Expert judgements and community values: preference heterogeneity for protecting river ecology in Western Australia*

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Abstract

Western Australia's Swan River is a complex asset providing environmental, recreational and commercial benefits. Agencies responsible for its management rely extensively on advice from experts, whose preferences may or may not align with those of the community. Using a choice experiment, we compared public and expert preferences for managing the river's ecology, and tested the application of budget-reallocation and personal-cost payment vehicles. The results indicate that the budget-reallocation method is a suitable payment vehicle for public and expert samples, although there are some differences to the more traditional personal-cost vehicles because of different tradeoffs involved. Modelling revealed heterogeneity in preferences. Expert and public preferences were statistically different from one another at the mean, but a significant amount of heterogeneity existed in the populations sampled. The differences in preferences across both public and expert groups suggests that the measurement of public values for the environment is still an important part of the management process, even when experts are providing advice.

1. Introduction

The management of iconic environmental assets is challenging, particularly when the asset is located close to a populated area. There are different and often conflicting uses to manage for, including recreational, commercial and environmental uses, and there are multiple stakeholder groups with varying preferences to consider. Governments often aim to manage for all of these preferences, and rely on expert advice about the most appropriate ways to do so.

Implicit in economic theory, and in arguments for democracy, is that the values or preferences used to evaluate different policy outcomes should be those of the general public. This is not inconsistent with experts playing key roles in supporting policy decisions by providing technical information, such as information about the functional relationships between actions and outcomes (Renn et al. 1993). Although there are initiatives that aim to facilitate and systematize the inclusion of public

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preferences in environmental policy, such as The Economics of Ecosystems and Biodiversity (TEEB 2011a, b), consideration of public preferences in policy decisions is often not systematic, representative or comprehensive. While this is obviously true of the technocratic governments of the formerly communist states of Eastern Europe (Pastorella 2016), it also applies to democratic governments such as Australia's. Numerous forms of public consultation have been implemented in democracies (Christie et al. 2012), but comprehensive consultation with the public (e.g. through detailed surveys) is too expensive to be practical for every policy decision (Adamowicz 2004; Rogers et al. 2013a).

In the absence of comprehensive public consultation, the role of experts may be broadened, at least implicitly. In many cases, experts are explicitly asked to go beyond the provision of impartial information and to make recommendations for policy decisions, which unavoidably involve value judgements. Implicitly or explicitly, these experts may be used as low-cost substitutes for public consultation (Rogers 2013). However, reliance on experts to make value judgements may pose risks if their values diverge significantly from those of the public.

In the context of invasive species risk management, Maguire (2004) noted that decision processes typically suffered from reliance on expert opinions to guide the process with their judgements being a mix of biological knowledge and personal preferences. Unstructured decision processes allowed for mixing the risks of what could happen (the technical elements) with the importance of potential outcomes (the values). The potential for the experts' judgements to incorporate their personal preferences was problematic when those judgements were being made on behalf of other individuals who might value things differently (Maguire 2004).

New systematic approaches to values elicitation from experts have since evolved. For example, the approach of Wallace (2012) and Wallace et al. (2016) places human values at the centre of environmental planning processes. However, the approach still relies on experts to act as stakeholder representatives for different groups, including the public, in the values elicitation process. While this approach makes the collection of value judgements explicitly distinct from technical judgements, it does not overcome the issue that a small number of experts may not accurately represent the values held by the broader community.

Discrete choice experiments have emerged as a convenient way to compare the value judgements of experts and the public. Carlsson et al. (2011) use a choice experiment to compare citizens' preferences with Environmental Protection Agency administrators in Sweden for valuations of a balanced marine environment and clean air. They found significant differences in willingness to pay (WTP), with values of environmental improvements tending to be higher for the experts. Rogers et al. (2013a) provide further evidence of differences existing between public and expert preferences in choice experiments on biodiversity values in the Southwest Australia Ecoregion, while Rogers (2013) found evidence of both convergence and divergence between public and marine scientists' preferences in a choice experiment of ecological values for two marine reserves in Australia. For the Ningaloo Marine Park, which was associated with higher levels of public awareness and charismatic attributes, values converged. For the Ngari Capes Marine Park, which was associated with a lower public awareness, there was a divergence in values, particularly for attributes that were not charismatic.

In the existing examples, payment vehicle treatments have varied across the public and expert samples. Selecting an appropriate payment vehicle is important in terms of the incentive-compatible properties of the choice experiment: the payment should mimic how funds would actually be collected if the hypothetical policy was to be enacted, thus making the scenario appear more realistic (Carson and Groves 2007). Rogers (2013) and Rogers et al. (2013a) used an increase in taxes as the payment vehicle, and asked both the public and expert samples to respond with their personal preferences; that is, acting as private citizens who consider the trade-offs implied by the policy cost with respect to their personal budget constraint. They recognise that this approach implicitly assumes there is a congruence between the experts' personal preferences and how they would actually act when making policy recommendations which may not always be true. However, the approach enables a direct comparison of the estimated public and expert values since the question being asked of each sample is identical.

Carlsson et al. (2011) also used a tax-based payment vehicle for the public sample, but asked their expert sample to recommend which policy they would choose in their professional capacity (recognising the cost that an alternative would impose on the public), rather than act as a private citizen (as someone who would bear that cost). The request for experts to provide professional recommendations is a more accurate reflection of how they would be consulted on an environmental policy. However, it means that the comparison with public values is indirect, as the experts are not bound by a personal budget constraint in making their choices. We expect their choices to be made using the mindset of a social planner, rather than a private consumer, although we expect their personal values to have an influence.

This study aims to contribute to the comparisons of public and expert values by providing further clarification of the suitability of expert judgement for reflecting community preferences. A choice experiment was used to measure and understand the differences between experts and the general public for the Swan Canning River System in Western Australia. This is an iconic river system that flows through the state capital of Perth, and is subject to multiple uses and complex management issues. The ecological health of the river is affected by a wide range of pressures including: nutrient and organic inputs, contaminants, foreshore degradation, invasive flora and fauna, decreased environmental water flows and climate change (e.g. Dollery 2013). In this study, values held by a sample of the West Australian population were compared with values held by two expert subsamples, namely environmental specialists and planners. Different types of experts were sampled to determine whether one type might better reflect public preferences than the other: it was hypothesised that people with pro-environmental sentiments may self-select themselves into environmental professions (e.g. Groom et al. 2007), and that this may lead to environmental specialists having stronger values for environmental protection relative to the public and other specialists.

An alternative payment vehicle was used to conduct the test in contrast to those used in previous public/expert comparisons. The choice experiment was framed for the public and expert samples using a budget reallocation payment, as opposed to a personal increased cost payment. The opportunity costs of the reallocation were demonstrated to respondents in terms of money being drawn away from other Government portfolios, which deliver services that were important to them. The approach of asking the expert sample to make professional recommendations rather than personal choices overcame limitations of the studies by Rogers (2013) and Rogers et al. (2013a) that framed the tradeoffs for experts as if they were making choices as private citizens. At the same time, it overcame a limitation of Carlsson et al. (2011) where the opportunity costs imposed on expert choices were not very direct.

To understand the applicability of the payment vehicle selected, we first test whether the use of a budget reallocation payment vehicle significantly increases the unobservable component of respondents' choices – that is, increases error variance – compared to a traditional payment vehicle. An objective of best-practice design in choice experiments is to minimise error variance, so this is important to consider for a reallocation scenario: not only might different people have different preferences for the attributes being valued, but they might also have different preferences for the explicitly defined bundle of goods from which funds are being reallocated, and we do not observe the latter. We test for this by comparing, for the public sample only, values collected with both budget reallocation and personal cost mechanisms. Using the budget reallocation scenario, we then test whether particular types of experts (planners or environmental specialists) have preferences that are similar to one another, and whether expert and public preferences are statistically similar. The results of this analysis are then discussed from the perspective of if and when expert judgement is a suitable substitute for public preferences in environmental decision making.

2. Methodology

The Swan and Canning rivers formed the case study area. These rivers combine into a single drainage system that collects water across the Perth metropolitan region in Western Australia and large sections of its semi-rural surrounds. This study focussed on the ecological values of the river system. A related study also measured preferences for the recreational values of the rivers (Rogers et al. 2013b). A choice experiment was designed to evaluate the preferences of both experts and the general public for the rivers.

Choice experiments are a survey-based technique used to investigate the trade-offs that people are prepared to make between different goods or policy outcomes. Respondents are presented with hypothetical choice scenarios, from which they must select their preferred outcomes. Conditional logit models are used to quantify the trade-offs between attributes that are implied by the choices made. One of the attributes that respondents are asked to consider is the cost required to achieve a particular outcome. The inclusion of this cost allows calculation of how much people are willing to pay for the various outcomes about which they are asked. Choice experiments are well-established and widely applied as an economic tool to estimate values for environmental assets (Bennett and Blamey 2001, Bateman et al. 2002). Many rivers and wetlands have been valued in the past using this method, with Rolfe and Brouwer (2013) identifying 145 separate value estimates for river protection in Australia from choice modelling studies, and Brouwer (2009) identifying a further 20 studies in Australia valuing wetlands protection.

2.1 Attribute selection

The selection of attributes for the choice experiment was informed by: a literature review on the ecological characteristics of the rivers; information gathered from observing focus group discussions with 143 river stakeholders (including community members, recreational users, and government representatives) on the topic of river health and use, undertaken as part of a separate project; and, participatory workshops with the senior management of the Swan River Trust (the governing body for the rivers), to ensure the attributes were meaningful for decision making.

Three ecological attributes were selected based on their relevance as either an indicator of river health or because they are a charismatic feature of the river, the ability of the Swan River Trust to manage identified threats and pressures associated with the attribute, and the availability of information to define the attributes adequately. The attributes are described in Table 1 and include foreshore (riparian) vegetation, fish, and dolphins.

Foreshore vegetation is important for river health given its ability to reduce nutrient and sediment run-off into the rivers, serving to both benefit water quality and reduce erosion of the foreshore (Hancock et al. 1996). Foreshore vegetation also offers additional benefits to wildlife, particularly waterbirds who rely on the vegetation for habitat, shelter and food provision (Department of Water 2007). The fish populations, including for instance, the recreationally important black bream (*Acanthopagrus butcheri*) in the river also have important links to river health as they are sensitive to algal blooms that can result in fish-kill events (Borusk 2004; Zammit et al. 2005). Thus, these attributes are important indicators of river health and it was anticipated that they would be highly valued by environmental specialists. The broader community might also value the attributes for these reasons or for the amenity and recreational opportunities offered (e.g. natural scenery; recreational fishing).

Bottlenose dolphins (*Tursiops aduncus*) were included as an attribute primarily because of their charismatic status. At the time of conducting the study, there was a small resident population of approximately 20-25 individuals (Holyoake et al. 2010). The ecological role of this small dolphin population is unknown: while the dolphins are likely to be dependent on the health of the river, the degree to which their presence contributes to river health remains unknown. Moreover, linkages between the Swan River dolphins and the much larger population of non-resident bottlenose dolphins outside the rivers are also unclear, with further research being required, particularly in terms of establishing dolphin demographic and ecological vulnerability (Holyoake et al. 2010). Therefore, it was hypothesised that dolphins may be valued highly by the community as a charismatic species, but less so by environmental specialists who might focus on improving the vegetation and fish attributes which have better established links as indicators of river health. Understanding any differences in value for this attribute was important for the Swan River Trust to establish how their management should be prioritised relative to those attributes that are more important for overall river health.

Note that the attributes were all defined to be independent of one another, in terms of the potential to implement management actions that could benefit one attribute while having no (or very limited) effect on the others. For example, the condition of foreshore vegetation could be improved by weed control, revegetation and managing access; fish kill events could be reduced through mechanical oxygenation in certain parts of the river; and dolphin health could be improved through removal of litter (particularly fishing lines, known to cause entanglements and increase risk of viral infections).

[*Table 1 around here*]

2.2 Payment vehicle

A budget-reallocation payment vehicle was used in this survey due to its relevance for the expert sample, relative to a traditional coercive payment vehicle. In the latter case, the respondent is (hypothetically) asked to pay for some increased-cost from their own pocket to achieve some beneficial change in the attributes they value, with an implicit relocation of their expenditure on other items. Stated preference surveys should be designed in a manner that is "consequential" to the respondent in a way that means they have an incentive to truthfully reveal their preferences when making their responses (Carson and Groves 2007). Increased-cost payment vehicles have been favoured in this regard to ensure that there is an opportunity cost to the respondent, in terms of trading off their disposable income (at least hypothetically). However, for a sample of experts who are being asked to select options based on the advice and recommendations they would give in their professional role, an increased- (personal) cost is not consequential (though such a payment vehicle may be appropriate if the experts are being asked to respond with their own personal preferences, as in Rogers 2013).

Budget-reallocations have emerged as a way to represent a more realistic payment scenario for experts (Bergstrom et al. 2004; Morrison and Hatton MacDonald 2011; Nunes and Travisi 2009; Swallow and McGonagle 2006). The theoretical basis of the welfare foundations for tax reallocation are detailed in Bergstrom et al. (2004). We would not argue that the measures of WTP derived from a private payment and tax re-allocation are equivalent, but that they can both be the basis for estimates of the trade-offs that respondents are prepared to make to achieve environmental outcomes. Environmental programs are often funded by a reallocation of an existing budget, rather than by increasing costs, which could be viewed as more realistic for experts working in this funding environment. Further, the opportunity costs of a budget reallocation can still be viewed as consequential in the case where an expert is making professional recommendations, as the trade-off for funds is against other areas of (environmental and other sector) decision-making, rather than an irrelevant private payment.

To maintain the principles of consequentiality, a budget-reallocation must be defined explicitly with respect to where money is being reallocated from, so that the respondent realises there is still an opportunity cost to him or her in that less money would be available for other things that they value. We addressed this by specifying a reallocation bundle that drew funds evenly from the major State Government portfolios (Table 1, Figure 1), such that the trade-off should have at least some consequence to all respondents. Specifically, respondents were advised that, under the survey's hypothetical scenario, any additional costs of managing the rivers would be met through reallocating funds evenly from the following main State Government sectors: (1) education; (2) health; (3) community amenities, safety and welfare; and (4) transport, communication, recreation, energy and other affairs. These four sector groupings each comprised of roughly one quarter of the WA budget in 2011-2012, meaning that an even reallocation would see equal amounts of money being taken from the four sectors.

[*Figure 1 around here*]

In the public survey, a survey treatment was also included where a set of choice questions were completed using an increased-cost (i.e. personal cost) payment vehicle[1]. This increased-cost was defined as a collection of "funds from West Australian households through a combination of: increased taxes by Commonwealth and State governments; higher rate payments to local councils; or higher prices for goods and services as businesses associated with the rivers meet more stringent environmental standards". Note that only the budget-reallocation data, which applied to both public and expert subsamples, was used for the comparison of public and expert WTP, but the availability of both payment vehicles for the public sample allows us to investigate whether the use of a budget re-allocation framework induces greater uncertainty in responses, via a consideration of the relative size of error variances.

2.3 Survey design and administration

The choice scenarios were designed with three options: a status quo option and two others (Figure 2). The experimental designs for the surveys were prepared using NGene (Rose et al. 2008). A Bayesian D-efficient design (see Scarpa and Rose 2008) was generated for the public survey[2] (D-efficiency statistic 0.219) and a D-efficient design was generated for the expert survey[3] (D-efficiency statistic 0.216). For the public survey, the same design was used for both budget-reallocation and increased-cost versions of the experiment, with 24 choice scenarios blocked into six groups of four. The expert design also comprised of 24 scenarios, but was blocked by a factor of two.

[*Figure 2 around here*]

A split-design approach was used to compare public and expert preferences (Table 2 identifies subsample descriptions and names). Two public subsamples were collected to manage potential ordering effects related to the payment vehicles: Public1, where respondents were presented with a block of four budget-reallocation scenarios, followed by four increased-cost scenarios; and, Public2, where the order was reversed. In each case, the block of questions from the 6 available was selected at random, and across the sample the full design was seen for both scenarios. For the expert subsamples (Environmental specialists and Planners), only the budget-reallocation payment vehicle was used, and respondents each saw 12 choice scenarios (a higher cognitive capacity was assumed, such that experts could manage with this many choice scenarios).

[*Table 2 around here*]

In each survey, first, the purpose of the survey was defined. For the public surveys, the purpose was to identify the values that West Australians place on the ecological features of the rivers, and to communicate the key findings of the research to management bodies. Similarly, the experts were informed that the purpose was to identify their values, but they were also made aware that the

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[1] We were unable to test this permutation on the experts due to the small size of the available sample.
[2] The MNL design used a uniform distribution of priors with parameter estimates of 0.01-1 for the first level increment for vegetation and dolphins, 0.5-1.5 for the second level increment, and 0.01-1.5 for fish. This acknowledged that we were reasonably certain preferences would be positive, but uncertain about magnitude.
[3] Priors used in the expert design were informed through the Swan River Trust participatory workshops: 0.5 and 1 for the two level increments for vegetation; 1 for the single level increment for fish; 0.3 and 0.6 for the two level increments for dolphins. We had more certainty about the relativity of these priors than those used for the public design.

results of their survey would be compared to those of the public surveys. In particular, experts were asked to complete the survey in a way that reflected the recommendations or advice that they would give based on their current position as an expert in the field (rather than as a general member of the Australian community).

Respondents were then asked about their past experiences with the rivers. This query was followed by information describing the relevant attributes in the choice experiment. A simple definition of each attribute was provided including a description of: the current condition of the attribute (see Table 1); the pressures or threats experienced by the attribute; and, examples of management approaches for dealing with the pressures. A description of the relevant payment vehicle was then given, and instructions on how to answer the choice scenarios. The choice experiment followed, along with debriefing questions and socio-demographics.

Before the full launch of the surveys, the public questionnaires were road tested across six focus groups consisting of 23 participants. The expert questionnaires were iteratively reviewed by Swan River Trust senior management. The full survey was administered online via The Online Research Unit, a market research company. For the public survey, members of the company's online panel were invited via email to participate in the survey, and offered a minor incentive for completion (entries into a prize draw). The sample was stratified based on the relevant population demographics for age and gender. Sampling was conducted in August 2012, with 664 respondents completing the two versions of the survey (Table 2).

Expert sampling took place between December 2012 and March 2013. The expert sample was constructed based on contacts provided by the Swan River Trust, with individuals identified as having specific expertise related to the Swan Canning River System. Experts were classified as "environmental specialists" if they had a background in environmental science, ecology or environmental management, or as "planners" if they had a background in land use planning or recreational and community management. This sampling strategy was adopted so that we would be able to explore whether preferences of river experts were dependent upon typology as an environmental specialist or planner. Note that another survey on the recreational values of the rivers was being conducted in parallel to this survey, and the experts were invited to participate in both. They were allocated first to the survey that most closely matched their expertise (i.e. environmental specialists were invited to respond to this ecological value survey first), and then were given the option to complete the alternate survey. Consequently, planners had already completed the recreational value survey before they responded to this survey. A total of 36 environmental specialists and 16 planners responded to this survey, resulting in 52 completed questionnaires (Table 2). We note that the expert sample sizes are comparatively small sample sizes relative to the public sample, reflecting the size of the available populations for sampling.

2.4 Data analysis

To test whether the budget-reallocation introduced more variance in the responses, we estimated a heteroscedastic multinomial logit model (Davis et al. in press, Swait 2006). This model allows the scale parameter to vary across individuals, enabling identification of the error variance by assuming that a relationship exists between the error variance and exogenous characteristics that vary across samples but are constant across alternatives: in this case which payment vehicle the sample received.

Formally, the probability that individual *n* selects option *i* from *J* alternatives is given by:

$$P_{ni} = \frac{\exp(\lambda_n \beta' x_{ni})}{\sum_j \exp(\lambda_n \beta' x_{nj})} \quad (1)$$

Where x_{nj} is a vector of observed variables relating to alternative *j*, and λ_i is the scale parameter, inversely related to the error variance. λ is conventionally assumed to equal 1, for identification, but the relative value of the scale parameter (and hence error variance) across groups can be identified if it is parameterised, that is:

$$\lambda_n = \exp(\alpha D_n) \quad (2)$$

Where D_i is a dummy variable identifying membership of the budget-reallocation subsample.

Data was pooled from the public samples using only the first four choice scenarios from each; that is, the four budget-reallocation scenarios for *Public1* and the four increased-cost scenarios for *Public2*. Note that because the blocks seen were selected at random, the full design for the increased-cost scenario and the full design for the budget-reallocation scenario are presented across the sample as a whole. By selecting this subset of the data, we minimised the potential for further noise to be introduced by ordering effects or fatigue if we had included, for example, the block of increased-cost questions that were answered *after* budget reallocation questions. Protest responses were removed from the sample, reducing the public sample size from 664 to 631. Protesters were defined as those who selected the status quo in all four choice occasions, and then in a follow up question selected an option that identified them as objecting to the specified payment vehicle or objecting to making these types of choices.

The scale parameter for the budget-reallocation sample (*Reallocation*) was allowed to vary, relative to the increased-cost sample, enabling estimation of the ratio between scale parameters, which is inversely related to the ratio between error variances. Parameters for the ecological attributes were held common between both samples: it is necessary to assume that some parameters in the models are common across samples for model identification, and it is reasonable to assume that the relative rates of substitution between ecological variables would be consistent, irrespective of the payment vehicle used (e.g. you would not expect the amount of vegetation people are willing to trade off for more dolphin protection to be influenced by the payment form). The cost coefficients were estimated separately for each sample, as well as the ASCs (you might expect the payment vehicle to influence selection of the zero-cost status quo). We conducted a log likelihood ratio test to confirm if the restriction of constraining the ecological attribute parameters to be equal across the increased-cost and budget-reallocation samples was rejected (independent of any consideration of error variance), where a restricted model is rejected as being statistically similar to an unrestricted model if the test statistic (calculated as 2 x (*unrestricted model LL – restricted model LL*) is greater than the chi squared critical value (p=0.05).

A statistically significant scale parameter would imply that the error variance was different between the two samples, and enabled testing of the first hypothesis:

H1: Error variance is not statistically different between the budget-reallocation and increased-cost survey treatments.

To compare public and expert preferences, generalised multinomial logit models were estimated in WTP space, using the budget reallocation data only, with normally distributed random coefficients on the ecological attributes[4].

Following Scarpa et al. (2008) we specify, in this model (and abstracting from differences in samples for the moment), that the utility obtained by individual *n* from alternative *i* is given by:

$$U_{ni} = -\lambda_n \cos t_{ni} + (\lambda_n w_n)' x_{ni} + \varepsilon_{ni} \tag{3}$$

where λ is the scale coefficient, *cost* the measure of budget reallocation, w_n a vector of WTP for a set of attributes *x,* and ε a random error. This specification of cost may be non-standard, but one can interpret it within a standard utility framework. Private income (and hence private consumption goods) are unchanged across options. The budget-reallocation measure will induce a uniform re-allocation of funds from other sectors of public expenditure, as described to respondents. This will then result in a reduction in utility arising from those changes which will be traded off against changes in the river-specific attributes. The perspective of respondents on the consequences of the change in budget may vary across individuals, but this is no different to the possibility of the marginal utility of private funds differing.

This WTP specification is open to a number of interpretations. We employ a GMNL-II model (Fiebig et al. 2010) where the scale effect is assume to follow the form:

$$\lambda_n = \exp(\overline{\sigma} + \theta m_n + \tau \varepsilon_{0n}) \tag{4}$$

Where *m* is a vector of observable individual characteristics, and ε_{on} is distributed as a standard normal. As only relative scale effects can be identified, some normalization is required, and here $\overline{\sigma} = -\tau^2 / 2$ which means that the expected scale coefficient =1 when ϑm_n=0. It should be noted that formally one cannot differentiate the source of the heterogeneity represented by λ_n between error variance and heterogeneity in the cost coefficient, or some mixture of both. For simplicity here we refer only to scale heterogeneity.

We assume that the distribution of individual specific WTP values are normally distributed. Given our interest in identifying differences across public and expert samples, we extend the definition of w_n to reflect which sample an individual is drawn from. We assumed that the distribution across samples has a common standard error and that sample differences manifest themselves as a shift relative to a base category; that is, for attribute *a*, and sample *s*, the distribution of WTP (*w*) is given by:

$$w_{as} \sim N(b_{ab}, \sigma_a^2) + b_{as} \tag{5}$$

[4] Other models were tested including a multinomial logit, mixed multinomial logit, and scale extended latent class model specification. The results of these alternative specifications are available from the authors on request, and were largely consistent with those of the GMNL model reported in terms of the translation to practical conclusions and implications. The model selected provided greater transparency in interpreting the results with respect for their relevance to decision making.

where b_{ab} is the mean of the distribution for the base sample, for attribute *a*, and b_{as} is the shift in the mean of the distribution for sub-sample *s*. Estimation is with the GMNL command, in Stata 15 (Gu et al. 2013). Note that we assume the effect of different samples manifests itself only as shifts in the mean of the distributions, and not in the variance. Although it would be possible to have a more general model with both mean and variances differing, the relatively small sample size for the experts, and the use of a full variance -covariance matrix for the random effects within the model, precludes that.

The scale λ, was allowed to vary parametrically with a self-reported measure of confusion about the survey (*Confused*), where respondents were asked to answer yes or no to the question "Did you find the scenarios confusing or particularly difficult to answer?". All categorical variables were correlated in the estimation to ensure model invariance in relation to the choice of the base level (Burton 2018). Protest responses were again removed from the sample, reducing the total available sample size to 689 (9 protestors removed from *Public1* and 18 removed from *Public2;* there were no protesters in the expert samples).

Data from all samples were pooled, and we considered there to be 4 samples (s=1 to 4, i.e. Public1, Public2, EnvSpecialists and Planners). We took as the 'general' model one that allowed there to be differences in the mean of the distributions for all 4 samples, and then conducted a sequence of restrictions, imposing equality of the means for subgroups. There are a number of paths through which one can pass, moving from general to restricted. That selected was: (1) restricting the two public samples to have parameters constrained to be equal, which implies there are no ordering effects related to whether the budget-reallocation scenarios were seen first or second; (2) conditional upon the results from (1):- restricting the two expert samples to have parameters constrained to be equal; and (3) restricting all four samples to have parameters constrained to be equal. The ASC was allowed to vary by sample in all cases. Log likelihood tests were used to confirm whether the restrictions could be rejected.

This sequence enabled testing of our next three hypotheses:

H2: The means of the WTP parameters for the ecological attributes are not statistically different between the Public1 and Public2 samples.

H3: The means of the WTP parameters for the ecological attributes are not statistically different between the EnvSpecialist and Planner expert samples.

H4: The means of the WTP parameters for the ecological attributes are not statistically different between public and expert samples.

For the preferred model, a question of interest is how the distributions of the WTP for the different samples compare, not just in terms of their means, but the extent to which the distributions overlap; that is, to what degree do the populations of public and experts have common preferences. We consider this in two ways. Firstly, we retrieve individual-level parameters (Revelt and Train (2000); Train (2009, chap. 11)) and show them graphically, by sample. Secondly, we can directly estimate the portion of the expert sample distribution that lies within the 95% interval of the public sample estimate (which is used as the base category), by evaluating the cumulative normal distribution using the estimates of the mean and standard error for the WTP distributions for both samples; that is:

$$\Phi(z_{a1}) - \Phi(z_{a2}) \quad (6)$$

Where $z_{a1} = \dfrac{1.96 * \sigma_a - b_{as}}{\sigma_a}$ and $z_{a2} = \dfrac{-1.96 * \sigma_a - b_{as}}{\sigma_a}$

And b_{as} is the estimate of the deviation of the mean of the expert sample from that of the public, and σ_a is the (common) standard deviation of the random parameter distribution, for attribute *a*.

3. Results

3.1 Payment vehicle comparison

The heteroscedastic multinomial logit model results, testing for effect of payment vehicle on error variance in the public sample, are reported in Table 3[5]. The statistically significant coefficient on the ASC shows that individuals in the budget-reallocation scenarios are more likely to select options that result in an ecological improvement than those who responded to an increased-cost scenario. The coefficient on the error heterogeneity is not significant, implying that there is no difference in the error variances across increased cost and budget reallocation, and that we cannot reject the null hypothesis H1. This suggests that the use of the budget reallocation vehicle did not, as one could hypothesise, induce greater uncertainty in interpretation of the public's choices. We continue with the comparison of public and expert preferences using the budget-reallocation payment vehicle.

[*Table 3 around here*]

3.2 Public and expert comparison of willingness to pay

The data comparing WTP of the public and experts contained four subsamples: *Public1* who saw the set of reallocation choices first in order, *Public2* who saw the reallocation choices second in order, Environmental specialists (*EnvSpecialists*) and *Planners* (Table 2). The log likelihoods of the models used to test appropriate sample restrictions are reported in Table 4. As noted above, the restrictions employed are on the mean of the distribution of WTP for the environmental attributes only. Model 1 reports the log likelihood for the model where the means of the distributions of the ecological attribute parameters (and the ASC) are all allowed to vary by sample i.e. b_{ax} are freely estimated. Model 2 reports the log likelihood for the model where the means of the ecological attribute WTP distributions are restricted to be equal for the *Public1* and *Public2* samples, reducing degrees of freedom by five (i.e. $b_{a1}=b_{a2}$). We cannot reject the null hypothesis H2, that the means are equal (likelihood ratio=0.6; critical value=11.07 at p=0.05). Model 3 reports the log likelihood for the model where the means of the WTP distributions are equal for the two public samples, and also that the means are equal for the two public expert samples, but they are different from each other (i.e.

[5] Note that a log likelihood test revealed the restriction of the ecological attribute parameters to be equivalent for both public samples was not rejected, relative to a model that allowed all parameters to be separately estimated across samples (prob>chi^2 = 0.5246, 5 degrees of freedom).

b_{a1}=b_{a2} and b_{a3}=b_{a4}). This implies ten parameter restrictions compared to Model 1, and the restrictions are not rejected (likelihood ratio=6.44; critical value=18.31 at p=0.05), meaning we cannot reject the null hypothesis H3. Model 4 reports information for a model where the ecological attribute parameters are assumed to be equivalent for all four samples (i.e. b_{a1}=b_{a2} = b_{a3}=b_{a4}), reducing degrees of freedom by 15 relative to Model 1. This model restriction is rejected (likelihood ratio=36.78; critical value=25.00 at p=0.05), and means that we reject the null hypothesis H4. Model 3 is therefore the preferred model, reported in full in Table 5.

[*Tables 4 and 5 around here*]

The attribute coefficients in Table 5 are reported in WTP space and are scaled by a factor of 100; that is, the values reported are reallocation amounts in hundreds of millions of AUD per year, to be reallocated for a ten year period. Vegetation in good condition is valued the most by the public sample, with a willingness to reallocate up to $158 million per year for an increase in area from 20% to 60%. A reduction in fish kills events from two to one per year is valued similarly to an improvement from 75% to 85% of dolphins in good health, at over $40 million per year, while improving the number of dolphins in good health from 75% to 95% is valued at $59 million.

There is a statistically significant shift in the mean WTP for the experts for increases in the area of vegetation in good condition, and for a reduction in the number of fish kill events. The experts are willing to reallocate $117 million to $167 million more than the public for increases in the area of vegetation in good condition from 20% to 40% or 60%, respectively. This approximately doubles the amount the public are willing to reallocate. A similar result is seen for the reduction in fish kill events with experts willing to reallocate almost $90 million in total. Experts are not willing to pay more than the public for improvements in dolphin health.

The WTP associated with the mean of the ASC is statistically significant and negative, indicating that all samples had a general preference for ecological improvements. This preference was intensified for the *Public1* sample (who saw the reallocation questions second in the sequence), who were willing to pay approximately $315 million to generate some amount of ecological improvement rather than maintain the status quo, relative to the $252 million that others were willing to pay.

To further explore differences in public and expert preferences, it is useful to consider the distributions of the WTP. In estimation (and not reported in Table 5) one obtains the estimates of the lower-triangular matrix used in the Cholesky factorization of the variance covariance matrix of the random parameters (Gu et al. 2013). Table 6 reports the implied variance covariance along with an estimate of the standard deviation of the distributions.

[*Table 6 around here*]

One can gain an intuitive evaluation of the distributions by comparing the size of the mean of the coefficient with the standard deviation, to get some indication of the degree to which preference heterogeneity leads to sign reversal (Table 6). Equal (absolute) values implies that 16% of the sample has preferences with a different sign to the mean: a larger (relative) standard deviation implies more respondents will have a WTP different to the mean. For the public, there are clearly some parts of the distribution that head into the ‘opposite’ signs (e.g. Veg60 has mean of 1.58 while the SD is 2.1). Experts tend to have higher values at the mean, and hence less sign reversal.

We now consider the relative values of the public and expert samples. Statistically we find that the means of the WTP distributions for the experts are different to those of the public, but this still allows for the possibility that there is considerable overlap in distribution of preferences. We start by evaluating the proportion of the expected distribution of WTP of the experts that lies within the predicted 95% range of the values of the public (i.e. evaluate equation 6 for each attribute). In order to account for sampling variation we draw a sample of 10,000 sets of the relevant parameters using the parameter point estimates and their standard errors, and report the mean and standard deviation of the proportion derived from those 10000 simulations (Table 7). This shows that although the means may be different for the vegetation and fish attributes, there is considerable overlap between the two sets of preferences; that is, the preference structures of the public and experts are not completely segmented.

[*Table 7 around here*]

Secondly, we compare the distributions of individual specific preference parameters retrieved ex post, after estimation, split into public and expert subsamples (Figures 3-8 below). Note the experts' y-axis scale is different, as there are fewer individuals. Here we can clearly see that the experts are 'shifted' out from the public for vegetation and fish. But there is also a clear overlap in the distributions. For dolphins, it is clear that they look very similar.

4. Discussion

This study makes two contributions to the literature as well as providing non-market values for the protection of foreshore vegetation, fish and dolphins in the Swan Canning River System in Perth, Australia. First we tested a methodological issue about payment vehicles by comparing a budget-reallocation version against a personal-cost version. This is relevant in a policy sense because many policy changes are funded through budget reallocations, and in an experimental sense because budget reallocation mechanisms are more incentive-compatible for experts and policy makers who may be involved in choice experiments. Second, we applied the budget-reallocation payment vehicle in choice experiments to compare public and expert preferences. This enabled us to provide some nuanced understanding of whether and when expert advice may be suitable as a substitute for public value judgements. We also tested whether the preferences of experts varied by the type of expert.

The results indicate that the budget-reallocation method is a suitable payment vehicle for public and expert samples. The ability to constrain ecological attribute parameters to be equal across the public payment vehicle samples revealed that the reallocation did not change the preference structure that individuals held for the ecological attributes. Conditional upon assuming that marginal utilities of the ecological attributes are the same, we found no differences in error variance between the personal cost and budget re-allocation samples (Hypothesis 1). However, some differences were found that may imply the response to opportunity costs was not equivalent between the payment vehicles. The cost coefficients were different in magnitude, which is to be expected as the scale of opportunity costs for an individual's money was not equivalent to that of the reallocated public money. As well, the ASC coefficient was significantly more negative for the budget reallocation than for the increased personal-cost payment vehicle, showing that individuals were more likely to select ecological programs for a cost in a reallocation scenario. While this could imply that individuals were not completely responsive to the opportunity costs of the reallocation, the effect is likely to exist because the general specification of the reallocation bundle may have meant that not all elements of the budget transfers were relevant to each respondent. Future applications should attempt to identify and potentially tailor the most important elements of a reallocation bundle for each respondent.

The nested tests of differences in the means of the WTP values for the ecological attributes of the Swan Canning River System identify that the order in which the public saw the budget reallocation choice sets (before or after the private cost) had no impact on preferences (Hypothesis 2), that environmental specialist and planning experts' preferences were not statistically different from one another (Hypothesis 3), but that expert and public preferences were statistically different from one another at the mean (Hypothesis 4). However, a large amount of heterogeneity in WTP existed in the populations sampled leading to a large degree of overlap in preferences. This finding is consistent with the literature where it is widely acknowledged that the general public have heterogeneous value structures (Renn 2006), and there is some evidence of value heterogeneity among experts (Sandbrook et al. 2010). Comparisons of expert and public preferences for the environment find instances of both preference convergence (e.g. Colombo et al. 2009; Kenyon and Edwards-Jones 1998; Rogers 2013) and divergence (e.g. Carlsson et al. 2011; Decker and Bath 2010; Kenyon and Edwards-Jones 1998; Rogers 2013; Rogers et al. 2013a), supporting our discovery of statistical separation of mean public and expert preferences, but with some overlap in the distribution of those preferences.

For policy guidance these results suggest that, at least in this instance, expert preferences would not provide an accurate representation of public values for all ecological attributes. While both the public and experts valued the area of vegetation in good condition the highest, such that expert prioritisation would reflect public preference for policies to protect foreshore vegetation, the magnitude of WTP was doubled in the expert valuation, relative to the public. A similar result was noted for valuations for reduction in fish kill events, where experts were again willing to pay roughly twice as much as the public to reduce the number of events from two to one per year. While the public and experts were aligned in their valuation of the dolphin population, the divergence in values for the other attributes means using expert judgement to inform a benefit-cost analysis, for example, would be inappropriate.

Our results indicate that experts are biased towards overstating the value of protecting non-charismatic attributes, evidenced by their higher WTP for the fish and vegetation attributes relative to the public's WTP. However, the similarities of WTP for dolphins, the positive WTP for all attributes, and the overlap in the distribution of values for all attributes between the public and experts would mean that, for less important decisions, and particularly those that are not relying on quantitative input on social value for a benefit-cost analysis, expert judgements about the selection of programs to protect river ecology could suffice and public welfare would not be substantially impacted in an adverse way.

Indeed, some would argue that expert judgement should always be used in lieu of public valuation for non-market goods, particularly those with non-use value. There is a long-standing debate about the appropriateness of using stated preference techniques to inform decision making, summarised in the Journal of Economic Perspectives (2012, vol. 26, issue 4). Hausman (2012, p.44) argues that “public policy will do better if expert opinion is used to evaluate specific projects, including non-use value”. This perspective is based on a perception that, in the context of a stated preference survey, the respondents are unable to be sufficiently well-informed about the environmental issue they are valuing. While there are undoubtedly cases where it is difficult to describe the complexities of environmental systems in a brief survey instrument, in the context of our study we argue that respondents are sufficiently familiar with all of the ecological attributes – foreshore vegetation, fish and dolphins – to have sufficiently well-informed views about their values. And we find differences compared to the expert values: this highlights the danger in ignoring public preference entirely. Our findings point towards the potential for expert judgement to be a suitable low-cost substitute for eliciting public preferences in some environmental decisions, such as those using decision frameworks other than benefit-cost analysis, but not in all cases. There is more work to be done to define the circumstances where the two sets of values are aligned.

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Figure legends:

Figure 1. Image capture of the reallocation specification presented to respondents.

Figure 2. Example choice scenario from ecological survey with budget-reallocation payment vehicle.

Figure 3. Histogram of individual-specific WTP estimates for improving the percentage area of vegetation in good condition from 20% to 40%, for expert and public subsamples.

Figure 4. Histogram of individual-specific WTP estimates for improving the percentage area of vegetation in good condition from 20% to 60%, for expert and public subsamples.

Figure 5. Histogram of individual-specific WTP estimates for improving the percentage of the dolphin population in good health from 75% to 85%, for expert and public subsamples.

Figure 6. Histogram of individual-specific WTP estimates for improving the percentage of the dolphin population in good health from 75% to 95%, for expert and public subsamples.

Figure 7. Histogram of individual-specific WTP estimates for reducing the number of significant fish kill events from 2 to 1 per year, for expert and public subsamples.

Figure 8. Histogram of individual-specific WTP estimates for the ASC, for expert and public subsamples.

Table 1. Ecological and cost attribute descriptions.

Attribute	Description	Attribute levels	Coding[a]
Foreshore vegetation in good condition	In 2008, a foreshore assessment report of the Swan and Canning rivers showed that about 20% of the foreshore vegetation was in good condition, 50% in moderate condition, and the remaining 30% in poor condition	20% (500 hectares) in good condition*; 40% (1000 hectares) in good condition; 60% (1500 hectares) in good condition	baseline veg40 veg60
Average frequency of significant fish kill events	Over the past decade, there has been an average of 2 significant fish kill events in the rivers each year, where more than 1000 fish have been killed each time	2 events each year*; 1 event each year	baseline fish1
Health of dolphin population	In 2009, about 75% of the river dolphins were in good health, in terms of being free from known entanglements and not showing any obvious signs of impairment	75% (17 dolphins) in good health*; 85% (19 dolphins) in good health; 95% (21 dolphins) in good health	baseline dolphin85 dolphin95
Budget-reallocation: reallocation amount each year from the State Government budget, for the next 10 years	Revenue sourced by reallocation of shifting money within the current State Government budget, meaning that money will be taken away from other State Government sectors	$0*; $50million total; $100million total; $150million total; $200million total; $250million total; $300million total	continuous
Increased-cost: cost to you each year, for the next 10 years	Collection of funds from West Australian households through a combination of increased taxes by Commonwealth and State governments; higher rate payments to local councils; or higher prices for goods and services as businesses associated with the rivers meet more stringent environmental standards	$0*; $50; $100; $150; $200; $250; $300	continuous

Notes: The levels for budget-reallocation and increased-cost were roughly equivalent on a per household basis (e.g. $50 tax reallocated by household, or $50 increased-cost to the household).
*Indicates the status quo level for each attribute.

[a]Unless coded as continuous, variables are dummy coded and = 1 if present, or = 0 otherwise. For dummy coded variables, the baseline is the omitted attribute level in the analysis (i.e. the status quo level).

Table 2. Survey versions and sample descriptions.

Sampled population	Payment vehicle	Number of choice scenarios	Number of choice blocks	Sample size	Subsample reference
Public – WA	Budget-reallocation / increased-cost	4 / 4	6	343	Public1
Public – WA	Increased-cost / budget-reallocation	4 / 4	6	321	Public2
Expert – environmental specialists	Budget-reallocation	12	2	36	EnvSpecialists
Expert – planners	Budget-reallocation	12	2	16	Planners

Table 3. Heteroscedastic multinomial logit model testing variance associated with the budget-reallocation.

	Coef.	z	95% Conf.	Interval
Fixed parameters				
ASC – increased cost	-0.414	2.66	-0.719	-0.108
ASC – budget reallocation	-0.748	2.01	-1.476	-0.020
Veg40	0.800	6.17	0.546	1.054
Veg60	1.093	7.05	0.789	1.397
Fish1	0.244	3.01	0.085	0.403
Dolphin85	0.321	3.85	0.157	0.483
Dolphin95	0.658	5.29	0.414	0.902
Cost – increased cost	-0.006	-6.88	-0.008	-0.004
Cost – budget reallocation	-0.002	-1.69	-0.005	0.000
Scale parameter				
Reallocation	-0.299	-1.47	-0.700	0.101
Log likelihood	-2635.47			
N choices	2524			
N individuals	631			

Table 4. Log likelihoods of restricted models for the means of the environmental attributes for public and expert samples.

	Model	Log likelihood	Restrictions relative to model 1
1	Public1, Public2, EnvSpecialists and Planners all different	-2550.07	-
2	Public1=Public2, EnvSpecialists and planners different	-2550.37	5
3	Public1=Public2, EnvSpecialists=Planners	-2553.29	10
4	Public1=Public2=EnvSpecialists=Planners	-2568.46	15

Table 5. Pooled public and expert generalised multinomial logit model estimated in WTP-space.

	Coef.	z	95% Conf.Interval	
cost	-1			
Mean of random parameters				
veg40	0.880	6.980	0.633	1.127
veg60	1.583	11.380	1.310	1.856
dolphin85	0.424	5.850	0.282	0.565
dolphin95	0.586	5.730	0.386	0.786
fish1	0.447	6.290	0.308	0.586
ASC	-2.524	-8.780	-3.088	-1.961
Estimate of fixed parameters				
Expert x veg40	1.170	4.380	0.646	1.694
Expert x veg60	1.674	5.480	1.075	2.273
Expert x dolphin85	0.150	0.810	-0.212	0.512
Expert x dolphin95	0.314	1.330	-0.147	0.775
Expert x fish1	0.448	2.550	0.104	0.791
Public1 x ASC	-0.622	-2.320	-1.147	-0.098
EnvSpecialist x ASC	0.131	0.330	-0.639	0.900
Planners x ASC	-0.601	-1.610	-1.335	0.133
Scale parameters				
Constant	1.105	3.820	0.539	1.672
confused	-0.932	-3.990	-1.390	-0.474
Lower-triangular matrix L, for the Cholesky factorization, not reported				
τ	1.078	4.720	0.630	1.526
Log likelihood	-2553.2698			
N Choices	3172			
N Individuals	689			

Table 6. Random parameter distributions: Variance covariance matrix with significance levels, implied correlation coefficients in upper diagonal, and standard deviations

	Veg40	Veg60	Dolphin85	Dolphin95	Fish1	ASC	Standard Deviations
Veg40	1.06***	*0.94*	*0.43*	*0.31*	*0.85*	*-0.17*	1.03***
Veg60	2.01***	4.28***	*0.51*	*0.54*	*0.90*	*-0.26*	2.07***
Dolphin85	0.31**	0.74***	0.49***	*0.87*	*0.29*	*-0.36*	0.70***
Dolphin95	0.45**	1.56***	0.86***	1.98***	*0.39*	*-0.51*	1.41***
Fish1	0.81***	1.72***	0.19*	0.51**	0.85***	*-0.24*	0.92***
ASC	-0.51**	-1.56***	-0.73***	-2.10***	-0.65*	8.47***	2.91***

Table 7. Percentage of experts who have a WTP value that lies within the 95% interval of the public's WTP distribution, derived from 10,000 simulations: mean and standard deviations reported.

	Mean	SD
Veg40	77%	11
Veg60	87%	4
Dolphin85	94%	2
Dolphin95	94%	1
Fish1	92%	2