Stated Preferences for Improved Cod Stock in Sweden

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Abstract. Historical records show that the stock density of coastal cod (*Gadus morhua*) in the waters off the Swedish west coast is extremely low. In 2001 the stock size was two percent of the size in the 1970s. Scarce fish resources imply conflicting interests among various user groups, and sustainable management requires necessary trade-offs to be made. An economic evaluation of the social benefits of the use of these fish resources is valuable for coastal managers in their decision making process. In this paper, we apply the contingent valuation method to estimate the willingness to pay for an increased cod stock in the coastal waters of the Swedish west coast. Also, we test the effects of using different elicitation formats and payment vehicles in the valuation process. We find that the dichotomous choice format yields higher values than the open-ended format and that the formats are statistically different. Surprisingly, we find no statistical difference between payment vehicles (tax versus license fee). The median values range from SEK 150 to 250 depending on estimation method, and mean values range from SEK 230 to 900. A relatively modest aggregation procedure gives an aggregate WTP equal to SEK 704 million. This is a reflection of the public concern for the coastal cod population in the Swedish waters.

Key words: cod, contingent valuation, recreational fishing, stated preferences, willingness to pay

JEL Classifications: D61, Q26, Q51

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1 Introduction

Cod (*Gadus morhua*) has for a long time been among the most important fish species for people in the coastal zones all around the North Atlantic Ocean. However, during the last few decades there have been reports of a growing scarcity. Degerman (1985) reports that the cod fisheries along the west coast of Sweden changed dramatically in the beginning of the 1980s. In the beginning of the 1990s the Canadian stocks of cod vanished (Hutchings 1996), and in Sweden the coastal cod stock declined drastically. The stock density of inshore demersal fish along the Swedish west coast is nowadays extremely low for most species. In 2001 the size of the stock was equivalent to a catch of approximately two kg cod per trawl hour, which corresponds to two percent of the size in the 1970s (Svedäng et al. 2001, Svedäng and Bardon 2003). In practice this means there is no recreational angling for cod.

Biological experiments by Pihl and Ulmestrand (1993) show that the young cod migrates to spawning areas in deeper waters mainly west to the North Sea. Biological research also indicates that the inshore area has become increasingly dependant on the transport of recruits from offshore spawning areas, and that the current situation is an effect of high fishing pressures in the North Sea and not of negative or hazardous environmental factors in the coastal zones (Svedäng 2003). This implies that better management of commercial cod fisheries in the North Sea is necessary to improve the cod stock along the Swedish west coast.

In Sweden, the old so-called "Right of Public Access" (Allemansrätten) gives everyone a freedom to enjoy the countryside as long as the activity is not disturbing or destructive. In recent years a political discussion has been brought up about introducing a license fee for recreational fishing to support local fish preservation (Bråkenhielm 2001). There has also been discussions about introducing a unilateral cod moratorium, and estimations of the economic consequences have been made (Fiskeriverket 2002). These discussions led to a public awareness of the declining fish populations, and the vanishing coastal cod stock has become a concern of the media. Fishery managers are becoming increasingly pressured to determine a sustainable allocation of fish resources. In this decision making process, reliable measures of the social benefits of recreational and commercial fishing are important along with potential non-use values. While statistics on commercial fisheries and their production values are readily available, less is known about recreational fishing and its economic values. Toivonen et al. (2000) conduct a contingent valuation (CV) survey in all five Nordic countries to estimate a value for preserving the existence of the total Nordic fish stock. For the area that we study, Paulrud (2001) is the only earlier study that has estimated the value of marine recreational fisheries. He uses the travel cost method as do most studies of recreational fishing (see e.g. Ledoux and Turner 2002 and Freeman 1995 for overviews).

In this paper, we use the CV method to estimate the willingness to pay (WTP) for an improved cod stock utilized for recreational fishing in the waters along the Swedish west coast. Further, the effects of using different elicitation formats and payment methods in the valuation process are explored. We also look at whether there are different WTP values for people who use the coastal zone for recreational angling and for people who do not.

2 Design and data collection

The survey concerns the willingness to pay for an increase in the coastal cod stock from the 2001 level to the 1974 level, in Skagerrak and Kattegatt off the west coast of Sweden. The change is described as an increase in the catch per trawling hour from two kilograms to 100 kilograms of cod.

Table 1

Sample design

	Open Ended	Dichotomous Choice
Tax	Sample 1	Sample 3
License fee	Sample 2	---

The design is a three-way split sample survey where two samples have the open-ended (OE) elicitation format and one sample has the dichotomous choice (DC) format. Empirical evidence shows disparities in WTP results between OE and DC formats (e.g. Cameron et al. 2002). By using split samples, we are able to test for disparity in the results between the elicitation formats. We further test for disparity between payment methods by splitting the OE data into one tax sample (called OE_tax) and one licence fee sample (OE_fee). The sample design is shown in *Table 1*. The respondents were randomly sampled from the counties of Västra Götaland and Halland in the southwest of Sweden. 600 respondents were sampled independently for each of the three formats and the questionnaires were mailed out in May 2002. The bids in the DC sample were set to cover the range of WTP found in a open-ended pretest. There were five bids, with the middle bid close to the mean WTP from the pretest plus two in each tail: SEK 50, 200, 350, 700, 1100. The questionnaires consisted of one section inquiring about habits of using the coastal area for recreation, one section eliciting the willingness to pay, and one section inquiring about socio-economic characteristics. The valuation question was preceded by a participation question in the DC sample, but not in the OE samples. The participation question asked whether or not the respondent had any WTP whatsoever for the good to be valued. In all other aspects the questionnaires were identical. A translation of the scenario and the valuation question is given in Appendix A.

3 The Model

When modeling the responses we follow Cameron and James (1987) and Cameron (1988) and model the WTP distribution directly. We assume that the cod stock is a normal good for most people. The negative social welfare effects of an increased stock are negligible and it is therefore reasonable to expect a non-negative willingness to pay for the good in question. Consequently, we assume an exponential WTP function:

$$\text{WTP} = \exp(\beta X + \varepsilon). \qquad (1)$$

Each individual is offered a randomly chosen bid t and if the bid is accepted we conclude that the true WTP is greater than the bid. We denote a "yes" response by y=1 and a "no" response by y=0. Assuming that the error terms are i.i.d. and normally distributed $\varepsilon \sim N(0, \sigma)$, we standardize the distribution to the standard normal $\varepsilon/\sigma \sim N(0, 1)$ and derive the probability of a yes response as:

$$\Pr(y=1) = \Pr(\text{WTP} > t) = \Pr(\varepsilon/\sigma > 1/\sigma \ln(t) - \beta/\sigma\, X) = 1 - \Phi[1/\sigma \ln(t) - \beta/\sigma\, X], \qquad (2)$$

where Φ denotes the standard normal cumulative density. This enables us to identify the standard deviation σ and thereby test for equal parameters across samples (Boyle et al. 1996, Cameron et al. 2002). The mean and median WTP are given by:

$$E[\exp(\beta Z+\varepsilon)] = \exp(\beta Z)\, E[\exp(\varepsilon)] = \exp(\beta Z + \tfrac{1}{2}\sigma^2) \qquad (3)$$

$$\text{Median WTP} = \exp(\beta X). \qquad (4)$$

4 Empirical Results

Table 2 shows that the response rates for the 600 respondents from each sample are approximately the same. A reason for the difference in participation rates between DC and OE samples might be that the OE questionnaire did not include a participation question.

Table 2

Response and Participation Rates

	DC	OE tax	OE fee
Response rate	61%	55%	61%
Participation rate (WTP>0)	58%	69%	65%

Our analysis of responses is conducted on the restricted group of respondents with a positive WTP, in the three samples respectively.[1] There are no statistically significant differences between sample means other than for the variables "Big City" and "Cottage". Since the three independent samples are so similar and the survey questionnaires are identical, we can reasonably assume that any differences in the valuation results can be attributed to the different elicitation formats being used. Descriptive statistics of the socio-economic characteristics are reported in *Table 3*. The share of respondents stating that they go angling when they visit the coast is around 30 percent. This is a slightly higher value than the population mean (Fiskeriverket 2000). Similarly, the sample is somewhat overrepresented by members of environmental NGOs (around 12 percent). This means that people interested in environmental issues in general, and fishing in particular, were somewhat more inclined to respond. Still, the samples are respresentative enough to allow for aggregation of our results.

Table 3
Descriptive statistics

		OE TAX (n=225)		OE Fee (n=217)		DC (n=210)	
Variable	Description	Mean	Std dev	Mean	Std dev	Mean	Std dev
Gender	1 = male 0 = female	0.50	0.5011	0.49	0.5010	0.50	0.5012
Age	18 - 65 years	41.7	13.10	41.9	12.99	42.0	12.68
Income	Equivalence scaled*)	12040.0	5548.4	12160.4	6684.6	11840.0	5803.6
Big City	1 = resident in Göteborg	0.36	0.48	0.26	0.44	0.29	0.46
Cottage	1 = access 0 = no access	0.23	0.42	0.15	0.36	0.17	0.37
Car	1 = ownership 0 = otherwise	0.88	0.32	0.91	0.29	0.90	0.31
Education	1 = University 0= No Univ	0.40	0.49	0.42	0.50	0.38	0.49
Coast	1 = house <1km distance	0.23	0.42	0.24	0.43	0.24	0.43
Boat	1 = No access 0 = Otherwise	0.61	0.49	0.57	0.50	0.56	0.50
Angler	1 = Yes 0 = No	0.32	0.47	0.34	0.48	0.36	0.48
House	1 = ownership 0 = otherwise	0.58	0.49	0.62	0.49	0.63	0.48
NGO	1 = supporter 0 = otherwise	0.12	0.33	0.12	0.32	0.11	0.31
WTP		352	371	371	397		

*) Adult#1 is given weight 0.95, adult#2 weight 0.95 and each child weight 0.61.

4.1 Nonparametric estimation of the WTP

We start by estimating the WTP with a non-parametric method which does not require any assumption about the distribution of the WTP. First we test whether the samples have been drawn from the same population distribution by using the Kolmogorov-Smirnov (KS) two-sample test. For this, we construct the cumulative survival function, $S(t_j)$, for each sample using the DC bid intervals as shown in *Table 4*.

Table 4

Survival functions for bid intervals.

t_j (SEK)	$S_{DC}(t_j)$	$S_{TAX}(t_j)$	$S_{FEE}(t_j)$
50	0,72	0,99	0,99
200	0,49	0,57	0,69
350	0,33	0,32	0,34
700	0,17	0,14	0,13
1100	0,07	0,08	0,04
	n=210	n=225	n=217

A large enough deviation at any point between two cumulative sample distributions is evidence for rejecting the hypothesis that they are drawn from the same population or from populations with the same distribution. According to this test, there is a significant difference between the DC and OE samples, but not between the two OE samples.[2] Following Kriström (1993), we can also apply a Chi-square test. Just like the KS test, the Chi-square test leads to rejection of the hypothesis that the OE and DC responses are generated from the same distribution.[3] However, we find no significant difference between the OE samples. This means that the payment vehicle does not seem to have any impact on the distribution of the WTP among respondents.

Next, we estimate the mean and median WTP. An estimate of the median WTP corresponds to that bid which half of the respondents would accept.[4] The mean WTP is the area bounded by the survival function, and for the DC case we use the so-called Spearman-Karber (SK) estimator:

$$E(WTP) = \sum_{j=0}[S_{DC}(t_j) - S_{DC}(t_{j+1})](t_j + t_{j+1})/2 \tag{6}$$

where $S_{DC}(t_j)$ is the share of respondents with a WTP equal to or greater than t_j. We assume that all respondents accept a zero bid and we assume that the maximum WTP in the DC sample is the same as in the OE samples, or SEK 3000. Thus, the bounds of the distribution are $S_{DC}(0)=1$ and $S_{DC}(3000)=0$. *Table 5* reports the mean and median measures of WTP together with confidence intervals for the mean estimates.[5] The mean WTP is around SEK 400 for the DC sample, which is higher than for the OE samples. The mean WTP is around SEK 370 for the OE_fee sample, and around SEK 350 for the OE_tax sample. The median WTP is SEK 200 in both OE samples, and approximately the same in the DC sample.

Table 5

Nonparametric Measures of WTP
(95% confidence interval in parentheses)

Sample	Mean	Median
DC	397 (326 - 468)	193
OE FEE	371 (318 – 424)	200
OE TAX	352 (303 – 400)	200

4.2 Parametric estimation of the WTP

In the parametric estimation of the WTP we assume a normal distribution as outlined in Section 3 above, and all models are estimated as described in equation (2). Since the scale parameter has been estimated directly, the coefficients can be compared across the models. In likelihood ratio tests we can reject the hypothesis that the underlying preference structure (Eq. 1) is the same for OE_tax and DC elicitation formats (Halvorsen and Saelendsminde 1998, Cameron et al. 2002). However, we find no statistical difference between the two OE samples. Thus, our parametric tests confirm our nonparametric tests. This means that the payment vehicle has no significant effect on the WTP in this study. Usually, the payment vehicle is expected to have an effect on the WTP since different vehicles provide different opportunities for free riders. People also have different views on the acceptability and credibility of various payment methods (Bateman et al. 1995, Garrod and Willis 1999, Champ et al. 2002).

Table 6 reports the coefficients from the estimations together with the corresponding p-values. Since we find no statistical difference between the OE samples, we estimate a model with the two OE samples pooled. This pooled model includes a dummy called TAX which equals one for the OE tax sample. The TAX coefficient is insignificant which again confirms the payment vehicle indifference. Although the samples are somewhat overrepresented by members of environmental NGOs and people who go angling at the coast, neither of these categories has any discernible impact on the WTP.

Table 6

ML estimation (p-values in parentheses)

	Model1 DC	Model 2 OE TAX	Model 3 OE FEE	Model 4 OE POOLED
Intercept	6.318	3.285	4.643	4.150
	(0.0000)	(0.0001)	(0.0000)	(0.0000)
Gender	0.0899	0.00992	-0.0754	-0.00909
	(0.7320)	(0.9415)	(0.5554)	(0.9218)
Age	0.00483	0.0631	0.0545	0.0558
	(0.9503)	(0.1010	(0.0695)	(0.0180)
Age^2	0.000337	-.000691	-0.000806	-0.000709
	(0.7066)	(0.1299)	(0.0262)	(0.0126)
Income	-0.0000116	0.0000398	0.00000774	0.0000204
	(0.6166)	(0.0020)	(0.3726)	(0.0007)
Big City	-0.386	0.198	0.320	0.245
	(0.2288)	(0.1689)	(0.0281)	(0.0134)
Cottage	-0.217	0.187	-0.336	-0.0443
	(0.5505)	(0.2222)	(0.0782)	(0.6859)
Car	-0.0785	0.0164	0.281	0.101
	(0.8681)	(0.9346)	(0.1379)	(0.4607)
Education	0.148	0.0252	-0.106	-0.0437
	(0.6112)	(0.8628)	(0.3935)	(0.6513)
Coast	0.0849	-0.0303	-0.0268	-0.0365
	(0.7763)	(0.8608)	(0.8592)	(0.7435)
No boat	-0.284	0.00255	-0.136	-0.0378
	(0.3273)	(0.9865)	(0.3508)	(0.7097)
Angler	-0.000213	0.00307	-0.187	0.0257
	(0.8746)	(0.0590)	(0.2122)	(0.8067)
House	-0.121	0.140	-0.0389	0.0522
	(0.6803)	(0.3383)	(0.7647)	(0.5875)
NGO	-0.128	0.0835	0.125	0.131
	(0.7362)	(0.6864)	(0.6026)	(0.3767)
TAX				-0.133
				(0.1354)
σ_{DC}	1.039			
	(0.0000)			
σ_{TAX}		0.893		
		(0.0000)		
σ_{FEE}			0.812	
			(0.0000)	
σ_{OE}				0.884
				(0.0000)
	LL=82.63	LL=293.74	LL=262.72	LL=572.84
	N=210	N=225	N=217	N=442

Table 7
Parametric Measures of WTP
(95% confidence interval in parentheses)

Sample	Median	Mean (UL = ∞)	Mean (UL = 3000)
DC	613	1052	913
	(292-935)	(300-1894)	
OE TAX	234	349	346
	(109-359)	(158-539)	
OE FEE	252	350	349
	(154-349)	(212-487)	
OE POOL	155	229	229
	(96-214)	(142-317)	

Remember that the mean WTP is the area bounded by the survival function, i.e. the integral of the function. In the parametric estimation of the WTP, the upper limit of this integral is usually infinity.[6] However, to be able to compare these measures to the nonparametric measures we impose the same upper limit (truncation point), or SEK 3000. *Table 7* reports the results from both the truncated and untruncated estimations as well as the median estimates.[7] Note that these measures do not take into account the zero bids. The estimates are naturally lower in the truncated estimation, but not by much. We see that the WTP measures for the DC version are much higher in the parametric model compared to the non-parametric model. The results for the OE do not vary as much, although the median values are slightly higher in the parametric estimations. The WTP measures for the pooled OE model are the lowest. While the DC elicitation format is the only format which is incentive compatible (Arrow et al. 1993, Carson Groves and Machina 1999), it results in very high WTP values in the parametric estimation. The OE format on the other hand is prone to induce strategic behavior, but does not seem to be as sensitive to the method of estimation.

5 Conclusions

In this paper we estimate the willingness to pay for an increased cod population along the Swedish west coast. Coastal cod is exploited through both commercial and recreational fishing, but over the last few years the poor status of the cod in Swedish waters has become a public concern in Sweden. This concern is reflected in our study by a substantial willingness to pay for cod stock improvement and by the fact that there is no significant difference in willingness to pay between recreational anglers and those who never fish. The overall poor significance in the models may result from relatively small sample sizes.

In line with earlier studies, we find a significant difference in the willingness to pay between the open-ended and dichotomous choice elicitation formats. However, we also test two different payment vehicles (tax versus license fee) and find no statistical difference between them. This is surprising since different vehicles provide different opportunities for free riders. People also have different views on the acceptability and credibility of taxes and fees (Bateman et al. 1995, Champ et al. 2002).

Concerning the WTP, we find that the dichotomous choice format yields higher values than the open-ended format. The median values range from SEK 150 to 250 depending on the estimation method, and mean values range from SEK 230 to 900 where the higher value comes from parametric estimation of dichotomous choice data. In the discussion in Sweden about introducing a license for recreational fishing, the suggested fee is SEK 200 (Bråkenhielm 2001). Our results show that among those who are willing to participate, there is a high enough mean WTP to cover this fee.

There are also estimations of the economic consequences of a Swedish moratorium on cod fishing. The social costs are estimated at around SEK 530 million (Fiskeriverket 2002). Although this study does not measure the WTP specifically for a cod moratorium, aggregating our results can still provide an indication of what the social benefits may be. A relatively modest aggregation procedure would assume that the participation rate of the Swedish population is the same as in our sample. The Swedish population in the 18-65 year range was around 5,5 million at the end of 2001. Using our median measure SEK 200 would then give an aggregate WTP equal to SEK 704 million. Thus, there is reason to believe that there is a high WTP also for a moratorium on cod. However, a unilateral moratorium is not likely to be sufficient to accomplish an increased cod population off the west coast of Sweden.

APPENDIX A.

English translation of the scenario and valuation question:

Cod availability

The availability of coastal cod was in the 1970s at a level equal to a catch of approx. 100 kg per hour of trawling with a research vessel. In 1992 the level was down to approx. 25 kg cod per trawling hour. In the 1990s the availability decreased so heavily that cod above the minimum length of 30 cm (2-3 hg) was almost entirely missing in the coastal area. Today's level is approx. 2 kg cod per trawling hour which broadly means that recreational fishing using a hook and a rod or a net does not yield any catch of cod.

The availability of cod along the Swedish west coast expressed as catch of cod:	
• Approx. 2 kg cod per trawling hour using research vessel	(today's level)
• Approx. 25 kg cod per trawling hour using research vessel	(1992 level)
• Approx. 100 kg cod per trawling hour using research vessel	(1974 level)

Cost

Imagine that the project is financed by municipalities within Dalsland, Bohuslän, Västergötland and Halland through a temporary fee for one year. The fee is paid by all 18-65 year old residents in the counties listed above. The project is implemented only if there is enough support for it.

Question

Given that the project is implemented and that it is financed as described above, how much would you be willing to pay <u>in total per year</u>, to be able to fish in the sea with a cod availability of the 1974 level, i.e. approx. 100 kg cod per trawling hour using a research vessel?

I would be willing to pay a maximum of SEK_________ in total per year.

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Notes

1 Respondents with WTP higher than 25 percent of their income were considered outliers and therefore removed.

2 The test statistic is the maximum deviation between two cumulative sample distribution functions, $D=\max|S_A(X)–S_B(X)|$. Between DC and OE_tax samples, $D=|0,72-0,99|=0,27$ at BID=50. Between OE_tax and OE_fee samples, D=0,12 at BID=200. The critical values are 0,1420 for $\alpha=0,025$ and 0,1564 for $\alpha=0,01$ (Siegel and Castellan 1988).

3 The test statistic is $\Sigma(O_i-E_i)^2 / E_i = 26,008$ where O_i is the observed number of people having a WTP higher than the bid t_i and E_i is the expected number of people. The critical value at $\alpha=0,01$ with 4 df is 13,277.

4 In the OE case, we just order the WTP data and take the middle observation as the median WTP. For the mean, we add the WTP values and divide the sum by sample size. Using the fractions in *Table 4* yields mean $WTP_{OE\_TAX} = 358$ and $WTP_{OE\_FEE} = 369$.

5 In the DC case, variance is given by the SK estimator as $\sum_{j=1}S_j(1- S_j)(t_j - t_{j-1})^2/n_j$ where n_j is the number of respondents (Miller 1973).

6 This comes from the relation between the mean of a random variable and the integral of its cdf. When WTP values are restricted to be non-negative, the expected WTP value is equal to the integral of its survival function 1-F(x), bounded by zero and infinity (Hanemann and Kanninen 1998).

7 The median is unaffected by this truncation. Hence we only report the mean estimates. The effect on the mean is sometimes referred to as the “fat tail” problem (Boyle, Welsh and Bishop 1988).

Heterogeneous preferences for marine amenities:
A choice experiment applied to water quality

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Abstract. The marine environment provides many goods and services dependent upon the quality of coastal waters. In this paper, we represent water quality by three different attributes, fish stock level, bathing water quality, and biodiversity level, and carry out a choice experiment among residents on the Swedish west coast to estimate the economic benefits of improved coastal water quality. We analyze data using the mixed multinomial logit model and explore various distributional assumptions and derive individual-specific parameters. Our results confirm heterogeneous preferences for these attributes and show that respondents have high levels of environmental concern and that substantial values are at stake. The most urgent action according to our findings is firstly to prevent further depletion of marine biodiversity and secondly to improve Swedish cod stocks.

Key words: choice experiments, mixed logit, stated preference, water quality,

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1. Introduction

The marine environment provides many goods and services such as food, recreation activities, coastal protection and breaking down degradable waste. These benefits have been hard to quantify in monetary terms, which has implied a risk of their being neglected in policy making. However, it is now well established that values from marine coastal waters can be substantial and avoiding potential losses is an important task for policy making. Marine water quality can be characterized in a number of ways. Any attempt to estimate values of improvement or prevention of deterioration must make the trade-off between the interest in various attributes on the one hand and a reasonable level of complexity on the other. Further, priority should be given to those attributes that are not only measurable but also demand-relevant and policy-relevant (Blamey et al., 2002). The failure to secure sustainable commercial fisheries and the implications of this to recreational fishing has generated great interest within the European Union. Similarly, attention has recently been focused on the costs and benefits of improving coastal water quality. Standards have been constructed for port and beach facilities, e.g. the 'Blue Flag'[1], to improve service to marine recreationalists and to control for pollution into the waters. The issue of securing marine biodiversity and its importance for sustainability has been at the top of the world agenda since the UN meeting in Rio 1992. Often, several aspects are relevant to a single decision. The current great interest in marine reserves concerns not only improvements of fish stocks and catches, but also benefits in terms of biodiversity (Roberts et al., 2001).

In this paper we estimate the benefits of improving coastal water quality with respect to fishing possibilities, bathing water quality and biodiversity levels for a random sample of individuals in the southwestern parts of Sweden. As we deal with both use- and non-use values, we apply a choice experiment where individuals are asked to choose between different alternatives for marine water quality improvements and a status quo alternative. This information together with socio-economic data is analyzed using the mixed multinomial logit model, which is the most promising discrete choice model currently available (Hensher and Greene, 2003). We test various distributional assumptions for the random parameters such as the normal and the triangular distribution with and without constraints. We also explore the technique of using population parameter estimates and conditioning them with individual choices to calculate individual level parameters, which e.g. is informative when a 'reversed' sign occurs for an attribute.

Further, we estimate individuals' marginal willingness to pay (WTP) for the various attributes and provide confidence intervals of the mean WTP estimates. Our results indicate that the respondents prioritize improvements of the Swedish cod stock levels and the prevention of further possible depletion of marine biodiversity. Improved water quality and improved marine biodiversity are also important, but less important than improving cod stocks and preventing biodiversity losses.

2. Valuing improvements in marine water quality

The coastal zone is an important habitat for many fish species in Swedish waters and the status of coastal fish stocks reflects the effects from commercial and recreational fishing as well as the coastal water quality in terms of feeding and breeding conditions. Individuals use the coastal zone for recreational purposes like fishing and bathing. The demand for bathing is dependent on characteristics such as water visibility, the level of organic material in the water, and the frequency of failures meeting the standards of bacteriological contamination. Recreational fishing and bathing are both examples of activities which we expect individuals to value in terms of use values albeit an element of non-use values may well be present. Biodiversity on the other hand, is an attribute which could be expected to be valued also in terms of non-use values. However, there are many aspects of biological diversity such as number of species, number of individuals within a species, and the diversity among the individuals within a certain species (Nunes and van den Bergh, 2001). Several studies estimate recreational benefits from improvements of marine water quality, applying the contingent valuation method, random utility models or the travel cost model (Freeman, 1995). Carson and Mitchell (1993) use the contingent valuation method to study the value of clean water on a national level and categorize the water quality as usable for boating, fishing, and swimming.

In addition to recreational values, potential categories relevant tovaluing marine water quality improvements are commercial fishing, health, non-use values, property values, and regional economic impacts (Morgan and Owens, 2001). Georgiou et al (2000) value bathing water improvements from a health risk perspective. Hanley, Bell and Alvarez-Farizo (2003) combine stated and revealed preference data to estimate economic benefits from improvements of coastal water quality in southwest Scotland.

In order to limit the complexity for respondents, we focus on recreational values and the value of various biodiversity levels in this study. Indirect methods like travel cost models cannot be used for estimating non-use values, which leaves us with direct methods such ascontingent valuation and choice experiments. In contingent valuation studies respondents are asked about a single event in detail, while choice experiments offer the possibility of asking about a sample of events. The latter has a potential benefit as it leads respondents to explicitly make trade-offs between the various attributes of the situation (Boxall et al., 1996). Choice experiments have become popular in environmental economics (Adamowicz et al., 1994; Layton and Brown, 2000). The freshwater recreation study previously mentioned (Adamowicz et al., 1994), the bathing water quality study (Hanley et al., 2003), and the study of polluted beaches, polluted rivers and low flow rivers by Garrod and Willis (1999), are so far, the existing choice experiment studies of water quality. Garrod and Willis (1999) is the exception as the study estimates both use- and non-use values, values often relevant to water quality. The current great interest in marine protected areas (e.g. Charles and Sumaila, 2002) is one example; biologists stress both stock enhancement effects and various non-use values like biodiversity (Novaczek, 1995) while economists, so far, have mainly focused on use-values such as stock enhancement and increased harvests (Sanchirico and Wilen, 2001).

The choice experiment traces its roots back to the psychologist's ambition of specifying and estimating a discrete choice model, which can predict chosen alternatives by different individuals (Thurstone 1927, Luce 1959). Later these ideas wererefined by economists and linked to the characteristics theory of value (Lancaster, 1966) and the random utility theory (Manski, 1977).[2] The basic idea is that individuals choose between different bundles of goods which are characterized by several attributes and the levels that these attributes take, where one attribute is price. Individuals are assumed to know their preferences[3] and to make choices maxmizingtheir utility, while these preferences are not fully known to the researcher. Based on the random utility framework and welfare theory in economics, it is then possible to calculate welfare estimates for various changes in levels of the different attributes. These benefits, specifically improvements, can then be related to costs in a standard cost-benefit analysis framework to provide policy guidance for decision makers. The easiest and most widely used discrete choice model is the multinomial logit (MNL) model. However, the MNL model imposes the property of Independence of Irrelevant alternatives (IIA), which can be a limitation. The IIA property implies a certain pattern of substitution across alternatives. An improvement in one alternative draws proportionately from the other alternatives. Imposing this proportionate substitution pattern can lead to unrealistic forecasts in many settings. (McFadden 1974; Train 2003). Furthermore, while the MNL model can represent systematic taste variation, the model is unable to represent differences in taste that cannot be linked to observed characteristics of the decision maker, i.e. the respondent. Taste can vary among people possessingthe same demographic characteristics for reasons the researcher must treat as random, just because people are different. To incorporate this random taste variation appropriately, a more general model has to be used instead. Hensher and Greene (2003) hold that the most promising discrete choice model currently available is the mixed multinomial logit model (MXL). The MXL does not exhibit the IIA property and it is well suited to explicitly account for unobserved heterogeneity in taste, since it allows parameters to have a distribution.

3. The Marine Water Quality Choice Experiment

3.1 SURVEY DESIGN AND DATA COLLECTION

The choice experiment concerns the coastal waters of the Swedish west coast, Skagerrak and Kattegatt. The respondents were sampled from the permanent population in the counties of the southwest part of Sweden, Västra Götalands- and Hallands län. A survey company constructed a random sample of 800 individuals aged 18-65 years from the Swedish Register of Inhabitants. The questionnaire was sent together with a complimentary lottery ticket in May 2002. A reminder was sent after three weeks to those who had not replied.

The questionnaire consisted of three parts; one with questions about socio-economic status and habits of using the coastal area, one with the choice experiment, and finally one with debriefing questions where the respondents could state certainty ofchoice, their motivations and other comments. The questionnaire was developed in collaboration with marine biologists and tested in several focus groups and in two pilot studies.[4]

The study was presented as a research project to elicit individuals' preferences for different aspects of water quality of the Swedish west coast and that the study could provideuseful information for policy makers concerning decisions of improving water quality. The choice experiment was introduced with a description of the three attributes used and the cost levels. The financing of a potential improvement project was described as a user fee to be collected for one year, also presented as a monthly cost, which would be collected from all permanent citizens aged 18-65 years in the municipalities of the four counties, given that sufficient support for the improvement was found. The respondents were provided with a separate fact-sheet with all the attributes and each respondent faced four choice sets. For each choice set they were asked to choose between three alternatives, where the third alternative was always the baseline or opt-out alternative, i.e. no improvements and no extra costs. Including an opt-out alternative prevents 'forced choices' by respondents, which could bias the results (Banzhaf et al 2001). The two otheralternatives offered various levels of improvements at various costs. The attributes and their levels are briefly described in Table I. In Appendix B, we provide a full description of the attributes, the scenario and an example of a choice set.

[Table I.]

When designing the choice sets for a choice experiment, the aim is to ensure that all different attributes can be estimated independently of each other. On the other hand it is unrealistic to assume that respondents will carry out a high number of choices. To managethis tradeoff, we use a fractional factorial design (Louviere, Hensher and Swait, 2000). The choice sets were constructed as a linear D-optimal main effects design, using the OPTEX procedure in SAS (Kuhfeld, 2001). The 32 choice sets were blocked into 8 groups of4 sets each.

3.2 ECONOMETRIC SPECIFICATION

Using the MXL[5] to analyze discrete choice data overcomes the two major limitations of the MNL model, i.e. the restrictive IIA property and the limited ability to explicitly account for unobserved heterogeneity in taste. However, using the MXL models raises new issues. Letting all parameters be random may lead to convergence problems and poorly defined WTP measures. A useful distribution like the normal often implies a non-negligible fraction of respondents with the reversed sign compared to the expected. The lognormal distribution leads to an unambiguous sign, but it is often problematic to reach convergency for the log-normal. One reason is that the parameters of log-normal distribution are hard to estimate with classical procedures, since the log-likelihood surface is highly non-quadratic (Train and Sonnier, 2003). The uniform distribution is judged suitable when dummy variables are used in a MXL setting. Finally, we have the triangular distribution, which in comparison to those mentioned above has been subject to less attention in applied econometrics so far.[6] However, the triangular distribution is a useful proxy for the normal distribution. In contrast to the normal, the triangular is bounded on both sides, which makes it easy to check whether the estimated bounds make sense. It is also possible to impose specification constraints on the triangular, the normal and the uniform distributions to avoid unacceptable signs on the random parameters (Hensher and Greene 2003). Finally, if heterogeneity in the sample population is confirmed, we may want to know how the individual respondents are distributed. Revelt and Train (2000) provide an approach for dealing with this, which we test on our data.

In the random utility model of the discrete choice family of models, we assume that a sampled individual (q= 1,…,Q) faces a choice among J alternatives in each of T choice situations. The individual is assumed to consider the full set of offered alternatives in choice situation t and to choose the alternative with the highest utility. The relative utility associated with each alternative j as evaluated by each individual q in choice situation t is represented in a discrete choice model by a utility expression of the general form:

$$U_{jtq} = \beta_q ' X_{jtq} + \varepsilon_{jtq} \tag{1}$$

where X_{jtq} is the observed attribute vector including attributes of the alternatives and socio-economic characteristics of the respondent. $\boldsymbol{\beta}_q$ is a vector of marginal utility parameters for individual q and ε_{jtq} is white noise, and where neitherof the latter two are observed by the researcher and treated as stochastic influences.

For the standard logit model we require that ε_{jtq} is independent and identically distributed (IID) extreme value type 1, which means that error components of different alternatives cannot be correlated. One way to relax this is to partition the stochastic component additively into two parts where one part is correlated over alternatives and heteroscedastic while the other is IID over alternatives and individuals (leaving the t subscript):

$$U_{jq} = \beta_q ' X_{jq} + \eta_{jq} + \varepsilon_{jq} \tag{2}$$

where η_{jq} is a random term with zero mean whose distribution over individuals and alternatives depends in general on underlying parameters and observed data relating to alternative j and individual q. The other term, ε_{jq}, is a random term with zero mean that is IID over alternatives and does not depend on underlying parameters or data.

MXL models assume a general distribution for η_{jq} such as normal or triangular and IID extreme value type 1 distribution for ε_{jq}. We denote the joint density of $[\eta_{1q}, \eta_{2q}, \ldots, \eta_{Jq}]$ by $f(\eta_q|\Omega)$ where the elements of Ω are the fixed parameters of the distribution and η_q denotes the vector of J random components in the set of utility functions. For a given value of η_q, the conditional probability for choice j is logit, since the remaining error term is IID extreme value type 1:

$$L_{jq}(\beta_q \mid \eta_{jq}) = \exp(\beta_q{}' X_{jq} + \eta_{jq}) / \sum j \exp(\beta_q{}' X_{jq} + \eta_{jq}) \quad (3)$$

The unconditional choice probability would be this logit probability integrated over all values of η_q weighted by the density of η_q:

$$P_{jq}(\beta_q \mid \Omega) = \int_{n1q} \int_{n2q} \ldots \int_{nJq} L_{jq}(\beta_q \mid \eta_{jq}) f(\eta_{jq} \mid \Omega) d\eta_{jq} \ldots d\eta_{2q} d\eta_{1q} \quad (4)$$

Models of this form are called mixed logit because the choice probability P_{jq} is a mixture of logits with f as the mixing distribution. The standard deviation of an element of the β_q parameter vector, which we denote σ_q, accommodates the presence of preference heterogeneity in the sampled population. The random parameters representation of this carries a challenge in selecting the appropriate distribution. Further, we do not know the location of each individual's preferences on the distribution. However, individual specific estimates can be retrieved by deriving the individual's conditional distribution based on their choices. Using Bayes Rule, we can define the conditional choice probability:

$$H_{jq}(\beta_q \mid \Omega) = L_{jq}(\beta_q \mid \eta_{jq}) g(\beta_q \mid \Omega) \mid \Omega) / P_{jq}(\beta_q \mid \Omega) \qquad (5)$$

where $L_{jq}(\beta_q)$ is now the likelihood of an individual's choice if they had this specific β_q, $g(\beta|\Omega)$ is the distribution in the population of β_qs, and $P_{jq}(\Omega)$ is the choice probability function defined in open-form (see Train 2003):

$$P_{jq}(\Omega) = \int_{\beta q} L_{jq}(\beta_q) g(\beta_q \mid \Omega) d\beta_q \qquad (6)$$

This shows how one can estimate the person specific choice probabilities as a function of the underlying parameters of the distribution of the random parameters.

The choice probability in (4) or (6) cannot be calculated exactly because the integral will not have a closed form. The integral is approximated through simulation (For more details see Train, 2003).

4. Results

In total, 343 of the 800 respondents returned the questionnaire, leading to a response rate of 43%. In the final analysis 324 of these could be used due to non-responses to various items. 317 respondents completed all four choice sets in the questionnaire. 22 respondents (7%) of these chose the status quo-alternative in all four sets and 9 respondents (3%) chose the status quo-alternative in three of the four sets. Table II presents some descriptive statistics of the sample.

We find that 47 percent of the respondents are male with an average age of 42. The average respondent's household consists of three persons, and the disposable household income after tax and benefit transfers is on average SEK 12750.[7] Forty percent have completed at least one semester at the university level. Eleven percent are members in an environmental non-governmental organization like World Wildlife Fund, Greenpeace orthe like. Twenty-five percent of the respondents live by the coast, i.e. less than 1 km from the sea. Sixteenpercent of the respondents have access to a summer cottage by the coast, 91 percent have a car, 100% of all the respondents spend at least one day by the sea and among non-residents the average is 16 days per annum. When visiting the sea, the most popular activities are swimming and sunbathing, practicedby 85% and 83%, respectively.

[Table II.]

Table III shows the results from our estimations. We start with the standard MNL as the base case. Since the model is generic, i.e. the alternatives are not labeled; we include equivalent alternative-specific intercepts for the two alternatives that imply changes in the marine water quality. We use Limdep (Nlogit) to estimate the mixed multinomial logit (MXL) models with simulated maximum likelihood and have a fixed intercept in order to compare the constrained and unconstrained distributions. In addition we provide estimates for unconstrained normal and triangular distributions with a normal distributed intercept.[8]

[Table III.]

Among the attributes, cost is held fixed to make the distribution of the marginal willingness to pay for an attribute equal to the distribution of that attribute's coefficient. A fixed cost variable is also beneficial in the sense that it results in the same sign for all individuals, i.e. non-positive in our study. An alternative to a fixed cost variable would be assuming a log-normal distribution, assuring a non-positive cost variable for all individuals. However, that could lead to extremely high marginal WTP values for the other attributes, as a value close to zero for the cost attribute is possible with the log normal distribution (Revelt and Train, 1998). Furthermore, allowing all coefficients to vary in the population often leads to problems with convergence and makes identification empirically difficult (Ruud, 1996). The log normal distribution is in general more demanding, which we experienced as the specifications with log normal distributed attributes did not converge.[9] All the parameter values for attributes are significant at the 1% level, except for both the constrained and the unconstrained models with fixed intercept and normally distributed parameters.

We note that allowing for distributed parameter estimates substantially improves the fit in terms of pseudo R^2 where values in the range 0.2-0.4 are considered extremely good model fits (Louviere et al., 2000). The best model fit is obtained for the model with normally distributed intercept and attributes. The attributes and the cost parameter are significant at the 1% level for all models, except for the water parameter in the constrained normally distributed MXL model. Heterogeneous preferences in the population are confirmed by the statistical significance of the standard deviation of the random parameters, which issignificant at the 1% level. The two exceptions, constrained and unconstrained normal distribution with fixed intercepts are significant at the 5% level. We find that among the socio-economic variables of age, owning a house, or owning a summer cottage by the coast, (which were significant for the standard MNL), it is only the summer cottage parameter that is significant for the $MXL(N)^e$ model. The respondents' age and whether or not he or she resides by the coast are not significant variables as it appeared from the MNL model, but that variation is more appropriately obtained by the normally distributed intercept.

Using normally distributed parameters implies that some people will most likely appear to have preferences with different signs than the mean, i.e. despite the fact that the fish parameter is positive and significant at the 1% level for all models, there is a fraction of the respondents who would prefer to see fewer fish in the sea. There is nothing in economic theory that tells us what kind of ecosystems the respondents prefer, but sometimes when dealing with attributes that we explicitly expect all respondents to share, an equal sign for such a situation seems problematic. One way to handle this is to constrain the distribution to one side of zero while another approach is to use a one-sided distribution like the log-normal. We tested constrained versions of the triangular and the normally distributed parameters, which implies that none of the area below the triangular distribution has an opposite sign while the normal distribution, due to its shape, still had a 0.62% area with an opposite sign, as reported in Table IV. Among the unconstrained models, the fixed intercept implies larger fractions of opposite sign. The best performing model is again that with intercept and attributes normally distributed, where the reversed sign is less than 20% for fish, water quality, and low biodiversity. We also note that the constrained models, as reported in Table III, have significantly lower model fit compared to the MXL(N)[e]. For our sample the benefits of restricting the parameter sign come at the expense of a reduced model fit, which also indicates that the seemingly significant socio-economic variables 'age' and 'owning a house by the coast' should not be interpreted as significantly different from zero.

[Table IV]

The interpretation of the coefficient values is not straightforward except for the significance. We can calculate the marginal rates of substitution between the attributes and cost, and therefore interpret the ratios as marginal WTP for a change in the attribute in question. With a fixed cost coefficient and normally distributed attributes, marginal WTP is also normally distributed. Researchers involved in applied welfare analysis stress the importance of providing estimates of the precision of welfare measures e.g. standard deviations or confidence intervals (Kling, 1991). Here, we obtain confidence intervals using the Krinsky-Robb method (Krinsky and Robb, 1986) with 8 000 replications. ‘Fish’ relates to the level of cod in the sea and the results indicate an average willingness to pay for improving cod level from the current level, which is the lowest level, to the highest level. Similarly, ‘Water’ relates to an improvement of bathing water quality from the worst to the best level. For the biodiversity level, the current level is the medium level and the estimates relate to an improvement or avoiding further deterioration of biodiversity, respectively. As expected, we find that the disutility from losing further biodiversity is higher than the corresponding gains from improving biodiversity. In Figure 1, we provide a graphic presentation of the mean marginal WTP for each attribute for the various models together with a 95% confidence interval, which is indicated by the vertical lines through the mean. We find that the mean values are fairly stable across models, with water, improved fish stock, high biodiversity, and avoiding lower biodiversity about SEK 600, 1200, 600, and 1400, respectively. The only exception is the fish attribute where the mean ranges from less than SEK 1100 to above SEK 1300. The corresponding 95% confidence intervals are about SEK 300-900, 800-1700, 300-900, and 1000-1800 for water, fish stock, high bio, and low bio, respectively. Here, the exception is the constrained model with normally distributed attributes and a fixed intercept, which yields substantially greater confidence intervals for all attributes with the high biodiversity parameter as the most striking example.

[Figure 1.]

Using the simulated maximum likelihood estimates and conditioning them with individual choices, as described in Section 3.2, we calculate individual level parameters, which are presented as frequency charts in Fig 2-5. The limited number of individuals, i.e. 324 in this case, implies that the distribution is discrete and does not perfectly mimic a smooth normal distribution, which we assumed for the corresponding population parameter model. This indicates one source of error when estimating a discrete set of values using a continuous distribution (Sillano and Ortúzar, 2003). The mean WTP values of the four attributes do not differ in statistical terms but are not identical for the two models. Mean WTP values for water, fish, highbio and lowbio are in turn, with population parameter model first, (589, 589), (1253, 1231), (604, 597), and (1442, 1465).[10]

[Figure 2]

[Figure 3]

[Figure 4]

[Figure 5]

Even more interesting is that we can determine how many of the individuals in fact represent a 'reversed' sign. The percentage of the individual parameter estimates for our sample with a 'reversed' sign is reported in Table V, together with the original estimates for the MXL (N)[e] model (also reported in Table IV).

[Table V.]

We find that the three attributes Water, Fish, and Low biodiversity with moderate numbers of 'reversed' signs for the population parameter model has even a smaller fraction of 'reversed' signs when we estimate individual parameters. This is of course case-specific, but indicates that a modest fraction of 'reversed' signs for a population parameter where an opposite sign doesn't make sense may not always be such a great problem. The fourth attribute high biodiversity has a substantial fraction of 'reversed' signs for the individual parameter estimate as well. This attribute is also conspicuous as the distribution for the individual parameter estimate is not unimodal but rather a distinct bimodal distribution[11], where there is a substantial group of respondents who in fact have a negative WTP for improving biodiversity. We do not know why these respondents are negative to an improvement of biodiversity, but one explanation may be that improving biodiversity is a vague project, which leaves respondents more skeptical. Improving bathing water quality and increasing the cod stock are well-defined projects, and avoiding loss of biodiversity seems more urgent than improving biodiversity.

5. Discussion and Conclusion

In this paper we use a choice experiment to elicit an individual's valuation of marine water quality improvements. We employ three different attributes as indicators of marine water quality; fish stock, bathing water quality, and level of biodiversity.The data is analyzed using various specifications of mixed multinomial logit models (MXL), which enables us to study heterogeneity in preferences for the different attributes. We also explore the use of individual parameter estimates. We find that the explanatory power of the logit models increases substantially when allowing for heterogeneity compared to the standard multinomial logit model. This finding applies to various distributional assumptions, i.e. constrained and unconstrained normal and triangular distributions with fixed and random intercepts. Estimated means and standard deviations for the quality attributes are significant, which confirm the existence of heterogeneous preferences for these attributes among the respondents in our sample. We also test the possibility of avoiding a reversed sign for a substantial part of the sample population, i.e. constraining distributions to one side of zero. In our approach to the constrained models, where we make the standard deviation or the spread a function of the mean, the gains from the constraint always come at the expense of reduced model fit. Sonnier and Train (2003) suggest a Bayesian estimation procedure, which for their sample leads to improved model fit for models where the normal distributions are censored from below at zero. A potential drawback with their approach is that instead of 40% and 20% 'reversed' signs for two attributes, they obtained 70% and 50% zeros, i.e. more than half of the respondents are completely unconcerned about these attributes. This phenomenon may well be inherent to their particular sample, but may also be the result of using too many attributes and making the choice experiment too complex. If respondents find a questionnaire too difficult to fill in, they may use a lexicographic strategy to facilitate the problem of solving the exercise, even though their true preferences may be more complex (Payne et al. 1993). This highlights the importance of choosing an appropriate amount of attributes and comparisons to be carried out by the respondents in a choice experiment. Whether Bayesian estimation procedures are preferable to classical methods is beyond the scope of this paper, yet we nonetheless refer to a recent paper which finds that the overall superiority of the Bayesian method is overstated (Hensher, Greene and Rose, 2003). The occurrence of a 'reversed' sign when applying MXL models may be puzzling and using individual parameter estimates is one approach to further investigate such cases. In our study, we found that a 'reversed' sign was not a real problem for two of the three attributes since the fraction of true 'reversed' sign respondents was so small. For the third attribute, we realized that choosing a reference level which enables either improvement or deterioration of an attribute most likely implies that the changes should be treatedseparately. Avoiding deterioration had a unimodal distribution and a very small number of 'reversed' sign respondents, while improving biodiversity was rejected by a substantial fraction of the respondents. The population parameter models all confirmed heterogeneity, i.e. significant standard deviation and improved model fit, but the individual parameter model also showed that concerning high biodiversity respondents were divided into two significant groups. Our guess is that this split is the result of the vagueness for ‘improving’ biodiversity and maybe also for ‘avoiding deterioration’ and ‘improving’ biodiversity. To explore why some respondents show a ‘reversed’ sign and link that question to, for example, uncertainty are issues for future research. Methodologically, there seems to be room for developing more suitable distributions for specific situations. In our case, an applicable bimodal distribution to model heterogeneity of the type where respondents are either for or against a change in the attribute level could be useful.

We calculate willingness to pay (WTP) for changes in the attributes, which are presented with estimated confidence intervals using the Krinsky-Robb method. Both mean values and confidence intervals are fairly stable across models with a few exceptions, such as constraining the distribution to one side of zero usually leads to either deviation in mean estimate or increased confidence intervals. The best-performing model possesses normally distributed intercept and parameters, but for this model as well, the confidence intervals are quite sizable. The changes measured for the fish stock and bathing water attributes are from the lowest to the highest level, while the biodiversity reference point is medium. The highest average marginal willingness to pay value, SEK 1400, is found for avoiding a reduction in biodiversity level.. The corresponding figure for improved biodiversity level is SEK 600, the marginal WTP for improved bathing water quality is SEK 600, and an improved fish stock leads to an average marginal WTP of SEK 1300. The studied area comprises 20% of the total Swedish population and has roughly one million inhabitants aged 20-64.. Assuming zero willingness to pay from all non-respondents implies that the respondents represent 40% of the entire population, which leads to a rough aggregate estimate of SEK 400-700 million for either improving the cod stock or avoiding deterioration of marine biodiversity. As a comparison we note that annual Swedish commercial landings of cod in total fish amounts to an ex-vessel value of SEK 300 million and SEK 1000 million, respectively (Sweden Statistics, 2003). The overall finding is that choice experiments offer a suitable way to assess multi-attribute values, as in the case of water quality. Marine reserves, currently a hot topic, provides a relevant example.Up to now, almost all economic research has focused on stock and harvest effects. The results of this study indicate that non-consumptive benefits like biodiversity should be addressed as well.

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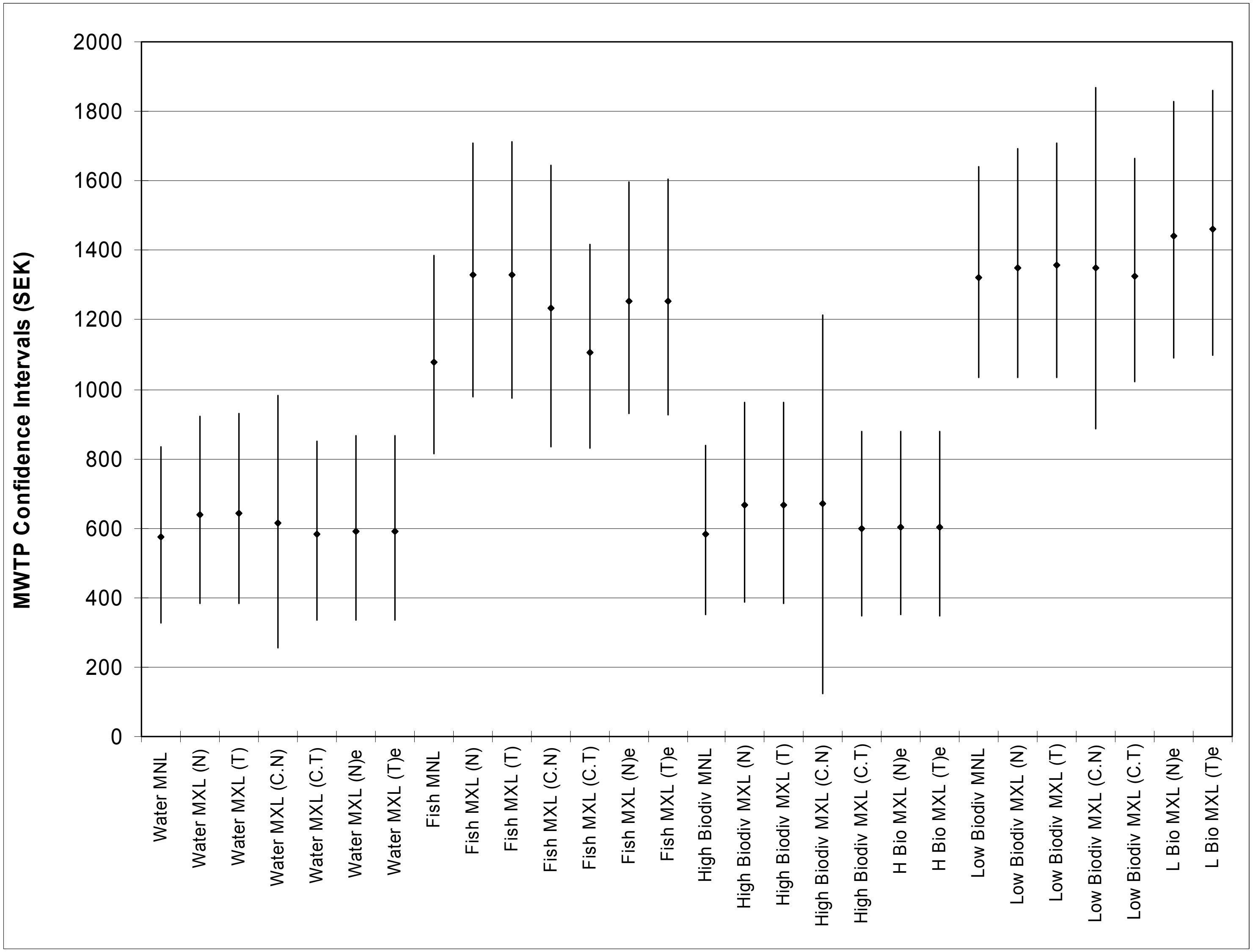
Figure 1 Marginal WTP with confidence intervals for various models



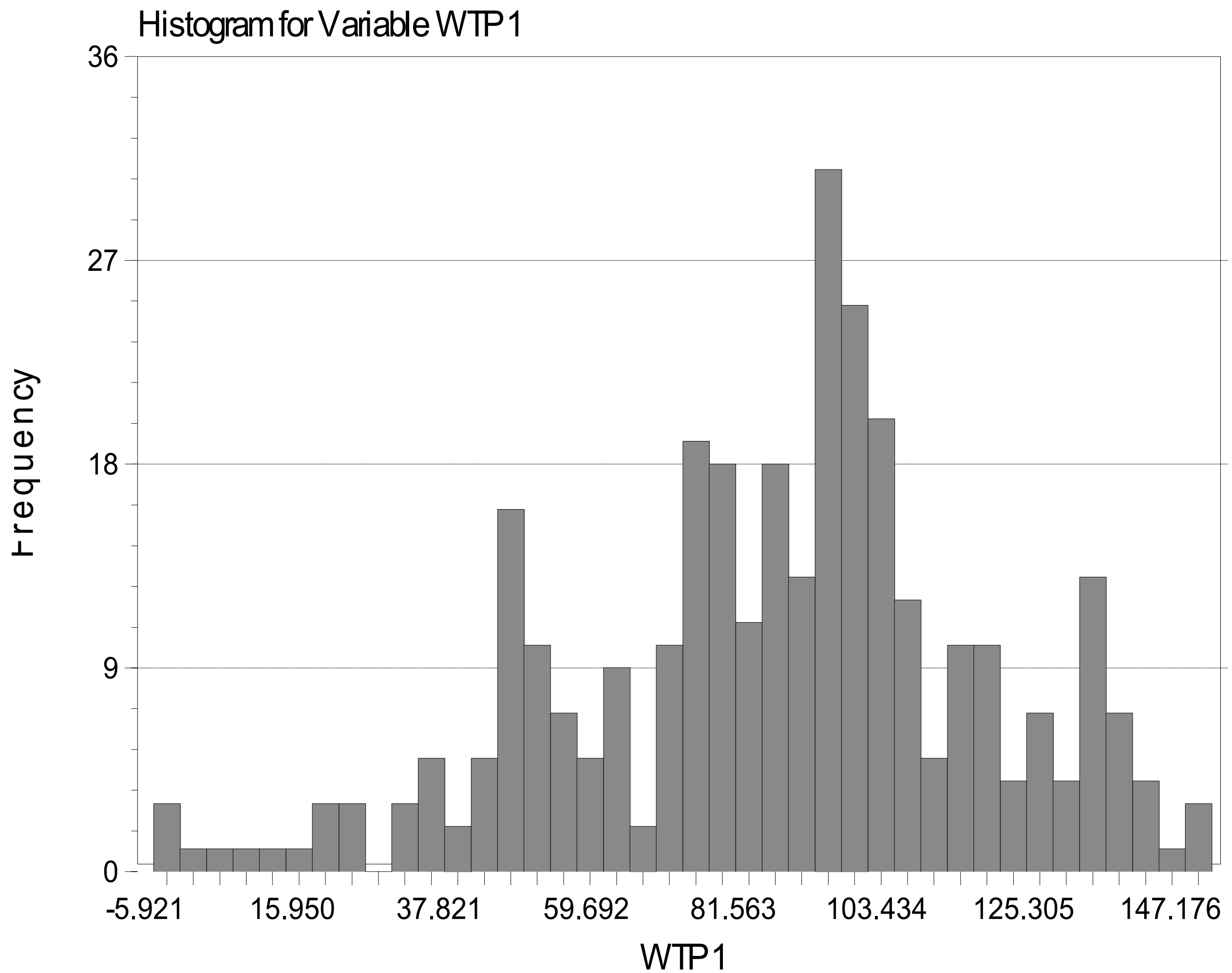


Figure 2. Histogram of Water quality attribute individual point estimates for sampled population

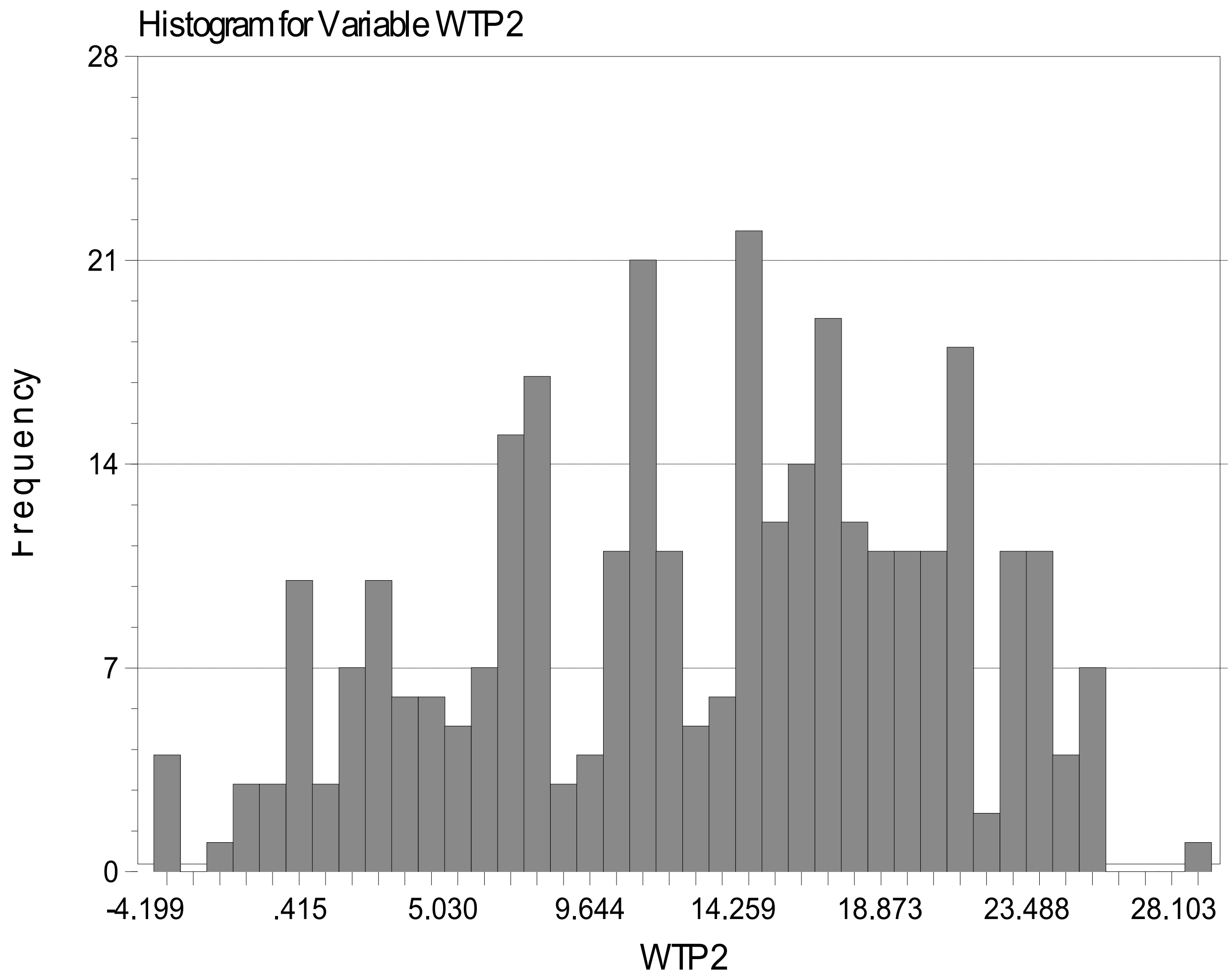


Figure 3. Histogram of Cod stock level attribute individual point estimates for sampled population.

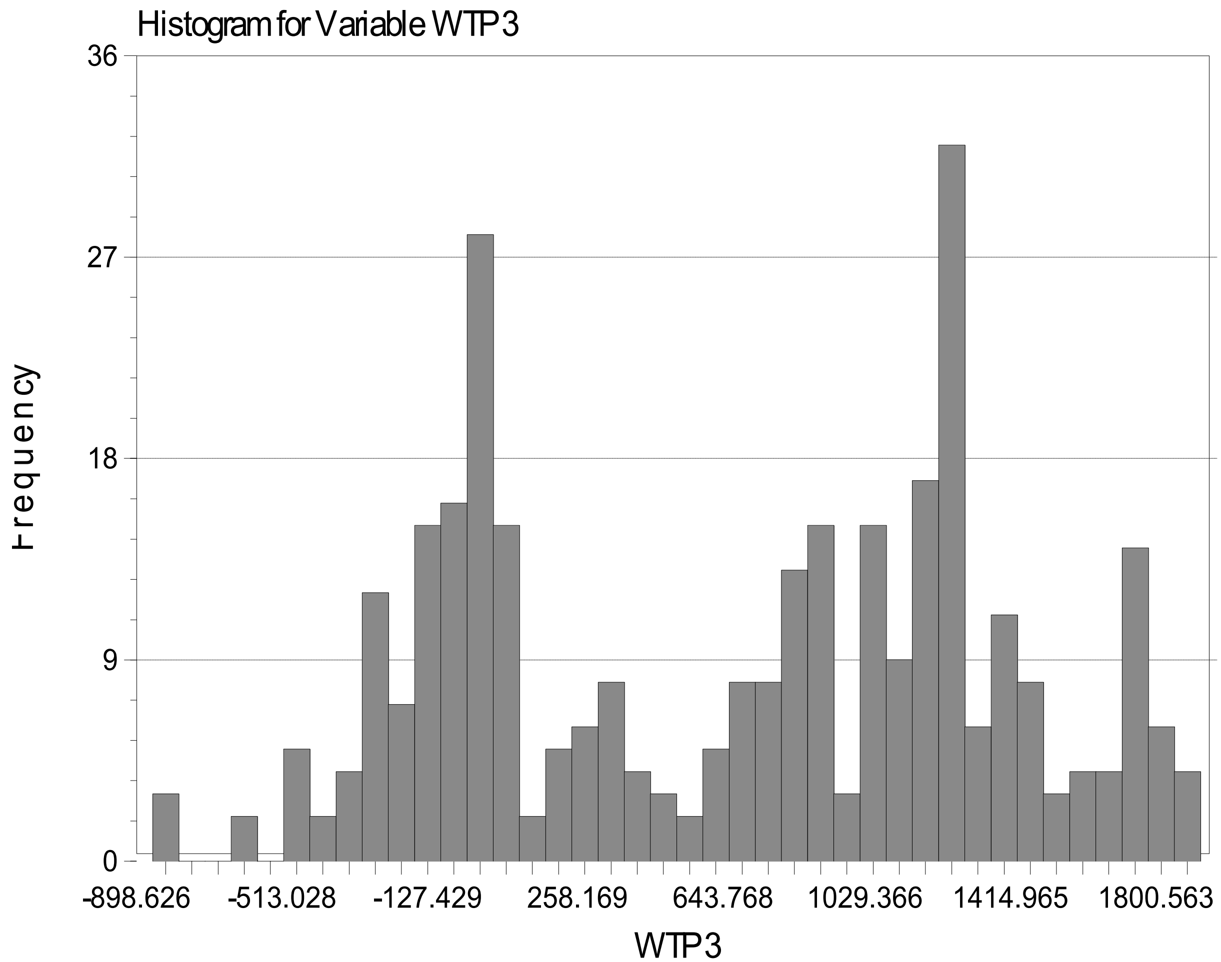


Figure 4. Histogram of High biodiversity level attribute individual point estimates for sampled population.

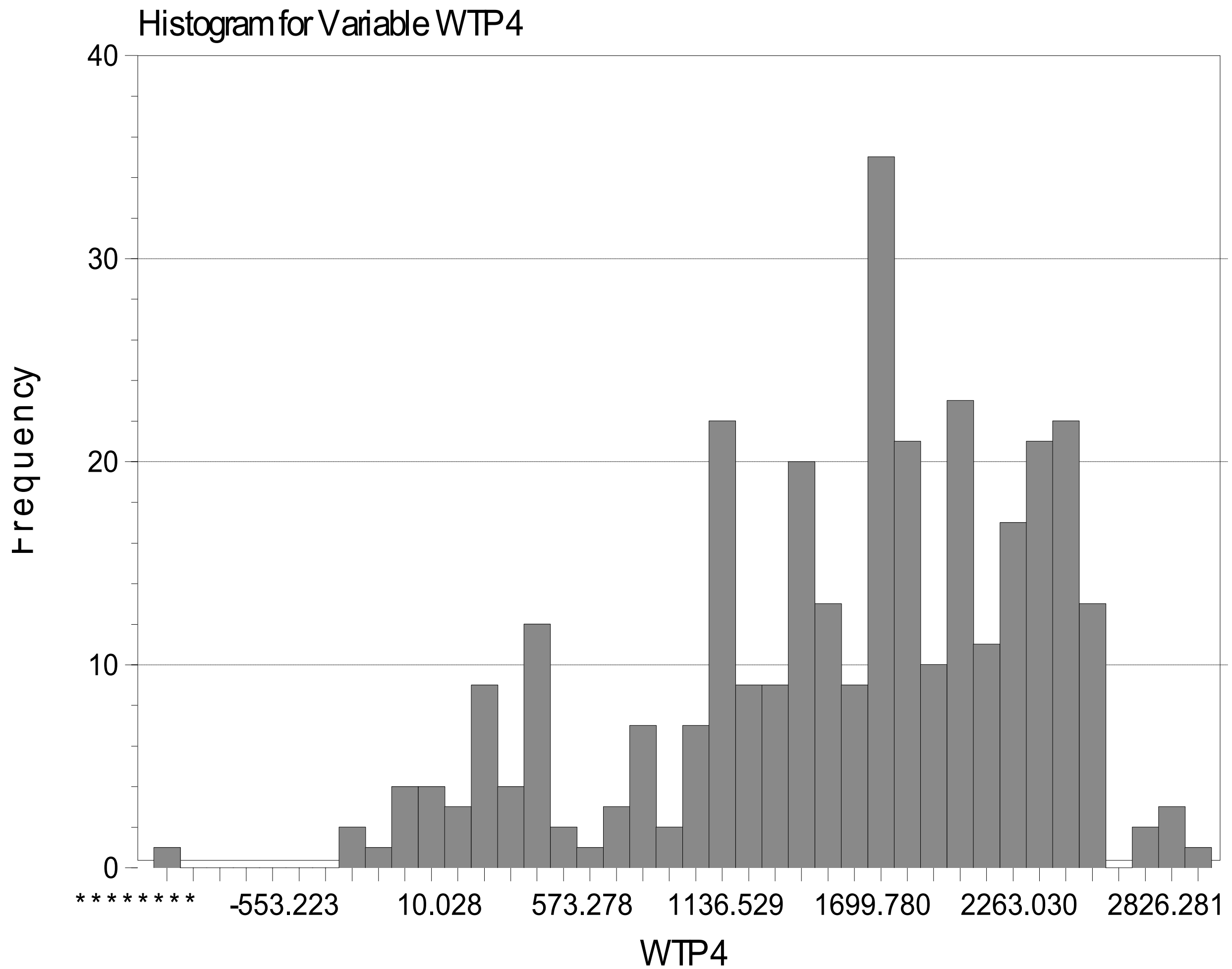


Figure 5. Histogram of Low biodiversity level attribute individual point estimates for sampled population

Table I. Brief description of the attributes and their levels (bold figures indicate baseline)

Attribute	Description	Levels
Bathing water quality (%)	Fraction of west-coastal sites violating the quality standard	**12**, 10, 5
Biodiversity	Biological diversity or ecosystem balance, where today's level is medium.	Low, **Medium** High
Cod stock (kg)	Catch per trawling hour with a research vessel.	**2**, 25, 100
Cost (SEK)	The total cost for an individual for each alternative	**0**, 120, 240, 600, 960, 1800

Table II. Descriptive statistics for respondents, n = 328.

Variable	Mean	Std dev	Min	Max
Male	0,47	0,50	0,00	1,00
Age (years)	42	13,29	18	65
Houshold size	3,0	2,8	1	50
Equivalenced Household Income (SEK)	12750	6364	1212	40000
University education	0,40	0,49	0	1
Member of env. NGO	0,11	0,31	0	1
Home ≤ 1km from the coast	0,25	0,43	0	1
Summer cottage ≤ 1km from the coast	0,16	0,36	0	1
Car ownership	0,91	0,29	0	1
Activities when visiting the sea				
Swimming	0,85	0,36	0	1
Sunbathing	0,83	0,38	0	1
Walking	0,64	0,48	0	1
Barbequing	0,44	0,50	0	1
Sailing / Yachting	0,37	0,48	0	1
Flora / Fauna watching	0,30	0,46	0	1
Fishing	0,25	0,43	0	1
Other activities	0,14	0,35	0	1
Canoeing	0,05	0,22	0	1
Scuba diving	0,04	0,19	0	1

TABLE III. Multinomial and Mixed multinomial logit estimations

	MNL	MXL (N)	MXL (T)	MXL (C. N)	MXL (C. T)	MXL (N)[e]	MXL (T)[e]
Water	-0,07861 [a]	-0,1495 [a]	-0,1494 [a]	-0,08904 [a]	-0,08021 [a]	-0,1368 [a]	-0,1357 [a]
Fish	0,01055 [a]	0,02215 [a]	0,02201 [a]	0,01280 [a]	0,01091 [a]	0,02080 [a]	0,02024 [a]
High Biodiv	0,5605 [a]	1,1340 [b]	1,1246 [a]	0,6753 [b]	0,5776 [a]	0,9905 [a]	0,9769 [a]
Low Biodiv	-1,2714 [a]	-2,2202 [a]	-2,2214 [a]	-1,3736 [a]	-1,2902 [a]	-2,3812 [a]	-2,3822 [a]
Intercept	1,523 [a]	2,0622 [a]	2,0595 [a]	1,471 [a]	1,5149 [a]	3,525 [a]	3,463 [a]
Cost	-0,000969 [a]	-0,001643 [a]	-0,001635 [a]	-0,001021 [a]	-0,000977 [a]	-0,001649 [a]	-0,001633 [a]
Age	-0,01243 [b]	-0,008473	-0,008960	-0,01285 [b]	-0,01251 [b]	-0,02058	-0,01915
Coast	0,3895 [b]	0,3780	0,3819	0,4058 [b]	0,3922 [b]	0,8196	0,7092
Cottage	0,7497 [a]	0,8077 [b]	0,8209 [b]	0,7837 [a]	0,7553 [a]	1,5500 [b]	1,4873 [b]
Education	-0,2994 [c]	-0,7648 [a]	-0,7594 [a]	-0,3191 [c]	-0,3019 [c]	-1,1020 [b]	-1,1451 [b]
STD DEV. OF RANDOM PARAMETER DISTR							
σ – Water	-	0,2373 [a]	0,5713 [a]	0,03562 [a]	0,03275 [a]	0,1582 [a]	0,4039 [a]
σ – Fish	-	0,02820 [a]	0,06621 [a]	0,005120 [a]	0,004455 [a]	0,02107 [a]	0,04710 [a]
σ –High Biodiv	-	2,3705 [b]	5,6281 [a]	0,27011 [b]	0,2358 [a]	1,9340 [a]	4,7707 [a]
σ –Low Biodiv	-	1,9411 [a]	4,6854 [a]	0,5494 [a]	0,5267 [a]	2,1404 [a]	4,7658 [a]
Intercept	-	-	-	-	-	2,9056 [a]	2,8192 [a]
Pseudo-R^2 [d]	0,183	0,240	0,240	0,191	0,184	0,311	0,310
No. of obs.	1249	1249	1249	1249	1249	1249	1249

a) significant at 1% level b) significant at 5% level c) significant at 10% level
d) Pseudo-R^2 is computed as 1- LL / (LL at 0) e) normal distributed intercept

TABLE IV. Probability of reversed sign

	MXL (N)	MXL (T)	MXL (C. N)	MXL (C. T)	MXL (N)[e]	MXL (T)[e]
Water	22 %	37 %	0,62 %	0 %	16%	34%
Fish	26 %	40 %	0,62 %	0 %	19%	37%
High Biodiv	32 %	42 %	0,62 %	0 %	30%	42%
Low Biodiv	13 %	33 %	0,62 %	0 %	13%	32%

Fixed intercep if not else indicated e) normal distributed intercept

Table V. Percentage of individual parameters with reversed sign (%)

Attribute	Percentage with reversed sign (%)	
	Sample population distribution	Individual parameters
Water	16%	1%
Fish	19%	4%
High biodiversity	30%	31%
Low biodiversity	13%	4%

APPENDIX A.

Table VI. Multinomial and mixed multinomial logit estimations, t-statistics in parentheses.

	MNL	MXL (N)	MXL (T)	MXL (U)	MXL (T/U)	MXL (N/U)
Intercept (N)[b]	1,523 (4,955)	3,525 (3,869)	3,463 (3,913)	3,383 (3,940)	3,495 (3,946)	3,458 (3,860)
Water	-0,07861 (-4,734)	-0,1368 (-4,533)	-0,1357 (-4,521)	-0,1343 (-4,666)	-0,1346 [t)] (-4,562)	-0,1380 (-4,588)
Fish	0,01055 (9,031)	0,02080 (6,843)	0,02024 (7,046)	0,02015 (6,994)	0,02015 [t)] (7,007)	0,02059 (6,934)
High Biodiv	0,5605 (5,194)	0,9905 (4,517)	0,9769 (4,419)	0,9475 (4,484)	0,9614 [u)] (4,409)	0,9804 (4,468)
Low Biodiv	-1,2714 (-10,240)	-2,3812 (-7,365)	-2,3822 (-7,378)	-2,4121 (-7,053)	-2,4180 [u)] (-7,040)	-2,4358 (-7,017)
Cost	-0,0009686 (-11,549)	-0,001649 (-9,731)	-0,001633 (-9,863)	-0,001605 (-10,092)	-0,001621 (-9,902)	-0,001636 (-9,825)
Age	-0,01243 (-2,096)	-0,02058 (-1,160)	-0,01915 (-1,112)	-0,01914 (-1,144)	-0,01992 (-1,162)	-,01984 (-1,130)
Coast	0,3895 (2,044)	0,8196 (1,507)	0,7092 (1,322)	0,7646 (1,477)	0,7041 (1,317)	0,7840 (1,457)
Cottage	0,7497 (2,937)	1,5500 (2,131)	1,4873 (2,081)	1,4595 (2,120)	1,4580 (2,046)	1,5308 (2,112)
Education	-0,2994 (-1,859)	-1,1020 (-2,193)	-1,1451 (-2,298)	-1,0550 (-2,215)	-1,1369 (-2,296)	-1,0623 (-2,148)
STANDARD DEVIATIONS OF RAND. PARAM. DIST.						
σ – Intercept (N)	-	2,9056 (7,306)	2,8192 (7,612)	2,7281 (7,851)	2,8139 (7,637)	2,8862 (7,380)
σ – Water	-	0,1582 (2,000)	0,4039 (1,988)	0,2090 (1,367)	0,3674 [t)] (1,708)	0,1557 (2,045)
σ – Fish	-	0,02107 (4,628)	0,04710 (4,371)	0,03424 (4,848)	0,04806 [t)] (4,391)	0,02097 (4,599)
σ –High Biodiv	-	1,9340 (5,330)	4,7707 (5,267)	3,0760 (5,646)	3,1861 [u)] (5,435)	3,1911 (5,601)
σ –Low Biodiv	-	2,1404 (4,919)	4,7658 (4,927)	3,4971 (4,974)	3,4145 [u)] (4,887)	3,5527 (4,993)
Pseudo-R^2 [a)]	0,18295	0,31055	0,30984	0,30945	0,30951	0,31022
No. of obs.	1249	1249	1249	1249	1249	1249

a) Pseudo-R^2 is computed as 1- LL / (LL at 0)
b) The intercept was always assumed normally distributed
t) Triangular distribution assumed
u) Uniform distribution assumed

APPENDIX B. **Scenario and example of choice question**

You have been randomly selected together with a large number of people living in west Sweden to participate in this survey. We are investigating individuals' choices for various measures affecting sea water quality (in terms of bathing, fishing, recreation etc.)

Below, we describe three factors characterizing water quality in Västerhavet. We ask you to consider these factors and the costs for carrying out various measures in the choice questions that follow in Section C.

Bathing water quality

The EU regulations for bathing water quality recommend testing every other week during the season for sites with a daily visitor rate above 100 for a "normal" summer day. The level of bacteria, chemicals, oil and other compounds are determined and if standards are not met, the site fails to pass the standard.

Bathing water quality is expressed as frequency of failure to pass standard	
● 12% of the sites fail	(today's level)
● 10% of the sites fail	(1998 level)
● 5% of the sites fail	(1995 level)

Cod abundance

During the 1970s the coastal cod level corresponded to a 100 kg per hour catch for a trawling research vessel. In 1992 this level had decreased to an hourly catch of 25 kg and today the level leads to a 2 kg per hour catch. Today's cod stock level implies that recreational anglers catch scarcely any cod at all.

Coastal cod abundance on the Swedish west coast:	
● 2 kg cod per trawl hour with research vessel	(today's level)
● 25 kg cod per trawl hour with research vessel	(1992 level)
● 100 kg cod per trawl hour with research vessel	(1974 level)

Biological diversity

Biodiversity in the sea consists of both richness in species and richness within each species. Biodiversity is important to the sea's resilience capacity to handle environmental disturbances, but also for productivity (e.g. for fish). It is hard to indicate the direct utility to humans derived from biodiversity.

Biodiversity in the sea is assumed to be represented by three levels:
● Low level of biodiversity - the ecosystem of the sea is not in balance.
● Medium level of biodiversity - the ecosystem of the sea is in balance (today's level).
● High level of biodiversity - the ecosystem of the sea is in good balance.

Cost
Assume that the project is financed by a temporary fee forone year. The fee is collected among all permanent residents aged18-65 in west Sweden, given that enough people support this. If support for the project is too low, no measures will be taken.

Cost due to the project: ● total cost SEK 120 (Implies paying SEK 10/month during the particular year) ● total cost SEK 240 (Implies paying SEK 20/ month during the particular year) ● total cost SEK 600 (Implies paying SEK 50/ month during the particular year) ● total cost SEK 960 (Implies paying SEK 80/ month during the particular year) ● total cost SEK 1800(Implies paying SEK 150/ month during the particular year)

There are no "correct" answers, but priorities have to be made. We ask you to carefully choose between the alternatives below understanding that these choices may be difficult. Please consider bathing water quality, cod abundance, and biodiversity level. Assume that the levels of these three attributes are independent of each other.. Please mark the preferred alternative and treat each alternative as if it is the only choice you make. Please feel free to go back and change your choice in a previous question.

Choices

Question 1

	EFFECTS OF PROJECT A	**EFFECTS OF PROJECT A**	NO MEASURES (TODAYS LEVEL)
Frequency of failure to pass bathing water quality standard	12 %	12 %	12 %
Kg cod per trawl hour with research vessel.	100 kg	25 kg	2 kg
Level of biodiversity	High	High	Medium
Total cost (cost per month)	SEK 1800 (SEK 150)	SEK 120 (SEK 10)	SEK 0 (SEK 0)
I prefer:			

[1] See further www.hsr.se or www.blueflag.org

[2] Alpizar et al. (2003), Hanley et al. (1998), and Hanley et al. (2001) offer more details on the method.

[3] Research in psychology has shown that individuals may well make choices without knowing why a particular choice was made (Nisbett and DeCamp Wilson, 1977).

[4] Initially, the questionnaire also included an attribute with levels of green algae, but that version received a substantially lower response rate in the pilot studies and was therefore excluded in the final version.

[5] The presentation in this section is based on Hensher, Greene and Rose (2003), which we find intuitive. Train's book (2003) is the standard reference for MXL models where all issues relating to this paper arecovered.

[6] For more details on the triangular distribution and its applications, see Johnson and Kotz (1999).

[7] In order to compare income between households, we employ the equivalence scale used by the National Tax Board (RSV) in Sweden. The scale assigns the first adult the value of 0.95, the following adults are set at 0.7 and each child at 0.61 units.

[8] We also tested the uniform distribution for the biodiversity parameters, which leads to a similar result with only a minor reduction in model fit compared to the normal; hence, we do not report any of these results.

[9] We also tested for correlation between attributes, but a majority of interaction terms being insignificant persuaded us to leave that option.

[10] We used the levels reported in Table 1, i.e. we multiply the values shown in Figure 2 by (12-7) to get the individual parameter mean WTP for water;correspondingly, fish is multiplied by (100-2).

[11] We also tested a uniform distribution for the highbio attribute, which leads to nonsignificant reduction in model fit.