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Elasticities of Demand and Willingness to Pay for Environmental Services in Sweden

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Abstract. Are environmental services luxuries or necessities? Are low-income groups relatively more willing to pay for environmental improvements than high-income groups? The discussion on the shape of the environmental Kuznets curve and environmental justice call for analyses that approach these questions. Following a survey-based approach for modelling the demand for public goods, this paper provides estimates of income and price elasticities of demand for reduced marine eutrophication effects in the case of the Baltic Sea, using data from five Swedish contingent valuation studies. Point estimates indicate that reduced marine eutrophication effects can be classified as a necessity and an ordinary and price elastic service. Confidence intervals show however that the classification as a necessity is not statistically significant. Income elasticities of willingness to pay, not to be confused with income elasticities of demand, are estimated for a broad range of environmental services in Sweden. A basic finding is that income tends to influence willingness to pay positively and significantly. The elasticity estimates are in most cases greater than zero, but less than unity, indicating that the benefits of environmental improvements tend to be regressively distributed. In a cost-benefit analysis of a project suggesting environmental improvements, distributional concerns therefore call for an introduction of weights or at least a sensitivity analysis of how weighting would change decisions about the project's social profitability.

Key words: demand for public goods, environmental justice, environmental services, eutrophication, income distribution, income elasticity, price elasticity

JEL classifications: D63, H22, Q20, Q21

1. Introduction

A traditional way to make intercommodity comparisons for consumers' economic behaviour is to estimate various elasticity measures for different goods and services. There are also reasons to study services provided by the environment in this way.[1] For example, there is a discussion on whether environmental services are characterized by an income elasticity of demand greater than unity or not, i.e. whether they can be classified as luxuries or not, cf. Pearce (1980) and Kriström and Riera (1996). This discussion is related to the possible existence of an "environmental Kuznets curve": an inverted U shaped empirical relationship between industrial pollution and per capita income, implying that pollution increases in early stages of economic development in a country, but with a turning point after which pollution decreases with increased per capita income. The existence of such a relationship is often regarded as a "stylized fact", cf. de Bruyn and Heintz (1999), but it seems to be a hasty conclusion that economic growth is a general cure for environmental damage (Arrow et al. 1995). Many explanations for the shape of the environmental Kuznets curve have been suggested; behavioural changes and preferences, institutional changes, technological and organizational changes, and international reallocation are potential explanations listed by de Bruyn and Heintz (1999). This suggests that while information on the relationship between income and demanded quantities of environmental services is relevant for explaining the shape of the curve, it is not enough for a full explanation.

Quite independent of the issue whether environmental services are luxuries or not, there are distributional reasons to be concerned about what income groups in society are relatively more willing to pay for an increased provision of environmental services, see Kanninen and Kriström (1992), Kriström and Riera (1996) and Ebert (2002). This calls for information on the magnitude of the income elasticity of willingness to pay for environmental services, i.e. a measure of how willingness to pay (WTP) is affected by changes in income.

For environmental policy-makers, another useful piece of information is how the demanded quantity of environmental services is affected by price changes. Technical innovations might imply reduced costs of supplying environmental services, and knowledge of price elasticities of demand might thus predict how consumers would respond to such a change. One might also be interested in predicting the response from introducing economic policy instruments such as taxes, charges or subsidies in order to influence people's and firms' behaviour vis-à-vis the environment.

How can income and price elasticities for environmental services be estimated? The typically public good nature of such services and the ensuing lack of markets introduce intriguing difficulties in estimating the demand for them. A number of estimation methods have been developed in order to resolve this problem, see, e.g., Freeman (1993). They are often referred to as indirect and direct methods, or as revealed preferences and stated preferences methods. The indirect methods rely on individuals' actual behaviour on markets for private goods whose relationship to the environmental service is characterised by weak complementarity or some other link that allows the demand for the environmental service to be revealed. The direct methods mainly rely on individuals' hypothetical behaviour on markets set up for the environmental service in some survey setting. That is, people get an opportunity to state what preferences they have. The contingent valuation method (CVM) is a widely used direct method, see, e.g., Mitchell and Carson (1989), Bateman and Willis (1999) and Carson (2000).

The purpose of this paper is to provide estimates of elasticities of demand and WTP for environmental services in Sweden. The focus on Swedes' economic behaviour vis-à-vis environmental services implies a simplification in the sense that potential international differences in such behaviour are not considered. The income elasticity of WTP is estimated for a broad range of environmental services, but a focus on the demand for one particular environmental service was needed in order to estimate income and price elasticities of demand. More precisely, this service was reduced marine eutrophication effects, which have been subject to several valuation studies. In Sweden, the contingent valuation method is the most widely used method for valuing environmental services, implying that the particularities of this method form a point of departure for our analysis.

The paper is organized as follows. The next section (2) gives a theoretical background and defines elasticity measures. Estimates of income elasticities of WTP for environmental services in Sweden are presented in Section 3. The demand for reduced eutrophication effects in the Baltic Sea is modelled and estimated in Section 4, which also includes estimates of income and price elasticities of demand for this particular environmental service. Finally, conclusions are found in Section 5.

2. Elasticities of Demand and Willingness to Pay

In order to derive expressions for elasticities, we follow Freeman (1993) and assume individuals to maximize utility (u), which is determined by the consumption of private goods (an n-vector $\boldsymbol{x}$) and the levels of public environmental services. The latter is for notational simplicity assumed to be a single environmental service z. In real-world settings, the level of z is typically rationed, and the public nature of z implies that no market price exists for this service. In a CVM setting, respondents are invited to behave as if there was a market for z. However, no real exchange of goods and money takes place. This means that one cannot take for granted that the responses obtained in a CVM survey correspond to the behaviour that would arise if the hypothetical CVM setting was turned to a real market situation. The suggestion that CVM responses are influenced by a "hypothetical bias" has been discussed and analyzed with mixed results elsewhere, see, e.g. Carson et al. (1996), Cummings et al. (1995), Frykblom (1997), Neill et al. (1994), and for recent reviews of the issues, Boyle and Bergstrom (1999) and Carson et al. (2001). In this paper, results from CVM studies are used without any attempt to adjust for the possible existence of a hypothetical bias.

A CVM market setting advanced enough to allow choices between different price and quantity combinations would imply that the individual can be assumed to maximize a utility function $u = U(\boldsymbol{x}, z)$ in $\boldsymbol{x}$ and z. The maximization is carried out subject to a budget constraint $\boldsymbol{qx} + pz = y$, where $\boldsymbol{q}$ is an n-vector of market prices of private goods, p is the virtual price of the environmental service, and y is income. Solving this maximization problem would give a set of Marshallian demand functions, including one for z: $z = D_z(\boldsymbol{q}, p, y)$. Inserting them in the utility function results in an indirect utility function $v = V(\boldsymbol{q}, p, y)$, where v is indirect utility.

From $D_z(\bullet)$, the price elasticity of demand (ε_p) and the income elasticity of demand (ε_y) are defined as:

$$\varepsilon_p = \frac{p}{z} \cdot \frac{\partial D_z}{\partial p} = \frac{\partial(ln\ D_z)}{\partial(ln\ p)} \tag{1}$$

$$\varepsilon_y = \frac{y}{z} \cdot \frac{\partial D_z}{\partial y} = \frac{\partial(ln\ D_z)}{\partial(ln\ y)} \tag{2}$$

where ε_p is used for defining Giffen goods ($\varepsilon_p > 0$), ordinary goods ($\varepsilon_p < 0$), price inelastic goods ($-1 < \varepsilon_p < 0$), price unit elastic goods ($\varepsilon_p = -1$) and price elastic goods ($\varepsilon_p < -1$), and ε_y is used for defining inferior goods ($\varepsilon_y < 0$) and normal goods ($\varepsilon_y > 0$), and necessities ($\varepsilon_y < 1$) and luxury goods ($\varepsilon_y > 1$).

In CVM studies, methodological and budgetary considerations often introduce restrictions in the market for the environmental service in the sense that only one particular change in the provision of the service is subject to study. In such a constrained setting, the individual cannot maximize $U(\boldsymbol{x}, z)$ in z. z thus becomes an argument in the indirect utility function, and the focus in the analysis is typically the welfare effect of the changed provision. In most CVM applications, welfare change is estimated as WTP, where the WTP for an increase in z from z^0 to z^1 is implicitly defined from the indirect utility function as $V(\boldsymbol{q}, y - WTP, z^1) = V(\boldsymbol{q}, y, z^0)$, i.e. WTP corresponds in this case to the compensating variation, see, e.g., Johansson (1993). The WTP is estimated from respondents' answers to a WTP question, which might be of a discrete choice (DC) type, so that respondents are asked to accept or reject to pay a given amount of money (a "bid") for obtaining the change in z. A main alternative is to pose an open-ended (OE) question. In this case, respondents are asked to state their maximum WTP for obtaining the change in z.

Such restricted CVM market settings do not allow the estimation of a demand function and, consequently, the elasticities defined above cannot be calculated. However, CVM studies often include an estimation of a function $WTP = W(\boldsymbol{r})$, usually referred to as a "value function", "valuation function" or "WTP function". Such a function tries to explain the variation in WTP by regressing WTP on a vector of explanatory variables $\boldsymbol{r}$, e.g. income and other socio-economic characteristics of the respondents to the CVM survey. The inclusion of income as an explanatory variable makes it possible to use an estimated valuation function for a computation of the income elasticity of WTP (ε_w):

$$\varepsilon_w = \frac{y}{WTP} \cdot \frac{\partial W}{\partial y} = \frac{\partial(ln\ W)}{\partial(ln\ y)}. \tag{3}$$

How are the three elasticities defined by Eqs. 1–3 related to each other? For example, does an estimate of ε_w give any information on ε_y, making it possible to use an estimated valuation function for concluding whether a particular environmental service is a luxury good or not? The results of Flores and Carson (1997) indicate that the answer is generally negative. Their analysis of a general case involving more than one public good showed that a substantial divergence is possible, so that, for example, an environmental service characterized by $\varepsilon_y > 1$ may have an income elasticity of WTP being greater than unity or less than unity. In general, estimates of ε_w are thus of little help for resolving discussions of whether environmental services tend to be necessities or luxuries.

However, estimates of ε_w are of interest for their own sake because of the information they give on the distribution of environmental benefits, at least when potential distributional impacts of taxes financing the provision of such benefits are not considered. Following Kriström and Riera (1996), if $\varepsilon_w < 1$, then $\partial(WTP/y)/\partial y < 0$, i.e. the proportion of income that is assigned as WTP for an increase in z decreases when income increases. If so, a project suggesting this particular environmental improvement would be relatively more beneficial for low-income groups than for high-income groups. ε_w thus indicates whether environmental benefits are distributed regressively ($\varepsilon_w < 1$), proportionally ($\varepsilon_w = 1$) or progressively ($\varepsilon_w > 1$). See Ebert (2002) for a detailed analysis. Furthermore, Ebert (2002) showed that in a case characterized by one environmental service and individuals not differing with respect to preference orderings but only with respect to income, there is a handy relationship between ε_y and ε_p and the income elasticity of WTP implying that benefits are regressively (proportionally) [progressively] distributed if $-(\varepsilon_y/\varepsilon_p)$ is less than (equal to) [greater than] unity.

Given no other weighting of WTP of different income groups than that implied by the existing income distribution, and the use of the Kaldor compensation criterion, a project that would result in a regressive distribution of environmental benefits is less likely to pass than a project that would primarily benefit high-income groups. This is because the sum of WTPs would decide the social profitability of the project, and rich people are less constrained by income than poor people. The consequences of introducing weights thus seem crucial to study in cases where $\varepsilon_w < 1$, cf. Kanninen and Kriström (1992).

3. The Income Elasticity of Willingness to Pay for Environmental Services in Sweden

To the knowledge of the authors, the presently most complete survey of Swedish studies valuing environmental change is available in Söderqvist (1996a). Most studies have used the contingent valuation approach, and the survey summarized about 40 CVM studies dealing with various environmental services. These studies constitute the population in our analysis of income elasticities of WTP. Some of the studies could however not be used for an estimation of ε_w because of at least one of the following obstacles: (1) a valuation function was not estimated; (2) income was not included as an explanatory variable in the valuation function; (3) there was not sufficient statistical information about the income variable or its covariates. Contacts with authors could to some extent solve these problems, but not completely.

Table I lists all studies included in the survey by Söderqvist (1996a) that estimated a valuation function with income as an explanatory variable; 24 estimated functions in total (i–xxiv). As a comparison, the table also includes four later and quite ambitious Swedish CVM studies providing five additional estimated valuation functions (I–V). The table reports the type of environmental service subject to valuation, the number of observations obtained through the CVM survey and the type of valuation function estimated. Most of the studies have used a simple OE question for eliciting WTP, and then simple linear or semilog regression models have been estimated. Tobit models have been used in some studies in order to take a large number of zero WTP responses into account. Other studies have employed DC WTP questions and primarily probit or logit models for studying the relationship between the answers to the DC questions and explanatory variables. As shown in Table I, some studies reported more than one estimated valuation function. Given that the studies did not provide any reason to prefer one or some of the estimated functions on the basis of, for example, goodness-of-fit, all estimated functions are reported in Table I.

The fifth column in Table I reports the sign of the estimated coefficient of the income variable in the estimated valuation functions, and on what level of significance a null hypothesis that the coefficient is equal to zero can be rejected. The sign is positive in 26 of 29 estimated functions, and the null hypothesis can be rejected at the 10% level of significance (or lower) in 23 of 29 cases. The three cases with negative coefficients include only one case where the coefficient is significantly different from zero. The standard result in the estimated valuation functions is thus a positive and significant coefficient estimate of the income variable.

The last column of the table reports the estimated income elasticity of WTP. In cases when the computations of ε_w required values of WTP, income and other explanatory variables, mean values of these variables were used. Due to the third type of obstacles mentioned above, i.e. lack of statistical description of the variables, ε_w could not be computed from all estimated valuation functions. Note also that the valuation functions estimated from DC question data imply a slight modification in the computation of ε_w; $\partial W/\partial y$ in Eq. 3 is replaced by $\partial E[WTP]/\partial y$, where $E[.]$ is the expectations operator.

The point estimates of ε_w vary between –0.71 and 2.83. However, only one of 21 estimated elasticities is negative, and only four of 21 are greater than unity. The mean and median values of ε_w are 0.68 and 0.46 respectively; 0.76 and 0.50 if the analysis is restricted to the studies included in the survey by Söderqvist (1996a). Hence, ε_w tends to take values between 0 and 1, and this is a finding consistent with the results reported by Kriström and Riera (1996).[2] The environmental services valued are highly diverse and on the whole difficult to categorize in groups, the exception being a few studies which all have valued reduced marine eutrophication effects. Considerably more attention is devoted to this environmental service in the next section; here it suffices to note that the estimates of ε_w associated with these studies fall within the quite narrow interval [0.24, 0.35].

4. Income and Price Elasticities of Demand for Reduced Marine Eutrophication Effects

As was mentioned in Section 2, CVM market settings only rarely allow choices between different price and quantity combinations. None of the CVM studies in Table I is advanced enough in itself to make an estimation of a demand function possible. Another option is to merge data from several CVM studies which all have considered a similar issue. In the case of environmental services in Sweden, Table I suggests that the only obvious choice for such a merging is the five CVM surveys on reduced marine eutrophication effects that have been carried out during the latter half of the 1990's.

The background to marine eutrophication effects in Sweden is the substantial increase in atmospheric and waterborne nutrient emissions to the sea during the 20th century (Larsson et al. 1985). The eutrophication caused by this inflow of nutrients involves an increased biological production and, consequently, more dead organic matter whose decomposition consumes oxygen (Bernes 1988). In the end, many eutrophication effects are likely to be detrimental to human well-being. More turbid water has been found to discourage people from seaside recreation (Sandström 1996), and fewer regions available for successful cod reproduction imply a reduction of catches of a fish species with a high commercial value.

The five CVM surveys on reduced marine eutrophication effects had the objective of estimating the benefits of reduced eutrophication effects in the Baltic Sea, though the focus of the studies has differed, see Table II for details.[3] Two of the five surveys considered the whole Baltic Sea, with all Swedish adult citizens as the population; two focused on the Stockholm Archipelago, a part of the Baltic Sea, with adult citizens in the Stockholm region as the population; and one considered an even smaller part of the Baltic Sea, the Laholm Bay in SW Sweden, with adult citizens in the Laholm Bay region as the population.

In all the five surveys, respondents were asked to consider their WTP for reduced eutrophication effects. We assume that this reduction is accomplished by a 50% reduction of the nitrogen load to the area in question, see Table II for quantities implied by this assumption. The precise relationship between reductions in load and reductions in eutrophication effects is subject to uncertainty, but model simulations suggest that the final result of a halved nitrogen load would result in a 30–50% reduction of nitrogen concentration levels in the sea (Gren et al. 1997). Such a reduction would imply a return to the concentration levels of the 1950's, i.e. a level consistent with the situation before eutrophication effects became evident. The effects of applying other assumptions concerning the required reduction in the nitrogen load are studied in a sensitivity analysis in Section 4.5.

Table I. Income elasticities of willingness to pay for environmental services in Sweden

Study[a]	Environmental service	Number of observations	Valuation function	Sign and significance of income variable[b]	Income elasticity of WTP
Bostedt (1995)	Right of public access to a forest area	58–60	i–ii: 2 Tobit models	+*** (1) +*** (2)	1.93 (1) 2.02 (2)
Bostedt (1995)	Right of public access to a forest area	43	iii–iv: 2 Tobit models	+** (1) + (2)	2.83 (1) 1.43 (2)
Bostedt and Mattsson (1991)	Use a recreational area	44	v: Simple linear	+*	0.46
Drake (1987, 1994)	Prevent spruce plantation at 50% of all agricultural land	922 (1) 143 (2)	vi–vii: Simple linear	+** (1) +** (2)	0.53 (1) 0.46 (2)
Drake et al. (1991)	Preserve the agricultural landscape	21	viii: Simple linear	+***	0.91
Fredman (1995), Li and Fredman (1994)	Prevent the extinction of the white-backed woodpecker in Sweden	216 (1) 216 (2)	ix–x: Probit (1) Semilog (2)	+*** (1) +** (2)	.. (1) 0.34 (2)
Frykblom (1998)	Reducing eutrophication effects in the Laholm Bay	294	I: Weibull	+***	0.35
Johansson (1990)	Licence for moose hunting next season	77	xi: Simple linear	+**	..
Katz and Sterner (1989)	Install sockets on gasoline pumps	238	xii: Simple linear	–	..
Kriström (1990)	Preserve 11 virgin forest areas in Sweden	454	xiii: Simple linear	+	0.32
Li and Mattsson (1995), Li (1994)	Right to visit and use forest areas "as usual"	389	xiv–xvi: Probit (1) Improved probit (2) Semiparametric (3)	+*** (1) +* (2) +*** (3)	0.20 (1) 0.59 (2) 0.45 (3)

Malmberg (1994)	Halve (1) or abolish (2) the use of pesticides in Swedish agriculture	168 (1) 171 (2)	xvii–xviii: Simple linear	+* (1) +*** (2)	0.32 (1) 0.66 (2)
Mattson and Kriström (1987)	Opportunities to hunt moose	<90	xix–xxi: Simple linear	+ (1) +*** (2) +** (3)	.. (1) .. (2) .. (3)
Silvander (1991)	No deterioration of angling due to eutrophication (1) Nitrate concentration in groundwater below the standard (2)	<95 (1) <601 (2)	xxii–xxiii: Simple linear	–* (1) – (2)	.. (1) –0.71 (2)
Söderqvist (1996b)	Reducing eutrophication effects in the Baltic Sea	311	xxiv: Logit	+*	0.24
Söderqvist and Scharin (2000)	Reducing eutrophication effects in the Stockholm Archipelago	1,552	II: Simple linear	+***	0.27
Svedsäter (1996)	Environmentally friendly car	307 (1) 193 (2)	III–IV: Simple linear	+* (1) +*** (2)	0.28 (1) .. (2)
Vredin (1997)	Preserve today's population of the African elephant	703	V: Simple linear	+**	0.30

[a]Studies in *italics* were not included in the survey by Söderqvist (1996a).
[b]The sign of the estimated coefficient of the income variable in the valuation functions is denoted with + and –. Significance levels are for a rejection of a null hypothesis that the coefficient is equal to zero. Levels are denoted with * (≤10%), ** (≤5%) and *** (≤1%).

Table II. Description of data sets

Scope and year of CVM study	Mean annual WTP per person (SEK)	Type of WTP question	Length of payment period (years)	Number of respondents	Nitrogen reduction per month (tonnes)
Laholm Bay, 1996	750	DC	20	335	213
Stockholm Archipelago, 1998	670	OE	10	1,810	424
Stockholm Archipelago, 1999	612	OE	10	641	424
Baltic Sea, 1995	1,030	OE	20	82	45,642
Baltic Sea, 1995	6,500–7,000	DC	20	319	45,642
Merged data set	··	DC	10	3,187	213–45,642

Sources:
CVM studies: Frykblom (1998), Söderqvist (1996b), Scharin and Söderqvist (2000).
Nitrogen reduction: SCB (1994, Table 5), Gren et al. (1997).

As indicated in Table II, three of the five studies used OE questions for eliciting WTP, whereas the two remaining employed a DC elicitation method. The latter surveys are the least informative ones in the sense that they just give a hint about the size of the respondents' *maximum* WTP. Being the least informative ones in this respect, they determine how the modelling of the demand for reduced eutrophication can be approached. This modelling is the subject of the next subsection. The procedure for creating a merged data set is then described in Section 4.2, an empirical model is specified in Section 4.3, estimation results are presented in Section 4.4, and Section 4.5 is devoted to a sensitivity analysis.

4.1. MODELLING THE DEMAND FOR REDUCED EUTROPHICATION EFFECTS

Two main approaches for estimating the demand for public goods – such as many environmental services, including reduced eutrophication effects – can be discerned from earlier studies. A collective choice approach based on the median-voter theorem has been dominating in the United States since the early 1970's (Sørensen 1995). Pioneering studies developing this approach were Borcherding and Deacon (1972) and Bergstrom and Goodman (1973). It is assumed that political decisions about the level of expenditures on public goods will be identical to the demanded quantity of the median voter. More precisely, the expenditure of any municipality on a certain public good is assumed to be an observation on the demand curve for the consumer characterized by the median income of that municipality. A demand function can then be estimated by matching observed expenditure levels in a sample of municipalities with characteristics of the median voter in each municipality. However, Sørensen (1995) concluded that the applicability of the collective choice approach is limited. The median-voter theorem cannot easily be invoked in political systems where citizens' votes are multidimensional in the sense that their votes concern more than the expenditure on one single public good.

The other main approach is survey-based and was introduced by Bergstrom et al. (1982), who estimated elasticities of demand for local public school services in the U.S. Other applications include Gramlich and Rubinfeld (1982), Husted (1990) and Sørensen (1995). This is the approach that is used in the following. More precisely, the merged data set from the five CVM surveys on reduced eutrophication effects makes it possible to observe respondents' reactions to suggested supplied quantities of nitrogen load reductions. Let a_i denote the reduction quantity suggested to the ith ($i = 1, \ldots, m$) respondent together with a bid b_i, implying a price $p_i = b_i/a_i$. The demanded reduction quantity (z_i) is assumed to depend on the following relationship:

$$ln\ z_i = ln\ D(\bullet) - ln\ e_i = \beta_0 + \beta_1\ ln\ y_i + \beta_2\ ln\ p_i + \Sigma_j \gamma_j\ ln\ s_{ij} - ln\ e_i \quad (4)$$

where $D(\bullet)$ is the demand function, y_i is the ith respondent's income, s_{ij} ($j = 1, \ldots, o$) are other variables that might influence demand, and $ln\ e_i$ is an independently and identically distributed random variable.

While the demanded quantity (z_i) is unobserved, the merged data set gives information on whether a respondent would be willing to pay a given price for a certain suggested reduction quantity or not. There are two possibilities:

1. If $z_i \geq a_i$, the ith respondent would accept to pay the price, and $z_i^* = 1$ is observed.
2. If $z_i < a_i$, the ith respondent would not accept to pay the price, and $z_i^* = 0$ is observed.

Using Eq. 4, these two conditions can be rewritten as:

1′. $z_i^* = 1$ if $ln\ e_i \leq ln\ D(\bullet) - ln\ a_i$
2′. $z_i^* = 0$ if $ln\ e_i > ln\ D(\bullet) - ln\ a_i$

Assume that the error term is normally distributed, so that $ln\ e \sim N(0, \sigma)$. Then $ln\ e/\sigma \sim N(0, 1)$ and the probability that a respondent would accept to pay the price can be written as follows:

$$Prob\{z_i^* = 1\} =$$
$$Prob\{ln\ e_i \leq ln\ D(\bullet) - ln\ a_i\} =$$
$$Prob\{ln\ e_i \leq \beta_0 + \beta_1\ ln\ y_i + \beta_2\ ln\ p_i + \Sigma_j \gamma_j\ ln\ s_{ij} - ln\ a_i\} =$$
$$\Phi[\beta_0/\sigma + (\beta_1/\sigma)\ ln\ y_i + (\beta_2/\sigma)\ ln\ p_i + \Sigma_j(\gamma_j/\sigma)\ ln\ s_{ij} - (1/\sigma)\ ln\ a_i] \quad (5)$$

where $\Phi[\bullet]$ denotes the cumulative standard normal distribution.

The coefficients β_0/σ, β_1/σ, β_2/σ, γ_j/σ and $1/\sigma$ in Eq. 5 can be estimated by a probit analysis, and Eqs. 1, 2 and 4 imply that the estimation results can be used for computing income and price elasticities of demand as:

$$\varepsilon_y = \beta_1 = (\beta_1/\sigma)/(1/\sigma) \tag{6}$$

$$\varepsilon_p = \beta_2 = (\beta_2/\sigma)/(1/\sigma). \tag{7}$$

4.2. CREATING A MERGED DATA SET

OE WTP questions were used in three of the five CVM surveys, and DC questions in the remaining two, see Table II. As was noted above, answers to DC questions are the least informative on maximum WTP. A consistent merged data set thus had to be created by transforming the answers to OE questions into a yes/no format. This was carried out by letting a Monte Carlo process assign a bid to each of the observations belonging to the data sets of the three OE surveys. The bids were randomly selected from the vector of bids used in the DC surveys. A uniform probability distribution was applied, so that every bid had the same probability of being selected. If the WTP amount actually stated by a respondent in the OE surveys was greater than or equal to the bid randomly assigned to this respondent, it was supposed that the respondent would have accepted to pay this amount of money. Consequently, z_i^* takes the value of 1 for this respondent. If the stated WTP amount was less than the randomly assigned bid, z_i^* takes the value of 0.

Another difference between the CVM surveys was the length of the payment period specified in the valuation scenarios, cf. Table II. The respondents to the Stockholm Archipelago surveys were asked to state the maximum monthly amount that they would be willing to pay during a period of 10 years, whereas the other three surveys used a 20-year payment period. Consistency in time horizon in the merged data set was accomplished as follows. Firstly, present values of suggested payments were computed as $PV[b_{it}] = \Sigma_t b_{it}/(1 + \rho)^t$, where $t = 1, 2, \ldots, 120$ in the Stockholm Archipelago surveys, $t = 1, 2, \ldots, 240$ in the other surveys, and ρ is the monthly discount rate. ρ was based on the average risk free interest rate of the period when the survey in question was carried out; the consequences of using other values of ρ are studied in the sensitivity analysis in Section 4.5. Secondly, $PV[b_{it}]$ was assumed to be paid during a 10-year period, which implies a time horizon consistent monthly bid $b_i^* = PV[b_{it}]/120$. Broadly speaking, this means that the total payment of the respondents who agreed to pay during 20 years is instead spread over 10 years.

4.3. SPECIFICATION OF EMPIRICAL MODELS

Available data in the merged data set allow the following two empirical specifications of the demand function in Eq. 4:

$$ln\ z_i = \beta_0 + \beta_1\ ln\ y_i + \beta_2\ ln\ p_i^* - ln\ e_i \tag{8}$$

$$ln\ z_i = \beta_0 + \beta_1\ ln\ y_i + \beta_2\ ln\ p_i^* + \gamma s_i - ln\ e_i \tag{9}$$

where $p_i^* = b_i^*/a_i$.

They correspond to a probit analysis of:

$$Prob\{z_i^* = 1\} = \Phi[\beta_0/\sigma + (\beta_1/\sigma)\ ln\ y_i + (\beta_2/\sigma)\ ln\ p_i^* - (1/\sigma)\ ln\ a_i] \tag{10}$$

$$\begin{aligned} Prob\{z_i^* = 1\} \ = \ & \Phi[\beta_0/\sigma + (\beta_1/\sigma)\ ln\ y_i + (\beta_2/\sigma)\ ln\ p_i^* + \\ & (\gamma/\sigma)s_i - (1/\sigma)\ ln\ a_i] \end{aligned} \tag{11}$$

The procedure of creating a merged data set implies that the observations of the dependent variable in the probit analysis (z_i^*) can be interpreted as respondents' answer to the following question: "Would you be willing to pay b_i^* per month in 10 years for reducing the nitrogen load to the Baltic Sea by a_i tonnes per month, implying a price p_i^* per tonne of nitrogen reduction?" y_i in Eqs. 8–11 is defined as monthly post-tax income per person; see Section 4.5.3 for a sensitivity analysis of different definitions of this variable.

Mean WTP estimates resulting from answers to an OE WTP question often differ from those resulting from answers to a DC question, even though the valued service is identical. The most common divergence seems to be that OE questions result in a lower mean WTP than DC ones (Kriström 1993, 1999; Söderqvist 1996a; Walsh et al. 1989). In order to take this methodological phenomenon into account, a dummy variable, *s*, is introduced in the second specification. *s* takes the value of unity if an observation originates from an answer to an OE question, and zero otherwise. See Table III for a statistical description of all variables.

4.4. ESTIMATION RESULTS

Limdep 8.0 (Greene 2002) was used for carrying out the probit analyses. Coefficient estimates for the two empirical specifications are presented in Tables IV and V, respectively. The coefficient signs correspond to those found by Bergstrom et al. (1982); income and suggested nitrogen load reduction have a positive effect on the probability of accepting a suggested price, whereas price is negatively related to the probability of accepting. As expected, the estimated coefficient of the methodological dummy variable for OE questions turns out to have a negative sign. Tables IV and V also show that a null hypothesis that coefficient estimates are equal to zero can be rejected at a significance level less than 1%. In addition, the results of

Table III. Statistical description of variables

Variable		Min	Max	Mean	Median	Std dev
z^*	Observed demand behaviour = 1 if the respondent accepted the suggested price = 0 otherwise	0	1	0.2	0	0.4
y	Monthly post-tax income per person (SEK)	0	60,000	11,297	11,000	6,258
b^*	Suggested monthly payment ("bid") per month in 10 years (SEK)	16	1,819	294	158	349
s	Methodological dummy variable = 1 if the respondent answered an OE WTP question = 0 otherwise	0	1	0.8	1	0.4
a	Suggested reduced nitrogen load to the Baltic Sea (tonnes per month)	213	45,642	6,091	424	15,007
p^*	Implied price per tonne of reduced nitrogen load (SEK); $p_i^* = b_i^*/a_i$	0.000383	2.32	0.547	0.310	0.641

the χ^2 tests indicate that the estimated models also work satisfactory as a whole. The significance of the methodological dummy variable suggests that it should be included in the analysis, and the results in Table V for the specification of Eq. 11 are referred to as the "base case" in the following.[4]

The elasticities in Tables IV and V are computed from the coefficient estimates, following Eqs. 6 and 7. The point estimate of the income elasticity of demand for the base case is 0.94, indicating that reduced eutrophication effects are a necessity. However, a 95% confidence interval for ε_y ranges from 0.58 to 1.31, which means that the necessity label is not statistically significant. That reduced eutrophication effects are an ordinary and price elastic good seems to be clear; a 95% confidence interval for ε_p around the point estimate of –1.86 is [–2.08, –1.64].

Finally, let us for a moment assume that the special case of Ebert (2002), referred to in Section 2, is valid. The point estimates of ε_y and ε_p then imply that $-(\varepsilon_y/\varepsilon_p) = -(0.94/-1.86) = 0.51$. This suggests that the benefits of reduced eutrophication effects in the Baltic Sea are regressively distributed, and this finding is consistent with the estimates of ε_w for the individual valuation studies on reduced eutrophication effects. As was noted in the end of Section 3, the estimates of ε_w for these studies ranged from 0.24 to 0.35.

Table IV. Estimation results for the specification of Eq. 10 (number of observations: 2,740)

	Estimate	Std error	p value
β_0/σ	–2.55	0.485	< 0.001
β_1/σ	0.234	0.049	< 0.001
β_2/σ	–0.553	0.029	< 0.001
$1/\sigma$	0.233	0.026	< 0.001
Log-likelihood = –1076			
Restricted log-likelihood = –1351			
$\chi^2(3) = 551$, p value of $\chi^2 < 0.001$			
ε_y	1.00	0.243	0.987[a]
ε_p	–2.38	0.186	< 0.001[b]

[a]Wald test of H_0: $\varepsilon_y = 1$.
[b]Wald test of H_0: $\varepsilon_p = -1$.

4.5. SENSITIVITY ANALYSIS

The creation of the merged data set involved a number of assumptions, and the effects on elasticity estimates of changes in the following assumptions are studied in turn below: (1) discount rate, (2) amount of nitrogen load reduction, and (3) definition of the income variable.

4.5.1. *Discount rate*

As mentioned in Section 4.2, the merging of data sets involved the transformation of 20-year payment periods to 10-year periods, given a positive discount rate, i.e. $\rho > 0$. However, it cannot be taken for granted that the respondents employed a discounting procedure when they answered the WTP questions. If they did not make any discounting of future payments, so that $\rho = 0$, respondents who accepted a payment per month in 20 years were simply willing to pay twice as much as those who accepted the same amount per month in 10 years. Another possibility is that respondents did not think about the time horizon at all, but just on the WTP amount per month *per se*. If so, a 10-year or 20-year perspective would not make any difference and the responses based on a 20-year payment period should not be transformed to a 10-year period. While such a neglect of the time horizon is quite unlikely, it is included here in order to illustrate the degree of sensitivity of the elasticity estimates for extreme assumptions.

As shown by Table VI, the absolute values of the elasticities are greatest when $\rho = 0$ and smallest when $\rho > 0$ (the base case). However, the differences are small and do not change any main conclusions about the elasticities. While the point estimate

Table V. Estimation results for the specification of Eq. 11 ("base case") (number of observations: 2,740)

	Estimate	Std error	p value
β_0/σ	–2.16	0.495	< 0.001
β_1/σ	0.290	0.051	< 0.001
β_2/σ	–0.570	0.030	< 0.001
γ/σ	–0.619	0.076	< 0.001
$1/\sigma$	0.307	0.028	< 0.001
Log-likelihood = –1043			
Restricted log-likelihood = –1351			
$\chi^2(4) = 616$, p value of $\chi^2 < 0.001$			
ε_y	0.945	0.185	0.766[a]
ε_p	–1.86	0.112	< 0.001[b]

[a]Wald test of H_0: $\varepsilon_y = 1$.
[b]Wald test of H_0: $\varepsilon_p = -1$.

Table VI. Sensitivity analysis: elasticity estimates resulting from different assumptions about discounting (standard errors in parenthesis)

	$\rho > 0$ (base case)	$\rho = 0$	Respondents do not care about the time horizon
ε_y	0.94 (0.185)	1.06 (0.216)	0.97 (0.188)
ε_p	–1.86 (0.112)	–2.08 (0.144)	–1.89 (0.116)

of the income elasticity becomes greater than unity when the discount rate is zero, a 95% confidence interval ranges from 0.64 to 1.48.

4.5.2. *Nitrogen load reductions*

There were three different suggested nitrogen load reductions in the base case; 213, 424 and 45,642 tonnes per month, all corresponding to 50% reductions of the present loads. While there are reasons to believe that a 50% reduction is consistent with the valuation scenarios in the CVM studies, this is not known with certainty. As mentioned above, there are indications that a halving of today's amounts would result in marine nitrogen concentrations corresponding to those measured in the 1950's. While it is true that eutrophication effects were not evident at that time, we cannot make certain predictions of how marine ecosystems would react to a reduced nitrogen load. There might be nonlinearities implying that a former equi-

Table VII. Sensitivity analysis: elasticity estimates resulting from different assumptions about the amount of nitrogen load reduction (standard errors in parenthesis)

Area for which the amount of reduction is changed in comparison with the base case		–50%	–25%	The base case	+25%	+50%
Baltic Sea	ε_y	1.08	0.99	0.94	0.91	0.89
		(0.223)	(0.198)	(0.185)	(0.177)	(0.172)
	ε_p	–2.13	–1.95	–1.86	–1.79	–1.75
		(0.171)	(0.131)	(0.112)	(0.101)	(0.0925)
Stockholm Archipelago	ε_y	0.94	0.94	0.94	0.94	0.94
		(0.181)	(0.184)	(0.185)	(0.186)	(0.187)
	ε_p	–1.83	–1.84	–1.86	–1.86	–1.87
		(0.104)	(0.109)	(0.112)	(0.115)	(0.116)
Laholm Bay	ε_y	0.86	0.91	0.94	0.98	1.01
		(0.166)	(0.176)	(0.185)	(0.193)	(0.201)
	ε_p	–1.70	–1.78	–1.86	–1.92	–1.98
		(0.0862)	(0.100)	(0.112)	(0.124)	(0.135)

librium cannot be reached without disproportionally great efforts today, cf. Mäler (2000). The effects on elasticity estimates of substantial changes in the suggested nitrogen load reductions have therefore been studied.

Proportional and simultaneous changes in all three reduction amounts do not change the elasticity estimates. Table VII shows the results from ±25% and ±50% changes in one reduction amount at a time, given no change in the other amounts. The elasticity estimates turn out to be quite robust. As before, point estimates of ε_y tend to be below unity, but confidence intervals around these estimates include values exceeding unity. If we allow a ±50% change, 95% confidence intervals range from 0.53 to 1.52. For the less extreme ±25% change, the intervals are narrowed to [0.56, 1.38]. The corresponding 95% confidence intervals for the price elasticity are [–2.47, –1.53] for the ±50% change, and [–2.21, –1.58] for the ±25% change.

4.5.3. *The income variable*

The definition of income showed some variation in the five data sets. In the merged data set, income was defined as monthly post-tax income per person, cf. Table III. Data on household income were however collected in the Laholm Bay survey and

Table VIII. Sensitivity analysis: elasticity estimates resulting from different assumptions about the definition of the income variable (standard errors in parenthesis)

	Assumed adjustment factor for converting household income to personal income				
	1	1.5	1.8	2 (base case)	2.5
ε_y	0.83 (0.187)	0.94 (0.193)	0.95 (0.189)	0.94 (0.185)	0.90 (0.173)
ε_p	–1.99 (0.136)	–1.92 (0.122)	–1.88 (0.116)	–1.86 (0.112)	–1.81 (0.106)

in one of the Stockholm Archipelago surveys. In order to convert this to income per person, household income were divided by an adjustment factor of two if the respondent were living together with some other adult; this information was available from the surveys. This is however quite a strong assumption since two persons living together only in special cases have identical incomes. Some other adjustment factors (1, 1.5, 1.8 and 2.5) were tried in order to study the effects on elasticity estimates, see Table VIII. An adjustment factor of unity represents the extreme case where household income is equal to income per person.

None of the adjustment factors results in a point estimate of ε_y being greater than unity. An adjustment factor of 1 results in the lowest point estimate (ε_y = 0.83), but again, a 95% confidence interval includes values greater than unity. Note also that the survey data indicate that such a low factor is highly unrealistic. The price elasticity estimates indicate a reduced elasticity when the adjustment factor is increased, but qualitative conclusions do not change; $\varepsilon_p < -1$ even in the case when the adjustment factor is 2.5.

5. Conclusions

That income tends to influence willingness to pay positively and significantly in the case of environmental services in Sweden is a basic finding from the analysis in Section 3. Consistent with the findings of Kriström and Riera (1996), the analysis also resulted in estimates of the income elasticity of WTP for a representative individual that tend to be greater than zero, but less than unity. Hence, as was noted in Section 2, this indicates that environmental improvements are in most cases relatively more beneficial to low-income groups. In a cost-benefit analysis of a project suggesting environmental improvements, distributional concerns are thus likely to call for an introduction of weights or at least a sensitivity analysis of how weighting would change decisions about the project's social profitability, cf. Kanninen and Kriström (1992).

The income elasticities of willingness to pay and demand are two separate entities which should not be confused with one another. In general, computing the income elasticity of demand requires an estimated demand function and thus data on people's choices among different combinations of prices and quantities. CVM studies are far from always enough advanced to allow such an estimation. This obstacle was overcome in our study by merging data from five different CVM surveys. Taken together, they involved a variety of suggested prices and supplied quantities of one particular environmental service, viz. reduced eutrophication effects in the Baltic Sea. The point estimates of the income elasticity of demand for this environmental service tended to be less than unity, suggesting that reduced eutrophication effects are a necessity. The sensitivity analysis in Section 4.5 showed that this finding is robust as far as the point estimates are concerned. However, it is not a statistically significant result, since confidence intervals included values greater than unity. However, the lower end of the intervals do not fall below about 0.5, which means that it can at least safely be concluded that reduced eutrophication effects are a normal good.

The confidence interval for the base case suggests that a 1% increase (decrease) in income would result in about a 0.6–1.3% increase (decrease) in the demand for reduced eutrophication effects. This indicates that income changes would indeed cause changes in the demand for this particular environmental service, but not any dramatic ones. With reference to the discussion on the shape of the environmental Kuznets curve, this result does not give any room for concluding that we are dealing with a general finding for environmental services. Environmental services are different in character, and people might very well conceive some of them as necessities and others as luxuries. This seems also to be true for other public goods than environmental services; the studies of the demand for public goods referred to in Section 4.1 resulted in income elasticities ranging from 0.2 to 1.3. It deserves to be emphasized that preferences govern whether a service happens to be a necessity or luxury. Preferences are changeable and so are thus classifications in necessities and luxuries; they might, for example, be influenced from a more widespread public knowledge of how environmental services provide support to society, cf. Daily (1997) and Daily et al. (2000).

Turning to the price elasticity of demand, the results clearly suggest that reduced eutrophication effects are an ordinary and price elastic good. According to the confidence interval for the base case, a 1% increase (decrease) in price would result in about a 1.6–2.1% decrease (increase) in the demand for reduced eutrophication effects. This suggests that technological innovations that would make it possible to supply reduced eutrophication effects at a lower cost would have a relatively large impact on the demanded quantity. Sewage treatment and wetland creation are two examples of nutrient abatement measures where technological progress might imply cost reductions.

Finally, the sensitivity analysis carried out in Section 4.5 suggests that the results obtained are robust. They are almost not at all affected by slight changes in the assumptions concerning discounting, nitrogen load reduction and personal income. In fact, quite extreme assumptions are required to change the results that the point estimate of the income elasticity of demand is less than 1 and that the point estimate of the price elasticity of demand is less than –1. In the same time, it should be acknowledged that a broader test of robustness would also take into account other demand model specifications and the effects of including more explanatory variables, such as the prices of other goods. Work that would relax the data availability restrictions we have faced and allow more advanced specifications of demand is left here as a suggestion for future research.

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Notes

1. For convenience, we use the label “environmental services” for all goods and services provided by the environment and the ecological systems, including environmental quality.
2. In a CVM study by Ready et al. (2002), ε_w was on average between 0 and 1. However, ε_w was found to approach unity as income increases. This relationship deserves further investigation for the studies in Table I.
3. Monetary amounts are expressed in Swedish crowns (SEK). In November 2002, SEK 1 $\approx$ US$ 0.11.
4. Assuming a logistic probability distribution for the error term and performing a logit analysis gave similar results.

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