Environmental Modeling and Assessment **5:** 193–203, 2000.

Cost effective management of stochastic coastal water pollution

Ing-Marie Gren [a,c], Georgia Destouni [b] and Henrik Scharin [c]

[a] *Department of Economics, Swedish University of Agricultural Sciences (SLU), Box 7013, 75005 Uppsala, Sweden*
E-mail: Ing-Marie.Gren@ekon.slu.se

[b] *Department of Civil and Environmental Engineering, Water Research Engineering, Royal Institute of Technology, 100 44 Stockholm, Sweden*
E-mail: Gia@wre.kth.se

[c] *Beijer International Institute of Ecological Economics, Box 50005, 10405 Stockholm, Sweden*
E-mail: henrik@beijer.kva.se

This study develops a theoretical tool for investigating the impact on cost effective coastal water management from explicit treatment of: coastal pollutant transports, stochastic pollutant transports in the catchment areas, and wetlands as a pollutant abatement option. It is applied to a relatively well investigated estuary, Himmerfjärden, south of the Swedish capital, Stockholm. The theoretical results indicate that all three factors influence cost effective allocation of measures and associated design of economic instruments. The consideration of stochastic pollutant transports will increase costs, but the direction of influence of the other two factors cannot be determined without empirical support. The application to nitrogen transport in Himmerfjärden shows that, for target nitrogen reductions given in terms of a percentage of pre-abatement loads, the inclusion of coastal transports in the cost calculations lowers the estimated total costs for targets interpreted in terms of nitrogen loads to the marine water. The alternative investigated target interpretation was in terms of nitrogen loads to coastal waters. Depending on the ability of wetlands to abate nitrogen and to change the variance in pollutant load to the coastal recipients, costs are either increased or decreased as compared to when wetlands are excluded as nitrogen abatement options.

Keywords: coastal water pollution, stochastic pollutant transports, wetlands, cost effectiveness, economic instruments

1. Introduction

Today, many coastal areas suffer from damages of eutrophication due to excessive nutrient loads (e.g. [1]). A challenge for mitigating such and other coastal pollution problems is posed by the quantification uncertainties associated with pollutant loads, which follow complex pathways from the emission sources to the coastal waters and then within and between different coastal water basins before entering the marine water. The coastal basins act partly as nutrient and pollutant sinks, which means that only part of the nutrient/pollutant load entering a basin is transported to other basins and to the marine water. In general, the coastal transports, i.e., the nutrient/pollutant transport within and between different coastal basins, are not accounted for when determining targets for marine water pollutant loads (e.g. [2]).

Furthermore, nutrient and pollutant transports in the coastal basins, as well as the transports from the catchment areas into the coastal basins, can be quantified only under conditions of uncertainty. This quantification uncertainty must be accounted for when assessing the resources needed for mitigating coastal eutrophication and pollution, in order to suggest appropriate regulation schemes. Given appropriate descriptions of stochastic pollution transports, the role of uncertainty may become influential on costs, which, in turn, could call for abatement measures that have impacts on both the mean and the variance of pollutant loads to the marine water under study. One such measure is creation of wetlands, which may reduce both the expected value and the variance of pollutant (such as nutrients and heavy metals) loads to downstream water [3]; reducing the load variance implies that wetlands have a "pointifying" impact on non-point sources.

The ultimate purpose of this paper is to calculate cost effective solutions to nutrient and pollutant load reductions and to derive the associated design of the charge and permit market systems. This is carried out with a specific focus on three different factors: (i) the role of nutrient/pollutant sinks within and fluxes between coastal basins for cost effective allocation of marine load reduction measures and the associated design of economic instruments, (ii) the impacts of deterministic versus probabilistic nutrient/pollutant reduction targets, and (iii) the potential of wetlands as cost effective solutions. The general analysis is applied to the specific case of nitrogen reductions to Himmerfjärden, an estuary located about 60 km south of Stockholm. This area has been subjected to about 20 years of nutrient load measurements and is therefore one of the few coastal zones equipped with the oceanographic descriptions necessary for including also coastal nutrient transports into cost calculations.

Important limitations of the present study are the static perspective, and the exclusion of environmental benefits and international aspects valid for the management of many marine waters. The static aspect is based on the indication that changes in nitrogen emission in the drainage basins, in general, generate quick responses in the coastal zones [1]. Environmental benefits are associated with improvements in coastal and marine water quality and also with investments in different abatement options and, in particular, in wetland creation [4]. The main reason for not including such benefits in this study is the difficulty of linking nutrient/pollutant transports to biological impacts measured in monetary terms (e.g. [5]). The international aspect of environmental management has been treated in many other studies (e.g. [6–10]). Most of this literature has assumed the existence of cost and benefit functions and then focused on the identification of conditions and mechanisms for co-ordinated actions among involved countries. The purpose of this paper is more basic, since we investigate the implications on cost-effective solutions of considering coastal transports, transport quantification uncertainty, and wetlands as a potential uncertainty limiting abatement option. Depending on the result, all these factors may influence the potential of co-ordinated actions, but we simplify the analysis by considering only a region within a country, where national decisions can be made on the implementation of cost effective pollutant reductions.

To our knowledge, the linkage of nutrient/pollutant transports in catchment areas with those within the coastal zones for calculating cost effective allocation of abatement measures has not been carried out before. The role of uncertainty on cost effective and efficient solutions to water pollution problems, however, has been analysed in several papers (e.g. [11–16]). In spite of the insights provided by these papers, we find relatively few applied studies. One reason might be the difficulties in actually measuring and quantifying nutrient and pollutant transports and the conceptualisation of the quantification uncertainty that is associated with these difficulties. A unique feature of Himmerfjärden, which we use for site-specific application in this study, is that the area is relatively well investigated with respect to coastal transports of nutrient. These previous investigations allow us to quantify coastal basin-specific nitrogen loads based on reported observation data, rather than by developing a mechanistic model of all the physical and biogeochemical processes that may determine nitrogen transport to and within the different coastal basins.

The paper is organised as follows. First, the principal model for minimising costs for achieving given water quality targets under stochastic conditions is presented. This model is then used for analysing the effective design of economic instrument. The focus of this general part of the paper is on the quantitative linkages between nutrient/pollutant transport in catchments, with that in the coastal basins, and with associated abatement costs. It is outside our scope to also provide the hydrological and biogeochemical models for independent prediction of the nutrient/pollutant transport problem itself. We just assume that the required transport quantification is available, either by appropriate predictive modelling, or by a direct, long-term field monitoring and observation programme, similar to that in Himmerfjärden. Section 4 contains the site-specific application of our generic model to nitrogen loads at the Himmerfjärden estuary. The paper ends with some concluding comments.

2. The model

In the general model developed here, we consider an estuary along a coastline divided into different characteristic coastal zones. Each coastal zone may then in general receive pollutants from several catchments, or drainage basins. For brevity and clarity, however, we will in this analytical part of the paper simplify the theoretical analysis by assuming that there is only one drainage basin associated with each coastal zone. Consideration of several drainage basins affecting the same coastal zone does not alter the qualitative results arrived at in this section. It will, however, have empirical implications, and is therefore accounted for in the applied section 3 based on available site data.

For the present generic analysis, we then have $i = 1, \ldots, m$ different coastal water and drainage basins. Two types of marine pollutant sources are identified in each drainage basin: (a) pollutant transport by soil, ground and surface water to the coastal water, with the pollutant load denoted N^{iL}, and (b) direct pollutant discharge into the coastal water basin, with the associated pollutant load denoted D^i. The specific pollutant may, for instance, be nitrogen that is transported with water from land to the coast, and the total dissolved amount of which may lead to eutrophication problems in the coastal and marine environment. The load terms N^{iL} (a) and D^i (b) would then represent the annual average mass of total dissolved nitrogen that: (a) is hydrologically transported to the coast through the soil, ground and surface water systems of a drainage basin, and (b) is discharged directly into the coastal water. An example of the direct deposition source (b) would then be the direct discharge of total dissolved nitrogen from sewage treatment plants into the coastal water.

The pollutant load N^{iL}, which we in the following also refer to as non-point source pollution, is the annual average result of all the various hydrological and biogeochemical processes that affect the dissolved pollutant concentration and mass flux along the source-to-coast pathway. As one possible abatement measure, the dissolved pollutant mass may be subjected to retention (mass removal from the dissolved aqueous phase) by combined hydrological and biogeochemical processes in created wetlands. The average annual mass removal of pollutant by wetlands is then denoted N^{iw} and is simply written as

$$N^{iw} = N^{iw}(N^{iL}, W^i; H^i), \tag{1}$$

where W^i is the area of wetlands within the drainage basin i, and H^i is a parameter vector that includes various hydrological and biogeochemical factors influencing the retention capacity of the wetlands. It is then assumed that $\partial N^{iw}/\partial W^i$ and $\partial N^{iw}/\partial N^{iL}$ are nonnegative, i.e., that the pollutant removal must increase, or at least remain the same, if the wetland area, W^i, or the pollutant load, N^{iL}, increase. The role of wetlands for nitrogen abatement has been debated among natural scientists for a period of a about 20 years, see [16] for a discussion of this in an economic context.

We wish here to focus on the role of quantification uncertainty in the transport through natural water systems and the resulting loads of pollutants from non-point, or diffuse land sources in to the coastal basins. We therefore assume that, on the one hand, the direct discharge loads, D^i, incur no uncertainty with respect to their impact on the coastal water. On the other hand, the pollutant transport through the natural water systems minus the wetlands' removal, N^i, is assumed to be predictable only under conditions of uncertainty, which is written as

$$N^i = N^i\left(N^{iL} - N^{iw}, \varepsilon^i\right), \tag{2}$$

where ε^i is the assumed additive stochastic pollutant transport term, which we assume has a zero mean and the variance σ^i. The transport of pollutant from a land area to the coast may take place along any, or all three, of the following pathways: (a) ground water flow discharging into the catchment stream network that discharges into the coastal water; (b) direct ground water flow into the coastal water; and (c) overland water flow to streams that discharge into the coastal water. All of these different transport pathways are difficult or impossible to describe and predict deterministically. Even though various modelling methods are possible, such as separation of hydrographs into different runoff components and estimation of the pollutant content of each component, all observation data available for the model construction, calibration and testing have limited support scales in time and space. The extrapolation that is required in both time and space, from the site and time specific hydrological observations to the large scale, long-term predictions that are relevant for applications to large coastal areas and their water quality management, will always be subject to uncertainty. One important reason for this uncertainty is the natural, and generally high and irregular, variability both in the temporal weather patterns that drive the water flow dynamics and in the highly heterogeneous spatial flow patterns though different subsurface (soil, aquifers) and surface (overland flow, streams) water systems. In order to extrapolate site and time specific observations of pollutant transport from land to coast with certainty, both temporal weather fluctuations and spatial flow patterns within a naturally heterogeneous catchment would need to be deterministically predictable, which is generally not the case (see, e.g. [17–23]). Various types of essentially random natural variability in space and time imply predictability limits and associated modelling/extrapolation uncertainties, which are the reasons why the pollutant transport from land to the coastal zone, through natural water systems, is considered to be a stochastic variable, ε^i.

A simplification is made in [2] by disregarding different locations of abatement measures within each drainage basin. Since the location of a non point emission source is quite likely to influence the load to the coastal water recipient, this simplification will imply an inefficiency in allocation of measures. This is a well known difficulty in the literature on non-point source regulation (e.g. [15]). We make here the simplifying assumption of uniform regulation of non-point sources within each drainage basin, the seriousness of which is determined by the relation between marginal costs of measures at different locations in the drainage basins and their marginal impacts on the coastal water recipients [25].

The pollutant load entering a certain coastal basin, i, is dispersed to other coastal basins and to the marine water. It is assumed that transports between coastal basins can be described by a coefficient matrix, where each element a^{ij} denotes the share of total load into the basin which is transported from basin i to j. For simplicity, these coefficients are assumed to be deterministic. The total pollutant load entering a coastal basin, T^i, is then written as

$$T^i = \sum_j a^{ji}\left(N^j + D^j\right). \tag{3}$$

The transports from the coastal basins to the marine water are assumed to be described by the coefficients m^i, which denotes the share of T^i that reaches the marine water.

It is also assumed that there exist one pollutant reduction measure for each type of load, and further that the reductions in pollutant loads can be associated with a continuous, increasing, and convex cost function in emission reductions from each source. The cost functions are then $C^{iW}(W^i)$, $C^{iL}(L^i)$ and $C^{iR}(R^i)$, where $L^i = (N^{iL'} - N^{iL})$, $R^i = (D^{i'} - D^i)$ and $N^{iL'}$ and $D^{i'}$ are pre-abatement emission levels, i.e., pollutant loads before any abatement measures have been taken.

Depending on the water quality in different coastal basins, it may be necessary to formulate different pollutant load targets for each coastal basin. We allow for this by formulating the cost minimisation problem with probabilistic pollutant load targets for each coastal zone, T^i*. However, we may also want to achieve large-scale improvements by restricting the total load of pollutants from all zones to the marine water, $M*$. The coastal zone manager is then assumed to choose the allocation of W^i, L^i, and R^i in different drainage basins which minimises total costs subject to probabilistic constraints on pollutant loads to each coastal zone and to the marine water according to

$$\text{Min} \sum_i \left(C^{iW}\left(W^i\right) + C^{iL}\left(L^i\right) + C^{iR}\left(R^i\right)\right) \tag{4}$$

$$\text{s.t.} \quad (1)\text{–}(3)$$

$$\text{Prob}\left(T^i \leqslant T^{i*}\right) \geqslant \alpha^i,$$

$$\text{Prob}\left(\sum_i m^i T^i \leqslant M^*\right) \geqslant \beta.$$

The solution to equation (4) is much simplified by replacing the constraint by its deterministic equivalent (see, e.g. [26]). Disregarding covariances between coastal basins, the constraints are rewritten as

$$\mu^i + K^{\alpha i}\left(\text{Var } T^i\right)^{1/2} \leqslant T^{i*},$$

$$\sum_i \mu^i + K^{\beta}\left(\text{Var } M\right)^{1/2} \leqslant M^*,$$

where $\mu^i = E\{T^i\}$ and $K^{\alpha i}$ and K^{β} represent the standard normal distributions, $\Phi^{\alpha i}$, where $\Phi(K^{\alpha i}) = \alpha^i$, and φ^{β}, where $\varphi(K^{\beta}) = \beta$. When $\alpha^i = 0.5$ for a coastal zone i we have that $K^i = 0$, which implies that the $\text{Var}(T^i)$ has no impact on the decision problem (4). As shown in, among others [11], this corresponds to a linear damage function in the monetary valuation of decreases in T^i*. However, a risk averse attitude or convex damage function is more likely, which implies that $\alpha^i > 0.5$. Then $K^{\alpha i} > 0$ and the consideration of the random impact will require a higher total pollution reduction requirement in coastal zone i and, hence, higher costs than when only expected outcomes are considered. The difference in minimum costs between the expected and chance-constrained outcomes depends on the chosen levels of α^i and β and the estimated $\mathrm{Var}(T^i)$ and $\mathrm{Var}(M)$, respectively.

However, not only the total costs are affected by the chosen specification of the decision problem, deterministic or probabilistic targets, but also the allocation of measures. This is most easily seen from differentiating equations (1–4) with respect to the different pollutant load reduction measures. Assuming that all covariances are zero we have that $\mathrm{Var}(M) = \sum_i (m^i)^2 \mathrm{Var}(T^i = \sum_i (m^i)^2 \sum_j (a^{ji})^2 \sigma^j$. The cost minimising levels of the reduction measures are then given by the first-order conditions

$$C^{iL}_{L^i} = \sum_j a^{ij}\big[\mu^i_{L^i}(\lambda^j + \gamma m^j) - \frac{a^{ij}}{2\sqrt{\sigma^i}}\sigma^i_{L^i}(\lambda^j K^\alpha j + \gamma K^\beta m^{j^2})\big], \tag{5}$$

$$C^{iW}_{W^i} = \sum_j a^{ij}\big[\mu^i_{W^i}(\lambda^j + \gamma m^j) - \frac{a^{ij}}{2\sqrt{\sigma^i}}\sigma^i_{W^i}(\lambda^j K^\alpha j + \gamma K^\beta m^{j^2})\big],$$

$$C^{iR}_{R^i} = \sum_j a^{ij}\mu^i_{R^i}(\lambda^j + \gamma m^j),$$

where

$$\mu^i_{L^i} = E\big[N^i_{L^i} - N^{iw}_{N^iL} N^{iL}_{L^i}\big] \leqslant 0,$$
$$\mu^i_{W\mathrm{supi}} = E\big[N^{wi}_{W^i}\big] \leqslant 0, \quad \mu^i_{R^i} = -1,$$

subindexes are partial derivatives, and γ and λ^i denote the Lagrange multipliers of the constraints in equation (5). They can be interpreted as the change in total costs associated with a marginal change in the constraint for coastal zone i. The left hand side of equation (5) are the marginal costs of each measure. The right hand sides measure the impacts on the pollutant targets. The higher impacts, more is used of the measure in question since the cost functions are increasing and convex in pollutant reductions.

We note, from the right hand sides of equation (5), that the consideration of coastal transports and targets, i.e., a^{ij} and λ^j, are likely to reallocate measures as compared to the case when the marine water target is included by only the marine transport coefficients, m^i. If, in a feasible solution, none of the coastal water targets is binding, the inclusion of coastal basin transport reallocates measures unless m^i and a^{ij} are the same for all coastal basins. Whether or not the consideration of coastal transports a^{ij} implies a higher total costs for a given marine water target depends on the relation between marginal costs of pollutant reductions to the coastal basin and the coastal water transport coefficients. For example, if the transports from a relatively low cost coastal basin, i.e., $a^{ii}m^i$, are high the consideration of these transports will decrease total costs. The reason is that the share of total pollutant transports from the low cost basin then increases.

According to the right hand sides of equation (5), the water target impacts of marginal changes in L^i and W^i are divided into four components: expected changes in coastal zone and marine pollutant load targets, and changes in respective variances in pollutant loads. When the latter two are zero, i.e., if changes in pollutant loads by these measures do not affect any variance, the first order conditions are reduced to the deterministic case where only expected impacts on coastal pollutant loads are included. If, on the other hand, the impacts are non zero, the variances can either increase or decrease from a marginal change in any of L^i and W^i. A decrease implies relatively higher impacts as compared to the deterministic case and vice versa.

The allocation of measures within drainage basin i is thus determined by the marginal costs, expected marginal decreases in pollutant loads, risk attitudes as expressed in the choices of α^i and β and the changes in variances of pollutant loads. Note also from the first condition that marginal changes in L^i negatively affects the impacts of wetland measures. The reason is that the pollutant removal effectiveness of wetlands is determined by upstream pollutant loads. Given a certain cost per ha of wetlands creation, a reduction in pollutant loads to the wetland implies an increase in costs of wetlands pollutant removal. The inclusion of wetlands thus increases the cost of marginal changes in upstream pollutant abatement measures.

In order to simplify subsequent calculations, we will in the sequel assume that the risk attitudes towards the different targets are the same expressed as the same probabilities for achieving the targets. This means that $K^{\alpha i} = K^{\alpha j} = K^\beta \equiv K^\alpha$. The first order conditions can then be written as

$$C^{iL}_{iL} = \sum_j (\gamma m^j + \lambda^j) a^{ij}(\mu^i_{L^i} - K^\alpha \zeta), \tag{5'}$$

$$C^{iw}_{iw} = \sum_j (\gamma m^j + \lambda^j) a^{ij}(\mu^i_{w^i} - K^\alpha \Psi),$$

$$\zeta = \frac{a^{ij}\sigma_{L^i}}{2\sqrt{\sigma^i}}, \qquad \Psi = \frac{a^{ij}\sigma_{w^i}}{2\sqrt{\sigma^i}},$$

and the first order condition for optimal use of measures reducing direct pollutant effluents, R^i, is unchanged.

3. Design of policy instruments

In this paper we analyse two types of policy instruments; pollution charges and market for pollution permits. The latter implies that permits are distributed to all involved firms, which then are allowed to trade permits. Under a charge system, each pollutant emission source is charged such that $(\sum_j a^{ij}\lambda^i + \gamma m^j) = t^i$, where t^i denotes the cost effective charge of coastal basin i. Each emission source is thus charged according to its weighted pollution impact on the targets. The larger the impact the higher is the charge. This can be seen by rearranging condition (5′) according to

$$t^{iL} = t^i\left(\mu^i_{L^i} - K^\alpha\zeta\right) = C^i_{L^i}, \tag{6}$$

$$t^{iR} = t^i\left(\mu^i_{R^i}\right) = C^{iR}_{R^i},$$

$$t^{iw} = t^i\left(\mu^i_{W^i} - K^\alpha\Psi\right) = C^{iw}_{W^i},$$

where t^{iL} is the charge of the non point source emission, t^{iR} is the charge of the point source emissions and t^{iw} corresponds to the charge of wetland emissions. Expression (6) thus states that the cost effective charges on emissions from the nonpoint, t^{iL}, wetland, t^{iw}, and point sources, t^{iR}, correspond to the effective charge at the target, t^i, times the respective impact of emission reductions on the coastal zone.

Under a permit market system each emission source is distributed permits, which in total corresponds to the pollutant load targets. If a competitive permit market is created, permit prices are established which reflects the marginal costs and impacts in the same way as the determination of charges in equation (6). However, in practice the establishment of competitive permit markets with cost effective equilibrium permit prices in different regions is far from a trivial matter (see, e.g. [27]). An alternative is then to trade permits at certain ratios. Choosing measures reducing direct deposition as the numeraire, the trading ratios between the pollutant mitigation measures are then determined by their relative impacts on targets according to

$$\frac{C^{iL}_{L^i}}{C^i_{R^i}} = \frac{\mu^i_{L^i} - K^\alpha\zeta}{\mu^i_{R^i}}, \tag{7}$$

$$\frac{C^{iw}_{w^i}}{C^i_{R^i}} = \frac{\mu^i_{w^i} - K^\alpha\Psi}{\mu_{R^i}}. \tag{8}$$

The left hand sides of equations (7) and (8) reflect the relation between marginal costs and the right hand sides that between impacts on the pollutant load targets. Similar results are obtained in [14] where only non point and point sources are included. The addition of wetlands, the effectiveness of which depends on the load from upstream non point source, implies a decrease in the trading ratio in equation (7) since a marginal reduction in their emissions reduces wetlands' impacts on the water targets.

The right hand sides of equations (7) and (8) reveal the difference in impacts on pollutant load targets between the measures changing non point source emissions and point source emissions, respectively. The measures L^i and wetlands W^i affect expected load to the coast and also the variance. Both these impacts are determined by combined hydrological and biogeochemical processes affecting pollutant transports in the drainage basin i. The impact of the measure reducing the effluents directly into the coastal water, R^i, occurs only through the change in expected load. Under deterministic specification of the coastal pollution target the impact on the variances is not included and, for given marginal costs, the allocation of these measures is determined by their impacts on the expected loads. When determining the impacts of deterministic versus stochastic specification on pollutant targets on the allocation of these three measures we can thus look at the role of K^α, ζ and Ψ.

The use of L^i and W^i are increased relative to that of R^i when the variance in pollutant load decreases from marginal increases in L^i and W^i, respectively. That is, if a measure has the ability, not only to reduce expected coastal load, but also the variance this measure has a cost advantage relative to a measure that has no or increasing impact on the variance. One example is provided by [3], who showed the potential of wetlands to reduce the variance by acting as a sink for nitrogen from non-point sources, mainly agriculture, where the variance in the pollutant outload from the wetland is lower than that of the inload. This impact of measures changing the variance in pollutant transports is enhanced for higher probabilities of achieving a certain target, i.e., for higher K^α.

4. Application to Himmerfjärden

From the theoretical sections 2 and 3, we identify three classes of data needed for calculating cost effective allocation of abatement measures: (i) pollutant reduction potential from different emission sources and associated abatement costs, (ii) impact of combined biological and chemical processes on the pollutant transports in the drainage basin from emission sources to the coastal waters, and (iii) information on transports and transformation of pollutant within and between coastal basins and to the marine waters. Unfortunately, we do not find any estuary or other coastal water with sufficient data for all three classes. To our knowledge, Himmerfjärden is unique in this respect since measurements have been made of the ecological impacts and transports of nitrogen for a period of approximately 20 years [1], i.e., data of class (iii). Therefore, it is interesting to apply the theoretical model to Himmerfjärden, although we still suffer data of high quality on costs of abatement measures and on nitrogen transport in its catchment region.

The drainage basin of Himmerfjärden is located about 60 km south of Stockholm. It covers an area of 1286 km^2 which corresponds to 5.5 times the water area. It constitutes a connection between the third largest lake of Sweden, Mälaren, and the Baltic Sea. Mean depth of the estuary is 17 m and it can be divided into four coastal basins as defined by sea bottom thresholds [28].

Like many other estuaries, Himmerfjärden suffers from ecological damages due to eutrophication, i.e., excessive loads of nutrients. According to (1), the limiting nutrient is nitrogen. Decreases in nitrogen load thus reduce damages from eutrophication. This has been recognised, not only for the Himmerfjärden estuary, but also for the entire Baltic Sea already in the 1970s. A ministerial declaration was made to reduce the load to the Baltic Sea by 50% [2]. It was not clear, however, if this reduction refers to reduction in the loads to the coastal waters or to the marine waters. In the following, we will therefore calculate costs of different probabilities of achieving 50% reduction in the nitrogen loads for both these options. We then consider only the nitrogen abatement options available within the catchment region. Since only a small fraction of atmospheric deposition of ammonium and nitrogen oxides within the catchment region are emitted within the catchment region (approximately 10–15%, e.g. [10]), these type of emission sources are excluded from the calculations. The leaching of manure nitrogen into soil and water is, however, accounted for.

4.1. Nitrogen transports in Himmerfjärden and its catchment area

There are four categories of nitrogen sources to Himmerfjärden; arable and forest land, atmospheric deposition, and household sewage. The contributions of nitrogen load from these sources to the estuary have been calculated by means of geographical information system data and a nitrogen leaching model, which is briefly presented in the following.

The entire drainage basin of Himmerfjärden is divided into nine different drainage basins characterised by differences in climate, fertiliser regimes, farming season and crop production. For each region, leaching coefficients are estimated for a combination of nine different crops, three types of soil and two different fertiliser regimes. Soils were further separated into three groups with regard to their particle size: sandy clay, light clay, and stiff clay. Due to variations in weather, leaching coefficients of a normalised year are used.

Information on nitrogen leaching from atmospheric deposition and forest areas are obtained from [30]. The estimated loads to the different coastal basins are then as presented in table 1.

The table reveals that the nitrogen emission in basin 2 accounts for about 2/3 of total nitrogen emission in the area. The results also show that direct discharges into the water streams by sewage treatment plants which are located at a coast account for approximately 60% of the total nitrogen emission.

Table 1
Nitrogen sources in the coastal basins of Himmerfjärden, tons of N/year.

Drainage basin	Atmospheric	Forest land	Arable land		Sewage land	Total
			Fertilizers	Manure		
1	12	21	32	19	0	84
2	8	24	35	22	572	661
3	45	50	53	34	16	199
4	3	8	9	6	0	26
Total	67	103	130	82	588	969

Table 2
Nitrogen loads to coastal waters and the Baltic Sea, tons of N/year.[a]

Target	Basin 1	Basin 2	Basin 3	Basin 4	All basins	Baltic Sea
Drainage basins	53	628	130	12	823	
Coastal transports	55	377	102	8	542	281

[a] Source: calculations from table 1 and table A.1 in appendix.

However, the estimated nitrogen emissions presented in table 1 do not correspond to the actual nitrogen loads into the coastal waters. Depending on the different water and nitrogen pathways from land to coast (ground water, overland flow, streams), the combined hydrological and biogeochemical conditions along these pathways, and the different pathway lengths, nitrogen will be removed from the mobile aqueous phase to different degrees. According to [31], the nitrogen retention in the Himmerfjärden area varies between 0.25 and 0.5. That is, the share of nitrogen leaching within the drainage basin that enters any of the coastal basins in the estuary ranges from 0.5 to 0.75. In the following, 0.372 of nitrogen leaching from nonpoint sources (table 1) is assumed to be removed from the mobile water on its various pathways to the coast. Accounting for this removal process in the transport of nitrogen from non-point sources does in turn imply that the role of point sources is accentuated. Specifically, the point sources then account for almost 2/3 of the total nitrogen load to the coastal water.

The need for nitrogen reduction to different coastal zones is currently scientifically unclear (1). It is recognised, however, that the reduction requirements to the Baltic Sea are larger than for the coastal zones presented in table 1. In order to calculate the load from the different coastal zones to the Baltic Sea we need information on nitrogen transports between the coastal zones and to the Baltic Sea. Based on such a matrix (see table A.1 in the appendix), calculations of loads to the coastal waters and the Baltic Sea are made. The results are presented in table 2, where the row "Coastal zones" denote the load from the drainage basins to the coastal waters and "Coastal transports" is the net nitrogen load to each coastal basin and to the Baltic Sea when accounting for all transports between coastal basins.

Considering the nitrogen transports between coastal basins, the final load to the Baltic Sea corresponds to about 1/3 of the nitrogen load entering the coastal basins. This is a reflection of the coastal zones' ability to act as nitrogen sinks.

4.2. Estimated minimum cost of nitrogen reductions

Since it is unclear how to determine nitrogen load targets for different coastal zones we estimate costs for reductions from Himmerfjärden to the Baltic Sea. In principle, two targets can then be identified. One is the currently applied principle of nitrogen reductions from the drainage basins to the coastal zones. If however, the aim is to improve the conditions of the Baltic Sea a more appropriate target is to consider the nitrogen transports between coastal zones and the resulting impact on the Baltic Sea. Thus, our two targets for minimum cost estimates are reductions from the levels of either 281 or 823 tons of nitrogen as shown in table 2. Comparing with the analytical discussion, we thus have no coastal zone targets but instead two variants of total load targets, with and without consideration of coastal nitrogen transports. The first case makes use of the transport matrix A1 in the appendix and the second assumes that $a^{ii} = m^i = 1$. In the following, results are presented for both these target formulations. We also consider reductions by 50%, which is based on a ministerial declaration from 1987 [2].

Three types of nitrogen removal options within the catchment region are included; point sources, changes in agriculture practices, and construction of wetlands. One point source mitigation measure is included, improvement of the nitrogen cleaning capacities at the sewage treatment plants. It is assumed that the cost corresponds to SEK 13/kg (8.74 SEK = 1 Euro, 18 June 1999) N reduction from sewage plants [33]. Two types of nonpoint source emission reduction measures are considered: reductions in the use of nitrogen fertilisers and cultivation of catch crops. Catch crops are sown at the same time as the ordinary crop but continues 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 [10]. Costs of nitrogen fertiliser reductions in each drainage basin are calculated as associated losses in profits. These profits are, in turn, calculated as changes in producer surplus from estimated nitrogen demand functions (see [33]). The cost of wetlands is estimated as the opportunity costs of land for wetland purposes. It is then assumed that wetlands are located only on arable land and the cost corresponds to foregone profits, which varies between SEK 1000 and 5000 per ha in the region depending on cultivated crop [34]. It is here assumed that the cost amounts to SEK 3000/ha wetland. Another important factor is the nitrogen sink capacity of wetlands, which varies considerably between different Swedish regions. It is here simply assumed that the retention capacity corresponds to 0.5 of the nitrogen load to the wetland. We also limit the area of land suitable for wetland restoration to 10% of the arable land.

Given these assumption, we calculate how minimum costs for achieving the two targets are related to changes in the standard deviation, measured as coefficient of variation, to changes in probabilities of achieving the targets, and to different nitrogen abatement capacities of wetlands. In figure 1, results are presented for a 50% nitrogen reduction under the two targets, *CW* (to coastal waters) and *BS* (to the Baltic Sea), respectively, for different levels of the coefficient of variation (cv). It is further assumed that the desired probability of achieving the target of 50% reduction is 0.8

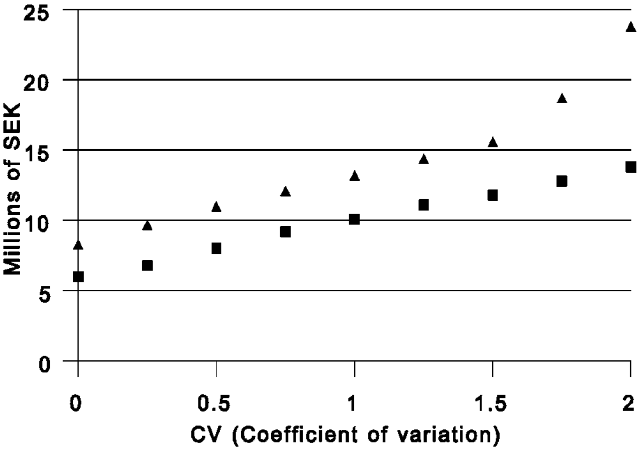

Figure 1. Costs of 50% N reduction pr. = 0.8; loads (■) BS and (▲) CW.

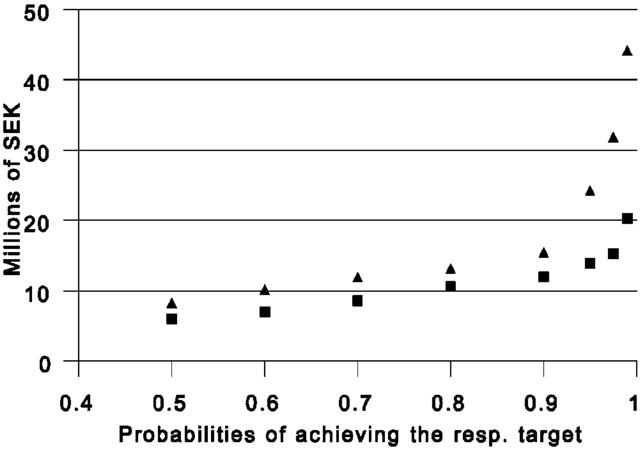

Figure 2. Costs of 50% N reduction, $cv = 1$; loads (■) BS and (▲) CW.

and that wetlands as removal options neither increase nor decrease the variance in the load to the coastal waters.

We can compare the results under deterministic formulation with the stochastic alternatives by relating the minimum costs of 50% nitrogen reduction to the Baltic Sea or the coastal zones when the standard deviation is zero. Minimum costs then amount to 6.0 and 8.3 millions of SEK, respectively. The reason why the nitrogen reductions to the to the Baltic Sea are less expensive than to the coastal waters is that the role of low cost measures at point sources is increased. Depending on the variance in drainage basin loads to the coastal zones the cost of probabilistic targets can be about three times as expensive as the corresponding deterministic target.

In figure 2, costs of 50% nitrogen reductions to the coastal waters for alternative probabilities of achieving the target is presented when the coefficient of variation is 1.

The results illustrated in figure 2 show considerable increase in costs of achieving a 50% nitrogen reduction to the coastal waters when the probability of achieving the target exceeds 0.9.

However, the impact of wetlands' nitrogen removal on the variance in nitrogen load to the coastal waters is far from a solved scientific issue. As demonstrated in the foregoing

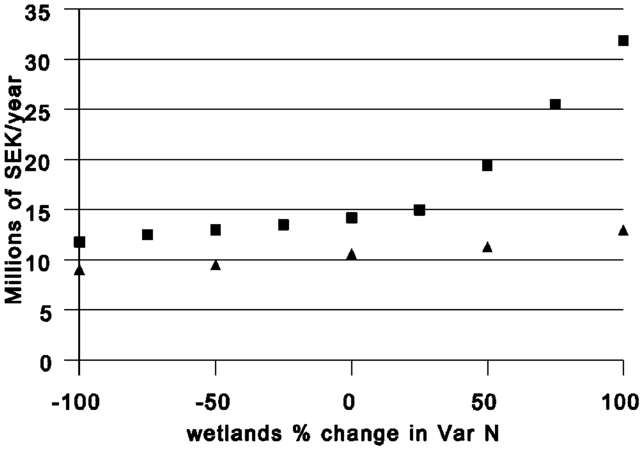

Figure 3. Costs and wetlands' impact on variance; loads (■) CW and (▲) BS.

section 2, the costs increase when wetlands imply higher variance and decrease when the variances to the coasts are reduced. As shown in figure 3, the difference in costs from wetlands impact on the variance can be quite large in magnitude when the coefficient of variation in the nitrogen load entering the wetlands is 1 and the chosen probability of achieving 50% nitrogen reductions is 0.8.

The horizontal axis expresses different impacts of wetlands' activities on the variance in nitrogen load to the coastal waters as measured in per cent. At one extreme, wetlands reduce the variance by 100%. Depending on the availability of land suitable for wetlands, the total variance could then vanish. In the case of Himmerfjärden, this does not occur at a land area that corresponds to 10% of the arable land. The costs show higher sensitivity to increases in the variance due to wetland nitrogen removal. If the variance is doubled the costs increase by three times.

4.3. Charges and permit trading ratios

Similarly to the results presented in the section above, the cost effective charge increase as standard deviations and probabilities for achieving the target increase. The cost-effective charge is then higher for target loads concerning the Baltic Sea than for targets on loads to the coastal waters. The simple explanation is the less impact on the Baltic Sea per unit of nitrogen reduction from all sources. The charges under deterministic conditions for the coastal water and the Baltic Sea targets are then SEK 15 and SEK 34/kg N reduction, respectively. Under stochastic conditions the charges increase more than ten times when the coefficient of variation exceeds 2.

For each cost-effective charge at either of the two targets, there is a set of charges for the nitrogen sources in each drainage basin. As shown in section 3, these charges are calculated as the impacts on the target times the cost-effective charge. Under a permit market system, the cost effective trading ratio between a point and nonpoint source is also determined by the relative impacts on the target. In table 3 we present charges for different nitrogen sources in

Table 3
Charges at nitrogen emission sources for 50% N reductions, $cv = 1$ and $pr. = 0.8$.

Basin	Load to the coasts (SEK/kg N emission)			Load to the Baltic Sea (SEK/kg N emission)		
	Non-point	Wetland	Point	Non-point	Wetland	Point
Basin 4	21	27	–	1.8	2.3	–
Basin 3	22	27	60	6.3	7.7	17
Basin 2	25	32	60	22	27	52
Basin 1	29	37	–	26	31	–

Table 4
Permit trading ratios for 50% N reductions, $cv = 1$ and $pr. = 0.8$.

Basin	Load to the coasts (SEK/kg N emission)			Load to the Baltic Sea (SEK/kg N emission)		
	Non-point	Wetland	Point	Non-point	Wetland	Point
Basin 4	2.8	2.2	–	28.8	22.6	–
Basin 3	2.7	2.2	1	8.2	6.7	3.1
Basin 2	2.4	1.9	1	2.3	1.9	1
Basin 1	2.1	1.6	–	2	1.7	–

the coastal basins when the standard deviation is equal to the mean value, i.e., $cv = 1$, and the probability of achieving the target is 0.8. The calculated cost effective charge for a 50% nitrogen reduction to the coastal waters is SEK 62 and that of a corresponding decrease to the Baltic Sea amounts to 105.

As shown by the results in table 3, there is a significant difference in charge levels on non-point source emissions and wetlands for basins 4 and 3 depending on choice of nitrogen target. The reason is the small impact from these basins on the Baltic Sea. We also see the higher charges on point sources which, as shown in section 3, is explained by their larger impacts on both targets. The lack of charge on point sources in basin 1 is due to the non-existence of point-source emissions in this region.

Assuming a permit market for the entire region, the trading ratios between nitrogen emission sources reflect the impacts of emission source in different drainage basins and coastal zones. In table 4 trading ratios are presented where we have chosen point sources in basin 2 as the numeraire. That is, each number in the table shows how many permits a point source in basin 2 requires in exchange for one permit.

The trading ratios measure the amount of nonpoint source and wetland permits, respectively, that can be substituted for 1 point source permit in basin 2 without changing the impact on the target. For example, in basin 4 approximately three nonpoint source permits are required in exchange for one point source permit when the target is to reduce nitrogen loads to the coastal waters. However, almost 30 permits are required if instead the target is to reduce the load to the Baltic Sea by the same percentage. It is noteworthy that the trading ratios are changed considerably when disregarding the stochastic impacts. Then, one point source permit requires about eight nonpoint source permits (see table A.2 in the appendix) when the target is to reduce nitrogen loads to coastal waters.

5. Conclusions

The main purposes of this paper has been to develop a tool for making analytical and empirical investigation of the role of coastal transports and wetlands as a pollutant abatement option for cost effective coastal management under conditions of stochastic transports of pollutant in the catchment areas. The stochastic component arises here as the result of uncertainties that are associated with quantification of land based loads, originating from non-point emission sources.

The theoretical analysis showed that inclusion of stochastic pollutant transports increases total cost for achieving predetermined targets when the chosen probabilities for achieving the targets exceed 0.5. Whether or not the consideration of coastal transport increases total cost for a given marine water target depends on the relation between the pollutant sink capacities of the coastal basins and the costs of reducing pollutant loads to the basins. The inclusion of wetlands as a mitigation measure has two impacts as compared to considering only non point and point sources. One is that the effect of pollutant emission reductions at non point sources is reduced since this implies less nitrogen abatement effectiveness of wetlands. The other implication is the eventual impact of wetlands on the variance of nitrogen loads to the recipient. When the introduction of wetlands reduces (increases) the variance, the optimal use of this measure increases (decreases) as compared to the allocation of measures under deterministic conditions. Thus, an increase in the variance in the load to the recipient as a result of wetland construction can offset the negative impact of non point source emission changes on wetland nitrogen abatement. Since, in theory, the optimal design of policy instruments accounts for differences in measures' impacts on pollutant load targets, the emission charges or trading ratios on a permit market are also determined by these two factors.

The application of the theoretical model, or tool, to Himmerfjärden showed that the consideration of coastal transports reduces total costs by at least 20% for a given nitrogen reduction to the marine water. The reason is that the basins with low cost measures have relatively larger nitrogen exports, or, equivalently, lower nitrogen sink capacity, than high cost basins. The inclusion of stochastic pollutant transports indicated large differences in costs depending on assumptions about the chosen probability of achieving a predetermined target in nitrogen reductions to the marine water and about the nitrogen load variance. For probabilities exceeding 0.9, total costs can be more than twice as high than the costs when the stochastic component is disregarded. The application also showed that the impacts of wetlands' ability to affect variation in nitrogen load to the coast on total costs is minor when they reduce total variation in the nitrogen load to the coastal zone. On the other hand, if instead wetlands increase the variance, the increases in total costs are significant.

An interesting empirical result is the relatively large differences in effective charges and permit trading ratios required for costs effective achievement of a given water target with and without the inclusion of coastal nitrogen transports. When coastal transports are included, there is a considerable differentiation in charges and trading ratios between coastal basins. The effective charge of a non point emission source can be ten times as high when coastal transports are included. Under deterministic pollutant transports, this difference is even larger. The relatively uniform charges and trading ratios when disregarding coastal transports is likely to facilitate the implementation and monitoring of the policies. A differentiated policy scheme associated with the different nitrogen sink capacities in the coastal basins is then worth while only if the transaction costs of enforcing a differentiated policy scheme are lower than the gains obtained from lower costs of achieving the water target.

Needless to say, the theoretical tool and its application to Himmerjfärden are carried out with several simplifying assumptions. The only type of uncertainty considered in the theoretical part is the stochastic pollutant transports by surface and subsurface water in the drainage basin. In practice, there are a number of additional uncertainties associated with regulating water quality, such as asymmetric information among the regulator and the firms, and stochastic pollutant transports and biological impacts in the coastal basins. 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. The impact on costs of uncertainties in nitrogen transports and biological effects depends on their correlation.

Although it would be theoretically feasible to account for the simultaneous existence of several types of uncertainties, it is a real challenge to obtain necessary data for empirical calculations. This is demonstrated by the many assumptions necessary for calculating costs of nitrogen reductions to Himmerfjärden, which is regarded as a relatively well investigated estuary with regard to pollutant transports.

However, in spite of these theoretical and empirical shortcomings, the empirical results in this paper show that costs may change considerably when accounting for stochastic pollutant transports and including coastal nitrogen transports. It could therefore be worthwhile to make further efforts to collect measurement on pollutant transports and quantification uncertainty, such as both mean and variances in pollutant loads, when designing cost effective programs for pollutant reductions to coastal and marine waters.

Acknowledgement

We are much indebted to Hans Andersson, Department of Economics, SLU, and an anonymous referee for valuable comments. Swedish Council of Agriculture and Forestry Research and the Fund for Strategic Environmental Research are gratefully acknowledged for financial support.

Appendix: Tables

Table A.1
Coefficient matrix, share of total inload to basin i which is transported to basin j and to the Baltic Sea.[a]

	Basin 1	Basin 2	Basin 3	Basin 4	Baltic Sea
Basin 1	0.60	0.29			
Basin 2	0.10	0.53	0.32		
Basin 3	–	0.27	0.21		0.51
Basin 4				0.60	0.40

[a]Source: calculations based on [28].

Table A.2
Charges for point and non-point emission for 50% nitrogen reductions to the coastal water and Baltic Sea, respectively, under deterministic nitrogen transports.

Basin	Load to coastal water (SEK/kg N red.)			Load to the Baltic Sea (SEK/kg N red.)		
	Non-point	Wetland	Point	Non-point	Wetland	Point
Basin 4	1.6	2.1	–	0.2	0.3	–
Basin 3	1.7	2.2	15	0.6	0.8	5.6
Basin 2	2.10	2.7	15	2.5	3.2	17.7
Basin 1	2.4	3.1	–	2.2	2.8	–

Table A.3
Trading ratios for point and non-point emission sources for 50% nitrogen reductions to the coastal water and Baltic Sea, respectively, under deterministic nitrogen transports, point source in basin 2 is numeraire.

Basin	Load to coastal water (SEK/kg N red.)			Load to the Baltic Sea (SEK/kg N red.)		
	Non-point	Wetland	Point	Non-point	Wetland	Point
Basin 1	9.3	7.1	–	88.5	59	–
Basin 2	8.9	6.8	1	29.5	22	3.2
Basin 3	7.1	5.6	1	7.1	5.5	1
Basin 4	6.3	4.8	–	8.1	6.3	–

References

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

[2] Helcom, The Baltic Sea joint comprehensive environmental action programme, Baltic Sea Environmental Proceedings, No. 48, Helsinki, Finland (1993).

[3] O. Byström, H. Andersson and I.-M. Gren, *Economic Criteria for Restoration of Wetlands under Uncertainty*, Beijer Discussion Papers Series No. 114, Beijer International Institute of Ecological Economics, Royal Swedish Academy of Sciences, Stockholm (1998).

[4] I.-M. Gren, The value of investing in wetlands for nitrogen abatement, European Review of Agricultural Economics 22 (1995) 157–172.

[5] I.-M. Gren, T. Söderqvist and F. Wulff, Nutrient reductions to the Baltic Sea: Economics and ecology, Journal of Environmental Management 51 (1997) 123–143.

[6] K.-G. Mäler, International environmental problems, in: *Economic Policy Towards the Environment,* ed. D. Helm (Blackwell, Oxford, 1991).

[7] S. Barett, Self-enforcing international environmental agreements, Oxford Economic Papers 46 (1994) 878–894.

[8] G. Chichilnisky and G. Heal, Global environmental risks, Journal of Economic Perspectives 7(4) (1993) 65–68.

[9] V. Kaitala, K.-G. Mäler and H. Tulkens, 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) (1995) 325–343.

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

[11] B. Beavis and M. Walker, Achieving environmental standards with stochastic discharges, Journal of Environmental Economics and Management 10 (1983) 103–111.

[12] J.S. Shortle, The allocative efficiency implications of water pollution abatement cost comparisons, Water Resources Research 26(5) (1990) 793–797.

[13] K. Segerson, Uncertainty and incentives for nonpoint pollution control, Journal of Environmental Economics and Management 15 (1988) 88–98.

[14] A.S. Malik, D. Letson and S.R. Crutchfield, Point/nonpoint source trading of pollution abatement: Choosing the right trading ratio, American Journal of Agricultural Economics 75 (1993) 959–967.

[15] J.S. Shortle, R. Horan and D. Abler, Research Issues in nonpoint pollution control, Environmental and Resource Economics 11(3–4) (1998) 571–585.

[16] O. Byström, The nitrogen abatement costs in wetlands, Ecological Economics 26 (1998) 321–331.

[17] V. Cvetkovic, A.M. Shapiro and G. Dagan, A solute flux approach to transport in heterogeneous formations, 2, Uncertainty analysis, Water Resources Research 28 (1992) 1377–1388.

[18] G. Destouni, Prediction uncertainty in solute flux through heterogeneous Soil, Water Resources Research 28 (1992) 793–801.

[19] G. Destouni, Stochastic modelling of solute flux in the unsaturated zone at the field scale, Journal of Hydrology 143 (1993) 45–61.

[20] G. Destouni and W. Graham, The influence of observation method on local concentration statistics in the subsurface, Water Resources Research 33 (1997) 663–676.

[21] A. Marani, R. Rigon and A. Rinaldo, A note on fractal channel networks, Water Resources Research 27 (1991) 3041–3049.

[22] A. Rinaldo, A. Marani and A. Bellin, On mass response functions, Water Resources Research 25 (1989) 1603–1617.

[23] X. Foussereau, W. Graham, A. Aakpoji, G. Destouni and P.S.C. Rao, Stochastic analysis of transport in unsaturated heterogeneous soils under transient flow regimes, Water Resources Research 36 (2000) 911–921, 22.

[24] A. Rinaldo, A. Marani and R. Rigon, Geomorphological dispersion, Water Resources Research 27 (1991) 513–525.

[25] R. Brännlund and I.-M. Gren, Costs of differentiated and uniform charges on polluting inputs: An application to nitrogen fertilizers in Sweden, in: *Environmental Economics and Regulation*, eds. M. Boman, R. Brännlund and B. Kriström (Kluwer Academic, Dordrecht, 1998).

[26] A. Charnes and W.W. Cooper, Chance-constrained programming, Operations Research 11(1) (1964) 18–39.

[27] T. Tietenberg, Design lessons from existing air pollution control systems: The United States, in: *Property Rights in a Social and Ecological Context – Case Studies and Design Applications*, eds. S. Hanna and M. Munasinghe (The World Bank, Washington, DC, 1995).

[28] A. Enqvist, Vatten- och närsaltutbyte i hela Himmerfjärden, in: *Himmerfjärden. Changes in a Nutrient Enriched Coastal Ecosystem* (In Swedish with an English summary), eds. R. Elmgren and U. Larsson Swedish Environmental Protection Agency, Report No. 4565 (1997).

[29] H. Johnsson and M. Hoffmann, Nitrogen Leaching from Swedish Arable Land, Report 4741, Swedish Environmental Protection Agency, Stockholm, Sweden (1997).

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

[31] B. Arrheimer, M. Brandt, G. Grahn, R. Roos and A. Sjöö, *Modelled Nitrogen Transport, Retention and Source Apportionment for the South of Sweden*, Swedish Meteorological and Hydrological Institute, Norrköping, Sweden (1997).

[32] Björklund, Ltd., *The Fluid beds at Himmerfjärdsverket*, Box 858, 183 22 Täby, Sweden (1998).

[33] I.-M. Gren, G. Destouni and O. Byström, Costs and instruments for management of stochastic water pollution, Working Paper Series no. 3, Department of Economics, Swedish University of Agricultural Sciences, Uppsala (1998).

[34] SLU info, *Områdeskalkyler,* Department of Economics, Swedish University of Agricultural Sciences, Uppsala, Sweden (1998).