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May 2007

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MPRA Paper No. 5867, posted 22 Nov 2007 06:09 UTC

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WORKING PAPER SERIES

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Working Paper N. 4

May 2007

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Abstract

This work aims at analysing the value of recreational water uses for the Idro Lake (Lombardy, Northern Italy), which has been experiencing dramatic fluctuations in its levels in recent years, due to excessive productive withdrawal that affected recreational uses. It estimates the economic benefits deriving from recreational uses, by considering the current recreational demand and the hypothetical one obtained by considering an "improved quality" scenario. Through an on-site survey, we built a panel dataset. Following Whitehead *et al.* (2000) and Hanley *et al.* (2003) we get welfare estimates by combining SP and RP responses. The present CS is estimated in €134 per individual, whilst the increase in CS is estimated in €173 per individual. These figures can be confronted with the economic value of competitive uses and with the clean up costs, respectively, to infer some policy indications.

Keywords: Recreational water uses, Water Framework Directive, Poisson and Negative binomial econometric models
JEL classification: D61, Q25, Q26

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1. Introduction

Recreational demand increased considerably in recent years as a result of increasing income, population and tourism facilities (Hanley *et al.*, 2003b; Krutilla, 1967). Several recreational activities rely on the availability of unspoiled natural environments. At the same time, many economic activities have an impact on the quality and availability of natural resources, since they are used as inputs in production processes. Water resources constitute a straightforward example of this kind of trade-off, since water can either be used for agricultural, industrial and hydroelectric uses or be conserved for recreational uses.

The role of economic analysis is acknowledged by scholars and it has been recognized even in recent EU legislation. In all cases where water resources are scarce, an allocative decision should be taken on which uses to favour. This decision should be made on a CBA basis, favouring the uses with the highest WTP (and consequently the highest social value). For this reason, the Water Framework Directive (2000/60/CE; WFD, hereafter) introduces economic analysis in water management planning process to allocate the resource and to justify the economic costs of conservation policies (on a benefit/cost ratio basis). In particular, the WFD requires that the economic analysis should be carried out by the competent authorities (normally the River Basin Agency) to attain allocative efficiency (WATECO, 2002).

The WFD can be considered the first European Directive that explicitly considers the economic analysis as a means to manage efficiently water resources. The reasons to introduce efficiency objectives are twofold. On the one hand, considering the quantity of the resource available, in many situations water is scarce, i.e. it is not able to satisfy the demand of several competitive uses. On the other hand, considering water quality, the productive inputs necessary to put in place the infrastructure needed to cope with environmental standards are scarce as well. In the former case, we refer to the resource opportunity cost; in the latter case we refer to the input opportunity cost (Massarutto, 2005). First, the WFD states that the economic analysis should be introduced at different stages of the planning process. It has to be considered in order to describe the current uses of the resource, thus allowing the planner to identify the different environmental functions and infer the economic value of the resource for the different uses. This analysis should allow the competent authority to allocate efficiently the resource among competitive uses (WATECO, 2002; Massarutto, 2005). Second, the Directive objective is the attainment of the "good status" at 2015 for all European waters, the only exception being the cases where this objective is deemed to costly with respect to the benefits derived from water protection. Obviously the assessment of these benefits is the first step necessary to characterise some protection measures as excessively costly. Finally, the implementation of measures which are not compulsory (i.e. not aimed at reaching the "good status") should be justified on a cost-benefit analysis ground, thus implying a benefit assessment.

This study aims at analysing the value of recreational water uses for the Idro Lake (Lombardy, Northern Italy), which has been experiencing dramatic fluctuations in its levels in recent years, due to excessive productive withdrawals that have affected recreational uses, and has a water quality classified as "sufficient", thus not compliant with the "good status" envisaged in the WFD. This work can be considered as a pilot study to infer the economic value of recreational water uses and, at the same time, assess the economic benefits related with a quality improvement (and consequently to identify measures whose costs are deemed disproportionate). Following the WFD, we carried out an assessment of the recreational water uses considering at the same time their current value and the value of the water quality improvement. This exercise would make possible to infer whether the current allocation of the resource among competiting uses is efficient, by considering the value of alternative uses, and to justify water quality improvement on a cost-benefit analysis ground, by considering the increase of the outdoor recreation demand following this quality increase.

2. *Recreational activities and environmental quality*

In last decades environmental issues have raised greater concern. Environmental quality can be conceived as a normal good, whose demand increases as the income increases (Ruttan, 1971). People's welfare can increase not only through direct consumption of goods and services, but also from the fruition of a clean environment. As prophetically stated by Krutilla (1967), advances in technology can decrease the impact of economic growth on the environment but cannot guarantee the provision of amenities associated with unspoiled natural environment. Economists tried to consider this change in consumer preferences by expanding the notion of economic value associated to natural resources. They started to be interested in assessing Total Economic Value (TEV), defined as the sum of all use and non-use values provided by the ecosystems (Perman *et al.*, 2003). The former refers to the dimension of value associated with a direct fruition of the environment (either direct or indirect, like the option value), whilst the latter refer to the economic value attached to a natural resource or ecosystem which is independent of its actual use (e.g. existence value). In the case of water resources, use values include benefits arising from withdrawal for drinking water, irrigation and industrial and hydroelectric purposes, in-stream uses (e.g. fishing, swimming, boating) and aesthetics for nearby uses (e.g. walking, picnicking, bird watching,…). Non-use values refer the benefits for future generation (bequest values) and from the ones deriving from the knowledge that the ecosystem has been preserved (existence value).

This work focuses on recreational uses associated with water ecosystems. The recreational demand for environmental goods has been increasing since the Second World War, as a result of an increase of the population, of the average income and of the recreational facilities (e.g. hotel, parking, motorways). From an economic point of view, recreational uses can be considered as open access resources, since it is impossible to charge a fee on the use of these resources for several reasons (Garrod and Willis, 1999)[1].

[1] They identify the following as the main reasons against the possibility to charge a fee to a recreational site:
- transaction costs associated with pricing policy are so high that they do not cover the fee revenues;
- Property rights on the environmental goods are either non existent or ill defined, and it is impossible to exclude on interested in
- Environmental good is provided like a public good
- It is not optimal to charge it given the public good nature of the environmental good: since MC of admitting an additional user is zero, charging it will entail that potential users will decide not to consume it.

As a consequence, public policy is needed to guarantee the provision of this good when other uses compete with recreational ones. Recreational activities related to environmental good uses are often in conflict with other activities that need a given resource. Consider, as an example, a lake that can be used for both productive and recreational uses. If productive uses compromise the amenity of the site (because they affect environmental quality or view) then the two kinds of uses are competitive and a decision should be taken on which of them to favour.

The role of economic analysis is acknowledged by scholars and it has been recognized even in recent EU legislation. Economic analysis plays a crucial role in regulation development and evaluation (Arrow *et al.*, 1996). First, it informs decisions makers on how to allocate scarce resources, so as to maximise social welfare (Hanley, 1993). Secondly, it identifies and quantifies the favourable and unfavourable consequences of a proposed policy change. Finally, it allows policy makers to gather and organize disparate information, thus improving the process and the outcome of policy analysis (Arrow *et al.*, 1996). Portney (1994) advocates the use of contingent valuation and warns economists on the importance on considering environmental valuation, as environmental laws can entail huge compliance costs. Since society has limited resources to spend on regulation, to find the most cost-effective alternative is crucial. Randall (1998) suggests that CBA could be utilised as the main tool for renewal resource management so long as the safe minimum standard for that resource is not violated. In other words, once the reproducibility of the resource is guaranteed, all policy actions should be justifies on a CBA ground, by comparing benefits and costs.

Another justification on the use of environmental evaluation techniques comes from the change in socio-economic patterns. Technological improvements have boosted the implementation of market instruments: however, market mechanisms are successful only if they reflect individuals' preferences (Garrod and Willis, 1999). Environmental valuation attempts to quantify the benefits of environmental projects and policies, so that they can be considered in any CBA.

Evaluation of costs is quite straightforward, since they can be identified with compliance costs. They consist of the costs necessary to put in place the assets needed to clean up the environment (in an *ex post* perspective) or to prevent pollution to occur (in an *ex ante* perspective). In the case of water resource management, compliance costs can be identified in investments needed to collect and treat wastewater or in remediation costs.

Evaluation of benefits is more problematic, since it depends on the choice of what kind of value to evaluate (use, non-use or both?) and on the elicitation method (are we interested in valuing willingness to pay, WTP, or to accept compensation, WTA?). Concerning the first point, the choice depends on the aim of the study. The second issue can be resolved by taking as reference the Consumer Surplus (CS) considered by Willig (1976) a good approximation of either the WTP or the WTA.

For what concerns the economic valuation of competitive uses, WTP for consumptive uses (i.e. household consumption, agriculture and industry) is given by the value of the foregone consumption or production entailed by the diminishing availability of water (provided that the resource use has only a private dimension, i.e. externalities or public good features are absent)[2].

For non-consumptive uses (i.e. amenity and recreational values) others non market techniques are suitable, such as revealed preferences (RP hereafter) and state preferences (SP hereafter) approaches. The former allows the researcher to infer the WTP for some environmental good by looking at the actual behaviour, thus being not suitable in valuing changing of actual environmental attributes (e.g. the improvement of water quality). As stated by Haab and McConnell (2003: 138) the travel cost is "a model of the demand for the services of a recreational site". In this approach, the number of trips depends from the travel costs to the site of interest (negatively), from the travel cost to alternative sites, from income and other socio-economics covariates. Alternatively, stated preferences models can be used to infer contingent behaviour by presenting hypothetical quality changes to the respondents, in terms of WTP, on the site choice or on the number of recreation trips (Mitchell and Carson, 1989). Contingent Valuation Methods (CVMs) assess the economic benefits related to an environmental resource by asking directly to people how much they are willing to pay or willing to accept as a compensation to see a policy implemented3, such that the environmental resource is protected or the environmental quality improved. The survey design is crucial to

[2] Market valuation techniques can allow us to determine, for example, the value of the lost crop for a drought or the KWh not produced for a water rationing. Gibbons (1986) reported several techniques. Income capitalisation approach is often used when one intends to compute a stream of net benefits attributable to water (i.e. the proportion of the benefits produced by the use of water is the value of the resource itself). Other market approaches include the least cost alternative estimates and the land value differential values (for agricultural uses).

[3] The choice between WTP and WTA depends on the distribution of entitlements.

avoid bias estimates. Several works show that responses are influenced by the information provided in the study (Champ et al., 2003). Content validity should thus be tested[4]. An alternative version of the stated preference approach consists in asking people how many times they are willing to visit a given recreational site instead of asking them how much they are willing to pay to have a given policy implemented (Englin and Cameron, 1996). This approach is usually referred to as contingent behaviour (CB, hereafter), since it focuses on hypothetical behaviour instead of hypothetical prices. Englin and Cameron (1996) advocate the use of this approach rather than a CV approach by arguing that it is easier for respondents to predict what one should do in a hypothetical situation than to guess an hypothetical price.

Both TCM and CVM have been criticised. TCM assumes a structure of preferences which may not be testable (Adamowicz *et al.*, 1994). CVM have been criticized because of its "hypothetical" features. Respondents do not face actual situations so their stated WTP can be different from the one relative to real situations. Other biases refer to the choice of payment vehicle, the starting point bias, the strategic bias and the embedding effect, extensively described in Diamond and Hausman (1994).

3. Combining stated and revealed preferences

A combination of the two approaches has been claimed to overcome these limits and to strengthen their advantages (Louviere *et al.*, 2000). Moreover, it makes possible to tackle the problem of water quality change evaluation. We will deal with this point in the next section. TCM only considers the status quo, i.e. the current quality level. However, it could be of interest to consider the effect on welfare measures of quality improvements, by considering the shifts in recreational demand following the quality improvements. Using only revealed preferences methods does not allow us to consider the shift in the demand curve that would follow an improvement in water quality (Loomis, 1995; Whitehead *et al.*, 2000).

Several studies exist that evaluate the benefit deriving from river quality improvements by combining SP and RP approaches.

[4] Assessing CV study validity consists in checking whether the interviewees will actually pay the amounts they would be willing to pay. Researchers can test content validity (i.e. the appropriate framing of the study), criterion validity (i.e. comparison of CV estimates with actual markets) and construct validity (convergence between a CV and other measures of the same good, e.g. TCM).

Cameron (1992) was the first to combine contingent and actual market behaviour to estimate the parameters of the utility function and the corresponding ordinary demand function related to a recreational fishery site. He asked the respondents to choose between (a) cease to use the resource and thus avoid paying a lump sum tax, T; (b) continue to fish at a lower income (given by Y-T). The aim of the study was to combine actual recreational activity decisions with stated intentions to pay for quality changes. His approach considers the information from the utility function relative to actual choice decisions and the one used to infer the willingness to pay. Parameters were estimated using information of both datasets assuming a quadratic utility function, through maximum likelihood. He estimated that demand elasticity at the means is -0.074 and an equivalent variation for a complete loss of access equal to $3,451.

Adamowicz *et al.* (1994) combine SP and RP methods using a random utility framework, in order to value environmental amenities. Individuals are asked to choose one alternative among three options (running water recreational site, standing water recreational site and other non water-based recreational activities). Each option is described through a set of characteristics (attributes such as the distance to the site, the water quality and the fishing rate) and the individuals must indicate their preference for the attributes. They also collect information about the actual behaviour of the respondents, in terms of choice among different sites (considering the actual attributes of each site). They first run two separate SP and RP model, and then a joint RP-SP model. They compare the joint likelihood to the sum of the separate likelihoods for the stated and revealed models. The null of equality of parameters is accepted if joint and summed separate likelihood do not differ statistically.

Englin and Cameron (1996) were the first to use a panel data approach to combine SP and RP data, studying the economic benefits of recreational fishing in Nevada. Their objective was to assess the change in trip frequency as the price changes. In this study, anglers were asked how many fishing trips they had taken during the previous year and how their total trips would change if travel costs increase by different percentages. They therefore obtained four price-quantity estimates for each respondent (one real and three hypothetical). They estimated four model specifications: a pooled standard Poisson, a panel standard Poisson, a pooled fixed effect standard Poisson and a panel fixed effect standard Poisson. They find that CS varies considerably among models between $752 and $2,685.

In their study, Whitehead *et al.* (2000) combine SP and RP to measure recreational benefits given by a water quality improvement, by considering both participants and non participants. They pool SP and RP responses in a unique dataset and estimate a joint recreation demand model through a random effect Poisson model (to consider heterogeneity among individuals) with dummies variables (to consider structural changes in different scenarios). They find that CS per trip is equal to $64.14 (with current quality) and $84.99 (with improved quality). The estimate changes substantially when only participants are considered. In this case the CS per trip is $105. They conclude that excluding non participants would overstate consumer surplus.

The combination of SP and RP seems particularly useful when interested in assessing recreational water uses. The utility of this approach is twofold: on the one hand, the travel cost exercise allows the researcher to assess the current recreational water demand. In other words, it gives a static picture of the uses as they are in the moment in which the economic analysis is carried out. On the other hand, the contingent behaviour exercise made possible to estimate the WTP for the improvements in water quality which, as a result, allows the decision makers to state in which cases the costs incurred to meet this quality objective are disproportionate. In the next section we will underline the policy relevance of the case study chosen and will describe the methodology more in details.

4. The case study and the methodology

The Idro Lake is located in the Brescia District (Lombardy Region, Northern Italy). It represents an interest case of water resource contested among different uses. At the same time it has an intrinsic value, forming wetlands which play a crucial role in preserving biodiversity. For this reason the lake has been classified as “a site having a European importance”, following the Directive 92/43/CEE. It is the only example in Italy of a natural lake whose natural flow been modified, upstream, through pipes which regulate the inflow, and downstream, through a dam. This dam was built during the 1920s, but it is still in use. It was built to divert water for hydroelectric and agricultural uses. Since then, the lake has been experiencing conflicts among competitive uses: during the ‘20s the disputes were between agricultural and hydroelectric uses. During the ‘60s, the conflict interested different agricultural uses. The present day disputes raised after the expiry of the 70 years withdrawal concession to a reclamation board, in 1987. Starting from that year, the State started an experimentation phase to decide how to share the resource among competitive uses. The fact that the water of the lake a scarce resource is highlighted in the 2004 Commissioner report, which emphasised in his conclusions that the resource is not still able to satisfy all the competitive uses. So far, only productive uses have been considered in the planning process, i.e. withdrawals for agricultural and hydroelectric purposes. However, in recent years other uses than hydroelectric and agricultural ones emerged, as a consequence of socio-economic changes. In particular, tourist uses and non uses (i.e. ecosystem protection) started to receive a greater attention[5]. Riparian local authorities and civil society started to ask for a greater role in determining the use of the resources, in particular for what concerns the protection of ecosystems and the exploitation of tourist vocation of the Sabbia Valley. Tourist activities consist mainly on beaching (even if swimming is not always possible, since the lake was declared not suitable for bathing in 1999 and 2003), fishing, windsurf and sky surf. In the surrounding mountains canoeing, parapenting and hang-gliding are done.

Apart for the relative scarcity between agricultural and hydroelectric uses, the water withdrawals cause the Idro Lake level to vary considerably (up to 6 metres[6]). The damages entailed by these fluctuations are twofold: on one side the wetlands, not receiving sufficient water, dry during summer, causing the death of fish eggs and other water species. As a consequence fishing recreational activities suffer from a diminished stock of fish available. On the other side, with less water, docks and other infrastructure become useless, since they are some meters above the actual level of the lake.

Apart from water scarcity, the lake suffers of water quality problems. Antrophic activities cause diffuse and point discharges, which entailed a steady increase over years of the nutrient level, from 50 μg/l during the ‘50s to the 300 μg/l. In the last decade the lake showed eutrophication and a consequent seaweed proliferation (ARPA, 2005). The impact of the water scarcity on the quality of the lake is not recognised

[5] Pesaro (1997) estimates that each year 500,000 visitors travel to the lake and 40% of the local population is employed in tourism business.

[6] For what concerns water resource planning, the lake inflow and outflow has been re-regulated in 2001, so as to increase the Idro Lake level in the season of maximum withdrawal. Lake fluctuations have been reduced to 3,25m (previously they were of 7 meters). However, the increase in the lake level is deemed insufficient to guarantee the minimum inflow. This is due to the fact that, for security reasons, the Dam Authority established the maximum level of the lake in 367m over the sea (there is the threat of a landslide), whilst the temporary regulation fixed the maximum level in 369,25m: as a result, actual fluctuation is more than the 3.25m allowed (being almost six meters).

univocally. In particular, while the Regional Environmental Agency in its report states that "the fluctuation of the [Idro] lake is one of the highest of the lakes analysed so far, and it entails several and severe environmental damages" (ARPA, 2005: 13), other experts tend to exclude that the lake quality depends on its level and they emphasised the need for infrastructural interventions (such as the collection and treatment of wastewater). While the former experts agree on the fact that the fluctuations of the lake level should be limited, in order to increase the water quality, the latter stress the importance of infrastructure development. In the first case, withdrawal for agricultural and hydroelectric purposes should be limited. A restriction of the amount of water available for productive uses means loss of output. In the second one, cleaning policies are necessary. The cost of these two different policies should then be compared with the benefits deriving from ecosystem protection and tourism development.

The objective of this study is to assess the benefits related to recreational activities considering different levels of water quality. Having this aim in mind, we decided to conduct a travel cost study, which is very popular among researchers interested to assess the amenity value of a given site (Garrod and Willis, 1999). Given the hypothetical feature of the "improved quality" scenario, we decide to combine SP and RP to infer the value deriving from the protection of the resource. Up to our knowledge, only Alberini *et al.* (2005) apply the same methodology to an Italian case study, but this study focused on estimation of the benefits deriving from an increase in sport fishing in the Venice Lagoon. Our work is thus the first that assesses the recreational water uses following the WFD, i.e. combining SP and RP to consider at the same time the status quo and an improvement in environmental quality.

For what concerns the Idro Lake, the quality improvement will affect local population (who will live in a cleaner and pleasant environment) and the actual travellers (who will enjoy a higher quality). Moreover, it is likely that improved water quality will increase the recreational demand from non-actual users. Since data on the actual trips were not available, they had to be collected through an ad hoc survey, combining the two approaches. In order to avoid to define the market of interest in an artificial manner (i.e. by considering all the inhabitants of the neighbouring districts or the whole population living X km far from the lake), we decide to collect primary data through on-site in person interviews to travellers[7]. By doing so we are keen

[7] We chose to survey only participants even because of the limited amount of time and finance available.

that we are excluding potential travellers and local resident. This point will be taken into account when we will discuss our results.

The questionnaire was piloted and revised during the month of June 2005 and the main survey was conducted in July 2005. The aim was to gather information on the actual trips to the lake, on the future trips on the lake at the current quality level and on the future trips on the lake, provided water quality improves. We also gather information about the type of recreational activity respondents conduct on the lake. We then ask them about the distance they cover to the lake and to an alternative site and on the relative travel costs as well. We finally collect some information relating their socio-economic characteristics. The purpose of the hypothetical question was to estimate the shift in the demand function, once the water quality improves. In order to make the problem more comprehensible to the respondents, we attached to the questionnaire a set of pictures, showing the state of the lake with different levels.

A total of 169 responses was obtained. Before undertaking the estimations, we eliminated 11 responses with missing data and 3 respondents who stated that they will stop to visit the lake, once the water quality improved (we assume that these people misunderstood the survey). Our panel data set is thus constituted by 155 individuals (465 observations). The questionnaire allows us to collect data on the variables described in table 1.

Considering the 155 responses we collected, the average number of trips in the past twelve months was 7.994, whilst in the next twelve months it should slightly increase to 8.439. If the quality improves the average number of trips will increase by 41% (11.865). Regarding participation, since we are sampling only participants obviously in the first period the rate of participation is 100% and it decreases in the following period (5 respondents who stated that they will not visit the lake in the next period if the quality of the lake remains the same will visit it, provided the environmental quality improves). However, overall the number of visits does not decrease, as a result of increasing visits per participant.

The majority of the lake visitors go to the lake to lay on the beach (63.23%) or to have a walk (51.61%). Only 6.45% go fishing. For what concerns the socio-economic characteristics, the majority of respondents has an income below 40,000 (44% between 0 and 19,999 and 43.8% between 20,000 and 39,999). 9% has income between 40,000 and 59,999 and 3.2% has an income greater than 60,000. The respondent majority works (52.9%) or is retired (21.3%). Students' number amounts at 9.03%. 14.8% of the respondents is housewife.

Regarding the total costs, average travel costs the Idro Lake amounts to €18.31, while the cost to the substitute site are €17.78. The distances covered are respectively km 138.91 and km 153.26[8]. Average respondent age is 44 years and 43.87% is a male.

Table 1 – Data collected through the survey

Variable	Description
Part1	Level of participation during the last 16 months (t=1)
Part2	Level of participation during the next 16 months (t=2)
Part3	Level of participation during the next 16 months provided that an improvement in the water quality occurs (t=3)
Trip1	Number of visit taken in t=1
Trip2	Number of visit taken in t=2
Trip3	Number of visit taken in t=3
Cost1	Travel costs to Idro Lake (€)
Cost2	Travel costs to an alternative site (€)
Dist1	Round trip distance (home respondent to Idro Lake)
Dist2	Round trip distance (home respondent to an alternative site) in Km
Cost_time	Opportunity cost of the time spent on travelling (€)
Age	Age of the respondent (years)
Stud	1 if the respondent is student, 0 otherwise
Empl	1 if the respondent is employed, 0 otherwise
Unempl	1 if the respondent is employed, 0 otherwise
Hous	1 if the respondent is housewife, 0 otherwise
Retired	1 if the respondent is employed, 0 otherwise
Oth	1 if the respondent is not in one of the economic categories listed above, 0 otherwise
Inc	1 if the respondent has an income between 0 and 19,999 €, 0 otherwise
Inc1	1 if the respondent has an income between 20,000 and 39,999 €, 0 otherwise
Inc2	if the respondent has an income between 40,000 and 59,999 €, 0 otherwise
Inc3	if the respondent has an income over 60,000, 0 otherwise
Male	1 if the respondent is a male, 0 otherwise
Env	The fact that the respondent was an environmental association member

[8] By considering only the travel costs, costs incurred to arrive at the substitute site are greater than the travel costs to the Idro lake.

Table 2 – Trips and participation

Variable	Mean	SD	Minimum	Maximum
TRIP1	7.994	8.641	1.000	50.000
TRIP2	8.439	8.778	0.000	50.000
TRIP3	11.865	10.994	0.000	50.000
PART1	1.000	0.000	1.000	1.000
PART2	0.929	0.258	0.000	1.000
PART3	0.968	0.177	0.000	1.000

Table 3 – Total travel costs and distance covered

Variable	Mean	SD	Minimum	Maximum
TCOST	18.306	22.244	0.500	150.000
DIST1	138.916	110.032	8.000	700.000
SUBCOST	17.785	24.708	0.000	150.000
DIST2	153.258	326.955	0.000	2600.000

In order to estimate the model of the demand for recreation the researcher has two alternative model structures (Haab and McConnell, 2003). She can model the choice among all sites or the demand for one site. In the latter case the aim is to estimate the demand function for a recreational site, whilst in the former one should begin by assuming a utility function specification, which considers the full set of choices, including the choice to participate at each site together with the number of visits.

In this study we decides to focus on the single site demand estimation, since we are interested in assessing economic benefits related to the Idro Lake (instead of modelling the choice to travel to the Idro Lake rather than other recreational site). Moreover, this approach seems more suitable than the RUM framework, since we are not interested in valuing different attributes of the lake, but the value of an overall water quality improvement.

In building our econometric model, we assume that the number of trips taken by the individual *i* in the scenario *t* (t = 1,2,3, see below) can be described through a Poisson distribution with mean and variance μ_{it}:

$$\Pr(x_i = x_i) = \frac{e^{-\mu_i}\mu_i^{x_i}}{x_i!}$$

The mean of the trips (μ_{it}) depends on different covariates for each individual, z_{it} (deterministic part, λ_i) and individual heterogeneity (stochastic part, u_i).

$$\lambda_i = \exp(\mathbf{z}_i \beta)$$

The sample loglikelihood function is thus given by

$$L(\boldsymbol{\beta}|\mathbf{z}, x) = \prod_{i=1}^{T} \frac{\exp(-\mu_i)\exp(\mu_i x_i)}{x_i!}$$

The Poisson model assumes that conditional mean and variance are the same, μ_{it}.. This is a very strong assumption, since often for recreational trip data variance is often greater than mean, implying overdispersion of the data (Haab and McConnell, 2003). In this case, a negative binomial model can be more appropriate. The Negative Binomial Model corresponds to a Poisson model with a gamma distributed error in the mean (Haab and McConnell, 2003).

$$\log(E(x_i)) = \mathbf{z}_i\boldsymbol{\beta} + \theta_i$$

where θ_i represents unobserved heterogeneity. In order to estimate this model, a distribution of θ_i must be assumed. In the negative binomial model it is assumed that θ_i are distributed with a gamma density with mean μ_i and variance $\mu_i(1+ \varphi\mu_i)$[9]. The null that $\varphi = 0$ can be used to test the appropriateness of the Poisson model.

Thus the distribution of trips, conditional on θ_i is given by:

$$\Pr(x_i \mid \vartheta_i) = \frac{\exp(-\exp(z_i\beta + \vartheta_i))\exp(z_i\beta + \vartheta_i)n}{x_i!}$$

[9] The general expression for a gamma distribution is (Haab and McConnell, 2003):

$$h(v) = \frac{v^{\alpha-1}\exp(-v/\beta)}{\Gamma(\alpha)\beta^{\alpha}}$$

The model has to be chosen by testing the coefficients. If overdispersion exists, the Poisson model will be rejected in favour of the binomial model. Even if our data were collected on site, we did not refer to a truncated specification because on the SP responses some zeros were observed.

The expected number of trips for a negative binomial model (i.e. the semilog demand function) can be written as follows, where $TCOST_i$ is travel cost for individual i, Zi is the vector of all explanatory variables and α and β_i are coefficients.

$$E\{x_i | z_i, TCOST_i\} = \exp\{z_i \beta_i - \alpha TCOST_i\} \equiv n_i$$

With this specification the consumer surplus for individual i is given by

$$CS_i = n_i / \alpha$$

Elasticity at the mean (η) can be computed easily considering the following expression.

$$\eta = z_{ij} \beta_j$$

Joint estimation of TCM and CB are discussed in McConnell et al. (1999) and Louviere *et al.* (2000). In this approach, RP data are used as a comparison basis, and SP are used only to ameliorate certain characteristics of the TCM data (like the strong correlation among attributes). This is the rationale of the "data enrichment paradigm" (Louviere *et al.*, 2000).

In order to combine SP and RP data, the SP and RP demand function must underline the same structure of preferences (i.e. parameters must be equal). This point is crucial for the consistency of the model (McConnell et al., 1999). To test this assumption, there are two alternatives. Swait and Louviere (1993) suggest to estimate separate models for each dataset; then to estimate a pooled model from the pooled data and finally to calculate the chi-square statistic for the hypothesis that the common demand parameters are equal[10].

[10] The chi-squared statistics is $LL = -2\left[\left(L^{RP} + L^{SP}\right) - L^{Joint}\right]$ (Swait and Louviere, 1993).

Alternatively, Whitehead *et al.* (2000) pool all the SP and RP data in a single equation, and then test for the equality of the parameters. The model specification used by these authors allows for differences in the underlying structure of behaviour. For that reason, we decided to adopt the same model in this work. The choice of the model is crucial since, as stated by Haab and McConnell (2003) different methods can lead to different estimates to WTP and consequently welfare measures. This depends on the fact that travel cost is a proxy for the price. Since travel costs among different sites are highly correlated, travel costs coefficients of different model specifications will be different and consequently welfare estimates vary. We will highlight these differences in the next section.

We pool in a single equation the visits in the three periods, as done in Englin and Cameron (1996), Whitehead *et al.* (2000) and Hanley *et al.* (2003). In particular, analogously to Whitehead *et al.* (2000), we build three scenarios, thus having to estimate defined as follows:

- t=1: RP behaviour (trips taken in the last 12 months):
- t=2: SP behaviour with water quality unchanged (i.e. trips taken in the next 12 months provided that the water quality remains unchanged)
- t=3: SP behaviour with water quality improvement (i.e. trips taken in the next 12 months in case the water quality of the lake improves)

These authors estimated joint recreation demand model, by pooling stated and revealed trips, with current and improved water quality. They use a random effects Poisson model (to take into account the heterogeneity among individuals) with dummy variables (to consider structural changes following quality improvements). They want to consider structural changes after the water quality improvement, because this can increase the number of participants (i.e. the demand curve shifts outwards) and because the inclusion of the new participants could change the shape of the demand curve (i.e. its elasticity).

We find that TCM and CB responses underlined the same preference structure and thus we can the two can be pooled. By considering the panel dataset, we find that a parallel outwards shift in the recreational water demand occurs.

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$$
\begin{aligned}
&\ln \mu_{it} = \ln \lambda_{it} + u_i = \alpha_i + \beta_1 TCP_{it} + \delta_1 TCF_{it} + \phi_1 INCOME_{it} \\
&+ \varphi_1 PAMLICO_{it} + a_2 D_2 + b_2 D_2 TCP_{it} + c_2 D_2 TCF_{it} + d_2 D_2 INCOME_{it} \\
&+ a_3 D_3 + b_3 D_3 TCP_{it} + c_3 D_3 TCF_{it} + d_3 D_3 INCOME_{it} + u_i
\end{aligned}
$$

Where the intercept dummy variables D2 and D3 allow differences in the intercept (thus accounting for parallels shift in the demand curve) and the slope dummy variables (D*t*TCP, D*t*TCF and D*t*INCOME) allow the demand elasticity to change.

5. *Results*

We select the model covariates following a "simple-to-general" procedure, i.e. starting with few independent variables and then adding only the ones whose coefficients are statistically different from zero, at a given significance level. Firstly, we checked for multi-collinearity. Since total costs (both to Idro Lake and to the substitute site) show positive autocorrelation, we only consider the total costs. Similarly, "age" and "retired" were positively correlated, so we decide to consider only the fact that a person is retired.

We then run both Poisson and Negative binomial models with Limdep 7.0 in their simplest formulation, i.e.

$$
\ln \mu_{it} = \ln \lambda_{it} + u_i = a_i + \beta_1 TCOST_{it} + \beta_2 SUBCOST_{it} + u_i
$$

The output table (see appendix C) shows that the coefficient of the total costs for the substitute site was not significant in both the model. This result tells us that the decision of how many trips to take is not influenced by the total costs necessary to arrive at the substitute site. At the same time, we checked the overdispesion parameter (α) and, since it was significant, we decide to continue the analysis with the Negative Binomial Model. Not surprisingly, the sample variance is greater than the sample mean and the Poisson model is not appropriate, since it has in its basic assumptions the equality between the sample mean and the sample variance.

Then we repeat this stepwise procedure. All the dummy variables indicating the income range were not significant at 5% significance level. This result should be interpreted in a cautious manner. It does not entail that respondents' income has no influence on the decision to travel to the Idro Lake. The income dummy variables consider different income ranges. Individual and overall insignificance of the income coefficients mean that behaviour of a respondent belonging to a particular range is not statistically different from the behaviour of a respondent having a different income.

Other socioeconomic characteristics result no significant at 5% significance level. Only total costs and the dummy variable indicating that a respondent was retired were significant (respectively at 1% and 5% significance level). The signs are the ones expected: total costs have a negative sign, showing that when they increase, recreational demand decreases, thus confirming the inverse relationship found in the literature between total costs (i.e. the price of travelling, see section 2) and travel (Q). Considering the "retired" covariate, the fact that one person is retired positively affects the decision to travel to the Idro Lake. Informal talks with respondents confirmed this result. Elderly people confirmed they do like the Idro Lake because of its tranquillity and because it is less crowded with respect to alternative sites (e.g. Garda Lake and Iseo Lake).

Our basic model is thus the following.

$$\ln \mu_{it} = \ln \lambda_{it} + u_i = a_1 + \beta_1 TCOST_{it} + \beta_2 RET_{it} + u_i$$

All the steps of the analysis and the relative outcomes can be read in appendix C. The statistical results of our model 1 are shown in table 4. Following Whitehead et al.'s analysis, we built a general recreation demand model adding some dummy variables in the mean, so as to take into account potential structural change in the recreation demand following the quality improvements. In order to take into account the structural changes induced by an improvement in the water quality, we modified the basic model by introducing dummy variables that allow us to consider changes in intercept and cross elasticities of the demand function.

Table 4 – Results with the random effect Negative Binomial Model

Variable of X	MODEL 1	MODEL 2	MODEL 3
Constant	2.2241 (0.0898)	5.0250 (0.6744)	4.9159 (0.5609)
TCOST	-.0 0073 (0.0017)	-0.0080 (0.0075)	-0.0075 (0.0019)
RET	0.5262 (0.2254)	0.5719 (0.2334)	0.5771 (0.2290)
D2	-	0.0427 (0.1622)	-
D2TCOST	-	0.0009 (0.0096)	-
D3	-	0.3848 (0.0852)	0.3710 (0.0354)
D3TCOST	-	0.0010 (0.0042)	-
a	0.8215 (0.1171)	25.8117 (14.9318)	22.9591 (11.2543)
b	-	1.2749 (0.2099)	1.2825 (0.2080)
Loglikelihood	- 1.309,230	-1.253,371	-1.254,355
Sample	155 individuals	155 individuals	155 individuals

We thus considered the following modified model.

$$\ln \mu_{it} = \ln \lambda_{it} + u_i = a_1 + \beta_1 TCOST_{it} + \beta_2 RET_{it} + a_2 D_2 + b_2 D_2 TCOST_{it}$$
$$+ a_3 D_3 + b_3 D_3 TCOST_{it} + u_i$$

As shown in table 4 (model 2), only the quality improvement dummy variable (D3) is different from zero at 1% significance level. This result indicates an outward shift in the recreational demand function if the water quality improves. We went in depth of this point through several hypothesis testing.

We conducted several tests, namely that there are no structural changes among the different scenarios (i.e. at = bt = 0); no structural change before and after the quality improvements (i.e. a3 = b3 = 0); no difference between revealed demand for this season, t = 1, and the revealed demand for next season, t = 2 (i.e. a2 = b2 = 0). Considering the null hypothesis of no structural change among the three scenarios (a2 = b2 = a3 = b3 = 0) we find that coefficients are jointly different from zero at 5% significance level. This means that following the water improvement, a structural change in the recreational demand occurs. To understand if it is coincident only with a parallel shift in the recreational demand or if the slope of the function changes as well, we test the null that cross elasticities are zero (b2 = b3 = 0). We find that slope does not change before and after the quality change, since coefficients are not significant at 5% significance level. We then tested that there is a structural change after the quality improvement, i.e. in t = 3. We cannot accept the null that both coefficients were zero. Data suggest that there is a structural change after the water quality improvement. To confirm this result, we test equality of stated coefficients with or without the quality change (a2 = a3; b2 = b3). We find that behaviour is going to change before and after the quality change (we cannot accept the null since Wald observed was 22.45 and $\chi 2 = 6$, df =2). We finally tested the null of a parallel shift occurs (a2 = a3 = 0). Observed Wald (65.15) confirmed that a parallel shift actually occurs ($\chi 2 = 6$, df =2).

When estimating joint recreational demand models, it is crucial to test that the stated and the revealed responses underlined the same behaviour. For this reason, we tested the null of coefficients in t = 2 being zero, i.e. insignificant (a2 = b2 = 0). We find that these coefficients are insignificant at 5% significance level (observed Wald is 0.22, $\chi 2 = 6$, df =2).

On the basis of these results we estimated a general recreational demand model which takes into account the demand shift following the quality improvement.

$$\ln \mu_{it} = \ln \lambda_{it} + u_i = a_1 + \beta_1 TCOST_{it} + \beta_2 RET_{it} + a_3 D_3 + u_i$$

The coefficient estimates, with the relative standard errors, are shown in table 4 (model 3).

Finally, we conducted a likelihood ratio test of the appropriateness of the panel random effect specification against the pooled model. We find that a random effect panel estimator is a better choice than a pooled model (observed $\chi 2$ = 498.32).

Our main finding is that, following a water quality improvement, a parallel shift of the water recreation demand occurs, with the recreational demand curve shifting outwards (see the positive sign of D3, the slope dummy). Our results partially diverge from that of Whitehead et al. (2000). In their study, they find that not only there is a shift of the demand, but there is also a change in the slope. This is showed in fig. 1, where D(q) shows the current water quality demand, D(q') the parallel shift in demand following a quality improvement and D(q')* a shift in demand with a change in slope following the policy implementation.

Our analysis indicates only a likely shift in the recreational demand function, without any change in the slope. The difference in these findings is explainable by considering the difference in the sample characteristics: whilst Whitehead *et al*. (2000) conducted an off-site sampling (thus having responses from both participants and non participants), we interviewed the respondents on site, thus having only participant data. As a consequence, the slope of the demand curve does not change since the respondent preferences remain the same in all the three scenarios. At the contrary, considering new participant can entail a change in the recreational demand curve slope, since they show preferences different from that of former participants.

Given the estimates showed in table 6, we can simply compute the elasticity of the demand curve at the mean (η) which is equal to -0.075 (SE = 0.0187) and significant at 1% significance level. This result is in line with the main literature findings. With these results, we can infer the economic value of the Idro Lake recreational uses by computing welfare estimates. The table 5 shows the aggregate CS, with current and improved quality level, and the difference between the two.

Table 5 – Welfare estimates (€)

Variable	Coefficient	Standard Error	b/St.Er.	P[|Z|>z]
Individual CS	134.0848619	33.570795	3.994	.0001
Seasonal CS with the actual water quality level	357.2712772	89.449924	3.994	.0001
Seasonal CS with improvement in water quality	530.2840022	132.76708	3.994	.0001
Change in seasonal CS following the water quality improvement	173.0127250	43.317154	3.994	.0001

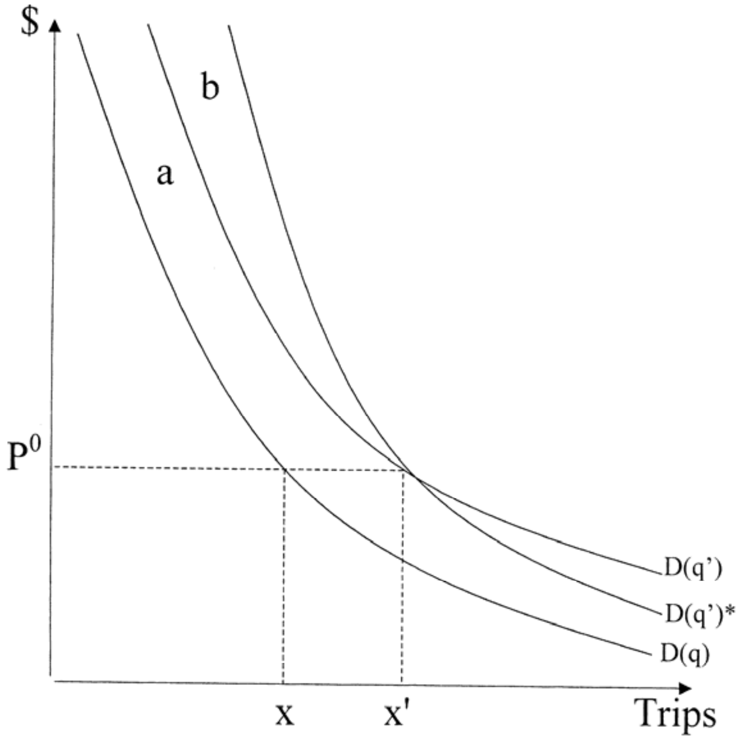
By considering the total number of trips taken by the survey respondents, we obtain an estimated individual CS per trip at the improved water quality level of € 134.08. In order to estimate the aggregate increase in CS, it will be necessary to know the actual number of visitors, a data which is not available11. Following Pesaro (1997), the total annual number of visitors amounts at about 500,000. This figure cannot be applied to this study, since informal talks with local residents and tourists confirm a steady decline in the number of visitors in the last decade. We will take as reference the number of tourists that stay in hotels, i.e. 126,684 individuals12. However, by considering this data (referring to 2004), it is possible to obtain the lower bound of the aggregate economic benefits associated with water quality improvement (and related to use values). With this assumption, we estimate aggregate CS equal to € 16,986,406. Similarly, the aggregate increase in CS is equal to € 21,917,944.

Excluding the non participants influences welfare estimates. Whitehead et al. (2000) demonstrate that excluding non participants overstates consumer surplus per trip. In their case, per trip CS considering only participants overestimated the CS considering both participants and non participants by 40%. This demonstrates that survey design can affect heavily welfare estimates. For instance, by assuming that our welfare estimate overestimates the real CS by the same percentage indicated by Whitehead et al. (2000), the individual CS in our case would then be equal to € 80.45, our aggregate CS €10,191,844 and the increase in CS will be € 103.81, giving an aggregate increase in CS of €13,150,767.

[11] Tourism Office normally gives the figures relative to the number of presences, i.e. the number of tourists who stay in hotels.

[12] This figure has been provided by the Assessorato al Turismo, Provincia di Brescia (i.e. Tourism Unit of the Brescia District).

Figure 1 – Recreational demand at current (q) and improved quality (q')



Source: Whitehead et al. (2000)

Given the assumption of weak comparability, the non use values related with biodiversity conservation are not taken into account. Holmes et al. (2004) and Loomis et al. (2000) measure the total economic value of ecosystem services (i.e. non-use values) considering a contingent valuation and a choice experiment approach, respectively. Holmes et al. (2004) find that benefits generated by full restoration households can be estimated in \$ 2,835,373 (equivalent to a cost ratio of 15.65). Loomis et al. (2000) find that household were willing to pay \$ 19 million for an increase in ecosystem services through an increase in water bills, well above the project costs, estimated in \$ 13.4 million. Inclusion of non-use values will increase economic benefits related to conservation. In our case, non-use benefits are likely positive and the total economic benefits are consequently greater than the use benefits we estimated.

Finally, TCM consider only benefits referring to travellers (actual or potential). The water quality improvement will affect the local inhabitants as well, but we do not consider this dimension of value, since another ad hoc study would be necessary. As a consequence, our results underestimate the total economic benefits related to water quality improvements and should be considered as a lower bound of the true economic benefits deriving from water protection policies.

Nevertheless, our findings are relevant for policy makers. Firstly, following the WFD, they made possible to assess the economic value of the recreational uses on the Idro Lake. This is equal to the estimated aggregate CS (€10,191,844). This figure can be compared with the economic value associated with other uses13, in order to allocate the resource to the most valuable uses. Economic benefits in productive uses can be assessed by considering direct benefits (i.e. produced output) and indirect ones (i.e. increase in employment rate associated with water-related activities). Secondly, these estimates can be compared with the costs associated to the water quality improvements (i.e. clean up costs). Our simulations make possible to compare the benefits arising from environmental protection with the costs incurred for that objective. It has been estimated (Regione Lombardia, 2004) that wastewater collection and treatment investments in the Idro water basin can be quantified in € 27 million. For simplicity, by considering the current year14, if the actual number of visitors is at least 336,000, then WDF compliance costs are justified only by considering recreational benefits.

6. Conclusions and further research

This work has been conceived by considering the increasing economic and social importance of recreational demand for natural resources. It has been highlighted that this is the result of changes in socio-economic patterns of behaviour and of an increased demand of environmental quality.

As a consequence, recreational water uses should be considered in water resource planning. In Europe water resources continue to be allocated through administrative decisions, however recent EU legislation emphasized the desirability to incorporate efficiency considerations in water resource planning process, through an appropriate economic analysis (WATECO, 2002).

[13] Up to our knowledge, no estimates exist on the value of water for agricultural and hydroelectrical uses in the area considered.
[14] This allow us to avoid to calculated net present value.

This study considers this claim, since it assesses the economic value related to recreational water activities, by estimating the actual recreational demand and the hypothetical one obtained by considering an "improved quality" scenario. Through an on-site survey, we built a panel dataset. Following Whitehead *et al.* (2000) and Hanley *et al.* (2003) we get welfare estimates by combining SP and RP responses. We find that TCM and CB responses underlined the same preference structure and thus we can the two can be pooled. By considering the panel dataset, we find that a parallel outwards shift in the recreational water demand occurs. The present CS is estimated in €134 per individual, whilst the increase in CS is estimated in €173 per individual. Following Whitehead *et al.* (2000), if we assume that the fact to consider only participants overestimates the true CS by 40%, we can correct the bias considering the present CS equal to €80.45 per individual and an increase in CS equal to € 103.81 per individual, with aggregate benefits equal to € 10,191,844. Similarly, the aggregate increase in CS is equal to € 13,150,767.

These figures can be confronted with the economic value of competitive uses and with the clean up costs, respectively, to infer some useful policy indications. On the one hand, available information does not allow us to infer if water resources are allocated efficiently among competitive uses (further research is needed to highlight the dimensions of value associated with competitive productive uses). On the other hand, we showed that WFD compliance costs can be covered by benefits associated with use values, provided that actual visitors amount at 336,000 individuals. Considering non-use values, which should be estimated in an *ad hoc* study, it is likely that water conservation policies show net social benefits.

This study allows us to infer the economic value associated to Idro Lake recreational uses, considering different water quality scenarios. It is the first assessment of this kind ever carried out on this recreational site. Our "better quality" scenario was built in a very naive way, by asking respondents about a general water improvement. This is because at the beginning of this work we were mainly interested in assessing if and to what extent the recreational demand would have shifted following a water quality improvement.

Once confirmed that a demand shift is likely to occur, it can be of interest to assess what attributes visitors deem relevant in their travel decisions. Following Egan *et al.* (2004) it would be possible to conduct a choice experiment on the single attributes of water quality. Obviously attributes would not be limited to indicators of environmental quality, e.g. the water transparency, but would encompass more general characteristics that tourist sites should possess in order to be able to welcome tourists (e.g. parking, picnic areas …). An analysis of this kind would require more time than the one available for this work and that is the only reason why we exclude this possibility.

The importance of this work goes behind the number it attaches to recreational activities. As stated by Portney (1994), environmental evaluation allows economists and policy makers to tackle with the public good nature of the benefits of preservation, which entails an under provision of this good as a consequence of the free riding problem. He argues that to consult people "has the potential to inform [them] about the nature, depth and economic significance of these values" (Portney, 1994: 15). The informational potential of evaluation exercises can be maximised by mixing evaluation techniques and post-questionnaire analysis, as proposed by Powe *et al.* (2005). In particular, the qualitative analysis helps the researcher detect individual preferences and opinions about alternative policy interventions.

This is what the art.14 of the WFD prescribes when it states that "Member States shall encourage the active involvement of all interested parties in the implementation of this Directive, in particular in the production, review and updating of the river basin management plans. Member States shall ensure that, for each river basin district, they publish and make available for comments to the public". Post-questionnaire analysis can be considered a way to involve citizens in water policy formulation.

Public participation is one of the main challenges facing environmental decision making. Economists and other social scientists should work together in order to put into practice this concept. One prerequisite of public involvement is information. Economists should play a key role in making economic values attached to natural resource uses explicit and thus increasing the information available to the public and to policy makers.

Acknowledgements

This study has been carried out under the supervision of Riccardo Scarpa, whose comments are aknowledged. I am also grateful to Viola Angelini for advice on the econometric analysis; to Antonio Massarutto for useful suggestions and to Nicola Cantore for constant support.

This paper has been presented at the Second International Conference on Tourism Economics, Palma de Mallorca, 18th – 20th May 2006. The author thanks participants for their comments. I am the only responsible for any errors and imperfections.

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