

ANALYSIS

# Costing an international public good: the case of the Baltic Sea

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## Abstract

The paper discusses an economically efficient and internationally equitable provision of a specific public good. The good in question, the Baltic Sea, satisfies a number of needs, but economic values implied may be different in various countries polluting and using the Sea. The level of eutrophication has been identified as a key characteristic of the Baltic Sea relevant for its value. The Chander–Tulkens model of cost-sharing is used to determine a hypothetical allocation of abatement costs across the countries around the sea. The application of the model is based on estimates of national abatement costs as well as national willingness to pay for reduced eutrophication. Estimating willingness to pay in a collection of countries (some of which undergo a transition from centrally planned systems) has been the main challenge of this study. The results were derived from a coordinated series of contingent valuation surveys in Lithuania, Poland and Sweden, combined with benefit transfer studies to cover the rest of the region. Predictions of cost-sharing schemes obtained from the Chander–Tulkens model are confronted with actual data on abatement expenditures and international assistance. Several recommendations on how to optimize the Baltic-wide programmes conclude the paper. © 1999 Elsevier Science B.V. All rights reserved.

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*Abbreviations:* CVM, Contingent Valuation Method; DC, dichotomous-choice question in CVM survey (e.g. would you be willing to pay a specified amount?); HELCOM, Helsinki Commission; JCP, Joint Comprehensive Environmental Action Programme; OE, open-ended format question in CVM survey (e.g. how much would you be willing to pay?); PPP, purchasing power parity.

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## 1. The Baltic Sea as a public good

The purpose of this paper is to discuss an economically efficient and internationally equitable provision of a specific public good. To be considered ‘public’, a good has to meet two conditions: (1) the non-exclusion, and (2) the non-rivalry principles. The former holds when it is not

possible to exclude anyone from using the good once its supply has been provided. The latter holds if using the good by one agent does not preclude using it by another agent. A typical problem with a public good is that agents have incentives not to pay for its supply expecting to benefit from having the good been provided by others. As a result, the good will be undersupplied unless a mechanism is created to let agents reveal their preferences and pay for its provision accordingly.

The Baltic Sea possesses some characteristics of a public good. The benefits derived from the Sea include existence values, the value of services provided by marine ecosystems, as well as direct use values such as recreation and fisheries. All of them depend on the state of the Sea. Simultaneously, countries may derive benefits from the Sea without interfering with interests of their neighbors, at least for some types of benefits (this certainly does not apply to fisheries). Thus one may expect to observe typical ‘free-riding’ behaviour with respect to the provision of these benefits, i.e. with respect to the protection of the Sea.

To overcome this market failure countries have sought institutional arrangements for a concerted action to allocate a sufficient effort towards a Baltic clean-up. In 1974 the Helsinki Convention on the Protection of the Marine Environment in the Baltic Sea Area was signed. In 1992, in Helsinki again, the Baltic countries signed a new, enhanced and more stringent convention which will supersede the previous one.

There are ten signatories to the (original) Convention presently. Nine of them, Finland, Sweden, Denmark, Germany, Poland, Russia, Lithuania, Latvia and Estonia, are coastal countries with immediate stakes in the improved quality of the Sea (the tenth signatory is the European Union). Norway is interested in the state of the Sea since a part of its southern coast is adjacent to Skagerrak where the Baltic Sea waters meet the North Sea. Four other countries, The Czech Republic, Slovakia, Ukraine and Belarus, belong to the Baltic drainage basin partially, but they are not connected to the Sea and thus may not derive tangible benefits from it.

The state of the Baltic Sea is characterized by many variables, which are shaped by both natural processes and economic activities. Eutrophication, i.e. excessive supply of nutrients such as phosphorus and nitrogen—originating mainly from inland activities—has been identified as the principal cause of the environmental degradation of the Baltic Sea and its coast (Sanden and Rahm, 1991; Wulff and Niemi, 1992). The discharges of phosphorus (P) and nitrogen (N) had increased until the 1980s when they stabilized at a level several times higher than in the first half of the century. They are decreasing now, although at a pace which—if sustained—can bring visible improvements but in a very distant future.

A large margin of uncertainty affects estimates of the total loads entering the Sea. In the first compilation (HELCOM, 1987), 528 000 t N and 48 500 t P were adopted as lower bounds for the early 1980s. The most recent total load estimates (HELCOM, 1996, pp. 20–26) are approximately 940 000 t N and 38 000 t P in 1992. Thus nitrogen discharges are still very high (and possibly growing), while those for phosphorus are lower.

“Forward to 1950” was suggested as a target for restoring the Baltic to a satisfactory condition. To this end, “nutrient discharges would have to be reduced to the level they were during the 1930s and 1940s, i.e. around 350 000 tons of nitrogen and 15 000 tons of phosphorus per year. For nitrogen this means a reduction of some 65% (...), based on today’s nutrient loading estimates which are by no means exact” (Stockholm Environment Institute, 1990, p. 22). To achieve such nutrient reductions, a massive abatement effort is required. Even a more modest target adopted by HELCOM in 1987, namely to reduce nutrient loads by 50%, requires serious economic commitments.

The relation between nitrogen and phosphorus—that of complementarity rather than substitutability—in producing eutrophic conditions implies that the time perspective has to be included in the analysis. It is observed that the limiting factor of eutrophication of the Baltic Sea is the load of nitrogen presently (Wulff and Niemi, 1992). Hence, immediate improvements can be achieved only by means of nitrogen reductions. In the long run, however, once further

nitrogen mitigation measures become costly, and when a substantial decrease in the phosphorus load is achieved, the latter will become the limiting factor. This means that, in order to achieve a sustainable improvement of the ecological conditions of the Baltic Sea, the emphasis may switch from one nutrient to another over time (starting with nitrogen, i.e. the current limiting factor).

Another reason to put emphasis on nitrogen rather than phosphorus is that, according to model studies, even large reductions of the latter are not likely to result in ambient improvements sooner than in a 25-year period due to a long residence period. In contrast, it would take only (!) 10 years for nitrogen concentrations to achieve reductions commensurate with the discharge reduction rate (Turner et al., 1995). In other words, it is possible to significantly reduce the eutrophication of the marine ecosystems now by controlling nitrogen rather than phosphorus. In the remainder of the paper we will thus treat the former as a key environmental characteristic of the Baltic Sea.

In our analyses we make a simplifying assumption that an ecosystem responds in a predictable way to a gradual reduction in pressure from human activities. Although ecologists provide many examples where this is not the case, modelling studies of the Baltic Sea suggest that—given the current level of their degradation—the sea ecosystems will respond by moving towards less eutrophic conditions and, eventually, a recovery.

By the very end of the second decade of the Helsinki Convention, in April 1992, its signatories committed themselves to the Joint Comprehensive (Environmental Action) Programme (JCP). The Programme lists priority investment projects (HELCOM, 1993a,b (No. 46)). However, it fails to create an effective mechanism for these projects to be undertaken. Some of them have already been launched, but some clearly need an additional stimulus to be carried out.

It has been recognized that countries need to create a basin-wide compensation mechanism to implement the JCP. Indeed some transfers between the wealthy northwestern and less economically developed southeastern parts of the drainage basin have taken place since 1990. Nevertheless it

is doubtful whether the absolute amount of these transfers and their breakdown by donors and recipients is consistent with any conscious policy to optimize the Baltic environmental recovery.

## 2. The Chander–Tulkens model

This section is based on the conceptual framework and notation introduced in Chander and Tulkens (1992). Their model of cost sharing can be summarized as follows.

There is a set  $N$  of  $n$  countries sharing a common resource. The function  $u_i(x_i, z)$  denotes the  $i$ th country's preferences over the consumption of a single private good  $x_i \geq 0$  and a single ambient characteristic  $z \leq 0$  of the common resource (being thus a public bad). The number

$$\pi_i = \partial u_i / \partial z : \partial u_i / \partial x_i \geq 0$$

is the country's marginal willingness to pay (WTP) for an improvement in environmental quality. The equation

$$f_i(y_i, p_i) = 0$$

defines the country's production function, linking its output  $y_i \geq 0$  of the private good with its discharges  $p_i \geq 0$  of a single factor degrading the common resource, and such that

$$\partial f_i / \partial y_i > 0$$

and

$$\partial f_i / \partial p_i \leq 0.$$

The number

$$\gamma_i = -\partial f_i / \partial p_i : \partial f_i / \partial y_i \geq 0$$

is the country's marginal cost of reducing its discharges (note that  $\gamma_i = -\partial f_i / \partial p_i : \partial f_i / \partial y_i = \partial f_i / \partial(-p_i) : \partial f_i / \partial y_i$ , and a positive  $\partial(-p_i)$  measures indeed a reduction, not an increase in pollution). The countries interact with each other (i) through transferring the private good in the amount of  $T_i$  ( $T_i < 0$  if given away by the  $i$ th country and  $T_i > 0$  if received by it); and (ii) by contributing to the degradation of the common resource:

$$z = -\sum p_i.$$

(here and below the  $\Sigma$  sign denotes summation over the set  $N$ ).

The transfers have to be consistent with the feasibility condition:

$$x_i = y_i + T_i, \quad \text{for } i = 1, \dots, n, \quad \text{with } \Sigma T_i = 0.$$

In a non-cooperative equilibrium of the Nash type (countries maximize their respective functions  $u_i$  subject to their production functions, with no transfers, and taking as given the other countries' discharges) the first-order conditions imply:

$$\pi_i = \gamma_i.$$

By contrast, a Pareto optimum is characterized by Samuelson-type first-order conditions

$$\pi_N = \gamma_i \quad \text{where } \pi_N = \Sigma \pi_i.$$

The Chander–Tulkens model takes a non-cooperative equilibrium as the initial condition of a resource allocation process defined in the following way (' denotes derivative with respect to the time variable which is omitted for brevity; all equations hold for each  $i$ ):

$$\begin{aligned} p'_i &= -(\pi_N - \gamma_i), & \text{'the driving force' of the} \\ & & \text{process: adjustments are} \\ & & \text{made as long as the Pareto} \\ & & \text{optimum conditions are} \\ & & \text{not satisfied;} \\ z' &= -\Sigma p'_i, & \text{change in the quality of the} \\ & & \text{common resource;} \\ y'_i &= \gamma_i p'_i, & \text{production function re-} \\ & & \text{sponse to abatement efforts} \\ & & \text{undertaken,} \\ x'_i &= y'_i + T'_i, & \text{the private good balance.} \end{aligned}$$

Chander and Tulkens have proposed the following form of the incremental transfer function:

$$T'_i = -\gamma_i p'_i + (\pi_i / \pi_N) \cdot \Sigma_j \gamma_j p'_j$$

countries have fully reimbursed their respective marginal costs of abatement ( $-\gamma_i p'_i$ ) but at the same time they give away  $((\pi_i / \pi_N) \cdot \Sigma_j \gamma_j p'_j)$ , a share in the sum of all countries' marginal costs ( $-\Sigma_j \gamma_j p'_j$ ) proportional to their marginal WTP divided into the sum of marginal WTPs of all countries  $(\pi_i / \pi_N)$ .

Under this assumption, the private good balance becomes:

$$x'_i = (\pi_i / \pi_N) \cdot \Sigma_j \gamma_j p'_j$$

This process has been proved to converge to a Pareto optimum (Chander and Tulkens 1992).

A number of issues need to be addressed before this approach can be applied to the Baltic Sea problem (Żylicz, 1993).

### 2.1. Multiple pollutants and their interaction

The model can be generalized to multiple characteristics of the common resource (represented as several different public goods). It can also be generalized to the case of multiple pollutants affecting a single characteristic of one common good, e.g. eutrophication of the Baltic Sea caused mainly by two nutrients: phosphorus and nitrogen.

### 2.2. Stock versus flow pollution

The model assumes that the common good in question is affected by flow pollution, whereas in fact the eutrophication of the Baltic Sea is more adequately described as a stock-type one. One way to deal with the problem (not addressed here) would be to develop a more complex modelling framework. Another one would be to treat the eutrophication problem as if it were a flow-type problem on the grounds that for policy (negotiation) purposes current discharges are more relevant than the accumulated stock of nutrients in the Sea. A flow model seems to be closer to the issues actually debated at political fora. Moreover it makes it easier to specify production functions  $f_i$  which actually depend on flows rather than stocks. (It should be noted, however, that, on the contrary, utility functions  $u_i$  depend on stocks rather than flows.) Reducing stock problems to flow-type ones is not uncommon in analyses of international environmental issues (see, e.g. the discussion of acidification by Germain et al. (1996)).

### 2.3. Numerical estimation of $p_i$ and $\gamma_i$

Abatement actions taken by countries are motivated mainly by domestic environmental concerns (river and lake clean-up). The most challenging problem is thus to define ‘incremental’ costs or projects undertaken for the sake of the Baltic Sea. The problem is even more complicated than that; the clean-up of various parts of the Sea (semi-enclosed bays) can be accomplished at the expense of increasing the eutrophication process elsewhere (in the Baltic proper). Such phenomena also point at the necessity of checking which aspects of the Baltic Sea qualify for being considered a public good (both non-exclusion and non-rivalry are violated in the case of semi-enclosed bays). This adds to the complexity of defining what is incremental from the common-good point of view.

### 2.4. Numerical estimation of $\pi_i$

This question is embedded in the previous one in a sense that we have to define precisely what is the common good before we can estimate its utility. It is our conjecture that many conventional valuation approaches, if applied in the Baltic context, would estimate the value of the sea shore as a local resource rather than a common one.

The approach taken in this paper is the following one. We make a simplifying assumption that the Baltic Sea eutrophication problem is caused by nitrogen emissions. We then review the nitrogen abatement measures  $-p'_i$  and their costs  $\gamma_i$ . We then turn to measuring benefits  $\pi_i$  and  $\pi_N$  through a series of contingent valuation method (CVM) surveys. Finally, we look at international assistance programmes from the point of view of their contribution to the Baltic environmental recovery. The actual international transfers are compared with hypothetical transfers derived from the Chander–Tulkens formula:

$$T'_i = -\gamma_i p'_i + (\pi_i / \pi_N) \cdot \sum_j \gamma_j p'_j$$

## 3. Estimating abatement cost functions

The Chander–Tulkens model assumes that marginal abatement costs,  $\gamma_i$ , are known for all countries concerned. From the construction of the model it follows that  $\gamma_i$ , in fact, are average abatement costs incurred by the  $i$ th country in period for which  $T'_i$  and  $p'_i$  are computed. In our analysis here we assume that 1 year is the length of a period for calculating incremental quantities such as  $T'_i$  and  $p'_i$ . In other words,  $-p'_i$  will denote an annual reduction in nutrient discharges, and  $-\gamma_i p'_i$  is the annual cost of the necessary measures in the  $i$ th country. In this paper, as elsewhere (e.g. Turner et al., 1995), nitrogen discharges are chosen as a proxy for the pollution variable  $p_i$ . This is motivated by the fact that nitrogen is considered the limiting factor for a major part of the Baltic Sea. Besides, nitrogen abatement measures are roughly an order of magnitude more costly than those targeted at phosphorus. Thus from an economic point of view the former is a key factor determining the international cooperation process.

Nitrogen abatement costs in the Baltic Sea Drainage Basin have been reviewed and analyzed quite extensively (see, e.g. Żylicz, 1994; Gren et al., 1995; Krysanova et al., 1995; Turner et al., 1995). Despite that, there are no definite answers as to what numbers can be best substituted for  $-\gamma_i p'_i$ . There are several reasons for this ambiguity.

Nitrogen in the Baltic Sea comes from rivers, surface runoff, and from atmospheric deposition. The waterborne nitrogen originates from within the drainage basin. In contrast, the airborne nitrogen may come virtually from all over Europe: either in the form of nitrogen oxides (mainly from car exhaust gases) or ammonia volatilized from the soil (mainly from agricultural activities). This does not allow us to confine analyses to the drainage basin. We will make, however, another simplifying assumption and disregard the depositions originating from countries other than those surrounding the Sea.

There are three types of nitrogen abatement measures: (1) for sewage treatment the costs range from 1 to 10 \$/kg N reduced; (2) for agricultural

Table 1  
Basin-wide cost-effective nitrogen discharge reductions<sup>a</sup>

Reduction rate (%)	5	10	15	20	25	30	35	40	45	50
Cost (10 <sup>9</sup> \$/year)	<0.1	0.1	0.1	0.2	0.3	0.5	0.8	1.2	1.9	4.1

<sup>a</sup> Source: Gren et al. (1995).

discharges there are options ranging from reduced fertilizer use to cultivation of catch crops, to better manure application (the corresponding costs range from 2 to 30 \$/kg N); (3) atmospheric depositions of nitrogen oxides originate from car exhaust pipes and power plants. Abatement costs—if related to the quantity actually deposited in the Sea rather than emitted from a source—are high. They range from 9 to as much 1300 \$/kg N depending on a type of a source and its distance to the Sea. In addition, waterborne nitrogen can be abated by means of restoring wetlands as nutrient sinks. The cost of abatement—measured as an opportunity cost of the land thus diverted from other uses—is estimated at 1 to 9 \$/kg N (Gren et al., 1995).

Both nutrients (phosphorus and nitrogen) responsible for the eutrophication reveal a high degree of technological complementarity in abatement processes. At low to moderate levels of abatement both nutrients are removed simultaneously, the abatement costs are relatively low, and it does not take much effort to reduce nitrogen discharge if phosphorus is removed and vice versa. The pattern changes at higher levels of abatement. Not only do the overall costs grow, but it costs much more now to remove nitrogen even if phosphorus has been removed. Basin-wide marginal abatement costs for nitrogen grow exponentially hitting \$4 billion per year once the reduction rate approaches 50% (see Table 1). These costs are indicative only, as they are affected by a wide margin of uncertainty.

There is no doubt that a part of any abatement effort is motivated by purely domestic concerns. By reducing nitrogen discharges countries serve their own needs by upgrading their private good production functions, as well as positively affecting their utility functions. This aspect is not explicitly dealt with in the Chander–Tulkens model.

Any abatement effort should be considered net of what would have been undertaken anyway, irrespective of the impact on the Baltic Sea, but there are no convincing methods of a priori separating these two motives, and no studies have been carried out to solve the issue empirically.

We assume further that the costs of nitrogen reductions in transition economies will be similar to those estimated using econometric techniques in Finland, Sweden, Denmark and Germany. By so doing we possibly overestimate these costs, since economies in transition may have some not-yet-harnessed potential for improving efficiency. To illustrate the scope of potential low-cost or no-cost nitrogen reduction options, one may refer to the Polish experience after the collapse of the central planning mechanism. Between 1989 and 1994 the fertilizer consumption fell by almost two-thirds from 195 to 71 kg NPK per hectare (in 1992 it plummeted to 62 kg/ha) without proportional losses in yields of major crops. These fell by approximately 20%, at least a part of which has been attributed to adverse climatic factors. Until 1998 no further adverse effects on crops were observed despite the continued low application of fertilizers. This is an example of an abatement measure whose opportunity cost is minimal if at all positive. It is widely held that examples like this are common in economies in transition.

Thus extrapolating western European abatement cost estimates to former centrally planned economies is likely to result in some overestimation. Nevertheless we follow this route encouraged by the fact that existing (fragmentary) evidence justifies such an extrapolation (Żylicz, 1994; Krysanova et al., 1995). Table 2 summarizes annual costs of reducing the nutrient load to the Baltic Sea by 50%. Approximately 80–90% of this sum represents the cost of nitrogen abatement. The costs were derived from an optimization

model which allocated abatement tasks among the Baltic countries so as to jointly meet the 50% target, at least cost (Gren et al., 1995). Investment costs were annualized (at 7% discount rate) and averaged over the 20-year period corresponding to the JCP time horizon.

#### 4. Measuring benefits from reduced eutrophication

Estimation of benefits that may be attributed to pollution abatement in the region seems to be an even more troublesome task than measuring the costs of pollution abatement. The CVM is considered to be the only economic method of natural resource valuation capable of capturing non-use value components. The methodology of CVM has improved significantly over approximately 30 years of its development. Nevertheless the method itself, its validity and reliability, are still a source of controversy. Moreover, as evident from the recent discussion in *Ecological Economics* on the global environment valuation exercise (Costanza et al., 1998), there are still unresolved problems even more fundamental than just valuation techniques.

According to Whitehead et al. (1995) *technical reliability* of WTP estimations is defined in terms of the extent to which the WTP measure is implied by nonrandom sources and the stability of the measure over time. *Validity* is the extent that the measured value corresponds to the theoretical

definition of value. Different biases may affect both validity and reliability of the WTP estimates. Such biases were tested in some experiments: see, e.g. Hanemann (1994) for well-known experiments with strawberries; Cummings and Harrison (1994) for a test on the strategic bias; Niewijk (1994) for an embedding bias. It is difficult to draw one conclusion based on these experiments—some studies tend to praise CVM while others deny its reliability.

In spite of the controversies regarding biases, the values produced in CVM experiments have been accepted by many researchers as well as by some policy-makers. One of the most recent handbooks devoted to environmental economics states: “The WTP (...) measures of economic value can be used as restrictions to guide policy or can be included, with caution, in the bottom-line cost–benefit analysis used to support public policy” (Hanley et al., 1997).

In the US, CVM has been adopted for practical use in several instances. The Department of Interior accepted CVM as the best available procedure for assessing monetary damages to resources. This decision was challenged by the State of Ohio in a Federal District Court in 1986, but the court affirmed it (Cummings and Harrison, 1994).

In 1994, National Oceanic and Atmospheric Administration (NOAA) recommended using CVM in oil spills damage assessments. NOAA’s decision has been taken after a panel of scientists had analyzed the usefulness and reliability of CVM in evaluating non-use values. Also EPA has begun to use CVM studies in cost–benefit evaluations of its regulations (Niewijk, 1994).

The NOAA panel report (Arrow et al., 1993) sets guidelines for researchers who undertake CVM studies. Compliance with its requirements is likely to alleviate biases, leading to higher reliability. It should be noted here that the CVM surveys described in the latter part of the paper conform to the rules set in the NOAA panel report.

Three Baltic countries, Poland, Lithuania and Sweden, have been the subjects of empirical studies aimed at eliciting aggregate national WTP for improvement of the quality of the Baltic Sea. The researchers’ additional task then was to extrapolate the results of empirical valuation from these

Table 2  
Annual costs of reducing the nutrient load to the Baltic Sea by 50%<sup>a</sup>

	Reduction (%)	Cost ( $-\gamma p_i'$ ) (10 <sup>6</sup> \$)
Finland	52	380
Sweden	42	710
Denmark	51	390
Germany	39	530
Poland	63	1280
Lithuania	55	330
Latvia	56	240
Estonia	55	200
Russia	44	80
Total	50	4140

<sup>a</sup> Source: Gren et al. (1995).

three countries to the whole Baltic region using a benefit transfer approach. According to Bergland et al. (1995) there are generally three types of benefit transfer approaches: (1) transferring mean unit values; (2) transferring adjusted unit values; and (3) transferring demand functions.

Often benefit transfer methods are not a reliable tool. It is not the purpose of this section to test reliability and validity of benefit transfers, although the empirical data collected provide quite a good starting point for a future work on this subject. What we try to achieve here is to compile empirical CVM results in such a way that both methodological discrepancies and country-specific factors may be accounted for as much as possible.

We will present findings obtained from five empirical CVM studies on country-wide samples: three from Poland, one from Sweden and one from Lithuania. All studies deal with the same problem, i.e. measuring benefits from reduced eutrophication of the Baltic Sea. The three Polish studies vary in methodology—the first, preliminary, survey employs an open-ended (OE) WTP question, while the other two studies use a dichotomous choice (DC) type of question. The last of the Polish studies was carried out as a mail survey, while the first two were carried out as face-to-face interviews. Yet another CVM study was carried out at the Polish Baltic Sea coast (see Żylicz et al. (1995) for the results). However, being aware of a bias resulting from a specific type of the sample (beach users), we will skip this one in our analysis.

#### 4.1. *Analysis of empirical results*

The two surveys presented in this paper are fully comparable. The Lithuanian survey carried out on 10–18 October 1994 was based on the earlier Polish pilot study implemented on 21–28 July of the same year. Both surveys were carried out by professional polling agencies of the respective countries on representative, country-wide samples. The Polish sample consisted of 1166 respondents and the Lithuanian one of 1002.)

*The Polish pilot survey* was the first in the series of CVM experiments. It was designed as a preparatory work before other surveys. The main

purpose of the pilot study was to check the adequacy of the valuation scenario and questions, as well as to estimate the range of initial bids to be used in the next surveys. The questionnaire was short and consisted of a scenario and three questions.

The question about support for a ‘Baltic tax’ was answered positively by 60% of respondents and negatively by 39.2%. A mere 0.9% of the respondents stated that it was difficult to say if they would or would not support the tax (ten cases). Those who gave a positive answer to the first question or said they did not know were subsequently asked about their opinion on an appropriate value of the Baltic tax (an open-ended question). Those who would not support the introduction of such a tax were asked about the reasons of refusal.

How to estimate mean WTP in situations where a significant number of the respondents do not answer the valuation question? Different authors take different approaches here: some (e.g. Langford and Bateman, 1993) tend to focus on positive respondents only while others (e.g. Gren et al., 1995) assume zero value for non-respondents.

An interesting approach was proposed by Bergland et al. (1995). Respondents who rejected the idea of payment are divided into two groups: ‘legitimate zero bidders’, who did not value the resource in question or could not afford to pay, and ‘protest bidders’, who viewed the idea of payments as not being their responsibility. We have followed this approach by making a distinction between protest and zero bidders according to different reasons for payment refusal.

Fig. 1 shows the breakdown of reasons for rejecting the idea of the Baltic tax. In the group of zero bidders we decided to place those who cannot afford to pay as well as those who think that there are other, more important goals to be financed. Assigning zero values to these categories does not necessarily mean that those respondents do not value the Baltic Sea or its recovery. Their WTP in the situation described in the valuation scenario, however, can be considered zero.

The reasons such as “I pay too much taxes already”, “I do not believe that my contribution may lead to any improvement” and “I am not

### Reasons for refusing to pay in the Polish pilot survey

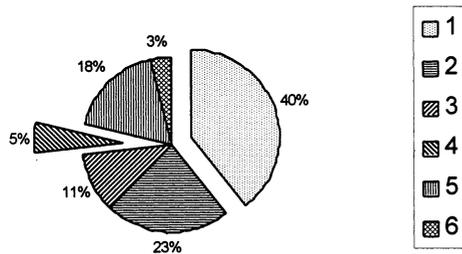


Fig. 1. Zero bidders (exploded) versus protest bidders in the Polish pilot study. (1) I cannot afford to pay but I would do so otherwise; (2) I think I pay too much in taxes already; (3) I do not believe that my contribution will lead to any improvement; (4) I do not think that this is the most important problem in our country; there are other, more important goals; (5) I do not feel responsible for the condition of the Baltic Sea; let the responsible parties pay; (6) other reasons.

responsible for the state of the Baltic Sea, let others, who are responsible, pay”, indicate that respondents have a particular reason for refusal, not connected with the value of payment. This is why we decided to exclude this group of respondents from WTP calculation (Fig. 1).

The mean value of  $WTP_{Pilot}$  including zero bidders simply as zero values and protest bidders as missing equals 31.98 PLN ( $\approx$  \$14 at the exchange rate of 2.32 PLN/\$ prevailing at the time of the survey),  $N = 820$ . Median value is 10 PLN ( $\approx$  \$4.3).

Let us compare these results with the results from the *Lithuanian CVM study*, which applied the same scenario and questions. The only difference was that the list of reasons for refusal in the Lithuanian study contains an additional option: “I am not able to value clean Baltic Sea” (for the full menu of other options, see the footnote to the Fig. 2).

The results of the Lithuanian survey are as follows. A total of 44.2% of the sample positively answered the question about support for the Baltic tax, while 41.6% of respondents said that they would not support the proposed action. A relatively high percentage of the sample, compared to the Polish study, did not know if they would or would not support a tax levied on all Lithuanians for the sake of protection of the

Baltic Sea (14.3%). Those who answered the first question positively or said that they did not know were asked about their WTP. As before, in the breakdown of reasons for refusing to pay we make the distinction between zero and protest bidders.

As can be seen from Fig. 2, the option “I cannot afford to pay” has been chosen less frequently relative to the Polish study. The mean and median were calculated as  $WTP_{Lithuania} = 27.92$  LIT  $\approx$  \$7 (at exchange rate 4 LIT/\$),  $M_{Lithuania} = 10$  LIT  $\approx$  \$2.5;  $N = 697$ . Regression models revealed no significant factors except for household income.

Except for the Polish pilot and Lithuanian experiments, all other Baltic CVM studies employ a DC-type of WTP question, which is preferred by many authors as more natural and resembling situation at real markets. Below the results of two other Polish CVM studies are summarized for comparison and discussion of the most reliable, approximate value of WTP for the Baltic Sea recovery.

The *main Polish study* was carried out on 25–29 November 1994 on a representative all-Poland sample of 1162 respondents, as a part of a larger sociological survey. Valuation scenario gave a short description of eutrophication effects ob-

### Reasons for refusing to pay in the Lithuanian survey

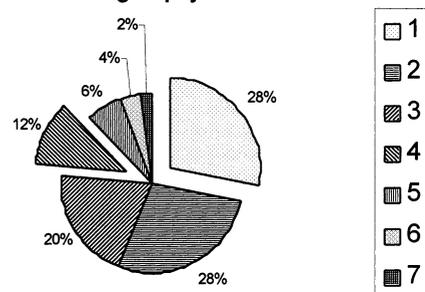


Fig. 2. Zero bidders (exploded) versus protest bidders in the Lithuanian study. (1) I cannot afford to pay but I would do so otherwise; (2) I think I pay too much in taxes already; (3) I do not believe that my contribution will lead to any improvement; (4) I do not think that this is the most important problem in our country; there are other, more important goals; (5) I do not feel responsible for the condition of the Baltic Sea; let the responsible parties pay; (6) other reasons; (7) I am not able to value clean Baltic Sea.

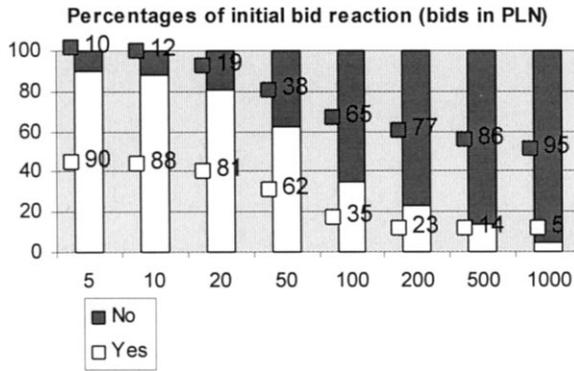


Fig. 3. Percentages of 'yes' and 'no' answers to the initial bid in the main Polish study.

served in the Baltic Sea and mentioned a possibility of carrying out an international clean-up action over the next 10 years. Such an action would call for collecting financial resources in each Baltic country in the form of an earmarked tax. The respondents were then asked if they would support such an action. A total of 62.5% of respondents said "yes", and 29.8% said "no"; 6.7% said they did not know.

A DC question on acceptance/rejection of an initial bid followed. Initial bids and the ratio of acceptance/rejection grouped according to initial bids that were randomly distributed across the sample are displayed in Fig. 3. Numbers displayed in bars represent percentage of yes/no answers in groups of respondents that were given a particular bid.

As expected, the percentage of acceptance falls as initial bids rise. For estimation of mean and median values of WTP, a linear logit model was used. The model with one explanatory variable—an initial bid value—has the following form:

$$\ln(\Pi_{\text{No}}/\Pi_{\text{Yes}}) = a + b \cdot \text{BID}$$

where  $\Pi_{\text{No}}$  and  $\Pi_{\text{Yes}}$  represent the probability of refusal and acceptance of a given initial bid (BID). The bid level revealed to be highly significant

$$\ln(\Pi_{\text{No}}/\Pi_{\text{Yes}}) = -1.1345 + 0.0067 \text{ BID}$$

(see Appendix A for statistical details of the equation).

For DC responses the mean WTP is given by the area under the cumulative probability (bid) function. This area may be truncated at zero and/or the value of the maximum initial bid. Here we have used a simple, not-truncated model. By allowing negative amounts we admit that we cannot exclude that someone who refused to pay anything would rather withdraw the funds that are already being spent by the economy on pollution abatement in the Baltic Sea. By not truncating the model at the maximum bid value, we take into account that the value accepted by a given respondent is not the maximum value this respondent might have accepted. For further discussion of this problem, see Langford and Bateman (1993).

In the linear logistic model integrated over the entire set of real numbers the estimate of mean WTP is the same as the median, since the density function is symmetrical about its mean value. By substituting the definition of a median ( $\Pi_{\text{No}} = \Pi_{\text{Yes}} = 0.5$ ) into the linear logit model we can calculate the median as  $-a/b$ . Thus in this special case also the arithmetic mean,

$$\text{WTP}_0 = -a/b$$

Based on the model coefficients we got the average value 169.33 PLN ( $\approx$  \$73) for positive bidders (713). According to the same criteria as before, 224 respondents were classified as zero bidders, and 225 as protest bidders. In order to modify the mean/median value by including zero bidders we have multiplied WTP achieved from the model by the factor 713/937 (the ratio of positive bidders to positive and zero bidders). Hence  $\text{WTP}_{\text{Main}} = 129$  PLN ( $\approx$  \$55.5).

Yet another CVM study was carried out in Poland in May/April 1995. This time *mail surveys* were used to compare the results with Swedish studies which employed that method. Unfortunately, the method is characterized by a relatively low response rate in Poland. Moreover, overestimation may occur resulting from the fact that those who respond are usually more interested in the subject.

In Poland the questionnaires with a Baltic valuation scenario were sent to 600 randomly chosen addresses; 304 have been filled out and returned,

Table 3

CVM studies results: mean WTP values in 1995 US\$ (including zero bidders and excluding protest bidders)

Country	Lithuania		Poland		Sweden
Type of empirical study	Pilot (OE)	Pilot (OE)	Main (DC)	Mail (DC)	Mail (DC)
WTP	7	14	56	102	458

giving the response rate of 50.7%. The majority of respondents (54.3%) said that they would be willing to contribute part of their income in the form of a special tax for the sake of protection of the Baltic Sea. A linear logit model was estimated as:

$$\ln(\Pi_{No}/\Pi_{Yes}) = -1.7787 + 0.0037 \text{ BID}$$

(see Appendix A for statistical details of the equation).

From this we obtain mean  $WTP_0 = -a/b = 480.73$  PLN (for positive bidders only). Including zero bidders we obtained the value of  $WTP_{Mail} = 236.32$  PLN ( $\approx \$102$ ).

The *Swedish mail survey* with the same DC type of WTP question was carried out on a 700-respondent sample (response rate 60%). The mean annual WTP reached 5800 SEK when only positive bidders were taken into account (Gren et al., 1995). Using the method of distinguishing zero bidders and protest bidders (based on (Söderqvist, 1998)) we estimated the mean  $WTP_{Sweden}$  at 3439 SEK which is equivalent to \$458 at the exchange rate of 7.5 SEK/\$.

#### 4.2. WTP estimates and benefit transfers

Table 3 summarizes WTP values obtained from various studies. In the subsequent part of the paper these values will be used to estimate the benefits ratios: proportions of each country's marginal WTP to the sum of marginal WTPs ( $\pi_i/\pi_N$ ), and then plugged into the Chander–Tulkens formula.

As we have the results from studies that employed three different methods of estimation, we have used the approach of translating the results from the Lithuanian OE study and Swedish mail DC study into hypothetical DC face-to-face type of study results, using the Polish experience from

all three types of surveys. In other words, the Polish main study results have been used as a common denominator in order to arrive at comparable estimates. Based on Table 3 we assumed that the coefficient for translating an OE study results into a DC face-to-face study results equals 4, and for translating mail survey results equals 0.55. We have therefore assumed that a hypothetical DC face-to-face survey in Lithuania would have yielded  $WTP = \$28$ , and in Sweden  $WTP = \$252$ .

To estimate the WTP for the entire Baltic region we applied the following approach. We have assumed that the three countries that were subject of CVM surveys are representative for three sub-regions: Sweden for Western Europe market economies, Lithuania for the former Soviet Union republics, and Poland (for Poland only).

We assumed that within each of the groups of countries (sub-regions) WTP is proportional to GDP per capita at Purchasing Power Parity. Six extrapolations were thus made. The Swedish WTP was extrapolated to Finland, Denmark and Germany using ratios of the Swedish GDP and those of respective countries. Likewise, the Lithuanian WTP was extrapolated to Latvia, Estonia and Russia using ratios of the Lithuanian GDP and those of respective countries. The GDP figures and their ratios,  $\beta_i$ , to the GDP of the reference country for each group are listed in Table 4. The same table contains WTP per capita estimates based on the WTP attributed to a reference country and modified by the  $\beta_i$  coefficient. Aggregate WTP for countries,  $\pi_i$ , are then calculated by multiplying mean WTP per capita by the number of adult population living in respective countries' Baltic drainage basin (Sweitzer et al., 1995; Rocznik, 1995).

Table 4  
Estimation of mean and total WTP values in countries of the Baltic Region<sup>a</sup>

Country	Nominal GDP per capita (US\$)	GDP per capita at PPP (US\$)	$\beta_i$	Mean WTP per capita (US\$)	Aggregate benefits $\pi_i$ (10 <sup>6</sup> US\$)
Finland	13 954	15 483	0.92	232	872
Sweden	17 777	16 821	1	252	1615
Denmark	21 791	19 306	1.15	289	997
Germany	19 688	18 541	1.10	278	676
Poland	1911	4588	1	56	1460
Lithuania	573	3632	1	28	73
Latvia	765	3058	0.84	24	46
Estonia	956	3823	1.05	29	33
Russia	1147	4970	1.37	38	276
Drainage Basin	6091	7998	–	110	6048

<sup>a</sup> Source of GDP data: OECD data files.

## 5. International cooperation

The calibration of the Chander–Tulkens formula is given in Table 5. According to the model there is no country within the region that achieves balance without a money transfer with nutrient load reductions shown in Table 2. In four cases (Finland, Sweden, Denmark and Russia) countries should contribute to a ‘common pool’ of money (negative  $T'_i$  values), and in five cases (Germany, Poland, Lithuania, Latvia and Estonia) countries should be able to use the ‘common pool’.

The total amount of transfers, i.e.  $\Sigma|T'_i|/2$  (for each dollar transferred there is one paying and one receiving party, but we count the amount

only once), equals \$947M per year which corresponds to 23% of the costs indicated in Table 2 and to 16% of the annual benefits. As one would expect, northwestern countries such as Sweden, Denmark and Finland are net payers owing to the fact that they value the Sea more than implied by their share in abatement expenditures. By the same token, southeastern countries such as Poland, Lithuania, Latvia and Estonia receive in net terms. Quite unexpectedly, two countries, Germany and Russia, end up in ‘wrong’ groups: Russia is a net payer (\$109M) while Germany receives a small compensation (\$67M).

The German–Russian paradox can be explained by taking under closer scrutiny our estimates  $\pi_i$  of these countries’ WTP. Only a very small part of the German population lives in the Baltic drainage basin and was thus considered in the benefits study. For instance, the several-million population of Berlin living just a 2-h drive from the Baltic coast has been excluded from the benefit aggregation procedure. And yet, if asked, probably they would have revealed a substantial WTP for improving the marine environment. It is therefore likely that the German benefits are greatly underestimated in our study.

At the same time, the Russian WTP was estimated by taking into consideration the country’s GDP per capita at purchasing power parity. There is an unusual discrepancy between this estimate (almost \$5000) and the nominal GDP (\$1147). Hence one may expect that the former is

Table 5  
Hypothetical money transfers  $T'_i$  in the Chander–Tulkens model

Country	$\pi_i / \pi_N$	$T'_i$ (10 <sup>6</sup> US\$)
Finland	0.144	–216.88
Sweden	0.267	–395.62
Denmark	0.165	–292.29
Germany	0.112	67.20
Poland	0.241	280.82
Lithuania	0.012	279.97
Latvia	0.008	208.79
Estonia	0.006	177.20
Russia	0.046	–109.19
Total	1.000	0.00

but a very disputable measure of wealth of an average Russian. It is likely that he or she could be inclined to relate the WTP for an international public good such as the Baltic Sea by looking at nominally priced private goods rather than by considering his or her purchasing power in real terms. As a result, the Russian WTP estimated in our study could have been overestimated.

## 6. Discussion and policy conclusions

Our study indicates that there exists an essential asymmetry in the distribution of costs of, and benefits from, the reduced eutrophication of the Baltic Sea. The analysis is a simplified one, non-Baltic countries intersecting with the drainage basin on small areas only (Norway, The Czech Republic, Slovakia, Ukraine and Belarus) were excluded from the study.

Germany and Russia are two large Baltic countries whose territories overlap with the drainage basin only partially. Consequently only small parts of their respective populations were considered in the aggregation of the total WTP. Even though the Baltic Sea is not a popular tourist destination worldwide, it may nonetheless be valued as a public good also outside its drainage basin. No such 'geographically external' values were accounted for in Table 4.

Likewise no abatement expenditures to be borne outside the drainage basin were taken into account in Tables 1 and 2, despite the fact that a significant portion of atmospheric depositions of nitrogen originates externally. Airborne nitrogen is emitted from power plants and car exhaust gases from all over Europe, and its abatement is quite costly. The exclusion of non-drainage basin areas may be particularly problematic in the case of Germany, the region's largest economy.

Another simplification was to consider countries' abatement measures as dedicated to the protection of the Baltic Sea, while at least some of these are undertaken in order to protect inland ecosystems or human health directly. Reducing nitrogen discharges in upstream locations, e.g. in Poland—only 20% of which would reach the Sea anyway—is primarily motivated by purely do-

mestic concerns. Thus the Polish costs estimated in Table 2, impressive as they are, may be overestimated as the country's contribution towards provision of the public good in question.

A large margin of uncertainty affects our benefit estimates. The series of surveys aimed at valuation of the same public good revealed a fair degree of internal consistency and reliability. All surveys carried out were based on the same valuation scenario, i.e. to reduce the eutrophication of the Baltic Sea to a tolerable level defined in terms of reduced algae bloom episodes, eliminated oxygen deficit conditions, and re-opened beaches. All surveys referred to the same payment vehicle, i.e. an internationally coordinated earmarked Baltic tax.

As anticipated by the economic theory, we have found that open-ended questions yielded consistently lower WTP responses. Likewise, we have found that mail surveys yielded higher WTP responses than face-to-face ones. It is not clear whether this can be attributed to a non-response bias, or to the fact that a written scenario may be perceived by respondents as more comprehensive and/or simply may be better understood. We have finally decided to base our conclusions on a face-to-face survey with a DC WTP question for two reasons. First, this type of survey produced 'middle' results. Second, it seemed to provide information sufficient to define the commodity to be valued without leaving the respondent overwhelmed by the message (as in the case of a long questionnaire received by mail).

Our analyses demonstrated that WTP revealed in various countries was correlated with the GDP per capita. However, they also revealed that money wealth alone is not sufficient to predict inter-country differences. A linear model whose parameters were estimated on a combined Polish–Lithuanian sample proved that neither income nor other socioeconomic characteristics of respondents were sufficient to explain the variance of WTP responses. A dummy variable to indicate respondents' home country turned out to be highly significant. Consequently we decided to minimize the need for benefit transfers by choosing three countries rather than one as extrapolation bases. We chose Swedish WTP regression

equations as representative for the other north-western countries, i.e. Finland, Denmark and Germany, and Lithuanian results for the countries of the former Soviet Union, i.e. Latvia, Estonia and Russia.

Sweden emerges as the single largest beneficiary of the recovery programme, with almost 27% share in total benefits. It is followed by Poland (24%) and Denmark (16%). The Baltic republics are located at the bottom of this list enjoying collectively less than 3% of the total benefits. Thus more than two-thirds of all benefits accrue to Sweden, Denmark and Poland, either because of the high average WTP (the first two cases) or because of the large population affected (the third case). It therefore comes as no surprise that these three countries have been most active in promoting coordinated, basin-wide approach to the problem at various Helsinki Commission fora.

Another important observation emerging from our analyses is that the total benefits  $\pi_N$  (\$6048M per year) outweigh the costs  $-\sum_j \gamma_j p_j'$  (\$4140M per year) by a factor of approximately 1.5. Let us also note here that cost estimates, taken from Gren et al. (1995), refer to average levels annualized over a 20-year period. They are higher than costs to be born in the first few years of the JCP, but lower than those to be born towards the end of the Programme.

A rough calibration of the Chander–Tulkens model allowed us to compute hypothetical money transfers  $T_i$  required to motivate countries to participate in the JCP up to the globally efficient level. Only fragmentary evidence of the actual environment-related money transfers is available (Laurson et al., 1995). Since 1989 Poland has attracted more international environmental assistance than other countries—approximately \$40–50M per year. In relative terms, however, this accounted for mere 4–5% of the country's environmental investment expenditures. Interestingly, Denmark turns out to be the largest single European donor with grants pledged totalling \$57.8M over a 7-year period (1990–1996). Sweden pledged \$24.2M, and Finland \$16.9M. Germany considered pledging \$42.3M, but only \$0.1M were actually spent by 1994 (Ochrona,

1995). The Polish case is rather typical for the region except for Estonia. The latter has received substantial Finnish assistance totalling almost a half of its environmental expenditures in the early 1990s.

Sweden—the single most generous international donor (in per capita terms) with a strong interest in the region—committed \$140M per annum (in 1993–95) to projects in Central and Eastern Europe. However, only a minor fraction of these funds was to be spent on the environment which competed with a number of other (political, educational, cultural, medical, and commercial) objectives (Ministry of Finance, 1992).

It is clear from these examples that actual transfers between the Baltic drainage basin countries are at least by one order of magnitude lower than those derived from the Chander–Tulkens model. Nevertheless some signs of emerging cooperation may be traced. First of all, the countries of the region are involved in a number of economic, social and political issues that are far more complicated than the Baltic question. It can thus happen that 'issue linkage' occurs, and other forms of incentives or pressures substitute for direct transfers. In addition, cost estimates that were taken as the basis for further modelling represent an approximate average level required until the year 2010 (the time horizon of the JCP). Given the steepness of the underlying abatement cost curves, the largest effort is yet to be undertaken. So far, countries have not left the relatively flat portions of their marginal cost curves. Hence it is likely that the actual level of transfers in the early 1990s was not very far from the hypothetically efficient one.

The latter argument leads to the conclusion that if, in the next decade, countries are to efficiently allocate their abatement effort they must establish a financial mechanism to carry out transfers on a much larger scale than currently observed. According to the model predictions, an efficient level of transfers corresponds to over 20% of the total region-wide costs. In the mid-1990s there were proposals at the Helsinki Convention fora to establish a Baltic-wide fund to stimulate an improved allocation of abatement effort. Even

though the proposals were very modest (corresponding to mere 5% of abatement costs), they did not gain political support. This casts serious

doubts on whether the signatories of the Helsinki Convention are ready to cost their public good adequately.

## Appendix A

(1) Linear logistic regression coefficients based on the main Polish study

Variable	<i>b</i>	S.E.	Wald	df	Sig	<i>R</i>
BID	0.0067	0.0007	97.1678	1	0.0000	0.308
Constant	−1.1345	0.1106	105.1476	1	0.0000	

− 2 log likelihood, 730.432; goodness of fit, 1689.327.

	$\chi^2$	df	Significance
Model $\chi^2$	268.871	1	0.0000

Percent of correct predictions, 77.84%.

(2) Linear logistic regression coefficients based on the mail Polish study

Variable	<i>b</i>	S.E.	Wald	df	Sig	<i>R</i>
BID	0.0037	0.0007	29.9593	1	0.0000	0.3726
Constant	−1.7787	0.2513	50.0771	1	0.0000	

− 2 log likelihood, 158.315; goodness of fit, 160.993.

	$\chi^2$	df	Significance
Model $\chi^2$	43.118	1	0.0000

Percent of correct predictions, 79.52%

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