Optimal Control of Nutrient Pollution in a Coastal Ecosystem: Agricultural Abatement versus Investment in Wastewater Treatment Capacity

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Optimal Control of Nutrient Pollution in a Coastal Ecosystem: Agricultural Abatement versus Investment in Wastewater Treatment Capacity*

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Abstract

We examine in a dynamic framework how public resources should be allocated to small-scale water protection efforts in agriculture or alternatively to investments in large-scale waste water treatment plants to control point source loads. The building of waste water treatment capacity is characterized by high set-up costs as compared to the operating costs. We determine the optimal timing of investment, the rate of nutrient load reduction from point versus non-point sources, and the optimal switching policies from control of non-point pollution only to control of both non-point and point sources. The results of the analytical model are illustrated with simulation of optimal abatement policies for the Finnish coastal waters in the Gulf of Finland.

Key words: nonpoint-source pollution, point-source pollution, timing of investment

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1 Introduction

Nutrient pollution is a serious environmental problem in many coastal ecosystems. Excessive nutrient concentrations cause eutrophication which manifests itself through increased water turbidity and algae production, oxygen deficiency in bottom sediments and changes in biodiversity (Gabric and Bell, 1993). In Europe the most heavily loaded coastal areas show symptoms of severe eutrophication: toxic algae outbreaks occur during the warm summer months and filamentous algae cover the seabed in coastal areas (see for example Ærteberg et al., 2001). Eutrophication results in damages both directly, and through lost value of fisheries and recreational activities. Valuation studies have attributed significant economic benefits to improving the state of eutrophied coastal zones (see for example Söderqvist and Scharin 2000, Söderqvist 1996, Markovska and Zylicz, 1999).

Eutrophication can be reduced through curtailing nutrient loading. The choice of measures is not straightforward - the costs of policies aiming at reducing nutrient loads vary substantially due to both economic and biophysical characteristics of nutrient sources. Many environmental assessments identify agriculture as the major cause of surface quality problems in developed countries (Shortle & Abler, 2001). For example in the Nordic countries, municipal and industrial nutrient loads have been reduced significantly during the last few decades but due to intense farming technologies the agricultural sector remains a significant source of eutrophying nutrients (Turner et al., 1999). Despite the relative ease of controlling pollution from point sources, poorly processed urban and industrial wastewaters still are a significant source of nutrient loading in particular in less developed regions because of lack of funding for high-cost, lumpy investment. For example in the Gulf of Finland, the most eutrophied sub-basin of the Baltic Sea, some 30 % of urban wastewaters from the densely populated St. Petersburg region enter the sea without treatment.

When diverse sources contribute to the accumulation of nutrients, an optimal policy to reduce nutrient loading has to account for the different characteristics of the nutrient sources and loading processes and balance the nutrient load reduction targets for the various sources accordingly. Previous studies analyzing the reduction of nutrient loads from both agricultural land and municipal point sources have applied a static framework to study the trade-offs between agricultural abatement and wastewater treatment assuming that abatement technology is already in place (e.g., Elofsson 2003, Malik et al. 1993). In reality removing nutrients from municipal wastewater requires an irreversible initial investment to set up wastewater treatment facilities and infrastructure

to transfer wastewater from households to the treatment facilities. Agricultural abatement on the other hand takes place through reversible small scale measures such as changes in fertilizer use, manure spreading and tillage practices. Moreover, as nutrients accumulate in the sea only slowly over time, the problem is essentially a dynamic one. Previous dynamic analyses on eutrophication have focused on agricultural nutrient loading. Hart and Brady (2002) and Hart (2003) study the impacts of alternative policy goals and time-lags of abatement due to upstream and downstream measures. Nævdal (2002) also used a dynamic framework to examine optimal regulatory policies when thresholds effects are present so that the eutrophication process is characterized by discrete jumps in the state variable. However, these dynamic models consider neither abatement measures nor costly investments required in municipal point sources.

This paper builds and solves a dynamic model that explicitly accounts nutrient loading from arable land and from municipal point sources, and for the irreversible investment required to establish wastewater treatment facilities for reducing nutrient loads from municipal point sources. Data for the Gulf of Finland are used to calibrate the model. We examine how public funding should be allocated between investments in large-scale wastewater treatment facilities and the operation of such facilities, and alternatively small-scale water protection efforts in agriculture. We conjecture that if there is uncertainty over the future cost of the investment, the opportunity cost of investing can be large. Consequently, uncertainty makes controlling municipal point sources less attractive relative to abatement of agricultural loadings. We focus on the following questions: Under what conditions should investment in wastewater treatment facilities be undertaken, and what determines the optimal time? How are the reductions in agricultural and municipal nutrient loads balanced in case that investment is undertaken? and How does the optimal agricultural abatement policy change once wastewater treatment facilities are established?

The paper is organized as follows. Section 2 presents the theoretical model, and Section 3 discusses the empirical work performed to calibrate the model. Section 4 characterizes the optimal policy and discusses its implications. Section 5 concludes.

2 The model

Consider a coastal zone that receives eutrophying nutrients via runoffs from agricultural land and municipal wastewater discharges. We focus on the solution of an environmental planner who seeks to minimize the environmental damages from nutrient accumulation through reducing

nutrient loading. If nutrients in municipal wastewater are to be removed, there is a necessary outlay on the establishment of a wastewater treatment facility at the initiation point of those operations. Reducing agricultural nutrient loads does not involve set-up costs. There are thus two potential phases of nutrient load reduction: Prior to undertaking the investment outlay, only agricultural nutrient loads can be controlled. If the capital outlay is incurred and a wastewater treatment facility is established, nutrient loads from both agricultural and municipal sources can be reduced.

Let $t = 1, 2, ...$ index the period. The agricultural nutrient load given profit maximizing production is denoted by L_A^* . The agricultural nutrient load can be reduced at the expense of agricultural profits by varying agricultural inputs. With a load reduction $R_A^t \leq L_A^*$,the agricultural nutrient load entering the ecosystem in period t is given by $L_A^t = L_A^*$ -R_A^t. The costs of agricultural abatement are given by $C_A\left(R_A^t\right) = \pi\left(L_A^*\right) - \pi(L_A^t)$, with $C_A'\left(R_A^t\right) > 0$ and $C_A''\left(R_A^t\right) \geq 0$. This cost structure follows from the standard assumption that agricultural profits are increasing and concave in the nutrient leachates. Let L_W^* denote the municipal nutrient load in the absence of wastewater treatment. The municipal load is largely determined by population size, which by assumption remains constant. For simplicity, the size of the investment required to set up wastewater treatment facilities is also fixed. Thus, the investment decision is a discrete choice $I^t \in \{0, K\}$.The size of the investment does not depend on the rate of cleaning wastewater. Further, it has no impact on the unit cost of cleaning wastewater. While these assumptions are probably an oversimplification, they illustrate the principle of having to incur a capital outlay in order to reduce nutrient loads from a point source. Let $t_1$denote the time of investment: an investment $I^{t_1} = K$ initiates the construction of wastewater treatment facilities in period t_1 but delays may occur in bringing the facilities online. The indicator function

$$\Delta^t = \begin{cases} 0 & t \leq t_1 \\ 1 & t \geq t_1 + 1 \end{cases} \tag{1}$$

takes up value 1 if the wastewater treatment facility is online, and 0 otherwise. Once a wastewater treatment facility is operational, the municipal nutrient load can be reduced at the rate $R_W^t \leq L_W^*$. The municipal nutrient load is then given by $L_W^t = L_W^*$ -R_W^t. The costs of nutrient removal are denoted by $C_W\left(R_W^t\right)$, with $C_W'\left(R_W^t\right) > 0$ and $C_W''\left(R_W^t\right) \geq 0$.

The stock of nutrients increases as agricultural or municipal nutrient loads, L_A^t or L_W^t, enter the ecosystem. The stock of nutrients, N, changes from one period to the next as follows:

$$N' = \begin{cases} f\left(N, L_A^*, L_W^*, R_A\right) & \text{if } \Delta = 0 \\ f\left(N, L_A^*, L_W^*, R_A, R_W\right) & \text{if } \Delta = 1 \end{cases} . \tag{2}$$

Finally, environmental damages are a function of accumulated nutrients, $D\left(N^t\right)$, with $D'\left(N^t\right) > 0$ and $D''\left(N^t\right) \geq 0$.

Having outlined the basic economic, ecological and technological relationships for the coastal ecosystem of concern, we next state the two-phase nutrient load reduction model. The environmental agency seeks to minimize the sum of environmental damages from nutrient accumulation and the costs of reducing agricultural and municipal nutrient loading. The problem entails determining optimally the rate of agricultural abatement R_A^t, the timing of investment to construct a wastewater treatment facility t_1, and finally the rate of municipal wastewater treatment R_W^t once the wastewater treatment facility is online. We allow for two sources of uncertainty in the construction of the wastewater treatment facility: bringing the facility online may be delayed, and additional costs may in this case be required to complete the construction process. Let P denote the probability that construction is completed as planned and the facility is online in period $t+1$ following investment outlay in period t, and let $(1-P)$ denote that probability that an additional financial outlay $\sum_{i=1}^{n} p_i X_i$ is instead required in period $t+1$ and the facility will not be online until period $t+2$. Assuming that the environmental agency's discount rate is constant, the environmental planner's problem can be written as

$$\begin{aligned} \max \sum_{t=0}^{t_1} \delta^t \left[-D\left(N^t\right) - C_A\left(R_A^t\right)\right] - \delta^{t_1} K \\ + P \sum_{t=t_1+1}^{\infty} \delta^t \left[-D\left(N^t\right) - C_A\left(R_A^t\right) - C_W\left(R_W^t\right)\right] \\ + (1-P)\delta^{t_1+1} \left[-\sum_{i=1}^{n} p_i X_i - D\left(N^{t_1+1}\right) - C_A\left(R_A^{t_1+1}\right)\right] \\ + \sum_{t=t_1+2}^{\infty} \delta^t \left[-D\left(N^t\right) - C_A\left(R_A^t\right) - C_W\left(R_W^t\right)\right] \end{aligned} \tag{3}$$

subject to the stock equation in (2), and

$$0 \leq R_A^t \leq L_A^* \tag{4}$$

$$0 \leq R_W^t \leq \Delta^t L_W^*. \tag{5}$$

The first line in the objective function (3) represents damages and abatement costs when nutrient leachates can be reduced only in agriculture, while the second line represents damages and abatement costs when a wastewater treatment facility is online in the period following investment, allowing nutrient removal from wastewater. The third line represents the case where an additional capital outlay is required to complete the construction, and bringing the facility online is delayed by one period.

We solve the model recursively using dynamic programming. When wastewater treatment capacity is online ($\Delta = 1$) the optimization problem for setting the optimal rates of agricultural abatement and wastewater treatment can be formulated as the following dynamic programming problem:

$$V^1(N,1) = \max_{R_A, R_W} \{-D(N) - C_A(R_A) - C_W(R_W) + \delta V(N',1)\} \tag{6}$$

subject to

$$N' = f(N, L_A^*, L_W^*, R_A, R_W\ ) \tag{7}$$

$$0 \leq R_A \leq L_A^* \tag{8}$$

$$0 \leq R_W \leq \Delta L_W^*. \tag{9}$$

Before an investment in wastewater treatment capacity has been undertaken, at the beginning of each period the environmental agency must decide (i) the optimal rate of agricultural abatement, and (ii) whether to invest in wastewater treatment capacity or not. The decision can be formulated as the following dynamic programming problem:

$$\begin{aligned} V^0(N,0) = \max_{R_A, I} \{ & -D(N) - C_A(R_A) + \delta V^0(N',0), \\ & -D(N) - C_A(R_A) - K + \delta\left[PV^1(N',1) + (1-P)V^2(N',0)\right]\} \end{aligned} \tag{10}$$

where

$$V^2(N',0) = \max_{R_A} \left\{ -\sum_{i=1}^{n} p_i X_i - D(N') - C_A(R_A) + \delta V^1\left(N^{"},1\right) \right\} \tag{11}$$

subject to

$$N' = f(N, L_A^*, L_W^*, R_A) \tag{12}$$

$$0 \leq R_A \leq L_A^* \tag{13}$$

The dynamic program asserts that if the enviromental agency does not invest in wastewater treatment capacity, its cost in the coming year is $-D(N) - C_A(R_A)$ and it begins the subsequent year with no wastewater treatment facility. If the agency does invest, with probability P it starts the next year with the wastewater treatment facility in place and bears a cost $-D(N) - C_A(R_A) - C_W(R_W)$; with probability $(1-P)$, however, an additional investment will be required and bringing the facility online will be delayed by one year, where the cost to the agency is $-\sum_{i=1}^{n} p_i X_i - D(N') - C_A(R_A)$. We solve this program numerically using the collocation method. The method entails discretizing the state space and approximating the value function by n order Chebychev polynomials that are satisfied in n collocation nodes. We first solve the programs in (6) and (11), and insert the value function approximants into (10). The solution yields policy functions $R_A(N, \Delta)$, $R_W(N, \Delta)$ and $I(N, \Delta)$ that map the optimal action with the current state $\{N, \Delta\}$.

3 Empirical model

We calibrate our analytical model for five main components in the empirical analysis: (i) the dynamics of the nutrient stock over time, (ii) the cost of agricultural nutrient abatement, (iii) the cost of municipal wastewater treatment, (iv) the investment cost of establishing wastewater treatment facilities, and (v) the environmental damages. Our data pertain to the Finnish coastal waters of the Gulf of Finland, which receive nutrients predominantly from agricultural runoffs and from municipal point sources. The city of St. Petersburg is the largest point source polluter within the Baltic Sea. Urban wastewaters from the St. Petersburg region reach the sea through the River Neva drainage basin, representing about 70 % of the total point source pollution of the Gulf of Finland. Significant investments are required to enable removing eutrophying nutrients from these discharges. The largest anthropogenic nutrient source within Finland is agriculture.

3.1 Nutrient stock dynamics

Eutrophication in the Gulf of Finland is governed by the availability of nitrogen and phosphorus, the two nutrients typically limiting primary production. We explicitly account for the dynamics of both nitrogen and phosphorus. The agricultural loads of nitrogen and phosphorus in the absence of load reduction measures are denoted by

$$L^*_{NA}, L^*_{PA}, \tag{14}$$

Finnish farmers use predominantly composite fertilizers with a fixed ratio of nitrogen and phosphorus. Moreover, agricultural abatement measures such as buffer strips reduce both nitrogen and phosphorus runoffs. There is then a fixed relationship between the agricultural load reductions for nitrogen and phosphorus,

$$R_{PA} = f\left(R_{NA}\right), \tag{15}$$

where R_{PA} is reduction in agricultural phosphorus load and R_{NA} is reduction in agricultural nitrogen load. The resulting loads of nitrogen and phosphorus entering the coastal ecosystem are

$$L_{NA} = L^*_{NA} - R_{NA}, \tag{16}$$

$$L_{PA} = L^*_{PA} - f\left(R_{NA}\right) \tag{17}$$

The municipal loads of phosphorus and nitrogen in the absence of load reduction measures are

$$L^*_{NW}, L^*_{PW} \tag{18}$$

Once wastewater treatment facilities are in place, nitrogen and phosphorus loads from wastewater are reduced in a ratio that reflects the technology adopted and the contents of each nutrient in wastewater. The relationship between the load reductions for nitrogen and phosphorus through wastewater treatment can be written as

$$R_{PW} = q_W \cdot R_{NW}, \tag{19}$$

where R_{PW} is reduction in municipal phosphorus load, R_{NW} is reduction in municipal nitrogen load, and q_W is the ratio of reductions in phosphorus and nitrogen loads that can be achieved through treating municipal wastewater.

The resulting loads of nitrogen and phosphorus reaching the sea are

$$L_{NW} = L^*_{NW} - R_{NW}, \tag{20}$$

$$L_{PW} = L^*_{PW} - q_W \cdot R_{NW} \tag{21}$$

Marine scientists use complex ecosystem simulation models to study the effects of anthropogenic nutrient loading on eutrophying nutrient stocks. We want to focus on the determinants of nutrient accumulation that can be controlled through abatement measures. The approach we adopt is to use a simple parametric model to describe the fundamental

characteristics of nutrient accumulation over time. In previous economic studies, simple nutrient turnover models have produced satisfactory results for the distribution of eutrophying nutrients in the Baltic (see e.g. Gren et al. 1997, Turner et al. 1999, Hart and Brady 2002). Thus, we describe the dynamics of the stocks of nitrogen and phosphorus as follows:

$$N' = \begin{cases} \alpha N + L^*_{NA} - R_{NA} + L^*_{NW} & \text{if } \Delta = 0 \\ \alpha N + L^*_{NA} - R_{NA} + L^*_{NW} - R_{NW} & \text{if } \Delta = 1 \end{cases} \tag{22}$$

$$P' = \begin{cases} \beta P + L^*_{PA} - f(R_{NA}) + L^*_{PW} & \text{if } \Delta = 0 \\ \beta P + L^*_{PA} - f(R_{NA}) + L^*_{PW} - q_W R_{NW} & \text{if } \Delta = 1 \end{cases} \tag{23}$$

A proportion $(1-\alpha)$ of the total stock of nitrogen is denitrified annually, and a share $(1-\beta)$ of the total stock of phosphorus is retained in the sediment layer. In ecosystem models of the Baltic Sea, the share of denitrification for bioavailable nitrogen in the Gulf of Finland has been estimated to be 50 % (Neuman 2000, Savchuk and Wulff 1999). Kiirikki et al. 2004 have estimated the share of chemically bound phosphorus to be 70 %. The corresponding values of α and β are 0.5 and 0.3.

As our initial values of N and P we use 40000 tn and 6000 tn. The values of L^*_{AN} and L^*_{AP} correspond to the agricultural nutrient loads in the absence of abatement. The values are based on VEPS estimates of the current loads from South-Western Finland and the model of agricultural production on the representative farm. The nutrient loads from municipal wastewaters, or values of L^*_{NW} and L^*_{PW} are based on estimates in Kiirikki et al (2003).

3.2 Cost of agricultural nutrient abatement

The costs of agricultural nutrient abatement are obtained from a representative farm model of South-Western Finland (see Laukkanen et al. 2005). By assumption, the environmental agency imposes environmental policies that limit the allowed load of eutrophying nutrients from agricultural land. Maximum agricultural profits in the absence of leaching restrictions are given by $\pi(L^*_{AN}, L^*_{AP})$. Abatement costs in year t are then measured by $C_A(R^t_{AN}) = \pi(L^*_{AN}, L^*_{AP}) - \pi[L^*_{AN} - R^t_{AN}, L^*_{AP} - f(R^t_{AN})]$, where $\pi[L^*_{AN} - R^t_{AN}, L^*_{AP} - f(R^t_{AN})] = \pi(L^t_{AN}, L^t_{AP})$ gives the maximum profits as a function of the allowed nutrient load. As Finnish farmers use predominantly composite fertilizers with a fixed ratio of nitrogen and phosphorus, and the abatement measures in agriculture, such as establishing buffer strips, reduce both nitrogen and phosphorus runoffs, the abatement and the cost of abatement are expressed in terms of ni-

trogen abatement. By assumption, a leaching restriction is imposed on nitrogen L_{AN}^t and the leaching of phosphorus follows (17).

As in Brady (2001) and Hart and Brady (2002), Laukkanen et al. (2005) constructed a single statistically representative farm to obtain the agricultural abatement costs. The crops cultivated and the agricultural practices available to the representative farm correspond to those typical in South-Western Finland. The crop choices include spring wheat, winter wheat, barley, oats, oilseed, sugarbeet, silage and fallow. The measures available to reduce nutrient runoff from the farm are reduced tillage and no till, establishing buffer strips along waterways, and reducing the rate of fertilization. The agricultural abatement cost function was derived based on deterministic economic and biophysical models of agricultural production and nutrient loading that pertain to regional averages. The abatement cost function thus has to be interpreted as giving the expected cost of agricultural abatement for the region as a whole. The nutrient load from the representative farm was scaled up to correspond to the load from the region estimated through the VEPS environmental accounting system used by the Finnish Environmental Agency.

The agricultural nutrient load in our model pertains to the nutrient load entering the sea. An average of 15 % of phosphorus runoffs and 5 % of nitrogen runoffs are retained along the way (personal communication, Antti Räike, Finnish Environmental Institute). As we are concerned with a representative farm, we abstract away from retention between the representative farm and the coastal ecosystem. In our model retention would have to be represented by the average values for the region, and is accounted for implicitly when we scale up the load from the representative farm to the load from the region as a whole. In a model accounting for heterogenous farms, the effect of differences in retention would be important. In terms of timing, we use a discrete time model, where the entire annual agricultural load enters the system in the beginning of the period.

The abatement costs $C_A\left(R_{AN}^t\right)$ were assessed by solving the representative farm's profit maximization problem for different levels of leaching restriction L_{AN}^t, with increments of 2 %. The abatement cost $C_A\left(R_{AN}^t\right)$ as a function of the load reduction was then obtained by fitting a quadratic cost function to the simulated data. The agricultural abatement costs in euros for abatement on nitrogen runoff in tonnes are given by

$$C_A\left(R_A^t\right) = c_A \cdot \left(R_{AN}^t\right)^2, \tag{24}$$

where the estimated value of the coefficient c_A equals 1.73.

3.3 Costs of municipal wastewater treatment and sewerage system in St. Petersburg

Due to lacking collector sewers and treatment facilities, some 30 % of the municipal wastewaters from St. Petersburg enter the sea without treatment. Currently, there are three major wastewater treatment plants (WWTP), one of which, Krasnoselskaya, constantly receives more waste water than it can treat. It will eventually be replaced by a new plant, the South-West WWTP. The construction of the newest plant started almost 20 years ago, and its total investment cost has been estimated to be about 240 million euro. Approximately 75 % of the investment cost was finally covered by international funding which proved crucial for completion of the construction work (http//swwwtp-project.se). As the plant is designed to operate with a biological nitrogen and phosphorus removal it is expected to reduce the annual biologically available nitrogen load to 11 000 ton and phosphorus to 1190 ton from St. Petersburg (Kiirikki et al 2003).

After the completion of the South-West WWTP, there are still additional measures needed to improve water treatment and sewer network in St. Petersburg. Some of the measures are related to the construction of sewage collectors and renovation of the sewerage system. The greatest uncertainty concerns the implementation options of the Northern collector sewer tunnel. The alternatives considered are completion of the existing collector sewer or construction of a completely new one. It is noteworthy that the construction of the Northern collector sewer tunnel (12 km) started already in 1987, and its completion according to the initial plan is estimated to cost at least up to 300 million Euro (Kiirikki et al p. 21; TOR, 2004 p.14). The major advantage of this tunnel would be that overloading could be avoided and the capacity of the existing WWTPs could be more efficiently utilized. The completion of the system would result in a direct annual load reduction of 1300 ton of the biologically available nitrogen and 100 ton of the biologically available phosphorus of the St. Petersburg municipal load. (Kiirikki et al 2003)

In addition, the nutrient removal efficiency of the existing old plants could be improved by introducing nutrient removal by chemical precipitation. In 2001, the average annual phosphorus removal was 66 % in the Central WWTP and 87 % in the Northern WWTP. It has been estimated that biologically available phosphorus could be reduced by 510 ton/year by introducing chemical phosphorus removal at the Central and Northern WWTP. The corresponding investment cost would be 1.5 million Euro with an annual operation cost of 6 million Euro. (Kiirikki

et al 2003)

All in all, the expected costs of improving municipal wastewater treatment in St Petersburg consist of additional investment and operation costs. Investment cost for the Northern collector sewer is highly uncertain, but it is estimated to lie in the range between 220 and 450 million Euro. The costs of wastewater treatment incur through operating wastewater treatment facilities. The cost of nutrient removal depends on the total volume of wastewater and the nutrient concentration. The most widely applied technology simultaneously removes nitrogen and phosphorus. Thus, the costs of removing nitrogen and phosphorus cannot be separated. We express the costs as a function of nitrogen removal

$$C_W(R_{NW}) = c_w R_{NW}. \tag{25}$$

The unit cost of nitrogen removal is approximately 2200 euros/tn, from Kiirikki et al. (2003).

3.4 Benefits of measures alleviating eutrophication

There are considerable challenges in estimating total benefits from a reduced eutrophication in monetary terms. In our application concerning the Gulf of Finland, we resort to the estimates available from previous valuation studies which indicate that inhabitants in the Baltic drainage basin region place a significant value on these benefits. Willingness to pay (WTP) for a reduction of the eutrophication from the current level (ne_1) to a level that the Baltic Sea can sustain (ne_0) has been estimated using contingent valuation (CV) method by Söderqvist (1996).[1] To use the benefit estimates we have to relate the valuation scenario described in the CV study to a specific reduction in the nutrient concentration which reflects corresponding benefits from avoided environmental damage. For this purpose, we assume environmental damages to depend on the total accumulation of eutrophying nutrients, nitrogen (N) and phosphorus (P). As most plants need nutrients in certain proportions we adopt the Redfield ratio, which is the proportion of nutrients that approximate optimal conditions for growth in algae ($N : P = 7.2$). Based on the Redfield ratio the amount of phosphorus can be recalculated and

[1] A valuation project of Baltic Drainage Basin was carried out as part of the EU Environmental Research Programme (see, e.g., Turner et al 1999). Willingness to pay (WTP) for decreasing the eutrophication in 20 years to a level that the Baltic Sea sustains resulted in Basin wide benefit estimates with national benefits of 215 980 million SEK for Finland and 69 761 million SEK for Russia (present value in 1999). Hence, the total WTP is € 38205 million (1 SEK=0,11 EURO; 2003 values derived using 5% interest rate).

expressed in nitrogen units (NE) (see, e.g., Kiirikki et al 2003). Thus, $NE^t = N^t + 7.2P^t$. The perceived benefits estimated in the CV study give a measure of consumer surplus (compensating variation) associated with the corresponding nutrient reduction. We can express the total willingness to pay $TWTP$ for the avoided damage (benefits) by

$$\int_{ne_0}^{ne_1} D(NE)dNE = TWTP, \quad \text{with } D(ne_0) = 0, \tag{26}$$

where the damage function receives a zero value when the sustainable level has been reached at $NE = ne_0$. Since eutrophication can be an irreversible process with a threshold when an infinite marginal damage occurs we assume that the damage function is exponential and fulfills the appropriate curvature properties being strictly convex. Hence, the damages are approximated by

$$D(NE) = a_d + e^{b_d/(NE-c_d)}, \tag{27}$$

where a_d , b_d and c_d are the parameters we estimate such that c_d gives the threshold level approached.

4 Simulation

4.1 Data

All the data used in the simulation are summarized in Table 1. The ecological parameters and cost estimates reflect the circumstances in the Finnish coastal waters of the Gulf of Finland as realistically as possible, but, given the data limitations, represent to certain extent crude approximations on the dynamics of the water ecosystem.

Table 1. Parameters used in the simulation

Stocks dynamics, initial values	
α	0.5
β	0.3
L^*_{AN}	8600 tn
L^*_{AP}	540 tn
L^*_{NW}	11000 tn
L^*_{PW}	1190 tn
P_0	6000 tn
N_0	40000 tn
Costs of agricultural abatement, $C_A\left(R^t_A\right) = c_A \cdot \left(R^t_{AN}\right)^2$	
c_A	1.73
Operation costs of wastewater treatment, $C_W(R_{NW}) = c_w R_{NW}$	
c_w	2200 euro/ton
Investment costs of wastewater treatment	
K_1	1.5 million euro
K_2	220 million euro
Damages	$D(NE) = a_d + e^{b_d/(NE-c_d)}$
a_d	-128353
b_d	-1500900
c_d	179200

4.2 The optimal policy

In this section we discuss the optimal nutrient abatement policy for the baseline calibration case described above. The optimal policy is a mapping from the current state (N, P, Δ) to the optimal abatement and investment decisions. In each period, a new state is inherited, and new abatement and investment decisions are made. The decision to invest in wastewater treatment capacity depends on the size of the lumpy investment cost and the relative costs of agricultural abatement and wastewater treatment. Our results indicate that it would be optimal to immediately undergo the relatively small investment outlay (K_1) required to improve the nutrient processing capacity of existing plants. The steady state values corresponding to both agricultural abatement and wastewater treatment at the capacity allowed by the improved processing capacity of existing plants are $N^* = 37500$ tn and $P^* = 5400$ tn. Given the high estimated construction cost of the Northern collector sewer tunnel (K_2), our results indicate that it is optimal to refrain from the investment in the tunnel: this investment would only be optimal for $N \geq 58000$ tn and $P \geq 7400$ tn, but even agricultural abatement alone will suffice to

obtain steady state nutrient stocks below these values. Figure 1 displays the optimal state paths for nitrogen and phosphorus. The estimated current nutrient stocks are relatively close to the optimal steady state levels. Of course, this result is partially driven by the estimated damage costs which are based on the willingness-to-pay study. It is likely that more significant damages would make the investment in the collector tunnel pay off.

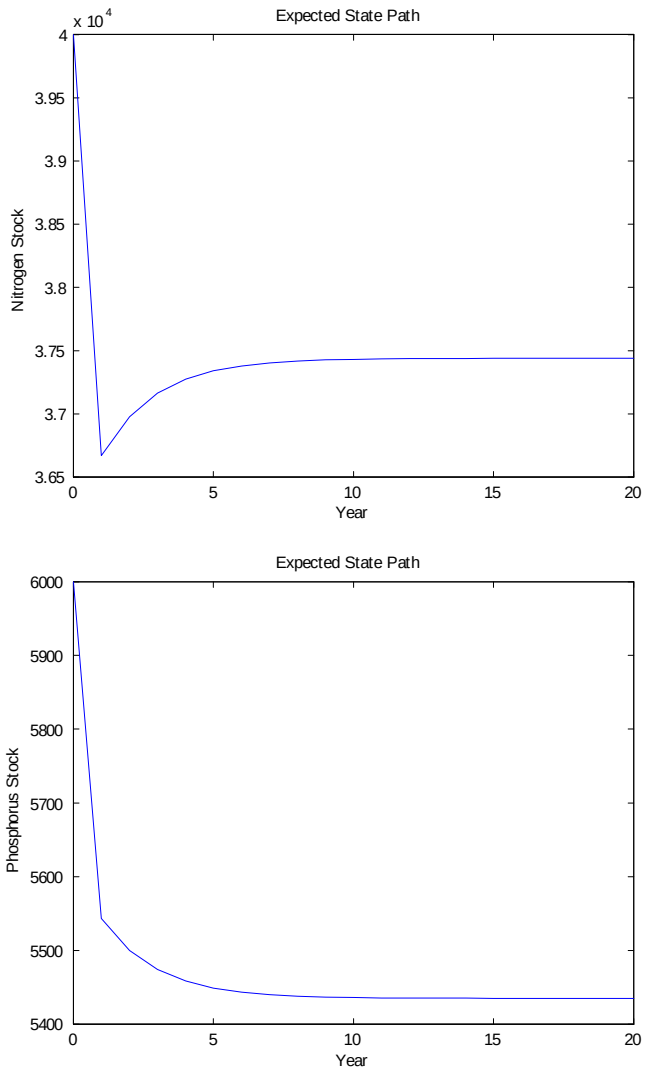

Figure 1. Solution to the nutrient abatement model: optimal state path for nitrogen and phosphorus.

5 Conclusions

We have examined optimal abatement of nutrient loading in an eutrophicated coastal zone where two sources contribute to the nutrient load: agricultural leaching and municipal waste water. A program to reduce the nutrient loads comprises two potential phases. Initially, small scale measures can be adopted to reduce agricultural loading. If investment is undertaken to establish waste water treatment capacity, nutrient loads from municipal waste water can also be controlled. We have formulated an investment and abatement model that incorporates both abatement technologies, and solved the model recursively using dynamic programming. The major findings relate to the rate of abatement in agriculture and in waste water treatment, and the optimal switching policies from agricultural abatement only to a regime where both abatement alternatives are used.

We outlined the conditions under which each of the following three abatement policies should be undertaken. First, in case investment in improving waste water treatment capacity is relatively inexpensive, the investment needed should be carried out immediately to adopt better treatment technology. Second, if the fixed investment cost is relatively high, only agricultural abatement should take place.

We illustrated the results of the theoretical model with a simulation on optimal abatement policies in the Finnish coastal waters of the Gulf of Finland. As expected, the investment in wastewater treatment capacity is highly dependent on the magnitude of the lumpy investment cost.

An interesting extension to this study would be to explicitly consider the uncertainties inherent to the management of nutrient loads also in agriculture. These uncertainties pertain most notably to precipitation influencing the nutrient loss which can vary considerably from year to year.

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