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Economic valuation for sustainable development in the Swedish coastal zone*

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Abstract

The Swedish coastal zone is a scene for conflicting interests about various goods and services provided by nature. Open-access conditions and the public nature of many services increase the difficulty in resolving these conflicts. ”Sustainability” is a vague but widely accepted guideline for finding reasonable trade-offs between different interests. The UN view on sustainable development suggests that coastal zone management should aim at a sustainable ecological, economic and social-cultural development. Looking closer at economic sustainability, it is observed that economic analyses about what changes in society imply an economic gain and what changes imply a loss should take into account the economic value of the environment. The methods available for making such economic valuation are briefly reviewed. It is observed that it is often a challenge to apply them, partly because the methods tend to be complex, and partly because they need quite detailed information on how humans are affected by changes in nature. It is also noted that the property rights context matters for the results of a valuation study. This general background is followed by a concise presentation of the design and results of valuation studies carried out in the Swedish research programme Sustainable Coastal Zone Management (SUCOZOMA). One study is about the economic value of a reduced eutrophication in the Stockholm archipelago. The other studies are a travel cost study about the economic value of improved recreational fisheries in the Stockholm archipelago, a replacement cost study on the value of restoring habitats for sea trout, and a choice experiment study on the economic value of improved water quality along the Swedish Westcoast.

1. Introduction

Nature provides humans with a number of different goods and services. In the Swedish coastal zone, such “ecosystem goods and services” include highly tangible natural resources such as fish for commercial or recreational fisheries, or less tangible services such as fish recruitment opportunities provided by marine habitats, or environmental amenities such as recreational opportunities, bathing water quality and attractive areas for housing (cf., e.g., 1-4). In Sweden, like in most other industralized countries with coastal areas, some coastal ecosystem goods and services are subject to increasing scarcity, implying different types of conflicts.

One type of conflict is between those who demand coastal ecosystem goods and services, for example, between residential people in the sometimes remote coastal areas and the growing urban population. The former group wants to make a living and have access to a good communication infrastructure, whereas the latter group is increasingly interested in high-quality recreation facilities as incomes in this group grow. The conflicts that such an increasing demand for coastal services might create are likely to be reinforced by the public nature of many ecosystem services, which implies difficulties for property-right holders, if any, to exclude other people from consuming the services.

Another type of conflict is between those who demand coastal ecosystem goods and services and those who influence their supply. For example, many Swedish coastal waters are negatively affected by eutrophication (5). But emitters of nutrients are to an important extent spatially and temporally separated from the eutrophication effects they cause; the main sources are agriculture and forestry in catchment areas, municipal wastewater treatment plants and atmospheric deposition from, for example, traffic (6, 7). In effect, these emitters use the coastal zone as a sink for their emissions.

These conflicts illustrate that coastal issues are in general not likely to be successfully managed without integrating not only ecological interrelationships, but also the surrounding economic, social and cultural landscape (e.g., 8-12). In this paper, we focus on two aspects of such an integration; how the environmental economics valuation approach can assess what solutions to conflicts are economically reasonable, and how this approach fits into the wider context of sustainable development.

2. Sustainable development and economic valuation

Attempts to resolve conflicts are likely to involve compromises, which imply the need of one or several criteria for normative judgments. Its wide acceptance in political spheres suggests that "sustainable development" is reasonable to use as a general, albeit vague, guideline. That is, solutions to be preferred for the coastal zone should be consistent with sustainable development. There is a wide literature about attempts to define sustainable development, and to make the concept operational, since its introduction into the global policy arena by the Brundtland Commission (13). Perman et al. (14) provide a recent overview. On a very general level, it seems to be widely accepted by now that sustainable development should be viewed as having at least three dimensions: an economic, a social-cultural and an ecological one (15). The only human activities that are consistent with sustainable development are viewed as the ones that are economically and socially-culturally desirable and in the same time ecologically sustainable.

What contributions can science make for finding human activities consistent with sustainable development? One commonly used attempt to make the concept more operational is based on the view that future human well-being is determined by the management of a number of different capital stocks. Suggested types of capital stocks include manufactured, human, social, cultural and natural capital. Ways to define them and the degree of substitutability between them are subject to debate (e.g., 16-20). Each of the stocks gives returns that are crucial for human well-being. Ecosystem goods and services can be viewed as the returns of natural capital.

Framing the dimensions of sustainable development into capital stocks helps emphasizing that they constitute the productive base in society. Our focus here will be on the economic dimension of sustainable development and the environmental economics approach to economic valuation of environmental changes. The purpose of measuring such values is to integrate them in judgements about what development is economically desirable. Referring to the types of capital, one can view this activity as measuring the returns of natural capital in economic terms. This means that environmental aspects are not to be covered by the ecological dimension solely, but also by the economic dimension. However, while these aspects are likely to enter in the ecological dimension in physical, chemical and biological terms, they appear in the economic dimension as their importance for human well-being expressed in economic terms.

More precisely, welfare economics theory suggests that changes in well-being can be measured as economic values as revealed by people's trade-offs between scarce resources. See, for example, Boadway and Bruce (21) and Johansson (22) for theory expositions and Hausman and McPherson (23) for a critical review. As a consequence, environmental change

– manifested in, for example, an increased supply of an ecosystem service – involves an economic value (might also be called benefits) as soon as people are willing to make trade-offs between such a change on one hand, and other resources, such as income or time, on the other hand. These trade-offs are typically measured as people's willingness to pay (WTP) for environmental improvements or for avoiding environmental damage. However, it is in some circumstances more appropriate to measure people's willingness to accept compensation (WTA) for environmental damage or no environmental improvement (e.g., 24).

If information about economic values of environmental change is made available, how can it influence what solutions are in fact chosen? At least three ways are available (25): (1) through inclusion in cost-benefit analyses (CBA), whose purpose is to reach a judgement whether a suggested (or realized) activity is (was) economically gainful to society or not; (2) through adjusting national accounts, which might result in "green GNP" indicators; and (3) through indicating what is proper pricing of ecosystem services in terms of levels of environmental charges and green taxes.

These are ways to make an environmentally adjusted picture of the economic dimension and thus to make it more complete in comparison to the information given by, for example, CBAs that do not take into account the environment. However, while this constitutes an improvement, it does not imply that an omnipotent basis for decisions is created. Because the economic approach to measure changes in well-being is based on its particular ethical and theoretical points of departure, it has in general to be complemented with other, non-economic, information in order to create a multidimensional basis for decisions. Multi-criteria decision analysis is one option available for accomplishing this, but any widely accepted framework does not exist (26, 27).

3. Methods for economic valuation

We begin this exposition by explaining the difference between economic and financial values, and then present valuation methods based on welfare economics theory. They are all about estimating either people's WTP for environmental improvements or avoiding environmental deterioration, or estimating people's WTA for environmental deterioration or not realized environmental improvements.

3.1. Economic values vs. financial values

Economic values are often confused with financial values. Examples of the latter are expenses for trips to fishing sites or incomes in the tourist industry. Financial values are usually not very helpful in a welfare economics setting, which takes as a point of departure that society strives for increased well-being among its citizens, not increased expenditures. Financial values might however be relevant for an analysis of other important types of economic phenomena, such as income and employment opportunities and tax revenues.

The difference between economic and financial values is illustrated by Figure 1. It shows a demand curve for some product that is subject to trade at a market. The demanders of the good are likely to have opinions about what quantity of the product they are willing to buy for a given price. It is usually reasonable to think that demanders would like to buy large quantities of the product when the price is low and small quantities when the price is high, given that other factors affecting the decisions, such as income and prices of other products, do not change. Demanders' behaviour is assumed to be a result of their striving for maximizing their well-being subject to the constraint implied by their limited income. This link between well-being and demand becomes evident if the demand curve is read off from another angle: for a given quantity of the product (q^0), the curve gives information about the highest price (p^0) that the demanders are willing to pay for this quantity. This implies that the shaded area under the curve measures the total willingness to pay for consuming this quantity. However, demanders have to pay the rectangular area A (p^0q^0) in expenditures for consuming q^0, which means that the net economic value (often referred to as the (net) consumer surplus) is equal to the triangular area B. The economic value of consuming q^0 (area B) is thus a different entity than the financial value of consuming q^0. The latter is typically measured as the expenses amounting to area A.

FIGURE 1

3.2. Valuation methods based on welfare economics theory

The fact that demand curves can be of help for estimating the WTP for a product suggests that it is fairly straightforward to assess the economic value associated with goods and services that are subject to trade at a market. However, this is of limited help when dealing with ecosystem services, which in many cases are contributing to human well-being without being traded at any market. To still be able to find out economic values associated with such non-market goods is a major methodological challenge. The methods developed for coping with this problem can be divided into *revealed preferences (RP) methods* and *stated preferences (SP) methods*. Freeman (24), Garrod and Willis (28) and Perman et al. (14) provide useful expositions of these methods.

The RP methods are based on the idea that information about people's (and firms') preferences for environmental improvements can be obtained from their behaviour at markets for goods and services that somehow are linked to the supply of ecosystem services. Four important examples of RP methods are the following:

1. The ecosystem service might be an input in the production of a market good. The *production function method* tries to specify this production relationship and a value of a change in ecosystem input is obtained through a study of how such a change affects the supplied and demanded quantity of the market good.

2. The *hedonic price method* studies the influence of a market good's characteristics on its market price. One example might be the relationship between ambient air quality and housing prices.

3. The *travel cost method* focuses on the costs of travels associated with use of the environment for recreational purposes. For example, the recreational value of angling might be estimated from information on how people travel to various fishing sites and use them. Use might be measured by a catch variable.

4. The *defensive expenditure method* studies the demand for a market good that at least to some extent substitutes for the ecosystem service; for example, the demand for water filters that give some protection against reduced drinking water quality. The *replacement cost method* is similar in nature, but focuses on the costs of programmes providing man-made substitutes for ecosystem services and thus not on individuals' market behaviour. Costly flood protection measures along a river might be an example, if they have to compensate for the loss of moderation of water flows resulting from destruction of wetlands. The fact that market behaviour does not constitute the basis for the replacement cost method implies that it result in valid estimates of economic values only if the following three conditions are fulfilled (24, 29, 30): (1) the human engineered system provide functions that are equivalent in quality

and magnitude to the ecosystem service; (2) the human engineered system is the least cost alternative way of replacing the ecosystem service; and (3) individuals in aggregate would be willing to incur the replacement costs if the ecosystem service was no longer available. Similar conditions would apply for other cost-based approaches to valuation, for example, costs for restoring nature or avoiding environmental damage.

While survey instruments such as mail questionnaires, telephone interviews and face-to-face interviews might be necessary for obtaining data for the use of RP methods, SP methods usually rely completely on the use of such instruments. There are some examples of SP methods using experimental markets including actual trading of a normally unpriced ecosystem service; a well-cited example is the second-hand market for hunting licenses set up by Bishop and Heberlein (31). However, SP methods are in most cases based on hypothetical market behaviour. That is, actual market transactions, including payments, do not take place.

The most used SP method so far is the *contingent valuation method* (CVM). A typical CVM application involves a description of an environmental improvement communicated to a sample of individuals, followed by questions about respondents' WTP for a realization of the improvement. CVM has been widely used in the last two decades (32), but has also been subject to criticism, partly because of its hypothetical nature. Carson et al. (33) present and discuss these controversies. However, the hypothetical nature of CVM implies a possibility to estimate potential economic values associated with non-use of the environment, such as the well-being derived from the mere knowledge of the existence of an environmental resource. This is also true for other SP methods, such as *choice experiments* (CE).

One main difference between CVM and CE is about how information about preferences is inferred. CVM focuses on the valuation of a scenario, describing a particular change in environmental quality, and the results provide information on preferences for the whole scenario. CE on the other hand can provide information on preferences for a certain characteristic or ‘attribute’ of the scenario, as well as for the scenario as a whole. This allows for increased flexibility in the analysis. While CE introduces some additional methodological complexity, the information about preferences for attributes can be a significant advantage compared to CVM. If there is some uncertainty about any of the final attribute levels, choice experiments can be used to determine the values for each possible outcome (28).

The most common approach in a choice experiment is to present individuals with a menu (or choice set) consisting of several alternatives (or ‘profiles’) and ask them to choose which one they prefer. Usually they are asked to perform a sequence of such choices, and each alternative is described by a number of characteristics or attributes. Often one of these attributes is a cost attribute, and the repeated choices reveal the trade-offs that the individual will make between the levels of the attributes. Based on the random utility framework, it is then possible to calculate WTP estimates for various changes in levels of the different attributes, which can be used in a CBA (34).

4. Economic values and property rights

Applications of the valuation methods presented in the preceding section do not always consider the property-rights status of the ecosystem service subject to valuation. This might be a serious weakness in a dynamic context, since the public nature of many ecosystem services might imply that economic values are eroded. For example, an improved recreational quality at the coast might have a considerable economic value in the short run, but the improved quality can result in an increased demand for recreation, implying a tendency towards decreased quality through, for example, congestion and increased use. At least at present, measures aiming at restricting recreational use of the Swedish coast would meet institutional barriers such as the Swedish right of public access and the general public's right to coastal recreational fishing with hand-tackle. In addition, these rights can be utilized free of charge, though a flat fee for recreational fishing has recently been suggested by a governmental committee (35).

A classical example of erosion of economic values is when increased fish stocks cannot be protected from increased harvest due to open access to fishing grounds. Gordon (36) showed that in an open-access situation, new fishermen can be expected to enter and harvest fish until profits are reduced to zero, i.e. the potential extra profit from a scarce resource is completely eroded. See also Eggert (37) and Hanley et al. (38). Swedish commercial fisheries can broadly be characterized as regulated open access fisheries, which means that participants are free to enter, subject to regulations like gear restrictions, area closures, and seasonal restrictions (39). However, seasonal restrictions have so far been kept at moderate levels in Swedish fisheries. Overall, Swedish fisheries like other regulated access fisheries show significant signs of stock sizes below optimal levels, over-capitalization and low profitability (40). The effects on demersal stocks, like cod, which are shared between several countries are even worse. According to ICES, a number of cod stocks are at serious risk of collapse (41). Large scale bottom trawling in the North Sea, Skagerrak and Kattegat is likely to generate negative externalities on coastal stocks, leading to negative impact on coastal fishing and recreational fishing possibilities. Coastal demersal stocks have experienced dramatic decline over the last 20 years. One example is cod, where trawling with a research vessel in the

Brofiord had average catches of 100 kg per hour in the 1970s and only two kg per hour in 1998 (42). The causes of this deterioration are not clear but coastal demersal stocks in Skagerrak and Kattegat seem to be dependent on intransportation of egg and larvae from the North Sea (43). The high fishing pressure from commercial catches in the North Sea may hence have had a negative impact on coastal stocks. A recent study recognises that fishing pressure and not variable environment via recruitment was the pivotal variable explaining dynamics of the cod stocks. In other words, cod stock abundance cannot be maintained even when the environmental conditions are permitting a high recruitment level, if not fishing intensity is reduced substantially (42). Another example is by-catches from bottom trawling. For a demersal species like the blue skate, they have been detrimental. In the 1940s an average trawl hour caught 0.5 blue skates, while the last ten years of research trawling has caught two blue skates. This species seems, at least at present, to be more or less extinct in Swedish waters (44).

5. Case studies

Coastal ecosystem goods and services have been subject to a considerable number of valuation studies (e.g., 45, 46). In the following subsections, we report results from some valuation studies of the Swedish coastal zone.

5.1. The economic value of a reduced eutrophication of the Stockholm Archipelago

The benefits to Swedes from reduced eutrophication effects in the Baltic Sea have been estimated from both regional and international perspectives (47, 48, 6). See Hökby and Söderqvist (49) for a meta-analysis of data from these studies. We will focus on the results of a regional study on the Stockholm Archipelago, which is the most detailed one from a CBA perspective.

The boxes in the upper part of Figure 2 give the background to the Stockholm Archipelago study. If there is a change in nutrient emissions in the catchment area of the archipelago, this influences the nutrient load to the sea. However, the impact on the load is dependent on where the emission reduction takes place. If it takes place in a sewage treatment plant situated on the coast, the reduction in load is equal to the reduction in emissions. If an inland measure is taken, part of the effect is lost, due to nutrient retention processes in the drainage basins. That is, the reduction in load will be less than the reduction in emissions. If the nutrient load is reduced, a decreased nutrient concentration in the coastal water can be expected, which in turn is needed for reducing eutrophication effects. The crucial economic question to be answered is whether the benefits of reduced eutrophication effects are large enough to outweigh the costs associated with the measures that are needed for accomplishing the reduced eutrophication effects.

FIGURE 2

In the Stockholm Archpelago study, this question was approached by specifiying a particular environmental improvement: a one-metre increase in the average water transparency during the summer, see the rightmost box in the lower part of Figure 2. At least in the inner parts of the archipelago, such an improvement would be noticed by recreationists. Historical relationships between nutrient concentration and water transparency in the archipelago suggest that at least a 30 per cent reduction of nitrogen concentration is needed to accomplish the one-metre improvement (50). To conclude what reduction in nitrogen load is required for accomplishing this reduction is tricky for an archipelago area, but an estimated load-concentration function for the Baltic Sea suggests that a minimum of a 40 per cent reduction in nitrogen load is needed (51). This minimum figure corresponds to an annual reduction of the nutrient load to the archipelago amounting to 2,725 tonnes (7). Such a reduction can be realized in many different ways, but the study aimed at identifying the particular combination of measures that would accomplish the 40 per cent reduction in nitrogen load to the lowest costs. The total costs of such a cost-effective combination of measures were estimated to be SEK 57 million per year (7). The measures were increased sewage water treatment and, to a considerably less extent, reduced fertilizer use.

What about the benefits? Recreational benefits were estimated by the travel cost method, where people's demand for recreation in the archipelago was estimated, with water transparency as one of the explanatory variables (52). The presence of other benefits than recreational ones was captured by a CVM study, where people's WTP for a nutrient abatement programme that would give a one-metre increase in water transparency was estimated (14). Data for the benefit studies were collected by two mail questionnaires to a total of 5,500 randomly selected adults living in the counties of Stockholm and Uppsala. Conservative estimates from the studies for the benefits of a one-metre increase in water transparency were SEK 60 million per year (travel cost method) and SEK 500 million per year (CVM).

While the estimated least total costs are about the same as the estimated recreational benefits, they are considerably lower than the broader benefits as estimated by CVM. This suggests that it would be profitable to society to reduce the nutrient load enough to generate the one-metre increase in water transparency.

5.2. The economic value of improved coastal recreational fisheries in Sweden

A brief survey of valuation studies in Karås et al. (53) shows that some aspects of the economic values of Swedish coastal recreational fisheries have been relatively well studied (e.g., 54-61). However, we have not found that they have included any attempts to link benefit estimates to the underlying ecosystem support to fish reproduction. This might be a serious weakness, since it reduces the potential to make use of the benefit estimates in a CBA setting. For example, the social profitability of projects aiming at increasing the fish stock by restoring or protecting spawning and nursery areas can only be assessed if the benefits of the results (increased catch rate due to a bigger fish stock) can be compared to the costs of measures causing these results.

A major problem in accomplishing such a linking between benefits and costs is that there is seldom sufficient quantitative natural scientific knowledge of relationship between habitat quality and quantity, fish recruitment and fish stocks. One main explanation is the fact that many fish species are not completely dependent on a particular habitat. However, relatively stationary species such as perch and pike might be suitable for a linking. For perch in the Baltic Sea, there are preliminary quantitative findings on the relationship between recruitment area size and perch stock (53). A travel cost study on recreational fishing in the Stockholm Archipelago with the aim of not only valuing a bigger fish catch, but also the underlying increased ecosystem support necessary for accomplishing this increase, has therefore been initiated.

The travel cost study requires information on travel behaviour in the archipelago, i.e. sites visited by respondents, the distance travelled, travel time, travel costs, catch rates etc. This information was gathered in questionnaire surveys in 2002 and 2003. The questionnaire was sent by mail to 500 randomly selected members of the Swedish Association for Recreational Anglers (Sveriges Sportfiske- och Fiskevårdsförbund) living in Stockholm County or the adjacent Uppsala County. This was done every third month during 12 months to gain information on potential seasonal differences. The average response rate for all seasons was 58 percent. Another, less detailed, questionnaire was sent to 2,000 randomly selected adults living in Stockholm and Uppsala counties, 52 percent of whom responded. Preliminary results indicate the relationship between the probability that a fishing site is chosen and the catch of fish. This relationship is positive for most species. However, negative signs exist which may be due to few catch observations. Further, there is a clear negative relationship between the probability that a site is chosen and travel cost. The estimated parameters are significant in most cases. Based on these results economic values are now being estimated for improved fishing in the archipelago. These benefit estimates can be compared to the costs for measures that would improve fishing conditions in the archipelago, and thereby the profitability to society of such measures may be assessed.

A different way to approach the linking was carried out in another SUCOZOMA case study. This study followed the replacement cost method for valuing improved habitats for sea trout reproduction in a watercourse in the Stockholm region (62). Pollution and physical change have implied a decreased sea trout production capacity in many watercourses. Sea trout is an important species in recreational fishing, and this has motivated measures such as stocking of reared sea trout and restoration of watercourses by, for example, removing physical barriers against migration, ensuring sufficient water supply, installing better sewage treatment, creating buffer zones between arable land and watercourses, and putting gravel on the bottom of watercourses for making them more suitable for spawning (63).

Kagghamraån in the SW part of Stockholm County is one watercourse where restoration measures have been taken and evaluated by measurements of sea trout density. There were strong indications that density changes were due to the restoration rather than to any other factor because the density was measured before and after the restoration at the places where restoration took place (64). This made it possible for Sundberg (62) to relate the costs of restoration to the effects accomplished: SEK 300,000 (once-for-all amount in 1995 prices) for a result of 68-260 sea trouts returning to the sea per year. Cost items such as time for planning and density measurements and maintenance of restoration measures were not included, but the maintenance necessary is judged to be negligible. These costs can be compared with those of another approach: increasing the sea trout stock in the sea directly by stocking sea trout. The present value of the costs of stocking 260 sea trouts per year were estimated to about SEK 100,000, again excluding cost items such as time for planning and density measurements. This thus seems to be a cheaper way to accomplish a given increase in the sea trout stock in the sea, but such a comparison ignores the fact that stocking of reared sea trout might reduce the genetic diversity of this species. Moreover, watercourse restoration for the sake of the sea trout might result in positive side-effects such as an increased biological diversity in the watercourse fauna and flora (65).

5.3. The economic value of water quality improvements at the Swedish Westcoast

Eggert and Olsson (66) study preferences for improved water quality, using a choice experiment framework. Marine water quality can be characterized in a number of ways and there is a trade-off between the interest and relevance of the attributes on the one hand and the level of complexity for the survey respondents on the other. The failure to secure sustainable commercial fisheries has generated great interest within the European Union. Similarly, attention has recently been focusing on coastal water quality. The issue of securing marine biodiversity and its importance for sustainability has been at the top of the world agenda since the United Nations summit in Rio 1992.

In the study, water quality was represented by; level of fish stock, bathing water quality and biodiversity level. A brief description of the attributes and their levels are shown in Table 1.

TABLE 1

The attribute ‘bathing water quality’ is measured in frequency of failures to meet the standards of, for example, bacteriological contamination for bathing sites along the coasts. The attribute ‘cod stock’ is measured as the catch in kilograms of cod above the minimum size 30 cm per trawling hour with a research vessel (which is used to estimate fish stocks). Here, the attribute ‘biodiversity’ is described merely as ecosystem balance, where the level of today is defined as a medium level. In comparison to the two former attributes, we expect non-use values to be relatively more important for people’s valuation of biodiversity.

The study analyzes preferences assuming different distributions of the taste parameters, but the mean WTP for a change in the attributes from current level to the highest level is shown in Table 2. For an improvement in bathing water quality, resulting in the fraction of bathing sites violating the standards decreasing to 5 percent, the individual WTP is on average SEK 600. Similarly, for an improvement in the cod stock to a level where the catch per trawling hour is 100 kg, the WTP is SEK 1200. For an improvement in the marine biodiversity, to a level than can be called ‘high’, the WTP is SEK 600, while the WTP to avoid deterioration is SEK 1,400.

TABLE 2

Assuming zero WTP from all non-respondents implies that the respondents represent 40 percent of the whole population, which leads to a rough aggregate estimate of SEK 500-600 million for either improving the cod stock or avoiding deterioration of marine biodiversity. The aggregate estimates for improved water quality or improved marine biodiversity are SEK 200-300 million.

6. Discussion

This paper provides some examples of how environmental change in the coastal zone can be economically valued. We note that estimated economic values are dependent on a number of circumstances. Firstly, the method selected for valuation influences the estimates; for example, some methods have the capacity to take potential non-use values into account whereas other methods have not. Secondly, the institutional context matters; for example, economic values might be eroded in open-access situations, and formal (e.g., law) and informal (e.g., norms) institutions in society are likely to determine people’s preferences and thus their willingness to make trade-offs for the sake of ecosystem services. Thirdly, nature’s heterogeneity might make value estimates unique for a specific setting; for example, the relatively low water transparency in the inner parts of the Stockholm Archipelago implies that a 1-metre increase would make a perceptible difference and it is therefore not surprising that it exists a WTP for such a change. However, such a WTP is not likely to be valid for settings with a considerably different baseline water transparency, for example that of the Swedish Westcoast. Fourthly, the economic context is of importance; for example, income is generally a significant determinant of WTP (49).

These circumstances are four reasons for why value estimates cannot easily be transferred from their original setting to other settings, and they also illustrate why it can be misleading to generalize average values of ecosystem services such as those provided by Costanza et al. (1). This is also acknowledged by the literature on benefit transfers (67, 68).

Why this emphasis on the fact that economic values are context-dependent and thus are likely to be difficult to generalize from one setting to another? The most important reason is perhaps that undertaking economic valuation involves a responsibility because its ultimate purpose is to advise decision-making. We have indicated what forms this guidance can take, and the Stockholm Archipelago study illustrated how estimated economic values can be used in a CBA. However, whether policy-makers will be influenced by such advice or not is another issue. While CBAs often are a requirement for developing countries when applying for environmentally related loans, there are few countries in the developed world in which CBA is in practice a generally accepted major decision-making tool (69). In Sweden, only a few government authorities make frequent use of CBAs (70).

This hesitance brings us back to the three dimensions of sustainable development. Even if all three dimensions cannot be covered by economic information such as that provided by environmentally adjusted CBAs, it is worth noting that some of the necessary non-economic information is likely to be produced when carrying out economic valuation (and CBA). For example, economic valuation requires an identification of the ways in which people are positively or negatively affected by environmental change. Moreover, survey-based valuation methods such as CVM and CE typically result in information about people's attitudes about the environmental change in question, and focus group sessions and other kind of preparatory work in valuation studies might result in important qualitative knowledge about existing conflicts. Valuation studies thus often provide important non-economic information alongside of the value estimates, and frameworks for taking care of this information should be developed.

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Figure 1. Financial values and economic values

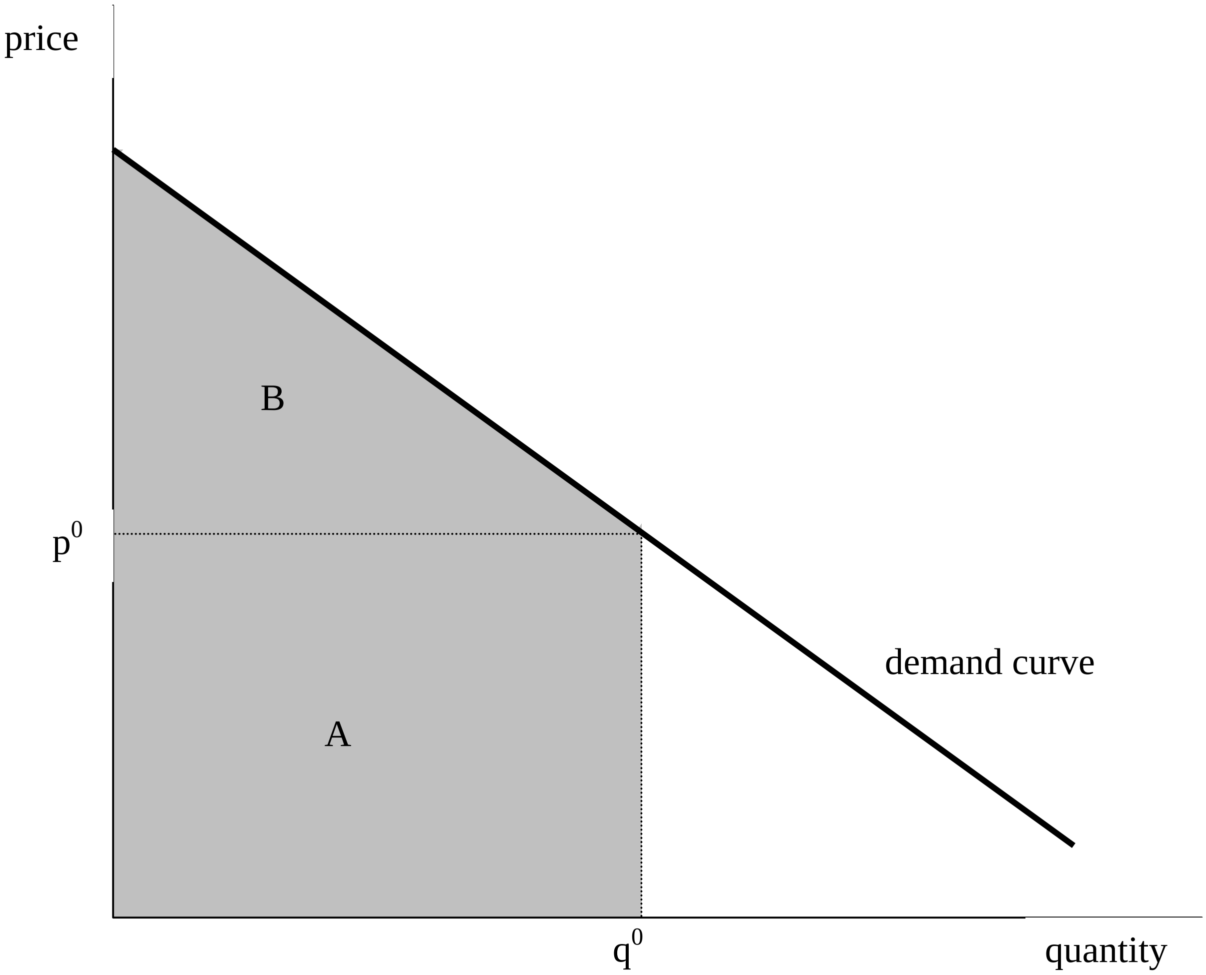

Figure 2. The Stockholm archipelago study (Δ=change in)

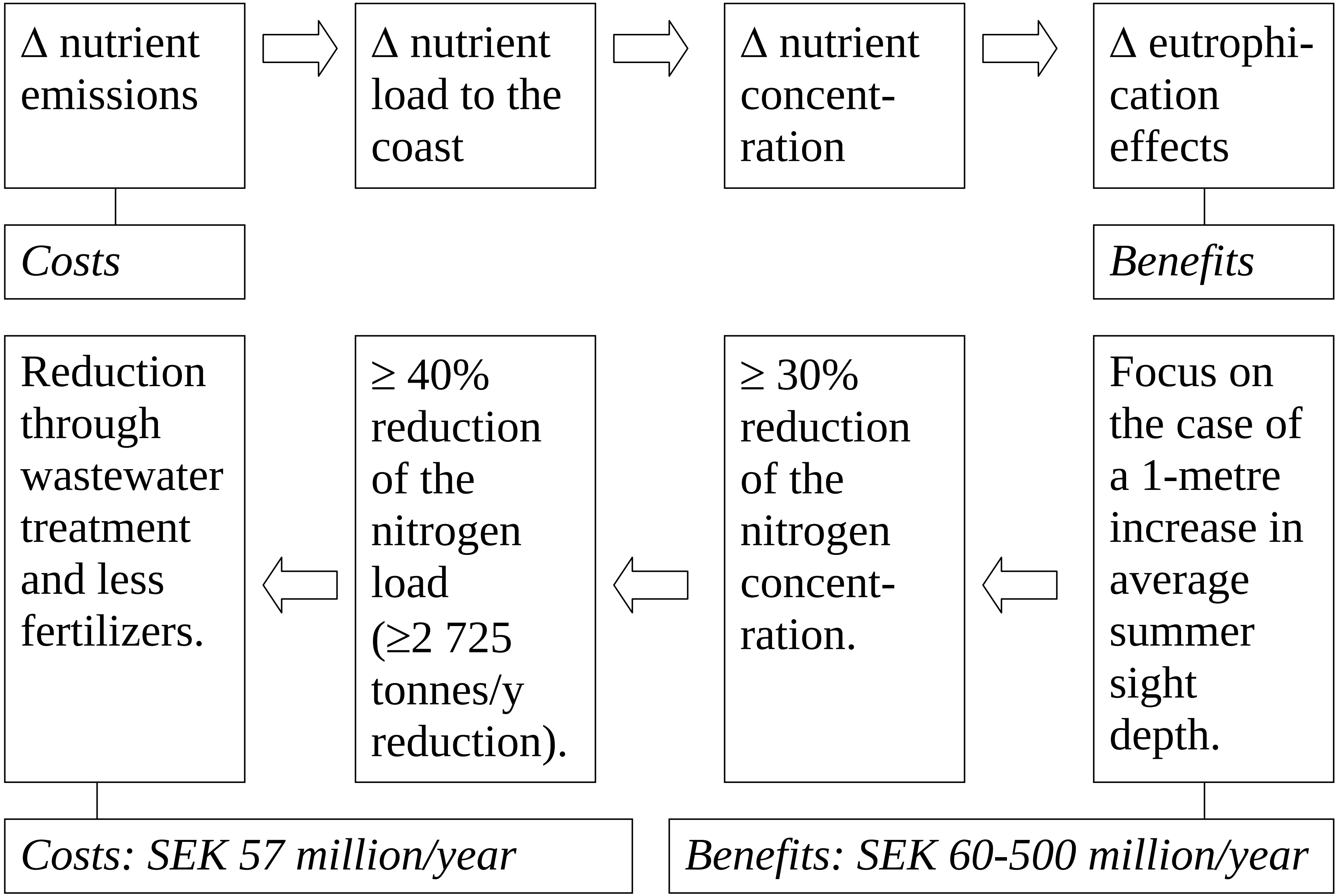

Table 1. Brief description of the attributes and their levels

(bold figures indicate baseline)

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

Table 2. Marginal willingness to pay (SEK) for a change from current level to highest level

	Attribute			
	Water	Cod	High Biodiv	Low Biodiv
MWTP	600	1200	600	1400