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Contingent Valuation Versus Choice Experiments: Estimating the Benefits of Environmentally Sensitive Areas in Scotland

Nick Hanley, Douglas MacMillan, Robert E. Wright, Craig Bullock, Ian Simpson, Dave Parsisson and Bob Crabtree

This paper reports results from a study of the economic value of the conservation benefits of Environmentally Sensitive Areas (ESAs) in Scotland. The main novelty of the approach taken is in comparing two direct valuation methods, namely contingent valuation and choice experiments, to value these benefits. The Contingent Valuation Method (CVM) is well-established as a technique for valuing the sorts of landscape and wildlife enhancements associated with ESAs. The CVM experiment reported here uses a dichotomous choice format, and includes a new correction for part-whole bias. Choice experiments are much less used as an environmental valuation technique. We note several advantages of such experiments over CVM, and then report characteristic values and 'programme values' estimated using the method. This application brings to light some problems in applying the choice experiment method. Finally, we discuss the issue of benefits transfer in the context of these two approaches to valuation.

1. Valuation Methods in Environmental Economics

Since the early 1970s, the use of environmental valuation methods has increased markedly. This increase has in part been due to external policy stimulation, such as the Superfund Act and Presidential Order 12291 in the USA; and the Department of the Environment's 'Policy Appraisal and the Environment' document in 1991. Recent moves to enshrine cost-benefit analysis within the statutory duties of the Environment Agency

■ Nick Hanley is Professor of Agricultural Resource Management, University of Edinburgh, Robert E. Wright is Professor of Economics, University of Stirling, Ian Simpson is a lecturer in Environmental Science, University of Stirling and Dave Parsisson is a graduate student at the Environment Department, York University. Douglas MacMillan, Craig Bullock and Bob Crabtree are at the Macaulay Land Use Research Institute, Aberdeen. The research reported on in this paper was funded by the Scottish Office Agriculture, Environment and Fisheries Department. We are grateful to all members of the Steering Group for helpful input, especially to Bob Henderson. The views reported in this paper are strictly those of the present authors, however, who also take full responsibility for all errors and omissions. Thanks to Jenny Milne and Begona Alvarez-Farizo for research assistance, and to Vic Adamowicz of the University of Alberta for much advice. Thanks also to two referees and Bruce Traill for comments on an earlier version.

(and in Scotland, the Scottish Environment Protection Agency) seem likely to continue this trend. This paper reports on the application of two direct valuation methods, namely contingent valuation and choice experiments, to estimate the wildlife and landscape benefits associated with the Environmentally Sensitive Areas scheme in Scotland.

Direct valuation methods rely on the concept of stated preference: namely, that individuals can be induced to reveal their true preferences for environmental goods through their behaviour in hypothetical markets. The Contingent Valuation Method (CVM) is now a well-established valuation technique in both North America and Europe (Hanley and Spash, 1994; Navrud, 1992). Respondents are asked their maximum Willingness to Pay (WTP) or minimum compensation sums for hypothetical increases or decreases in environmental quality (Mitchell and Carson, 1989). Much controversy surrounds the technique, both in terms of its ability to deliver reliable estimates of WTP, and the correct design of CVM surveys (Diamond and Hausman, 1994). Critical design issues are bidding formats (open- versus closed-ended) and the level of information provision. CVM has also been criticised for suffering from a part-whole bias problem (Kahneman and Knetsch, 1992). This occurs when respondents bid for a more inclusive category of the good being valued, rather than the good itself (for example, all rivers in Scotland rather than one river). Proponents of CVM have argued that such an effect is to be expected, since it reflects the substitution possibilities inherent in any neo-classical model of demand (Carson and Mitchell, 1995). Carson and Mitchell have argued that embedding should be controlled for by informing respondents about these substitution possibilities, and by issuing reminders about what good is being valued.

Finally, the National Oceanographic and Atmospheric Administration (NOAA) has issued guidelines on the design of CVM studies in environmental damage suits (Arrow *et al.*, 1993). Among the most important are that WTP estimates should be sensitive to the scope (scale) of environmental change, that in-person surveys be used, that a Dichotomous Choice (DC) framework be used for eliciting bids and that CVM results should be 'calibrated' against experimental or actual market findings. Whilst there has been much criticism of the NOAA guidelines, they seem likely to be influential in future CVM work. Criticisms can be made for example, of both the insistence on in-person surveys and the use of a DC framework. With respect to the former, well-designed mail surveys may well offer advantages over in-person surveys (MacMillan *et al.*, 1996). With respect to the latter, it is now well known that DC designs yield higher WTP amounts than open-ended (OE) designs. This may be due to so-called 'yea-saying' in DC designs (where respondents say 'Yes' to a bid amount greater than their true WTP to register an environmental vote); to preference uncertainty; or to strategic behaviour (under-bidding) in OE designs (Boyle *et al.*, 1996). OE designs are also expected to produce lower-variance estimates of WTP than DC alternatives.

Choice Experiments (CE) also make use of stated preference data. Respondents are asked to choose between different consumption bundles, described in terms of their attributes and the levels taken by these attributes. One of these attributes is typically a price term. By repeating such choices, and varying attribute levels, the researcher can infer four pieces of information:

(i) which attributes significantly influence choice

(ii) the implied ranking of these attributes
(iii) the marginal WTP for an increase or decrease in any significant attribute
(iv) implied WTP for a programme which changes more than one attribute simultaneously.

These economic interpretations are made possible by the combining of random utility theory (Luce and Suppes, 1965; Manski, 1977) with limited-dependent variable econometrics (Greene, 1990; Maddala, 1983). The CE method is closely related to conjoint analysis, a well-established technique in marketing, which estimates 'part-worths' for different characteristics of a product (Green and Srinivasan, 1990); the CE method as described here, however, seeks to place choices in a welfare-theoretic framework. Since the CE method is less familiar than the CVM method, we now describe it in more detail.

The CE method is an application of probabilistic choice theory (Ben-Akiva and Lerman, 1985), whereby we seek to model choices between a non-continuous set of choice possibilities (thus violating the continuity assumption of standard neo-classical choice models). Assume that utility depends on choices made from some set C. For any individual n, a given level of utility will be associated with any alternative i. Alternative i will be chosen over some other option j if $U_i > U_j$. Utility for any option is assumed to depend on the attributes Z of that option; these attributes may be viewed differently by different agents, whose socio-economic characteristics S will also affect utility. We can thus write:

$$U_{in} = U\ (Z_{in}, S_n) \qquad (1)$$

Assume now that the utility function can be partitioned into two parts; one deterministic and in principle observable, and one random and un-observable. Ben-Akiva and Lerman attribute this randomness to (i) unobserved attributes (ii) unobserved taste variations (iii) measurement errors and (iv) the use of instrumental variables (rather than the actual variables that appear in U(.)). Then (1) can be re-written as:

$$U_{in} = V\ (Z_{in}, S_n) + \varepsilon\ (Z_{in}, S_n) \qquad (2)$$

and the probability that individual n will choose option i over other options j is given by:

$$\text{Prob}\ (i \mid C) = \text{Prob}\ \{V_{in} + \varepsilon_{in} > V_{jn} + \varepsilon_{jn},\ \text{all } j \in C\} \qquad (3)$$

In order to estimate (3), assumptions must be made over the joint probability set of the error term. One convenient assumption to make is that the errors are Gumbel-distributed (McFadden, 1974). This implies that the probability of choosing i is given by:

$$\Pr(i) = \frac{\exp^{\mu V_i}}{\sum_{j \in C} \exp^{\mu V_j}} \quad (4)$$

and if V(.) is linear-in-parameters such that $V = \beta(X_n)$, where X is a vector of observable attributes, and ß a vector of parameters to be estimated, by:

$$\Pr(i) = \frac{\exp^{\mu\beta' X_{in}}}{\sum_c \exp^{\mu\beta' X_{cn}}} \quad (5)$$

Here, μ is a scale parameter, which is usually assumed to be equal to 1 (implying constant error variance). As $\mu \rightarrow \infty$, the model becomes deterministic.

Since CE models share the same random utility framework as Dichotomous Choice (DC) CVM models (Hanemann, 1984), the welfare estimates from each should be directly comparable. This is important for what follows, in that it implies that the estimates of WTP for the environmental good of interest in this paper should be the same when estimated by CE as when estimated by DC-CVM. In environmental applications of the CE technique, the attributes chosen have been related to the environmental management problem being studied. For example, Adamowicz *et al.* (1994) used the CE method to value water quality improvements on the Highwood and Little Bow rivers in Alberta, and included attributes such as fish size, water quality and terrain. Boxall *et al.* (1996), in a study of habitat changes in moose hunting areas, used attributes such as access, distance from home, forestry activity and moose population. Choice sets are usually designed such that attribute levels are kept perfectly orthogonal across choices. A decision must also be made as to whether to specify a 'main effects only' design, or to allow interactions between attributes.

Relative to CVM, CE offer three advantages:

(i) It is easier to disaggregate values for environmental resources into the values of the characteristics that describe the resource. This might be important from two perspectives; first, that managers and policy makers may be more interested in the marginal value of changing certain attributes (such as percentage of open space or age diversity in a forest) rather than the value of the forest itself. Second, knowing about attribute values could be very important in being better able to 'transfer' benefit figures from sites/contexts where empirical work has been carried out, to other sites of interest (Willis and Garrod, 1995).

(ii) CE avoids the part-whole bias problem of CVM, since different levels of the good can be easily built into the experimental design.

(iii) CE avoids the 'yea-saying' problem met in DC-CVM (Adamowicz, 1995).

2. The Environmentally Sensitive Areas Scheme and the Case Study Area

The ESA Scheme

Environmentally Sensitive Areas (ESAs) are designated areas of the UK which are of special landscape and/or conservation interest, where traditional farming methods are considered to be essential to maintaining this wildlife and landscape quality. In Scotland, 10 ESAs have been designated since 1987, and now cover some 1.4 million ha. Farmers may 'join' an ESA scheme by signing a 10 year agreement based on a plan which meets the conservation aims and objectives of the particular ESA. In return for agreeing to these restrictions on activities, farmers qualify for annual per hectare payments on two different levels: tier one (aimed at the preservation of conservation features at existing levels) and tier two (aimed at enhancement and extension of conservation features beyond existing levels). ESAs thus involve the state paying farmers to produce environmental public goods, in terms of wildlife and landscape quality, a notion which finds many echoes throughout the OECD (Hanley *et al.*, 1996a).

This paper reports on data obtained with regard to the Breadalbane ESA in Highland Perthshire.[1] Breadalbane ESA comprises 179,284 ha of mountain and valley lands. The land cover is made up of grasslands, heather moorland, wetlands, and birch and ash woodlands, with increasing amounts of conifer plantation in upland areas. Farming is a mixture of upland sheep and suckler cows plus intensive grassland cultivation on intensively cultivated grazing land close to the farmstead. ESA payments are conditional on the management of broadleaved and native woodlands, wetlands, herb rich pasture, heather moorland, dykes, hedges and archaeological features.

Predicting Environmental Change in the ESAs

The ESA prescriptions will produce quite complex changes in flora, fauna and landscape, and will also have implications for archaeological features not protected under existing legislation. Two stages of the research were thus *prediction* of changes to these features, and *representation* of these changes, in the form of with/without scenarios. With regard to prediction, the land area within each ESA was divided up into km^2 land class types, using the Institute of Terrestrial Ecology's land classification system. For each class, we predicted changes in land cover resulting from changes in management (for example, from changes in stocking rates or fertiliser use), using the NVC classification system for vegetative cover. Changes were predicted using succession models (Simpson *et al.*, 1996). The impact of these likely biological successions on the conservation status of each land class was then assessed, using three criteria: biodiversity (number of species per square metre), presence/absence of key indicator species, and relative rarity. These predictions were also discussed with local agricultural advisors and farmers. Changes in bird numbers/species type were predicted in consultation with the Royal Society for the Protection of Birds (RSPB), Scottish Office Agriculture, Environment and Fisheries Department (SOAEFD) and Scottish Natural Heritage (SNH). Changes in archaeological features were predicted in consultation with Historic Scotland. These changes were all then set in the context of 'with' and 'without' the ESA prescriptions, by predicting likely changes in the absence of the scheme.

Representation of these predicted changes was accomplished by producing 'information packs' for each ESA. These accompanied the CVM questionnaires, which

[1] This was part of a larger study, which also applied CVM and CE techniques to the Machair of the Uists, Benbecula, Barra and Vatersay ESA in the Western Isles. See Hanley *et al.* (1996b) for details.

gave background details on the ESA scheme in general. Changes were shown as *with* and *without* the ESA scheme in place, using colour photograph pairs produced in *Adobe Photoshop*. For Breadalbane, changes to the appearance of 'farmland' and 'moorland', to archaeological features and to the number and types of flowers were included.

3. Design and Implementation of the Surveys

The initial stage in the CVM survey was to conduct a pilot 'attitudes' survey amongst 300 respondents drawn randomly from the general public in England and Scotland. A large majority of the sample were in favour of paying farmers both to produce food and to look after the countryside. In terms of how environmental payments in ESAs should be made, the most popular choice was for visitors to these areas to be charged. However, due to the likely existence of non-use values for ESAs, and to practical and cultural problems of excluding beneficiaries from these areas, we chose the second most favoured bid vehicle, namely increases in income taxation. Focus groups were then used to pre-test the wording of the CVM questionnaire, and the photographs and text in the information packs.

The target populations for the CVM survey were three-fold: the UK general public, residents in the ESA and visitors. Responses were collected both by mailing out surveys to a random selection of addresses, and by in-person interviews. For residents, these interviews were conducted in people's houses. For visitors, they were conducted in popular tourist locations in or close to the ESAs. In general, the survey design follows NOAA guidelines, except that we compare open-ended and dichotomous choice formats, and mail shot with in-person surveys. The mean response rate for the mail shot was 44 per cent, and these responses made up 40 per cent of the total sample returned (thus in-person interviews yielded 60 per cent of the sample returned). Respondents were repeatedly reminded that they were being asked their WTP for the environmental improvements at one ESA only, and that extra spending would be necessary for all other ESAs and for all other environmental policies. Results from the open-ended CVM study may be found in Hanley *et al.* (1996a) and Alvarez-Farizo *et al.* (1996). Briefly, this survey performed well on a number of validity tests, and yielded mean WTP amounts of between £31.43 per annum (residents) and £22.02 per annum (general public), with confidence intervals of £20.62-£42.24 and £14.50-£29.54 respectively. Significant non-use values were found, in that people who had not lived in nor visited the area were still willing to pay to achieve the environmental gains of the programme. No significant difference between mail-shot and in-person responses was found. Results from the dichotomous choice survey are reported in section 4.

For the CE experiment, an initial pilot survey showed that face-to-face interviews were necessary to obtain satisfactory completion and understanding of the questionnaire. Interviews were undertaken by a market research company, with a sample drawn from residents and visitors in the ESA. A total sample of 256 interviews was obtained.

4. Dichotomous Choice CVM

The NOAA panel's espousal of the Dichotomous Choice(DC) design for CVM stems from two features of this design. First, that it is alleged to be incentive-compatible (Hoehn and Randall, 1987), and second that the decision framework it imposes (take-it-or-leave-it) at a given price is more akin to normal market transactions. However, DC designs are also known to result in upwardly-biased WTP estimates, due to the 'yea-saying' problem, whilst WTP estimates are also very sensitive to the experimental design and econometric procedure adopted, particularly with regard to the treatment of the upper tail of the distribution. In this study, we use the DC version of CVM to estimate WTP for the ESA for three groups: the general public, visitors, and residents. Data for the latter two groups were collected by in-person interviews; for the general public sample, they were collected by both in-person interviews and mail shots. DC design involves setting a price (bid) for the environmental improvement, and asking respondents if they would be willing to pay it. Possible responses are 'yes', 'no' and 'don't know'. In all cases, we use a reciprocal transformation of the bid, to prevent estimated WTP being negative over any of its range (Buckland *et al.*, 1996).

Respondents were first asked if they would be willing to pay anthing, even a small amount, for the environmental improvements shown. This established three groups of respondents; those prepared to pay in principle, non-payers and protesters. The proportion of those willing to pay in principle ranged from 56 per cent to 67 per cent, with highest percentages being recorded for residents and lowest percentages for the general public. Non-payers accounted for 12-33 per cent of respondents, and protesters 6-25 per cent. Most common motives for protesting were disagreement with the bid vehicle; and that the 'government should pay'.

Results

The bid curve was estimated using a step-wise log-logistic regression, having rejected both non-payers (since it is assumed that the upper asymptote of the bid curve is unity) and those who 'did not know' whether they would pay the bid amount. The zero value non-payers place on the ESA is allowed for by weighting mean WTP by the proportion of

Table 1 Logistic Bid Curves for DC Data: Breadalbane

Variable	*Coefficient*	*Asymptotic t-statistic*
GP-mail		
Bid	-0.818	-4.64
Envpref	-0.484	-2.12
Familiar	-0.486	-2.11
GP-in person		
bid	-0.014	-3.20
age	-0.354	-2.21
Visitors		
bid	-0.018	-4.08
envpref	-0.589	-2.10

Variable definitions: bid = bid amount (possible values in £s: 5, 10, 14, 21, 33, 47, 72, 103, 210, 490)
envpref = rank score for 'protecting the environment and countryside as a policy goal'
familiar = ordinal variable: 1 = lived in/visited area, 2 = never lived in or visited but heard of it, 3 = never heard of it
age = age of respondent in years

respondents in this group (Buckland *et al.*, 1996). The bid amount was forced into the step-wise estimation, since estimation of WTP is based on its parameter value. Table 1 gives the reduced-form equations for the two general-public samples, and for visitors, where the dependent variable is the probability of accepting the bid amount. As may be seen, 'bid' is always negative and significantly related to the probability of acceptance. Surprisingly, income was not significant in any of these three equations at the 95 per cent level, although it was always positively signed. Table 2 gives the estimates of WTP based on the information in Table 1.

It was not possible to estimate WTP for residents in Breadalbane using this DC design, since too many of the sample said 'yes' to the highest bid value: this reflects a failure in the design of the survey in the case of this group of respondents. Further references to residents' WTP values in this paper (e.g. in Section 6) thus refer to data from the open-ended CVM exercise.

Table 2 WTP Estimates for Breadalbane: DC Data

Group	*Median WTP (£)*	*Mean WTP (£)*	*95 Per Cent ci for Mean (£)*	*Sample Size*
General public, mail	0	47	34-61	325
General public, in-person	0	60	42-99	249
Visitors	62	98	53-135	235

Testing and Correcting for Part-Whole Bias

As was mentioned in Section 1, part-whole bias is often cited as a problem area in CVM. This, as will be recalled, occurs when WTP for a good is higher when that good is valued in isolation than when it is valued as part of a more exclusive group. In this study, the good being valued (the ESA programme in Breadalbane) is explicitly nested within the more exclusive good of 'all ESAs' in the questionnaire. A related issue is commodity mis-specification, whereby respondents value a more inclusive category of goods X rather than some member of this set x, where x is the good which the researcher is concerned with, possibly due to the symbolic nature of the good or poor questionnaire design (Kahneman and Knetsch, 1992). We can test for part-whole bias in the DC design framework. Respondents who said they were willing to pay the bid price £z were asked why they were willing to pay. Possible responses were:

(i) I especially care about the landscape/wildlife in Breadalbane

(ii) I care about the overall ESA programme

(iii) I care about the countryside in general

Respondents who answered (ii) or (iii) were then asked *"if you answered (ii) or (iii), remember that the £z extra would only go to Breadalbane ESA. Are you willing to pay this amount just for Breadalbane ESA only?"* Only those people who answered 'yes' to this question were kept as 'yes' responses in the bid curve. This procedure resulted in the re-classifying of around 10 per cent of 'yes' responses as 'no'. Re-estimating mean WTP, we get the results shown in Table 3. As may be seen, correcting for part-whole bias in this case results in a downwards adjustment in WTP.

Table 3 Original WTP Estimates and Estimates Corrected for Part-Whole Bias

Group	*Original WTP*	*Corrected WTP*
General public, mail shot	£47	£42
General public, in person	£61	£57
Visitors	£98	£73

5. Choice Experiment Results

The first step in implementing the Choice Experiment (CE) approach was to decide on which characteristics to specify in the design for each ESA. This was done by first considering which attributes were affected by the ESA management provisions. For Breadalbane, this list comprised broadleaved woodland, moorland, wetland, dry stone dykes and archaeological sites. These characteristics were then tested in focus groups to see whether respondents included them in their descriptions of ESA landscapes. Each attribute took one of two values: a level corresponding to our predictions for the 'no ESA management agreements' case, and a level corresponding to our forecast for the 'with ESA management agreements' case. See Simpson *et al.* (1997) for further details on these predictions. Eight price levels were used, based on the distribution of WTP from the open-ended CVM exercise. A main-effects,[1] perfectly orthogonal design was then constructed, creating pair-wise comparisons. This gave a possible (2^5 . 2^5) design size. Focus group work showed that respondents could cope with up to 8 choice pairs each, and this gave a final sample size of 256 persons. In each choice pair, respondents were asked to select choice A, choice B, neither (i.e., the status quo), or respond that they 'did not know' which option to choose. All interviews were carried out face-to-face in the ESA area, using the information packs used in the CVM exercise to provide background information. Data were also collected on respondents' socio-economic characteristics and membership of conservation groups.

Results are shown in Table 4. No *a priori* consensus exists on the appropriate functional form to fit in CE studies, so we experimented with both linear and quadratic forms (with a negative squared term on the tax variable). We also experimented with alternative specific constants representing choices A and B in both data-sets; however, including such constants made little change to the models, so only the 'constants included' versions are reported here. The ranking obtained from the CE is intuitively appealing in both the linear and quadratic models, with broadleaved woods ranked highest and archaeological features ranked lowest. This implied ranking accords exactly with the mean ranking scores obtained by a supplementary question which simply asked respondents to explicitly rank the five characteristics in order of importance (also reported in Table 4). Note that all characteristic parameter estimates are statistically significant, and that 'tax' is negatively signed, indicating that people prefer cheaper packages. The tax squared term is positive and significant, with the utility curve becoming positively sloped in utility/tax space at an additional tax of £181, which is greater than the highest tax value used in the choice sets.

Given that the squared term in the quadratic model is significant, this implies a preference for the quadratic model over the linear alternative. Marginal willingness to

[1] That is, ignoring characteristic interactions such as (broadleaved forest * drystone walls).

Table 4 Choice Experiment Results

Attribute	*Linear Model, Parameter Value (t stat)*	*Quadratic Model Parameter Value (t stat)*	*Implied Ranking, Both Models*	*Stated Ranking*	*'Marginal' WTP, Quadratic Model (£)*
Woods	0.575 (16.0)	0.576 (16.0)	1	1	50.46
Archaeology	0.075 (2.2)	0.076 (2.2)	5	5	6.65
Heather Moors	0.260 (7.5)	0.262 (7.6)	2	2	22.95
Wet Grasslands	0.236 (6.8)	0.238 (6.9)	3	3	20.85
Dry Stone Walls	0.128 (3.8)	0.129 (3.8)	4	4	11.30
Tax	-0.007 (-8.223)	-0.0137 (4.3)			
Tax^2	-	0.000038 (2.2)			
n	1480	1480			
Log L (max)	-1281.564	-1279.199			
Log L (0)	-1625.946	-1625.946			

Note: income, age and membership of conservation organisation were also included in the estimation, but results are not shown here. Alternative specific constants were also included.

pay figures varied from £82.85 for woodlands to £10.00 for archaeological sites in the linear model, and were obtained by dividing the parameter on the characteristic of interest by the parameter on 'tax', since this represents the marginal utility of money in the estimated indirect utility function. In the quadratic model, marginal willingness to pay varies from £50.46 for woodlands to £6.65 for archaeology. Note that the term 'marginal' with regard to the environmental characteristics should be interpreted with care, since it represents the value of a discrete change in the characteristic from 'policy off' to 'policy on'. Overall WTP for the ESA policy is calculated using the formulae provided by Morey *et al.* (1993). For the quadratic form this implies solving for the compensating surplus, CS, in the following equation:

$$V_1 = a_1(Y) - a_2\,(Y^2) + b(Q_1) = a_1\,(Y\text{-}CS) - a_2\,(Y\text{-}CS^2) + b\,(Q_2) = V_2 \qquad (6)$$

where V_1 is utility without the ESA, V_2 is utility with the ESA, Q_1 is environmental quality in the policy-off setting, Q_2 is environmental quality in the policy-on setting, and Y is income. This gives a value of £107.55 per household per year for the quadratic version. For the linear form the total programme value is more straightforward to calculate, giving a value of £182.84 per household per year. The quadratic estimate can be compared with the open-ended CVM estimate for residents of £31.43 and the dichotomous choice estimate for visitors of £98. Whilst the CE estimate is greater than either of these, it lies within the 95 per cent confidence interval for the visitors' WTP estimate (although it is outside the 95 per cent interval for residents).

6. Benefits Transfer

Benefits transfer is currently one of the most important questions in environmental valuation (Luken *et al.*, 1992). For valuation to be policy-useful at anything lower than the strategic/large project level, it seems essential that WTP estimates from 'study' sites be capable of being transferred to 'policy' sites with reasonable accuracy, given the high cost of rigorous valuation studies[1] and the time necessary to undertake original work. Bergland *et al.* (1995) note three approaches to benefits transfer: transfer of mean values; transfer of adjusted mean values; and transfer of benefit functions (bid curves). Bergland *et al.* were not able to show that any of these three methods was justified in their empirical study of two river sites in Norway. Transfer of mean values from this study to other ESAs is clearly only acceptable if (i) the policy site is identical to either of the two ESAs and (ii) the benefiting population is identical (Boyle and Bergstrom, 1992; Desvouges *et al.*, 1992). Even with respect to the first of these requirements, problems obviously arise since every Scottish ESA is unique in terms of the combination of land forms, vegetation and wildlife.

Transferring of adjusted mean values implies an ability to find some variable which explains how WTP is expected to change across sites. This is the approach taken, for example, in the Foundation for Water Research (FWR) handbook on benefits transfer for UK inland waters (FWR, 1996). Yet CVM as a method is likely to do poorly in this respect, since it does not break down environmental goods into their constituent characteristics, thus it is harder to identify which variable should be used to transform WTP (for example, we do not know the implied marginal value of the percentage of heather moorland, or number of plant species, nor which is more important). Finally, since bid curves show that many site- and respondent- specific variations are important, transferring this entire function might offer better opportunities to estimate WTP at other ESAs. Yet, as Bateman *et al.* (1994) and Turner *et al.* (1992) found, little variation in WTP estimates across UK CVM studies is statistically explainable, whilst the benefit transfer method rests on the un-validated assumption that bid curves are parametrically stable across sites. Thus we may conclude that prospects for benefits transfer from CVM in this instance are poor, except in the 'order-of-magnitude' direction.

In contrast, the choice experiment (CE) approach seems to offer greater potential for benefits transfer. This is due to the decomposition of total value for any environmental resource into characteristic values. Whilst a methodological objection can be maintained against such a de-constructionist approach, if environmental goods can be at least partly described in terms of their measurable characteristics, then CE estimates of marginal WTP for changes in these characteristics could be used to estimate benefits at other sites. This procedure, however, would not allow for differences in the composition of the beneficiaries themselves (for example, in terms of income). However, we note that whilst the characteristic valuations obtained in this study are a step in this direction, their usefulness is limited in that they value 'less- to- more' changes in each characteristic (e.g. less heather moorland without the ESA scheme to more with it), rather than measurable changes in each characteristic. However, this shortcoming could be overcome in future applications of the CE technique, by specifying different physical quantities or cardinally

[1] This study, for example, cost over £80,000.

measurably-different variations in characteristic levels (such as percentage woodland cover) in the choice sets.

7. Discussion and Conclusions

This paper has used the Contingent Valuation Method and Choice Experiments to estimate the economic value of the conservation and landscape benefits of ESAs in Scotland. The purpose of this exercise was to obtain policy-useful information (whether benefits outweigh costs), but also to undertake a methodological comparison. We found that both contingent valuation (CVM) and choice experiments (CE) could estimate the value of the ESA package as a whole, but that the latter method is more suited to measuring the (marginal) value of the individual landscape and wildlife characteristics that make up ESAs. An unresolved issue, though, is how to choose such characteristics from the very large set available, and how this choice impacts on the 'total package' welfare measures obtained from choice experiments. Such choices are not necessary in CVM, which may imply that if the main objective of analysis is estimating the value of some overall policy package (or environmental resource), then CVM is better than CE. However, some caveats should be attached to these CE results. First, the revealed ranking of attributes is partly a function of the relative difference between 'good' and 'bad' levels as suggested in the information packs (in other words, CE, like CVM, is sensitive to the information set both held by and presented to respondents). Second, the welfare estimates obtained from CE are, as we have seen, sensitive to functional form choice. Third, it may be an error of aggregation to assume that the total value of the ESA scheme can be inferred from adding up characteristic values, since there may be other attributes which are important to preferences but which are not included in our design (although the use of focus groups should help minimise this specific problem). Our very simple experimental design also ignores interactions between attributes.

It is of interest to compare our CVM results for the Breadalbane ESA with those obtained for other ESAs in the UK. Table 5 summarises results from these other studies, and includes equivalent estimates from this study. ESA benefits in the UK have previously been obtained by Willis *et al.* (1993) for the South Downs and Somerset Levels in England; Gourlay (1996), for Loch Lomond and Stewartry, Scotland; and Bullock and

Table 5 Comparing CVM Estimates for ESAs in the UK (WTP, £/hsld/yr)

ESA	*Residents*	*Visitors*	*General Public*
This Study			
Breadalbane	31.43[1]	73.00[2]	22.02[1]; 42.0-57.0[2]
Willis et al. 1993			
South Downs	27.52[1]	19.47[1]	1.98[1]
Somerset Levels	17.53[1]	11.84[1]	2.45[1]
Gourlay, 1996			
Loch Lomond	20.60[1]	1.98 per visit[1]	n/e
Stewartry	13.00[1]	2.53 per visit[1]	n/e
Bullock and Kay, 1996			
Southern Uplands	n/e	69.00[2]	83.00[2]

Notes: 1 = open-ended CVM; 2 = dichotomous choice CVM; n/e = not estimated.

Kay (1996) for the Southern Uplands, Scotland. Comparison is made difficult since some authors use open-ended estimates only and others dichotomous choice only; it is well-known that the former are often smaller than the latter, in this study by 1.9-5.7 times and in Willis *et al.* by 3.8 times. Furthermore, Gourlay only quotes per visit figures for visitors and does not estimate general public WTP, whilst Bullock and Kay do not measure residents' WTP. Finally, the ESAs themselves are very different in nature; for example, Breadalbane has little in common with the Somerset levels.

Three main points emerge from Table 5. First, the WTP figures for residents are fairly similar across all three studies. Second, whilst the visitor values from Breadalbane are comparable with those for the Southern Uplands, they are much higher than those for the two English ESAs. However, this is probably because whilst both the Scottish studies giving annual WTP figures are dichotomous choice estimates, the English estimates are open-ended. Finally, whilst both the English ESA and our ESA estimates for general public values noted in the Table are from an open-ended format, the former are much smaller than the latter. This difference probably arises due to the method of presenting the good. The South Downs and Somerset Levels studies both asked respondents to bid for *all* ESAs in England; and then used a points scoring system to apportion this across individual ESAs. This resulted in rather less information being available on each ESA than was the case in this study, where respondents were asked to bid for Breadalbane ESA directly, whilst being reminded about the need to pay for other ESAs and for other environmental policies, and that the questionnaire was only concerned with WTP for Breadalbane.

This difference in general public values then, may be due to both a difference in the ESAs themselves, and a difference in the strategy researchers adopt to cope with part-whole bias. The strategy adopted by Willis *et al.* was to nest the good they were interested in a more inclusive good (all ESAs), seek WTP bids on the inclusive good, and then use other information to estimate WTP for the individual ESA of interest. Whilst this approach has the merit of reminding respondents of the opportunity cost of 'paying' for one ESA (that less points from the fixed total can be allocated to others), it is known from the work of, for example, Carson and Mitchell (1995) that this approach will lower WTP for the good of interest. The nesting approach suffers from one other problem, concerned with information provision. It might be argued that the same level of information should be provided on all components that make up the inclusive good on which people bid; yet if there are many such goods (ten, for Willis *et al.*), then this greatly restricts the amount of information that can be given on each. It might also be argued that WTP for all ESAs should first be nested in WTP for all agri-environmental policies, which should in turn be nested in WTP for all environmental policy: the implications for providing sufficient information to produce 'reliable' bids at the most inclusive level rapidly become apparent. Finally, if more information is provided on goods lower down, then respondents might want to revise their earlier inclusive bid and/or their points allocation.[1]

In conclusion, CVM and CE offer rather different merits to the policy researcher, CVM seems best suited to valuing the overall policy package, and CE to valuing the individual

[1] The obvious counter-criticism is that the approach adopted in our study, despite all reminders to the contrary, will suffer from part-whole bias.

characteristics that make up the policy. Should researchers make progress in solving the problem of dis-aggregating policies/resources into appropriate characteristic sets and levels, then the other advantages of CE (such as with regard to benefits transfer) may lead it to becoming preferred in valuing total packages. Finally, we note that the consistently large numbers generated by CVM for environmental benefits in the UK and elsewhere has led to it becoming a popular valuation technique with policy makers enthusiastic about environmentally-friendly policy (how many environmentally-beneficial policies/ projects subject to a CBA which includes a CVM analysis fail the CBA test?). Yet this very feature may undermine its long-term usefulness and acceptability, if CVM is not seen as being sufficiently discriminatory.

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Input Controls, Input Substitution and Profit Maximisation in the English Channel Beam Trawl Fishery

Sean Pascoe and Catherine Robinson

Studies of input substitution in fisheries have tended to focus on substitution between physical inputs and/or time fished. However, input controls may create incentives for substitution of other inputs into the production process. For example, fishers faced with constraints on access to particular areas of the fishery may substitute physical inputs for fishing location. This was the case with the UK beam trawl fishery in the English Channel. Constraints were imposed on access to particular areas of the fishery through restrictions on engine power. This created incentives for a number of fishers to reduce their engine power to meet the conditions of the input control. The relative contribution of the boat inputs and location in the production process and the potential for substitution were examined using a translog production function. The results suggested that the apparent input substitution was consistent with profit maximising behaviour.

1. Introduction

The need to regulate fisheries for both biological and economic reasons has been well established in the literature (see, for example, Scott, 1955; Anderson, 1986; Cunningham *et al.*, 1985; Hannesson, 1993). Most of the problems associated with overexploitation have been attributed to the open access nature of the resource. In the absence of regulations limiting entry, the existence of above normal profits encourages new entrants to the fishery. Equilibrium is achieved at the point where marginal cost and average cost equals average revenue. This has proven to be non-optimal in most cases, with over-exploitation of the fish stocks, over-capitalisation of the fishing fleet, and the dissipation of economic rents.

Fisheries management aims to reduce or control the level of fishing effort to achieve a better allocation of resources. The term 'effort' is used as a proxy for the combination of inputs, including everything from the number of boats fishing, the length of time fished,

■ Sean Pascoe and Catherine Robinson are respectively Senior Research Fellow and Research Associate in the Centre for the Economics and Management of Aquatic Resources (cemare), Department of Economics, University of Portsmouth, Locksway Road, Portsmouth PO4 8JF. The study was funded by the Ministry of Agriculture, Fisheries and Food as part of the Channel fisheries bioeconomic modelling project (Project number MF 03 08). The authors would also like to thank Guy Judge, Richard Harris, Bruce Traill and the two anonymous referees for their useful comments on the paper.