parameters rather than generic ones. It can therefore account for possible differences in service provision between sites. Finally, model III offers the possibility to estimate substitution effects by including the attributes of other alternatives in the specification of the value function of each alternative. Full model details are found in Schaafsma et al. (2008). The results for model III show that the standard assumptions underlying choice models are violated due to site-specific differences in ecosystem service provision. The two sites Braakman and Breskens with bathing opportunities are closer mutual substitutes than Saeftinghe. Disregarding these substitution effects results in overestimations of the WTP welfare measures up to almost 100% (Table 9.3). Table 9.3 presents the changes in WTP for quality improvements at the three sites from current bad conditions to moderate and good levels.

Table 9.2: WTP (€ per household per year) for site characteristics

Attributes	**MODEL I** Standard choice model, generic values	**MODEL II** Site specific value functions	**MODEL III** Site specific value functions accounting for substitution effects
Bathing – moderate (Braakman)	12.87 (6.43)	31.32 (7.70)	30.55 (7.67)
Bathing – good	47.97 (4.87)	-	-
Breskens bathing – good	-	31.84 (5.30)	31.34 (5.29)
Braakman bathing – good	-	79.40 (8.43)	82.60 (8.79)
Breskens nature – good	-	44.46 (6.17)	43.63 (6.16)
Braakman nature – good	-	53.56 (7.01)	56.25 (7.12)
Saeftinghe nature – good	-	65.51 (7.32)	64.62 (7.26)
Breskens * Braakman nature moderate	-	-	-18.32 (5.60)
Saeftinghe * Braakman bathing good	-	-	13.75 (6.48)

Note: Standard errors are given in parentheses.

The results in Table 9.3 show that the same WFD water quality improvements can generate significantly different values at different sites due to site-specific characteristics. In this case study, respondents attach a significantly higher value to bathing water quality at Braakman compared to Breskens, and good nature conditions in Saeftinghe compared to Breskens. The nature improvement at Saeftinghe has the highest value. This implies that the direct substitution effects of these attributes differ among sites. Thus, the site-specific models II and III suggest that water quality improvements at Braakman are most beneficial to society, whereas the standard model I cannot differentiate between Braakman and Breskens at good quality levels. The site-specific attribute values also lead to significant differences in WTP welfare estimates across sites.

Table 9.3: WTP (€ per household per year) for different policy scenarios

	MODEL I	**MODEL II**	**MODEL III**
Policy scenario	Generic values	Site specific value functions	Site specific value functions accounting for substitution effects
1a) Breskens: all attributes moderate	41.64 (9.01)	42.76 (6.64)	
1b) with contextual change: Braakman nature moderate	-	-	21.94 (8.88)
2) Braakman: all attributes moderate	54.51 (9.25)	74.09 (10.46)	
3a) Saeftinghe: all attributes moderate	41.64 (9.01)	42.76 (6.64)	
3b) with contextual change: Braakman bathing good	-	-	54.02 (9.04)
4a) Breskens: all attributes good	154.64 (11.57)	131.47 (12.55)	
4b) with contextual change: Braakman nature moderate	-	-	111.17 (13.57)
5) Braakman: all attributes good	154.64 (11.57)	188.14 (16.16)	
6a) Saeftinghe: all attributes good	106.67 (9.13)	120.68 (11.11)	
6b) with contextual change: Braakman bathing good	-	-	132.90 (12.62)

Note: Standard errors in parentheses.

Model III identifies two significant substitution effects. The first one shows the substitution effect of nature at moderate level at Braakman. This substitution effect, present in the value function for Breskens, shows that improving nature to a moderate level at Braakman has a stronger negative effect on WTP for Breskens than Saeftinghe. Choice simulations based on model III show that improving nature at Braakman to a moderate level would only draw choices away from Breskens, and not reduce the share of choices for Saeftinghe. On the contrary, model I and II would predict an equal loss of choices for Saeftinghe for such a scenario. In other words, the results for model III demonstrate that stronger substitution takes place between the nature benefits at Breskens and Braakman compared to Saeftinghe with its relatively unique nature and wildlife conditions.

The second substitution effect is found for good bathing water quality at Braakman. The positive value means that an improvement of bathing water quality at Braakman to a good level draws fewer choices away from Saeftinghe than from Breskens. Choice simulations here show that such an improvement would result in only 3% fewer choices for Saeftinghe, compared to 11% fewer choices for Breskens. This is most likely due to the absence of bathing possibilities at Saeftinghe and reflects a shift in choices of those respondents most interested in bathing water quality, indicating that Braakman and Breskens are also more substitutable when it comes to changes in bathing possibilities.

Both the site-specific values and the substitution effects in model III result in different WTP welfare estimates under different ecological quality conditions (Table 9.3). An improvement of nature at Breskens is valued €18 per year lower if, at the same time, nature at Braakman changes to moderate (Table 9.2). Similarly, if bathing water quality at Braakman changes to good, an improvement of bathing water quality at Saeftinghe is valued €14 per year higher.

Accounting for substitution effects, shown separately in the last column of Table 9.3, implies that WTP for Breskens - if all attributes increase to moderate levels (scenario 1b) - is significantly lower (€22) if a simultaneous change takes place in which Braakman's nature raises to moderate level compared to a situation where the situation at Braakman does not change (€40). So, the models without substitution effects would overestimate welfare improvements for Breskens if improvements would take place simultaneously at Braakman.

The WTP estimates from model III also differ significantly from model I if all attributes at Breskens were to be improved to a good status and natural conditions at Braakman improved to a moderate level at the same time (scenario 4b: CS is €111 compared to €155 if conditions at Braakman stay the same). If all site conditions at Saeftinghe are improved to a good level, this yields a value of €107 in model I, compared to €133 in model III if bathing water quality conditions at Braakman improve at the same time. The impact of the substitution effect on welfare changes suggests that synergy effects in water management investments can be achieved if both Braakman and Saeftinghe are improved simultaneously.

In conclusion, the results show that the non-market benefits of ecological quality improvements at one water body is dependent on changes at other water bodies in the same catchment. Both the site-specific characteristics and changes in substitute sites significantly affect welfare estimates for environmental improvements of a particular site. In this case study, bathing water quality and nature conditions were valued significantly different across sites. Disregarding site-specific values results in an underestimation of WTP values up to almost 140%. This in turn questions the validity and reliability of existing welfare estimates from single-site studies in which such site-specific and substitution effects were ignored. The results presented here urge future stated choice researchers to pay more attention to the dependence of WTP values for non-market benefits at one site on environmental changes at nearby sites.

9.3 *References*

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10. Sensitivity to scope and procedural variance

10.1 *Introduction*

For most non-market goods there can be no criterion validity test (Mitchell and Carson, 1989), i.e. there is no external objectively 'correct' measure (akin to a market price) against which we can assess study results. Given this, the broad approach to validity is to examine whether or not findings conform to prior expectations, with those expectations being principally drawn from economic theory. It was this reasoning that led the so called NOAA Blue-Ribbon Panel on contingent valuation (CV) to recommend that the principal validity test for CV studies should be the 'scope test' (Arrow et al., 1993). This is based upon the expectation that (Arrow et al., 1993, p.4604):

"[u]sually, though not always, it is reasonable to suppose that more of something regarded as good is better so long as an individual is not satiated. This is in general translated into a willingness to pay somewhat more for more of a good, as judged by the individual".

The NOAA Panel recommended a number of conditions for survey validity, of which "responsiveness to the scope of the environmental insult" (Arrow et al., 1993, p. 4614) has come to be "regarded by many as an acid test" (Carson et al., 1996, p. 3) of survey-derived values. However, as Banerjee and Murphy (2005) point out, sensitivity to scope is neither necessary nor sufficient for preference consistency. There are very few non-market goods for which we have prior expectations regarding what degree of increase in WTP might be reasonable. Indeed, given that individuals may become satiated with environmental goods at low levels of provision (e.g. it would be reasonable for a respondent to think that once they had access to one nearby clean river they were not willing to pay anything for a second), then the only expectation that economic theory provides for us is that WTP should not be negative for an increase in provision of a good. This is hardly an adequate test of study validity. The claim to validity arising from studies that merely report a statistically significant increase in WTP as scope increases seem insufficient (Bateman and Brouwer, 2006).

Another well-documented validity test concerns the degree to which value functions satisfy procedural invariance. In essence, economic theory posits that individuals have well formed preferences, conforming to standard assumptions and robust against what theory would see as irrelevant information, such as the way in which a given question was framed. We believe that, in aggregate and particularly over long periods, the economic model is a good guide to predicting behaviour. However, there is ample evidence that individuals can exhibit non-standard preferences particularly over the short term, and in situations in which those individuals have low prior experience or are asked for behavioural intentions as in stated preference research. In such cases individuals often determine their assessments of a situation not solely by reference to what might be recognised as their economic preferences, but also by inferring information from the manner in which a question is framed from which they draw heuristic rules of thumb to 'construct' preferences (e.g. Lichtenstein and Slovic, 1971; Tversky and Kahneman, 1974; Kahneman et al., 1982; Slovic, 1995; Hsee, 1996; Birnbaum, 1999; Ariely et al., 2003; Kahneman, 2003; Bateman et al., 2007). The problem with such constructed preferences is that they are highly malleable; changing the frame within which a decision or survey question is posed alters that decision or response. Such phenomena have excited considerable interest within the valuation literature (e.g. Kahneman and Knetch, 1992; Schkade and Payne, 1994; Bateman et al., 2008) with commentators arguing that framing effects have to be addressed within the design of studies (e.g. DeShazo, 2002; Bateman et al., forthcoming).

The common presupposition that, for validity purposes, valuation studies should contain a scope sensitivity test provides an ideal motivation for integrating procedural invariance testing within our own specific study design. For scope sensitivity testing we require at least two levels of provision of the good in question, which for the moment we refer to as a Small and Large improvement in the resource. Scope sensitivity would normally simply examine whether WTP increases significantly between these respective levels (although for the reasons spelt out previously this is not actually a strong expectation and hence not an ideal test). There are a number of ways in which the data required for such a test can be gathered. One simple route is to ask each respondent to state their WTP for both the Small and Large improvement. This allows a simple yet powerful test of procedural invariance, if we randomise the order in which these two questions are presented across respondents, thereby avoiding the sequencing problems highlighted by Carson et al. (1998). Provided that for both questions use an 'exclusive list' format (Bateman et al., 2004) by asking respondents to state their value from the current status quo to the level specified, then the expectation is that values for each improvement should not be dependent upon the order in which they were elicited[7].

We can now formalise our procedural invariance test as follows defining

A^1 = Small quantity improvement, for which willingness-to-pay is denoted WTP (A^1)
and
A^2 = Large quantity improvement, for which willingness-to-pay is denoted WTP (A^2)

To ensure that our test is not undermined by quality differences, A^2 is defined so that it contains all of A^1 plus an additional quantity of the good (i.e. A^1 is 'nested' within A^2). Therefore, in both quantity and quality terms, $A^2 > A^1$. By varying the order of presentation randomly across respondents and denoting the 1st and 2nd question by subscripts, we therefore define the following four improvements over the status quo: , , , and their corresponding WTP measures:

, , ,

We can now define both scope sensitivity and procedural invariance tests. Because of the diminishing marginal utility associated with many environmental goods the utility of A^2

[7] Arguably, the ordering of the two scope questions may affect their incentive compatibility. Carson and Groves (2007) argue that once a respondent realises that there is more than one valuation question, they may feel they can influence the provision price by pretending to have a lower WTP than they really do. The exclusive list format is intended to avoid this problem.

is not necessarily greater than A^1 utility[8]. Therefore combining a weak scope sensitivity expectation with our procedural invariance expectation that (within an exclusive list format) WTP for a given good should not vary by order of presentation:

$$= \quad \leq \quad = \qquad (3)$$

10.2 *Case study illustration*

In implementing the case studies in AquaMoney, the value transfer design principles set out previously were adhered to. The study design was developed from economic theoretic principles which were in turn used to derive testable hypotheses for verifying the validity of findings, including various economic-theoretic and procedural invariance tests. A common approach to the valuation method, information and change in provision[9] was used in all cases and an identical WTP question frame was employed to elicit respondent valuations of both a smaller and larger improvement in water quality. Response options and data coding was common across studies and data were pooled into a single analysis. GIS was used to assess the distances for use value decay and substitute effects, and monetary variables were PPP adjusted.

A vital element of any SP valuation study is the definition of the good concerned, its status quo conditions and the change in provision which survey respondents were asked to value. This in turn requires an understanding of the physical science determining these states. However, individuals do not hold values for reducing pollution per se, but rather for the effects which such reductions may induce in terms of recreation suitability and ecological quality. Although the pathways linking pollution to ecological impact are still to considerable degree the subject of ongoing research, this does not prevent the analyst from valuing certain states of the world on the assumption that ongoing research will indicate how such states might subsequently be attained. The generic water quality ladder presented in chapter 5 ties together the ecological and use quality of water bodies, applicable to lowland slow flowing rivers as well as lakeshores. Qualitative one-to-one testing with a pilot sample confirmed that this form of information was clearly comprehended by respondents who were able to recall patterns in quality change following the interview process.

With the nature of the good clarified, the next task was to define the current level of provision and provision change. When considering natural resource goods, provision issues are inherently spatial; the provision of a given level of quality in a specified location delivers a differing amount of utility depending upon the distance from that site to the respondent's home. Therefore, an early question in all studies was to record the

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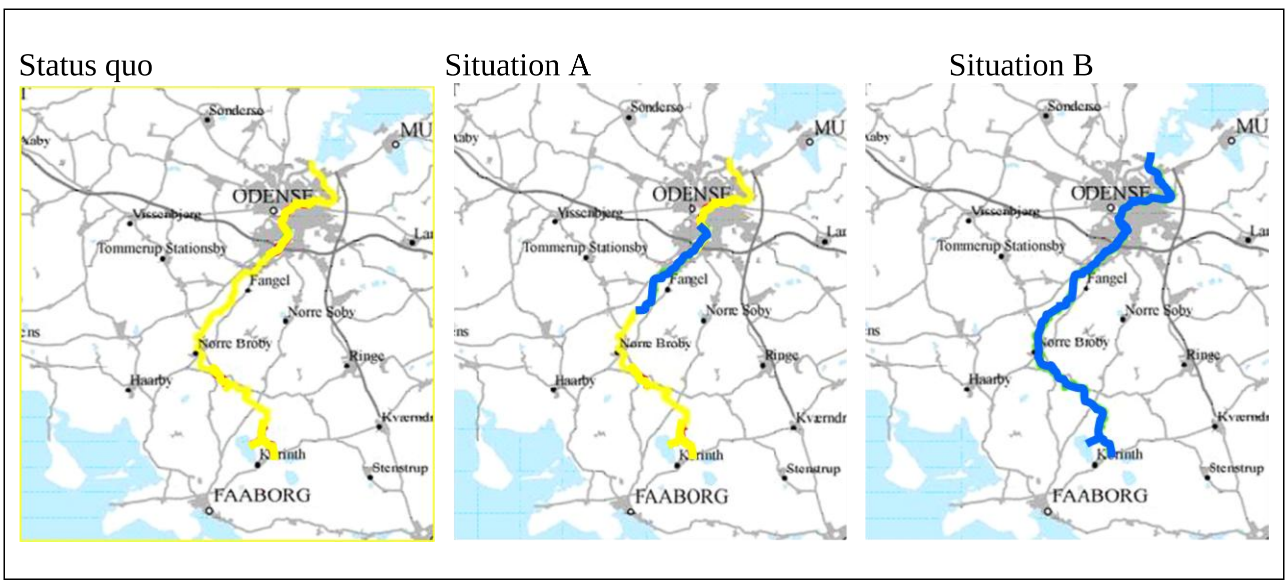
[8] Note that Hsee (1996) shows that a further anomaly can arise when a high quality good is given a higher preference rating than the same good plus an additional inferior (but still in its right utility enhancing) good. List (2002) shows that the same result is replicated within an incentive compatible, real payment framework.

[9] The change in location between study sites means that provision changes are unlikely to be precisely uniform. Differences such as substitute availability etc are controlled for in the design and analysis and strenuous efforts were made to ensure that the central provision change was common.

respondents home address. This was generally achieved both through conventional means (recording street name and full postcode) but in three countries (Denmark, Norway, UK) this was supplemented by utilising an interactive website or a custom written Computer Assisted Personal Interviewing (CAPI) program incorporating a touch screen map for recording home location, which was subsequently linked to a GIS to allow automatic calculation of distances and travel times to the improvement site and all substitutes.

The value of an improvement is clearly affected by the distribution and quality of other relevant resources which act as substitutes or complements to that which is under evaluation. Therefore, it is clearly insufficient to describe a change in water quality merely in terms of a shift from one level to another on a water quality ladder (although numerous studies have done so). It is vital to also describe the spatial location of this shift. Therefore, in all case studies within the analysis, provision change was described by combining the water quality ladder information with maps of the area around a respondent's home, using the quality level colours given in the ladder to indicate the quality of both the site where change is occurring and at all substitute sites (see chapter 6). Respondents were initially presented with a single map of the status quo situation. This was then supplemented by a further map detailing the first improvement they were asked to value (either the smaller or larger improvement; depending upon the question ordering they were randomly assigned to). A valuation response was then elicited. Following the 'exclusive list' approach we then returned to the status quo baseline and elicited WTP for the second improvement, again presented in map form. Figure 10.1 illustrates maps from the Danish case study. Examining Figure 10.1 shows that the change in provision within the Danish case study was from a 'yellow' quality river to the 'blue' quality. This change was repeated in all other case studies.

Figure 10.1: Example of the map used in the Danish case study



Given that there is a clear literature showing that changes in the elicitation method used to pose WTP questions have significant impacts upon responses (Bateman et al., 1995; Bateman and Jones, 2003), this method was standardised across all case studies using the common payment card, which was prefixed by a standard budget constraint reminder. Although presented in local currency units, when converted into Euros the payment card included the same amounts for all countries. The payment card amounts were chosen after considering the differences in purchasing power between countries and the impact upon the statistical efficiency of WTP estimates of different payment card levels.

A follow up question sought respondents' motivations for their WTP response, which also allowed assessment of any protest responses rejecting the valuation scenario (Bateman et al., 2002). The common questionnaire also contained uniform questions regarding respondent and household socio-economic and demographic characteristics, usage of water and other outdoor recreation resources, etc.

Sample sizes were designed to support not only conventional parametric validity testing but also cross sub-sample analyses of the procedural invariance tests, with a minimum subsample size of 200 respondents in all cases. In all countries, the sampling strategy ensured a wide geographical distribution of respondents over the study area in order to capture variation in distance data. Some differences arose in terms of survey implementation. In Belgium, Denmark and Norway, the surveys were conducted online, through a marketing company in the two latter countries. In Lithuania and the UK, surveys were conducted by face-to-face interviews. Response rates ranged from 12 % in Belgium up to 55% in Lithuania. In the latter countries surveys were conducted in March 2008, whereas the Danish and Norwegian studies were held in July and September-October 2008 respectively and the UK surveys were conducted throughout the summer months spanning the above periods. A total of 3,589 questionnaires were completed across the five study countries. Table 10.1 presents descriptive statistics of each sample together with WTP sums disaggregated by ordering, size of provision change and a combination of both factors.

The wider representativeness of the final sample is satisfactory with most sample descriptors differing by less than 5% from national statistics. Compared to the all country average, PPP-adjusted household income is lowest in Lithuania. The wide areas used for all sampling means that in all countries, respondents typically have a substitute water body which is closer to them than the study site. This indicates that, if substitute relationships do apply, then the data should reflect them. Protest rates were typically low and within the bounds of acceptability suggested by the literature (Mitchell and Carson, 1989; Champ et al., 2003). Following guidelines for international value transfers (Navrud and Ready, 2007b), WTP responses (and income data) were corrected for PPP differences World Bank data (2008) and converted to Euros. In line with expectations, mean WTP values broadly reflect income constraints, being lowest in Lithuania and highest in Norway and Belgium. Mean values appear to reflect scope sensitivity; however recalling our caution regarding such tests we now turn to consider this and related results in greater detail.

Table 10.1: Descriptive statistics and WTP by country

	Belgium	Denmark	Lithuania	Norway	UK	Total
Sample characteristics						
Number of respondents	768	754	500	1133	434	3589
Protest bids (% of country sample)	5%	2%	8%	12%	2%	7%
Respondents' characteristics						
Mean Age	45	50	48	45	50	47
Gender (% women)	36%	44%	49%	48%	46%	45%
Household income per year before tax	40877	34854	9531	24884	26686	28310
(in € PPP) (s.d. in brackets)	(19002)	(17708)	(7823)	(11452)	(16709)	(17730)
Urban (% urban)	45%	79%	63%	41%	78%	58%
Mean distance to the improved site (in km)	21	30	20	22	10	22
Mean distance to unimproved substitute site (km)	3	24	1	7	5	9
WTP values (in € PPP, standard deviation in brackets)						
Average WTP- Question1	47	29	7	46	24	34
	(68)	(41)	(36)	(83)	(32)	(63)
Average WTP- Question2	42	31	6	43	21	32
	(66)	(50)	(22)	(85)	(32)	(63)
Average WTP- Small	47	25	6	42	19	31
	(66)	(38)	(23)	(82)	(29)	(61)
Average WTP- Large	48	36	8	47	26	37
	(70)	(52)	(38)	(86)	(35)	(66)
Average WTP- Question 1-Small WTP ( )	50	29	6	45	22	34
	(67)	(42)	(15)	(88)	(32)	(64)
Average WTP- Question 1-Large WTP ( )	49	31	10	45	25	36
	(70)	(41)	(52)	(78)	(32)	(64)
Average WTP- Question 2-Small WTP ( )	43	21	6	38	16	28
	(66)	(33)	(29)	(76)	(25)	(57)
Average WTP- Question 2-Large WTP ( )	47	40	7	48	26	37
	(69)	(61)	(15)	(93)	(38)	(69)

The results suggest that each ordering treatment appears to yield scope sensitive results (with the larger improvement being accorded higher WTP). However, there appears to be considerable differences across the two treatments, with greater scope sensitivity in the top-down treatment and a substantially higher WTP for the smaller improvement when presented as the first good encountered by respondents. The results show great similarity to those of Bateman et al. (2004). The test results for scope and ordering effects suggest the presence of framing effects.

10.3 *The effect of geographical scale on scope effects*

Valuation studies are required to define sampling areas large enough to test for the presence of distance decay effects in the valuation of water quality improvements. The documentation of distance decay effects provides further internal validity to the valuation results, in comparison with studies that do not sample populations far from the policy relevant water body. The definition of "far" in distance terms is generally interpreted to mean far enough for other substitute water bodies to be as close, or closer, than the water body that is the focus of policy assessment. This consideration often determines the geographical scale of case studies.

This section argues that the effects of distance decay, substitution and scope may be hard to disentangle. There is a trade-off between identifying scope effects and distance decay in the same study while also accounting for substitutes. Distance decay may be easier to observe with larger geographical scale/sampling area just because distance to the water body is greater from the furthest household sampled. Larger scale studies will also include more substitute sites reinforcing the distance decay effect. Substitute effects strengthen, or may be confounded with distance decay effects. The section argues that substitution and distance decay effects make it harder to observe scope effects in the "one versus two water bodies" scope tests. At the same time greater demands are placed on conducting benefits transfer, because scale, substitutes and scope are expected to be significant elements of the context of the valuation case studies.

The numbers in Figure 10.1 symbolize different water bodies inside and outside a catchment draining to the sea (#1). A valuation practitioner must make a decision on the geographical scale of the sample (sample 1, 2 or 3). Figure 10.1 summarises the context dependence of the valuation results and sets the stage for a discussion about the trade-off between observing distance decay and scope effects. Increasing the geographical scale of the sampling increases the distance to site #2 of households at the edge of the sampling area and increases the number of substitute sites, both increasing the expectation of distance decay effects in the willingness to pay (WTP) for improvements in site #2. Increasing the geographical scale of the study increases the range of possible scope effects, as different water bodies are expected to have varying water quality levels and other site characteristics. The more variation in site characteristics, the more sites are possible substitutes for site #2.

Figure 10.1: Context dependence of a valuation study

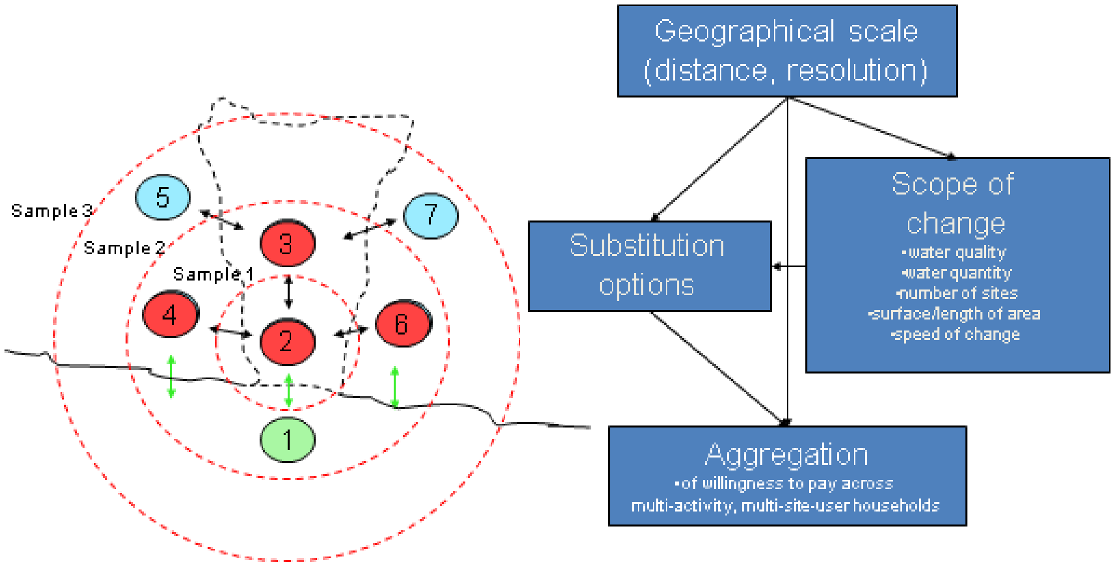


Figure 10.2 shows that different lakes have different status quo conditions. Water quality and recommended uses are illustrated using the 'WFD water quality ladder' with four levels. The choice of geographical area determines the variation in status quo water quality to be considered in a valuation scenario. It also determines the variation in scope that is relevant to test relative to the status quo water quality level. Furthermore, it determines the number of potential substitute water bodies that will be shown to respondents within the sampling area. In this example from South Eastern Norway, a number of lakes are of particular interest for valuation, including (a) "Vestre Vansjø ", (b) "Mingevatnet/Glomma", and (c) "Femsjøen". Imagine 5 different households - households #1 - #5 - living at different distances from these lakes. We are interested in finding whether their WTP drops with distance and also the presence of substitutes such as the sea and other lakes (in blue quality).

Figure 10.3 shows a conceptual illustration of a WTP function with distance decay concentrically away from "lake b". Suppose households #1 and #5 are about 50 km from "lake b" located at the origin. If distance decay in WTP is present we would expect WTP to be highest for household #3 and lowest for households #1 and #5. In this example households #2 and #4 have willingness to pay at or near zero, for a water quality improvement ΔQ_b of '1 class' on the water quality ladder. If there were no substitutes and the lake was equally accessible from all directions, we would expect WTP to drop in a radius around the water body as illustrated in the upper right panel.

Figure 10.2: Lake water qualities and example household sampling locations

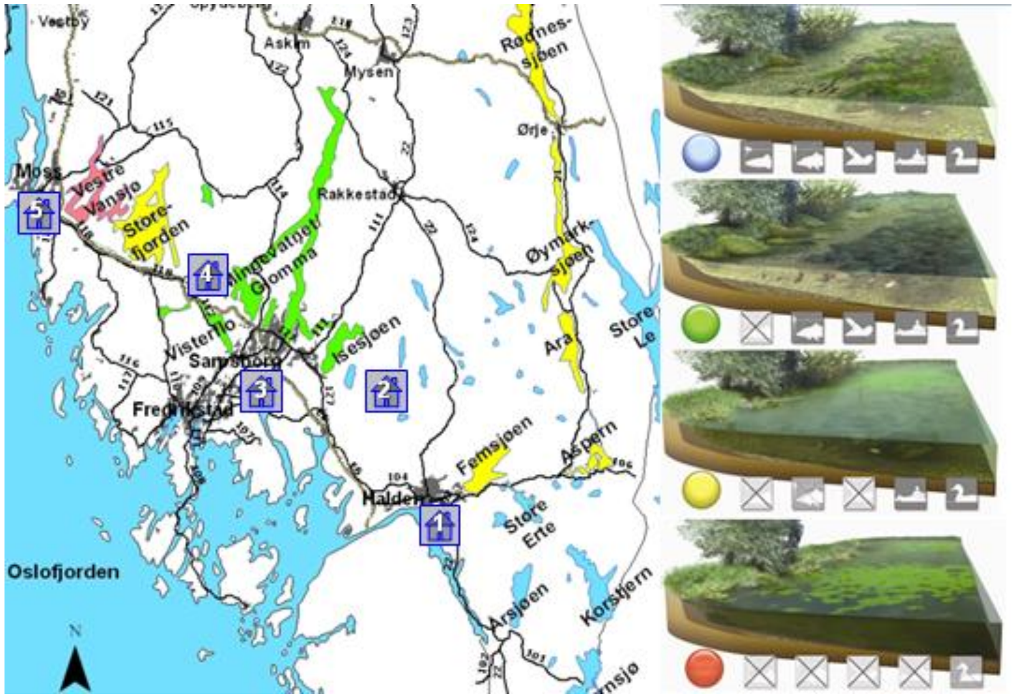


Figure 10.3: Distance decay in willingness to pay

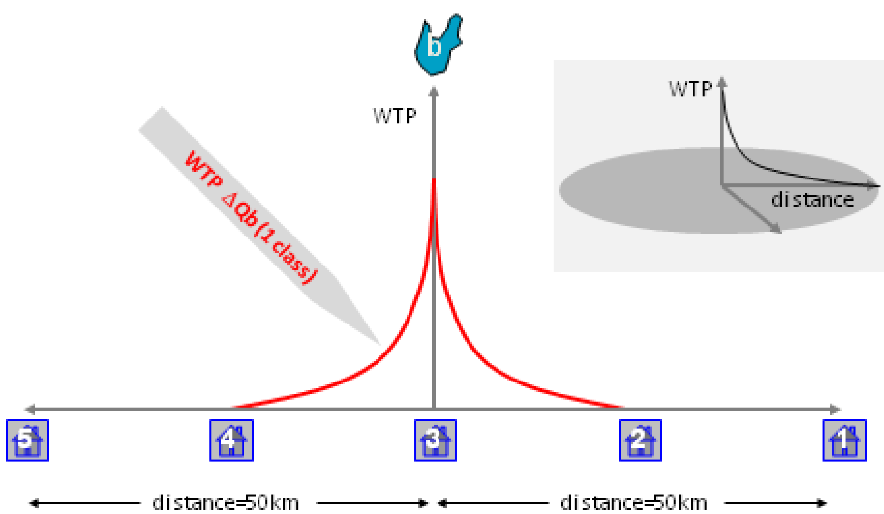
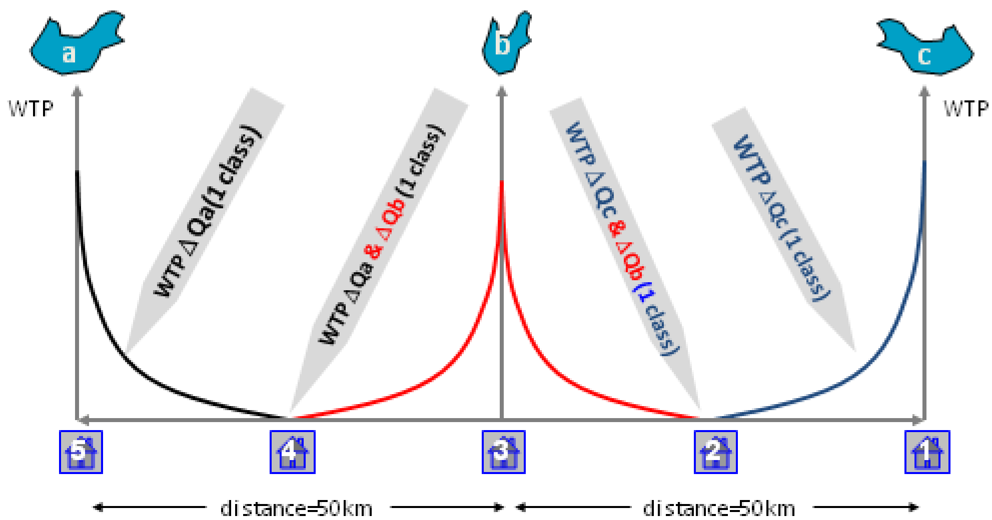


Figure 10.3 shows that as long as there are no substitute sites, WTP is expected to drop with increasing distance (or increasing access time). We would expect the "radius of WTP" to increase if the water quality improvement was 2 classes instead of 1 class.

Suppose that household # 1 is asked to state their WTP for an improvement in “lake b” first and then “lake b” and “lake c” as an example of a scope test (one versus two lakes). Suppose furthermore that we ask the same question also to households #1-#3. Figure 10.3 shows that WTP for a 1 class quality improvement is not sufficient to uncover a scope effect because nowhere do any of the households live close enough to lakes b and c to have WTP for both lakes simultaneously (the valuation functions/lines do not cross in Figure 10.3). Households #1 and #5 can be expected to have positive WTP for their local lakes. Households #2 and #4 live at equal distance from all lakes considered and still have near zero WTP for a 1 class quality improvement. If these households are representative of the population, there would be no significant scope effect observed in the study for scenarios improving “lake b” and “lake a” or “lake b” and “lake c”.

Figure 10.4: Distance decay in WTP and substitute sites



Hence, a lack of scope effect is not necessarily because households are not sensitive to the scope of water quality improvements. The combination of the scale of the study, the resulting substitutes considered and the water quality improvement proposed to households together may constitute a **study context** where there are no observable scope effects.

With a 2 class water quality improvement, WTP is positive at larger distances from the lakes assuming the same distance decay function. Figure 7.6 illustrates WTP functions for the three lakes improved by 2 water quality classes. For households #2 and #4 WTP functions for adjacent lakes now overlap, implying that households in these areas consider the lakes to be substitutes and would be willing to pay for improvements in either. They have additional WTP for improvement in 2 lakes versus in 1 lake (illustrated by the thin lines in the upper panel). Scope effects would only be observed for households that had the lakes as substitutes, while not being observed in other parts of the study area: households #1, #3 and #5 do not have any substitutes for their local lakes ("lake c", "lake b" and "lake a" respectively).

Figure 10.5: Sensitivity to scope when the water quality improvement is large enough

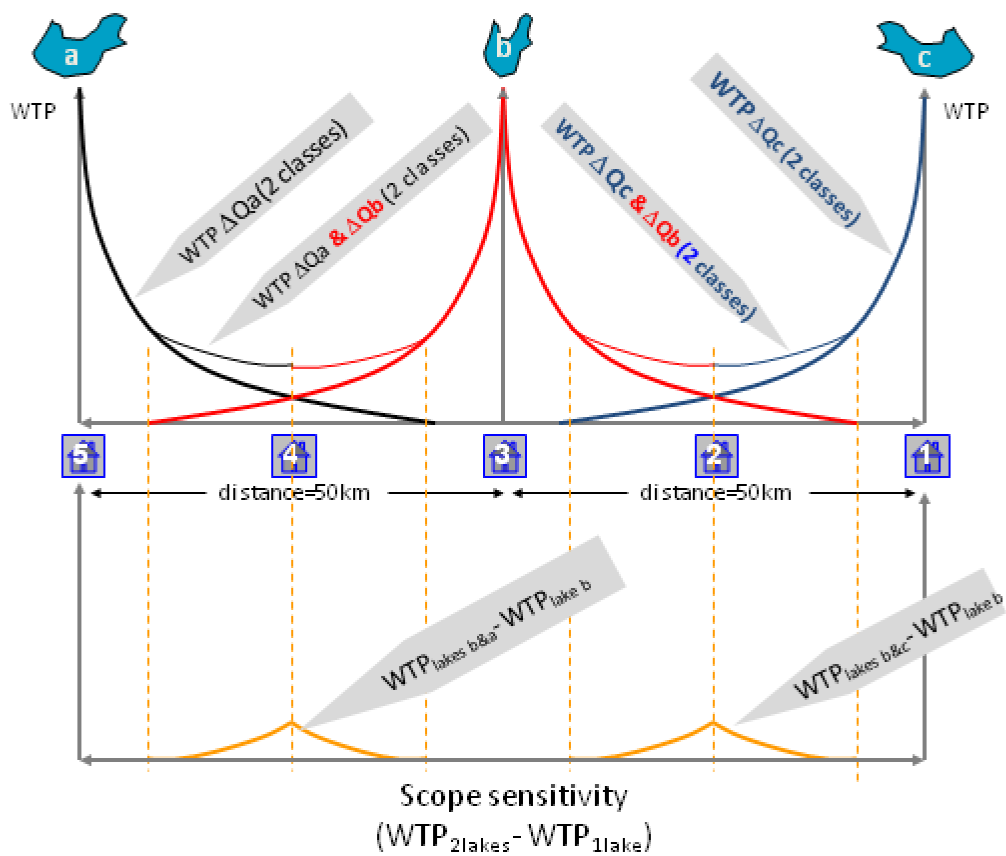
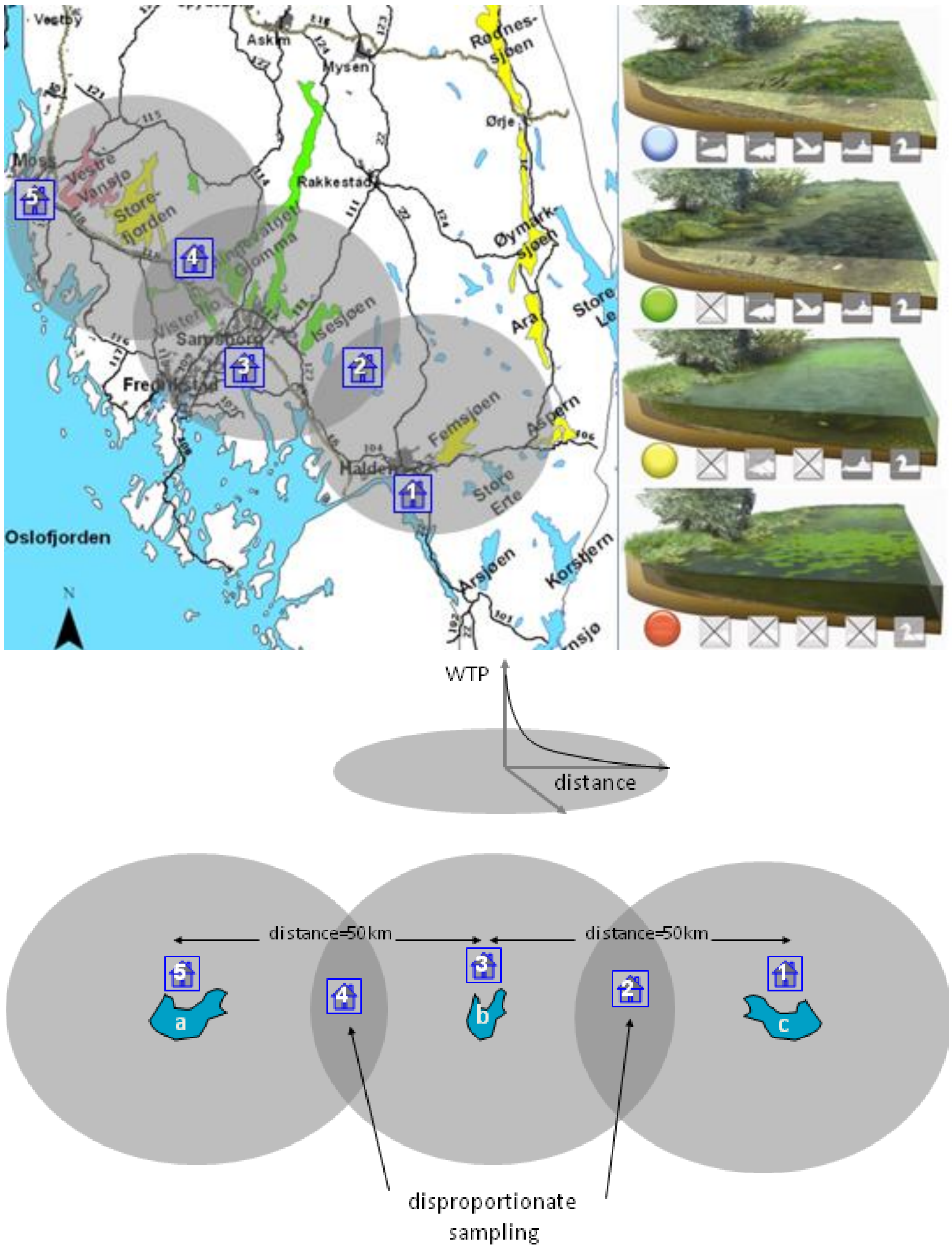


Figure 10.5 illustrates that households may be sensitive to scope where the lakes included in the scope test are considered to be substitutes. In other words, in designing a scope test, presentation of water quality improvement scenarios in water bodies that are as close to each other as possible is advisable in order to increase the probability of finding scope effects (if indeed they are there).

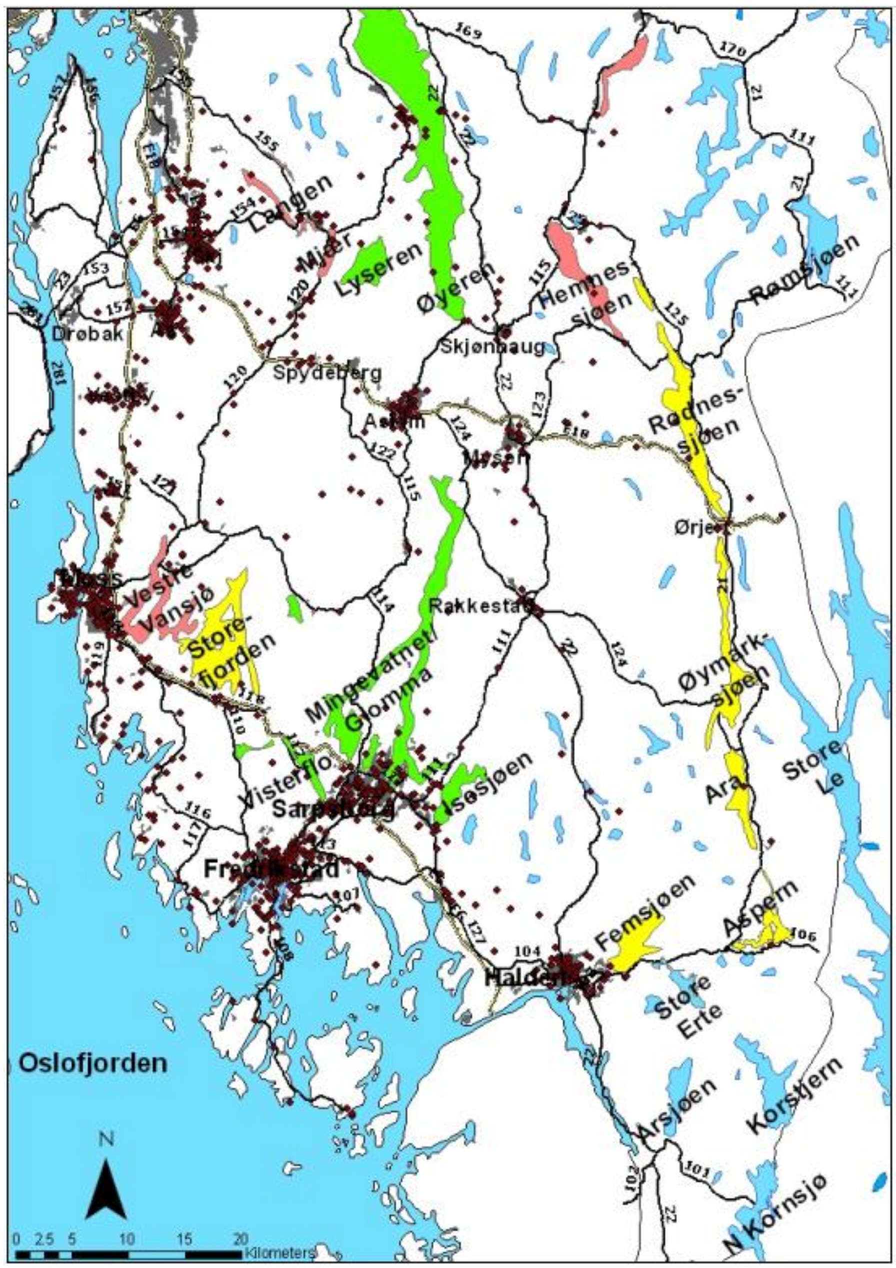
Figure 10.6 shows that the actual geographical configuration of the lakes in the Norwegian case study, their status quo quality, the size of the improvement proposed and the sampling strategy of the study. These issues all combine to make it more or less likely that a scope effect will be identified. This issue suggests that the sample frame itself will determine the likelihood of observing scope effects. Looking at Figure 10.6, if WTP for lakes (a) "Vestre Vansjø ", (b) "Mingevatnet/Glomma", and (c) "Femsjøen" is positive in the radius suggested (upper panel) then over-proportional sampling of households in locations #2 and #4 might be necessary to detect a significant scope effect in the sample (see lower panel). With (b) “Mingevatnet/Glomma” already in class “green” the scope test might also be limited by the status quo water quality situation (it is npt possible to propose a 2 class increase in this lake).

Figure 10.6: Adjusting household sampling strategies to detect scope effects



In practice the ‘lumpy’ geographical distribution of households in urban areas may not permit sampling households of the type #2 and #4 in order to get observations across different distances from the lakes in question. In Østfold County in Norway households cluster in towns near the water bodies but not in rural areas in-between (Figure 10.7). In summary, observation of scope effects - in terms of one lake versus two lakes - would seem less likely (i) the further away lakes are from each other, (ii) the stronger distance decay effects are, (iii) the more the lakes differ from each other (the less they are substitutes), and (iv) the more a population is clustered around the lakes.

Figure 10.7: Distribution of the sampled households (dots) in the Norwegian study



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11. Payment certainty calibration

11.1 *Introduction*

Stated preference methods such as contingent valuation (CV) face a number of biases, of which payment certainty related to the method's hypothetical bias received a lot of attention after publication of the NOAA Blue Ribbon Panel recommendations (Arrow et al., 1993). Payment certainty refers to the empirical finding that respondents are unsure about their value statement for hypothetical changes in the provision level of a private or public good (e.g. Li and Mattson, 1995; Ready et al., 1995; Champ et al., 1997). Values are systematically overstated when elicited under hypothetical conditions compared to real purchase decisions (e.g. Johannesson et al., 1998; Veisten and Navrud, 2006). The extent to which hypothetical responses are overstated is influenced by the value elicitation method, including whether respondents were asked for their willingness to pay (WTP) or willingness to accept (WTA), and whether the good concerns a private or public good (List and Gallet, 2001).

To account for this overstatement, a number of calibration approaches have been advocated in the literature. The most important ones are ex post decision ratings and the use of polychotomous elicitation formats. In the former case, the respondent is asked in a follow-up question to indicate the certainty of his WTP reply on a scale from 1 to 10 or 0 to 100 percent. In the latter case, respondents are able to express their certainty through the dichotomous choice (DC) WTP question self, for example by answering 'definitely yes, probably yes, don't know/not sure, probably, definitely no' to the presented bid amount. Typically, asymmetric approaches are applied based on self-reported payment certainty, where uncertain yes responses to a DC WTP question are recoded as certain no responses. This automatically reduces estimated mean WTP. In the limited number of studies exploring at which certainty cut-off value hypothetical WTP best simulates actual market behavior (i.e. where uncertain yes responses are recoded to certain no responses), values vary between 6 and 10 using a scale from 1 to 10 (e.g. Champ and Bishop, 2001; Poe et al., 2002). Polychotomous elicitation formats have been used to identify similar threshold values with the help of multinomial choice models where respondents switch between certain and uncertain WTP replies (e.g. Welsh and Poe, 1998; Alberini et al., 2003). In other applications, polychotomous elicitation formats were used as a WTP follow-up question and only the 'definitely' or 'absolutely' yes responses appeared to match actual purchase behavior for private goods (e.g. Blumenschein et al., 2008).

Although there exists no consensus in the literature about the most appropriate payment certainty elicitation format, the available empirical evidence listed above suggests that both approaches can help to reduce hypothetical bias in stated preference research. However, the evidence is limited and a fair share of the studies focus on private market goods. More research is needed in the area of public good valuation, also regarding underlying sources of payment uncertainty, which are rarely investigated, but expected to provide important insight into the design of more reliable stated preference survey formats and WTP values.

11.2 *Case study illustration*

The data used in this paper are taken from an international water quality CV study conducted in the Scheldt river basin. The Scheldt is 350 km long and flows through three countries. The river originates in France, runs through Belgium and ends in the Netherlands where it flows into the North Sea. The international river basin covers an area of over 36 thousand square kilometres and has almost 13 million inhabitants. This corresponds with an average population density of 350 inhabitants per square km, which is almost three times the European average. The Scheldt is one of the pilot river basins in the implementation of the WFD. Current Scheldt water quality was classified as ‘moderate to poor’ in the first WFD regulatory assessment report (Scaldit, 2006). In order to elicit public preferences for water quality improvement and obtain measures of WTP for the WFD water quality objectives, a CV questionnaire was sent out to a random selection of 17,000 households across the Scheldt basin. In particular, questionnaires were sent to 5,000 households in Artois-Picardie in northern France, 9,000 in Flandres, Belgium, and 3,000 households in the Dutch part of the basin.

The common survey design was developed together with water experts from the responsible water management authorities and was pre-tested in French, Flemish and Dutch. The questions are identical in the three versions of the questionnaire except for the description of the current situation, which was modified to the specific prevailing circumstances and conditions in the three different parts of the river basin. A map and common water quality ladder were used to depict the current situation and show respondents the location and quality levels of the river relative to their place of residence. Following a series of introductory questions, respondents are presented with a one-page information statement in the second part of the questionnaire, in which the actual water quality situation is described with the help of a map and a brief explanation of the WFD. After the information statement, respondents are asked how familiar they are with the presented information and how important it is to them that the WFD objective of ‘good ecological status’ is reached. This is then followed by a DC WTP question using 10 different bid amounts ranging between 5 and 250 Euros and a post-decisional payment certainty question on a scale from 0-100% with 10 percent intervals. It is explained that 0 means not certain at all and 100 means completely certain. An open-ended follow-up question is used to enable those respondents who are not completely certain to specify why not (see Box G). The questionnaire ends with a number of questions about the questionnaire self and in particular the WTP question to examine respondent understanding of the WTP question, the perceived credibility of the valuation scenario and the difficulty experienced in answering the WTP question.

The overall response rate was 18 percent, which is not unusual for these kinds of surveys in the three countries where the survey was conducted. As expected (based on available country specific demographic and socio-economic statistics), respondents differ significantly between countries except for age. Although the response rate was not very high, the three samples approximately represent the average inhabitant of the three countries.

Box G. WTP and preference certainty questions

Q. Are you as a household willing to pay every year € X in extra income tax over the next 10 years in order to reach a good ecological water quality status in 2015 in your part of the river basin?

Note: this money will only be used to finance the additional measures needed to reach a good ecological water quality status in 2015 in your part of the river basin as indicated in the map.

1 Yes
2 No

Q. Can you explain why you are willing to pay this specific amount of money, or if you are not willing to pay this specific amount of money why not?

………

………

Q. How certain are you that you will actually pay this specific amount of money, or if you are not willing to pay this specific amount of money how certain are you that you will not pay?

(please circle the appropriate percentage)

0% certain	10%	20%	30%	40%	50%	60%	70%	80%	90%	100% certain

Q. If you are not completely (100%) certain, can you explain why not?

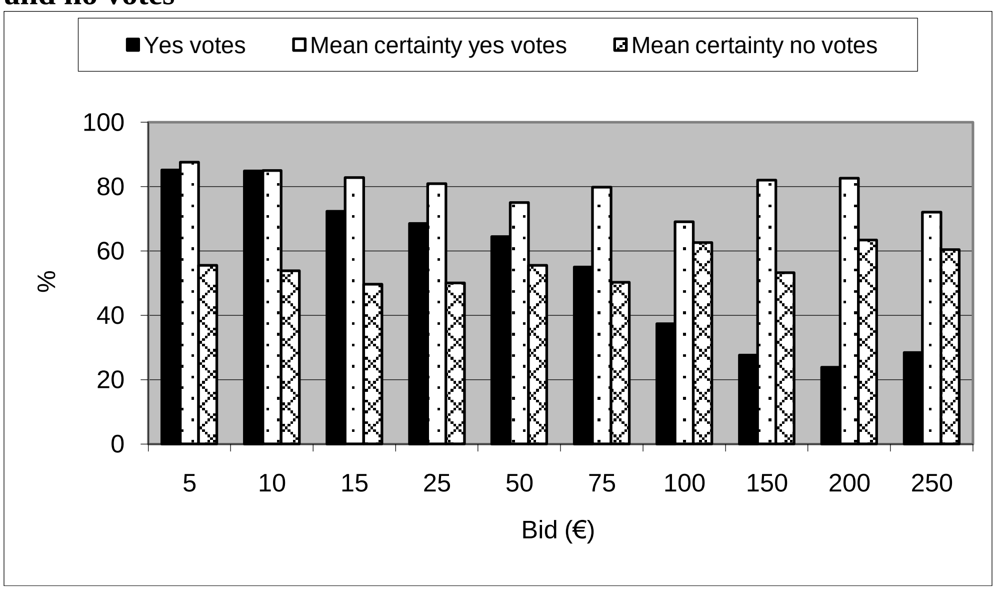
………

………

Respondents were asked for their WTP through extra annual taxes until and including 2015 to reach a good ecological status for all water bodies in their part of the Scheldt basin. Just over half of all respondents (52%) said they were willing to pay extra for this purpose, and this share is more or less the same across the three river basin countries. Most of the respondents who were not willing to pay extra for the proposed water quality improvements motivated this by saying they lacked sufficient income, 'the polluter should pay', 'I already pay (enough) taxes for water quality', and 'there are more important other things I prefer to spend my money on'. About five percent believe that current water quality is good enough or consider water quality improvements not important enough to pay for. The share of respondents protesting against the WTP question in this study based on considerations such as the polluter should pay, lack of trust in the feasibility of the proposed program of measures or the responsible authorities is 9.5 percent of the total sample[10]. Protest rates vary slightly across the three countries, from 7 percent in Belgium to 11 percent in the Netherlands and 13 percent in France.

The probability distribution function of the positive responses to the DC WTP question is presented in Figure 11.1. As expected, the higher the bid amount, the lower the probability that the respondent is willing to pay. The certainty experienced when answering the DC WTP question is also presented in Figure 11.1. A distinction is made between yes and no votes. A small, but significant negative correlation exists between bid price and self-reported certainty for the yes votes and a significant positive correlation for the no votes. Corresponding with previous findings, yes voters are more confident overall about their answer than no voters.

Figure 11.1: Probability distribution function yes votes and payment certainty yes and no votes



Respondents are significantly more certain that they are willing to pay the lowest bid (€5) than the highest bid (€250). No significant difference can be found between the payment certainty levels for respondents not wanting to pay the lowest and highest bid. Payment certainty is significantly lower in France compared to Belgium and the Netherlands. No significant differences can be found between Belgium and the Netherlands.

Respondents who were not 100% certain about their stated WTP were asked in an open-ended follow-up question why not. The reasons given were carefully analyzed and

[10] Protest bidders typically object against the imposed market structure in a CV study (e.g. Meyerhoff and Liebe, 2006).

categorized. The main sources of uncertainty underlying stated WTP are presented in Table 11.1. Payment uncertainty can be related to imperfect knowledge and information about (i) the good to be valued (which is a function of information provision and experience), including its provision level now and in the future (referred to here as ‘supply uncertainty’), (ii) the utility derived from different ‘consumption’ levels (referred to here as ‘demand uncertainty’), which is a function of individual respondent characteristics such as household income levels and corresponding purchasing power (now and in the future as CV research often asks respondents to pay over a specified period of time in the future), (iii) particular simulated market conditions (referred to here as ‘survey instrument uncertainty’), such as respondent trust in property right security when paying for a public good, public good suppliers (e.g. government or other) and related payment mechanisms (e.g. tax or user fee), and (iv) price levels.

Almost half of all self-reported payment uncertainty (45%) is related to the survey instrument, followed by respondent uncertainty about his or her future income situation (mentioned by a fifth of the uncertain sample), and current and future price levels (17%). A high cost price can be seen as a choke price and hence as another demand related source of uncertainty. Within the category ‘survey instrument uncertainty’, three sub-groups are distinguished. Policy scenario uncertainty constitutes the largest source of uncertainty. This includes lack of trust that the money paid will actually be spent on the improvement of water quality, lack of trust in the government and water managing institutions as the main providers of the good for which respondents were asked to pay, the perceived inefficiency of public administration and lack of control over how public money is spent. This category is closely related to and overlaps with the ‘supply’ related sources of uncertainty as it also refers to doubts about the provision of the environmental good in question. Another source of survey instrument uncertainty relates to the appropriateness of taxes as the payment mode or whether individual households are the right target group for this particular problem.

Although it was emphasized in the questionnaire that the contingent market simulation is based on the polluter pays principle, fifteen percent of the uncertain respondents question whether this is actually the case, including how much surrounding countries will do to solve the problem and to what extent all households will pay for this. Together with the lack of trust in the government, these latter reasons are usually classified as protest response in CV. Protest beliefs may hence be an important source of payment uncertainty. On the other hand, respondents who are uncertain may also be more inclined to resort to protest beliefs when trying to explain why they are uncertain due to instable preferences.

Interesting differences were found when looking at the main sources of uncertainty across the three countries. In France, doubts about household income are the main source of uncertainty, and also in the Dutch sample most respondents are worried about their future employment status. These concerns are much less a source of uncertainty in Belgium where most respondents lack trust in the authorities and doubt that the money will actually be spent on the improvement of water quality. This reason comes second in the Netherlands, and plays almost no role in France. Also the feasibility of reaching a good

Table 11.1: Main sources of payment uncertainty

	Share (%)
Uncertainty related to the good self and its future supply	
Feasibility reaching good ecological water status	7.2
Insufficient information about the good and its supply	2.3
Doubt effectiveness of measures to be taken	1.3
Subtotal	10.8
Future demand uncertainty	
Future household income	17.3
Rising other household expenditures in future	2.4
Future situation in general	1.8
Subtotal	21.5
Price uncertainty	
High cost price	13.6
Future development cost price/tax	2.0
Calculation cost price	1.8
Subtotal	17.4
Survey instrument uncertainty	
General survey instrument uncertainty	
Doubt own contribution to the problem	3.3
Appropriateness of the tax instrument	2.8
Doubt influence on political decision	2.1
Existence other possible solutions	1.6
Doubt whether households are the right target group	0.9
Doubt whether paying extra is the solution to this problem	0.4
Subtotal	11.1
Policy scenario uncertainty	
Disbelief that the money will be spent on water quality improvements	13.0
Mistrust of the government	4.9
Control over how money will be spent and monitoring results	1.0
Subtotal	18.9
Uncertainty market conditions	
Whether polluters will pay	9.5
Whether everybody else will pay too	3.3
What surrounding countries will do	2.1
Subtotal	14.9
Other reasons	5.4
Total	100.0

ecologic water status plays an important role in Belgium and the Netherlands, but much less in France. Uncertainty about whether the polluter will pay plays a stronger role in France than in Belgium and the Netherlands. Related to this, a remarkable finding is that uncertainty about what surrounding countries will do is mentioned by almost 10 percent of all respondents in the Dutch (downstream) sample, less than one percent in Belgium and never in the French (upstream) sample. The high bid price is an important source of uncertainty in all three samples.

Self-reported (un)certainty was accounted for in the WTP estimation procedure and the results are presented in Table 2, including the uncorrected WTP ignoring preference uncertainty. Mean WTP values are furthermore presented for those respondents who are and those who are not 100% certain and based on different certainty calibration cut-off points. The statistical efficiency of the WTP estimates is measured with the help of the mean squared error (MSE).

A first important observation from Table 11.2 is that respondents who are 100% certain about their stated WTP are willing to pay, on average, significantly more than respondents who are not 100% certain. Not accounting for payment uncertainty hence results in an overestimation of the WTP welfare measure. The statistical efficiency of the welfare estimates is also highest for respondents who are 100% certain when examining the MSE values. This implies that WTP values are most accurate when respondents are completely certain.

A second observation is that, as the MSE show, the statistical inefficiency of the calibrated welfare estimates (i.e. recoding of uncertain WTP based on different certainty cut-off points) increases as the restrictions imposed on payment certainty are more stringent (and the welfare estimates decrease accordingly as expected). Hence, as demand for payment certainty of hypothetical WTP responses increases for welfare estimation purposes, the practice of recoding uncertain responses results in gradually less precise welfare estimates.

Hence, the study confirms what other CV studies found before, that is, that respondents face considerable uncertainty when participating in a simulated hypothetical market, and this uncertainty significantly affects their stated WTP. Not accounting for payment uncertainty results in an overestimation of the welfare measure. Comparing respondents who are and respondents who are not completely certain about their stated WTP, the latter are significantly less willing to pay than the former and the estimated welfare measure is less precise. A strong correlation was found between payment certainty and respondent familiarity with the public environmental good in question and belief in the presented valuation scenario points out the importance of the role of information in stated preference research.

Table 11.2: Mean WTP with certainty corrections

Summary statistics	Uncorrected WTP		100% certain	<100% certain		Certainty ≥50%	Certainty ≥60%	Certainty ≥70%	Certainty ≥80%
Mean WTP (€/year)	107.4		127.4	86.3		92.8	63.1	50.9	22.3
95% confidence interval	97.6-117.2		113.3-141.6	71.8-100.8		83.1-102.6	52.9-73.3	40.5-61.2	10.1-34.6
Mean squared error (MSE)	0.201		0.168	0.218		0.202	0.211	0.218	0.238
N	1662		701	961		1662	1662	1662	1662

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12. Transfer errors

12.1 *Introduction*

Value transfer exercises typically involve estimating the value of a given change in provision of an environmental good at some target 'policy site' from previous analysis undertaken at one or more 'study sites'. Analysts have for many years sought methods which will reduce decision costs and the extrapolation of assessments from one case to another is clearly attractive. Given the significant costs of valuing preferences for non-market goods it is not surprising that this area has now generated a considerable literature concerning the transfer of benefit and cost estimates, most particularly in the area of environmental valuation (Brouwer, 2000).

To date the most common approach for such study site analyses is some variant of stated preference (SP) technique such as contingent valuation (CV) or choice experiments (CE) (Bateman et al., 2002). Although CE are considered superior to CV for the purpose of value transfer (Morrison et al., 2002), the most fundamental problem for value transfers is not regarding which technique to apply, but rather in assessing whether a given transfer is correct or not when the 'true' value of the policy site is a-priori unknown. Because of this problem, the literature has placed great emphasis upon the development and testing of value transfer methods (e.g. Bergland et al., 1995; Downing and Ozuna, 1996; Brouwer and Spaninks, 1999; Ready et al., 2004; Moeltner et al., 2007). Here researchers typically undertake studies at two or more sites and test the effectiveness of a transfer method by dropping data from say one site (i.e. designating it as the policy site) and using information from remaining (study) sites to estimate values for the policy site. Errors are then assessed by comparing transferred value estimates with those obtained from the study conducted at the policy site itself.

The methods used for transferral can be broadly categorised into two types (Navrud and Ready, 2007). The simplest approach is to attempt to find study sites which appear similar to the policy site and transfer mean values from the former to the latter (e.g. Muthke and Holm-Mueller, 2004). Such 'univariate' transfers are frequently used in practical decision making, but are crucially dependent upon the pertinence of differences between transfer sites. Clearly all sites are to some degree dissimilar (e.g. unique ecosystem habitats or the spatial pattern of substitutes around a site is unique); it is the degree to which this dissimilarity affects values which will determine the appropriateness of unconditional mean value transfers. It is because of such concerns that value function or 'multivariate' transfer approaches have been developed. Here statistical techniques are used to estimate value functions from study site data. These are then used to predict new values for policy sites. This is achieved by assuming that the underlying utility relationship embodied in the parameters of the estimated model applies not only to individuals at the study sites but also to those at policy sites. Usually, these parameters are kept constant, while the values of the explanatory variables to which they apply are allowed to vary in line with the conditions at the policy site. In those cases where results are used from studies carried out years ago, an important question obviously is to what extent preferences and parameters have changed (Brouwer, 2006).

Pearce et al. (1994) argue that because value function transfers allow the analyst greater control over differences across sites, they should in principle yield lower transfer errors than simple mean value transfers. However, empirical evidence regarding this assertion is mixed with sometimes the opposite result being observed (e.g. Bergland et al., 1995; Barton, 2002; Ready et al., 2004). This is partly due to the lack of a systematic assessment of a set of (theoretically driven) baseline conditions needed to be in place for valid and robust value transfer. In this paper, we argue that the reason for the perverse results reported in the literature is in part because of misspecification of empirical value functions. Note that we do not claim that the value functions used previously are statistically mis-specified; we argue that the statistical selection process underlying the generation of value functions is part of the problem, for what is appropriate for statistical specification (maximization of explained variance) may be inappropriate for transfer purposes (minimization of transfer error). Efforts to describe determinants of willingness to pay (WTP) in a specific case study through maximization of the value function's explanatory power may transfer poorly if those functions include highly site-specific factors, which are of little or no relevance at all at other sites, resulting in large transfer errors (Brouwer and Bateman, 2005). The driving issue is one of changing contexts.

In a previous cross-country comparison, Brouwer and Bateman (2005) show that a simple unadjusted unit value transfer works best for similar case study sites, while errors generated by simple mean value transfer are considerably larger than those arising from function transfer across dissimilar case study sites. As expected, if conditions are not the same across study and policy sites, some degree of adjustment helps reducing the error. In an international water quality transfer study based on unadjusted average values between Norway and Germany using the RFF water quality ladder (Muthke and Holm-Mueller, 2004), significant differences are found between WTP values exceeding error margins of 60% due to differences in population and site characteristics and baseline conditions that could not be controlled for. Similar large errors (87-130%) are found in Barton and Mourato (2003) when transferring WTP for coastal bathing water quality in Portugal and Costa Rica and reductions in associated health risks. Lower errors based on international transfer of unadjusted unit values of around 40% are found, on average, in Ready et al. (2004) in the context of ill health episodes across five European countries, and 25% in Rozan (2004) for improved air quality and related health impacts in France and Germany. Average transfer errors are around 30% for unadjusted value transfer and 10% for function transfer for dissimilar case study sites in Europe and New Zealand facing global health risks in Brouwer and Bateman (2005). Obviously, the accuracy of value transfer is conditioned on the measurement errors contained in original studies, and part of the measurement error transferred from the original value estimation may be amplified in a value transfer if care is not taken to minimize such effects (Wilson and Hoehn, 2006).

12.2 *Case study illustration: international transfer of water quality values*

One of the objectives of AquaMoney was to test, and as a result identify, the baseline conditions needed to be in place in order to make value transfer work in the context of water quality improvement across European river basins by minimizing transfer errors.

For this, a common valuation design was applied in five different EU countries, including the newly developed water quality ladder presented in Chapter 7 to accommodate the international transfer of the non-market benefits of water quality improvements.

Economic theory suggests that the value of improvements to a given, spatially confined, environmental resource such as water should be determined by the change in provision, core characteristics of the valuing individual common to all contexts (e.g. their income), characteristics of the site (e.g. distance from the valuing individual) and the availability of substitutes. Study selection criteria for value transfer in the 1992 *Water Resources Research* special issue on benefits transfer relate accordingly to similarity of the goods to be valued, the change in their provision level, the sites where these goods are found, and the population of beneficiaries and their characteristics, including their distribution around the sites.

The validation of value transfer analyses is essentially twofold. First, the analyst wishes to test whether a value function estimated from some study site(s) accurately estimates values for some policy site(s). The evaluation of transfer validity then consists of calculating the transfer error as the difference between the value for a site as estimated from the function estimated from other sites, compared with the value of that site as estimated from its own data. In the case of unadjusted mean WTP value transfer, the null hypothesis is simply $WTP^p = WTP^s$ where WTP^p is actually observed mean WTP at the policy site and WTP^s mean WTP transferred from the study site to the policy site. Equality of mean WTP values is tested in this study using the non-parametric Mann-Whitney test. The relative transfer error equals $(WTP^s - WTP^p)/WTP^p * 100\%$. The null hypothesis in the case of the value function transfer is: $WTP^p = \beta^s X^p$. WTP at the policy site is predicted using the parameter estimates from the study site and the values of the explanatory factors at the policy site (e.g. Bergland et al., 2005). Whether in this case study the estimated functions for individual countries originate from the same underlying generic distribution is tested using dummy variables for the different countries in a pooled model (choosing one of the countries as the baseline category). If the country dummies are statistically significant in the estimated pooled model, we conclude that the value function is not transferable across countries (e.g. Downing and Ozuna, 1996). The errors associated with the transfer of the value function are based on an out-of-sample prediction approach. That is, WTP is predicted for one country using the value function estimated based on the data from the four other countries.

Prior to our value function transfer exercise, we conducted a number of preliminary transfers of non-pooled, individual country, values for each scope and ordering good. Ignoring framing effects resulted in rejection of all the equality tests and underlines the need to control for framing effects. The hypothesis of equal means of WTP is rejected across countries and between any one country and another[11]. Overall, these results reinforce the need to obtain and pool sufficient data when dealing with the transfer of variable values across multiple spatial contexts. Small sample transfer exercises based upon just a few survey sites are highly liable to failure.

[11] The only exceptions being the Large1 and Small1 values in Denmark and Norway, and the Small2 value in Denmark and the UK.

Using the full set of data, we start by undertaking simple univariate transfers, controlling for framing effects, to provide a reference point against which we can subsequently assess function transfer findings. Here, data from all countries are pooled except the one which we are transferring to. The former are our 'study' sites from which we conduct an out of sample transfer to the latter 'policy' site. Transfer errors are calculated as usual by comparing this predicted value with the actual WTP elicited at that fifth country with transfer errors calculated as a percentage. We repeat this exercise for each country in turn and each scope-order combination. However, as we find no significant difference between the Large1 and Large2 values, we further merge responses to undertake univariate transfers for a combined 'Large' good. Errors for these univariate transfers are presented in the upper part of Table 12.1. As might be expected given the care expended to ensure commonality of approach and good across studies, even these simple univariate transfers yield lower rates of error than most previous international transfer studies. However, there are clear exceptions, the most obvious being the case of Lithuania, and the overall transfer error is more than 100% which would typically devalue the use of such estimates for practical decision making purposes.

Moving to estimating a value function and using this estimated function to transfer between countries, we follow the theory driven methodology in selecting predictor variables for our value transfer function by initially only adding in those variables which economic theory suggests should impact upon WTP in a consistent manner, avoiding contextual variables even though these may maximise the degree of statistical fit at individual sites. The impact of adding even a single theory driven variable (income) is dramatic, immediately cutting average transfer error from 116% to 38%. In the case of the UK, the errors are even reduced to single figures. This provides the strongest evidence in favour of our proposed methodology and transforms, so we argue, value transfer into a workable policy tool. We subsequently add two further theory driven variables, distance to the improvement site (capturing distance decay) and distance to the nearest substitute site. This induces a further reduction in transfer error to 35%.

To test the contention that the inclusion of variables regarding which economic theory has no prior expectations or which would just be chosen because they improve statistical fit at study sites induces context specific error into the transfer process, two further variables are added regarding which economic theory has no prior expectations, i.e. *Age* and *Urban*. Repeating our transfer exercise we duly find that errors increase very substantially over the theory driven models increasing by more than half to stand at an average of 56%. While this still outperforms univariate transfer (a result which may reflect the close commonality of design and provision change across studies), this represents a major reduction in performance compared to the theory driven models.

Table 12.1: Transfer errors (%) from univariate and multivariate value function transfers of pooled models

WTP measure	**Belgium**	**Lithuania**	**Denmark**	**Norway**	**UK**	***Average errors (weighted)***[1]
UNIVARIATE TRANSFERS						
Small improvement first	49	508	6	41	48	*130 (102)*
Small improvement second	53	392	23	43	69	*116 (94)*
Large improvement	39	391	10	37	34	*102 (81)*
Average errors (weighted)[2]	*47 (45)*	*430 (420)*	*13 (12)*	*40 (40)*	*50 (46)*	*116 (90)*
MULTIVARIATE TRANSFERS						
Variable = INCOME						
Small improvement first	45	67	20	62	0	*39 (40)*
Small improvement second	50	28	26	67	0	*34 (39)*
Large improvement	30	90	29	51	3	*41 (40)*
Average errors (weighted)[2]	*42 (39)*	*62 (69)*	*25 (26)*	*60 (58)*	*1 (2)*	*38 (40)*
Variables = INCOME, DISTANCE TO SUBSTITUTES AND SITE						
Small improvement first	51	12	14	66	1	*29 (34)*
Small improvement second	54	70	14	71	4	*43 (45)*
Large improvement	37	39	25	58	2	*32 (35)*
Average errors (weighted)[2]	*47 (45)*	*40 (40)*	*17 (19)*	*65 (63)*	*2 (2)*	*35 (37)*
Variables = INCOME, DISTANCE TO SUBSTITUTES AND SITE, AGE AND URBAN-DUMMY						
Small improvement first	65	112	8	71	48	*61 (58)*
Small improvement second	70	32	5	77	61	*49 (51)*
Large improvement	51	115	20	63	41	*58 (55)*
Average errors (weighted)[2]	*62 (59)*	*86 (93)*	*11 (13)*	*71 (69)*	*50 (48)*	*56 (55)*

1. Raw averages given outside parentheses. Figures in parentheses are average errors per country weighted by relative sample size (sample country/total country)

2. Raw averages given outside parentheses. Figures in parentheses are average errors per scenario are weighted by the number of times each scenario is included (Small1:Small2:Large=0.25:0.25:0.5)

In summary, the central contention underpinning the case study presented here is an agreement with the Pearce et al. (1994) argument that, because value function transfers allow the analyst greater control over differences across sites, they should yield lower transfer errors than simple univariate transfers. That this needs assertion and support arises from the mixed evidence presented in the literature wherein several empirical studies have reported the opposite result, finding that function transfers yield higher errors than univariate transfers. We argue that the reason for apparently perverse results reported in the literature is in part because the value functions employed for transfers have been designed following statistical best fit principles rather than being designed for transfer purposes. In doing so, statistically driven functions include context specific variables rather than the generic drivers of preference highlighted by economic theory. The starting point for the transfer exercise was that a common design format for the source studies used in developmental work is needed (Desvousges et al., 1992). However, we argue that, in addition, theoretically driven value functions are warranted to avoid context specific transfer errors.

Therefore, we propose to extend the general principles of design and analysis that should be followed for reliable multi-country benefit transfer exercises with the following:

- The description of the good should be common in all study sites; we have employed a common water quality ladder, based on a new classification based on the WFD linking ecological and physical determinants to resultant impacts upon flora and fauna and suitability for different uses, which we hope to be applicable to future cases studies in this area;
- Study design should be developed from economic theoretic principles and employ a theoretically consistent utility specification, for subsequent robust validity testing of theoretical expectations and transfer errors;
- The framing of valuation questions and the method through which valuation responses are elicited should either be common or should vary sufficiently to ensure that the effects of such variation can be assessed.

12.3 *Case study illustration: international transfer of water conservation values*

Compared to CV, choice experiments are believed to be better equipped for value transfer. Their multi-attribute nature allows for the valuation of marginal changes in good and site characteristics and hence account for differences in environmental quality when transferring values between sites (Morrison et al., 2002). Accounting furthermore for respondent preference heterogeneity using a random parameters model has also been shown to reduce the magnitude of the transfer error (Colombo et al., 2007). An overview of the limited number of studies testing the reliability of value transfer based on choice experiments is provided in Morrison and Bergland (2006).

This second example is based on the choice experiment study presented in Chapter 6 examining public preferences for water conservation policies related to outdoor domestic use and environmental non-use values. Table 6.2 presents the estimated choice models in Greece, Italy and Spain and Table 6.3 the estimated welfare measures related to 5 different possible policy scenarios. Formal tests of transferability and transfer errors will be presented here in this section based on these two tables. The prime interest is to assess the transferability of the estimated choice models and compare the predicted WTP values for the five policy scenarios from these models. Simple tests of the equality of the estimated implicit prices presented in Chapter 6 show that these are more or less the same for domestic water use, but not for environmental water use. In this section, three additional tests will be performed.

First, the equality of the estimated choice models is tested using the Swait and Louvier (1993) test procedure. The null hypothesis of equal coefficient estimates when rescaling the models for the purpose of cross-country comparison is convincingly rejected in all cases, despite the equality of MWTP for domestic water use between Greece and Italy and Greece and Spain. Hence, none of the estimated models presented in Table 6.2 are the same and as a consequence non-transferable. Given this outcome, transferring the models is expected to result in prediction (transfer) errors. The question, however, is how large these errors are and what can be done to reduce these errors. This will be examined in the next two steps.

Second, we examine differences in prediction error when transferring 'unit values' estimated from a standard conditional logit model including the choice attributes only (UVT in Table 12.2) compared to value function transfer where we account for preference heterogeneity in the choice model (FVT in Table 12.2). In the latter case, the estimated models presented in Table 2 are used to predict the CS welfare measures across different countries (e.g. the Greek function is used to predict the CS of the 5 policy scenarios in Italy etc.). This functional approach to value transfer (FVT) is theoretically expected to be superior to unconditional unit value transfer (UVT) since effectively more information can be transferred. Like Colombo et al. (2007), we find that the transfer error is on average lower when accounting for preference heterogeneity through a functional transfer. Across all transfers, the average error is reduced by 23 percent from 115 to 89 percent when accounting for preference heterogeneity in the different country samples[12]. Function transfer is superior to unit value transfer in 77 percent of the 30 investigated cases in the upper part of Table 12.2.

Transfer errors are generally lower for the first two policy scenarios where the likelihood of a domestic water use restriction is reduced from 4 times in the next 10 years to 2 times or 1 time than for the third and fourth policy scenarios where environmental quality is improved from poor to good or very good water status. This is especially the case for transfers involving Greece and Spain and Greece and Italy where implicit prices for a reduced likelihood of domestic water use restriction were more or less the same. Irrespective of the transfer method (UVT or FVT), transfer errors are, on average, in these cases (between Greece and Italy and Greece and Spain) about twice as high when transferring CS welfare measures for environmental quality improvements compared to the transfer of CS welfare measures for reduced domestic water use restrictions.

The transfer error is lowest when predicting CS for the domestic water use policy scenarios based on UVT for Greece and Spain (15-17%). This can be explained by the fact that the null hypothesis of equality of their implicit prices for domestic water supply security could not be rejected at the 10 percent level. Adding in these cases more control for respondent characteristics only increases the error. Accounting for preference heterogeneity has most effect, i.e. results in the highest reduction of transfer errors, when transferring the choice model from Spain to Italy (the transfer error is reduced by almost half), followed by the transfer from Greece to Italy. Transfer errors are also highest when transferring the estimated choice models in Greece and Spain to Italy. In other words, CS

[12] The average transfer error across 27 different soil erosion policy scenarios between two Spanish watersheds in Colombo et al. (2007) was reduced more substantially by 57% from 154 to 66% when controlling for preference heterogeneity.

Table 12.2: Absolute transfer errors (%) between countries and across policy scenarios

		Policy scenario											
		1 Restrictions from 4 to 2 in 10 years		2 Restrictions from 4 to 1 in 10 years		3 From poor to good ecological status		4 From poor to very good ecological status		5 Combination scenario 2 and 4		Average	
Study site	**Policy site**	**UVT**	**FVT**	**UVT**	**FVT**	**UVT**	**FVT**	**UVT**	**FVT**	**UVT**	**FVT**	**UVT**	**FVT**
Greece	Italy	157	184	157	160	398	272	398	269	316	203	285	218
Italy	Greece	61	61	61	59	80	70	80	71	76	65	72	65
Greece	Spain	15	57	15	40	89	87	89	79	50	46	52	62
Spain	Greece	17	40	17	32	47	49	47	46	34	33	33	40
Italy	Spain	67	48	67	49	62	51	62	53	64	53	64	51
Spain	Italy	202	80	202	87	163	96	163	105	176	108	181	95
Average		**87**	**78**	**87**	**71**	**140**	**104**	**140**	**104**	**119**	**85**	**115**	**89**
Pooled: Italy+Spain	Greece	41	51	41	46	14	59	14	58	7	49	23	53
Pooled: Greece+Spain	Italy	180	97	180	96	280	149	280	157	246	136	233	127
Pooled: Greece+Italy	Spain	22	22	22	28	64	6	64	8	55	21	45	17
Average		**81**	**57**	**81**	**57**	**119**	**71**	**119**	**74**	**103**	**69**	**100**	**66**

Absolute transfer errors are calculated as follows (Brouwer and Bateman, 2005): $|(WTP_{studysite}-WTP_{policysite})/WTP_{policysite}|*100\%$.

UVT: unit value transfer based on conditional logit model including attributes only.

FVT: functional value transfer based on mixed logit models presented in Table 2 accounting for preference heterogeneity.

values are difficult to predict for the Italian sample. The other way around, transferring the estimated Italian choice model to Greece and Spain yields substantially lower prediction errors, varying between 49 and 80 percent. Note that transfers from Greece to Italy and Spain and Spain to Italy always result in an overestimation and transfers from Italy to Greece and Spain always in an underestimation of actual WTP.

Third, we also investigate to what extent pooling the data from two samples helps to better predict the CS for the third. Colombo and Hanley (2008) show that this may increase the probability of yielding smaller transfer errors, but also result in even higher errors. In our international case study, pooling the data yields a systematically lower transfer error in one third of all transfer cases, namely when transferring the pooled Greek and Italian samples to Spain across all policy scenarios (Table 12.2). Transfer errors range between 6 and 28 percent in that case.

This is lower than the results found in the study by Colombo and Hanley (2008) where the pooled approach outperformed the single sample approach in 58 percent of the transfer cases. However, overall, using the pooled approach reduces the transfer error by 26 percent compared to the single sample approach, which is more than the 15 percent found in Colombo and Hanley (2008)[13]. When pooling the samples, function approach systematically outperforms the unit value transfer approach when predicting welfare measures for Italy, but not for Greece and Spain. In the latter case the function approach yields lower transfer errors for the environmental quality improvement scenarios and the fifth combined policy scenario. The predictive power of the single Spanish and Italian models is higher than pooling these models to estimate the CS welfare measures for the 5 policy scenarios in Greece.

In conclusion, formal transferability tests show that the estimated choice models differ significantly from each other across the countries, undermining their transferability. Significant differences exist between the implicit prices for environmental water use across the three countries, but not for domestic water use. Securing domestic water use is valued more or less the same across two of the three South European countries, whereas preferences differ in these three countries when it concerns the allocation of limited water resources to the environment.

Using the choice models and underlying utility functions to predict the monetary welfare implications of different policy scenarios to improve water availability for domestic or environmental use or combinations of the two across countries, errors range between 15 and 398 percent. CS values are particularly difficult to predict for the Italian sample, and adding control here for preference heterogeneity in the environmental policy scenarios is most effective in reducing transfer errors. As expected, a substantial reduction in transfer error of almost 25 percent is achieved when controlling for preference heterogeneity in the estimated choice models. This provides additional empirical support for the superiority of the functional approach to value transfer. Transfer errors are lowest between Greece and Spain and highest between Greece and Italy.

[13] The average transfer error is much higher in their study of landscape restoration (410%) compared to this study where the average overall error using the pooled approach (based on Table 12.2) is 65%.

Promise is also found in pooling samples. Transfer errors can be reduced by another 25 percent when pooling two country samples to predict welfare measures in a third country instead of using single country sample transfers. Lowest errors varying between 6 and 28 percent are found when using the pooled Greece-Italy model to predict the policy scenario welfare measures in Spain.

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13. Value aggregation

13.1 *Introduction*

The issue of defining the population of beneficiaries is a particular challenge for water valuation studies for reasons to do with defining the relevant extent of the market and complications related to the distinction between user and non-user vales. However, also the spatial distribution of water bodies and their users plays a role here. In practice, it is often the aggregation procedure, which receives most criticism when using non-market valuation study results to inform policy and decision-making. The average values may remain undisputed, but the population of beneficiaries across which these values are added up may result in very high or low values depending on the chosen market size. An example is the public inquiry carried out in the UK after a consultant used average economic values for water abstraction values from the existing literature and decided to aggregate these values across the political boundaries of a water company many times the size of the original study with substantially higher population density (e.g. Bateman et al., 2000).

The natural and economic system demarcate the boundaries of the market over which one can aggregate the WTP function, i.e. a value function has to be estimated, which takes into account these spatial characteristics and patterns and the influence they have on human behaviour and valuation. In particular, aggregation problems arise as a result of (a) separation across water bodies with different goods and services and substitution possibilities and aggregation to river basin level; (b) separation across populations; (c) with the separation of TEV parts – use and nonuse and double counting; (d) double counting because of ecosystem functioning, e.g. which are not necessarily spatially defined.

While the adoption of the aquatic ecosystem functional approach is used to identify water resource good and services, if each of them is identified separately, and then attributed to underlying functions, there is the likelihood that benefits will be double counted. Benefits might therefore have to be explicitly allocated between functions. For instance, Barbier (1994) notes that if the nutrient retention function is integral to the maintenance of biodiversity, then if both functions are valued separately and aggregated this would double count the nutrient retention which is already ‘captured’ in the biodiversity value. Some functions might also be incompatible, such as water extraction and groundwater recharge, so that combining these values would overestimate the feasible benefits to be derived from the ecosystem. Studies that attempt to value the aquatic system as a whole based on an aggregation of separate values tend to include a certain number of functions although these studies do not usually claim to encompass all possible benefits associated with the system. Some functions may be mutually exclusive and, therefore, cannot be aggregated. For example, aggregation of the values for both extraction of water and recharge of water would overestimate the benefits that could feasibly be derived from a water resource. Interactions can occur between functions, and some functions may be complementary.

In practice, the ability to use water resources repeatedly or simultaneously for different uses means that competition and complementarity are important considerations in valuation. This means that total valuation (estimation of the full value of an aquatic ecosystem) is undertaken only when necessary. Assessments are more commonly based on partial valuation, based on a sectoral approach or on a specific good or service or impact.

In considering how broadly individuals' benefits be aggregated (i.e. their marginal benefit schedules be vertically summed to give the extent of the 'market'), one can distinguish between the political jurisdiction, concerning some administrative area, and the economic jurisdiction incorporating all those who hold economic values (or benefit) from the good(s) or service(s) in question. From an economic cost-benefit point of view, the notion of accounting stance defines the relevant jurisdiction for assessing benefits. The accounting stance should be such that it captures all Pareto-relevant impacts. Where political and economic jurisdictions coincide, there is said to be *fiscal equivalence*.

The relevant population for an economic assessment will also depend in part on the type of function that is being valued. Given unlimited sampling resources, one would ideally sample from the entire feasible economic jurisdiction. The problem however, is that, *a-priori*, the extent of this area is unknown. As such, studies have tended to define the accounting stance with reference to a political jurisdiction based on some administrative area, for example, water company operations boundaries.

Another approach however is to define the economic jurisdiction through the identification and estimation of a spatially sensitive valuation function. In defining the economic jurisdiction, such an approach is able to adjust for the fact that underlying values for changes to some spatially confined resource like water are likely to decay with increasing distance from the resource. Aggregation procedures need to be able to recognise and address this problem. The use of a spatially sensitive valuation function explicitly incorporates distance decay relationships into defining the limits of the economic jurisdiction while allowing for variability in the socioeconomic characteristics of the encompassed population within the aggregation process. The economic jurisdiction is defined by the area within which there are positive values. The maximum limits of this areas can be found, given a significant distance decay relationship, by predicting the point at which WTP values decline to zero using the estimated value function that calculates the level of WTP as distance increases and holding non-distance explanatory variables constant (at their mean) to estimate the distance at which WTP is zero. As discussed in chapter 7, it may be necessary to take into account substitution effects arising from quality change at multiple sites. In such cases the predictions of single site studies may be poor guides to the impact of more widespread changes, as envisaged by the WFD. Research suggests rapidly diminishing marginal WTP for additional sites once an initial level of recreational or non-use provision is made (Bateman et al., 2005).

13.2 *Case study illustration*

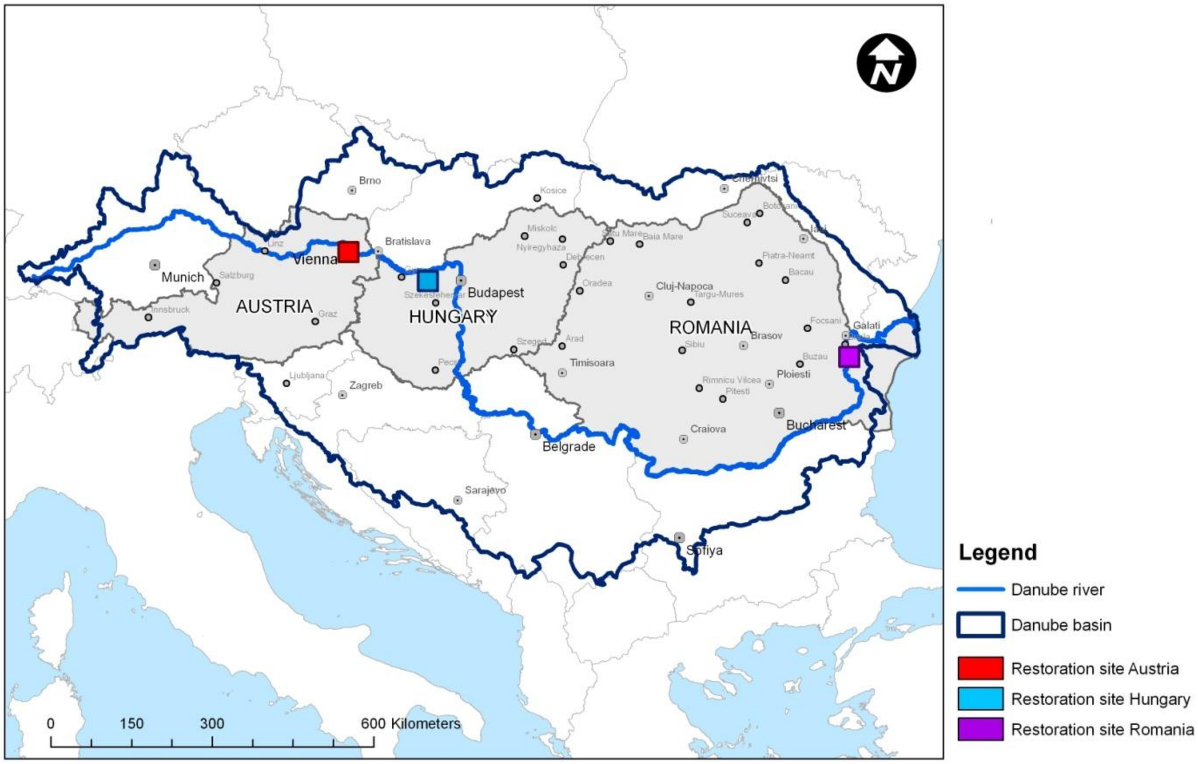
The example presented below taken from the AquaMoney case study ecological floodplain restoration of the international Danube river basin illustrates the impact of the aggregation procedure on the estimated welfare measure.

With a length of 2,780 km and a catchment area of more than 800,000 km^2 the Danube is the second largest river in Europe. The river has been subject to anthropogenic modification and environmental pressures over the past centuries, including canalization, construction of embankments, navigation, use of hydropower and pollution. Although some parts of the river still are in a near-natural state, most river stretches have been classified as heavily modified under the WFD due to embankment and regulation works, and intensive navigation. The shape of the river has been drastically changed and large parts of the associated formerly-waterlogged area have been drained for agricultural purposes, reducing the connectivity between the Danube and the surrounding area and its tributaries to small patches. Only in small areas, such as the Danube National Park in Austria, are the original Danube ecosystem and adjacent wetlands still intact. Restoring the river to a 'good ecological status', as required by the European WFD, is only feasible if parts of the river are transformed back as close as possible to their original state with regard to natural hydro-morphological conditions. Various river restoration projects have been identified in the Danube river basin. Three of them were included in the AquaMoney case study (Figure 13.1): the Donau-Auen National Park in Austria, Által-ér in Hungary, and the Islands of Braila in Romania.

Through the application of a common choice experiment design, the economic values of river restoration were compared in Austria, Hungary and Romania. GIS was used to aggregate economic welfare implications of these policy scenarios across the relevant population of beneficiaries in the three countries, controlling for distance-decay effects and differences in income distribution across the international river basin.

The common choice experiment assesses the non-market value of the ecosystem services associated with river restoration, and consists to this end of two exclusive categories of benefits: the impact of river restoration on floodwater storage and a corresponding reduction in flood risk, and the river's nutrient retention capacity and hence water quality. These benefits make up two of the three attributes used to evaluate river restoration policy alternatives. A monetary cost price was included as a third attribute to enable monetization of the benefits of different river restoration projects. The inclusion of a monetary attribute in the choice model allows for the estimation of monetary WTP welfare measures for different river restoration policy scenarios and changes in individual components of these scenarios. These monetary welfare measures form the basis for the international comparison and transferability tests.

Figure 13.1: Location of the Danube river restoration projects



The alternatives in the CE describe different end states created through river restoration measures. The link between these end states and river restoration projects is explained in the introduction to the CE. Variations in end states are caused by different degrees of river restoration and corresponding scale effects. In this way, respondents were not asked to value the river restoration measures per se, but rather their outcomes in order to avoid correlation due to causality. To increase the realism of the presented alternatives, respondents were shown existing river restoration plans on a map. The same map was used in the three case study applications: a 2000 CORINE land cover map at 1:100,000 scale, displaying human settlements, agriculture, forests and meadows, wetlands and freshwater ecosystems.

The main survey was carried out simultaneously in all three countries in November 2007. In Austria, the main survey targeted a random online sample of 1,977 households from a representative household market panel in Vienna and Lower Austria. The response rate was 26 percent. Stratified sampling procedures were followed in Hungary and Romania, based on gender, age and representative shares of the rural and urban population living along both sides of the Danube river. In Hungary, 892 people were asked to be interviewed face-to-face, of whom 471 agreed to participate (a response rate of 53%), while in Romania 519 of the 850 respondents who were asked to participate in the face-to-face interviews completed the questionnaire (a response rate of 61%).

Based on the estimated choice models (full details are provided in Brouwer et al., 2009), a number of policy scenarios were simulated and their welfare implications estimated, changing flood frequency and water quality simultaneously (Table 13.1). Tests were carried out to see to what extent these CS measures were transferable across the three river basin countries. For the purpose of international comparison, the estimated welfare measures were adjusted as in the previous chapter for differences in purchasing power (Ready and Navrud, 2006). Two policy scenarios involve the improvement of water quality to a good ecological status, with flood risk variations of once every 25 and 50 years, and three policy scenarios involve water quality improvements up to very good ecological status with flood risk reductions varying from once every 25 years to once every 100 years. Average values were used for respondent characteristics, including income and distance, under the assumption that the samples were representative for the whole river basin in each country.

Table 13.1: WTP welfare measures for different policy scenarios (€/household/year)[1]

	Policy scenario		**CS**		
	Flood risk	**Water quality**	**Austria**	**Hungary**	**Romania**
1	Once every 25 yrs	Good	69.6	20.2	4.8
			(37.5-101.8)	(15.2-25.3)	(4.0-5.5)
2	Once every 50 yrs	Good	73.5	20.2	4.8
			(40.8-106.2)	(15.2-25.3)	(4.0-5.5)
3	Once every 25 yrs	Very good	85.0	32.1	9.5
			(50.2-119.8)	(26.3-37.9)	(6.8-12.2)
4	Once every 50 yrs	Very good	88.8	32.1	9.5
			(53.4-124.2)	(26.3-37.9)	(6.8-12.2)
5	Once every 100 yrs	Very good	96.4	32.1	9.5
			(59.6-133.3)	(26.3-37.9)	(6.8-12.2)

[1] 95% confidence intervals between brackets.

In Austria, the WTP welfare measures for the policy scenarios increase gradually for both the flood return period and water quality. The results for Hungary and Romania only differ due to water quality changes (higher quality yields, as expected, a higher value) as flood risk appeared not to be a significant determinant of choice behavior. The estimated WTP therefore are the same when keeping water quality levels constant and varying flood risk.

As expected based on the observed income differences, the WTP in Austria is significantly higher than WTP for the same policy scenario in Hungary, while WTP in Hungary are significantly higher than WTP in Romania for each policy scenario. Based on the results in Table 4, transfer errors vary between 62 and 263 percent for transfers between Austria and Hungary (average 138%), 89 and 1442 percent for transfers between Austria and Romania (average 580%), and 70 and 325 percent for transfers between Hungary and Romania (average 172%). However, transfer errors are reduced substantially when transferring the entire choice model from one river basin country to another instead of the predicted unadjusted average values in Table 4. On average, errors are reduced by 45 percent based on the estimated value functions as compared to unadjusted value transfer for transfers between Austria and Hungary, 62 percent for transfers between Austria and Romania and 73 percent for transfers between Hungary and Romania. However, the predicted values remain untransferable if an error tolerance level of 20 percent is used for value transfer. Errors are lowest when transferring the estimated choice models between Hungary and Romania (39-52%), and highest when transferring the models between Austria and Romania (60-519%). To what extent these errors are acceptable in a cost-benefit policy evaluation depends on policy-maker demand for accurate estimates.

A final step in the welfare estimation procedure is the aggregation of the estimated WTP values across the population benefiting from the welfare gains associated with the river restoration policy scenarios. This step is often critical to arrive at valid and reliable (and hence credible) estimation of total economic value (TEV). Most studies simply use the number of people living in an administrative unit or geographic jurisdiction (e.g. county, province, state or country), and average values are transferred unconditionally (i.e. uncorrected) across the population living within the boundaries of this geographical unit. Depending on area size and population density, aggregated TEV can differ enormously (Bateman et al., 2006). Strict guidelines for welfare aggregation do not exist, making the estimation procedure vulnerable to manipulation. In view of the fact that the population from which the samples in this study were drawn and their characteristics are unevenly distributed over space, the aggregation procedure is carried out using GIS.

To this end, GIS data about Europe's major rivers including the Danube from the 2008 ESRI database were combined with (1) the JRC 100 by100 meter population density grid (Gallego, 2008) and (2) NUTS-3 level information about per capita income disaggregated from NUTS-3 regions to 100 x 100 m grid cells[14]. Euclidian distances were calculated per 100 by 100m grid cell to the Danube river in meters. A TEV is calculated for two policy scenarios: improvement of water quality in the Danube river to (1) good and (2) very good conditions (keeping flood conditions in both cases constant). The importance of accounting for preference heterogeneity in welfare aggregation procedures, in this case income and distance-decay for which secondary GIS data were available and additional calculations could be made, will be illustrated by comparing the outcomes of two different TEV approaches: TEV calculated by aggregating the unadjusted average values across the whole population living within the boundaries of the administrative units surveyed in this case study (TEV_1)[15] and TEV adjusted for the estimated income and distance-decay effects (TEV_2). In the former case, the sample population and the

[14] The Nomenclature of Territorial Units for Statistics (NUTS) is a breakdown of territorial units to harmonize regional European statistics. NUTS-3 is the lowest aggregation level, and usually follows a European member state's own regional administrative structure.

[15] The administrative units surveyed in this case study were the states of Lower Austria and Vienna in Austria, Komárom-Esztergom County in Hungary and the counties Braila, Constanta, Ialomita and Tulcea in Romania.

estimated WTP per capita are assumed to represent the population at large in the administrative units. The steps in the latter aggregation procedure are summarized below. First, information about population density was converted to number of inhabitants per 100x100m grid cell. Second, the average WTP for reaching good and very good water quality with the help of river restoration were converted from per household to per capita values based on each sample's average household size, and multiplied with the number of inhabitants in each 100 by 100m grid cell. For each grid cell also average per capita income was determined based on the specific geographical NUTS-3 area to which a grid cell belonged. This average per capita income was subtracted from mean per capita income in the sample and multiplied on a cell by cell basis with the estimated income coefficient for each country and the number of people living in each cell[16]. In this way, the economic value per grid cell was modified upwards or downwards depending on the income difference.

The distance of each grid cell to the Danube river was used to correct the economic value per capita per grid cell for the distance-decay effects detected in each country[17]. The estimated distance-decay factor was multiplied by the calculated distance of each grid cell from the river in kilometres and multiplied by the number of people (illustrated for good water quality in Austria in Figure 13.2). In a final step, the income and distance adjusted values (in the case of Hungary only distance adjusted) are added up to estimate the TEV of river restoration to good and very good water quality (illustrated for very good water quality in Romania in Figure 13.3). The results of the two aggregation procedures are presented in Table 13.2. As before, the values are adjusted for differences in purchasing power between the three countries.

[16] Except in Hungary. The area surveyed in Hungary falls within the boundaries of a single NUTS-3 region, so no income variation can be found within this area based on the available GIS data.

[17] Modified for the percentage of respondents who visited the case study area in Romania and the percentage of flood affected households in Hungary (see Table 3).

Figure 13.2: Illustration of distance-decay effects for good water quality in Austria

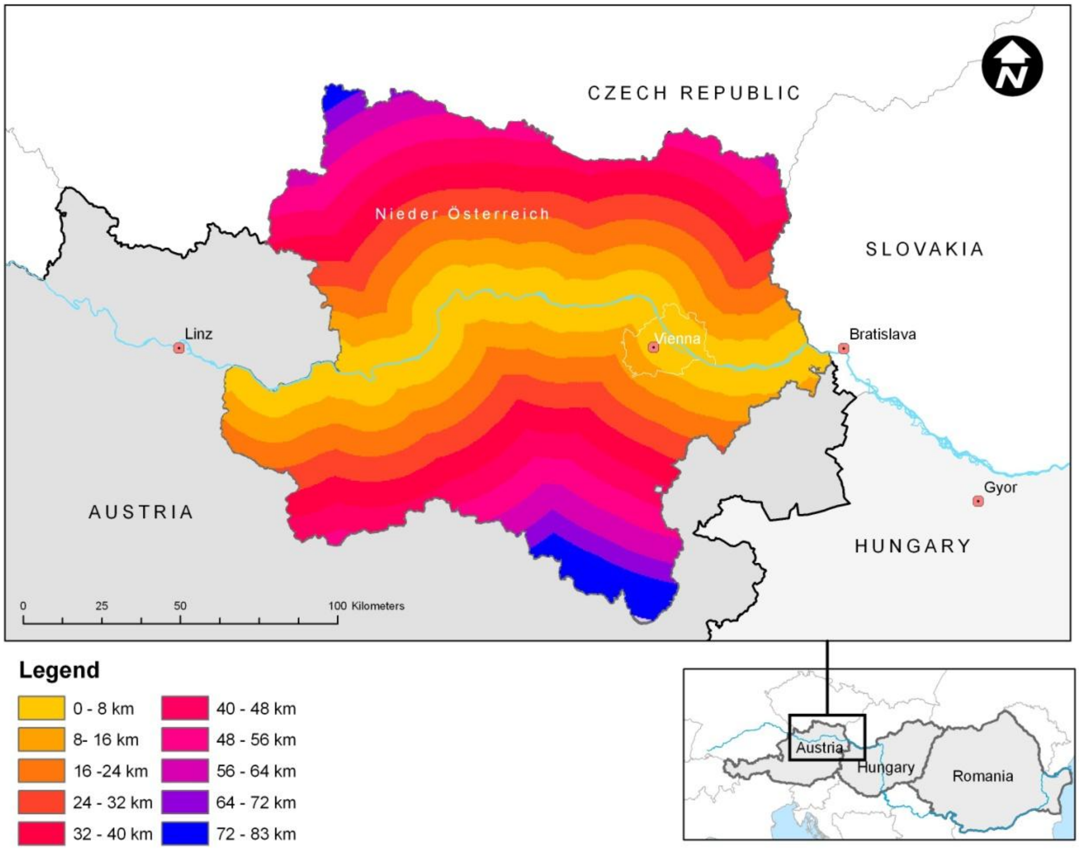
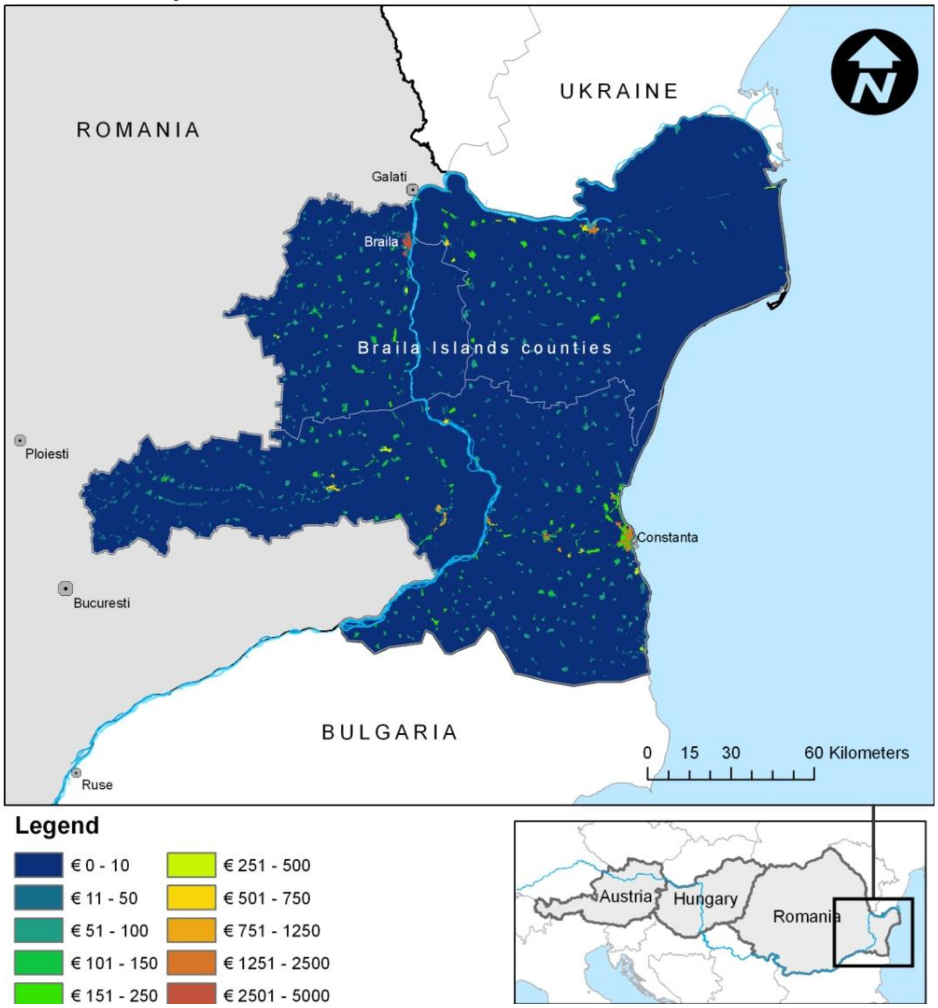


Figure 13.3: Illustration of TEV for very good water quality in Romania adjusted for distance-decay



The most striking observation from Table 13.2 is the difference between TEV_1, calculated by simply multiplying the number of inhabitants living inside the administrative units, and TEV_2 where the CS is adjusted for (a) distance-decay only and (b) distance-decay and income differences. On average, accounting for distance-decay yields a 30 percent lower TEV than the unadjusted TEV, while the additional income correction reduces TEV by another 10 percent. These differences are most pronounced for Romania. Accounting for distance-decay results in Romania in a 35 to 50 percent lower TEV (for very good and good water quality respectively). In other words, not accounting for distance-decay overestimates the TEV by a factor of 1.5 to 2. Based on the available NUTS-3 information about average income levels in the study area (which were lower than the sample average possibly due to the fact that the sample consisted of a relatively high share of more higher educated respondents), TEV is reduced by an additional 9 to 12 percent (for very good and good water quality respectively).

Table 13.2: TEV for good and very good water quality based on two aggregation procedures (€10^6/year)

	TEV$_1$ unadjusted aggregation	**TEV$_2$ GIS adjusted aggregation**	
		Distance-decay correction	Distance-decay and income correction
Austria			
Good water quality	54.0	46.3	41.9
Very good water quality	73.8	na	69.4[a]
Hungary			
Good water quality	2.0	1.4	na
Very good water quality	3.9	3.3	na
Romania			
Good water quality	10.2	5.0	4.4
Very good water quality	15.3	9.9	9.0

[a] Income correction only. na: not available.

In Hungary, unadjusted TEV is 42 percent higher than TEV adjusted for distance-decay in the case of good water quality, and 17 percent in the case of very good water quality. No income correction could be carried out because the survey area falls completely within one NUTS-3 region, and so there was no variation in income levels within that area.

In Austria, accounting for distance-decay produces a TEV for good water quality that is 14 percent lower than the unadjusted TEV. Because the distance-decay effect was only significant when included as an interaction term with good water quality, no adjusted TEV for very good water quality is presented for Austria in Table 13.2. Adjusting TEV also for the spatial variation in income levels across the survey area lowers TEV by another 10 percent. The sample income average was somewhat higher than the average in the whole survey area, possibly due to the overrepresentation of respondents from Vienna where average income levels are relatively high compared to the rest of the country. The income adjusted TEV for very good water quality is 6 percent lower than unadjusted TEV.

Summarizing, the estimated choice models in the three countries were significantly different, both in terms of model parameters and variances, indicating that their potential for transferability is limited. Observed and unobserved heterogeneity plays a significant role in the three countries, and is an important reason for the non-transferability of the results. The PPP adjusted marginal values for water quality improvements were the same in Hungary and Romania, but the PPP adjusted CS estimates for different floodplain restoration policy scenarios were different. None of the estimated WTP welfare measures were transferable at the 20 percent error tolerance level imposed. When controlling for respondent heterogeneity, most of the transfers also exceeded for instance a 50 percent tolerance level (transfer errors were lower than 50% in only 30% of the cases), even through transfer errors were reduced by up to almost 75 percent. Hence, the ‘function approach’ to value transfer clearly outperformed the ‘unadjusted value approach’, but transfer errors remained relatively high, varying on average between 48 and 380 percent for different floodplain restoration scenarios. Accounting for heterogeneity due to distance-decay and income effects in the welfare aggregation procedure of the WTP estimates across the population of beneficiaries also consistently resulted in lower TEV across the three countries. Compared to unconditional aggregation of average WTP estimates over the population living within the boundaries of the administrative units from which the survey samples were drawn (common practice in value transfer studies), the calculated TEV turned out to be 10 to 50 percent lower in the more sophisticated GIS-based aggregation procedure presented in this paper.

In conclusion, the welfare impacts of floodplain restoration in the context of the WFD differ significantly throughout the Danube river basin, i.e. within and between the countries through which the river flows. Errors are substantial when using country-specific values and models to predict values elsewhere, even if one takes preference heterogeneity and different socio-economic conditions in the three countries that were part of this study, such as purchasing power, into account. Similarly, substantial error may be created if average values are aggregated in the same country without controlling for preference heterogeneity and socio-economic conditions that are unevenly distributed in space. As such, both sources of error will affect the outcome of the policy analysis for which the values are to be used.

13.3 *References*

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14. Best practice recommendations

14.1 *Introduction*

Broadly speaking uncertainty regarding the benefits of measures estimated in a non-market valuation study can be introduced at the following stages of the study:

- study design
- sampling method and strategy
- analysis
- reporting.

Good valuation **study design** reduces uncertainty in respondents' willingness to pay responses because the concepts presented in the study generate less random responses. Good study design is also expected to reduce 'bias' relative to theoretical constructs the valuation survey is trying to measure. In study design we seek to minimise uncertainty regarding the elicitation of individual or household WTP estimates. In **sampling design** we aim at representative surveying the 'economically relevant' population so that aggregation of individual/household WTP does not miss out populations who have positive WTP or assign WTP to populations who do not care for the water body. In the **analysis** stage, model assumptions can increase or decrease individual/household WTP estimates by determining econometric model parameters. Finally, the way economists **report** individual/household WTP estimates and total benefits can determine which of the estimates are used by policy-makers in benefit-cost analysis. Which results are emphasised is particularly important when valuation approaches (e.g. CVM versus CE) or modelling assumptions (e.g. conditional logit versus mixed logit) have a significant impact on WTP estimates. Contextual information, such as high protest rates or zero WTP responses may also be important for how managers and policy-makers interpret the reliability/uncertainty of valuation results. Overall, the role of reporting is to give a fair picture of the different uncertainties/assumptions in the study.

In this section we do not present a complete and exhaustive review of such best-practices - a number of valuation textbooks and manuals exist, providing best-practices for valuation study design (see annex I). Instead we summarise the best-practices offered by authors of the various case studies in the project AQUAMONEY. We list a number of "problems" or challenges experienced in the AQUAMONEY case studies. Although many of the issues raised are generic, we have tried to focus on valuation design, sampling and analysis issues that are specific to the economic valuation of water bodies.

14.2 *Recommendations*

Table 14.1: Study design

ISSUE	RECOMMENDATION
A water quality ladder creates a standard hierarchy of **water use suitability thresholds** for users. However, users of water bodies are heterogenous within populations and between study sites; they have preferences for different types of recreation, and the conditions under which they will practice them will vary between study sites.	Ask respondents their use suitability thresholds explicitly, in a pilot, or at least prior to showing the water quality ladder that will be used in the valuation exercise. In choice experiments, use mixed logit models with interaction effects that account for heterogeneity in respondents suitability thresholds
Accessibility to water quality or quantity is not determined solely by respondent distance to the water body.	Ask respondents their current use levels for their favourite water body and closest substitute sites. Use this data to evaluate whether distance is a good proxy indicator for accessibility.
In a choice experiment there is a tension between the 'ideal' **orthogonal experimental design** where attributes (different ecological status characteristics; different water bodies) are uncorrelated vs. a "realistic" **correlation** between attributes as water quality characteristics due to ecological conditions (e.g. water colour and sight depth) and ecological functions relating water body attributes. For example hydrological connectivity (e.g. adjacent river stretches and lakes in a catchment have correlated water quality); biodiversity indicators are spatially correlated	Conduct simulations with different experimental designs allowing certain realistic correlation patterns between attributes; evaluate the trade-offs between these realistic constraints and experimental design efficiency (software such as Sawtooth allow simulation of the effectiveness of different designs).
Water bodies have **multiple uses** and values, often held by the same individual. Depictions of ecological status may prompt respondents to think of one or a combination of values.	Use questions on use suitability of ecological status for different uses, and follow-up questions to identify the combination of motives for WTP. Identification of users in order to cross check distance decay of WTP (a large proportion of non-users may lead to weak or insignificant distance decay).

ISSUE	RECOMMENDATION
Perceived rights to historical water quality levels may determine household expectations of a programme of measures in valuation scenarios. Water quality, restoration improvements that are too small relative to expectation may even lead to negative marginal utility	A study of historic water quality, flooding frequency, availability and historical uses under these conditions will help to determine the scale of improvements that may be considered consequential.
The distance decay in WTP reduces the significance of an ”additional” water body improved in the valuation scenario. If **scope of improvements** is evaluated as an additional water body improved distance decay may be the explanation for 'scope insensitivity' seen in several of the case studies. Where scope of improvements is defined as increase in water quality, use suitability thresholds may be an explanation for insensitivity to scope.	Choosing adjacent water bodies for the scope test will reduce the likelihood that distance decay of WTP leads to scope insensitivity. Define scope tests of water quality improvement which cross the use average suitability thresholds, revealed in the pilot study (see above).
Scenario pictograms and attribute wording simplify the **dimensions of value** of water	Complement WTP information with qualitative evaluation of scenarios from focus groups. Use open-ended questions in pilot surveys to evaluate whether water quality scenario descriptions capture characteristics of interest to the respondent.
'Information bias' or '**framing effect**' of policy relevance of water quality in valuation surveys	Identify the a priori importance of water resource issue among other societal policy objectives in questions before the valuation scenario; compare the proportion of a priori concerned respondents with the *a posteriori* proportion who responded WTP>0 to the scenario. A large difference indicates a framing effect
A significant number of respondents are expected to **protest** non-market valuation for a number of reasons determined by the study context	Include follow-up questions to differentiate respondents who disagree with the legitimacy of surveys as a forum for social value articulation; those who protest to the institutional or technical setting of the valuation scenario; and respondents who accept valuation surveys, but have zero WTP e.g. due to non-use.

ISSUE	RECOMMENDATION
Choosing the appropriate **mapping scale and resolution**	Map scale is needed in order to represent substitute water bodies, while resolution is required for respondents to identify the location of their household relative to water bodies. Sufficient map resolution combined with explicit questions about household location in the maps used encourage respondents to consider the importance of distance in valuation questions. Pilot tests at different distances will help to optimise the scale of the study area that captures a significant proportion of non-users and distance decay of WTP to zero.
Symbols and pictograms, such as quality ladders, may be interpreted differently by respondents and researchers	Respondent understanding of pictograms should be pretested. It should be evaluated in the main survey also using e.g. simple ranking exercises of attribute levels, as a matter of external validity testing.
Status quo water quality, habitat condition and water accessibility often vary greatly. Making a case study as relevant as possible for a specific water body, reduces the transferability of valuation estimates by making them less generic. Is it possible to select **'generic' study sites?**	A study should be adapted to local conditions to the extent that the improvements are consequential for local water users, rather than abstract or generic. Water bodies and their population of users will always be heterogeneous; heterogeneity can be controlled for across some theoretically expected (generic) population characteristics (distance/travel cost, substitute sites, income). Use choice experiments with attribute ranges should cover plausible variation within sites (but be aware that attribute ranges may still not cover variation in conditions between sites).
Programmes of measures defined in valuation scenarios may have distributional implications and perceived fairness, even where distributional issues are not focused on. Particularly in the valuation scenario for water scarcity issues, above some minimum water allocation fairness between users in a catchment may be as important as absolute scarcity for individual WTP	Pre-test respondents interpretations of the 'fairness' of proposed policy solutions, particularly households water and co-financing shares versus neighbours, adjacent communities etc.
Water bodies are part of a landscape and often linked to public commons. In addition to private use and non-use values they may invoke social **altruistic values**. In some institutional and cultural contexts individual-focused, survey-based WTP studies may capture only part of the value spectrum for water bodies.	Combine several different non-market valuation formats in the same study, and triangulate with other revealed and stated social preference methods focusing on social value elicitation (e.g. voting patterns on environmental referenda, citizens juries)

Table 14.2: Sampling method and strategy

ISSUE	RECOMMENDATION
Trade-off between study objectives: **policy welfare analysis versus methodology testing**	If policy analysis is the aim, focus attention on obtaining a representative random sample of the population (residential surveys; in-person, mail, web). If the focus is methodology testing a targeted/quota based sampling is more cost-effective (intercept surveys; in-person)
Internet versus postal and in-person **survey formats** Ensuring representation across water body substitutes and across river basin	Internet panels are pre-recruited and offer little flexibility in the geographical sample design. Population weighted sampling within administrative areas that approximately cover the watershed, ensure sampling in both rural and urban areas.
Internet versus postal and in-person survey survey materials **- maps**	Internet surveys limit the map scale or resolution more than postal surveys and in-person surveys, making it comparatively harder to study spatially heterogeneous goods and distance decay of WTP
Spatially distributed goods requires **spatially distributed** sampling	Due to 'lumpy' population distribution around urban areas, disproportionate sampling may be needed in rural areas and between substitute water bodies in order to observe significant distance decay. In-person surveys offer more control of cluster sampling strategies.
Ensuring **representation** of age, sex, income levels	Quota based sampling (internet-panel or intercept) can ensure socio-demographic representation.
Timing of survey implementation	Implement the survey immediately following a water use season in order to maximise recall rates, while describing average conditions across a full season
Increase respondent **convenience**	Combine internet with postal surveys
Minimising data **recording and entry errors**	Computer-based in-person or internet surveys are an advantage

Table 14.3: Analysis

ISSUE	RECOMMENDATION
Choice of experimental design software	A number of software packages are available. Sawtooth is a commercial package offering experimental design and simulation of sample efficiency. A drawback is that utility functions cannot be defined (e.g. including interaction effects between attributes and respondent variables). NLOGIT-LIMDEP, STATA, LATENT GOLD are commercial software packages that allow specification of utility functions. BIOGEME is a freely downloadable package.
Advanced analysis Choice of software for discrete choice models	STATA, NLOGIT, BIOGEME and LATENT GOLD all allow random parameters mixed logit models. LATENT GOLD is specialized on models to identify segments of respondents. This might be an advantage if valuation survey results are to be used to devise policies differentiated across different groups (e.g. differential water pricing)
Multiple approaches to **coding** attributes levels in choice experiment (effects, dummy or continuous coding?)	Use effects coding if you believe utility is non-linear (e.g. in evaluating suitability thresholds for water quality). Dummy coding may be easier to interpret than effects coding, and convenient if there is no no status quo option. Use effects coding to avoid confounding status quo mean with grand mean.Be careful with the economic interpretation of implicit prices derived from an effects coded model: implicit prices are not welfare estimates.
Demonstrating **validity** to colleagues is not the same as demonstrating validity to policy-makers	The following study results should be of interest to both scientists and policy-makers, but the presentational form should be differentiated technical/non-technical: Representativity: Response rate; Protest rates; Use versus non-use population; Importance of a priori knowledge and concern External validity: Convergence of CE and CV estimates and with prior valuation studies Internal validity: Scope sensitivity; Distance decay; Sensitivity to scale / substitutes; Lack of ordering effects; Sensitivity to time lag between payment and improvement; Interaction effects with status quo; Income effect and with attributes
The analysis shows lacking significance of **distance decay** of WTP (but this is due to small sample size)	Estimate distance decay indirectly by identifying geographical distribution of users/residents & non-users/non-residents and their respective WTP

Table 14.4: Reporting

ISSUE	RECOMMENDATION
Tables and graphs are too abstract for policy-makers or do not capture the **distributional and political-administrative implications** of WTP estimates.	Use maps to convey geographical coverage and sampling intensity of the study. Generate a WTP ”heat map” to show the spatial extent of the “economic concern” for lake quality improvements across administrative boundaries Map mean WTP and simulated voting by municipality for hypothetical scenarios
Policy-makers do not trust the **robustness** of the policy conclusion based on WTP results.	Illustrate WTP sensitivity at a site to changes in characteristics of substitutes. Use choice experiments to generate WTP for different ”policy packages” other than achieving GES;
Relative **reliability** of WTP	Include confidence bounds of WTP estimates versus engineering costs
WTP estimates from CVM or CE meth are not **decision-relevant** on their own (except perhaps for water pricing design) Decision-support role of WTP	Compare aggregate WTP to costs of measures in cost-benefit analysis of derogations under the WDF. Simulate voting scenarios on particular policy scenarios (proportion “yes”votes for given combinations of price levels and ecological status achievement in water bodies).
WTP results are not accepted as valid or sufficiently reliable for **policy analysis** under WFD.	As input to authorities’ **communication strategy under WFD,** provide policy-makers with qualitative information on protest rates to programmes of measures proposed in the valuation scenarios. Provide a priori knowledge of water body issues. **Opinion polling**: WTP can be interpreted as an indicator of attitudes of social concern. Valuation surveys may still be useful as opinion polls, despite WTP estimates themselves not being employed in policy-analysis.

14.3 *Using non-market values for water services in policy and decision-making*

In order to arrive at useful, authorized economic non-market values, they have to be accepted by scientists, policy and decision-makers and stakeholders involved, i.e. those affected by public policy or decision-making and bearing the costs or benefits. An important first step in this authorization process is the institutional and political embedding of such values in official public policy documents. Here we present two examples where the authorization process is mainly considered from the point of view of those who participated in the social surveys, i.e. the public at large, as opposed to policy maker authorization, by explicitly assessing the validity, reliability and acceptability of the obtained values as a separate step in the survey design and implementation. We start with an example from the Netherlands, followed by an example from Norway.

Example from the Netherlands

The use and acceptability of large-scale social surveys in the domain of water, including questions related to public willingness to pay (WTP), has increased significantly in the Netherlands, for example, over the past 5-10 years. A number of large-scale stated preference, in particular contingent valuation (CV) surveys, have been carried out since 2000, which aimed to inform public policy and decision-making at national level related to the revision of the European Bathing Water Quality Directive BWQD, the implementation of the European Water Framework Directive (WFD), and the European Marine Strategy Framework Directive. The emphasis in the WFD on public participation had some effect on the use of social survey methods to elicit public opinions and perceptions towards socially acceptable levels of water quality. These studies were all commissioned by the responsible water authority. Although not sufficient, this in itself is an essential step towards their authorized use.

Each of the studies was scrutinized for their validity and reliability in a number of ways. Internal consistency and validity was thoroughly tested through conventional statistical analysis, confirming *a priori* expectations regarding the direction of influence of statistically significant explanatory factors of stated WTP such as bid price and household income level. The explanatory power of the estimated models varied between 16 and 62 percent. Consistency and validity was furthermore carefully checked by analyzing protest bids and respondent understanding of the information provided in the survey and the WTP question. Examples are summarized in Table 14.5.

Protest bidders are respondents who refuse to pay for different provision levels of the public good in question because they disagree with the proposed payment structure for other than *a priori* expected economic reasons. A high protest rate questions the validity of the valuation design (Arrow et al., 1993). Criteria to determine whether or not a respondent is a legitimate zero bidder to a WTP question or a protest bidder often are arbitrary, if reported at all in CV studies. No strict guidelines exist regarding acceptable protest rates. The notion of acceptability is highly subjective in this context. However, in order to help policy and decision-makers to judge the validity of CV findings, a protest rate less than ten percent should be aimed for in order to produce acceptable (valid and reliable) results. Protest rates between 10 and 25 percent may still be considered acceptable, but should be interpreted carefully, while a protest rate higher than 25 percent is believed to invalidate a stated preference study (see Box H). Clearly, other subjective or arbitrary threshold values may also apply depending upon the criteria used by the researcher or policy maker to identify protest bidders.

Table 14.5: Summary assessment self-reported validity indicators stated preference studies related to water

CV study	Year	Sample size	Survey method	Elicitation format	Protest response	Difficult answering WTP question?[a]	Information supply sufficient?[b]	Clear what exactly you are being asked to pay for?[c]
1. Ecological rehabilitation regional Lake district	2002	670	Face- to-face	OE	14.3%	70% yes	70% yes	Not asked
2. Bathing water quality improvements revised BWQD	2002	5000	Mail	DC	8.8%	20% yes	93% yes	Not asked
3. Good water status EU WFD	2003	5000	Mail	DC	16.5%	20% yes	92% yes	94% yes
4. Biodiversity conservation and contaminated sediment clean-up	2004	5000	Mail	DC	13.5%	33% yes	78% yes	96% yes

<u>Explanatory notes</u>:
[a] Share of the sample population who said they found it (very) difficult to answer the WTP question, measured on a 5-point scale ranging from '*not difficult at all*' to '*very difficult*'.
[b] Share of the sample population who said they felt that the information provided in the questionnaire to answer the WTP question is (more than) sufficient, measured on a 4-point scale ranging from '*more than sufficient*' to '*completely insufficient*'.
[c] Share of the sample population who said they know exactly what they are being asked to pay for, measured on a 4-point scale ranging from '*completely clear*' to '*completely unclear*'.

Box H: Guidelines to assess the reliability of stated preference studies based upon protest response rates

Protest response rate	Conclusion
≤10%	(Almost) no problems with the proposed market construct → **results are reliable**
10-25%	Some problems with the proposed market construct → **results are reliable, but need to be interpreted carefully**
>25%	Serious problems with the proposed market construct → **results are not very reliable and their use is questionable**

Important: Calculated protest rates should always be presented together with the criteria used to identify WTP protest responses as these obviously affect the calculated protest rate. Criteria are not always straightforward. Important means to reduce protest responses include thorough pre-testing, including the use of alternative payment vehicles.

Examples of *a priori* expected economic reasons include low or no preference (e.g. respondents who attach no value to the good in question or who have no problem with the current situation), income constraints (respondents who say they have insufficient income to pay) or substitution effects (respondents prefer to spend money on other things than the good in question). The average protest rate of 13 percent found in the four Dutch water studies reported and discussed here is considered too low to invalidate the results, but do indicate that the results have to be interpreted with the necessary care as approximately one in every eighth respondent seems to object against the proposed market construct and valuation scenario. The most important reasons found in all four studies for protest are '*the polluter should pay*' and '*I already pay enough taxes*'. These are typically reasons which do not say much about the real value attached to the good in question (positive, zero or negative), but relate to the proposed market construct. In some cases also respondent distrust that the money would actually be spent on the good in question played a role. However, the two first reasons were predominant in each study, even though it was emphasized in some studies that those responsible for the problem would pay based on the Polluter Pays Principle and respondents were offered the opportunity to pay in another way (e.g. through water prices or a one-time off donation).

Other indicators of the studies' validity include questions in the questionnaire about the ease with which respondents answer the WTP question, the amount and quality of the information provided in the questionnaire to answer the WTP question and in the last two studies also how clear respondents are about what exactly they are being asked to pay for (Table 14.5). When looking at the difficulty respondents experienced in answering the WTP question, the open ended question in the first study clearly was more problematic than the dichotomous choice questions in the other studies. In all studies except the bathing water quality study, the degree of difficulty to answer the WTP question appears to be a significant determinant of stated WTP in the multiple regression analysis. The more difficulty respondents experience answering the WTP question, the lower, on average, stated WTP. Finally, a majority of respondents feel that the information supplied suffices to answer the WTP question, while almost all respondents in the last two studies are clear what exactly they are being asked to pay for. These findings are also important additional indicators of the reliability of the CV results.

Example from Norway

The acceptance of non-market valuation surveys as legitimate information for making collective decision about pollution mitigation measures may be contested. A large enough proportion of such respondents may make decision-makers uncertain about how to use non-market valuation estimates for decision-making. In the Norwegian AQUAMONEY case study, following the willingness to pay questions and the choice experiment questions respondents, respondents were asked to rate their agreement with a series of statements regarding the valuation tasks in the survey. Figure 14.1a shows respondents' level of (dis)agreement regarding whether willingness to pay responses are a measures of the value they place on water quality. A little more than a third of the sample partly or strongly agree that their willingness to pay responses express values they hold regarding water quality.

Figure 14.1b similarly asks whether the choice questions and trade-offs between water and sewage fees and water quality are a correct way to express values respondents hold regarding water quality. A similar proportion of the sample as above agrees or strongly agrees to these statements.

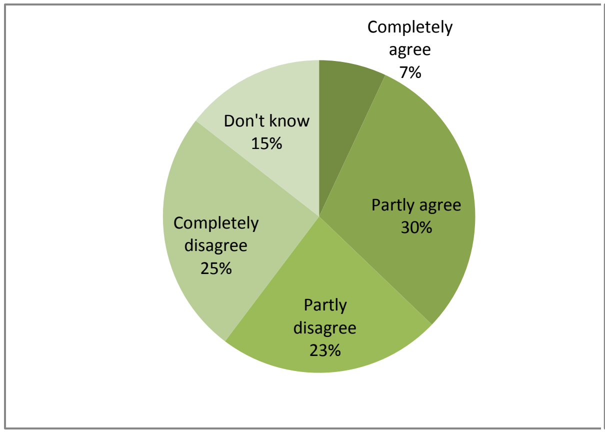
From these two questions we concluded that respondents see no difference in the content validity of contingent valuation versus choice experiments as methods for valuing water quality. It is notable that half the sample disagree that these valuation methods capture the values they hold regarding water quality. In defence of the methods we can argue that the water quality ladder was only meant to capture use values, and so respondents with non-use values (altruistic or existence values) could disagree with these statements (while accepting they are valid for use values only).

In Figure 14.1c we asked a wider question of whether willingness to pay responses should be used by authorities as a basis for making decisions about water quality measures. A majority of respondents disagree with this statement. We would need to conduct follow-up interviews to discover whether this was synonymous with respondents not wishing authorities to use valuation methods for decision-making at all, or whether valuation results could be used for specific policy analysis purposes if they are explained (e.g. disproportionate cost analysis under the Water Framework Directive (WFD). The potential legitimacy problem illustrated above can be compared to the 'democratic deficit' problems of e.g. municipal governments elected with very low voter turnout[18].

[18] An example of in-depth ex post goup discussions about the validity, reliability and legitimacy of using non-market valuation outcomes in actual policy and decision-making can be found in Brouwer et al. (1999).

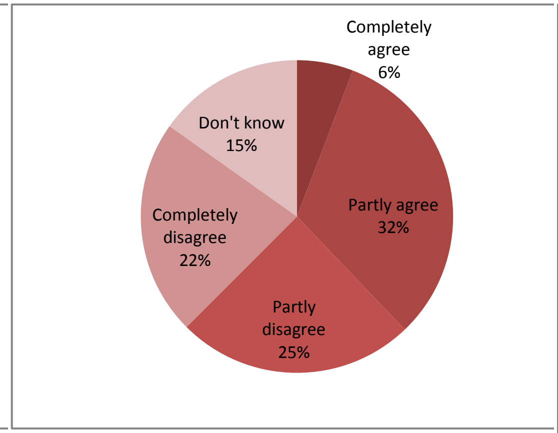
Figure 14.1: Respondent opinion about the WTP questions

Figure 14.1a



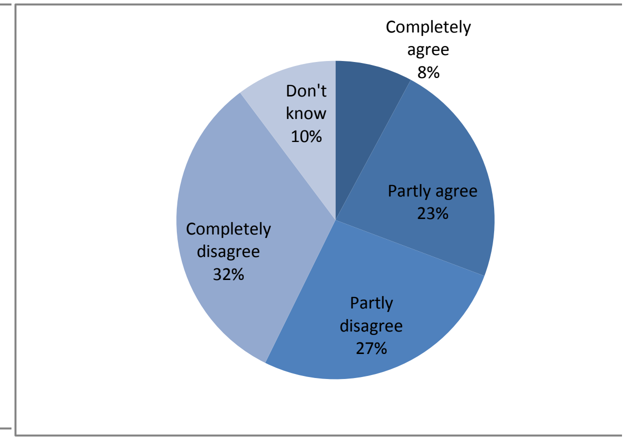
"Saying how high an increase in my water and sewage fee I would be willing to pay for a water quality improvement is a correct way to express the values I hold regarding water quality"

Figure 14.1b



"Choosing between different scenarios of water and sewage fee increases and water quality is a correct way to express the values I hold regarding water quality"

Figure 14.1c



"It is reasonable that decisions about measures to improve water quality are made based on how much people are willing to pay"

14.4 *References*

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Annex I: List of manuals and guidelines used in water valuation

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Annex II: Aquatic ecosystem functions and applications of valuation techniques

Hydrological functions

Flood water control: the short or long term detention and storage of water from over bank flooding and/or slope runoff.

- *Hedonic pricing.* Hedonic pricing can be used to analyse the price differential for properties that are at risk from flooding. It entails analysis of all variables that could affect price, such as location, size, aspect, and age of property. Use of hedonic pricing requires existence of a property market and existence of known and distinct risks of flooding. It is a complicated procedure that is subject to various complications. Impacts of flooding on property prices can be countered by defensive expenditures to reduce flood damage (Holway and Burby, 1990). A further complication is that perceived risk of flooding and the resultant impact on house prices can diminish as memories of previous flooding episodes fade (Tunstall, Tapsell & Fordham, 1994). Also, house prices may reflect flood hazards only where flooding has occurred relatively recently, regardless of the expected frequency of flooding (Tobin & Newton, 1986).

- *Contingent valuation.* Flood water control can be valued by asking the affected population what they would by hypothetically willing to pay to either avoid flooding (of some area of interest) or to avoid an increase in the frequency of flood episodes. Given the analytical and resource demands of contingent valuation survey, this is best limited to valuation of the impacts of flooding that are not non-marketed, such as impacts on unique ecosystems.

- *Damage costs avoided.* The costs that would be incurred if flood control provision (e.g. the flood protection provided by a wetland) was not present are given by the damage costs. These can be divided into direct costs, indirect costs and intangible costs.

The direct costs of flooding are incurred by physical contact with the floodwaters. Costs of damage to the built environment are determined by the type of building (e.g. residential, commercial, industrial) and factors such as the design, function, density and age of the buildings. Cost estimates can be obtained from relevant publications (e.g. the 'FLAIR blue book' used in the UK (N'Jai et al, 1990), government agencies, or site specific surveys conducted by government agencies or insurers. In determining the costs of damage to movable assets, account needs to be taken of avertive action. For example, Tunstall et al (1994) found in a study of flooding in Maidenhead in the UK that the reduction in damages due to avertive action was 'substantial'. Higgs (1992) allows for a 5-10% reduction in damage costs due to items being moved away in advance from flood prone areas.

Flooding also imposes costs on productive activities in the non-built environment. Damage to natural ecosystems (wetlands, woodlands and meadows) may be minor and temporary. However, the costs can be substantial for intensive agriculture. Losses in returns to agricultural production are determined by the depth, extent and duration of flooding, the effluent and silt content of the flood waters, types of crop, expected yields and price. Silt exacerbates the volume of flooding and is itself a cause of damage; Clark (1985) estimated that silt accounted for 20% and 7 % of urban and rural flood damage respectively for a study in the US. Returns from agriculture, as the opportunity cost of wetland preservation, are calculated by Turner et al (1983) based on detailed analysis of output, fixed and variable costs and transfer payments (agricultural subsidies). Estimates of standard losses in agricultural gross margins due to flooding may also be available from official publications (e.g. the Farm Management Pocketbook for the UK). Long term impacts on agricultural production through continued exposure to inundation are reflected in the value of the land. Flooding affects the land use categorisation of land and this is reflected in the market price (Boddington, 1993); average price data for land use categories is often available from official publications.

Flooding also results in indirect and intangible costs. Indirect costs are caused by disruption to physical and economic linkages in the economy. They include costs of implementing immediate emergency measures; reduced production, and the knock-on effects of this on production elsewhere; impacts on transport; and increases in living expenses. Intangible costs by definition cannot be readily quantified. Examples include psychological effects (stress caused by flooding and worry about future events) and poor health caused by flooding. Some costs formerly described as intangibles are now being quantified such as the effects of disruption and evacuation (Green & Penning-Rowsell, 1986; 1989). Intangible costs could be more significant that the direct damages of a flood episode (Green & Penning-Rowsell, 1989). It is best to acknowledge that such costs are expected but cannot be valued and that the total cost of damage (and hence the value of the wetland flood protection function) is underestimated as a result.

- *Defensive expenditures.* Defensive expenditures provide only a minimum estimate of the benefits of flood water control as they may omit costs of flooding against which defensive actions are not taken. Furthermore, defensive expenditures tend to be low relative to potential damages as individuals underestimate the likelihood of flooding and over-estimate their ability to cope with its effects (Tunstall et al, 1994). Defensive expenditures include relocation of assets (buildings, nature reserves and livestock (Boddington, 1993), rewiring of electrical points above expected flood levels and raising of houses on stilts or piles (Tunstall, Tapsell & Fordham, 1994). Relocation may not be to a site that is a direct substitute. Costs of relocation therefore need to be attributed accordingly between the various benefits, and any disadvantages also taken into account.

- *Replacement Cost.* The replacement cost of flood control can be determined, for example, through the use of shadow projects. In the case of the flood water control function of a wetland, a shadow project could entail creation or restoration of another wetland that would performed the same function within a given catchment. This would also replace other functions of the wetland and would, therefore, be particularly appropriate in a situation where total loss of a wetland is threatened. Locally relevant costings for wetland creation or restoration are likely to be sparse (though mitigation banking has led to considerable creation and restoration of wetlands in the U.S.). There is uncertainty associated with ‘engineered’ ecosystems and the functions that they can perform. This is more pronounced if the location of the shadow project is distant from the original site and the benefits are consequently derived by a different population. For shadow projects that entail a change in land use for a site (e.g. taking land out of agricultural production) the opportunity costs of this must also be included in the analysis.

Groundwater recharge: the recharge of groundwater by infiltration and percolation of detained floodwater into a significant aquifer.

The groundwater recharge function is only of value if the recharged groundwater is of some benefit to society. The benefits may be direct, such as abstraction of the water for irrigation or domestic use, or indirect, such as the maintenance of water table levels. In addition to these use values of recharging groundwater, there may also be non-use values of maintaining groundwater supplies. Non-use values can be attributed to the maintenance of groundwater supplies for subsequent generations, but only if use of the reserves is anticipated.

The potential extractive and in-situ uses of groundwater acting as a water reserve (stock) were shown in table 2.1 in Section 2. As far as the extractive uses are concerned, the techniques involved in assessing the economic value of groundwater recharge are much the same as those outlined for the 'groundwater discharge/surface water generation' function below. Studies that have considered values for groundwater supply are outlined in that subsection as they illustrate techniques that may also be useful for assessing the value of surface water. They include hedonic pricing based on variations in availability of groundwater irrigation supplies; costs of establishing substitute well sites; and contingent valuation of willingness to pay for alternative piped water supplies. A number of studies have assessed values associated with maintaining the quality of groundwater (which may be relevant to the in-situ uses of the recharge function) and these are considered under the 'nutrient retention' function. Two other in-situ use values arising from groundwater recharge include prevention of land subsidence and salt water intrusion. We consider the valuation of each in turn.

<u>Prevention of land subsidence:</u>

- *Hedonic property pricing.*- Hedonic pricing is used to analyses a price differential in property that is attributable solely to the risk of subsidence. If identical sets of housing exhibit variation in prices, and the only non-constant attribute is the risk of subsidence, then price differences can be related to the buyers' willingness to pay to avoid subsidence. However, it is necessary to assess all relevant variables that could affect price (e.g. location, size, aspect, age etc), and to isolate the effect of subsidence from these.

- *Damage costs avoided.* Predominant land uses are identified and the various costs of a potential subsidence assessed. Estimation of the damage costs that are avoided due to ground water recharge provides an upper bound estimate of the value of this function as it does not technically value society's willingness to pay to avoid the subsidence. Instead, it values the full extent of costs expected to result from subsidence, which could exceed the cost of alternative measures that might be used to negate the economic impacts. However, it may not be feasible to estimate all the costs involved, particularly the intangible costs (a full appraisal of this valuation approach was given above under the flood water control function).

<u>Salt water intrusion</u>

- *Residual imputation and variants.* Intrusion of salt water can occur due to falling groundwater in levels in areas in proximity to the coast. Salt water intrusion can deem groundwater unusable for irrigation, thereby impinging directly on agricultural production. The change in net returns that this would cause can be used to assess the value of maintaining ground water levels to prevent the intrusion of salt water.

Groundwater discharge/surface water generation: the discharge of groundwater into the surface water system.

Groundwater discharge and surface water generation can be considered as identical functions for the purposes of valuation. Whether water originates from direct precipitation, groundwater discharge or another source does not influence the value attributable to surface water generation. The surface water that is generated contributes to the stocks and flows of surface water, which support a variety of in-situ and extractive uses as well as non-use values.

Extractive uses of surface water include use of water for irrigation and domestic purposes. In-situ uses of surface water are more varied and can include maintenance of habitats and provision of aesthetic and recreational value. A number of the in-situ uses of surface water are also considered within other functions. For instance, the reliance of characteristic wetland ecology on surface water and anaerobic conditions resulting from inundation (and the subsequent capacity to retain excess nutrients) are considered under the ‘ecological’ and ‘nutrient retention’ functions, respectively. Downstream habitat and biodiversity maintenance are considered below only in so far as they might contribute to recreational and amenity value. Other benefits associated with maintaining biodiversity could be significant (for instance non-use values) and valuation methods for these are outlined under the subsection on ‘ecological’ functions.

The techniques outlined for valuing extractive and in-situ uses of surface water are also applicable to the extractive uses listed under the groundwater recharge function.

- *Market based transactions.*– Surface water abstraction for use in irrigation can be valued using market prices observed in rentals and sales of water rights. In order for traded water rights (either for use of water over a specified period or for a permanent right to water use) to reflect the economic value of water use, allocation and enforcement of property rights is required. If necessary, prices should be adjusted to reflect long term considerations (i.e. social values). In practice rental rates may be affected by factors other than the marginal value of water. Although observations of prices on markets for perpetual water rights are more appropriate for long run planning contexts, some degree of care is required in converting this capitalised asset value into the annual values conventionally used in planning and policy analysis (Young, 1996). Furthermore, the use of water right prices, in circumstances where crop prices are supported by agricultural subsidies, will lead to overestimation of the social value of irrigation water.

- *Residual imputation and variants.* This is one of the techniques that is most widely used to value irrigation water. It was employed by Ruttan (1965) in an early study that demonstrated the difference between the value of irrigated and non-irrigated agricultural production. Some degree of care has to be taken with its use to ensure that statistical problems, such as multicollinearity between variables, do not bias the analysis. Linear programming models have been applied to farm budget data to derive shadow values on irrigation water (e.g. Colby, 1989). Here, crop type is the most important determinant of the marginal value of irrigation water. The presence of uncertainty makes valuation of agricultural water use difficult, due to uncertainty in the need for irrigation (arising from climatic variation, for example) and in water supplies. Farmers’ attitudes towards risk must therefore be considered when undertaking studies. Furthermore, market distortions and externalities also need to be taken into account. The annex contains a case study example for a multiple use irrigation system in Sri Lanka.

- *Derived Demand Functions.* This technique has been used to estimate households’ valuation of domestic water supplies employing relatively easily acquired price and quantity data, for example in Young and Gray (1972) and Gibbons (1986). Though these studies address households’ valuations of a given quantity of water, they did not address complications created by variations in water quality or service reliability. These issues have been considered in studies using contingent valuation and avoided cost approaches.

- *Hedonic pricing.* Hedonic pricing can, in principle, be used to derive the value of maintenance of river flows by surface water generation (Loomis, 1987). Individuals or businesses (including farms) might pay a premium for property located close to a river. It may be difficult to distinguish from use of water in the river from other locational factors, such as benefits associated with aesthetics, recreation or transportation that result from proximity to the river. It is also difficult to determine the contribution made by the discharge of ground water to maintenance of water levels in the river. Few studies have used hedonic pricing to value surface water generation, presumably due to these complications and demands of the technique. One of the few examples decomposes the value of agricultural land as a function of its attributes including the use of irrigation water (Faux and Perry, 1999). An example is also given in the annex of a detailed case study of water supply in the Philippines.

- *Replacement cost/avoided cost.* Avoided cost has been used to value hydroelectric power generation (see Gibbons, 1986). The cost that would be incurred if the capacity to generate power was provided from an alternative source is used to impute the value of the hydroelectric power generated. However the method is problematic as it ignores the price elasticity of demand for electric energy. The approach can also be applied to valuation of surface water generation where water is abstracted to provide drinking water. The expense of finding an alternative water supply though considerable is, however, likely to be exceeded by benefits of continued use of the existing source. The technique may be particularly suited to this application though if there is difficulty in valuing the health implications of restrictions in water supply.

<u>Biogeochemical functions</u>

Nutrient retention and export: the storage and removal of excess nutrients (nitrogen and/or phosphorus) from water, via biological, biochemical and geochemical, physical and land management processes.

The main impact of storage of nitrogen and phosphorus is improved water quality; thus this function is discussed here with respect to water quality. Improvements in water quality have a number of benefits. A few illustrative examples of valuation of the benefits of improved water quality are outlined below. Impacts on recreation can be valued using the travel cost method. The benefits for drinking water supplies can be considered, for instance, via defensive expenditures. Indeed, nutrient retention benefits can be considered generally in terms of the costs of providing substitute treatment facilities. Potential increases in the costs of industrial production processes are not considered here. The residual imputation methodology, discussed with reference to the 'surface water generation' function, could also be employed to value the benefits of improved water quality as an input to production processes.

- *Contingent valuation.* CV based research has been widely used to consider water quality. Jordan and Elnagheeb (1993) used the approach to assess households' valuations of improvements in drinking water supplies (due to reductions in nitrate levels). One of the most challenging aspects of using this approach is the manner in which water quality information is conveyed to survey respondents, and specifically whether objective or subjective measures of water quality are used. Poe (1998) argues that objective measures are preferable because people do not have reliable and well informed reference points with regards water quality. Conjoint analysis and contingent ranking has also been used to value water quality improvements. For example, Georgiou et al (2000) used contingent ranking to value urban river water quality improvements. The annex contains a case study that examines water quality in the Philippines

- *Travel cost.* The travel cost method has been used to assess the value of improved water quality at recreational sites. A complex form of travel cost analysis, which includes measures of water quality as independent variables, is applied to sites which vary in water quality (but are similar in other attributes) or to one site for which water quality changes over time. This is an extremely involved procedure, which measures only the recreational benefits associated with improved water quality downstream. There are a number of difficulties with such analysis. In particular, for a multi-site study, the influence of water quality between sites needs to be isolated from other varying attributes that might affect recreation demand. In the case of a single site temporal study, changes in water quality need to be isolated from other attributes that might change over time. Smith and Kaoru (1990) undertook a meta analysis of travel cost studies that relate to water-based recreational values. They found that the following five features consistently had an influence on results: type of recreational site, the definition of a site's usage and quality, measurement of the opportunity cost of time, the description of substitutes, and specification of the demand model.

- *Hedonic price method.* The price of properties in close proximity to water bodies can be affected by the quality of the water and therefore by nutrient retention. The value of the nutrient retention function is derived from (a) property values that are attributable to water quality and (b) the role of the function in maintaining the water quality. This entails analysis of prices for otherwise similar properties that are located close to polluted and unpolluted water bodies. The data demands are, however, considerable. A rough approximation of value can be derived directly from a summation of adjustments in property prices, which could be based on assessments of experts, such as estate agents, rather than actual observed price differentials in the property market.

- *Defensive expenditures/avoided cost.* The value of improved water quality can be estimated based on the expenditures undertaken by people to avoid consumption of poor quality water. The sum of defensive expenditures on marketed goods such as water purification equipment represents the lower boundary on society's willingness to pay for improved water quality. This accounts only for changes in behaviour made by consumers in response to poor water quality. It does not take into account consumers who do not undertake defensive actions but would nonetheless prefer improved water quality. Such individuals may be inhibited from acting by inconvenience associated with the defensive activities, or lack of information about pollutant levels and possible adverse effects. Abdalla et al (1992) use this valuation approach to determine the time and money that households expend to avoid risk arising from groundwater contamination. Their approach assumes that households undertake a two-step decision-making process in which they first decide whether to undertake any avertive action, and then decide on the intensity of those actions. Abdalla (1994) also provide a survey of the literature on averting cost methods.

- *Replacement cost.* The retention of nutrients can be valued using the replacement cost in terms of the cost of substitutes. Substitute activities include reduction of nitrate and phosphate pollution at source by limiting applications of agricultural fertilisers or the installation of water treatment facilities. The replacement cost is particularly useful for situations where the benefits of reduced nutrient loading are difficult to estimate. Examples include estimation of the benefits of avoiding deleterious health effects or the benefits of maintaining water quality and ecosystems for future generations.

Sediment Retention: the net retention of sediment carried in suspension by surface water including runoff from the contributory area and over bank flooding.

Retention of sediment reduces the sediment load in water downstream and thereby improves water quality. The value of this may be most readily estimated in terms of the additional costs that would be incurred by industrial and municipal users of water through the necessity for water treatment in the absence of sediment retention. Higher water quality may also lead to increased opportunities downstream e.g. for recreation and commercial fisheries and may have biological impacts on survival of habitats and species. Habitats and biodiversity are considered here only in so far as they might contribute to recreational and amenity value. Techniques for valuing the benefits of improved water quality (or, conversely, the costs of poor water quality), were covered under the nutrient retention function.

Additional benefits of reduced sediment loads result from decreased levels of sediment in water downstream. These include mitigation of damages to water conveyance facilities. Such damages can occur through deposition of sediment in rivers, drainage ditches and irrigation canals, which can lead to adverse effects on navigation and water storage capacity and can increase flooding. Some of the techniques that may be used to value those benefits that have not already been considered under other functions are discussed below.

- *Avoided cost/damage costs.* The benefits of maintaining navigation can be estimated in terms of the avoided costs of alternative transport. This approach does not usually account for the differences in speed between alternative modes of transport. Alternatively, the benefits of maintaining navigation can be valued as the damage costs avoided in terms of reduced accidents and groundings. However, values are likely to be low, especially if the costs of infrastructure have already been accounted for. The benefits of mitigating damages to water conveyance facilities, such as deposition in drainage ditches and irrigation canals, is peculiar to the sediment retention function. Estimating the damage costs avoided, in terms of the costs of reversing possible adverse impacts is the most appropriate valuation technique to use.

- *Residual imputation and variants.* The presence of fine silt particles in water used for irrigation can lead to a loss in productivity, as they can seal the surface of the soil, making it impermeable. However, the addition of sediment can increase soil fertility and thereby improve productivity. As sediment impinges directly on agricultural production, for which market prices exist, then changes in marketed outputs can be used to assess the value of sediment retention.

<u>Ecological functions</u>

Ecosystem maintenance: the provision of habitat for animals and plants through the interaction of physical, chemical and biological processes.

Economic value of ecological function is generally only derived through contact with or concern for species or habitats that are components of an ecosystem. The economic value of ecological functions consequently focuses on aspects of biodiversity (both quantity and variety of organisms) With respect to biodiversity maintenance the valuation techniques of relevance include:

- *Contingent valuation.* Contingent valuation can be particularly useful for assessing the value of biodiversity maintenance, indicating willingness to pay for conservation of biodiversity.

Contingent valuation is the only technique currently regarded as suitable for estimating non-use values associated with the maintenance of species diversity and population sizes. By definition, these values are not reliant on individuals visiting the site (so are not associated with measurable changes in behaviour). Brouwer et al (1999) provide a meta-analysis which attributes values to various ecological functions estimated from a large number of contingent valuation studies.

- *Hedonic Pricing.* Differences in property prices that can be attributed to aesthetic and amenity benefits of proximity to a wetland can provide a value for maintenance of biodiversity on the wetland site. This requires analysis of prices for otherwise similar properties that are located close to and distant from wetlands with a diversity of species.

- *Replacement costs.* The replacement cost of the biodiversity maintenance function is based on the costs of creation or renovation of an alternative. To provide a replacement, the alternative is required to provide similar habitats to the original site. Indeed, a possible management option would entail relocation of species to an alternative site. To have corresponding value, the alternative site is required to provide the same benefits. These are influenced by the location of the alternative site: the proximity to population centres, ease of access and availability of substitutes sites. There is also the question of ‘authenticity’: the original naturally occurring site may be preferred to an exact replica, thereby affecting amenity and non-use values. Valuation using the replacement cost is most straightforward for sites that predominantly provide the single function of biodiversity maintenance. For sites that provide multiple functions, the costs of replacement are attributed between the respective functions. Opportunity costs of the conversion of the alternative site and any externalities are also taken into account.

In addition to biodiversity maintenance, anthropogenic export of this biodiversity which is also an important ecological functions. This has consumptive use value associated with commercial exploitation, subsistence provision, or recreational use. The value of commercial exploitation of fish, shrimp or timber harvesting can generally be assessed through analysis of market prices. Subsistence value can be harder to estimate because the products are not marketed, but market prices may exist for the products, alternative products or inputs to production that can be used as surrogates for prices. Consumptive recreation activities most often involve fishing and hunting, These can be assessed using the travel cost method or the contingent valuation method.

Annex III: Economic valuation techniques used to value water resources

Valuation Method	market-based transactions
Description and Basis of Approach	Direct market based transactions in water rights between buyers and sellers.
Applicability	This technique has mainly been used to estimate the demand for agricultural water use. Applicable to other areas where data is available, e.g. to value wetlands, one could look at what organizations such as nature conservation organisations have paid to purchase wetlands.
Procedure	
Data Requirements & Implementation issues	
Advantages (strengths)	Based on actual market transactions.
Disadvantages (weaknesses)	May require shadow pricing; sparcity of such market transactions and even when they do exist, scarcity of carefully collected data
Validity	High level of validity since based on actual market relationships
Value Concept Compatibility with TEV (incl. non-use)	Use value only; only lower bound estimate of TEV.
Value Measure	Depending on study: Marginal value based on price; not Max WTP; sometimes average value;
Reference	Studies of such transactions have been conducted in the south-western states of the U.S. (Saliba and Bush, 1987), and elsewhere in the world (Easter and Hearne, 1995).

Valuation Method	Production Functions and Derived Demand Functions
Description and Basis of Approach	Production and demand functions are sometimes fitted to production data with econometric techniques as a basis for estimating the value of water. A production function of the general form Y= f(materials, labour, natural resources, capital, water) can be fit to empirical data with econometric techniques. The approach calculates the marginal productivity of each input, including water. This then allows the marginal value of water to be calculated by multiplying the marginal product of water by the output price to yield the value marginal product. At an efficient level of production the marginal products must be equal across inputs. Derived Demand Functions: This makes use of a household's or firm's inverse demand function to estimate the user's willingness to pay for water. Based on observations of water use behaviour. The approach combines concepts from consumer theory with observed water usage and makes use of econometric procedures to derive a demand curve for water. Transactions concerning water are observed between water utility suppliers and individual water users, usually involving a 'take it or leave it' price schedule. Despite the usual monopolistic nature of supply, because the buyer can buy as much as desired at the price schedule it is possible to derive inferences on willingness to pay and demand, as long as sufficient observations are observed across variations in real price.
Applicability	At site values for agricultural and industrial production uses; site demands for municipal uses
Procedure	1. Collate data; this includes market data on price, output and demand, as well as data on factor inputs and prices (in the case of the CFA). A measure of the environmental input is also required. 2. Derivation of production function and/or cost function; here for example the relationship between the environmental input and output of the final market good is estimated. Production and cost functions may be estimated by econometric methods that include the environmental quality variable or by simulation approaches which model the behaviour of producers and responses to changes in environmental quality. 3. Estimate changes in consumer and producer surpluses. Here the effects of changes in production and costs are analysed in relation to final market supply and demand in order estimate changes in surplus.
Data Requirements & Implementation issues	Production function requires a considerable amount of data concerning the final goods market and factor inputs. It is also necessary that the production function and market structure be specified. Data are obtained for a cross section of farmers or manufactureres from controlled crop experiments performed by agronomists, from interview surveys emphasizing use of water and other inputs and production or from secondary data sources. Derived demand data are obtained preferably from observations on water use behaviour of individual households, though this can be costly so aggregate data from suppliers is often used. Statistical regression analysis is employed to estimate the parameters of the production function and demand equations.
Advantages (strengths)	Sometimes have readily available data on price of water, elasticity of demand for water and quantity if water consumed.
Disadvantages	Cost of data; wide variation in production technologies; Estimates lower bound of value??

(weaknesses)	Due to confidentiality reasons, data on cost structures and functions may be difficult to obtain. Generally, the effect of the environmental input on output is more observable, and typically more often analysed Estimates highly sensitive to functional form assumed for demand curve
Validity	Theoretically sound technique.
Value Concept Compatibility with TEV (incl. non-use)	Use value only
Value Measure	Marginal value, net average value and gross average value depending on purpose of study
Reference	Walker et al (2000)

Valuation Method	Optimisation models (incl. mathematical programming)
Description and Basis of Approach	*Optimisation models*. Optimisation models are used to provide mathematical solutions to problems that entail maximisation or minimisation of an economic objective subject to specified constraints to the economic objective. In generating the optimal solution to the problem, the models reveal the associated economic value of all inputs. Two types of optimisation model are discussed here: mathematical programming models and dynamic optimisation models. Mathematical programming models tend to be static one-period models. They are used to model economic problems in which the economic agent (consumer, central planner, or firm) seeks to optimise (maximise or minimise) a single objective function (e.g., surplus, costs, profit, or revenue) over a specific time period, whilst facing constraints that restrict choice to certain levels of inputs or outputs. The models can be used to determine marginal or non-marginal values for use of water as an input. Water enters mathematical programming models as an input constraint, such that its marginal value is found by relaxing the water constraint by adding a unit to the water available for production and calculating the difference between the optimal value before and after relaxing the constraint. This marginal value of water is also known as the "shadow value" of water. Non-marginal changes can be evaluated similarly, and also changes in the shadow value of water can be calculated for exogenous changes in output prices, input prices, or constraints. Dynamic optimisation models are used to indicate the optimal outcomes for separate periods in frameworks that involve multiple time periods. In a similar manner to mathematical programming models, they can be used to determine marginal and non-marginal values for water and the impacts of changes in other variables on the value of water.
Applicability	Models applied to one sector, such as agriculture to determine optimal mix of crops; to a watershed to determine optimal allocation of water among all users; Mathematical programming models are often used to determine the value of irrigation water and groundwater in situations where detailed data are available for a few representative agents. Dynamic optimisation models have been used to measure the value of water in allocation schemes, irrigation policies, and water quality projects.
Procedure	
Data Requirements & Implementation issues	
Advantages (strengths)	
Disadvantages (weaknesses)	
Validity	
Value Concept Compatibility with	Use value only

TEV (incl. non-use)	
Value Measure	Calculate shadow price or marginal values for all constraints including water; Optimisation models estimate marginal values based on ‘optimal’ allocation of water.
Reference	Bouhia (2001); Diao and Roe (2000)

Valuation Method	Residual Imputation and variants (Yield Comparison, Value added, Change in Net rents)
Description and Basis of Approach	The use of water in a production process can be determined using the residual imputation approach. This is a form of a budget analysis technique in which water is treated as one input into the production of a good. The approach seeks to find the maximum return attributable to the use of water by calculating the total returns to production and subtracting all non-water related expenses. The value of the product is allocated amongst the range of marketed inputs that go into its production. The residual value is assumed to equal the returns to water and represents the maximum amount the producer would be willing to pay for water and still cover input costs. If only variable input costs are subtracted then a short run measure of the value of water is derived; if the costs of all non-water inputs are subtracted (including a normal rate of return on capital) then a long run value is obtained. The approach is sometimes categorised as a farm crop budget technique in applications to agriculture. Variants of the residual imputation approach include: *Yield comparison approach.* In its application to irrigated agriculture, for example, this approach values irrigation water as the difference in per acre returns between irrigated and non-irrigated land, using observed farm budget data. It is assumed that the additional net returns obtained from the use of irrigation in the production process represents the maximum amount that the producer would be willing to pay for use of irrigation water. *Value added approach*. The residual not only includes the contribution of water but also the contribution of all primary resources included in the value of the final product. The concept of value added includes expenditures on several primary factors, including wages and salaries, depreciation, profits, rents to other natural resources, interest and indirect taxes. It is possible to use the marginal cost of water recirculation as a proxy for the value of industrial intake water. The value-added imputation of the value of a resource is simply the value added by the sector divided by the total physical quantity of water consumed by that sector. By attributing the total value added to water, the assumption is made that the shadow price of the other primary resources results in an almost certain overestimation of the true value of water. This approach is not used except in the case where it is not possible to derive economic value sing another method. *Changes in net rents approach.*
Applicability	Limited to uses of water in production process; especially irrigation, industry; It is most suitable for use in cases where the residual input contributes significantly to output.
Procedure	
Data Requirements & Implementation issues	The key assumptions here are that all other inputs are priced and that total value of output can be apportioned according to the marginal product of the inputs. Data derived from either extensive surveys of crop production and inputs, or from secondary data that is used to derive average crop yields and production costs. Secondary data may differ considerably from actual inputs and yields of the farming area being assessed. Assuming model specification is accurate, the prices for all inputs and products must be reviewed because some inputs, notably family labour may not be paid and the prices of others may be distorted due to taxes, subsidies, trade protection, etc. A great deal of

	judgement is thus required to determine whether non-water inputs require shadow prices, and how to estimate these shadow prices. Calculation of residual values requires considerable information and accuracy in allocating contributions among the range of resource inputs. Essential that all inputs are included and that they be priced at their marginal economic value
Advantages (strengths)	Conceptually straight forward and relatively easy to apply
Disadvantages (weaknesses)	Imposes rigorous conditions. Market distortions need to be accounted for. Difficult to implement when the input being considered has a small cost share (as is usually the case for water). A difficulty is that the residual return (after subtraction of the costs of all measured non-water inputs) is the return to water plus all unmeasured inputs, and hence will result in overstatement of the value of water. The approach is also extremely sensitive to small variations in assumptions concerning the nature of the production function or prices. Yield Comparison approach: assumes homogeneity in land, crops, husbandry, quality of produce and price between irrigated and non-irrigated production. The heterogeneity that occurs in these factors in reality brings into question use of the difference in net returns as the net WTP for irrigation water. Value added approach: can lead to correct results if, extreme caution is exercised with respect to the disaggregation of value added so that the opportunity cost of all other resources are empirically identified and deducted; or, the assumption that zero opportunity costs of the other primary resources is a valid assumption. If these assumptions are violated it can be expected that the resulting estimated value of the resource will be overstated. Using this approach as a method of ranking the value of water may also be misleading since this approach provides no indication of the value of water to the various users.
Validity	Validity of the approach requires, firstly, that profit maximising producers employ productive inputs up to the point at which marginal product is equal to the opportunity cost; secondly, that the total value of the product can be divided, so each input can be ‘paid’ according to its marginal productivity and the total value of product is thereby exhausted.
Value Concept Compatibility with TEV (incl. non-use)	Non-use value not measured
Value Measure	Estimates Producers surplus which can be converted to net average value; Marginal value also from residual value, change in revenue/productivity approaches
Reference	Heady (1952) (Gray and Young, 1984:171).

Valuation Method	Hedonic Price Method (HPM)
Description and Basis of Approach	Hedonic pricing employs differences in the prices of marketed goods to derive the value of environmental characteristics. Marketed goods can be viewed as comprising a bundle of characteristics; for some goods these include environmental characteristics. Individuals' preferences for environmental quality are reflected in the differential prices that they pay for such goods. Statistical analysis of the prices and characteristics of the goods is employed to derive an implicit value for environmental quality. In the case of housing, hedonic pricing assumes that the expected stream of benefits of living in a property is capitalised into the market value of the property. In this way, two properties in areas popular for water based recreation that differ only in respect of water quality have different market values, due to people's preferences for the difference in water quality. Hedonic pricing uses this difference in value as the implicit price of the difference in water quality. With adequate data and analytical skills, it is possible to determine the implicit price for environmental quality for properties that differ in not just one, but many factors. The implicit price of the environmental characteristic of interest is given by the responsiveness of property prices to change in the characteristic, as specified by the partial differential of the hedonic price function. A functional form for the hedonic price function is identified that best fits the data. This determines the functional form of the marginal implicit price function: the price is not necessarily constant; it might fall with increases in the characteristic, or it might be dependent on the level of another property characteristic. To obtain a value for changes in the environmental characteristic of interest, its implicit price (as indicated by the hedonic price function) is regressed against physical and socio-economic variables that are thought to influence demand for housing. The supply of housing is assumed to be fixed in the short run to enable identification of the demand or bid function, which is required for benefit estimation. Hedonic pricing rests on a number of stringent assumptions. It assumes a freely functioning and efficient property market and that individuals have perfect information and mobility. These conditions need to be met for individuals to buy the property and the associated characteristics that they desire and so reveal their demand for environmental quality. In reality, a large part of the housing stock may be in the public sector and so subject to price controls. The market may be segmented resulting in restrictions in mobility between areas. Individuals may not be fully informed about the environmental characteristics of properties prior to purchase. The market may not be in equilibrium, resulting in implicit prices that represent upper or lower boundary estimates of the true price. In many developing countries the property market is administered, preventing free operation of the price mechanism. Even where the markets do operate freely, records of transactions are not usually kept in any detail, severely restricting availability of data. For this reason, hedonic pricing is rarely applied in developing countries. A further assumption is that the measure used for the environmental characteristic in hedonic pricing reflects individuals' perceptions. Though an objective quantitative measure is required for the analysis, it may be that people perceive the environmental characteristic qualitatively. A broader measure of environmental quality may be needed if individuals do not discern changes in an individual variable. A further complication arises in the statistical analysis: if correlation occurs between variables, a trade-off has to be made between

	multicollinearity and bias due to the omission of significant explanatory variables.
Applicability	Applicable only to environmental attributes likely to be capitalised into the price of housing and/or land. Most applied to water quality and flooding. Also sometimes used in municipal, commercial, industrial, agricultural and recreational water uses.
Procedure	Hedonic pricing employs multiple regression econometric techniques and requires two stages of analysis: 1. Estimation of the hedonic price function - in this stage, variation in property prices is explained by regressing property price on explanatory variables relating to the attributes of properties (such as those listed above); i.e. the hedonic price function summarises the relationship between a property's market price and its characteristics. From the hedonic price function it is possible to derive the price differential for the characteristic of interest (e.g. the environmental good), which plots the additional price that must be paid by any household to move to attain a higher level that characteristic, holding all other variables constant. Mathematically, the price differential of the characteristic of interest is the partial derivative of the hedonic price function with respect to that characteristic, and is typically termed as the 'implicit price function'. 2. Derivation of demand curves and underlying values - property prices are determined by the interaction of supply and demand in the property market, and hence do not reflect the excess of willingness to pay over price paid, i.e. consumer surplus. The second stage of the HPP analysis therefore seeks to estimate the demand curve for the characteristic of interest so that full economic value may be inferred. Practically, the demand curve is mapped out by regressing quantity of the characteristic (the environmental good) on the price paid for the property, plus other additional relevant variables, such as socio-economic characteristics of households. From the demand curve, the change in consumer surplus due to a change in the environmental good may be calculated by the integral of the demand function with respect to the quantity of the environmental good between its initial level and final level.
Data Requirements & Implementation issues	As with all econometric analyses, the implementation of HPP is dependent on the quality of data, the specification of an appropriate functional form for the hedonic price function (e.g. linear, etc), the inclusion or omission of explanatory variables in the initial analysis, etc. Hedonic pricing requires data that can be used to relate house prices to relevant characteristics of individual properties (characteristics of the house e.g. number of rooms, type of neighbourhood, and environmental characteristics e.g. noise, water quality). Data on sale prices for actual market transactions are preferred over individual's own valuations of their property. The data are used to estimate a hedonic price function, which describes all points of equilibrium in the housing market between sellers' offers and buyers' bids for the environmental characteristic of interest. Practical applications of HPP require large amounts of data, particularly on property prices and property characteristics. Compilation of data on determinants of house prices may be difficult to measure and obtain. Data may be expert opinion of property values, self-reporting or related to actual sales, although the latter is the most accurate. Sources include the Land Registry, which reports price paid, or building society data on mortgage acceptances (although this will exclude those who buy outright). If data are collected from secondary sources (e.g. Land Registry, Census data etc.), it is generally not possible to link a particular individual buyer to the purchase price and other

	characteristics of a given property. Instead, the socio-economic characteristics of the neighbourhoods are used as these are assumed to be sufficiently similar to the individual buyers who live in or move into these neighbourhoods. However, HPP studies could also collect primary data by implementing surveys of home owners, providing a direct link between socio-economic characteristics and house prices. One solution to data collection and sorting difficulties is through the application of geographical information systems (GIS) to measure and compile data. If house price data are sufficiently disaggregated (e.g. to the level of an individual property) GIS can be useful in determining accessibility variables for individual properties (e.g. travel time, distance to amenities) as well as linking socio-economic and demographic census data to neighbourhood quality variables. Application of GIS to noise modelling impact analysis, for example from transport, also facilitates use of the HPP to derive values for actions such as noise mitigation, while using GIS to generate digital elevation models (DEMS) allows the impact of topography (the type and quality of view), an attribute which is typically omitted from HPP studies, to be incorporated into analysis. Aside from data issues, practical implementation of the HPP requires statistical (econometric) expertise. Depending on the ease of access to good data, the process can take six months to a year. Identification of a bid function is problematic. All consumers within a housing market face the same equilibrium price schedule, or hedonic price function. Hence, the observation of a single consumer's behaviour provides only one point on that consumer's bid function. Other marginal prices are observed only for other individuals, so they provide no indication of the likely reaction of the original consumer to varying prices. A number of solutions to the identification problem have been proposed. One option is to restrict variables or functional forms so that they are different between estimation of the hedonic price function and estimation of the bid function. The preferred alternative is to use data from spatially or temporally-separated markets, so that individuals do not face the same hedonic price function. This does, however, require that consumers are similar between these separate markets.
Advantages (strengths)	A distinct advantage of revealed preference techniques such as HPP is the use of readily available market data from actual behaviour and choices. Hedonic pricing is grounded firmly in the principles of economic theory, relying on the derivation of demand curves and elasticity estimates. Moreover, the theoretical expectations of HPP have typically been borne out by empirical studies.
Disadvantages (weaknesses)	Disadvantages of HPP lie in the requirement for copious amounts of data and specialist econometric expertise. In terms of undertaking hedonic analysis, other weaknesses arise from issues of identification and complementarity. More important from a decision-making perspective though, is that HPP is not suited for application where environmental impacts are not perceived (or observed) in property purchasing decisions, or where environmental impacts are yet to occur, since fundamentally environmental values are revealed from situations with precedence, i.e. difficulty in detecting and isolating effects of changes in prices based on changes in water. However, the corollary of this is that HPP may be suitable for long-standing environmental effects.
Validity	Hedonic pricing is founded on a sound theoretical basis and is capable of producing valid estimates of benefits as long as individuals can

	perceive the environmental change of interest. Market failures may mean that prices are distorted, that is markets may not behave as required by the approach. Data on prices and factors determining prices often difficult to come by. Hedonic pricing has been employed to produce reliable estimates of the values of actual environmental changes such as improved water supply. Limited tests of convergent validity but generally encouraging results.
Value Concept Compatibility with TEV (incl. non-use)	Does not measure non-use value
Value Measure	Marginal value if second stage of analysis undertaken, otherwise average value of water derived
Reference	Bockstael and McConnell (2006).

Valuation Method	Travel cost method (TCM)
Description and Basis of Approach	The travel cost approach takes the costs of travel that are incurred by individuals in visits made to recreational sites as a proxy for the value of recreation. The costs of travel (the costs of transport plus the value of time) are used as implicit prices to value the service provided and changes in its quality. Expenses will differ between sites (or for the same site over time) with different environmental attributes. Travel costs measure only the use value of sites and is usually limited to recreational use values; the option and existence value of the sites are measured using other techniques. There are two variants of the simple travel cost visitation model. The first can be used to estimate (representative) individuals' recreation demand functions. The visitation rate of individuals who make trips to a recreational site are observed, as a function of the travel cost. The value of the recreation site to an individual is measured from the area under his or her demand curve: the total recreation (use) value of a site is the area under each demand curve summed over all individuals. This 'individual' travel cost model requires that there is variation in the number of trips that individuals make to the recreational site, in order to estimate their demand functions. The second variant, known as the 'zonal' travel cost model, estimates aggregate or market demand for a site using standard statistical techniques. The unit of observation is the 'zone' as opposed to the individual. Zones are specified as areas with similar travel costs; the region surrounding a site is divided into zones of increasing travel cost. The method entails observation of the number of visits to the recreational site per capita of population for each zone. Data are again collected through a survey of visitors to the site. For both variants, the demand curve is estimated by the regression of the visit rate against socio-economic factors (such as income), the travel cost of visiting the site and some indicator of site quality. The data requirements are, therefore, considerable. For the individual model, data is required on each individual's socio-economic characteristics; in the case of the zonal model, these data are required for the population of each zone. Data are also required on the nature of each trip to the site, the distance traveled, time taken and cost of travel. The data are usually gained from existing or specially commissioned surveys. The method also requires a measure of site quality, which can be an intangible variable. A measure of site quality can range from angling catch rates to biochemical indicators such as concentrations of dissolved oxygen. The key issue is that the measures of site quality used are robust in relation to measures that individuals perceive as relevant. Unless the site that is being valued is unique, individuals have access to substitute sites that they can use for the same or similar recreational activities. Omission of substitute sites from the analysis creates a source of bias in the analysis. There is, however, no simple means of incorporating substitute sites into the individual and zonal travel cost models presented here. Multi-site models can be used. These vary in their complexity and their ability to explain substitute behaviour. Judgement on the part of the analyst is required to determine which substitute sites to include. Restrictions are often placed on site characteristics (some studies are limited to 'typical' sites) or demand equations (such as the use of 'pooled' models). Morey's (1984, 1985) 'share' model considers the allocation of an individual's fixed time budget between sites. This accounts for site substitution, but at the expense of explaining the total amount of time allocated to recreation.

Applicability	Generally limited to recreational water uses.
Procedure	The travel cost method is a survey-based technique implemented as follows: 1. A questionnaire is administered to a sample of visitors to a site in order to ascertain their place of residence, demographic and attitudinal information, frequency of visit to the site and other similar sites, and trip information (purposefulness, length, associated costs etc). 2. From the survey data, visit costs can be calculated and related, with other relevant factors, to visit frequency so that a demand relationship may be established via econometric (statistical) analysis. 3. The demand function can then be used to estimate the recreation value of the whole site in terms of consumer surplus, which when consider in together with 'price' paid (e.g. travel costs) yields a WTP estimate of a site's recreational value. More advanced studies attempt to estimate demand equations for differing attributes of recreation sites and estimate values for these individual attributes. The ZTCM is implemented as follows: 1. Collect data via on-site surveys on the number of visits made by households in a period and their origin. 2. The area encompassing all visitor options is sub-divided into zones of increasing travel cost. Household visits per zone are calculated by allocating sampled household visits to their relevant zone of origin. The household average visit rate in each zone is calculated by dividing the number of household visits in each zone by the zonal population. 3. Zonal average cost of a visit is calculated with reference to the distance from the trip origin to the site. 4. The demand curve is fitted relating the zonal average price of a trip (the travel cost) to the zonal average number of visits per household. For each zone, household consumer surplus is calculated by integrating the demand curve between price (travel cost) of visits actually made from each zone and the price at which the visitor rate would fall to zero (i.e. the vertical intercept of the demand curve). Implicit in the estimation of the demand curve is the assumption that households in all distance zones react in a similar manner to visit costs (i.e. when faced with the same cost, they would make the same number of visits). 5. Annual total consumer surplus for the site recreation experience is estimated by the dividing total household consumer surplus by the zonal average number of the visits made, giving the zonal average consumer surplus per household visit. This is then multiplied by the zonal average number of visits per annum to obtain annual zonal consumer surplus. Totalling annual zonal consumer surplus across all zones gives an estimate of total consumer surplus per annum for the recreational experience of visiting the site. The principal distinction between the ITCM and ZTCM approaches is the definition of the dependent variable. Using the ITCM, where this is defined as the number of visits per period by an individual (or household) to a site, the derived demand curve relates an individual's annual visits to the cost of those visits. This allows various individual specific variables to be incorporated in the trip-generating function. With the ITCM approach, the demand curve is derived from the change in visits over the change in travel costs (i.e. the first order derivative

	from the trip-generating function). Integration of the area under the curve yields an estimate of consumer surplus per individual; overall consumer surplus for the site is estimated by multiplying this by the number of individuals visiting the site annually.
Data Requirements & Implementation issues	The TCM survey is required to collect data on place of residence of visitors, demographic and attitudinal information, frequency of visit to the site and other similar sites and trip information (e.g. purposefulness, length, associated costs etc). Practical application of the ZTCM requires data concerning population of the each of the travel cost zones that are identified. Data on explanatory variables which are also likely to influence visit rates, e.g. income, preference and availability of alternative sites is also important, as is the mode of travel (car, rail, etc) to the site. Limited availability of data are likely to mean that only reduced forms of the trip-generating function can be estimated. A particular problem associated with the ITCM model is that variation in trips is not always observed, especially as not all individuals make a positive number of trips to a recreational site - some individuals do not make any. If standard statistical techniques such as Ordinary Least Squares (OLS) are used in the data analysis, non-participants are excluded from the data sets. This exaggerates participation rates and results in the loss of potentially useful information about the participation decision. However inclusion of data on individuals in the sampling area requires use of more complex statistical methods – in particular, discrete choice models. Use of geographical information systems (GIS), particularly in applications of the ZTCM, can help define travel costs zones with to account for areas with similar travel costs, availability of substitute sites and socio-economic characteristics (Bateman et al., 2005 and 2006). In addition to collecting appropriate data, the TCM also requires econometric expertise. The timescale for analysis will typically depend on the length of survey stage. For certain sites it may be necessary to sample at different times of the year in order to provide an accurate account of seasonal variations in visitor patterns and number. Given this implementing, the TCM approach may require a time frame from 6 months to (potentially) one and half years, allowing for data analysis.
Advantages (strengths)	Its strength is that, in theory, it is based on observed behaviour. The TCM is a potentially useful tool for producing estimates of the use value associated with well-defined recreation sites. A distinct advantage that estimated values are revealed from actual behaviour of individuals and the formulation of demand curves. Analysis of demand curves can also yield significant input to analysis of visitor rates and changes in these, which can aid the management of these sites.
Disadvantages (weaknesses)	The technical and data requirements should not be underestimated; travel cost is unlikely to be a low cost approach to valuation of non-marketed services. Practical applications of the approach, may be limited by data availability. More methodological concerns may disadvantage the use of TCM results, particularly with regards to different estimates of consumer surplus that may arise as a result of adopting either the ITCM or ZTCM approach, as well as the treatment of substitute sites, the choice of appropriate functional form and the calculation of the value of time (all as discussed above). Finally, the TCM is not able to account for environmental goods (or bads) that are imperceptible to short-term visitors.
Validity	The travel cost method is a technically well-developed valuation approach, which has been extensively employed over the past two decades. Theoretically correct, but complicated when there are multi-purpose trips and competing sites. Some doubts about ‘construct validity’ in that

	number of trips should be inversely correlated with ‘price’ of trips –that is, distance travelled. Some UK studies do not show this relationship. Convergent validity generally good in US studies. Generally acceptable to official agencies and conservation groups. The individual travel cost model is generally preferred to the zonal variant. The latter is statistically inefficient, as it aggregates data from a large number of observations into a few zonal observations. Also, it assumes that the cost of travel to the site for all individuals within each zone is equal, which is often not the case.
Value Concept Compatibility with TEV (incl. non-use)	Cannot be used to estimate non-use values.
Value Measure	Usually consumer surplus based and hence average value of water; sometimes marginal value??
Reference	Bockstael and McConnell (2006).

Valuation Method	Contingent valuation (CVM)
Description and Basis of Approach	A survey instrument is used to measure individuals' maximum willingness to pay (WTP) for an aspect of a water resource, presented to them in a hypothetical market with a proposed improvement. CVM can also be used to measure what people are willing to accept (WTA) by way of compensation for a deterioration in quality of a water resource. The CVM is based on the consumer demand theory which explains the factors determining demand, in this case, for environmental goods and services. These factors include tastes, attitudes, socio-economic characteristics, characteristics of the environmental good and service, the cost of (or avoiding) the environmental change and the price of other goods and services. Therefore, before asking a WTP or WTA question, a CVM questionnaire provides information on: An introduction to the general decision-making context; A detailed description of the good or service offered to the respondent; The institutional setting in which the good or service will be provided; The way in which the good or service will be paid for; and Reminders about respondents' budget constraint including other things they may wish to purchase. Essentially, this information describes the hypothetical market which respondents are required to engage in. The questionnaire also collects information about tastes, attitudes, prior experience of using or knowing about the good or service in question and the socio-economic characteristics of the respondents. The survey may either elicit the willingness to pay measure of economic value or the willingness to accept measure of economic value. Given that the WTP and WTA responses are elicited in the context of the hypothetical market presented in the questionnaire, the economic values estimated via the CVM are contingent upon this hypothetical market. A respondent's choice or preference can be elicited in a number of ways. The simplest is to ask the respondent a direct question about how much he or she would be willing to pay for the good or service (known as continuous or open-ended questions). High rates of non-responses can be a problem with this approach. Alternatively, a respondent can be asked whether or not he or she would want to purchase the service if it cost a specified amount. These are known as discrete or dichotomous choice questions, and may be favoured because they do not give the respondent any incentive to answer untruthfully, i.e. the approach is 'incentive compatible'. A hybrid approach is the 'bidding game', where respondents are asked a series of questions to iterate towards a best estimate of their valuation. Alternatively, respondents may be shown a list of possible answers – a 'payment card' – and asked to indicate their choice, though this requires a careful determination of the range of possible answers. Each approach implies particular requirements in terms of statistical methods, and the appropriate choice for a specific problem is a matter of judgement on the part of the analyst.
Applicability	Extensive, since it can be used to derive values for almost any environmental change. This explains its attractiveness to 'valuers'. The contingent valuation method can be useful for eliciting the value of several aspects of water resources including water quality, recreation and biodiversity. CVM can be employed to calculate both use and non-use values including option and existence values.
Procedure	Applying the CVM requires a number of steps: 1. Development of the survey instrument (i.e. the questionnaire) for elicitation of respondents' preferences for the non-market good - this

	requires (i) design of the hypothetical scenario, (ii) determining as to whether WTP or WTA is to be sought and (iii) specification of the scenario concerning payment or compensation. The initial design stage of the CVM implementation process is typically augmented by focus groups, one-to-one interviews, or workshops to aid the development of the survey instrument. In addition to the valuation scenario and other questions mentioned above, it is common practice to include a 'debriefing' section in which respondents state why they answered certain questions in the way they did. A final element of the design stage is typically a pilot survey which administers a draft questionnaire to a sample of respondents in order to test the survey instrument in the field. 2. Implementation of the CVM survey instrument with a sample of the population of interest - the survey instrument may be administered in a number ways: on site or door to door face to face interviews, via remote telephone or mail surveys, or web-based surveys. Face to face surveys are the recommended approach. 3. Analysis of the survey responses – this typically has two elements: (i) estimation of average WTP/WTA from the sample data for the population of interest; (ii) estimation of WTP/WTA functions via econometric analysis to assess the determinants of WTP/WTA and to judge the validity and reliability of the surveys results. The analysis undertaken will depend on the elicitation format (the way in which the WTP/WTA questions in asked). These include open-ended (respondents can state any amount), dichotomous choice (respondents refuse or accept pre-specified amounts), or payment ladder (respondents are given a choice of amounts). Estimation of aggregate or total WTP/WTA for the specified change in the provision of the non-market good of interest over the relevant population. This estimate of overall WTP may then be applied in decision-making exercises, for example, for use in cost-benefit analysis.
Data Requirements & Implementation issues	Primarily, the CVM survey instrument will be designed to collate all data required for estimating WTP/WTA values and functions for the determining the main influences on respondents' WTP. As mentioned above, consumer demand theory helps in deciding which factors to include in these functions. In addition, a crucial aspect of CVM survey design is ensuring that the survey sample is representative of the population of interest. Generally representativeness will be based on socio-economic characteristics (e.g. sourced from census data), since data typically do not exist on other factors that may be relevant, for instance prior experience of the environmental good or service in question. Reliable CVM studies are not simple (or inexpensive) to implement. Proper practice in CVM studies requires time to develop the survey instrument and to ensure that the non-market good or service to be valued is clearly explained along with the constructed market and payment method. Overall, from the initial design stages of the survey instrument to aggregating and reporting of results, practical implementation of the CVM could require six months to a year depending, particularly on aspects such as complexity of the issue of concern and sample size. Considering the constant developments in stated preference techniques (including CVM), leading practitioners should be involved in a study at least in a peer review capacity. Analysis of a CVM dataset; the estimation of WTP/WTA values and functions, as well as validity and reliability testing requires econometric expertise.
Advantages (strengths)	Primarily, stated preference techniques (including CVM) are the only approach that can estimate non-use value associated with

	environmental goods and services. Furthermore the CVM approach to valuing environmental goods and services offers a great deal of flexibility; in particular the construction of a hypothetical market can be envisaged for numerous environmental goods and services at differing degrees of quality irrespective as to whether they have precedence. In addition the CVM enables a great deal of information to be collated and analysed from the target population concerning their attitudes towards, use and experience of environmental goods and service in addition to eliciting WTP/WTA amounts and WTP functions concerning the determinants of WTP.
Disadvantages (weaknesses)	Economic analysis has typically favoured evidence based on actual market behaviour over hypothetical approaches. However, it would appear that is necessary to trade-off this 'real' market data with data from hypothetical markets in order to account for non-use values, which can form a substantial proportion of total economic value and provide justification for preservation of the natural environment. Much emphasis should also be placed on ensuring that practical application of the CVM are guided by current best practice, particularly since the method is a focal point for much discussion concerning the measurement of individual's preferences. This may imply that the approach is relatively expensive to undertake, although the cost of undertaking a CVM study should be viewed in comparison to the actions which are the concern of the decision-making context.
Validity	The literature has identified various forms of potential bias. 'Strategic bias' arises if respondents intentionally give responses that do not reflect their 'true' values. They may do this if they think there is potential to 'free ride'. However, there is limited evidence of strategic bias. 'Hypothetical bias' arises because respondents are not making 'real' transactions. Costs of studies usually limit the number of experiments involving real money (criterion validity), but some studies exist. Convergent validity is good. Construct validity – relating value estimates to expectations of values estimated using other measures –is debated, especially the marked divergence in many studies between WTP and WTA compensation.
Value Concept Compatibility with TEV (incl. non-use)	Together with choice experiments only method for eliciting non-use values.
Value Measure	Average, Marginal or Total value can be estimated depending on purpose of study
Reference	Bateman et al (2002), Mitchell and Carson (1989)

Valuation Method	Choice Modelling
Description and Basis of Approach	Choice modelling approaches are based around the notion that any good can be describe in terms of it characteristics (or 'attributes') and the levels that these characteristics take. For example, a lake may be described in terms of its ecological quality, chemical water quality of appearance. Likewise, woodland can be described in terms of its species diversity, age and recreational facilities. The following briefly summarises the different choice modelling approaches: • Choice experiments – in this approach respondents are presented with a series of alternatives and are asked to choose their most preferred. In order for estimates of economic value to be derived, a baseline option corresponding to the status quo or a 'do nothing' option is included in the choice set presented to respondents. With choice experiments, respondents are required to trade-off changes in attribute level against the cost of these changes. In addition, though, the baseline option implies that respondents can opt for the status quo at no additional cost. Data from choice experiments is typically analysed by econometric techniques based on the theory of rational probabilistic choice, enabling estimates of willingness to pay to be derived for different attributes of environmental goods and services. • Contingent ranking – in this approach respondents are required to rank a set of alternative options. Each alternative option is characterised by a number of attributes which vary in level across different options. In order for results from contingent ranking exercises to be consistent with economic theory, one of the options presented to respondents must represent the status quo. If the status quo is not included, then respondents are effectively 'forced' to choose one of the alternative options (neither of which they may actually prefer). • Contingent rating – with this approach respondents are presented with a number of scenarios and are asked to rate each one on a numeric or semantic scale. Notably contingent rating does no involve the direct comparison of alternative options. • Paired comparisons – in this approach respondents are required to choose their preferred alternative out of a set of two choices and to indicate their strength of preference on a numeric or semantic scale. Effectively a paired comparison exercise combines elements of choice experiments (selecting the most preferred alternative) and rating exercises (rating strength of preference).
Applicability	Extensive, since it can be used to derive values for almost any environmental change. Important advantage compared to CVM is the valuation of environmental attributes, making it possible to derive marginal economic values for a range of environmental changes. This explains its attractiveness to 'valuers'. Requires substantial knowledge of econometric analysis. Together with CV the only method for eliciting non-use values.
Procedure	Regardless of the specific approach, choice modelling exercises typically feature the same common stages of implementation 1. Development of survey instrument (i.e. the questionnaire) – primarily, this initial step involves identifying the relevant attributes of the non-market environmental good or service in question. Attributes may be based on impacts arising from policy or project options or those thought to be significant to the preferences of respondents. Focus groups and/or cognitive testing are typically useful for this

	process. Once the attributes are determined, the survey instrument is developed via the assignment of levels to attributes (which should be realistic and span the range over which respondents are expected to have preferences). Statistical design theory is then used to combine the levels of attributes into a number of alternative scenarios to be presented to respondents. From this, 'choice sets' may be constructed which provide the alternative options with which respondents are presented. As with the contingent valuation method (CVM), the choice modelling survey instrument will typically be tested via a pilot survey prior to its full implementation. 2. Implementation of the survey – the survey instrument is administered to a sample of the population of interest. As with the CVM, the survey instrument may be administered in a number ways: on site or door to door face to face interviews, via remote telephone or mail surveys, or web-based surveys. Aside from the choice modelling exercise, the questionnaire will also collate information on the respondent's attitudes, experience and use of the environmental good or service in question as well socio-economic characteristics in the same way as a CVM survey. 3. Analysis of the survey responses – analysis of choice modelling data sets can be quite involved. Initially data must be organised and coded according to the choice sets and attribute levels faced by each respondents. In the case of choice experiments, the dataset is typically estimated on the basis of limited dependent variable models, while contingent ranking datasets may be analysed by censored dependent variable models. Analysis of choice experiment data is in fact similar to that of the random utility model (see separate fiche, in particular the property of independence of irrelevant alternatives) where econometric methods focus on the probability that a respondent will choose a particular option (e.g. binary logit or multinomial logit/probit model). With choice experiment data, estimates of WTP are derived from the parameter estimates of the choice models. As with CVM surveys, analysis of choice modelling data also entails validity testing, particularly through the estimation of WTP functions. Aggregation of results – the final step of the analysis (in choice experiments) is to aggregate WTP estimation for the specified change in the provision of the non-market good of interest over the relevant population. This estimate of overall WTP may then be applied in decision-making exercises, for example, for use in cost-benefit analysis.
Data Requirements & Implementation issues	Primarily, the choice modelling instrument will be designed to collate all data required for estimating preferences environmental goods and services and in the case of choice experiments, estimating WTP functions for the determining the main influences on respondents WTP. Aside from data on respondent WTP and information from debriefing questions, the dataset will also include information on respondent socio-economic and demographic characteristics, as well as information on respondent attitude towards the environmental good or service and their prior experience of the good or service. In addition, a crucial aspect of choice modelling survey design is ensuring that the survey sample is representative of the population of interest. Generally representativeness will be based on socio-economic characteristics (e.g. sourced from census data), since data typically does not exist on other factors that may be relevant, for instance prior experience of the environmental good or service in question. As with CVM studies, reliable choice modelling exercise are not simple to implement and take time to develop to ensure that the survey instrument and choice sets cover the range of scenarios required and that the procedure is clearly explained to respondents. Overall, from the

	initial design stages of the survey instrument to aggregating and reporting of results, practical implementation of the CVM could require six months to a year depending, particularly on aspects such as complexity of the issue of concern and sample size. Considering the constant developments in stated preference techniques (such as choice experiments), leading practitioners should be involved in a study at least in a peer review capacity. Analysis of choice modelling datasets; the estimation of WTP/WTA values and functions, as well as validity and reliability testing requires econometric expertise.
Advantages (strengths)	One distinct advantage of choice modelling is that it can be seen as a generalised form of a discrete choice CVM study (e.g. a change or no-change scenario). However, in the CVM approach it is not possible to analyse the attributes of the change in question without designing different valuation scenarios for each level of the attribute, which would be a costly undertaking. However, since choice experiments can incorporate more than two alternatives, they are more suited to this form of analysis. In addition, choice experiments are more suited to measuring the marginal value of changes in the characteristics of environmental goods, which may be useful from a management of resources perspectives, rather than focusing on either the gain or loss of the good and more discrete changes in attributes. Choice experiments may also avoid some of the response difficulties which are encountered in CVM studies. For instance CVM studies using a dichotomous choice format may be subject to 'yea-saying' where respondents see a positive answer as a socially desirable response or as a strategic response. However, in a choice experiment setting, respondents get many chances to express a positive preference for a good over a range of payment amounts; hence such behaviour will likely be avoided.
Disadvantages (weaknesses)	Disadvantages of choice modelling include the fact that more complex choice modelling designs may cause problems for respondents leading to an increased degree of random error in responses. Therefore it should be expected that as the number of attributes (or rankings increase) the likelihood of inconsistent responses will also increase due to limits in cognitive ability. Additionally, contingent rating and paired comparisons will not yield values consistent with economic theory due to the absence of a status quo option for respondents.
Validity	Use and applicability is increasing in the literature over the past 5 years. From a welfare-theoretical point of view superior to contingent ranking. Important methodological issue include the independency between attributes and the orthogonal design of the experiment.
Value Concept Compatibility with TEV (incl. non-use)	Together with contingent valuation only method for eliciting non-use values.
Value Measure	Marginal, Average and Total value can be estimated depending on purpose of study??
Reference	Kanninen (2007), Bateman et al., 2002; Louviere et al., 2000

Valuation Method	Avertive behaviour & defensive expenditures/ Avoidance Costs (structural adjustment, abatement)
Description and Basis of Approach	Costs and expenditures incurred in mitigating the effects/avoiding damages of reduced environmental functionality. Adverse impact can be avoided by buying durable foods, non-durable goods, changing daily routines to avoid exposure to pollutants, etc. Perfect substitutability provides the basis for the averting behaviour and defensive expenditures technique, which focuses on averting inputs as substitutes for changes in environmental characteristics. For instance, expenditures on sound insulation can be used to indicate householders' valuations of noise reduction; and expenditure on liming might reflect the value of reduced water acidification. Fairly crude approximations can be found by looking directly at changes in expenditure on a substitute good that arise as a result of some environmental change. Alternatively, the value per unit change in an environmental characteristic can be determined. This involves determining the marginal rate of substitution between the environmental characteristic and the substitute good, using known or observed technical consumption data. The marginal rate of substitution is multiplied by the price of the substitute good to give the value per unit change in the environmental characteristic. Variants of avoidance cost approaches include: Structural adjustment costs: Are those costs incurred to restructure the economy (production and consumption patterns) in order to reduce water pollution or other forms of environmental degradation to a given standard. It addresses both production activities and consumption. Measuring the cost of structural change often requires complex economy wide modelling. Abatement Costs: Expenditures on introducing technologies to prevent water pollution. These technologies include both end of pipe or change in process solutions. Inputed abatement costs should always be calculated as the sum of direct and indirect cost effects of additional prevention measures. Abatement costs describe the costs of technical options for reducing a certain type of pollution. This is presented as cost functions (abatement cost curves). Typically these cost functions display increasing marginal costs, though exceptions exist.
Applicability	Limited to cases where households spend money to offset environmental hazards, but these can be important – for example water filters. Potentially useful in Irrigation, industry, municipal use
Procedure	Whilst used comparatively rarely, the approach is potentially important. Expenditures undertaken by households and designed to offset an environmental risk need to be identified. Examples include noise abatement, reactions to radon gas exposure –for example purchase of monitoring equipment, visits to medics, and so on. Econometric analysis on panel and survey data is sometimes needed. Can be fairly expensive.
Data Requirements & Implementation issues	The approach requires data on change in an environmental characteristic of interest and its associated substitution effects. Three types of data are required to derive abatement cost curves: Data on emissions by economic activities and underlying production processes disaggregated according to technical characteristics; parameters of available abatement techniques/ measures (e.g. reduction potential, actual rate of application for each production process/economic activity); and costs data for these measures. The physical data will very often come from emissions data and technical abatement data often originating from technology oriented databases. Costs data are

	available on a micro-economic scale relating to a particular technology or process. Aggregation will thus be needed to arrive at sector level and river basin indicators. However if measures are applied to a whole industry this can affect the whole cost structure, prices and level of output in the industry. Consequently, the accounting of abatement cost may be different for the micro-economic, meso- economic and macro-economic accounting levels. Furthermore, Abatement cost reflect the technological possibilities and knowledge available in a given account year. Cost will tend to decrease over time with technological progress and economies of scale such that cost data need to be updated regularly. To obtain total costs of technical measures a modelling approach is necessary, which will normally have an input-output core that is closely linked with direct cost calculations. Direct abatement cost calculations should take account of interactions between technical abatement measures, interactions between pollutants, incompatibility of abatement measures and the uncertainties of measurement of integrated techniques. Such problems are rarely covered in primary data sources. It is not always easy to distinguish between costs for environmental protection and other costs, for example associated with increased production efficiency of some technological improvement that reduces pollution.
Advantages (strengths)	Although the technique has rarely been used, it is a potentially important source of valuation estimates since it gives theoretically correct estimates which are gained from actual expenditures and which thus have high criterion validity.
Disadvantages (weaknesses)	If observed averting behaviour is not between two perfect substitutes, the value of the environmental characteristic is underestimated. For example, if there is an increase in environmental quality, the benefit of this change is given by the reduction in spending on the substitute market good required to keep the individual at their original level of welfare. However, when the change in quality takes place the individual does not reduce spending (in order to stay at the original level of welfare). Income effects cause reallocation of expenditure between all goods with a positive income elasticity of demand and consequently the reduction in spending on the substitute for environmental quality does not capture all of the benefits of the increase in quality. Further problems with the approach are that individuals may undertake more than one form of averting behaviour in response to an environmental change, and that the averting behaviour may have joint costs and multiple outputs/other beneficial effects that are not considered explicitly (e.g. air conditioner cools and cleans air at the same time; the purchase of bottled water to avoid the risk of consuming polluted supplies may also provide added taste benefits). Furthermore, averting behaviour is often not a continuous decision but a discrete one – a water filter is either purchased or not, for instance. In this case the technique again gives an underestimate of benefits unless discrete choice models for averting behaviour are used. Simple avertive behaviour models can, therefore, give incorrect estimates of value if they fail to incorporate the technical and behavioural alternatives to individuals’ responses to change in environmental quality. Abatement cost curves may not be stable over time.

Validity	Theoretically correct. Insufficient studies to comment on convergent validity. Although the technique has rarely been used, it is a potentially important source of valuation estimates since it gives theoretically correct estimates which are gained from actual expenditures and which thus have high criterion validity. Abatement costs do not represent the severity of the environmental problems; rather they represent the effort in terms of costs, of taking measures to rectify the environmental problems
Value Concept Compatibility with TEV (incl. non-use)	Does not measure non-use values, though arguable that payments to some wildlife societies could be interpreted as insurance payments for conservation
Value Measure	Lower bound estimate of WTP; Marginal or Average value depending on nature of study
Reference	Dickie, Gerking and Agee (1991); Baumol and Oates (1971)

Valuation Method	Replacement/Restoration Cost and Cost Savings (opportunity cost)
Description and Theoretical Basis	Potential expenditures incurred in replacing/restoring the function that is lost; for instance by the use of substitute facilities or 'shadow projects'. The replacement cost estimates the benefits of an environmental asset based on the costs of replacement or restoration. The replaced or restored asset is assumed to provide a direct substitute for the original. The technique is widely used because the data that are required are usually readily available from actual expenditures or estimated costings. The underlying assumption is that the costs of replacement equal the benefits that society derives from the asset. However, the benefits derived from the asset could substantially outweigh the costs of renovation or restoration, in which case the technique will underestimate the value of the asset. The replacement cost is thus a valid measure of economic value only in situations where the remedial work is required to comply with an economically determined environmental standard. Use of the replacement cost assumes that complete replacement or restoration is feasible. In the case of environmental assets this often is not the case. There are also temporal issues as replacement or restoration of an alternative water resource, e.g. a wetland, may not coincide directly with the damage or loss of the original resource. Because of the potential for confusion between costs and benefits, the replacement cost technique should be used with some care, and only when benefits cannot easily be estimated. The cost savings method is similar to the replacement cost, but determines the value of water in terms of the savings in costs made through use of a good or service provided by water versus the next best (cheapest) alternative source of the good or service. The method is fairly commonly employed to value use of water for transportation, and can be applied to other uses of water as well. The value of using water as a means of transporting goods is measured in terms of the cost savings that result from not transporting the same goods via an alternative means, typically by train. For example, the value of using water to transport goods can be determined as the difference between the cost of transporting goods by train and the cost of transporting goods by boat. The approach does not allow for the large differences in time costs between different transport modes. The method has also been used to value hydroelectric power generation by estimating the difference between the cost of generating hydroelectric power and the next cheapest alternative method of power generation (e.g. coal-fired). As with replacement costs, the approach equates cost savings with value, and hence can be criticised on the grounds that it implicitly assumes demand will be unresponsive to changes in costs.
Applicability	Replacement cost approaches also widely used because it is often relatively easy to find estimates of such costs. Replacement cost approaches should be confined to situations where the cost relates to achieving some agreed environmental standard, or where there is an overall constraint requiring that a certain level of environmental quality is achieved.
Procedure	
Data Requirements & Implementation issues	
Advantages (strengths)	

Disadvantages (weaknesses)	Data limitations on production cost and alternative production processes
Validity	Replacement Cost: validity limited to contexts where agreed standards must be met.
Value Concept Compatibility with TEV (incl. non-use)	
Value Measure	Net average value based on market price of replacement; Can be used as proxy for Marginal value
Reference	

Annex IV: AquaMoney surveys

To be included

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