ALTERNATIVE DECISION SCHEMES FOR COASTAL WATER MANAGEMENT

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Abstract: The purpose of this study is to investigate the relative advantage of two alternative decentralised decision units for water quality management: jurisdictional and drainage basin. It is assumed that there is a relatively large coastal water recipient is assumed which can be divided into several coastal basins. Further, there are transports of pollutants among coastal basins which constitute spill-over effects among decision regions. There is also a choice between two decision rules: maximisation of net benefits, or minimisation of costs for achieving prespecified targets. Analytical results show that relative performance between the two decentralised decision units depends on asymmetry in pollution benefits, pollutant transports within and among regions, and choice of decision rule. An application to an estuary located approximately 100 km south-west of Stockholm, Sweden, shows that the relative advantage of the jurisdictional (municipality) and drainage basin decision regions depends on choice of decision rule, and – under maximisation of net benefits – the size of benefits of water quality changes.

Key words: coastal water management, maximum net benefits, jurisdiction decision, drainage basin decision.

JEL: D62, H7, Q25, R1

1. Introduction

Today, many coastal areas suffer from damages, such as eutrophication, due to excessive pollutant loads. One challenge for mitigating this problem is provided by the pollutant loads which follow complex pathways from emission sources to coastal water recipients, and subsequently within and between marine water basins. Since these pathways do not coincide with jurisdictional borders, water quality management might be improved by allowing for decision making at a drainage basin level. Traditionally, environmental policy has been determined and implemented at the jurisdictional level. This introduces a risk for inefficiency unless the natural and jurisdictional borders coincide so that the jurisdiction includes all costs and benefits associated with water pollution.

Inconsistencies between natural and jurisdictional borders were recognised already in the 1960s in, for example, France, where drainage basin water management has been in practice for several decades (see e.g. Gustafsson, 2000). Moreover, a major change to water management based on natural borders is at present taking place in the EU. The Water Framework Directive adopted by the European Parliament and the EU Council in 2000 instructs all member states to identify their individual drainage basins and assign them to drainage basin districts, to which appropriate administrative arrangements are to be introduced (EU, 2000). An important integrative feature of the directive is that coastal waters are also to be associated to the districts. However, the directive does not indicate the desirable size of the districts, which might result in differences among the member states. In the case of Sweden, a governmental committee has recently suggested that Sweden is divided into five drainage basin districts with sizes ranging from 26,056 to 162,160 km^2 despite that the number of main drainage basins is considerably greater: 119 (SOU, 2002). While the committee also considered the option of having smaller districts, as suggested by e.g. the Swedish Association of Local Authorities, the committee judged that having relatively large districts would simplify coordination and increase consistency (SOU, 2002).

The purpose of this study is to investigate whether or not the tendency towards letting natural borders be more influential in policy is a good one to society. Or, more precisely, under what conditions is social net welfare higher under drainage basin management than under jurisdictional management? Two decision rules are investigated; maximisation of net benefits and minimisation of costs for pre-specified targets. The analysis is applied to Himmerfjärden, an estuary situated 100 km south-west of Stockholm, Sweden.

The study focuses on management of coastal waters. Typical for coastal water pollution is that, in addition to drainage basins and jurisdictions, pollutant transports between coastal water basins determine water quality. This means that several drainage basins and/or jurisdictions can share a common coastal basin and depend on each other through transports between coastal basins. Studies on management of large-scale water bodies, such as Paulsen and Wernstedt (1995), Gren et al. (1997) and Turner et al. (1999), investigate policies for achieving full co-operative solutions, but they do not consider implications of alternative decentralised decisions.

The environmental economics literature related to location of decisions – environmental federalism – is usually about one type of decentralisation, viz. the jurisdictional one, and its efficiency in comparison with that of centralised regulation (e.g. Oates and Scwab 1988; Fredriksson and Millinet 2002; Levinson 1997; Markusen et al 1995). Jurisdictions are then assumed to compete with each another for polluting firm location under deterministic conditions. We also compare centralised and decentralised decisions, but our analysis differs from the literature on environmental federalism in two respects: two alternative decentralisation schemes are compared, and competition among decentralised decision units is excluded.

Another related class of literature is about efficient provision of an international environmental public good (e.g. Barrett, 1990; Mäler, 1991 and 1993; Elofsson, 2002; Hoel, 1992; Kaitala *et al.,* 1995; Folmer and van Mouche, 2000; Gren, 2001; Folmer and van Mouche, 2002). This paper follows this literature, but instead of one type of players (usually jurisdictional), two Nash solutions for each of the jurisdictional and drainage basin players are compared with the co-operative solution. Further, this is made for full and limited discretion for the players.

Decision units at the centralised and decentralised decision levels are assumed to have full discretion where they decide on optimal emission/abatement level. The analytical results show that efficiency losses at the decentralised level depend upon the symmetry among regions, and their area coverage of marine basins. These results can be compared with those from the literature on regulation of heterogenous pollutant sources where uniform policies are compared with differentiated (e.g. Helfand and House 1995; Brännlund and Gren 1999). The outcomes of these policies depend on symmetry among sources with regard to emission benefits and environmental damage costs, which also influence the relative performance of the two decentralised decisions in this study.

The rest of this paper is organised as follows. A model for maximising net benefits under different regional decision systems is presented in section 2. Section 3 contains the application to Himmerfjärden. The paper ends with a concluding section 4.

2. The model

Two types of decision rules are investigated; maximisation of net benefits and minimisation of costs for centrally given water quality targets. Three types of regional scales are included, the central or cooperative scale, drainage basin and jurisdictional. The regional decision units are assumed to have total discretion over abatement for both decision rules. Separability is further assumed between water quality and other regional issues such as employment and income, so that competition among regions for, for example, firm location is excluded (see e.g. Oates and Schwab 1988; Levinson 1997; Markusen et al 1995; Fredriksson and Millinet 2002; Kunce and Shogren 2002). Although these issues might be of concern, a justification for excluding them is our focus on different regional scales.

The catchment of the marine recipient contains $h=1,...,m$ jurisdictions, and $d=1,...,p$ drainage basin regions. Each emission source is located in one jurisdiction and one drainage basin, with emission reduction l^{dh}. Each jurisdiction thus contains emission reductions corresponding to $\sum_{d=1}^{p} l^{dh}$, and, similarly, each drainage basin includes $\sum_{h=1}^{m} l^{dh}$ total emission reductions. For each emission location, we assume there is a cost function for emission reduction, $C^{dh}(l^{dh})$, which is increasing and convex in l^{dh}. Further, there is a capacity constraint on each reduction possibility, $\bar{l}^{dh}$, such as a maximum reduction of fertilizers or land use changes. In the long run, such capacity constraints may not be applicable, but considering the static model used here there are short run limits to the magnitude of land use changes. For example, assume that one type of measure is to construct wetlands as nutrient sinks. This can be done in a short period on land particularly suitable for this purpose like arable land that once was converted from wetlands. However, to convert other types of land to wetlands may require long time before it can be regarded as a functioning wetland (see e.g. Mitsch and Gosselink, 1988).

Emission from each source is either discharged into a stream, deposited on land, or emitted into the air before it enters the coastal water recipient. During the transport from the emission source to the recipient, the pollutant is subject to transformation, which implies that less of the pollutant finally enters the coastal water of marine basin i, where $i=1,..,s$. This transformation is characterised by stochastic and dynamic processes in the drainage basin, which means that neither the final deposition into the coastal water nor its timing can be predicted with certainty (e.g. Gren et al. 2002). Nevertheless, in order to focus on the role of decentralised decision schemes, these complexities are ignored, and a simple linear relation between emission and deposition into the coastal water is assumed. It is allowed to vary for different drainage basins and denoted by b^{di}, which gives deposition reduction from a certain emission source into the coastal water of a marine basin as $e^{dhi}=b^{di}\, l^{dh}$.

Total pollutant reductions to a marine basin, T^i, can be determined by emission reductions from either all its jurisdictions $T^i=\Sigma_j a^{ji}\Sigma_d e^{dhj}$, or from all drainage basins $T^i=\Sigma_j a^{ji}\Sigma_h\, e^{dhj}$. For both summations, a^{ji} is the transport of nitrogen from basin j to basin i and e^{dhj} is the load impact of emission reduction at a source located in jurisdiction h and drainage basin d with load to marine basin j. It is further assumed that water quality improvements in each basin are determined by its total load reduction as $D^i=D^i(T^i)$, where $D^{i\,\prime}>0$ and $D^{i\,\prime\prime}\leq 0$, and that there exists a value function of water quality improvement for each marine basin, $V^i(D^i)$ and $V^{i\,\prime}>0$ and $V^{i\,\prime\prime}\leq 0$. For simplicity, the regional value function is assumed to constitute a share, v^k where $k=d,h.$ of the total value of damage reduction in a marine basin so that $V^{ik}(D^i)=v^k V^i(D^i)$.

Given these assumptions of emission benefits and water quality improvement functions, three regional decision units are identified: co-operative, jurisdiction and drainage basin. At the co-operative level the sum of all net benefits, π^C, are maximised. The jurisdiction and the drainage basin decision unit are assumed to maximise their own net benefits, π^h, and π^d respectively.

2.1 Maximisation of net benefits

At the co-operative scale all benefits and costs of pollutant emission reductions are included, which gives net benefits as

$$\pi^{C} = \sum_{j=1}^{s} V^{i}(D^{i}(T^{i})) - \sum_{k=1}^{n} C(l^{dh}) \tag{1}$$

s.t $\quad l^{dh} \leq \bar{l}^{dh}$

Solving for the optimal choice of either *h* or *d*, gives

$$\frac{\partial C}{\partial l^{dh}} + \theta^{dh} = \sum_{j=1}^{s} V^{j}_{D^{i}} D^{j}_{T^{i}} a^{ij} b^{di} \tag{2}$$

where subscripts denote partial derivatives and θ^{dh} is the Lagrange multiplier denoting the shadow value of a relatively low cost option. According to (2), the optimal allocation of emission reductions among emission sources occurs where marginal benefit, the right-hand side of (2), equals marginal cost. The former includes benefits for all marine basins from a marginal emission reduction in l^{dh}. At the co-operative scale, the decision-maker allocates emissions reductions towards sources with relatively large impact on water quality, i.e. high marginal benefits, and low marginal costs.

At the jurisdictional or drainage basin scale, a Nash solution is assumed where the decision-makers maximise net benefits given other regions' best response emissions. It is assumed that the decision makers have discretion over abatement only in their own regions, and the net benefit function of the jurisdiction is then written as

$$\pi^{h} = \sum_{i=1}^{s} v^{ih} V^{i}(D^{i}(T^{ih}), D^{i}(T^{-ih*}) - \sum_{d=1}^{p} C^{dh}(l^{dh}) \tag{3}$$

s.t $\quad l^{h} \leq \bar{l}^{h}$

The right hand side of (3) shows all benefits and costs of emissions reductions in marine basins, i, for which $v^{ih} > 0$. Environmental improvements occur from pollutant transport among basins of concern for the hth region due to emission reduction, T^{ih}, in the own region and also from optimal emission responses to T^{ih} in all other regions, T^{-ih*}. The net benefit function for the drainage basin, π^{d}, is the same as (3) except for the change in the summation of benefits from emission reductions from h to d, and in costs of emission reductions from d to h.

When maximising the net benefits of the regional decision unit, we obtain

$$\frac{\partial C^{dh}}{\partial l^{dh}} + \theta^{h} = \sum_{j=1}^{s} v^{jh} V^{j}_{D^{j}} D^{j}_{T} a^{ij} b^{di} \qquad (4)$$

The differences as compared to optimal abatement at the co-operative scale are the change in summation over marine basins, where (4) includes only these basins for which $v^{jh} > 0$. Since the number of marine basins belonging to a certain regional decision unit cannot exceed the total number of basins, total marginal benefit must be smaller at the jurisdictional than at the co-operative scale. The right-hand side of (4) is then lower than that of (2), which, from the assumption of a concave benefit function, implies less emission at the co-operative scale.

By comparing first-order condition for optimal cleaning levels under the jurisdictional, l^{h*} and drainage basin regional scales, l^{d*}, it is found that

$$l^{d*} \geq l^{h*} \quad when \quad \sum_{j=1}^{s} v^{jd} V^{j}_{D^{j}} D^{i}_{T} a^{ij} b^{di} - \theta^{d} \geq \sum_{j=1}^{s} v^{jh} V^{j}_{D^{j}} D^{i}_{T} a^{ij} b^{di} - \theta^{h} \qquad (5)$$

From (5) it is seen that first- best optimum is obtained under both types of regional decision scales when marginal benefits from emission reductions plus the shadow values of cleaning options are the same for all regions. For non-binding cleaning capacity constraints, the common marginal benefit then creates efficient allocation of emission reductions among regions. This analytical result can be compared with the impact of perfectly mobile capital, which may create efficiency among regions being able to tax the immobile labour input (e.g. Oates and Schwab, 1988).

In the case of equal marginal benefits and non-binding cleaning capacity constraints both sides of (5) must be the same for all regions. The equality of the right hand side among regions requires *i),* identical transport coefficients and marine basins for all decision regions, which is the same as one single marine basin, or *ii*) identical and constant marginal benefits from emission reductions. These requirements are quite restrictive and probably not fulfilled for any marine water and its catchment. Therefore, it is an empirical issue which decision scale gives the largest net benefits.

However, one advantage of decision at the regional scale, in the perspective of the region, is the guarantee of avoiding net losses. A co-operative outcome might very well result in some region being a net loser, if this region has relatively low pollutant abatement costs and benefits (e.g. Gren 2001). Depending on the abatement costs and benefits for jurisdictions and drainage basins, occurrences of net losers in a cooperative solution might differ between different regional decision scales. If there is at least one net loser, cooperation generating first-best solution is likely not to occur unless there are binding agreements on compensation to the net loser.

2.2 Cost effective management

Although benefits from water quality improvements are the aim of many current water policies, most targets are expressed in reductions in pollutant loads from the mainland to coastal waters without consideration of transports among water basins. One example is the ministerial agreement on a 50 per cent reduction of the nitrogen load to the Baltic Sea (Helcom, 1988). Minimum cost under the cooperative decision scale, C^C, for achieving a certain minimum coastal water target reduction, E, is written as

$$Min \qquad C^C = \sum_{k=1} C^{dh}(l^{dh}) \tag{6}$$

s.t.

$$\sum_i \sum_d e^{di} \geq E$$

$$l^{dh} \leq \bar{l}^{dh}$$

where k is either d or h. The first-order condition for optimal emission reduction is then $\frac{\partial C^{dh}}{\partial l^{dh}} + \theta^{dhc} = \alpha b^{di}$ where α is the Lagrange multiplier on the load target E. Now, only land pollutant transport coefficients, i.e. b^{di}, and pollutant abatement costs at the emission sources determine the optimal solution.

Under regional decision making, emission reduction targets are $\Sigma_d\, e^{dh} \geq E^h$ at the jurisdictional scale and $\Sigma_h\, e^{dh} \geq E^d$ at the drainage basin scales. The targets E^h and E^d are calculated as a percentage of total load from the hth and dth region respectively. The total load reduction from both decision scales then corresponds to the overall reduction E at the cooperative scale. The optimal conditions for the regional decision scales are $\frac{\partial C^{dh}}{\partial l^{dh}} + \theta^{dhd} = \alpha^d b^{di}$ and $\frac{\partial C^{dh}}{\partial l^{dh}} + \theta^{dhh} = \alpha^h b^{di}$ at the drainage basin and jurisdictional scale respectively. For this target, there are no spill-over impacts among regions, and they therefore do not need to take each others' decisions into consideration when making their own optimal choices.

For non-binding capacity constraints, the only difference between the conditions for cost effective solutions between the decision regions is the Lagrange multipliers α, α^h and α^d respectively. The Lagrange multiplier measures the cost for tightening the load target by one unit, and the larger it is the higher is the cost of achieving the same load targets. Since there are more abatement locations to choose among at the cooperative decision scale, costs for achieving the targets under this decision level must be the same or lower than under any regional decision scale. When comparing the two regional decision scales, their difference is expressed by α^d and α^h respectively, and the size of these Lagrange multipliers is determined by the costs and availability of measures within their decision territory. When these are the same, there is no difference between the regional decision scales.

3. Application to Himmerfjärden

In order to make comparisons among different decision levels data are needed on abatement measures, their costs and impacts on different marine basins. Further, information on associated values in monetary terms is required. A water which is relatively well investigated with respect to these data is a bay of the Baltic Sea called Himmerfjärden. It is situated 100 km south-west of Stockholm, Sweden. The relation between nutrients and water transparency in this estuary has been investigated for approximately 25 years (Elmgren and Larsson, 1997), and studies of emission sources in the catchment have been made (Johansson, 1989; Scharin 2003). Further, valuation studies of changes in water transparency including this estuary have been carried out (Söderqvist and Scharin, 2000).

The drainage basin of Himmerfjärden covers an area of 1268 km^2. While this is a small area in comparison with the drainage basin districts suggested for Sweden, it still includes several municipalities. Further, the basin is typical with regard to distribution of emission sources, where point source emissions are usually concentrated in densely populated areas and non-point source emissions in rural regions. Therefore, while absolute numbers are only valid for conditions in Himmerfjärden, the results can still be indicative for other, larger areas.

3.1 Brief description of emission sources and transport.

The catchment area and Himmerfjärden contain five jurisdictions (municipalities), four marine basins, and eleven drainage basins. However, since these drainage basins are small, they are grouped into four main drainage basins, see Table A1 in Appendix A for a description. The areas of the drainage basins correspond to coastal basins as defined by sea bottom tresholds (Engqvist, 1997) according to the map in Figure 1.

The major part of the four main drainage basins is situated west of Himmerfjärden in the municipalities of Gnesta, Södertälje, and Trosa. The rest is located in the municipalities of Nynäshamn, Botkyrka, and to some extent Södertälje. Basin 2 is the largest one and includes parts of three municipalities. Basin 4 is almost to its whole extent found within the municipality of Södertälje. Data on the sizes of these municipalities and how large part of their area that is situated within the main drainage basins are presented in Table A2 in Appendix A.

Nitrogen loads from each region to the coastal basins include point source emissions from sewage treatment plants, and non-point source emissions. The latter type of emission includes leakage from agriculture, forests and atmospheric deposition over the water surface within the drainage basin. Data on nitrogen loads from the four main drainage basins are available in

Johansson (1989) and Gren et al. (2000). Data on point source emissions from municipalities can be found in Scharin (2003), but non-point source emissions and loads into coastal waters need to be calculated.

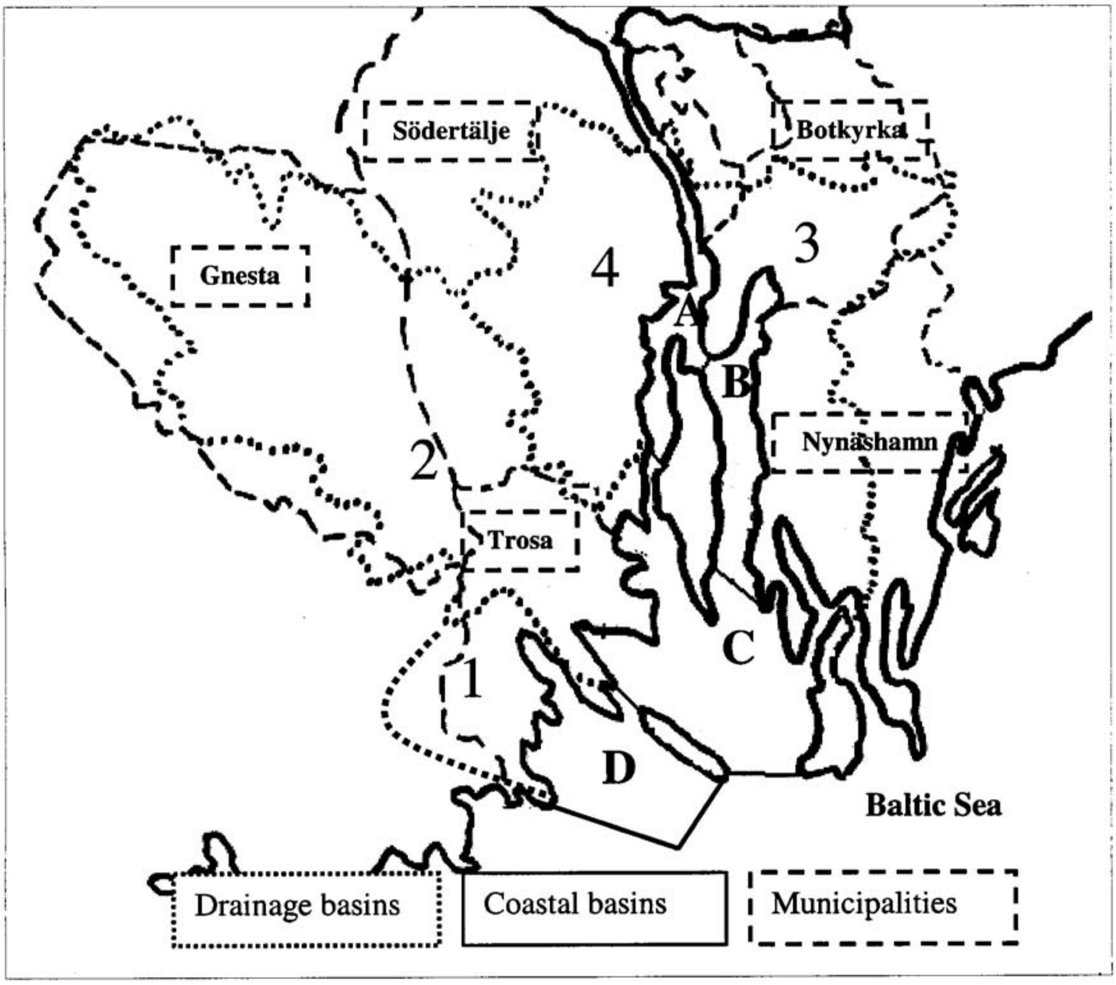

Figure 1.: Regions of the Himmerfjärd catchment. Drainage basin 1 discharges its nitrogen load into coastal basin D, drainage basin 2 into coastal basin C, drainage basin 3 into coastal basin B, while finally drainage 4 discharges its nitrogen load into coastal basin A.

In order to calculate nitrogen load from municipalities, each municipality is divided into several parishes for which crop production data are available (SCB, 1997). In the used database some parishes were divided into two areas. Most of the parishes had their total area within one drainage basin, but there were some of which certain parts were located within another basin than the one observed. Parishes were included that totally or to a large part are situated within one of Himmerfjärden's drainage basins. Those with a major part of their farmed land within another drainage basin were excluded from the study. In Table A1 in Appendix A, the area for each coastal basin's drainage basin is described. As can be seen from this table, 19% of the drainage basin's area is used for crop production.

However, data on loads to the coastal water from inland non-point sources do not account for the retention of nitrogen during its transport from sources to coastal waters. Unfortunately, there are no estimates of retention in the drainage basins. As an approximation of the retention, we compared data on nitrogen emissions to measurements of nitrogen in the coastal water (Johansson, 1989, p 36). This results in estimates of nitrogen retention in the different drainage basins that vary between 0.4 and 0.5. Accounting for the existence of retention from non-point emission source accentuates the role of the point sources. The latter source accounts for about 3/4 of the total nitrogen load.

In order to calculate the load from the different coastal zones to the coastal basins, we need information on nitrogen transports between the coastal basins. In Table A3 in Appendix A, a coefficient matrix is presented which shows the proportion of the load to a coastal basin that is being transported to the other coastal basins. The final, or steady state, deposition are reflected by these proportions of nitrogen transported from basin i to j, i.e. they correspond to a^{ij} in Section 2.

Based on these data, nitrogen loads to the coastal waters, to all coastal basins, and to the own coastal basins are calculated, the results of which is shown in Table 1. The nitrogen load refers to emission sources within the catchment region, and does not include atmospheric deposition from outside emission sources.

Table 1 shows that there are one dominating municipality and one dominating drainage basin region, Botkyrka and Basin 3 respectively. The reason is that these regions contain the largest single nitrogen emission source, the Himmerfjärden sewage treatment plant. It accounts for about 2/3 of the total load to the coastal waters, see Table A4 in Appendix A. The column 'N own basin' in Table 1 shows the load from emission sources within the region that is deposited on the marine basin of the decision region. For example, the municipality of Södertälje deposits in total 40.4 tons of N in the coastal basins, of which 5.6 tons are deposited in the basins where Södertälje discharges nitrogen directly. Thus, Södertälje exports 34.8 tons of N to coastal basins of other municipalities.

Table 1: Nitrogen loads from to coastal waters and basins from municipalities and drainage basins, tons of N per year.

Municipalities: Regions	*N coast*	*N basins*	*N own basin*	*Drainage (and marine) basins:* Regions	*N coast*	*N basins*	*N own basin*
Trosa	52.9	20.9	4.3	1	5.8	3.5	3.5
Gnesta	67.5	57.3	6.4	2	131.7	89.2	15.8
Södertälje	72.6	40.4	5.6	3	616.9	194.5	166.7
Botkyrka	593.8	13.6	1.9	4	55.6	25.9	6.1
Nynäshamn	23.2	180.9	53.0				
Total	**810.0**	**313.1**	**71.2**		**810.0**	**313.1**	**192.1**

3.2 Benefits and costs of nitrogen reductions

In order to conclude whether nutrient abatement measures would involve a net social gain, it is necessary to quantify the increase in people's well-being that an improved water quality in Himmerfjärden would cause. This requires information on biological damage from nitrogen loads to the coastal basins and also peoples' preferences of these damages expressed in monetary terms. Based on this, we can then compare these gross benefits with costs of nitrogen abatement and calculate the level of abatement which gives the maximum net benefits for the different solution concepts analyzed in section 2.

No studies of benefits from nitrogen abatement are available for Himmerfjärden exclusively. However, there are data from one benefit study carried out in 1998 for the whole Stockholm Archipelago, of which Himmerfjärden is one part. This study followed a contingent valuation approach. A random sample of adult inhabitants in the Stockholm region received a questionnaire. They were asked to state their willingness-to-pay (WTP) for a nutrient abatement programme that would improve the water quality in the archipelago, see Söderqvist and Scharin (2000) for details. In the valuation scenario, it was said that the programme would involve measures in agriculture and at sewage treatment plants. The results of the programme were quantified as follows: The water transparency in the inner and central parts of the archipelago would on average increase about 1 metre in 10 years. This would mean that, for example, in the inner parts of the archipelago, the sight depth would increase from the present average of about 1 metre in summers to about 2 metres in 10 years.

Assume now that the inhabitants in Himmerfjärden's catchment area are only concerned about the water quality in Himmerfjärden, not in any other part of the Stockholm archipelago. This assumption allows a computation of aggregate WTP for a 1-metre increase in water transparency in Himmerfjärden by multiplying the number of adult inhabitants in the catchment area by the mean WTP of those respondents who lived in this area. It is true that people living in Himmerfjärden's catchment may very well care also for the water quality in other parts of the archipelago. However, in order to obtain at least indications on the size of the benefits for a water quality improvement in Himmerfjärden, we consider this assumption to be the most reasonable way to use the existing data set.

65 of 1,840 respondents turned out to be inhabitants in Himmerfjärden's catchment area. They had a mean WTP of SEK 69 per month (std. dev.: 95; median: 50). Since this indicates a close similarity to the corresponding estimate for the whole survey (mean: 71; std. dev.: 115; median: 50), we stick to the results obtained for the whole survey as a basis for aggregating WTP, see Söderqvist and Scharin (2000) for details. Taking into account non-respondents by using results from a follow-up survey and protest answers to the valuation scenario, this results in a present value of aggregate WTP of SEK 19-32 million per year, see Table 2. When asking about whether the respondents were willing to pay anything for the nutrient abatement programme, they could choose between "yes, definitely", "yes, probably" and "no". The lower end of the aggregate WTP interval corresponds to a conservative case where the WTP of those respondents who answered "yes, probably" were assumed to have a zero WTP. An aggregate WTP of SEK 32 million corresponds to a case when the stated WTP amounts of those who responded "yes, probably" were included in the computation.

Table 2: The aggregate WTP for a reduced eutrophication in Himmerfjärden

Case	*Mean WTP per adult resident, year 1, SEK*	*Aggregate WTP[a] year 1, SEK million*	*Aggregate WTP, present value[b], SEK million*	*Aggregate WTP, present value per year,SEK million*
Conservative	436	23.9	193.8	19.4
Non-conservative	725	39.7	322	32.3

[a] The population is the 54,800 residents of age 18-75 years in Himmerfjärden's catchment area
[b] Time horizon: 10 years (as specified in the valuation scenario). Discount rate: 4 %

In order to relate the estimated WTP to nitrogen loads, we use Färlin's (2002) estimated relationships between nitrogen load and concentrations of nitrogen and between nitrogen concentration and water transparency. These calculations are based on measurements at a station located in coastal basin 3, with an average water transparency of 6 m. The estimated relationship between nitrogen concentration in mg/m^3 (C_n), and nitrogen load in tons of nitrogen (N) is

$$C_n = 0.2\ N + 275$$

The estimated relationship between C_n and water transparency measured as Secchi depth in metres (S), for the same station is

$$\log S = 4.754 - 1.641 \log C_n$$

Calibration is made at the average water transparency of 6 m for each basin. When calculating the water transparency and, hence benefit functions, from nitrogen reductions at the jurisdictional scale when jurisdictions share the same marine basin, benefits are allocated according to nitrogen load. The resulting equations are presented in Appendix B.

When estimating costs of nitrogen reductions, three classes of measures are included: (1) improvement of the cleaning capacity of point sources, (2) reductions in nonpoint source emissions at the emission sources, and (3) construction of wetlands as nitrogen sinks. One type of point source mitigation measure is included: improvement of the nitrogen cleaning capacities at the sewage treatment plants. It is assumed that the associated cost corresponds to

SEK 15/kg N reduction from sewage plants. Three types of nonpoint source emission reduction measures are considered: reductions in the use on nitrogen fertilizers, cultivation of catch crops, and construction of wetlands. Catch crops are sown at the same time as the ordinary crop but continue to grow, and thereby make use of residual nitrogen in the soil, when the ordinary crop is harvested. The unit cost per kg N reduction for catch crops is assumed to amount to SEK 20. Costs of nitrogen fertilisers reductions in each drainage basin are calculated as associated losses in profits, see Appendix C. Wetland construction costs are based on calculations in Byström (1998), which gives a total cost, consisting of opportunity cost of land plus management cost, which together amount to SEK 7420 per hectare.

3.3 Maximum net benefits

Maximum net benefits under the various solution concepts are calculated by use of the Gams software (Brooke et al., 1998). We first present maximum net benefits for the co-operative case, when the entire catchment is the decision unit. Table 3 presents estimates for the two cases of estimated benefits of water quality improvements, SEK 19 million per year (conservative estimate) and 32 million per year (non-conservative estimate).

Table 3 Net benefits in millions of SEK per year for different decision units under co-operative solutions

Municipalities:			*Drainage basins:*		
Regions	*19*	*32*	*Regions*	*19*	*32*
Trosa	1 649	2 703	Basin 1	87	279
Gnesta	5 100	9 598	Basin 2	7 969	14 483
Södertälje	3 367	4 174	Basin 3	8 894	24 733
Botkyrka	-2 749	3 860	Basin 4	2 076	1 744
Nynäshamn	11 644	20 873			
Total	**19 011**	**41 208**		**19 026**	**41 239**

Total net benefits are more than doubled for the higher marginal value of water quality improvement. This is due mainly to the larger marginal value; there is only a slight decrease from 278 tons of N to 275 in total load to all coastal basins. This is, in turn, explained by the large proportion (90 per cent) of abatement by Himmerfjärden sewage treatment plant. The fact that the plant is situated in the municipality of Botkyrka also explains why Botkyrka experiences negative net benefits in the conservative benefit case.

Maximum net benefits for the two regional decision systems, at the municipal or drainage basin level, are calculated by inserting the first-order optimality conditions (4) as restrictions for each regional maximisation of net benefits in the optimisation program. The results are presented in Table 4.

Table 4: Net benefits in millions of SEK per year for different decision units under jurisdictional and drainage basin decisions

Municipality decisions:			*Drainage basin decisions:*		
Regions	*19*	*32*	*regions*	*19*	*32*
Trosa	89	289	Basin 1	87	280
Gnesta	51	162	Basin 2	2 713	8 088
Södertälje	2	1 895	Basin 3	8 642	21 769
Botkyrka	4	2 417	Basin 4	960	2 395
Nynäshamn	6	478			
Total	**152**	**5 241**		**12 402**	**32 532**

For both decision regions, total net benefits are – as expected – less than those of the co-operative solution. In the conservative case, only one region at each scale, Trosa and Basin 3, carries out abatement. For both these regions, the impact on the own region is relatively large so that marginal benefit equals marginal cost at a positive abatement level. The relative advantage of the two regional decision systems remains for the non-conservative case. Abatement then takes place in two municipalities, Trosa and Botkyrka, and in one drainage basin, Basin 3. The reason for the differences in total net benefits is that the lowest cost options are available to a relatively large extent in Basin 3. This means that decision at this scale includes all nitrogen reduction benefits and costs, where benefits from reduced nitrogen transports are obtained also in other basins. The municipalities of Botkyrka and Nynäshamn are both located in Basin 3, but they capture only a part of nitrogen reduction benefits, which reduces overall nitrogen reductions.

Another interesting result is that Botkyrka would gain from the non-cooperative solution in the conservative case, and Basin 4 would make a gain from not co-operating in the non-conservative case. If co-operation is legally possible, this is thus more likely to occur at the basin scale for the lower value of water quality improvements and at the municipality scale at the larger value of water quality improvements. This result is similar to that of Gren (2001), where it is shown that the distribution of net gains among Baltic Sea countries in co-operative and non-cooperative solutions depends on how much water quality improvements are valued.

3.4 Minimisation of costs

The countries surrounding the Baltic Sea have made an agreement on reducing nitrogen loads to the Baltic Sea by 50 per cent (Helcom, 1988). It was not specified whether each country should make such a reduction, or if it could be made on an overall level. As demonstrated in Gren et al. (1997), total costs increase by four times if each country reduces its load by 50 per cent, as compared to an overall reduction. The main reason is the differences in abatement costs among countries.

The 50 per cent nitrogen reduction target is also used here to show differences in outcomes between the two regional scales, see Table 5.

Table 5. Minimum costs for a 50 per cent nitrogen reduction under regional and co-operative decision scales

Municipalities:			*Drainage basins:*		
Regions	*Cooperation*	*Regional*	*Regions*	*Cooperation*	*Regional*
Trosa	0	1 476	Basin 1	0	160
Gnesta	0	3 580	Basin 2	0	5 678
Södertälje	0	3 133	Basin 3	8 099	6 169
Botkyrka	8 099	5 952	Basin 4	0	2 247
Nynäshamn	0	849			
Total	**8 099**	**14 990**		**8 099**	**14 254**

Due to the dominating role of discharges from and the low abatement costs at Himmerfjärden sewage treatment plant, the region where this plant is situated makes all required abatement in the coordinated solution. This solution is about 40 per cent less expensive than regional decision-making where each region minimises its cost for a 50 per cent nitrogen reduction.

The total cost is approximately 5 per cent larger for municipality decision unit scales than for drainage basin scales. The reason, as demonstrated in section 2, is the higher availability of low cost abatement measures in each of the basins as compared to each of the municipalities. This result is similar to that of maximisation of net benefits, where these are larger at the drainage basin decision-making. However, one difference is the divergences in costs for the region, which undertakes all required abatement in the case of co-operation, i.e. Botkyrka and Basin 3 respectively. In the minimum cost decision rule, the loss from changing from regional decision making to the cooperative scale is larger for the municipality decision scale than for the drainage basin decision scale. It thus seems as if the results indicating relative advantages of basin scale decision-making in the case of a minimum cost decision rule are relatively robust as compared to the net benefit maximisation decision rule.

4. Conclusions

The main purpose of this paper was to make an analytical and empirical investigation of the role of two types of regional decision units for efficient and cost effective water quality management of a relatively large coastal region: municipality and drainage basin scales. The entire region was used as a reference scale since all benefits and costs of emission reductions are included in decisions at this scale. Efficiency can be reached at both regional decision scales either under the highly restrictive condition that benefit and cost functions are the same in all regions, or if all regions cooperate. Cost effective pollution reduction is obtained if the marginal cost functions for reducing pollutant loads to the marine basins are the same. If none of these circumstances prevails, differences in divergences from the first best solution between the two regional scales depend on asymmetry between decision regions with regard to environmental benefits and costs of emission reductions, and the regions' coverage of marine basins within their territory.

In our application to Himmerfjärden estuary, the same abatement measures were assumed in the decision regions. As a consequence, differences in abatement costs among decision regions depend on various conditions for transports of nitrogen from the emission sources to the coastal waters, and on availability of abatement capacity for different measures. Environmental benefits from emission abatement were based on estimated relations between nitrogen emissions and water quality – measured as water transparency – and on a contingent valuation study on changes in water transparency. The results showed that the total net benefits are higher under drainage basin decision for both a conservative and a non-conservative estimation of the benefits of water quality improvements. It was also demonstrated that costs for a 50 per cent nitrogen reduction are lower at the drainage basin scale. However, under a cooperative scale, there is a net loser under a jurisdictional regional division, but not for any drainage basin authority. Thus, although net benefits and abatement costs are at the advantage for the drainage basin decision scale, cooperation among regions to reach first-best optimum is less likely for this regional scale.

Needless to say, the analysis and the application to Himmerfjärden involve several simplifying assumptions. For example, the analysis assumed separability between water policy and other regional issues. In practice, various regional policy issues are likely to be linked and balanced against each other, and there might be a risk for lax water regulation (e.g. Oates and Schwab, 1988). If these impacts vary among different decentralised decisions, their relative performance would be affected. Also different types of uncertainty may affect the results, such as stochastic pollutant transports and asymmetric information among the regulator and regulated firms. The existence of asymmetric information is likely to increase total costs since the regulator may have to pay informational rents to firms with relatively low costs. On the other hand, this may be an important argument in favour of delegation of decisions due to the better use of information at different scales. If this occurs, efficiency losses as calculated in this study can be outweighed by reduced uncertainty of environmental impacts of emissions.

The difficulty of obtaining data is demonstrated by the many assumptions necessary for calculating net benefits of nitrogen reductions to Himmerfjärden under various regional decision systems, despite the fact that this estuary is relatively well investigated with regard to pollutant transports. The results show that the advantage of the two regional systems may change depending on assumed benefits from water quality changes. It was also demonstrated that the likelihood of cooperation among regions differs depending on the choice of decision rule: maximisation of net benefits or minimisation of costs for pre-specified targets. It could therefore be worthwhile to make further efforts to collect data on coastal basin pollutant transports, biological impacts and economic valuation of water quality improvements when designing efficient or cost effective programmes for pollutant reductions to coastal and marine waters.

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References

Barrett, S. (1990). Global environmental problems. *Oxford Review of Economic Policy* 8(1):68-79.

Brooke, A., Kendrick, D., and A. Meeraus (1998). GAMS. A user's guide. San Fransisco: The Scientific Press.

Brännlund, R. and I-M. Gren., (1999). *Costs of differentiated and uniform charges on polluting inputs: An application to nitrogen fertilizers in Sweden*. In Boman, M. Brännlund, R. and B. Kriström. (Eds.) *Environmental economics and regulation*. Kluwer Academic Publishers, The Netherlands.

Byström, O. (1998). The nitrogen abatement costs in wetlands. *Ecological Economics* 26:321-331.

Elmgren, R. and U. Larsson, (1997) "*Himmerfjärden . Changes in a nutrient enriched coastal ecosystem*" (In Swedish with an English summary). Swedish Environmental Protection Agency, Report No. 4565.

Elofsson, K. (2002). *Economics of marine pollution*. Doctoral thesis. Department of Economics, Swedish University of Agricultural Sciences, UppsalaElofsson, 2002

Engqvist, A. (1997). *Vatten- och närsaltutbyte i hela Himmerfjärden*. In Elmgren, R. and U. Larsson (eds.) "Himmerfjärden . Changes in a nutrient enriched coastal ecosystem" (In Swedish with an English summary). Swedish Environmental Protection Agency, Report No. 4565.

EU, (2000). *Establishing a framework for community action in the field of water policy*. Directive of the European Parliament and of the Council 200/60/EC., Brussels, Belgium.

Folmer, H. and P. van Mouche (2000). *Transboundary pollution and international cooperation.* In: T. Tietenberg and H. Folmer (eds) The International Yearbook of

Environmental and Resource Economics 2000/2001. Edward Elgar, Cheltenham.

Folmer H. and P. van Mouche (2002). The acid rain game: a formal and mathematically rigorous analysis. Working paper. Department of General Economics. Wageningen University.

Fredriksson, P.G., and D.L. Millinet, (2002). Strategic interaction and the determination of environmental policy across US states. *Journal of Urban Economics* 51:101-122.

Färlin, J. (2002). *Secchi disc relations and the willingness to pay for improved water quality in the Stockholm archipelago*. Master thesis 2002:2. Department of Systems Ecology, Stockholm University, Sweden.

Gren, I-M., and R. Brännlund. Enforcement of regional environmental regulations: Nitrogen fertilizers in Sweden. In Hanna, S. and M. Munasinghe (eds.), "*Design Principles of Property Rights System*", The World Bank, Washington, 1995.

Gren, I-M., Elofsson, K., and P. Jannke. Cost effective nutrient reductions to the Baltic Sea. *Environmental and Resource Economics*, 10(4):341-362, 1997.

Gren, I-M.. International versus national actions against pollution of the Baltic Sea. *Environmental and Resource Economics*, 20(1): 41-59, 2001

Gren, I-M., Destouni, G., and R. Temponi, Cost effective policies for alternative distributions of stochastic water pollution. *Journal of Environmental Management,* 66(2):127-157, 2002.

Gustafsson, J-E. (2000). *Environmental charges in Franch agriculture- Advanced agriculture with obstacles.* TRITA-AMI, report 3076, Royal Institute of Technology, Stockholm

Helcom, (1988). *HELCOM Declaration*, Helsinki Commission, Helsinki, Finland

Helfand, G. and B. House, (1995). Regulating nonpoint source pollution under heterogeneous conditions. *American Journal of Agricultural Economics* 77(4):1024-1032.

Hoel, M., (1992). International environment conventions: The case of uniform reductions of emissions. *Environmental and Resource Economics* 2:141-159.

Johansson, S. (1989). *Näringsämnesbelastning från Himmerfjärdens tillrinningsområde En översikt av olika källor*. Askölaboratoriet, Technical report No. 5, Sweden.

Johnsson, H., and M. Hoffman (1997). *Nitrogen leaching from Swedish agriculture – calculations of normal leaching and possible measures.* The Swedish Environmental Protection Board, Report 4741.

Kaitala, V., Mäler, K-G. and H. Tulkens, (1995). The acid rain game as a resource allocation process with an application to the international cooperation among Finland, Russia and Estonia. *Scandinavian Journal of Economics* 97(2): 325-343.

Kunce, M., and J.F. Shogren (2002). On environmental federalism and direct emission control. *Journal of Urban Economics* 51:238-245.

Levinson, A. (1997). A note on environmental federalism: Interpreting some contradictory results. *Journal of Environmental Economics and Management* 27:49-63.

Markusen, J. E. Morey, and N. Olewiler, (1995). Competition in regional environmental policies when plant locations are endogenous. *Journal of Public Economics* 56:55-77.

Mäler, K-G., (1991). International environmental problems. In D. Helm (ed.), *Economic policy towards the environment*, Blackwell, Oxford.

Mäler, K-G., (1993). The acid rain game II. Beijer Discussion Papers Series No. 32, Beijer International Institute of Ecological Economics, Stockholm, Sweden.

Oates, W.E. and R. Schwab, (1988). Economic competition among jurisdictions: Efficiency enhancing or distortion inducing? *Journal of Public Economics* 35:333-354.

Paulsen, C.M., and K. Wernstedt, (1995). Cost-effectiveness analysis for complex managed hydrosystems: An application to the Columbia river basin. *Journal of Environmental Economics and Management* 28:388-400.

SCB (1992). *Atlas över rikets indelningar: i län, kommuner, församlingar och tätorter. Andra utgåvan.* Statistics Sweden, Stockholm.

SCB (1995). *Atlas över rikets indelningar: tabelldel 1995.* Statistics Sweden, Stockholm.

SCB (1997). *Åkerarealens användning 1994* (1997). Statistics Sweden, Stockholm.

SCB (1998), Rolf Selander, Lantbruksregistret, Statistics Sweden; personal communication.

SCB (2002a). *Agricultural Statistics,* 2002, Table 10.7. Statistics Sweden, Stockholm.

SCB (2002b). *Tillförsel av kväve efter produktionsområde, grödgrupp ochgödselslag.* År 1998/1999-2000/2001. Statistics Sweden, Stockholm.

Scharin, H. (2003). *The efficient environmental state of coastal zones – A study of eutrophication in the Stockholm archipelago*. Licentiate thesis, Department of Economics, Swedish University of Agricultural Sciences, Uppsala, Sweden.

SMHI (1998). Ylva Westam, Swedish Meteorological and Hydrological Institute; personal communication.

Söderqvist, T., and H. Scharin, (2000). *The regional willingness to pay for a reduced eutrophication in the Stockholm archipelago*. Beijer Discussion Paper Series No. 128. Beijer International Institute of Ecological Economics, The Royal Swedish Academy of Sciences, Stockholm.

SOU (2002). *Klart som vatten*. SOU 2002:105, Utredningen Svensk Vattenadministration, Ministry of Environment, Stockholm.

Turner, K., Georgiou, S., Gren, I-M., Wulff., F., Barett, S., Söderqvist, T., Bateman, I.J., Folke, C., Langaas, S. Zylicz, T., Mäler, K-G., and Markowska, A. (1999). Managing nutrient fluxes and pollution in the Baltic: An interdisciplinary simulation study. *Ecological Economics* 30:333-352.

Appendix A. Details about Himmerfjärden catchment

Table A1. Studied drainage basins and municipalities

Drainage basin	*Area, km^2*	*Arable land*	*Municipalities within drainage basins*
1	96.0	13.7(14%)	Trosa
2	570.0	102.5 (18%)	Gnesta(64,2%), Trosa(19,6%), Södertälje(16,2%).
3	317.4	65.2 (23%)	Nynäshamn(60%), Botkyrka(40%),
4	284.9	62.3 (22%)	Södertälje
Total	**1268.3**	**243.7 (19%)**	

Source: Calculations based on SCB (1992, 1995, 1997, 1998) and SMHI (1998).

Table A2. Municipalities in Himmerfjärd catchment

Municipality	*Total area, km^2*	*Area within a drainage basin*	*Arable land*	*Arable land within drainage basin*
Botkyrka	196.4	148.9 (76%)	32.0	27.4
Södertälje	675.5	377.5 (56%)	116.5	78.7
Nynäshamn	356.5	168.5 (47%)	59.7	37.8
Gnesta	460.5	365.5 (79%)	82.5	66.5
Trosa	207.9	207.9 (100%)	33.5	33.5
Total	**1896.8**	**1268.3 (67%)**	**324.2**	**243.9**

Source: Calculations based on SCB (1992, 1995, 1997, 1998) and SMHI (1998).

Table A3. Coefficient matrix, share of total load to basin *i* transported to basin *j*

From/to	*Basin D*	*Basin C*	*Basin B*	*Basin A*
Basin D	0.11	0.29	0.10	
Basin C	0.03	0.27	0.11	
Basin B		0.09	0.12	
Basin A				0.60

Source: Calculations based on Engqvist (1997).

Table A4. Land area and nitrogen sources in Himmerfjärden region

	Area km²	***Non-point emission***	***Direct emission***	***Total N***
Coastal basins A, drainage basin 4:				
Bränningeån	58	6.3	-	6.3
Vaskabäcken	10.7		0.3	0.3
Järnaån	92.3	34.9	0.3	35.2
Enebyån	32.9	7.5	2.5	10.0
Hölöån	30.9	17.5	-	17.5
Mörkö	55.1	16		16
Lake Mälaren			155	155
Total A:				*240.3*
Coastal basin B, drainage basin 3:				
Fitunaån	73	50	5.4	55.4
Saxbroån	100	39	0.6	39.6
The Himmerfjärd sewage treatment plant			572.0	572.0
Total B:				*667.0*
Coastal basin C drainage basin 2:				
Tullgarnsån	18	16.1		16.1
Trosaån	587	167.6		167.6
The Trosa sewage treatment plant			15.7	15.7
Total C:				*199.4*
Coastal basin D drainage basin 1:				
Örholmsån	29	25.6		25.6
Atmospheric			91	91
Total D:				*116.6*
Total				***1223.3***

Source: Calculations based on SCB (1992, 1995, 1997,1998, 2002a, 2002b), SMHI 1998, and Johnsson & Hoffman (1997).

Appendix B. Secchi depth equations at the drainage basin and the municipal scales

Coastal basins

Basin D:	$S^D = 62193*(279+0.022*N^D)^{-1.641} - 5.99013$
Basin C:	$S^C = 74981*(281+0.054*N^C)^{-1.641} - 5.98919$
Basin B:	$S^B = 66903*(290+0.024*N^B)^{-1.641} - 5.98724$
Basin A:	$S^A = 60610*(275+0.12*N^A)^{-1.641} - 5.99546$

Municipalities

Trosa:	$S^T = S^D + 0.196*S^C$
Gnesta:	$S^G = 0.64*S^C$
Södertälje:	$S^S = S^A+0.164*S^C$
Botkyrka:	$S^B = 0.9*S^B$
Nynäshamn:	$S^N = 0.1*S^B$

Appendix C. Cost functions for nitrogen fertiliser reductions

The costs of reducing the use of nitrogen fertiliser from current level, N^*, is calculated as the associated changes in producer surplus, by means of the nitrogen demand function. The nitrogen demand function is assumed to be linear according to

$$N = a - bP \tag{C1}$$

where P is the price of nitrogen. The values of a and b are found by applying results from estimated nitrogen demand functions in Gren and Brännlund (1995). The study contains estimates of nitrogen demand in different Swedish regions. The results from the Mälar region are applied here since Himmerfjärden drainage basin is a part of that larger region. The estimated elasticity is -0.34, that is

$$\frac{\partial N}{\partial P}\frac{P}{N} = -0.34 \tag{C2}$$

For given values of P and N we can solve for b in (C2) since from (C1) we have that $-b=\partial N/\partial P$. Then, for the same given P and N we can also solve for a in (C1).

The net producer surplus at the given values of P^* and N^* is then simply calculated as the integral of the inverted demand function (C1) minus the cost of fertilisers at the price P^*, according to

$$PS^* = \int_0^{N^*} P\,dN = \int_0^{N^*} (ab^{-1} - b^{-1}N - P^*)\,dN \tag{C3}$$

The cost of reducing N from the level of N^* is calculated as PS^* minus the net producer surplus at N, which gives

$$TC = \int_N^{N^*} (ab^{-1} - b^{-1}N - P^*)\,dN \tag{C4}$$

Cost functions according to (C4) are calculated for the 10 drainage basins in Table A4, with a further division of Trosaån into three jurisdictions. Common assumptions for all regions is that the nitrogen demand elasticity is -0.34, the nitrogen fertilizer price, P^*, is SEK 6.79/kg N (in 1998), and that the nitrogen fertilizer use amounts to 105 kg N per hectare (SCB 2002a,b). The total initial use of nitrogen, N^*, will then differ between the three regions according to their area of arable land. The estimated quadratic cost functions, measured in thousands of SEK, for the different drainage basins and regions are then

Fitunaån:	$TC = 3485 – 25.4N + 0.04331(N)^2$,
Saxbroån:	$TC = 1615 – 25.4N + 0.05587(N)^2$,
Bränningeån:	$TC = 1449.4 – 25.4N + 0.10417(N)^2$,
Vaskabäcken:	$TC = 334 – 25.4N + 0.45168(N)^2$,
Järnaån:	$TC = 2626 – 25.4N + 0.05748(N)^2$,
Enebyån:	$TC = 1474 – 25.4N + 0.10423(N)^2$,
Hölöån:	$TC = 1480 – 25.4N + 0.10204(N)^2$,
Tullgarnsån:	$TC = 1088 – 25.4N + 0.13866(N)^2$,
Trosaån:	
Trosa	$TC = 3485 – 25.4N + 0.04331(N)^2$
Gnesta	$TC = 11393 - 25.4N + 0.01325(N)^2$
Södertälje	$TC =2880 - 25.4N + 0.0524(N)^2$
Örholmsån:	$TC = 556 – 25.4N + 0.2516(N)^2$.