Using economic valuation techniques to inform water resourcesmanagement: A survey and critical appraisal of available techniques and an application

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Abstract

The need for economic analysis for the design and implementation of efficient water resources management policies is well documented in the economics literature. This need is also emphasised in the European Union's recent Water Framework Directive (2000/60/EC), and is relevant to the objectives of Euro-limpacs, an EU funded project which inter alia, aims to provide a decision-support system for valuing the effects of future global change on Europe's freshwater ecosystems. The purpose of this paper is to define the role of economic valuation techniques in assisting in the design of efficient, equitable and sustainable policies for water resources management in the face of environmental problems such as pollution, intensive land use in agriculture and climate change. The paper begins with a discussion of the conceptual economic framework that can be used to inform water policy-making. An inventory of the available economic valuation methods is presented and the scope and suitability of each for studying various aspects of water resources are critically discussed. Recent studies that apply these methods to water resources are reviewed. Finally, an application of one of the economic valuation methods, namely the contingent valuation method, is presented using a case study of the Cheimaditida wetland in Greece.

Keywords: Water resources; Economic value; Cost-benefit analysis; Hedonic pricing; Travel cost; Contingent valuation; Choice experiment

1. Introduction

The Water Framework Directive (WFD) of the European Union (EU) (2000/60/EC) defines water resources to include surface water, groundwater, inland water, rivers, lakes, transitional waters, coastal waters and aquifers (Chave, 2001). Together, these water resources are crucial to human health and the natural environment, and are vital to the European economy. Water resources are necessary inputs to production in economic sectors such as agriculture (arable and non-arable land, aquaculture, commercial fishing, and forestry), industry (e.g. power generation) and tourism, as well as to household consumption (UNEP, 2005).

Over time, however, water resources have been degraded and depleted. With respect to water quantity, these trends have grown stronger within the past century during which global freshwater-use increased six-fold, and 50% of global wetlands were lost (IUCN, 2005). Water quality has arguably improved over the last century in the original EU member states as a result of more sophisticated wastewater treatment since the Industrial Revolution (Burton, 2003) and a decline in acid deposition resulting in the recovery of some lakes (Stoddard et al., 1999). There is now an emphasis on diffuse rather than point sources of water pollution. These adverse effects on water are a result of increasing water demand from agriculture, industry, hydroelectric generation, and continued pollution. The effects are further exacerbated by population growth, rapid urbanisation and climate change (UNEP, 2000). From an economic perspective, water resources are over-extracted and are not efficiently allocated. This is due in part to the existence of market and government failures at the local, national and international levels. Private costs and benefits diverge from social costs and benefits, leading to social welfare losses (Pearce and Turner, 1990).

In recognition of the deterioration in the quantity and quality of water, several initiatives have been undertaken to ensure the sustainable management and conservation of this valuable resource. The EU's WFD aims to protect and achieve a “good status” for all water resources by 2015, with a combined approach of emission limit values, quality standards, and the introduction of more efficient water prices. There are also international efforts to conserve water resources, such as the 1971 Ramsar Convention on Wetlands of International Importance, providing a framework for national action and international cooperation for the conservation and wise use of wetlands (Ramsar, 1996). Relevant to these efforts is a recent EU-funded project called Euro-limpacs, which is designed to assess the effects of future global change on Europe's freshwater ecosystems (Wade, 2006-this volume). The work presented aims to contribute to this project by providing a decision-support system for valuing changes in environmental quantity and quality. There are three objectives: first, to highlight the need for economic analysis in the design and implementation of efficient and effective water resources management strategies and policies; second, to explain and critically assess the suitability of various economic valuation techniques for this purpose; and third, to demonstrate how these methods can be used in the development of appropriate policies for sustainable water resources management.

The paper is structured as follows: The next section discusses the role of economic analysis in efficient water resources management. In Sections 3 and 4, the most commonly used economic valuation methods, namely revealed preference methods and stated preference methods, are described. The context in which each of these methods can be used and their respective limitations are explained. The theory is illustrated with examples of existing studies that have employed these methods to estimate the values of water resources. Section 5 presents an example of one of the economic valuation methods, namely the contingent valuation method, using the Cheimaditida wetland in Greece as a case study. Finally Section 6 concludes and discusses implications for policy.

2. The economics of water resource depletion and degradation: a conceptual framework

Although water resources are vital for the functioning of any economy, they continue to be depleted and degraded at an unsustainable rate. This is true for both developed and developing countries alike, and is due to the nature of the economic development and growth path that has been chosen thus far, which has readily substituted environmental resources (such as water) for other forms of economic resources such as capital and labour for the production of goods and services that are deemed to be more productive and yield higher returns (Swanson and Johnston, 1999). This path has been chosen because the value of environmental resources has often been overlooked in development decisions. Economic efficiency occurs at the point where net social benefits (i.e., benefits minus costs) of an economic activity are maximised, or equivalently, when the marginal benefits are equal to marginal costs. To implement the most efficient social and economic policies that prevent the excessive degradation and depletion of environmental resources, it is necessary to establish their full value, and to incorporate this into private and public decision-making processes.

A widely accepted and often used framework for decision-making is Cost Benefit Analysis (CBA). CBA is an analytical tool based in welfare theory, which is conducted by aggregating the total costs and benefits of a project or policy over both space and time (Hanley and Spash, 1995). A project or policy represents a welfare improvement only if the benefits net of costs are positive. Different management options will yield different net benefits and the option with the highest net benefits is the preferred or optimal one.

A CBA of a policy or project with environmental impacts is complicated because many environmental resources (including most water resources) are public goods. A good is public to the extent that consumption of it is non-rival and non-excludable; it is non-rival if one person's consumption of the good does not reduce the amount available to others and non-excludable if it is not possible to supply the good only to those who choose to pay for it and exclude everyone else. Pure public goods cannot be provided by the price mechanism because producers cannot withhold the good for non-payment, and since there is no way of measuring how much a person consumes, there is no basis for establishing a market price. Public goods are therefore not traded in markets as private goods are, and are thus often under-produced or exploited by the market. This phenomenon is called a 'market failure' in economic terms. Both surface water and groundwater have public good characteristics in that people who extract them and use them are not paying their scarcity rents (both in terms of quality and quantity); they only pay the private extraction costs. When scarcity rents go unrecognised, this results in inefficiently high extraction or pollution rate over time and space (Koundouri, 2000). Other causes of market failure include insufficient or non-existent property rights, externalities, the lack of perfect competition (e.g., market power) and lack of perfect information. The property rights issue is especially important in the context of water resource management. If there were private property rights, then for example an upstream polluter of water would be legally required to compensate the downstream property rights owner for damages, thus leading to the 'optimal' level of pollution. Externalities are defined as benefits or costs, generated as a byproduct of an economic activity, that do not accrue to the parties involved in the activity. An externality can be local, in which case it is confined to a specific location, or global, and it can be positive or negative. Where market failures exist, government must intervene to allocate the resources efficiently. Generally, governments do not intervene to correct these failures because environmental conservation is not a high priority. In the case of water supply, a basic human necessity, the government has a stronger incentive to intervene to provide the population with clean water. Although this is true for both developed and developing countries, water quality standards in developing countries tend to be lower than in the developed countries (e.g. EU standards for drinking water quality are stricter than those of the World Health Organisation), and government intervention in the developing world is often slower due to budget constraints and incomplete or non-existent infrastructure and institutions. In addition, certain government policies such as subsidies distort the prices of environmental resources thereby not accounting for their economic scarcity. These result in the phenomenon of 'government failure'.

To correct for these failures, the value of all the benefits provided by environmental resources need to be captured. Environmental economists have been at the forefront arguing that individuals may derive values from non-market goods, especially environmental resources, through many more sources than just direct consumption (Pearce and Turner, 1990). More specifically, they refer to the importance of considering the Total Economic Value (TEV) of an environmental resource. TEV recognises two basic distinctions between the value that individuals derive from using the environmental resources, i.e. use values, and the value that individuals derive from the environmental resource even if they themselves do not use it, i.e. non-use values. Use values can be further classified into three broad categories: Direct use values, indirect use values, and option values. Direct use values come from the

Table 1
Components of TEV of water resources and appropriate economic valuation methods

TEV component	Economic valuation methods [a]
Direct use values	
Irrigation for agriculture	PF, NFI, RC, MP
Domestic and industrial water supply	PF, NFI, RC, MP
Energy resources (hydro-electric, fuelwood, peat)	MP
Transport and navigation	MP
Recreation/amenity	HP, TC, CVM, CEM
Wildlife harvesting	MP
Indirect use values	
Nutrient retention	RC, COI
Pollution abatement	RC, COI
Flood control and protection	RC, MP
Storm protection	RC, PF
External eco-system support	RC, PF
Micro-climatic stabilisation	PF
Reduced global warming	RC
Shoreline stabilisation	RC
Soil erosion control	PF, RC
Option values	
Potential future uses of direct and indirect uses	CVM, CEM
Future value of information of biodiversity	CVM, CEM
Non-use values	
Biodiversity	CVM, CEM
Cultural heritage	CVM, CEM
Bequest, existence and altruistic values	CVM, CEM

With modifications adopted from Barbier (1991), Barbier et al. (1997), Woodward and Wui (2001), Brouwer et al. (2003), and Brander et al. (2006).

[a] Acronyms refer to production function (PF), net factor income (NFI), replacement cost (RC), market prices (MP), cost-of-illness (COI), travel cost method (TCM), hedonic pricing method (HP), contingent valuation method (CVM), and choice experiment method (CEM).

consumptive use of the environmental resource itself. With regard to water resources, these include drinking water, irrigation, or as an industrial input (Table 1). For most private (normal) goods, value is almost entirely derived from their direct use. Many environmental resources however perform an array of functions that benefit individuals indirectly: indirect use values of water resources include benefits such as flood control, nutrient retention, and storm protection. Finally, option value recognises that individuals who do not presently use a resource may still value the option of using it in the future. The option value for water resources therefore represents their potential to provide economic benefits to human society in the future.

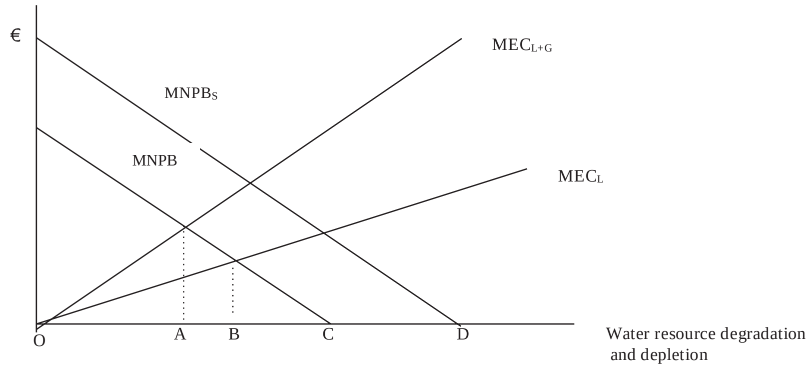
A further major expansion of value of an environmental resource is the inclusion of non-use values (Krutilla, 1967). These are values that individuals may derive from environmental resources without ever personally using or intending to use them. These can be further classified into three categories, namely existence value, bequest value, and altruistic value.

Existence value refers to the value individuals may place upon the conservation of an environmental resource, which will never be directly used by themselves or by future generations. Individuals may value the fact that future generations will have the opportunity to enjoy an environmental resource, in which case they might express a bequest value. And finally, altruistic value states that even if the individuals themselves may not use or intend to use the environmental resource themselves, they may still be concerned that the environmental good in question should still be available to others in the current generation.

These concepts are illustrated in Fig. 1. The MNPB curve represents the marginal net private benefits of using water resources, where $MNPB_S$ curve represents the marginal net private benefits of using water resources exacerbated by subsidies to their use. The MEC_L is the marginal external costs borne locally from use of water resources and the MEC_{L+G} is the local and global marginal external costs from use of water resources, measured by the TEV of the water resources. These curves result in four equilibria, with four levels of water resource use. Point C is the local private optimum, where all externalities are disregarded and there are no subsidies to water use.

Point D is the local private optimum, where, again all externalities are disregarded and water use is subsidised. Point B is the local social optimum, where local externalities are internalised but global externalities are ignored, and point A is the global social optimum, where all externalities are internalised. When an externality is internalised, the market and government failures have been corrected to the point where economic efficiency has been attained. The government failure is measured by distance CD–i.e. the quantity and quality of water resources that is lost due to its conversion for use in economic activities (e.g., irrigation for agriculture or a waste sink for pollution run-off from industry) as a result of government subsidies. Local market failure is measured by BC, and global market failure by AB. The distance AD reflects the inefficiency of water resource use, as shown by the divergence between the private and social optimum. The efficient use of water resources occurs at OA (Pearce, 2001).

To summarise, values of water resources are not straightforward to estimate for CBA purposes. This is



Source: Adopted from Pearce (2001).
X axis is the decline in quantity and quality of water; y axis is the monetary costs and benefits

Fig. 1. Impacts of market and government failure and population growth on water use.

not only because many of the water resources are public goods in nature, and hence do not have readily available monetary values attached to them, but also because their value is more complex compared to private goods. This complexity arises from the fact that the value of water resources are composed of both use and non-use values. Capturing the TEVof water resources is crucial to policy and management decisions because they can guide resource allocations among water resource conservation and sustainable management and other socially valuable endeavours, as well as within water resources, thus enabling society to allocate its scarce economic and environmental resources efficiently. Establishing the TEV would also assist in the design of economic incentives and institutional arrangements, and help to identify potential gainers and losers from current depletion and degradation of water resources (Drucker et al., 2001).

Various economic methods have been developed to capture the TEV of environmental resources. Table 1 lists the main economic methods that can be used to estimate the values of water resources. The advantages and disadvantages of each of the methods, along with their uses in capturing the value of water resources, is the subject of the subsequent two sections.

3. Revealed preference methods

Revealed preference methods, also known as indirect valuation methods, look for related or surrogate markets in which the environmental good is implicitly traded (i.e., if it is one of the many components of a good that is purchased by the consumer; Lancaster, 1966). Information derived from observed behaviour in the surrogate markets is used to estimate willingness to pay (WTP), which represents individual's valuation of, or the benefits derived from, the environmental resource. Two such methods prevalent in the environmental economics literature are the hedonic pricing and the travel cost methods. These methods are suitable for valuing those water resources that are marketed indirectly and are thus only able to estimate their use (direct and indirect) values.

3.1. Hedonic pricing method

The hedonic pricing method (HPM) is based on Lancaster's characteristics theory of value (Lancaster, 1966), which states that any good can be described as a bundle of characteristics and the levels these take, and that the price of the good depends on these characteristics and their respective levels. It is commonly applied to variations in housing prices that reflect the value of local environmental resources. The price of a house will reflect its relevant characteristics i.e., number of bedrooms, number of bathrooms, size, schools in the neighbourhood, level of crime, etc., in addition to the local environmental resources such as ambient air quality, noise levels, aesthetic views, water quantity or quantity.

It follows that an implicit price exists for each of the characteristics and an implicit marginal WTP, which represents an individual's valuation of the incremental unit of the environmental resource can be identified statistically. A limitation of the HPM is that it only measures direct use values of water resources as perceived by the consumers' of the good in which it is implicitly traded. Services such as flood control, water quality improvement, habitat provision for species, and groundwater recharge may provide values that benefit individuals far away, beyond the consumers of the good, which the HPM is unable to capture (Boyer and Polasky, 2004).

The HPM was developed by Griliches (1971) to estimate the value of quality change in consumer goods. The earliest examples of HPM applied to irrigation water valuation are by Milliman (1959) and Hartman and Anderson (1962). The relationship between land prices and surface and groundwater access (both in quantity and quality terms) has been studied in a hedonic framework by Miranowski and Hammes (1984), Gardner and Barrows (1985), Ervin and Mill (1985) and King and Sinden (1988). On average these studies place a value of €0.43 per m^3 of clean water. Torell et al. (1990) compared sales of irrigated and non-irrigated lands to estimate the value of groundwater in the southern High Plains in the US. Results indicate the water value component or irrigated farm sale transactions ranged from 30% to 60% of the farm sale prices, depending on State. Faux and Perry (1999) applied HPM to agricultural land sales in Malheaur County, Oregon, to reveal the implicit market price of water in irrigation. The value of irrigation water in this location is estimated at €7.7 for an acre-foot (0.41 ha) on the least productive land irrigated, and up to €37.5 per acre-foot (0.41 ha) on the most productive land.

HPM has also been applied to wetland valuation. Mahan et al. (2000) used data on more than 14000 home sales in Portland, Oregon metropolitan area to estimate the effect of proximity to wetlands on property values. They found that a decrease in the distance to the nearest wetland by 1000 ft (304.8 m) from an initial distance of

1 mile resulted in an increase in property value of €371.6. Doss and Taff (1996) found similar results using data from Ramsey County, Minnesota. They also found a preference for open-water wetlands and scrub-shrub wetland types over emergent vegetation and forested wetlands.

3.2. Travel cost method

The travel cost method (TCM) is used to estimate use values associated with ecosystems or sites (such as forests, wetlands, parks, and beaches) that are used for recreation to which people travel for hunting, fishing, hiking, or watching wildlife. The basic premise of the TCM is that the time and travel cost expenses that people incur to visit a site represent the "price" of access to the site. Thus, peoples' WTP to visit the site can be estimated based on the number of trips that they make at different travel costs. This is analogous to estimating peoples' WTP for a marketed good based on the quantity demanded at different prices. The TCM encompasses a variety of models, ranging from the simple single-site TCM to regional and generalised models that incorporate quality indices and account for substitute sites (CGER, 1997).

The method can be used to estimate the economic benefits or costs resulting from changes in access costs for a recreational site, elimination of an existing recreational site, addition of a new recreational site and changes in environmental quality at a recreational site. There are however several limitations to TCM. Defining and measuring the opportunity cost of time is complicated since there is no strong consensus on appropriate measure. Substitute sites are only taken into account in the random utility approach to TCM, which uses information on all possible sites that a visitor might select, their quality characteristics, and the travel costs to each site. This approach yields information on the value of characteristics in addition to the value of the site as a whole. TCM however can only be used to value goods consumed in situ and, similar to HPM, it cannot capture the non-use values of environmental resources.

The TCM was first proposed by Hotelling (1931) and subsequently developed by Clawson (1959) and Clawson and Knetsch (1966). Such models have been employed to measure the welfare effects to changes in water quality of recreational sites (e.g. Caulkins et al., 1986; Smith and Desvousges, 1986; Bockstael et al., 1987). Bell and Leeworthy (1990) investigate the tourists' recreational demand for saltwater beach days in Florida and find the daily consumer valuation to be nearly €29. Cooper and Loomis (1991) estimated the value of 7 wildlife reserves in the San Joaquin Valley in California at €47.23 per waterfowl hunter per season. The total consumer surplus from hunting is estimated at €2.6 million annually. Choe et al. (1996) employed the TCM to estimate the local community's valuation of surface water quality improvements in the rivers and seawater in Davao, Philippines. They find that the values are quite low, both in absolute terms and as a percentage of household income, suggesting that water pollution control is simply not a high priority for local residents. Bowker et al. (1996) used the TCM to study the value of guided white-water rafting on Chatooga and Nantahala Rivers in southern US. They estimate a value between €75.9 and €243.7 per visitor per trip, depending on modelling assumptions and river quality. Yapping (1998) employed the TCM in China to estimate the value of improving the water quality of East Lake in Wuhan. The results reveal that lake users are WTP significant amounts for the use of the lake and its facilities, thus offsetting some of the cost of maintaining water quality for recreation. Total value of an improvement in water quality to boatable level is estimated at €21.4 million, whereas this value is as high as €54.8 million for swimmable quality level and €97.4 for drinkable quality level. Loomis (2002) employed the TCM to estimate the recreation use values from hypothetically removing dams and restoring free-flowing rivers, with an application to the Lower Snake River in Washington, US. He found that if the four dams are removed and the 225 km river is restored, the value of river recreation is as high as €264.2 million, which exceeds the loss of reservoir recreation, but is about €51.1 million less than the total costs of the dam removal alternative.

3.3. Other revealed preference methods

In addition to the HPM and the TCM, there are also other revealed preference methods that are not as widely used in the context of environmental resources valuation; however they can be useful in certain situations. These are described below.

3.3.1. Replacement cost method

This method values the costs of replacing damaged assets, including environmental assets, by assuming these costs are estimates of the benefit flows from avertive behaviour. This method assumes that the damage is measurable and that the value of the environmental asset is no greater than the replacement cost. It also assumes that there are no secondary benefits arising from the expenditures on environmental protection. This method is particularly applicable where there is a standard that must be met, such as a certain level of water quality (Markandya et al., 2002).

3.3.2. The avertive expenditures method

This method is based on the household production function theory of consumer behaviour. The household produces consumption goods using various inputs, some of which are subject to degradation by pollution. In the context of water resources, households may respond to increased degradation of these inputs in various ways that are generally referred to as averting or defensive behaviours so as to avoid the adverse impacts of water contaminants. This includes buying non-durables (e.g., bottled water), making expenditures on liming to reduce water acidification, and changing behaviour to avoid exposure to the contaminant (e.g., boiling water for cooking and drinking or reducing the frequency or length of showers if a volatile organic chemicals were present).

There are however important limitations to this method. Individuals may undertake more than one form of averting behaviour in response to an environmental change and the averting behaviour may have other beneficial effects that are not considered explicitly (e.g., 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, e.g. a water filter is either purchased or not. Generally, the averting expenditures does not measure all the costs related to pollution that affect household utility and are therefore only able to provide a lower bound estimate of the true cost of increased pollution.

Abdalla (1994) discussed five studies that have used this method to measure household-level costs associated with groundwater contamination. Annual costs from the household averting expenditure studies reviewed generally ranged from €106.5 to €281.3 per household. Annual costs for expenditure on bottled water to address organic contamination alone ranged from €27.3 to €281.3 per year. McConnell and Rosado (2000) have more recently estimated the non-marginal benefits from improvements in drinking water quality using defensive inputs in Guarapari and Grande Vitoria, Espirito State, Brazil. Um et al. (2002) estimated improved drinking water quality in Pusan, Korea and find that marginal WTP estimation results for a small reduction, $10 \mathrm{mgl}^{-1}$ of suspended solid concentration in tap water from $335 \mathrm{mgl}^{-1}$ range from €0.60–1.50 per month per household.

3.3.3. Production function approach

This approach can be used to value non-marketed goods and services that serve as an input to the production of marketed goods. The approach relates the output of particular marketed goods or services (e.g. agricultural production, timber, fish catch) to the inputs necessary to produce them. These include marketed inputs such as labour, capital, and land, as well as non-marketed goods and services such as soil stability, air quality, or water quality and quantity. Thus, the implicit value of water can also be calculated by measuring the contribution of water to the profit in cases where water is an important component of a production process and the producer's cost structure is known. If water supply is unrestricted, a producer will continue to use units of water up to the point where the contribution to profit of the last unit is just equal to its cost to the firm. Even if water is “free”, there will be costs to the producer associated with water use (including pumping and delivery costs). If water supply is restricted (for example, by quotas or water rights), the producers may cease use of water before the equality is met. The level of water use at varying costs to the producer defines a “derived” demand relationship, since the demand for the water is derived from the demand for the output of the producer (e.g., agricultural commodities).

An example of this method is Acharya and Barbier (2002), which uses the production function approach to estimate the value of groundwater recharge in the Hadeja-Jama'Are floodplain, Northern Nigeria. They find the value of the recharge function is €11104 perday for the wetlands and the average welfare change for a 1-m change in water levels is approximately €0.1 perhousehold.

3.3.4. Net factor income

The net factor income estimates changes in producer surplus (i.e., the monetary measure of net benefit to a firm of producing a good) by subtracting the costs of other inputs in production from total revenue, and ascribes the remaining surplus as the value of the environmental input (Brander et al., 2006). Thus, for example, the economic benefits of improved water quality can be measured by the increased revenues from greater agricultural productivity when water quality is increased. Alternatively, water quality affects the costs of purifying municipal drinking water hence economic benefits can be measured by the decreased costs of providing clean drinking water.

3.3.5. Cost-of-illness (COI) method

Another approach is the cost-of-illness (COI) method in which the benefits of pollution reduction are measured by estimating the possible savings in direct out-of-pocket expenses resulting from illness (e.g., medicine, doctor and hospital bills) and opportunity costs (e.g., lost earnings associated with the sickness). Two important limitations of this approach is that it does not consider the actual disutility of those who are ill, nor does it account for the defensive or averting expenditures that individuals may have taken to protect themselves (CGER, 1997).

3.3.6. *Market prices*

Market prices are used to value the costs/benefits associated with changes in quality and quantity of environmental goods that are traded in perfectly functioning markets. They are generally used with other revealed preference methods (e.g. cost-of-illness approach, replacement costs approach), which assume that market price represents the opportunity cost of water resources.

4. Stated preference methods

Stated preference methods (SPM), also called direct valuation methods, have been developed to solve the problem of valuing those environmental resources that are not traded in any market, including surrogate ones. In addition to their ability to estimate use values of any environmental good, the most important feature of these survey-based methods is that they can estimate the non-use values, enabling estimation of each component of TEV. Since many of the outputs, functions and services that water resources generate are not traded in the markets, SPM can be used to determine the value of their economic benefits.

4.1. *Contingent valuation method*

The purpose of the contingent valuation method (CVM) is to elicit individuals' preferences, in monetary terms, for changes in the quantity or quality of non-market environmental resources. With CVM, valuation is dependent or ‘contingent’ upon a hypothetical situation or scenario whereby a sample of the population is interviewed and individuals are asked to state their maximum WTP (or minimum willingness to accept (WTA) compensation) for an increase, or decrease, in the level of environmental quantity or quality. To conduct a CVM, special attention needs to be paid to the design and implementation of the survey. Focus groups, consultations with relevant experts, and pre-testing of the survey are important pre-requisites. Decisions need to be taken regarding how to conduct the interviews (in-person, via mail or via telephone surveys); what the most appropriate payment bid vehicle is (e.g., an increase in annual taxes, a single-one-off payment, a contribution to a conservation fund, among others, see Champ et al. (2002) for more on this); as well as the WTP elicitation format (see Hanemann, 1994; Bateman et al., 2003). Ultimately, the mean WTP bids that have been obtained from the sample can then be extrapolated across the population to obtain the aggregate WTP or value of the environmental resource (Mitchell and Carson, 1989).

With regard to water resource applications, CVM is useful for examining direct use values such as recreational fishing and hunting, and indirect use values such as improved water quality. Unlike revealed preference methods, CVM is also able to measure the option use values of water associated with biodiversity, as well as the non-use values. Despite the strengths of CVM regarding its ability to estimate non-use values and evaluate irreversible changes, this method has been criticised for its lack of validity and reliability (Kahneman and Knetsch, 1992; Diamond and Hausman, 1994). This is on account of potential problems including information bias, design bias (starting point bias and vehicle bias), hypothetical bias, yea-saying bias, strategic bias (free-riding), substitute sites and embedding effects (see Appendix A for a detailed description of these biases). To address these, the Blue Ribbon Panel under the auspices of U.S. National Oceanic and Atmospheric Administration (NOAA) (Arrow et al., 1993) have made recommendations regarding best practice guidelines for the design and implementation of contingent valuation studies that will form the basis of natural resource damage litigation actions. To date more than 5000 CVM studies have been conducted in over 100 countries, most of which make reference to the guidelines of the NOAA panel, and a large proportion of CVM studies have examined water quality and quantity issues specifically.

The earliest CVM related to water valuation is a 1969 estimation of net benefits (or “consumer surplus”) for wildlife hunting in the wetlands of the U.S. Pacific western flyway (Hammack and Brown, 1974). Desvousges et al. (1987) estimated the option price bids for the improved recreation resulting from enhanced water quality in the Pennsylvania portion of the Monongahela River. Boyle et al. (1993) conducted a CVM to estimate the WTP values for changes in water flow for white-water rafting in the Grand Canyon. On a larger scale, Carson and Mitchell (1993) evaluated the national water quality benefits from the Clean Water Act by examining the WTP for increased water quality for all rivers in the US. They find that WTP per capita per annum is €118.5 for water quality improvement from an unusable to a boatable level, and €175.6 for further improvement to a swimmable level. The incremental value of improvement from boatable to fishable is €32.4 and from fishable to swimmable, €23.9. The CVM has also been applied to evaluate water supply issues, such as in Briscoe (1990) who examined drinking water supply in Brazil. Choe et al. (1996) compare the results from CVM and TCM to evaluate surface water quality improvements in the rivers and sea-water near the community of Davao, Philippines. Their CV results indicate that household WTP for environmental amenities such as improved water quality is low (€0.9 per month). Cooper et al. (2004) examined the use and non-use benefits associated with three nested schemes for improving water quality in a lake in Norwich, UK, and find that these range from €18–36.8 depending on the scheme. Furthermore, a large number of CVM studies focus on the use and non-use values of wetlands. This is because of the substantial local and global indirect and non-use values inherent in this resource (see Crowards and Turner, 1996; Brouwer et al., 2003 for a review). Pate and Loomis (1997) found that WTP for a wetlands improvement program in California, USA, is about €183.3 per household and that this value decreases as the distance from the site increases. Oglethorp and Miliadou (2000) for example find that mean per capita WTP per year for use and non-use values of Lake Kerkini in Greece is €22.5. Finally, Brouwer et al. (2003) used 30 wetland CV studies to conduct a meta-analysis of wetland valuation studies, where a meta-analysis is the statistical analysis of the summary findings of empirical studies (Champ et al., 2002). They find that use values (such as flood control, water generation and water quality attributes) have a stronger influence on WTP than non-use elements such as the biodiversity function of wetlands.

4.2. Choice experiment method

A relatively new addition to the portfolio of SPM, the choice experiment method (CEM), is theoretically grounded in Lancaster's characteristics theory of value (Lancaster, 1966) and based on random utility models (RUMs) (Luce, 1959; McFadden, 1974). RUMs are discrete choice econometric models, which assume that the respondent has a perfect discrimination capability, whereas the analyst has incomplete information and must therefore take account of uncertainty (see Manski, 1977 for more information). A choice experiment is a highly 'structured method of data generation' (Hanley et al., 1998), relying on carefully designed tasks or "experiments" to reveal the factors that influence choice. The environmental resource is defined in terms of its attributes and levels these attributes would take with and without sustainable management of the resource. For example one attribute that can be used to describe the quality of coastal waters is bathing water quality. The levels of this attribute could be high, medium, and low. One of the attributes is a monetary one, which enables estimation of WTP. Profiles of the resource in terms of its attributes and attribute levels is constructed using experimental design theory, a statistical design theory which combines the level of attributes into different scenarios to be presented to respondents. Two or three alternative profiles are then assembled in choice sets and presented to respondents, who are asked to state their preference (Hanley et al., 1998; Bateman et al., 2003).

Similar to CVM, CEM can estimate economic values for any environmental resource, and can be used to estimate non-use as well as use values. CEM however, enables estimation not only of the value of the environmental resource as a whole, but also of the implicit value of its attributes, their implied ranking and the value of changing more than one attribute at once (Hanley et al., 1998; Bateman et al., 2003). Another advantage of CEM over CVM is that respondents are more familiar with the choice rather than the payment approach. Moreover, CEM can solve for some of the biases that are present in CVM; the strategic bias is minimised in the CEM since the prices of the resources are already defined in the choice sets. Further, yea-saying bias (or warm glow effect) is also eliminated because the choice approach does not allow for the respondent to state a value for the resource even if they do not value it. Finally, the risk of insensitivity to scope (or embedding effect) in CEM is reduced. If the choice sets offered to respondents are complete and carefully designed, the respondent would not mistake the scale of the resource or its attributes for something else that it could be embedded in (Bateman et al., 2003).

Although CEM has been applied to valuation of environmental resources only in the past decade, there have been some noteworthy applications of this method to water resources valuation. Morrison et al. (1999) employed a choice experiment to estimate the non-use values of environmental, as well as social and economic attributes of the Macquarie Marshes wetland in Australia. They find that the Australian public is WTP substantial amounts (€13.3 to €60.9 per household depending on the model and management scenario employed) in order to increase the wetland area, to improve the biodiversity found in the wetland, and to increase irrigation related employment, revealing the

Table 2
Advantages and disadvantages of economic valuation methods

Method	Advantages	Disadvantages
Hedonic pricing method (HPM)	Based on observable and readily available data from actual behaviour and choices.	Difficulty in detecting small effects of environmental quality factors on property prices. Connection between implicit prices and value measures is technically complex and sometimes empirically unobtainable. Ex post valuation. (i.e. conducted after the change in environmental quality or quantity has occurred). Does not measure non-use values.
Travel cost method (TCM)	Based on observable data from actual behaviour and choices. Relatively inexpensive.	Need for easily observable behaviour. Limited to in situ resource use situations including travel. Limited to assessment of the current situation. Possible sample selection problems. Ex post valuation. Does not measure non-use values.
Replacement cost method	Based on observable data from actual behaviour and choices. Relatively inexpensive. Provides a lower bound WTP if certain assumptions are met.	Need for easily observable behaviour on averting behaviours or expenditures. Estimates do not capture full losses from environmental degradation. Several key assumptions must be met to obtain reliable estimates. Limited to assessment of current situation. Ex post valuation. Does not measure non-use values.
Production function method	Based on observable data from firms using water as an input. Firmly grounded in microeconomic theory. Relatively inexpensive.	Understates WTP. Ex post valuation. Does not measure non-use values. Omits the disutility associated with illness.
Cost-of-illness method	Relatively inexpensive.	Understates WTP because it overlooks averting costs. Limited to assessment of the current situation. Ex post valuation.
Market prices	Based on observable data from actual choices in markets or other negotiated exchanges.	Does not provide total values (including non-use values). Limited to assessment of current situation. Potential for market distortions to bias values.
Contingent valuation method (CVM)	It can be used to measure the value of anything without need for observable behaviour (data). It can measure non-use values. Technique is not generally difficult to understand. Enables ex ante and ex post valuation.	Subject to various biases (e.g., interviewing bias, starting point bias, non-response bias, strategic bias, yea-saying bias, insensitivity to scope or embedding bias, payment vehicle bias, information bias, hypothetical bias). Expensive due to the need for thorough survey development and pre-testing. Controversial for non-use value applications.
Choice experiment method (CEM)	It can be used to measure the value of any environmental resource without need for observable behaviour (data), as well as the values of their multiple attributes. It can measure non-use values. Eliminates several biases of CVM. Enables ex ante and ex post valuation.	Technique can be difficult to understand. Expensive due to the need for thorough survey development and pre-testing. Controversial for non-use value applications.

Adopted from CGER (1997).

conflict between uses of this water resource. Carlsson et al. (2003) used the CEM to estimate both non-use and use values of the Staffanstorp wetland in Sweden, to help design wetland conservation and management programmes that yield the highest public benefits. They rank several wetland attributes (e.g., biodiversity, crayfish, surrounding vegetation, fish, walking facilities) according to public's valuation, and the results reveal that the fenced waterline and introduction of crayfish decrease social welfare whereas biodiversity and walking facilities increases it. Carlsson et al. state that a natural extension to this study is the estimation of marginal costs of providing the different attributes of a wetland, so that a CBA can be carried out to construct a socially efficient design of the wetland. Similarly, Othman et al. (2004) employed a CE to assist decision makers in determining the optimal management strategy for the Matang Mangrove Wetlands in Perak State in Malaysia. They estimated the values for environmental attributes (e.g., the area of environmental forest protected, the number of bird species protected and the recreation use of the area) as well as the value of a social attribute (i.e., the employment of local people in wetland-based extractive industries). Othman et al. found that the households are WTP −€2.7 to €3 for management of the wetland, depending on the management scenario and estimation method employed. The negative WTP implies that households experience negative utility from reduced employment and hence demand compensation. Moreover, this study also reveals that CEM can be employed successfully in a developing country.

The CEM has also been applied to estimate the value of improved water quality and water services. Willis et al. (2002) employed the CEM to investigate the preference tradeoffs of water company customers between increasing security of water supply and the potential impacts of this on biodiversity in the local wetland sites and river flows in Sussex, UK. They find that consumers' valuation of increasing security of water supply is insignificant; however they value conservation of wetland habitats and river flows, with WTP values of €2.1 for a unit increase in the former and €6.3 for the latter. Abou-Ali and Carlsson (2004) investigated the welfare effects of improved health status through increased water quality in Cairo, Egypt. They find that the mean WTP to improve health problems caused by poor water quality was €1.1 per month per household. The estimated WTP, however, is fairly low compared with the costs of a program that would achieve these improvements. Finally, Hensher et al. (2004) employed the CEM to estimate the Australian consumers' WTP to avoid interruptions in water service and overflows of wastewater, differentiated by the frequency, timing and duration of these events. They find that consumers are WTP €71.7 to reduce the frequency of interruptions when they face one interruption in ten years, while the average WTP is only €6.1 when customers face monthly interruptions. Consumers' WTP to reduce the length of water services interruptions ranges from €34.7 for interruptions of 1h to €2.8 for interruptions of 24h. The authors state that this is because customers faced with more interruptions in water supply are more likely to take actions to reducing their impact, such as storing water, and because psychologically, a reduction of frequency of water supply interruptions from 12 to 11 seems less important than a reduction from 2 to 1. For wastewater services, consumers' WTP for reduction of frequency of overflows range from €134.4 for one flow in ten years to €49.3 for two flows a year. These values of water consumption and disposal services are crucial information for establishing service levels and tariffs, and for agencies to find cost effective ways of delivering services at prices that customers consider value for money. The advantages and disadvantages of the valuation methods described in Sections 3 and 4 are summarised in Table 2.

5. Case study on Cheimaditida wetland

5.1. Background

During the last century, several lakes in Greece were drained to generate hydroelectric power or to expand agricultural land, resulting in biodiversity loss. A drought between 1987 and 1993 diminished both water quantity and quality in rivers and lakes, causing chronic water shortages in the largest cities and limiting growth of natural vegetation. Further, Greece lost 63% of its wetlands between 1920 and 1991 (Barbier et al., 1997). As an EU member state and a signatory to the Ramsar convention, Greece is obliged to conserve, sustainably manage and improve the conditions of its remaining wetlands and other water resources. The aim of this case study is to estimate the non-use values of the Cheimaditida wetland in Greece using the CVM method which is one of only two valuation techniques able to estimate non-use values of environmental resources. The CVM was thought simpler to implement than the choice experiment. These non-use values can be combined with use values of the Cheimaditida wetland (see Psychoudakis et al., 2005 for estimates) to obtain its TEV which can then be used for CBA of management strategies for this wetland.

The Cheimaditida wetland is located 40 km southeast of Florina in Northwest Greece. The wetland contains one of the few extant freshwater lakes in the country and generates several important ecological functions, such as providing refuge to a great diversity of fauna and flora. Many of the species, such as the Dalmatian pelican and the lesser kestrel are under protection, and 11 of the mammals, 7 amphibians, 7 reptiles and 8 fish are listed in Annex II and IV of the EU Habitats

Directive (92/43/EEC). The wetland also supports six habitat types listed in Annex I of the EU Habitats Directive (92/43/EEC), one of which is a *priority natural habitat* under Article 1. The main economic activities in the area are irrigation and fertiliser-intensive agriculture and fishing. Both are adversely affecting water quantity and quality of the wetland, thereby reducing its ability to maintain biodiversity and other life support functions (Seferlis, 2004).

5.2. Methodology

Given that the purpose of this study is to elicit the non-use values associated with the wetland, we are restricted to using either a CVM or a choice experiment method. As discussed in Section 4.1, CVM is a survey-based valuation method and can be used to elicit non-use values of an environmental good or service. In this study, respondents were asked to state their valuation (WTP) for an improvement in the quantity and quality of the environment. Based on expert consultations, literature review and focus groups, four characteristics, expected to generate non-use values, were selected. These were (a) biodiversity, (b) open water surface area, (c) inherent research and educational values that can be extracted from the wetland and (d) values associated with environmentally friendly employment opportunities. To date the majority of the non-use values associated with wetlands that were estimated were attributed to biodiversity (see Brouwer et al. (2003) and Brander et al. (2006) for a list of these valuation studies). This is because many species of animals, plants and their habitats depend on wetlands for their continued existence. Open water surface-area and the natural vistas associated with them are expected to create non-use values through feelings of serenity and tranquillity. Further, larger open water surface-areas provide water quantity required for sustaining the wetland's biodiversity. Research and educational extraction from the wetland was expected to contribute to non-use values associated with cultural heritage and scientific knowledge. Finally, re-training of locals in environmentally friendly occupations were expected to generate non-use values to the wider Greek public as non-use values may be derived from economic and social factors in addition to environmental factors (Portney, 1994). These are non-use values because only the preferences of non-users were elicited (i.e., from the general public, rather than tourists visiting the area or locals in the vicinity). Other examples in the literature that estimate similar non-use values include Morrison et al., 1999; Bennett et al., 2004; Colombo et al., 2005; Bergmann et al., 2006.

Using these characteristics, and after extensive consultations with scientific experts from EKBY, the Greek Biotope and Wetland Centre, a business-as-usual (BAU) scenario and two management scenarios were designed:

Scenario A: no management, BAU. Biodiversity deteriorates to a low level i.e., a 10% decline in population and size of habitats. Open water surface declines by approximately 3–10%. Educational and research extraction potential deteriorates due to lack of investment in existing facilities, and 65 locals would become unemployed because the wetland will no longer be able to support agriculture and fishing.

Scenario B: managing the wetland to maintain current conditions. Biodiversity would be maintained at the current level. Open water surface area is maintained at the current level of 20% (with the remaining 80% covered by reed beds). Educational and research extraction is maintained, and 75 locals would be re-trained in environmentally friendly employment such as arid-crop production and eco-tourism.

Scenario C: managing the wetland to improve current conditions. Biodiversity levels would increase by 10%, and open water surface area would increase to 60%. Educational and research extraction would be increased by funding better facilities, i.e., larger information centre with microscopes, binoculars, books, and information leaflets. Finally, 150 locals would be re-trained in environmentally friendly employment such as arid-crop production and eco-tourism.

The payment vehicle used in the CV study was a one-off payment in terms of an increase in taxes for the year 2005–2006, which would be channelled to a 'Cheimaditida Wetland Management Fund' and managed by a trustworthy and independent body. Taxation was preferred over voluntary donations since respondents may have the incentive to free-ride with the latter (Whitehead, 2006). Due to limited time and resources, an open-ended approach was used to elicit WTP values for the two different wetland management scenarios. An open-ended (OE) elicitation format asks the respondent "How much are you willing to pay to…", rather than a close-ended format which asks "Are you willing to pay €X to…". Although the OE format is not the approach recommended by the NOAA blue ribbon panel, Langford et al., 1998 conclude that useful information can still be obtained from OE elicitation studies. The respondents were asked whether they are WTP to move from scenario A to B (i.e., *are* you willing to contribute to the wetland management fund?) and if yes, to state their maximum WTP (i.e., how *much* are you WTP?, to elicit their valuation). Similarly the respondents were asked whether or not they are WTP to move from scenario A to C and if yes, to state their maximum WTP. While stating their WTP values the respondents were reminded of their budget constraints, household expenses, as well as other substitute sites in Greece and the payments they make or would like to make for other environmental goods or services.

5.3. Results and policy implications

The contingent valuation survey was conducted in January 2005 on 122 respondents. This is a relatively small sample size (primarily due to budget constraints) but given that the aim of the valuation exercise was to determine whether the Greek public attach non-use values to this wetland, and if so, to identify what the determinants of value might be, the sample size is sufficient. The respondents were randomly selected from city centres of Greece's two largest cities, Athens and Thessaloniki (Table 3). In addition to the WTP questions, information was collected on respondents' social, demographic and economic status. Compared to the Greek national average, a statistically higher proportion of the respondents was female and employed, had a university education, and children, and was located in urban areas. This was likely due to the fact that our sample was collected from Athens and Thessaloniki, the largest and wealthiest locations in Greece. Overall, 84.4% of the respondents indicated a positive WTP to move from scenario A to B, with a mean WTP of €22.3 (median €10) and 83.6% indicated a positive WTP to move from A to C, with a mean WTP of €34.9 (median €20).

As reported in Table 4, 18 respondents indicated a zero WTP to move from scenario A to B, and 19 respondents indicated a zero WTP to move from scenario A to C. To discriminate true zero WTP values from protest responses (i.e., when respondents do value the resource, but state a zero value because they object to an element of the survey), four follow-up questions were asked (Federal Guidelines, 1983; Haab, 1999). These were: a) I do not care about wetlands; b) I cannot afford to contribute to the fund; c) The government is responsible for wetland management; and d) I do not believe the funds will be used appropriately. The first two categories were classified as true zero values whereas the remaining two categories were considered protest responses, since they do not reflect the respondents' true valuation of the non-use values of the wetland. Only 3 respondents (i.e. 2.5% of the sample) had true zero WTP, whereas 15% of the responses were protest votes, constituting a substantial portion of zero bids (Mitchell and Carson, 1989).

Table 3
Means and standard deviations (in parenthesis) of survey respondent characteristics and Greek national averages

Parameters	All responses [a]	Greek average [b]
WTP to move from scenario A to B (%)	84.4	–
Mean WTP (in €, scenario A to B)	22.3 (33.7)	–
Median WTP (in €, scenario A to B)	10	–
WTP to move from scenario A to C (%)	83.6	–
Mean WTP (in €, Scenario A to C)	34.9 (43.5)	–
Median WTP (in €, Scenario A to C)	20	–
Heard of the wetland (%)	16.4	–
Visited the wetland (%)	3.3	–
Age	33.5 (9.7)	40.2 [c]
Gender (% female)	57	50.5
Education (% with university degree and above)	62	18
Household size	2.8 (1.4)	3.5
Children (% with children)	38.1	30.4
Employed (% with full-time employment)	75.2	47
Household income (net, in € per month)	2356.9 (1317.4)	1358
Tenure (% own property)	69.4	80
Urban (% located in cities)	72.5	58
Sample size	122	10, 628, 113

[a] Cheimaditida Wetland Management Contingent Valuation Survey, 2005.
[b] National Statistical Service of Greece (NSSG) (2003) www.statistics.gr.
[c] Median age.

In both of the improvement scenarios (i.e., A to B, and A to C), the respondents with positive responses had statistically higher education levels, were more likely to be female, have larger households, and are more likely to own property. Those who protest to the move from A to B were more likely to be in full-time employment and those who protest the move from A to C were less likely to be from urban locations. A higher percentage of those respondents who stated positive responses to move from A to C had heard of the site. In both cases, however, those who visited the site were more likely to be protesters. This might be because respondents, who had visited the site in the past for free, might find it unacceptable to make a payment for these wetland goods and services, independently of the value they attached to the site. If these social and economic variables influence the WTP for improvement of the

Table 4
Means and standard deviations (in parenthesis) of respondent characteristics by group

Parameters	Improvement from scenario A to B		Improvement from scenario A to C	
	Protesters	Positive responses	Protesters	Positive responses
Mean WTP	...	26.7 (35.3)	...	41.7 (44.5)
Median WTP	...	15	...	30
Heard	16.7	15.7	10.5	16.7
Visited	16.7	1	10.5	2
Age	36.2 (10.5)	33.2 (9.5)	35.6 (11)	33.2 (9.4)
Gender	27.8	62.7	42.1	59.8
Education	52.9	64.7	44.4	65.7
Household size	2.2 (1.4)	2.8 (1.3)	2.3 (1.5)	2.8 (1.3)
Children	35.3	39.4	27.8	40.4
Employed	94.1	72.5	72.2	76.5
Income	2029.1 (898.8)	2407.5 (1378.6)	2034.1 (1040.1)	2387 (1334.1)
Tenure	58.9	70.6	50	73.5
Urban	76.5	73.3	55.6	75.2
Sample size	18	102	19	102

Cheimaditida Wetland Management Contingent Valuation Survey, 2005.

wetland, we can expect that the final estimates obtained from the sub-sample of non-protesters are likely to be affected by sample selection bias (Strazzera et al., 2003).

Table 5 reports the results of the parameter estimates of the best fitting models for the two management scenarios. Two steps were involved in the decision-making process. First, was the individual's decision on whether to contribute to the fund (i.e. to participate) or not (i.e. to protest). Second, was the individual's decision regarding their WTP (i.e., valuation). Due to this two-step process in the question format and the large proportion of protesters which may be affected by sample selection bias, a Heckman 2-step sample selection model was estimated for both management scenarios. In the first improvement scenario (A to B), there was evidence of sample selection bias (as indicated by the inverse Mill's ratio (λ) which was significantly different from zero (Maddala, 1983). Regression of the estimated inverse Mills' ratio against the parameters of the valuation equation tests for collinearity, and produced an R^2 value of 0.10 which indicated an insignificant level of correlation. Thus the 2-step model was appropriate for estimating the participation and valuation decisions for improvements from scenario A to B (Strazzera et al., 2003). As can be seen from the contribution equation in the first column in Table 5, females, those with university degrees, larger households, and respondents with higher income levels, were more likely to choose to contribute to the wetland management fund. In contrast those with children were less likely to contribute to the fund. This can be explained by the fact that those households with

Table 5
Probit, OLS and Heckman's 2-step sample selection models

Parameters	Improvement from scenario A to B	Improvement from scenario A to C
	Contribution model [coefficient (S.E.)]	
	Heckman's 2-Step (Probit)	Probit
Constant	0.28 (0.92)	−0.4 (0.9)
Heard	9.72 (148298)	585.3 (74899.7)
Visited	−12.8 (148298)	−586.1 (74899.7)
Age	−0.12 (−0.22)	−0.02 (0.02)
Gender	1.9 (0.56)***	0.43 (0.3)*
Education	0.98 (0.42)**	0.40 (0.35)
Household size	0.36 (0.58)**	0.2 (0.15)
Children	−0.36 (0.19)**	−0.19 (0.16)
Employed	−0.48 (0.46)	0.56 (0.40)*
Income	0.002 (0.0008)**	−0.003 (0.02)
Tenure	−0.49 (0.42)	0.26 (0.39)
Urban	0.003 (0.009)	0.58 (0.3)*
Sample size	120	88
% Correctly predicted	90	121
Log likelihood	−27.3	−36.6
Significance level	0.00	0.00
	Valuation model [coefficient (S.E.)]	
	Heckman's 2-Step (OLS)	OLS
Constant	1.08 (0.60)**	2.88 (0.41)***
Heard	0.41 (0.47)	0.22 (0.34)
Visited	−1.56 (1.66)	1.42 (0.8)**
Age	−0.001 (0.001)	−0.0008 (0.001)
Gender	1.05 (0.43)***	0.22 (0.24)
Education	0.48 (0.32)*	0.16 (0.24)
Household size	0.13 (0.1)	0.06 (0.07)
Children	−0.13 (0.1)	−0.06 (0.07)
Employed	0.1 (0.32)	−0.19 (0.26)
Income	0.001 (0.0008)*	0.001 (0.0005)***
Tenure	−0.03 (0.3)	0.27 (0.25)
Urban	−0.002 (0.001)	0.0001 (0.001)
Sample size	102	102
λ	1.74 (0.89)***	–
Adjusted R^2	–	0.03

Cheimaditida Wetland Management Contingent Valuation Survey, 2005.
***1% significance level, **5% significance level, *10% significance level with two-tailed tests.

children may have more binding budget constraints. In accordance with economic theory, the results from the valuation equation reveal that the amount that respondents were WTP increases with education and income. In addition females were more likely to attach higher values to non-use values of wetlands. Previous findings from CV studies indicate that the impact of gender on WTP is mixed (see Bord and O'Connor, 1997; Brown and Taylor, 2000; Berrens et al., 1997). Stern et al. (1993) argue that women are more attentive than men to links between the environment and the things they value.

In the second improvement scenario (A to C), the inverse Mill's ratio, λ, was not significantly different from zero, thus a two-part model for the two separate stochastic processes (contribution and valuation) was estimated (Strazzera et al., 2003). The contribution equation was estimated with a Probit model. This is a model whereby the choice probability P_i is related to explanatory factors in such a way that the probability remains in the [0,1] interval. It is therefore suitable for estimating the determinants of the binary choice of whether or not to contribute to pay for the sustainable management of the wetland. The valuation equation was estimated via ordinary least squares (OLS). The results from the former (contribution) model indicate that females, those in full-time employment, and those located in urban areas were more likely to decide to contribute to the wetland management fund. The results from the valuation model reveal that those who visited the wetland and those with higher incomes were more likely to attach higher values to the non-use values of the wetland. The fact that there was sample selection bias for the first improvement scenario (A to B) and no such bias for the second improvement scenario (A to C) suggests that the sample in the first scenario failed to be representative of the whole population (and was therefore corrected for), whereas the sample for the second improvement scenario was representative of the whole population. Thus, the Greek public would prefer a greater improvement in management of the wetland to a smaller one.

Overall, the results of this contingent valuation case study indicate that the Greek public attaches positive and significant non-use values to the Cheimaditida wetland, and that the impacts of the social, demographic and economic characteristics of respondents on their contribution and valuation conform to economic theory. These results assert that CVM can produce valid non-market estimates of non-use value. These non-use values can be combined with direct and indirect use values of the Cheimaditida wetland to estimate its TEV, which can provide policy makers with the necessary economic information to carry out a CBA, and thus to ensure the sustainable and efficient management of the Cheimaditida wetland.

6. Conclusions

Values of environmental resources such as water are not straightforward to assess due to the public good nature of this resource. This paper presents a non-technical introduction to the economic valuation techniques that can be used to capture the total economic value (TEV) of changes in the quantity and quality of environmental resources, with a specific focus on water. Capturing the TEV of water resources is an integral part in the design of economic incentives and institutional arrangements that can ensure their sustainable, efficient and equitable allocation. The methodological framework presented here should form an important component of a decision-support system especially for projects such as Euro-limpacs, which is designed to assess the effects of future global change on Europe's freshwater ecosystems.

The paper provides a brief overview of important applications of valuation techniques that have been conducted in this field, enabling the interested reader to refer to these for more information. In addition, it provides an applied example of one of the environmental valuation methods, namely the contingent valuation method, to estimate the non-use values of the Cheimaditida wetland in Greece. The results indicate that the Greek public does attach positive and significant non-use values to the Cheimaditida wetland. Such non-use values can be combined with direct and indirect use values of the Cheimaditida wetland, to estimate the TEV of the wetland. A TEV estimate provides policy makers with the necessary economic information for the construction of sustainable and efficient management strategy of the Cheimaditida wetland. Finally, the results from the case study imply that, given the current mandate under the EU's WFD and the obligations of the Ramsar Conventions, non-use values from other wetlands in Greece, as well as in other European countries, should be included in decision-making processes for the development of efficient and effective strategies for sustainable wetland management. These valuation techniques enable a movement away from a biased calculation of private costs and benefits of a project or policy, to an estimation of the social costs and benefits of an economic activity. CVM and other economic valuation techniques are useful as they quantify how the public perceives the importance of ecosystem health in their locality, nationally and internationally, and illustrate how public participation, which is central to the WFD, can be further incorporated into decision-making processes.

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Appendix A

The possible biases that might arise in a CV study include the following:

- Starting point bias usually arises in bidding games and suggests that the WTP bid is anchored on the first suggested bid price.
- Interviewing bias indicates that the attitude of the surveyor can influence the values given by respondents.
- Non-response bias may arise if those that refuse to answer the survey are not a random part of the population but those with a particular attitude (e.g. strongly against the proposed project).
- Strategic bias occurs when respondents deliberately under- or overstate their WTP. Respondents may understate their WTP if they believe that the actual fees they will pay for provision of the environmental resources will be influenced by their response to the CV question. Conversely, realising that payments expressed in a CV exercise are purely hypothetical, respondents may overstate their true WTP in the hope that this may increase the likelihood of a policy being accepted.
- Yea-saying bias indicates that respondents may express a positive WTP because they feel good about the act of giving for a social good although they believe that the good itself is unimportant.
- Insensitivity to scope or embedding bias implies that WTP is not affected by the scale of the good being offered. If people are first asked for their WTP for one part of an environmental resource and then asked to value the whole resource the amounts stated may be similar.
- Payment vehicle bias indicates that respondents may state different WTP amounts, depending on the specific payment vehicle chosen. Payment vehicles such as a contribution or donation, may lead people to answer in terms of how much they think their fair share contribution is, rather than expressing their actual value for the good.
- Information bias contends that the WTP that an individual expresses in response to a CV question is not a reflection of preferences they held previously but are constructed in the interview procedure.
- Hypothetical bias contends that respondents may be prepared to reveal their true values without strategic bias but are not capable of knowing these values without participating in a market in the first place (Bateman et al., 2003).

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