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# Improving value transfer through socio-economic adjustments in a multicountry choice experiment of water conservation alternatives*

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This study tests the transferability of the nonmarket values of water conservation for domestic and environmental purposes across three south European countries and Australia applying a common choice experiment design. Different approaches are followed to test the transferability of the estimated values, aiming to minimise transfer errors for use in policy analysis, comparing both single- and multicountry transfers, with and without socio-economic adjustments. Within Europe, significant differences are found between implicit prices for environmental water use, but not for domestic water use. In the Australian case study, alleviating restrictions on domestic water use has no significant value. Pooling the three European samples improves the transferability of the environmental flow values between Europe and Australia. Results show that a reduction in transfer error is achieved when controlling for unobserved and observed preference heterogeneity in the single- and multicountry transfers, providing additional support for the superiority of socio-economic adjustment procedures in value transfer.

**Key words:** benefits transfer, choice experiment, preference heterogeneity, water conservation.

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## 1. Introduction

Given the costs of nonmarket valuation studies, it is not surprising that a considerable literature concerning the transfer of nonmarket value estimates has emerged (Brouwer 2000; Barton 2002; Ready *et al*. 2004; Navrud and Ready 2007; Johnston and Rosenberger 2010; Bateman *et al*. 2011; Martin-Ortega *et al*. 2012). Transfer exercises typically involve approximating the value of a given change in the provision of a good at some 'policy site' from analyses undertaken previously at one or more 'study sites'. The literature has placed great emphasis upon the development and testing of stated value transfer methods (e.g. Desvousges *et al*. 1992; Downing and Ozuna 1996; Brouwer and Spaninks 1999; Rolfe and Bennett 2006; Moeltner *et al*. 2007; Zandersen *et al*. 2007; Johnston and Duke 2009; Moeltner and Woodward 2009). These methods can be broadly categorised into two types. The simplest approach is to transfer mean values from study to policy sites. Such transfers are frequently used in practical decision-making, but have been shown to generate significant transfer errors (Jiang *et al*. 2005; Johnston 2007). The alternative is to use value functions which relate the value of interest to ecological and socio-economic characteristics (Wilson and Hoehn 2006; Navrud and Ready 2007).

Value functions can be estimated using data from one original study (Loomis 1992), or by using a meta-analysis from several case studies on similar sites (Rosenberger and Loomis 2000; Woodward and Wui 2001; Bateman and Jones 2003). These are then used to predict new values for policy sites. This 'value function transfer' approach assumes that the underlying utility relationship embodied in the parameters of the estimated model applies not only to individuals at the study site(s), but also to those at the policy site. While parameters are kept constant, the values of the explanatory variables to which they apply are allowed to vary in line with the conditions at the policy site. The latter 'adjusted approach' to value transfer is theoretically expected to be superior to unconditional mean (unit) value transfer, since effectively more information is transferred (Pearce *et al*. 1994).

Despite the efforts in this area of research, the reliability of transfer procedures is still under discussion (Wilson and Hoehn 2006; Moeltner *et al*. 2007; Colombo and Hanley 2008; Johnston and Duke 2010; Johnston and Rosenberger 2010; Johnston and Thomassin 2010). Compared to contingent valuation (CV), choice experiments are believed to be better equipped for value transfer (Morrison and Bergland 2006; Rolfe and Bennett 2006). Their multi-attribute nature automatically allows for the valuation of marginal changes in good and site characteristics and can hence account for differences in environmental quality when transferring values between sites (Morrison *et al*. 2002). Johnston and Duke (2010), however, find that attribute-adjusted value transfers do not always outperform unadjusted transfers, while socio-economic adjustments are equally likely to reduce than to increase transfer accuracy, contrary to the findings in Colombo *et al.* (2007), who show that accounting for respondent preference heterogeneity reduces the magnitude of the transfer error.

The number of studies testing transferability based on choice experiments is very limited and largely confined to Australia (for an overview see, for example, Rolfe and Brouwer 2012). Benefit transfers across countries would allow much greater use of source studies. However, there are even fewer tests of the international transferability of nonmarket values, and almost only based on CV studies (e.g. Barton and Mourato 2003; Muthke and Holm-Mueller 2004; Ready *et al.* 2004; Brouwer and Bateman 2005; Bateman *et al.* 2011). In the USA, Rosenberger and Phipps (2001) find that CV-based transfer errors between states are larger than transfer errors within states. Johnston and Thomassin (2010) show that international transfers between Canada and the USA based on a CV meta-function outperform single-study value transfers in terms of prediction accuracy. In the only international choice experiment (CE) we know, focusing on commuting trips in Australia, Chile and Taiwan, Rose *et al.* (2009) find that the estimated choice models are not transferable across different countries and cultures.

The main objective of this study was to test the international transferability of the nonmarket values estimated through choice experiments and to identify how socio-economic adjustments and pooling source data help to reduce transfer errors. To this end, a common CE valuation design is applied at the same point in time in four different countries. The policy context in this study is the valuation of freshwater conservation in three water scarce and drought-prone Mediterranean countries (Italy, Spain, Greece) and Australia. The study aimed at minimising transfer errors by systematically increasing control for (i) unobserved and observed preference heterogeneity in the estimated choice models and (ii) single- and multicountry data sources. The methodological approach will be further clarified in the next section.

## 2. Transfer model and test procedure

Choice experiments fall in the class of attribute-based methods in which the deterministic part of utility $U^c_{\mathrm{ijt}}$ of individual *i* for good *j* in choice task *t* is described in (1) as a linear function of its attributes $X^c_{\mathrm{ijt}}$ and other explanatory variables $Z^c_{\mathrm{ijt}}$ (Train 2009):

$$U^c_{\mathrm{ijt}} = \beta^c X^c_{\mathrm{ijt}} + \alpha^c Z^c_{\mathrm{ijt}} + \varepsilon^c_{\mathrm{ijt}} \forall j \in D^c_{\mathrm{it}}, c = 1, \ldots, C \quad (1)$$

The superscript *c* indicates that preferences may vary across countries. In each choice task, the respondent is presented with a limited set of policy proposals $D^c_{\mathrm{it}}$, each offering a change in water availability and environmental quality (see Section 3). The stochastic term $\varepsilon^c_{\mathrm{ijt}}$ is assumed to follow an i.i.d.

extreme value distribution of type 1. To account for preference heterogeneity, we allow the preference parameters for the attributes to vary across respondents in random terms. Equation (2) describes the mixed logit probability of individual $i$ from country $c$ selecting alternative $j$ in choice task $t$. Here, $\Delta(\beta_i|b)$ represents the mixing density distribution for the attribute parameters $\beta_{\mathrm{i}}$.

$$P^c_{\mathrm{ijt}} = \int \left( \frac{\exp\left[\left(\beta^c_i X^c_{\mathrm{ijt}} + \alpha^c Z^c_{\mathrm{ijt}}\right)\right]}{\sum_{j \in D^c} \exp\left[\left(\beta^c_i X^c_{\mathrm{ikt}} + \alpha^c Z^c_{\mathrm{ikt}}\right)\right]} \right) \Delta(\beta_i|b) d\beta_i, \forall j \in D^c_{\mathrm{it}}, c = 1, \ldots, C \quad (2)$$

The equivalence of the CE design across countries offers the opportunity to conduct various types of transferability tests for the estimated choice models. Two main approaches are taken in this study.

In the first approach, the same model is estimated in each country, with and without socio-economic adjustments, in what we call a 'single-country' transfer approach. The estimation results are used to derive country-specific confidence intervals around willingness to pay (WTP) for alternative policy scenarios. 'Unadjusted' WTP values estimated from a standard conditional logit model including only the choice attributes are compared with the 'adjusted' WTP values estimated from a mixed logit model, accounting for (i) unobserved preference heterogeneity in the choice attributes and (ii) observed preference heterogeneity by including a common set of theoretically driven explanatory variables. In the latter case, the settings of the policy site are plugged into the estimated model for the study site to derive a confidence interval for WTP at the policy site. This addresses one of the limitations of existing transfer applications detected by Johnston and Duke (2010), who noted that CE transfer applications often omit socio-economic adjustments.

In the second approach, adjusted and unadjusted transfers are also compared, but the observations from two or more countries are combined into a 'pooled' model to predict transfer values in another country. These 'multicountry' transfer values are subsequently compared with the results from the single-country transfer approach. The 'multicountry' transfer approach requires us to test whether the utility parameters and their confounded scale parameters are equivalent across countries (e.g. Morrison *et al*. 2002) and therefore whether data from different countries can be pooled without producing biased welfare estimates.[1] We use the Swait and Louviere (1993) procedure to identify whether the data from two or more countries can be combined.

[1] The confidence intervals obtained in the single-country approach are independent of scale. Hence, the potential transfer errors obtained in the previous step are a consequence of differences in preferences across countries.

## 3. Choice experiment design

The CE was developed by four research groups over a 6 month time period in order to inform national and regional decision-makers about the nonmarket value of changes in freshwater availability. The case studies involved the Po basin in the Italian region Emilia-Romagna, the Serpis basin in the provinces Alicante and Valencia in Spain, the Kalloni catchment on the island of Lesvos in the Greek Aegean Sea, and the Fitzroy basin in Queensland, Australia. The case studies were selected after careful screening by international water experts in the European research project AquaMoney. The south European case studies share similar challenges in that the target of achieving good ecological water status as required by the European Water Framework Directive (WFD) is compromised by structural water scarcity and competing demand for water use. Similar conditions hold for the Fitzroy basin in Australia where high rates of extraction and commitment limit availability and ecological function in drought years (Rolfe and Windle 2005). These conditions are expected to be exacerbated by the impact of climate change and growing water demand.

Preceding the survey, focus groups were organised to discuss the specific CE design with lay public in each country, followed by two to three rounds of pretesting focusing specifically on the selection of attributes and levels, and clarity and credibility of the policy scenarios and baseline conditions. The CE involved four choice tasks, with two policy alternatives described in terms of three attributes. Two water allocation attributes were included in the CE, involving water conservation for domestic household use and the environment. In the latter case, water availability has a direct impact on the ecological status of water resources. In the former case, water conservation reduces the risk that households face water use restrictions in the future. It was explained to respondents that these water use restrictions would only affect outdoor water use such as sprinkling gardens, washing cars, filling swimming pools and other secondary uses of water during certain hours of the day in the summer. The full survey text introducing the CE, including the definition of the attributes and their levels, is included in the online Appendix to this study.

For both alternate water allocations, two possible levels of improvement were proposed in the CE. The ecological status of the freshwater resources in the region where the respondents live could be improved from current low stream flow levels and corresponding poor to moderate environmental quality to good and very good levels. These levels were based on the ecological standards in the European WFD and described in detail in the survey (see the online Appendix). Using qualitative attribute-level descriptions allowed the same design to be applied to the four case study regions where baseline conditions were similar. Domestic water supply security was represented as the frequency with which households face water use restrictions over the next 10 years. Future domestic water use restrictions in the baseline scenario are expected in four of the next 10 years based on climate change and demand projections. In the presented policy alternatives, this could be reduced to three, two or one year every 10 years, depending on additional measures taken to conserve and secure future water supply.

Respondents were told that taking additional measures to secure future water supply for households and the environment will come at a cost. The payment vehicle used here was an increase in the household water bill in the next 10 years. The trade-off therefore was the price that respondents were willing to pay for the presented domestic use-related water security benefit and the nonuse-related water conservation benefit for the environment. Six price levels were used of equal increments of 20 Euros on top of a respondent's annual water bill. The price levels were based on pretests of maximum WTP for price increases and available information about the average water bill.[2] Table 1 presents the attributes and levels applied in the CE.

In the CE, respondents were asked to choose between two possible policy alternatives in which water supply in the next 10 years was conserved for domestic or environmental use compared to a baseline situation where water scarcity problems remain the same or deteriorate. The baseline option described the situation that currently exists with zero additional cost. The two policy alternatives in each choice task showed possible improvements for either one or both water uses at the same time with an associated cost within a 10 year time frame. Hence, respondents were asked to value improvements in

**Table 1** Choice experiment attributes and levels

| Attributes | Levels |
|---|---|
| (1) Ecological status related to water flows | Poor or moderate (baseline) |
| | Good |
| | Very good |
| (2) Frequency of outdoor water use restrictions | Restrictions in 4 out of the next 10 years (baseline) |
| | Restrictions in 3 out of the next 10 years |
| | Restrictions in 2 out of the next 10 years |
| | Restrictions in 1 out of the next 10 years |
| (3) Increase annual household water bill | €0 (baseline) |
| | €20 |
| | €40 |
| | €60 |
| | €80 |
| | €100 |
| | €120 |

[2] In Australia, price levels were converted into Australian dollar equivalents.

public good provision that could be expected from water-saving measures. Respondents were asked to answer truthfully, and there were no indications of respondent mistrust related to the presented information or that others would not pay as well (free riding). Respondents were told that all users would pay. Confidence in the design was furthermore enhanced by the very low protest rate (see Section 4.1).

Although there is typically no explicit decision or provision rule presented in stated CEs, possibly undermining their incentive compatibility (Collins and Vossler 2009), there is no strategic incentive in our design for respondents to lie. Respondents were told that the survey served to inform actual decision-making (see the online Appendix), and were hence prompted to view their responses as consequential (Carson and Grooves 2007). In the focus group discussions and pretests, it became clear that water scarcity is perceived as a problem and also in the main survey a significant part of the target populations cared about the benefits of water conservation. Interviewers were furthermore carefully instructed to ensure that respondents understood that their answers could potentially influence local or regional decision-making related to water allocation. Respondents were told to answer each choice task independently of the previous one so that each choice in the choice sequence could be interpreted as an independent vote (Vossler *et al.* 2012). There was no mention of what would happen in the case of under- or overprovision of the public good (Rondeau *et al.* 1999), in view of the fact that the design captured incremental changes in the existing provision level for which respondents were already paying. The payment mechanism was applied in several other stated preference studies before (e.g. Genius *et al.* 2008; Brouwer *et al.* 2010a,b). During the pretest, there was furthermore no protest to the proposed payment mechanism, and no questions were raised related to the provision point mechanism.

Respondents were given 4 choice tasks following the recommendation by Louviere *et al.* (2003) to limit the cognitive burden of the choice experiment.[3] Using a D-efficient fractional factorial main-effects design, the alternatives and attributes were combined over 6 sets of 4 choice tasks. The allocation of one of the six sets across respondents was random. An example choice card is presented in Figure 1.

The choice experiment was part of a wider questionnaire survey, which was implemented in each country over a 3 month time period (July–September 2008) through door-to-door interviews by hired professional interviewers, targeting a random selection of households living in urban and rural municipalities in the river basins. The number of interviews was 241 in Italy, 312 in Greece, 394 in Spain and 300 in Australia.

[3] The choice experiment in Australia used 5 cards, but the fifth card was the same as the first card to test choice consistency (Brouwer *et al.* 2010a). Because this card was shown at the end of the choice sequence, it could be excluded from the analysis and comparison presented here.

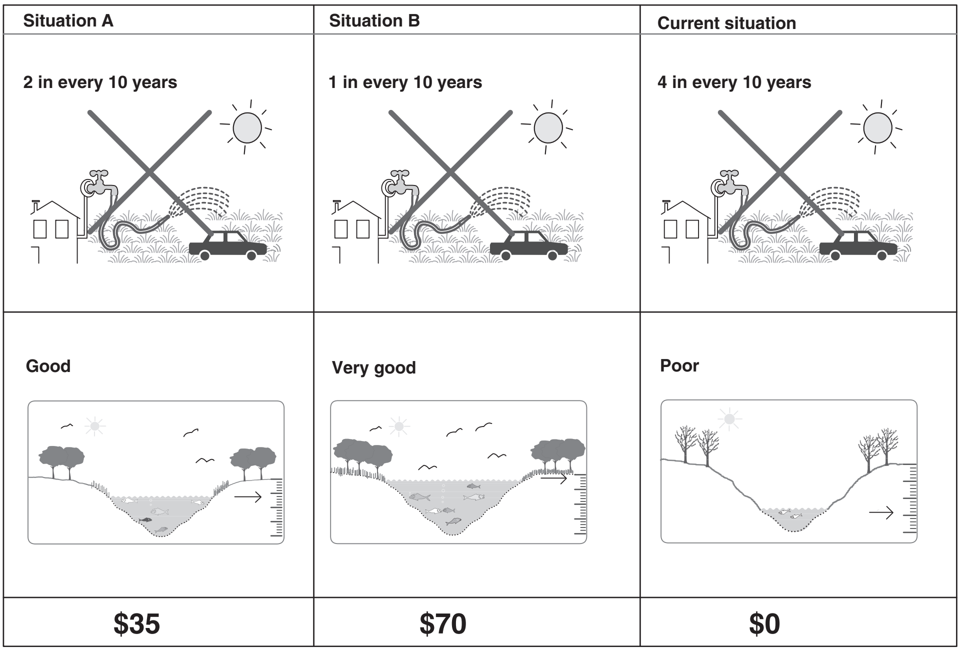


**Figure 1** Example of a choice card.

## 4. Results

### 4.1. Sample characteristics

The socio-demographic sample characteristics are presented in Table 2. Respondents in the four country samples differed significantly in terms of age and household income at the five per cent level.[4] In Europe, average household income was the lowest in Italy and the highest in Spain. Other differences between the samples include the relatively large share of male respondents and farmers in the Greek sample, and the relatively large share of female respondents in the Australian sample. The Greek case study area was more rural compared to the other case study areas, and when interviewers asked to interview the head of the household, women were more inclined to refer to their husband. Otherwise, the samples were fairly representative of the population from which they were drawn in each of the regions.

The number of respondents who consistently chose the opt-out during the four choice tasks for protest reasons (e.g. due to lack of trust that the increase in their water bill would be spent on water-saving measures) was very low across the three south European countries, and zero in the Australian sample, indicating that the trade-offs were accepted by nearly all respondents.

[4] Test results are available from the authors.

**Table 2** Summary statistics of the samples' main characteristics

| Socio-demographic characteristics | Greece | Italy | Spain | Australia |
|---|---|---|---|---|
| Share male (%) | 74.4 | 59.3 | 50.5 | 41.3 |
| Average age (min-max) | 42.7 (18–86) | 40.8 (18–82) | 46.0 (18–91) | 47.6 (18–91) |
| Average household size | 3.2 | 3.1 | 3.0 | 2.9 |
| Share with children (%) | 37.5 | 26.9 | 33.8 | 39.7 |
| Average disposable household income (€/year) | 19,565 | 18,530 | 21,990 | 33,375* |
| Share completed secondary school (%) | 21.9 | 43.0 | 38.3 | 46.3 |
| Share completed university degree (%) | 28.1 | 29.0 | 24.4 | 24.3 |
| Share unemployed (%) | 3.2 | 2.1 | 4.3 | 3.0 |
| Share retired (%) | 11.2 | 16.2 | 16.5 | 17.6 |
| Share full-time farmer (%) | 13.8 | 0.8 | 1.8 | 0.0 |
| Share member environmental organization (%) | 4.5 | 4.1 | 2.8 | 3.3 |
| Perceptions, beliefs, attitudes | | | | |
| Share perceiving water scarcity as a problem (%) | 52.3 | 21.6 | 65.9 | 41.6 |
| Share who experienced water use restrictions (%) | 55.1 | 29.7 | 21.0 | 43.1 |
| Average years of restrictions over the past 10 years | 5.5 | 3.7 | 1.6 | 3.9 |
| Share professionally affected by water scarcity (%) | 23.8 | 6.6 | 8.1 | 4.4 |
| Survey results | | | | |
| Share familiar with presented information (%) | 64.6 | 54.0 | 53.3 | 82.3 |
| Protest rate (%) | 3.5 | 0.8 | 2.0 | 0.0 |

*Gross instead of net annual income.

## 4.2. Estimated choice models

Three choice models are presented in Table 3(A–C) for all four countries: the conditional logit model including the attributes only (Model I), the same mixed logit model accounting for unobserved preference heterogeneity in the choice attributes (Model II), and the mixed logit model including a limited set of theoretically expected explanatory variables (Model III). In the mixed logit models, control is included for the panel structure of the data by allowing the random parameters to vary over individuals, but not over the choice sequence.

In all three models, the choice attributes are highly significant at the one per cent level, except for domestic water use restrictions in Australia, and have the expected signs. The likelihood of domestic water use restrictions and the increase in the water price are coded in their original values (the likelihood of a restriction varying from 0.4 to 0.1) and always have a

**Table 3** Estimated conditional logit choice Model I (attributes only) (A), Estimated mixed logit choice Model II (attributes only) (B), Estimated mixed logit choice Model III with socio-economic covariates (C)

| Variables | Greece | Italy | Spain | Pooled EU | Australia |
|---|---|---|---|---|---|
| (A) | | | | | |
| ASC | 0.762*** (0.191) | 1.192*** (0.178) | −0.153 (0.150) | 0.462*** (0.095) | 2.024*** (0.186) |
| Attributes | | | | | |
| Water use restriction | −2.414*** (0.490) | −2.493*** (0.571) | −3.980*** (0.447) | −2.979*** (0.281) | 0.220 (0.483) |
| Good ecological status | 0. 895*** (0.100) | 0.576*** (0.108) | 0.840*** (0.090) | 0.781*** (0.056) | 0.991*** (0.098) |
| Very good ecological status | 1.783*** (0.118) | 0.983*** (0.121) | 1.357*** (0.097) | 1.387*** (0.062) | 1.421*** (0.117) |
| Water price | −0.010*** (0.002) | −0.025*** (0.002) | −0.013*** (0.002) | −0.015*** (0.001) | −0.015*** (0.002) |
| Model summary statistics | | | | | |
| Log Likelihood | −981.298 | −952.582 | −1521.607 | −3562.071 | −969.228 |
| Number of observations | 1204 | 956 | 1544 | 3704 | 1180 |
| Number of respondents | 301 | 239 | 386 | 926 | 295 |
| (B) | | | | | |
| ASC | 8.975*** (1.417) | 3.550*** (0.456) | 7.060*** (1.146) | 6.378*** (0.667) | 18.461*** (4.973) |
| Attributes | | | | | |
| Water use restriction | −4.641*** (1.113) | −3.575*** (0.786) | −7.672*** (1.226) | −4.977*** (0.510) | −0.297 (0.670) |
| Good ecological status | 1.550*** (0.172) | 0.828*** (0.131) | 1.592*** (0.184) | 1.253*** (0.083) | 1.325*** (0.130) |
| Very good ecological status | 3.539*** (0.370) | 1.428*** (0.176) | 2.797*** (0.330) | 2.323*** (0.131) | 2.235*** (0.233) |
| Water price | −0.022*** (0.005) | −0.044*** (0.004) | −0.042*** (0.005) | −0.037*** (0.003) | −0.030*** (0.004) |
| St. dev. random parameters | | | | | |
| ASC | 8.500*** (1.260) | 3.597*** (0.452) | 10.770*** (1.385) | 7.459*** (0.636) | 12.848*** (3.417) |
| Water use restriction | 11.800*** (1.584) | 4.892*** (1.347) | 12.360*** (1.968) | 8.115*** (0.785) | 4.580*** (1.425) |
| Good ecological status | 0.243 (0.649) | 0.037 (0.629) | 0.882 (0.693) | 0.005 (0.302) | 0.025 (0.390) |
| Very good ecological status | 4.104*** (0.547) | 1.533*** (0.391) | 3.630*** (0.587) | 2.708*** (0.215) | 2.596*** (0.372) |
| Model summary statistics | | | | | |
| Log Likelihood | −761.924 | −814.117 | −1038.573 | −2725.486 | −785.822 |
| Number of observations | 1204 | 956 | 1544 | 3704 | 1180 |
| Number of respondents | 301 | 239 | 386 | 926 | 295 |
| (C) | | | | | |
| ASC | −1.010 (1.216) | 1.255* (0.709) | 3.689*** (1.160) | 0.124 (0.632) | 17.064*** (3.704) |
| Attributes | | | | | |

**Table 3** (*Continued*)

| Variables | Greece | Italy | Spain | Pooled EU | Australia |
|---|---|---|---|---|---|
| Water use restriction | −4.498*** (1.196) | −4.226*** (0.824) | −7.274*** (0.960) | −5.216*** (0.539) | −0.429 (0.682) |
| Good ecological status | 1.680*** (0.192) | 0.757*** (0.136) | 1.455*** (0.146) | 1.285*** (0.086) | 1.326*** (0.134) |
| Very good ecological status | 3.760*** (0.420) | 1.311*** (0.181) | 2.542*** (0.234) | 2.397*** (0.140) | 2.323*** (0.248) |
| Water price | −0.023*** (0.005) | −0.044*** (0.004) | −0.040*** (0.005) | −0.037*** (0.003) | −0.032*** (0.004) |
| St. dev. random parameters | | | | | |
| ASC | 9.974*** (1.545) | 3.114*** (0.419) | 14.187*** (2.102) | 6.618*** (0.541) | 12.423*** (2.664) |
| Water use restriction | 13.834*** (2.052) | 4.356*** (1.338) | 9.836*** (1.422) | 8.783*** (0.767) | 4.528*** (1.268) |
| Good ecological status | 0.400 (0.470) | 0.071 (0.569) | 0.076 (0.730) | 0.066 (0.313) | 0.114 (0.394) |
| Very good ecological status | 4.075*** (0.575) | 1.638*** (0.393) | 3.041*** (0.382) | 2.857*** (0.224) | 2.598*** (0.396) |
| Socio-economic covariates | | | | | |
| Restriction experience | 1.322 (0.963) | 1.821*** (0.640) | −4.990*** (1.373) | 2.136*** (0.664) | −3.923** (1.651) |
| Household income | 0.489*** (0.100) | 0.049 (0.031) | 0.265*** (0.060) | 0.192*** (0.031) | 0.090** (0.039) |
| Environmental disposition | 2.978*** (1.026) | 2.261*** (0.633) | 5.304*** (1.267) | 4.201*** (0.642) | 3.382 (2.103) |
| Model summary statistics | | | | | |
| Log Likelihood | −727.905 | −731.574 | −1016.230 | −2596.362 | −737.780 |
| Number of observations | 1204 | 956 | 1544 | 3704 | 1180 |
| Number of respondents | 301 | 239 | 386 | 926 | 295 |

$*P < 0.10$; $**P < 0.05$; $***P < 0.01$.

significant negative effect on choice probability. Utility for the policy alternatives hence increases if the frequency of domestic water use restrictions and price levels decrease. The environmental quality attributes have been dummy-coded with poor to moderate water quality as the baseline category. The positive signs indicate that respondents prefer improved environmental conditions of the water resources. The differences between the coefficient estimates for good and very good ecological status are all statistically significant based on the Wald test.

The alternative specific constant (ASC) is consistently significant and positive across most models, indicating preferences for a change away from the status quo. There exists a considerable degree of heterogeneity in the coefficient estimates between countries, including the ASC, but this is not unexpected in view of the differences observed in Table 2 between countries in terms of public perception of water scarcity problems, experiences with water use restrictions, and other case study conditions. The standard deviations of the random parameters are based on a uniform mixing distribution for the two ecological dummy variables (e.g. Hensher *et al.* 2005) and a normal distribution for the ASC and water use restriction attribute. One hundred Halton draws were used to improve the efficiency of the maximum simulated likelihood estimation procedure (Bhat 2001). Domestic water use restrictions are valued highest in the three models in Spain, while Greek respondents value very good ecological status the highest in all three models. Respondents in Italy are most sensitive to increases in the water bill.

In Model III, control for observed preference heterogeneity is included by interacting individual respondent characteristics with the ASC: respondent experience with water use restrictions (dummy variable with the value one if the household faced water use restrictions over the past 10 years), attitude towards environmental conservation (dummy variable with the value one if the respondent believes that water should be allocated to the environment first in times of scarcity instead of agriculture or industry) and household income (measured in thousands of Euros per household per year). Income has, as expected, a significant positive effect on choice probability in all country samples except in Italy (the higher the income level, the more likely someone is willing to pay extra). The respondent's disposition towards the environment has a significant positive effect in all three European countries at the one per cent level, except in Australia. Respondent experience with domestic water use restrictions has a significant positive effect on choice probability in Italy, but a negative effect in Spain and Australia. Hence, although water use restrictions are valued highest in Spain and not significantly different from zero in Australia, if someone in Spain or Australia suffered from water cuts in the past, the likelihood of choosing one of the water conservation policy alternatives and WTP is significantly lower.

Testing the equality of the estimated choice models using the Swait and Louviere (1993) method shows that the null hypothesis of equal preference parameters is rejected in all cases.[5] Similar results have been found, for example, in Morrison *et al.* (2002) and Colombo *et al.* (2007). Given this outcome, transferring the models is expected to result in prediction errors. The question is how large these errors are and what can be done to reduce them by accounting for preference heterogeneity and socio-economic adjustments.

### 4.3. Willingness to pay estimates

Implicit prices reflecting marginal WTP and mean WTP to allocate water for domestic and environmental use according to specified policy scenarios were calculated based on the estimated choice models. Results for marginal WTP values are similar across the three models. Differences across countries are illustrated in Figure 2, showing implicit prices and their 95 per cent confidence intervals based on the mixed logit models including covariates. Standard errors and confidence intervals are constructed applying the Krinsky and Robb (1986) procedure using 10,000 draws (Hole 2007).

Marginal WTP to reduce the likelihood of domestic water use restrictions over the next 10 years is significantly different from zero in south Europe, but not between the three south European countries. This implies that securing domestic water use is valued the same across these countries.[6] Marginal WTP to improve the ecological status of water resources over the next 10 years, either to a good or very good state, differs significantly between the European countries and is the lowest in Italy and the highest in Greece. Marginal WTP for reaching a very good ecological water status in these two south European countries is also significantly different from the value found in Australia, while no significant difference can be detected between Australia and Spain. Pooling the three European samples improves the transferability of the environmental flow values between Europe and Australia.

The compensating surplus (CS) welfare measures of different policy scenarios are also estimated (e.g. Bennett and Blamey 2001) based on Models I-III. The scenarios differ to which degree they capture domestic use values and environmental nonuse values. Two policy scenarios focus on an increase in domestic water use security (Scenarios 1 and 2), two on an improvement of the environmental conditions of water resources (Scenarios 3 and 4), and one policy scenario (Scenario 5) consists of a combination of the two. Mean WTP values for the five policy scenarios are presented in Figure 3, again based on the mixed logit model including covariates. The welfare measures are relatively high compared to the highest price level in the CE. This is partly due to the inclusion of the positive ASC in the welfare calculation procedure and respondents' eagerness to move away from the status quo, but also because of the relatively high share of respondents who chose one of the two hypothetical policy scenarios even at the highest price level, especially in

[5] Test results are available from the authors.
[6] Differences were tested using the *t*-test and the Poe *et al.* (2005) test.

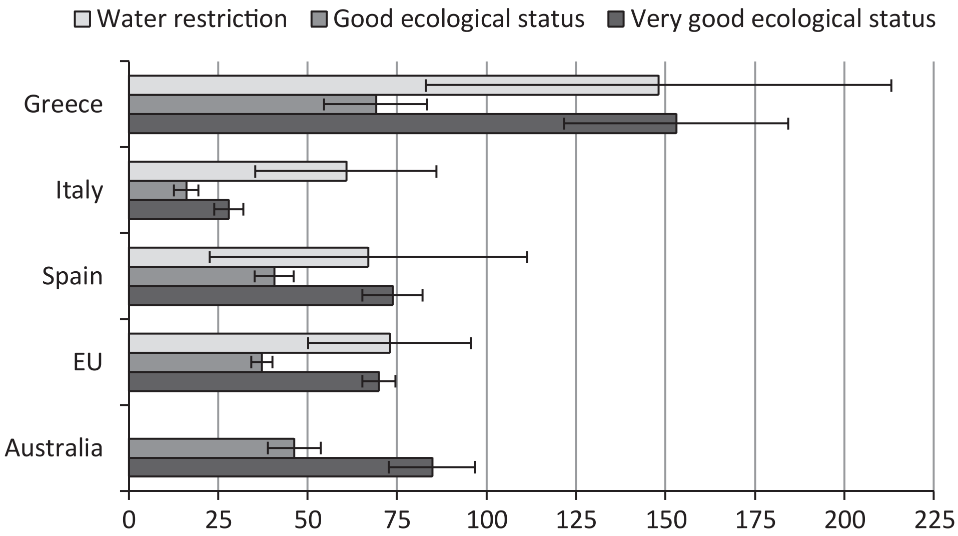


**Figure 2** Marginal WTP values (€/household/year). Note: error bars represent Krinsky and Robb 95 per cent confidence intervals.

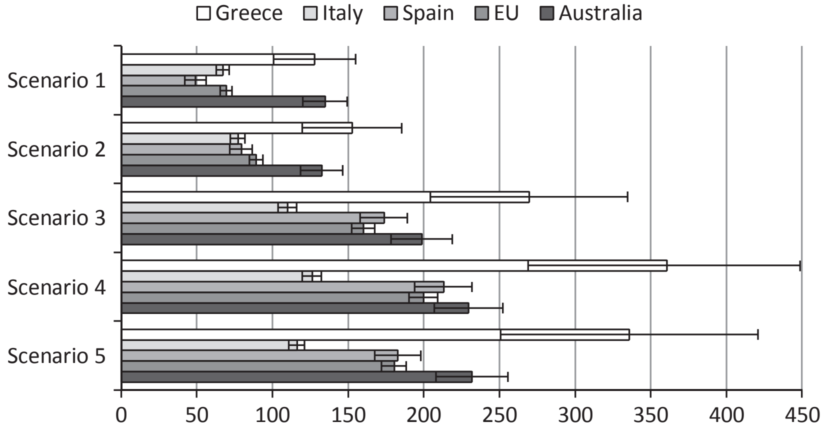


**Figure 3** Compensating surplus estimations for alternative policy scenarios (€/household/year). Note: error bars represent Krinsky and Robb 95 per cent confidence intervals.

Greece and Australia.[7] Despite thorough pretesting, the latter suggests the presence of a fat tail and a high choke price even though the bid levels were based on increments of the current average water bill up to 80 per cent. Compared to disposable household income, the highest WTP values are, on average, <1 per cent.

[7] Two-thirds of the estimated WTP values in Figure 3 are higher than the highest price level in the CE, mainly in Greece and Australia. The share of respondents who accepted to pay the highest price level in these two samples is 84 and 90 per cent, respectively.

As expected, the estimated CS increases when improving water use security (Scenario 2 compared to Scenario 1) and environmental water conditions (Scenario 4 compared to Scenario 3) and is highest for the fifth policy scenario where water is secured for both domestic use and environmental nonuse, except for Australia because of the nonsignificance of the domestic water use restriction. The WTP values are significantly different among the three south European case studies across the five policy scenarios. The same applies to the observed differences between Australia and the three south European countries and their pooled sample, except between Australia and Greece.

## 4.4. Transfer results

### *4.4.1. The 'single-country' approach*

We examine differences in prediction errors when transferring 'unadjusted' WTP values estimated from the standard conditional logit Model I, including the choice attributes only (Table 3A), compared to 'adjusted' value transfer using the estimates of the mixed logit Models II and III (Table 3B, C). The transfer results are presented in Table 4. Transfer errors are calculated using a standard approach (e.g. Bateman *et al.* 2011).

Among the three south European countries, transferring values based on Model II is superior to the unadjusted value transfer based on Model I in 57 per cent of the 30 cases. Accounting for socio-economic characteristics in Model III reduces the transfer error in an additional 8 cases (27 per cent). This is a substantial deviation from the results found in Johnston and Duke (2010), where the transfer error decreased in just over 50 per cent of the 30 cases they examined as a result of socio-economic adjustments. Note that income was included here like the other covariates as an interaction term with the ASC, meaning that it serves as a demand shifter explaining choices for the policy alternatives over the opt-out, not preference heterogeneity around the mean attribute estimates. Johnston and Duke (2010) included a dummy variable for high-income households (>$60,000) in their choice model, both as an interaction term with the ASC and the policy cost.[8] We find that including control for both unobserved and observed preference heterogeneity in Model III for the three south European samples yields an improvement in transfer errors in 70 per cent of the cases compared to Model I. That is, transfer errors are reduced in a majority of the cases when adjusting for preference heterogeneity and socio-economic characteristics at the policy site using choice Models II and III compared to Model I, also for transfers between the south European countries and Australia.

[8] Also Colombo *et al.* (2007) included income as an interaction term with the ASC, while income was not included as a covariate in Colombo and Hanley (2008). Instead, a dummy was used indicating whether or not a respondent was an active worker.

**Table 4** Absolute transfer errors (%) between countries across alternative policy scenarios based on Models I, II and III

| Predicting WTP in: | Scenario | Based on estimated choice model from: | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Italy | | | Spain | | | Australia | | | Pooled south Europe | | |
| Greece | | I | II | III | I | II | III | I | II | III | I | II | III |
| | 1 | 47 | 52 | 49 | 62 | 43 | 49 | 2 | 10 | 47 | 53 | 48 | 53 |
| | 2 | 50 | 54 | 49 | 48 | 46 | 49 | 16 | 20 | 48 | 48 | 50 | 54 |
| | 3 | 59 | 55 | 56 | 69 | 34 | 56 | 16 | 5 | 44 | 63 | 45 | 54 |
| | 4 | 67 | 64 | 64 | 65 | 43 | 64 | 14 | 14 | 49 | 66 | 55 | 60 |
| | 5 | 65 | 65 | 64 | 46 | 48 | 64 | 34 | 29 | 50 | 57 | 58 | 60 |
| | | Greece | | | Spain | | | Australia | | | Pooled south Europe | | |
| Italy | | I | II | III | I | II | III | I | II | III | I | II | III |
| | 1 | 90 | 109 | 94 | 27 | 19 | 11 | 93 | 87 | 85 | 6 | 46 | 31 |
| | 2 | 98 | 118 | 97 | 3 | 17 | 11 | 67 | 74 | 68 | 29 | 47 | 30 |
| | 3 | 142 | 120 | 132 | 25 | 45 | 39 | 182 | 131 | 141 | 28 | 69 | 67 |
| | 4 | 203 | 176 | 192 | 6 | 58 | 56 | 162 | 138 | 151 | 72 | 96 | 100 |
| | 5 | 189 | 190 | 184 | 57 | 50 | 49 | 91 | 105 | 101 | 101 | 94 | 87 |
| | | Greece | | | Italy | | | Australia | | | Pooled south Europe | | |
| Spain | | I | II | III | I | II | III | I | II | III | I | II | III |
| | 1 | 161 | 75 | 71 | 37 | 16 | 19 | 165 | 57 | 58 | 67 | 8 | 3 |
| | 2 | 93 | 87 | 74 | 3 | 14 | 18 | 62 | 49 | 45 | 20 | 12 | 4 |
| | 3 | 225 | 52 | 65 | 34 | 31 | 33 | 278 | 59 | 66 | 88 | 5 | 5 |
| | 4 | 185 | 75 | 84 | 6 | 37 | 40 | 146 | 51 | 56 | 51 | 2 | 3 |
| | 5 | 84 | 94 | 88 | 36 | 33 | 36 | 21 | 37 | 32 | 2 | 5 | 1 |
| | | Greece | | | Italy | | | Spain | | | Pooled south Europe | | |
| Australia | | I | II | III | I | II | III | I | II | III | I | II | III |
| | 1 | 2 | 12 | 79 | 48 | 47 | 43 | 62 | 36 | 30 | 46 | 32 | 22 |
| | 2 | 19 | 26 | 93 | 40 | 42 | 37 | 38 | 33 | 23 | 31 | 26 | 15 |
| | 3 | 14 | 5 | 51 | 64 | 57 | 56 | 74 | 37 | 35 | 59 | 40 | 32 |
| | 4 | 16 | 16 | 62 | 62 | 58 | 58 | 59 | 34 | 31 | 46 | 36 | 29 |
| | 5 | 51 | 41 | 89 | 48 | 51 | 48 | 18 | 27 | 19 | 19 | 24 | 16 |
| | | Australia | | | | | | | | | | | |
| Pooled south Europe | | I | II | III | | | | | | | | | |
| | 1 | 87 | 48 | 50 | | | | | | | | | |
| | 2 | 44 | 35 | 37 | | | | | | | | | |
| | 3 | 142 | 66 | 65 | | | | | | | | | |
| | 4 | 86 | 55 | 55 | | | | | | | | | |
| | 5 | 23 | 32 | 31 | | | | | | | | | |

Across all south European transfers, the average error is reduced from 76 to 64 per cent when accounting for unobserved preference heterogeneity in Model II. Adjusting values further for specific socio-economic characteristics in Model III does not reduce transfer errors any further. Transfer errors based on Model III are on average lower when predicting WTP in Spain based on the Italian sample (transfer errors vary between 18 and 40 per cent) and vice versa, predicting WTP in Italy based on the Spanish sample (transfer errors vary between 11 and 56 per cent). Transfer errors are the highest when predicting welfare measures in Italy (94–192 per cent) based on the estimated choice model in Greece.

Comparing transfer errors between the individual south European countries and Australia, similar results are found: Model II outperforms Model I in 60 per cent of the cases and Model III outperforms Model II in an additional 30 per cent of the cases. Transfer errors are higher when predicting WTP values in the three south European countries based on the choice data from Australia than the other way around. The average error of predicting WTP in Greece, Italy and Spain across the five policy scenarios based on Model I for Australia is 90 per cent, whereas the average error is 41 per cent when predicting mean WTP in Australia based on the estimated choice models for Greece, Italy and Spain. These errors are lower for Model II and III when transferring the estimated choice models from Australia to the individual south European countries (the average error across all policy scenarios is 60 per cent), while predicting WTP in Australia based on the more complex south European choice models accounting for preference heterogeneity increases the average transfer error to 50 per cent.

### *4.4.2. The 'multicountry' approach*

We also investigate to what extent combining stated preference data from two or more countries in a multicountry transfer helps to better predict welfare measures in another country. The results are shown in the last column in Table 4. The pooled south European sample refers to the two other south European country samples when predicting CS values in Greece, Italy and Spain separately or all three south European country samples when predicting WTP values for Australia. So, for instance, the pooled south European column in the top right-hand corner in Table 4 for predicting WTP in Greece consists of the merged choice data collected in Italy and Spain.

The multicountry approach yields a lower transfer error in a majority of 61 per cent of all transfer cases between the south European countries compared to individual country transfer, irrespective of the estimated choice model. This is only slightly better than the results found in the study by Colombo and Hanley (2008) where the pooled approach outperformed the single-study approach in 58 per cent of the transfer cases between two regions in Spain. Examining the transfers between the pooled south European samples and Australia, these appear to result in an improvement of the transfer error in 76 per cent of the cases. That is, WTP values in Australia are better predicted in three quarters of the cases when pooling the three south European samples than based on individual country sample transfers. Using the pooled country approach between the three south European countries reduces the overall transfer error on average by 35 per cent across the three different model types from 70 to 45 per cent, which is substantially better than the 15 per cent reduction found in Colombo and Hanley (2008).[9]

## 5. Conclusions

The main objective of this study was to test the international transferability of nonmarket values associated with public water conservation to inform water allocation decision-making. A common CE design was developed and implemented across different regions in south Europe and Australia, all regularly facing freshwater supply restrictions, to elicit nonmarket values for domestic water use and environmental flows. Specifically, we examined how adjustments for preference heterogeneity and socio-economic characteristics help to reduce transfer errors. On the one hand transfer errors are expected to diminish by including control for influencing socio-economic factors, while on the other hand the inclusion of context specific control factors may also reduce the transferability of the estimated values.

Two transfer approaches were tested: one based on the transfer from one country to another, and another in which estimates from a pool of countries are used to estimate the benefits in another country. Our expectation was that pooling source data may help to reduce the context specificity of the estimated transfer models and hence reduce transfer errors. In each approach, unadjusted and adjusted WTP estimates controlling for preference heterogeneity and socio-economic variables were compared. Although income, experiences with water use restrictions and environmental disposition are, as expected, significant determinants underlying choice behaviour and stated WTP, the observed differences in these factors are unable to fully explain the variation found in estimated WTP values between country samples.

Using the single-country approach, our results show that an overall reduction of just over 20 per cent in average transfer error is achieved when controlling for unobserved preference heterogeneity in the estimated choice models. Adjusting WTP values further for socio-economic differences between the study and policy sites does not necessarily always reduce the average transfer error further due to the inclusion of additional local context specificity. However, overall, transfer errors are still reduced from approximately 60–70 per cent. This provides additional empirical support for using socio-economic adjustment procedures in value transfer. Results from the multicountry approach also reduce transfer errors and are possibly even more promising, because they seem to reduce the degree of context specificity in the

[9] The average transfer error in Colombo and Hanley (2008) in their study of landscape restoration (410%) was also much higher than in our study.

estimated choice models. Here too transfer errors are reduced by just over 20 per cent when accounting for unobserved preference heterogeneity, and even further to 30 per cent when also including control for observed sources of preference heterogeneity.

The classic dilemma facing value transfer practitioners remains that as long as preference structures underlying choice models are not the same across sites or unstable in time, differences in good, site or population characteristics between study and policy sites will inevitably result in some degree of transfer error. These differences are more likely to play a role when trying to transfer stated preferences across different countries and cultures. This study nevertheless adds to the increasing empirical evidence base that, although not fully unavoidable, transfer errors can be reduced through socio-economic adjustments and data pooling and that source studies can be drawn from international contexts.

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## Supporting Information

Additional Supporting Information may be found in the online version of this article:

**Appendix S1.** Choice experiment.