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Balancing Agricultural and Municipal Abatement Measures

Marita Laukkanen & Anni Huhtala

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Abstract

Agriculture and municipal wastewater are the principal sources of eutrophying nutrients in many water ecosystems. We develop a model which considers the characteristics of agricultural and municipal nutrient abatement. The model explicitly accounts for the investment needed to set up wastewater treatment facilities, and makes it possible to determine the optimal timing of investment as well as the optimal agricultural and municipal abatement levels. We apply the model to the Finnish coastal waters of the Gulf of Finland. Our results indicate that substantial savings in abatement costs and the damage associated with eutrophication could be obtained by constructing the facilities needed to process all the wastewaters entering the coastal ecosystem. The optimal timing of investment is shown to hinge on both the economic and ecological characteristics of the ecosystem.

Key words: eutrophication control, nutrient abatement, endogenous regime switches, environmental policy

*Corresponding author. MTT Economic Research, Agrifood Research Finland, Luutnantintie 13, 00410 Helsinki, Finland, marita.laukkanen@mtt.fi

†MTT Economic Research, Agrifood Research Finland, Luutnantintie 13, 00410 Helsinki, Finland, anni.huhtala@mtt.fi

1 Introduction

One key environmental concern today is reducing the nutrients that lead to excessive growth of phytoplankton and eutrophication of water ecosystems. Eutrophication is manifested as decreased water transparency, disproportionate growth of filamentous algae and aquatic plants, and mass blooms of toxic blue-green algae (see, e.g., Ærteberg et al., 2001; Gabric and Bell, 1993). Eutrophication can affect people's health directly, and cause losses to fisheries and recreational activities. Valuation studies have attributed significant economic benefits to improving the state of eutrophied inland waters and coastal zones (see, e.g., Söderqvist and Scharin 2000, Söderqvist 1996, Markovska and Zylicz 1999, Pretty et al. 2003).

Many environmental assessments identify agriculture as the major cause of surface quality problems (Shortle & Abler, 1999), particularly as municipal and industrial nutrient loads have been reduced considerably during the last few decades. However, in many regions urban and industrial wastewater treatment facilities are still lacking and untreated wastewaters remain a significant source of nutrient loading in addition to agricultural runoff. For example, several EU Member States are behind schedule in establishing the sewage treatment capacity required by the Urban Waste Water Treatment Directive (EEA 2005). Abatement costs in agriculture and in wastewater treatment have a fundamentally different character. While agricultural abatement takes the form of reversible small-scale measures such as changes in fertilizer use, manure spreading and tillage practices, an irreversible initial investment is needed to set up wastewater treatment facilities and the sewage infrastructure needed to transport wastewater from households to treatment facilities. When abatement measures entail discrete investments that impose considerable sunk costs on society, the investment costs should be appropriately accounted for in policy choices (see e.g. Pindyck 2000).

This paper explores optimal nutrient abatement policies when a significant discrete investment is required for reducing nutrient loads from municipal point sources and agricultural abatement measures represent a backstop technology. The study contributes to the existing literature in several important aspects. While the dynamics of the eutrophication process have been described in detail in analytical models (e.g., Carpentier et al. 1999, Naevdal 2001, Mäler et al. 2003, Ludwig et al. 2003), the investment required to set up wastewater treatment facilities has, to our knowledge, not been considered in previous papers. Studies analyzing nutrient abatement in both agricultural and municipal sources have applied a static framework and assumed that the requisite abatement technology is already in place (e.g., Elofsson 2003, Malik et al. 1993, Gren et al. 1997). Furthermore, empirical applications with dynamic models of nutrient accumulation are relatively rare. Hart and Brady (2002) and Hart (2003) studied optimal abatement of nutrient loading in the Baltic Sea in a dynamic setting but considered only one nutrient, nitrogen, and one nutrient source, agricultural leaching.

Here, we contribute to the literature on nutrient abatement by modelling explicitly the decision to invest in wastewater treatment capacity and by analyzing the optimal allocation of agricultural versus municipal abatement effort in a dynamic setting.[1] We apply the model empirically to study nutrient abatement in the Gulf of Finland, one of the most eutrophied sub-basins of the Baltic Sea. The assertion that nitrogen limits primary production in the marine environment has been challenged recently in the scientific community (see, e.g., Boesch et al. 2006). We augment the present empirical modelling of eutrophication by considering the two nutrients that are necessary for primary production: nitrogen and phosphorus. For economic variables, we take into account the uncertainty pertaining to the duration and cost of constructing underground sewage tunnels and wastewater treatment facilities. Furthermore, we explicitly model the benefits of improved water quality, where previously only total costs and benefits have been compared in a static framework assuming that the marginal costs and benefits of abatement depend on flows of nutrients (e.g., Turner et al. 1999, Gren 2001, Gren and Folmer 2003). In our model, the benefits are attributed to the state of the water ecosystem as measured by nutrient mass, whereby the optimal abatement for given costs can be endogenously determined.

We focus on the following questions: Under what conditions should investment in wastewater treatment facilities be undertaken? What determines the optimal time to invest? How are agricultural and municipal nutrient abatement balanced where investment is undertaken? and How does the optimal agricultural abatement policy change once wastewater treatment facilities are operational? The empirical results suggest that in the case of the Gulf of Finland, the investment required to process the currently un-

[1] In the literature, the optimal investment policy has typically been analyzed as a two-stage optimal control problem (see, e.g., Amit 1984, Tomiyama 1985, Makris 2001). A similar logic is adopted here, but the specific approach taken is discrete-time dynamic programming.

treated wastewaters should be undertaken immediately. The optimal abatement policy decreases the phosphorus stock but allows the nitrogen stock to increase, which highlights the importance of describing the dynamics of both nitrogen and phosphorus stocks when the nutrients differ in residence times and their contribution to damage**.** A sensitivity analysis helps unveil the reasons behind the still prevalent non-compliance with sewage treatment requirements (see, e.g., EEA 2005): when agricultural abatement is relatively inexpensive, the municipal nutrient load is relatively small, or the accumulation of nutrients is slow, it may be optimal to refrain from investment.

The following section presents the theoretical model, and section 3 details the empirical work performed to calibrate the model. Section 4 then goes on to characterize the optimal policy and discuss its implications. Section 5 concludes.

2 The model

Consider a coastal zone that receives eutrophying nutrients from two principal sources - agricultural runoff and municipal wastewater discharges. An environmental planner wishes to control nutrient loading in order to minimize the both the total environmental damage caused by nutrient accumulation and the cost of nutrient abatement. Agricultural nutrient abatement does not involve set-up costs. In contrast, removing nutrients from municipal wastewater requires an outlay of capital for treatment facilities. There are thus two potential phases of nutrient abatement: prior to the investment, only agricultural nutrient loads can be controlled; if the necessary outlay is made and wastewater treatment facilities are built, nutrient loads from both agricultural and municipal sources can be reduced.

We next describe the basic economic, ecological and technological conditions in the coastal ecosystem concerned. Our model accounts for the accumulation of two nutrients: nitrogen and phosphorus. Nutrient load reductions in agricultural and municipal sources are used as the control variables. The notation is as follows: $t = 1, 2, ...$ indexes the period; $S^t = (N^t, P^t)'$ is a vector containing the stocks of nitrogen and phosphorus; $L_A^* = (L_{AN}^*, L_{AP}^*)'$ is a vector of agricultural nitrogen and phosphorus loads in the absence of abatement measures; $R_A^t = (R_{AN}^t, R_{AP}^t)'$ is a vector of agricultural nitrogen and phosphorus abatements; $L_W^* = (L_{WN}^*, L_{WP}^*)'$ is a vector of municipal nitrogen and phosphorus loads in the absence of abatement measures; $R_W^t = (R_{WN}^t, R_{WP}^t)'$ is a vector of nitrogen and phosphorus abatements through wastewater treatment; and K is the cost of establishing wastewater treatment facilities.

The agricultural nutrient load in the absence of abatement, L_A^*, is defined as the load resulting from unconstrained maximization of farm profits. Agricultural nutrient load abatement bears a cost $C_A(R_A^t)$, where the cost function is increasing and convex in the abatements (R_{AN}^t, R_{AP}^t). This cost structure follows from the standard assumption that agricultural profits are increasing and concave in nutrient loading. As the volume and nutrient content of untreated wastewater are largely determined by population size, we proceed from the assumption that the municipal load in the absence of abatement, L_W^*, is 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 wastewater cleaning rate nor does it have any impact on the unit cost of cleaning wastewater. While these assumptions are probably an oversimplification, they illustrate the principle of a capital outlay being required in order to abate municipal nutrient loads. Once wastewater treatment facilities are operational, the costs of nutrient removal are denoted by $C_W(R_W^t)$, where the cost function is increasing and convex in the abatements (R_{WN}^t, R_{WP}^t).

We allow for two sources of uncertainty in the construction of wastewater treatment facilities: bringing the facilities online may be delayed and additional capital outlays may be required to complete the construction process. Construction of sewer infrastructure, for example, involves underground excavation, where the duration and difficulty of a project are fully revealed only as work proceeds. In what follows, p denotes the probability that construction is completed as initially planned in T_0 periods and that the facilities are online in period $\tau + T_0$ following investment outlay $I^\tau = K$ in period τ. With probability $(1 - p)$, period $\tau + T_0$ reveals a delay and an additional financial outlay that is necessary to complete the project. For simplicity, we assume that the set of possible additional outlays, X, is finite and has n elements, with p_i the probability that expense $x_i \in X$ will arise. In the case of delay, facilities will be online with certainty in period $\tau + T_1$, where $T_1 > T_0$. The indicator function Δ^t takes on a value of 1 if the wastewater treatment facilities are online, and 0 otherwise:

$$\Delta^t = \begin{cases} 0 & \text{wastewater treatment facilities not online} \\ 1 & \text{wastewater treatment facilities online} \end{cases} \tag{1}$$

The stock of nutrients increases as agricultural or municipal nutrient loads, L_A^t or L_W^t, enter the ecosystem. The stock $S = (N, P)'$ changes from one period to the next as follows:

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

Finally, environmental damage is a function of accumulated nutrients, $D(S)$, which is increasing and convex in $S = (N, P)$.

We next state the two-phase nutrient abatement problem. The environmental planner's objective is to minimize the total environmental damage caused by nutrient accumulation and the cost of nutrient abatement. The problem entails determining the optimal rates of agricultural abatement, (R_{AN}^t, R_{AP}^t), the timing of investment to construct wastewater treatment facilities, τ, and the optimal rates of municipal wastewater treatment, (R_{WN}^t, R_{WP}^t), once the wastewater treatment facilities are online. Let δ denote the environmental planner's discount factor, which by assumption is constant. The problem can then be written as

$$\begin{aligned} \max_{R_A, R_W, \tau} - \Bigg\{ & \sum_{t=0}^{\tau+T_0-1} \delta^t \left[D\left(S^t\right) + C_A\left(R_A^t\right)\right] + \delta^\tau K \\ & + p \sum_{t=\tau+T_0}^{\infty} \delta^t \left[D\left(S^t\right) + C_A\left(R_A^t\right) + C_W\left(R_W^t\right)\right] \\ & + (1-p) \left\{ \delta^{\tau+T_0} \sum_{i=1}^{n} p_i x_i + \sum_{t=\tau+T_0}^{\tau+T_1-1} \delta^t \left[D\left(S^t\right) + C_A\left(R_A^t\right)\right] \right. \\ & \left. + \sum_{t=\tau+T_1}^{\infty} \delta^t \left[D\left(S^t\right) + C_A\left(R_A^t\right) + C_W\left(R_W^t\right)\right] \right\} \Bigg\} \end{aligned} \tag{3}$$

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 term in the objective function (3) represents damage and abatement costs when only agricultural nutrient loading can be controlled. The initial investment outlay is captured by the second term. The third term represents damage and abatement costs when wastewater treatment facilities are brought online in T_0 periods following the investment. The last term represents the case where an additional capital outlay is required to complete construction, and bringing the facility online is delayed until period $\tau + T_1$.

As long as no investment outlay has been made, at the beginning of each period the environmental planner must decide on the optimal rate of agricultural nutrient abatement and whether to invest in the construction of wastewater treatment facilities or not. If wastewater treatment facilities are built, once they are online the environmental planner must choose the optimal rate of agricultural abatement and the optimal rate of wastewater treatment. We solve the environmental planner's problem recursively, starting from the situation where treatment capacity is online. We proceed by writing out the social value of current and future damage and abatement costs in each phase, given the nutrient stock S and the status of wastewater treatment facilities indicated by Δ. In any period t such that $\Delta^t = 1$, that is, wastewater treatment facilities are operational, the value of current and expected future rewards satisfies

$$V^1\left(S^t\right) = \max_{R_A^t, R_W^t} \left\{-D\left(S^t\right) - C_A\left(R_A^t\right) - C_W\left(R_W^t\right) + \delta V^1\left[S^{t+1}\right]\right\}. \tag{6}$$

Consider first the case where construction is delayed so that completing the project takes T_1 years. Thus, if an investment $I^\tau = K$ is made in period τ, capacity will be online in period $\tau + T_1$. In period $\tau + T_1$, the value of current and expected future rewards is $V^1\left(S^{\tau+T_1}\right)$. In the intermediate periods from $\tau + T_0$, where a delay is revealed, to $\tau + T_1$, the value of current and expected future rewards is

$$J^{T_1-1}\left(S^{\tau+T_1-1}\right) = \max_{R_A}\left\{-D(S^{\tau+T_1-1}) - C_A(R_A^{\tau+T_1-1}) + \delta V^1\left(S^{\tau+T_1}\right)\right\} \tag{7}$$

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$$J^{T_0}\left(S^{\tau+T_0}\right) = \max_{R_A}\left\{-D(S^{\tau+T_0}) - C_A(R_A^{\tau+T_0}) + \delta J^{T_0+1}\left(S^{\tau+T_0+1}\right)\right\}. \tag{8}$$

Consider next the periods between initial investment and the revelation of a possible delay, i.e., from $\tau+1$ to $\tau+T_0$. Where construction is not delayed, the value of current and expected future rewards in period $\tau+T_0$ is

$$V^1\left(S^{\tau+T_0}\right). \tag{9}$$

Thus, in the periods from $\tau+1$ to $\tau+T_0-1$, we have

$$\begin{aligned} J^{T_0-1}\left(S^{\tau+T_0-1}\right) = \max_{R_A}&\left\{-D(S^{\tau+T_0-1}) - C_A(R_A^{\tau+T_0-1})\right. \\ &\left.+\delta\left[pV^1\left(S^{\tau+T_0}\right) + (1-p)J^{T_0}\left(S^{\tau+T_0}\right)\right]\right\} \end{aligned} \tag{10}$$

$$J^{T_0-2}\left(S^{\tau+T_0-2}\right) = \max_{R_A}\left\{-D(S^{\tau+T_0-2}) - C_A(R_A^{\tau+T_0-2}) + \delta J^{T_0-1}\left(S^{\tau+T_0-1}\right)\right\} \tag{11}$$

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$$J^0\left(S^\tau\right) = \max_{R_A}\left\{-D(S^\tau) - C_A(R_A^\tau) + \delta J^1\left(S^{\tau+1}\right)\right\} \tag{12}$$

Finally, as long as no investment outlay has been made, the value of current and expected future rewards is given by

$$V^0(S^t) = \max_{R_A^t}\left\{-D\left(S^t\right) - C_A\left(R_A^t\right) + \delta V^0\left[S^{t+1}\right]\right\}. \tag{13}$$

We are now set to investigate the questions of whether and when to invest in wastewater treatment facilities. Three possible cases arise:

Case 1. When wastewater treatment is relatively inexpensive compared to agricultural abatement, the municipal nutrient load relatively large, or the rate of nutrient accumulation high, it may be desirable to invest immediately. Formally, for immediate investment at $\tau = 0$ to be optimal, the condition

$$J^0\left(S^0\right) - K \geq V^0(S^0) \tag{14}$$

must hold. This means that the social value generated by commencing wastewater treatment as soon as possible minus the initial capital cost of establishing wastewater treatment facilities must be at least as large as the social value arising from agricultural abatement only.

Case 2. When agricultural abatement is relatively inexpensive, the municipal load relatively small, or the rate of nutrient accumulation slow, it may be desirable to practice agricultural abatement alone. Formally, it will be optimal to refrain from investment if

$$J^0\left(S\right) - K < V^0(S) \tag{15}$$

for all S. This condition means that for any amount of accumulated nutrients S, abating only agricultural loads must generate a social value that exceeds the value of abating in both sources when the cost of initial capital outlay is accounted for.

Case 3. If equation (14) does not hold but a stock level is eventually reached at which the social value of treating wastewater in addition to abating agricultural loads, minus the initial capital outlay, equals but does not exceed the social value under agricultural abatement only, it will be optimal to invest after a period of only agricultural abatement. Formally, investment is undertaken once the nutrient stock reaches a level where

$$J^0\left(S\right) - K = V^0(S). \tag{16}$$

The value functions in (6) to (13) are solved for numerically using the collocation method. The method entails discretizing the state space and approximating the value function by m^{th} order Chebychev polynomials that are satisfied in m collocation nodes. The solution yields policy functions $R_A(S, \Delta)$, $R_W(S, \Delta)$ and $I(S, \Delta)$ that map the optimal action with the current state $\{S, \Delta\}$ .The solution was implemented using the CompEcon Toolbox for Matlab.[2] The Matlab code is available from the authors upon request.

3 Empirical application

We illustrate the preceding model by applying it to waters of the Gulf of Finland along the Finnish coast, where the principal external nutrient sources are agricultural runoff from southwestern Finland and municipal wastewaters from the St. Petersburg region in Russia. The Gulf of Finland is one of the most eutrophied sub-basins of the Baltic Sea, and nutrient enrichment has led to marked increases in algae biomass, frequent blooms of toxic blue-green algae and oxygen depletion, including anoxic "dead zones" in bottom waters. Although the external loading to the gulf has decreased considerably during the past decade, its trophic status has not changed correspondingly. The lack of positive response is partly explained by the substantial internal loading of nutrients in the area. Moreover, the nutrient loading to the Gulf of Finland is still 2–3 times that of the Baltic Sea average. (Boesch et al. 2006; Lehtoranta, 2003; Pitkänen et al. 2001).

Comparison of abatement measures in agriculture and municipal wastewater treatment facilities is highly relevant in the northern part of the Gulf of Finland. Along the Finnish coast all municipal wastewater is treated before it enters the sea but agriculture remains a significant nutrient source, comprising 42% of the load from Finland to the Gulf of Finland (Kauppila et al., 2001). Sewage infrastructure is lacking in St. Petersburg and significant investments will be required to enable the removal of nutrients from all municipal discharges. Currently the wastewaters of some 500,000 residents are released untreated into the Neva River, from which they enter the Gulf of Finland. These wastewaters represent about 70% of the total point source pollution of the Gulf of Finland. Significant delays have occurred in past construction projects: the building of the newest, and one of the largest, treatment plants in St. Petersburg took over 20 years due to unrealistic initial budgeting, with the total investment cost ultimately approaching 240 million euro.

The ecological parameters and cost estimates used in our empirical model reflect

[2] The CompEcon Toobox is a library of MATLAB functions for numerically solving a variety of problems in economics and finance that was developed to accompany Miranda and Fackler (2002). The library is downloadable at http://www4.ncsu.edu/~pfackler/compecon/toolbox.html.

the circumstances in the Finnish coastal waters of the Gulf of Finland as realistically as possible, given data limitations and tractability requirements. (See Table 1 for a summary of the parameter values.) The empirical model has four main components, which we discuss in detail in the following subsections: (i) the dynamics of the nutrient stock over time, (ii) the cost of agricultural nutrient abatement, (iii) the cost of municipal wastewater treatment and sewerage system, and (iv) the environmental damages.

[Table 1 about here]

3.1 Nutrient stock dynamics

While marine scientists use complex ecosystem simulation models to study the effects of anthropogenic nutrient loading on nutrient stocks, previous economic studies of the Baltic Sea region have produced satisfactory results for the distribution of eutrophying nutrients using simple nutrient turnover models (see, e.g., Gren et al. 1997, Turner et al. 1999, Hart and Brady 2002). We follow this approach and adopt a simple parametric model to describe the fundamental characteristics of nutrient accumulation. Accordingly, the relevant nutrient stock dynamics are summarized in the following vector equation:

$$S^{t+1} = \begin{cases} \alpha S^t + L_A^* - R_A^t + L_W^* + L_E^* & \text{if } \Delta = 0 \\ \alpha S^t + L_A^* - R_A^t + L_W^* - R_W^t + L_E^* & \text{if } \Delta = 1, \end{cases} \tag{17}$$

where $\alpha = diag(a_N, \alpha_P)$ is a matrix of annual carry-over rates for nitrogen and phosphorus, and $L_E^* = (L_{EN}^*, L_{EP}^*)'$ is a vector of nitrogen and phosphorus loads from other anthropogenic and natural nutrient sources that cannot be controlled by active abatement measures and in this sense are exogenous to the environmental planner.

Ecosystem models of the Baltic Sea indicate that 50% of the bioavailable nitrogen in the Gulf of Finland is denitrified annually (Neuman 2000, Savchuk and Wulff 1999), which corresponds to a nitrogen carry-over rate of $\alpha_N = 0.5$ in our model. The estimates of the proportion of phosphorus that is buried in the bottom sediments of the Gulf of Finland range from 0 to 70% depending on the availability of oxygen (Kiirikki et al. 2006). As no distribution is available for the proportion of inactivated phosphorus, we proceed from the assumption that the values are uniformly distributed between 0 and 70% and use the mean value of 35 % phosphorus inactivation, which yields a carry-over rate of $\alpha_P = 0.65$. The study area covers the part of the Gulf of Finland where nutrient loads from Finnish agriculture and the municipal wastewaters of St.Petersburg have an impact, i.e., coastal areas of the northern part of the gulf with a volume of 200 km^3. The nutrient concentrations are estimated to be 200 mg/m^3 for nitrogen and 30 mg/m^3 for phosphorus. (Heikki Pitkänen and Pirkko Kauppila, The Finnish Environment Institute (FEI), personal communication). Hence, the initial stocks of nitrogen and phosphorus, denoted by N^0 and P^0, are 40,000 t and 6000 t. The agricultural nutrient loads in the absence of abatement, L^*_{AN} and L^*_{AP} were derived from Helin et al. (2006). The nutrient loads from municipal wastewater, L^*_{WN} and L^*_{WP}, and exogenous loads, L^*_{EN} and L^*_{EP}, correspond to estimates by Kiirikki et al. (2003) and Heikki Pitkänen (personal communication).

3.2 Agricultural nutrient loads and abatement costs

The Finnish coastal waters of the Gulf of Finland receive agricultural runoff primarily from the provinces of Uusimaa and Varsinais-Suomi in southwestern Finland. The costs of agricultural nutrient abatement in the region are derived from a study by Helin et al. (2006), which ascertained abatement costs based on deterministic economic and biophysical models of agricultural production. The estimated cost function thus has to be interpreted as a mapping of the expected costs of agricultural abatement.

Helin et al. proceed from the assumption that environmental authorities limit the allowable agricultural nutrient load, and estimate costs in terms of the farm profits that are foregone due to the restrictions. As the principal abatement measures applicable in the region - buffer strips, conservation tillage and changes in crop mix - reduce both nitrogen and phosphorus runoffs, a fixed relationship is assumed between nitrogen and phosphorus abatements and the abatement costs are derived in terms of nitrogen load reductions. Maximum agricultural profits without load restrictions are denoted by $\pi\left(L^*_{AN}\right)$, where L^*_{AN} is the unconstrained agricultural nitrogen load. The abatement costs in year t are measured by

$$C_A\left(R^t_{AN}\right) = \pi\left(L^*_{AN}\right) - \pi\left(L^*_{AN} - R^t_{AN}\right), \tag{18}$$

where R^t_{AN} is the reduction in agricultural nitrogen load required by the authorities and $\pi\left(L^*_{AN} - R^t_{AN}\right)$ the maximum agricultural profits under the load constraint. The representative farm's profit maximization problem was solved for different abatement targets, and a quadratic cost function was fit to the simulated data. The resulting nitrogen abatement costs in euros, for abatement in tonnes, are given by

$$C_A\left(R_A^t\right) = c_A \cdot (R_{AN}^t)^2, \tag{19}$$

where the estimated value of the coefficient c_A is 1.68. For any nitrogen abatement rate and the associated cost, the phosphorus load is given by

$$R_{AP}^t = q_A R_{AN}, \tag{20}$$

where q_A is a constant whose estimated value is 0.0039. The unconstrained nitrogen load L_{AN}^* is 7764 t and the phosphorus load L_{AP}^* 522 t.

An average of 15% of the phosphorus loads and 5% of the nitrogen loads from agricultural land in Uusimaa and Varsinais-Suomi are retained by lakes and rivers along the way to the Gulf of Finland (personal communication, Antti Räike, FEI). In a representative farm model, retention would have to be described by the average values for the region. Helin et al. account for retention implicitly through model calibration, and the unconstrained agricultural loads produced by their model are the loads entering the sea. Consequently, our analysis also abstracts away from retention.

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

There are currently three major and several small wastewater treatment plants (WWTP) in operation in St. Petersburg. Yet, some 20% of the municipal wastewaters enter the Gulf of Finland without treatment due to a lack of collector sewers and treatment facilities. Construction of sewage collectors and renovation of the sewage system will be needed to prevent untreated sewage effluence into water bodies, but the extensive investment program planned (Krasnoborodko et al 1999) has not been carried out due to limited funding. The most expensive investment required is a main tunnel sewer, which is necessary for the transportation of discharges to the existing WWTPs. A major advantage of the tunnel sewer would be that overloading of the sewage system could be avoided by utilizing the capacity of the existing WWTPs more efficiently. As an additional measure, the nutrient removal efficiency of the existing plants could be improved by introducing nutrient removal through chemical precipitation. (Kiirikki et al. 2003)

Our cost estimates are based on the investment outlay that would enable the construction of the tunnel sewer and the enhancement of nutrient removal at two major WWTPs. The investment cost has been estimated to be 330 to 440 million euro, and the construction work is estimated to take 4-8 years according to a preliminary schedule (Vodokanal 2005). We proceed from the assumption that construction is completed in 4 years with probability 0.5, and delayed by another 4 years with probability 0.5. Once capacity is online, the costs of wastewater treatment are those of operating the treatment facilities. The cost of nitrogen and phosphorus removal depends on the total volume of wastewater and on the nutrient concentrations. Nitrogen and phosphorus are removed in a ratio that reflects the technology adopted at the treatment facilities and the amount of each nutrient in wastewater, which is constant by assumption. The ratio of nitrogen and phosphorus abatement through wastewater treatment is then captured by

$$R_{WP} = q_W R_{WN}, \tag{21}$$

where $q_W = 0.45$ corresponds to the treatment capacity that the investment in the tunnel sewer and additional nutrient removal by chemical precipitation would provide. Operating costs arise from chemical and biological processing of wastewater, and we proceed from the assumption that the unit cost remains constant. Given the fixed ratio of nitrogen and phosphorus removal, we express the operational costs of wastewater treatment as a function of nitrogen removal

$$C_W(R_{NW}) = c_w R_{WN}. \tag{22}$$

The estimated value of c_w is 4460 euro per tonne. For these overall expenditures, a maximum reduction of 2285 tonnes of nitrogen can be achieved. (Vodokanal 2005; H. Pitkänen, FEI).

3.4 Damage from eutrophication

Even though there are considerable challenges in estimating the total benefits of reduced eutrophication in monetary terms, the empirical literature on valuation of water quality improvements is extensive (see, e.g., Freeman 1996, Wilson and Carpenter 1999). In our application, we rely on benefit estimates available from a previous contingent valuation study by Söderqvist (1996), who carried out a valuation project of Baltic drainage basin as part of an EU Environmental Research Programme (see also Turner et al. 1999). The study indicated that inhabitants in the region place a significant value on the benefits: willingness to pay (WTP) for reducing eutrophication in 20 years from its current level to a level that the Baltic Sea can sustain resulted in a basinwide estimate for total benefits of about 7600 million euro per year. The corresponding annual WTPs per adult were 600 euro in Finland and 100 euro in Russia. We assume that the inhabitants in the coastal areas in Finland and Russia are only concerned about the water quality in the Gulf of Finland, not in any other part of the Baltic Sea. When only the people living within the Baltic drainage basin in these countries are taken into account, the total present value WTP for the 20-year period approaches 55,000 million euro.[3]

We relate the WTP measure for avoiding eutrophication to a specific reduction in the nutrient stock. Hence, environmental damage is assumed to depend on the level of eutrophication, which is governed by the total amount of eutrophying nutrients, nitrogen (N) and phosphorus (P), accumulated as a stock in the Gulf of Finland. The N:P ratio of phytoplankton averages 7.2:1 (mass:mass, Redfield et al., 1963). We use this ratio to convert the amount of phosphorus into nitrogen equivalent units, E, which we use as an indicator of eutrophication (for a similar approach using a phosphorus-based nitrogen equivalent, see, e.g., Kiirikki et al., 2003 and Anon., 2004). The nutrient stock measured in nitrogen equivalents is given by $E^t = N^t + 7.2P^t$. While the damage is a function of an aggregate nutrient stock measured in nitrogen equivalents, separate equations of motion for nitrogen and phosphorus enable us to account for the differences in the accumulation rates of the two nutrients as well as the differences in the effects of abatement measures on the respective loads of each.

The perceived benefits estimated by the WTP in the contingent valuation 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

[3] 2003 values in euro were recalculated from Table 5 in Turner et al. (1999) using a 5% interest rate and an exchange rate of 1 SEK=0.11 euro.

$$\int_{e_0}^{e_1} D(E)dE = TWTP, \quad \text{with } D(e_0) = 0, \tag{23}$$

where $e_1 = 83,200$ t is the current level of the nutrient stock measured in nitrogen equivalents. The damage function receives a zero value when the sustainable level has been reached at $e_0 = 51,600$ t. Because severe eutrophication may result in infinite marginal damage at a certain threshold level, we assume that the damage function is exponential and fulfills the appropriate curvature properties, being strictly convex. The approximated damage takes the form

$$D(E) = a_d + e^{b_d/(E-c_d)}, \tag{24}$$

where $c_d = 179,200$ indicates the threshold level approached and $a_d = -170,190$ and $b_d = -1,536,900$ have been determined numerically for the given $TWTP$ estimate.

4 Results

This section discusses the optimal nutrient abatement policy, first for the baseline calibration case and then for alternative parameterizations. The optimal policy is a mapping from the current state (N^t, P^t, Δ^t) to the optimal abatement levels (R_A^t, R_W^t) and, where the construction of wastewater treatment facilities has not already begun, the optimal investment decision I^t. In each period, a new state is inherited, and new abatement and investment decisions are made.

4.1 The optimal policy: baseline calibration case

We first discuss how establishing wastewater treatment facilities affects the optimal agricultural abatement policy and how wastewater treatment and agricultural abatement are optimally balanced once treatment capacity is online. Figure 1 shows the optimal agricultural abatement policy for the case where no investment has been made and only agricultural nutrient loads can be reduced. The optimal abatement rate is increasing and convex in the stocks of nitrogen and phosphorus and approaches the upper limit of $L_{AN}^* = 7764$ t when both nutrient stocks are very large. For comparison, Figure 2 displays the optimal agricultural abatement policy in the case where wastewater treatment facilities are in use. As could be expected, for any stock level the optimal agricultural abatement rate is now substantially smaller than when wastewater treatment is not an option. Figure 3 shows the optimal wastewater treatment policy, which is also increasing in the stocks of nitrogen and phosphorus. The upper bound of wastewater processing at the projected capacity, $\overline{R}_{WN} = 2285$ t, is reached at moderate nutrient stock levels.

[Figures 1, 2, and 3 about here]

Table 2 reports the steady state abatement rates and nutrient stock levels for the cases of (i) agricultural abatement only and (ii) both agricultural abatement and wastewater treatment. The optimal policy in the absence of wastewater treatment requires cutting back agricultural nitrogen loads by more than 50% relative to the profit-maximizing level, which would entail substantial abatement costs, 27 million euro per year. When wastewater treatment is possible, abatement consists primarily of removing nutrients from wastewater. At the initial stock levels, the optimal wastewater policy requires treatment at full capacity. As the nutrient stocks approach their optimal steady state levels, the optimal treatment rate falls to approximately 90% of the maximum treatment allowed by the projected capacity.[4] Agricultural abatement is modest. Significant cost savings are achieved by establishing wastewater treatment facilities: total abatement costs fall to approximately 9 million euro per year, and damage to 3 million euro per year, which is less than 10% of the damage associated with the case of no wastewater treatment.

Agricultural abatement alone achieves only minor reductions in phosphorus loading and consequently in the amount of accumulated phosphorus. While the steady state phosphorus stock in case (i) is smaller than the current stock level, it remains more than twice the size of the steady state stock in case (ii). The projected investment enables a much higher rate of phosphorus removal relative to the removal of a unit of nitrogen than is possible with agricultural abatement. Interestingly, the steady state nitrogen stock is above the current level in both cases. When nitrogen equivalents are used as an indicator of eutrophication and the N:P ratio of 7.2 is employed to convert phosphorus into nitrogen equivalents, phosphorus receives considerable weight in the

[4] That wastewater treatment does not occur at full capacity, despite the constant marginal cost, derives from the nonlinearities in other model functions.

damage function and thus becomes the more important abatement target. Measured in nitrogen equivalents, the nutrient stock will increase when only agricultural abatement is possible but decrease when wastewater treatment is also available.

[Table 2 about here]

In light of the differences in the abatement costs and damage associated with the steady states using agricultural abatement alone vis-à-vis both agricultural abatement and wastewater treatment, it is not surprising that investing in wastewater treatment capacity is found to be optimal in the baseline calibration case. The condition $J^0(N^0, P^0) - K > V^0(N^0, P^0)$ holds at the initial stock level (N^0, P^0). Thus, it is optimal to invest immediately. Figures 4 and 5 display the state and policy paths for a twenty-year time span, starting from the current state. Following an investment in period $t = 0$, capacity will be online at time $t = T_0 = 4$ if no delay occurs, and time $t = T_1 = 8$ if construction is delayed. The state and policy paths for no investment are provided for comparison. Once investment has been undertaken, the nitrogen stock is allowed to increase slightly more than in the case of no investment. If a delay is revealed in period T_0, agricultural abatement increases, and the nitrogen stock is brought back into line with the path that is optimal without investment. As period T_1 approaches, agricultural abatement declines, and the nitrogen stock is allowed to increase in anticipation of initiating wastewater treatment at T_1. Once online, wastewater treament facilities are operated at full capacity for four periods, after which the treatment rate fall slightly. As only modest phosphorus abatement can be achieved through agricultural measures in the present model, the state path of phosphorus after investment follows that emerging without investment until capacity is online at period T_0 or T_1. Investment in wastewater treatment capacity halves the level of phosphorus stock.

[Figures 4 and 5 about here]

4.2 Sensitivity analysis

The previous section discussed the optimal abatement and investment policies under the baseline calibration. This section reports the results from a number of alternative parameterizations. The analysis serves to study the sensitivity of the results to the specific parameter values and to illustrate how the various forces at play affect the optimal timing of investment. The optimal abatement policies are qualitatively similar to the baseline scenario, and we therefore only report the steady state values of the key variables here (Table 3).

While immediate investment is optimal in the baseline case, the result is sensitive to the changes in the key parameters, in particular those describing the ecological model. A 10% decrease in all nutrient loads in the absence of abatement, L_A^*, L_W^* and L_E^*, renders it optimal to refrain from investment. That is, equation (15) holds everywhere along the path from the current state to the steady state under agricultural abatement alone. A 10% decrease in the annual carry-over of phosphorus, α_P, also results in no investment. In these two cases, the optimal agricultural abatement rate without wastewater treatment is markedly lower than in the baseline scenario. As nutrient accumulation is also more moderate, agricultural abatement suffices to reduce the stock measured in nitrogen equivalents to below the level in the baseline case. The anticipated value of abatement cost savings and of damage avoided through the construction of wastewater treatment facilities does not offset the investment cost.

Among the economic parameters, those describing the damages associated with eutrophication are the most uncertain. As an alternative parameterization of the damage function, we lowered the willingness to pay for a reduction in eutrophication by 50% relative to the baseline case. The resulting damage function parameters are $a_d = -98,484, b_n = -1,467,100$ and $c_n = 179,200$. The decision to invest is robust to the willingness to pay measure: immediate investment was still optimal. The same holds for a 10% increase in the operating costs of wastewater treatment facilities. In contrast, a 10% decrease in the agricultural abatement cost parameter, c_A, makes it optimal to refrain from investment. The steady state agricultural abatement rate and stock levels remain relatively close to the baseline case, but the smaller abatement costs suffice to make investment unprofitable.

Changes in the probability of delay, the maximum construction time, or the investment outlay only affect the value of the investment while the steady state policies and stock levels remain unchanged. In case of an 80% probability of delay, or a maximum construction time of 20 years, immediate investment was still optimal. Finally, a 100% increase in the cost of investment K postpones the investment slightly: it is optimal to invest after an initial phase of agricultural abatement. Figure 6 depicts the value of the investment and the state path under agricultural abatement. Initially, the value of investment falls below the value of the current and expected future rewards under agricultural abatement alone. The white region in Figure 6 contains the state space for which $J^0(N, P) - K < V^0(N, P)$ holds and refraining from investment is optimal. The shaded region contains the state space for which $J^0(N, P) - K \geq V^0(N, P)$ and investment is optimal. The current stock level lies in the region where refraining from investment is optimal. However, as the stocks of nitrogen and phosphorus evolve along the path associated with the optimal agricultural abatement policy, a region is reached where condition (16) holds, and investment becomes optimal.

[Table 3 about here]

We conducted a variety of additional experiments with alternative values of the ecological and economic parameters. Due to space limitations, the results are not reported here but are available from the authors upon request. All in all, the results are not excessively sensitive to reasonable changes in model parameters. The steady state nitrogen stock levels range from 55,000 to 61,000 tonnes. Abatement technology has a greater effect on the steady state phosphorus stock, which ranges from 2300 to 2900 tonnes when wastewater treatment is available, and from 4000 to 4800 tonnes when only agricultural abatement is possible. Due to the curvature of the agricultural abatement cost function, parameter changes affect agricultural abatement policy more than they do wastewater treatment policy.

5 Conclusion

We have examined optimal abatement of nutrient loading when two sources contribute to the nutrient load: agricultural loading and municipal wastewater. The program to reduce the nutrient loads comprises two potential phases: initially, small-scale measures can be adopted to reduce agricultural loading; if investment is then undertaken to establish wastewater treatment facilities, nutrient loads from municipal wastewater can also be controlled. On this basis we have formulated an investment and abatement model that incorporates both abatement technologies, as well as the irreversible investment required to set up wastewater treatment facilities.

The model developed has been applied to study optimal abatement policies for the Finnish coastal waters of the Gulf of Finland, which are exposed to agricultural nutrient loads from southwestern Finland and wastewater from St. Petersburg. The empirical results suggest that it would be optimal to invest immediately in construction of wastewater treatment capacity that would enable processing all of St. Petersburg's wastewaters. Wastewater treatment would then become the principal abatement measure. However, the result that the investment should be undertaken immediately is not self-evident. While the perceived damage, or willingness to pay for reducing eutrophication, is high enough to justify active measures, the optimal allocation of resources to control nitrogen and phosphorus was found to hinge on the ecological parameters: a sensitivity analysis showed that the decision to invest in wastewater treatment capacity is robust to changes in the parameters describing the damage, which are the most uncertain economic parameters, whereas relatively small changes in the ecological parameters reversed the outcome. The finding underlines the need to reconcile economic and ecological models to provide guidelines for nutrient abatement policies that are sound in both areas. Moreover, the emphasis of previous economic analyses on nitrogen may have overly distracted attention from phosphorus discharges, which also play a significant role in the dynamic eutrophication process and which our findings suggest are the principal abatement target.

If the discrete investment cost structure hinders municipalities from investing at the optimal time due to difficulties in arranging financing, cost-efficient abatement measures cannot be obtained in a timely manner. An obvious policy conclusion is that long-term investment programs may be required as part of policy implementation. Finally, an interesting extension to this study would be to explicitly consider the uncertainties inherent in the management of nutrient loads in agriculture in particular. Further, in our model the coastal ecosystem is managed by a single authority. A worthwhile dimension to accommodate in future research would be transboundary cooperation.

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References

[1] Amit, R. (1986). Petroleum reservoir exploitation: switching from primary to secondary recovery. *Operations Research* 34: 534-539.

[2] Anon. (2004). Nutrient and sediment tributary strategy, Government of the District of Columbia, Department of Health, Environmental Health Administration, June 2004. 90 p. http://app.doh.dc.gov/services/administration_offices/ environmental/watershed/pdf/dc_tributary_strategy_2004o.shtm

[3] Boesch, D., Hecky, R., O'Melia, C., Schindler, D. and Seitzinger, S. (2006). Eutrophication of Swedish seas, Final Report, Swedish Environmental Protection Agency, Report 5509, March 2006. 68 p.

[4] Carpenter, S.R., Ludwig, D. and Brock, W.A. (1999). Management of eutrophication for lakes subject to potentially irreversible change, *Ecological Applications* 9(3): 751-771.

[5] EEA (2005). Effectiveness of urban wastewater treatment policies in selected countries: an EEA pilot study, European Environment Agency Report No 2/2005, EEA Copenhagen. 51 p.

[6] Elofsson, K. (2003). Cost-effective reductions of stochastic agricultural loads to the Baltic Sea. *Ecological Economics* 47: 13-31.

[7] Freeman III, A. M. (1995). The benefits of water quality improvements for marine recreation: A review of the empirical evidence. *Marine Resource Economics* 10: 385-406

[8] Gabric, A. J. and Bell P. R. F (1993). Review of the Effects of Non-point Nutrient Loading on Coastal Ecosystems. *Australian Journal of Marine and Freshwater Research* 44: 261-83.

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

[10] Gren, I.-M. and Folmer, H. (2003). Cooperation with respect to cleaning of an international water body with stochastic environmental damage: the case of the

Baltic Sea, *Ecological Economics* 47: 33-42.

[11] Gren, I.-M., Söderqvist, T., and Wulff, F. (1997). Nutrient reductions to the Baltic Sea: Ecology, costs and benefits, *Journal of Environmental Management* 51: 123-143

[12] Hart , R. (2003). Dynamic pollution control – time lags and optimal restoration of marine ecosystems. *Ecological Economics* 47: 79-93.

[13] Hart, R., and Brady, M. (2002). Nitrogen in the Baltic Sea – policy implications of stock effects, *Journal of Environmental Management* 66: 91-103

[14] Helin, J., Laukkanen, M. and Koikkalainen, K. (2006). Abatement costs for agricultural nitrogen and phosphorus loads: a case study of South-Western Finland. Mimeo, MTT Agrifood Research Finland.

[15] Kauppila P., Korhonen, M., Pitkänen, H., Kenttämies, K., Rekolainen, S. and Kotilainen, P. (2001). Loading of pollutants. In: Kauppila, P. & Bäck, S. (eds) The state of Finnish coastal waters in the 1990s. The Finnish Environment 472: 15–29.

[16] Kiirikki, M., Lehtoranta, J. M., Inkala, A., Pitkänen, H., Hietanen, S., Hall, P. O. J., Koponen, J. and Sarkkula, J. (2006). A simple sediment process description suitable for 3D-ecosystem modelling - model development and testing in the Gulf of Finland. Forthcoming in *Journal of Marine Systems.*

[17] Kiirikki, M., Rantanen, P., Varjopuro, R., Leppänen, A., Hiltunen, M., Pitkänen, H., Ekholm, P., Moukhametsina, E., Inkala, A., Kuosa, H. and Sarkkula, J. (2003). *Cost effective water protection in the Gulf of Finland. Focus on St. Petersburg.* The Finnish Environment, 632, Finnish Environment Institute.

[18] Krasnoborodko, K.I., Alexeev, A.M., Tsvetkova, L.I., and Zhukova, L.I. (1999). The development of water supply and sewerage systems in St.Petersburg, *European Water Management* 2(4): 51-61.

[19] Lehtoranta J. (2003). Dynamics of sediment phosphorus in the brackish Gulf of Finland. Monographs of the Boreal Environment Research No. 24, 58 p.

[20] Ludwig, D., Carpenter, S. and Brock, W. (2003). Optimal phosphorus loading for a potentially eutrophic lake, *Ecological Applications* 13(4): 1135-1152.

[21] Makris, M. (2001). Necessary conditions for infinite-horizon discounted two-stage optimal control problems. *Journal of Economic Dynamics and Control 25: 1935-1950.*

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

[23] Markowska, A. and Zylicz, T. (1999). Costing an international public good: the case of the Baltic Sea. *Ecological Economics* 30(2): 301-316.

[24] Miranda, M. and Fackler, P. (2002). *Applied Computational Economics and Finance*, MIT press.

[25] Mäler, K-G., Xepapadeas, A. and de Zeeuw, A. (2003). The economics of shallow lakes, *Environmental and Resource Economics* 26(4):603-624.

[26] Neumann, T. (2000), Towards a 3D ecosystem model of the Baltic Sea, *Journal of Marine Systems* 25, 405-419.

[27] Nævdal, E. (2001). Optimal regulation of eutrophying lakes, fjords, and rivers in the presence of threshold effects. *American Journal of Agricultural Economics* 83:972-984.

[28] Pindyck, R.S. (2000). Irreversibilities and the timing of environmental policy, *FEEM Working Paper* No. 5 2000.

[29] Pitkänen, H. (ed.) (2004). Rannikko- ja avomerialueiden tila vuosituhannen vaihteessa, Suomen Itämeren suojeluohjelman taustaselvitykset, Suomen ympäristö 669, Suomen ympäristökeskus.

[30] Pitkänen H., J. Lehtoranta & A. Räike, (2001). Internal nutrient fluxes counteract decreases in external load: the case of the estuarial eastern Gulf of Finland, Baltic Sea. Ambio 30: 195–201.

[31] Pretty, J.N., Mason, C.F., Nedwell, D.B., Hine, R.E. (2003). Environmental costs of freshwater eutrophication in England and Wales. *Environmental Science and Technology* 37 (2), 201–208.

[32] A.C. Redfield, B.H. Ketchum and F.A. Richards (1963). The influence of organisms on the composition of seawater. In: M.N. Hill, Editor, The Sea, vol. 2, Wiley, New York.

[33] Savchuk, O. and Wulff, F. (1999). Modelling regional and large-scale response of the Baltic Sea ecosystem to nutrient load reductions, *Hydrobiologia* 393, 35-43.

[34] Shortle, J.S. and Abler, D.G. (1999). Agriculture and the environment. In J. van den Bergh (ed.), Handbook of Environmental and Resource Economics. Edward

Elgar, Cheltenhamn. p. 159-176.

[35] Söderqvist, T. (1996). Contingent valuation of a less eutrophicated Baltic Sea, Beijer Discussion Paper Series No. 88.

[36] Söderqvist, T. and Scharin, H. (2000). The regional willingness to pay for a reduced eutrophication in the Stockholm archipelago, Beijer Discussion paper No. 128.

[37] Tomiyama, K. (1985). Two-stage optimal control problems and optimality conditions. *Journal of Economic Dynamics and Control* 9:317-337.

[38] Tuominen L., Heinänen, A., Kuparinen, J. and Nielsen, L. (1998). Spatial and temporal variability of denitrification in the sediments of the northern Baltic proper. *Marine Ecology Progress Series* 172, pp. 13-24.

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

[40] Vodokanal (2005). Cost Effective Pollution Reduction Investments in St. Petersburg, Draft Final Summary Report, December 9, 2005.

[41] Wilson, M.A. and Carpenter, S.R. (1999). Economic valuation of freshwater ecosystem services in the United States: 1971-1997, *Ecological Applications* 9(3): 772-783.

[42] Ærteberg, G., Carstensen, J., Dahl, K., Hansen, J., Rygg, B., Soerensen, K., Severinsen, G., Nygaard, K., Schrimpf, W., Schiller, C., Druon, J. N. and Casartelli, S. (2001). *Eutrophication in Europe's coastal waters.* European Environment Agency. Topic Report No. 7. 115 p.

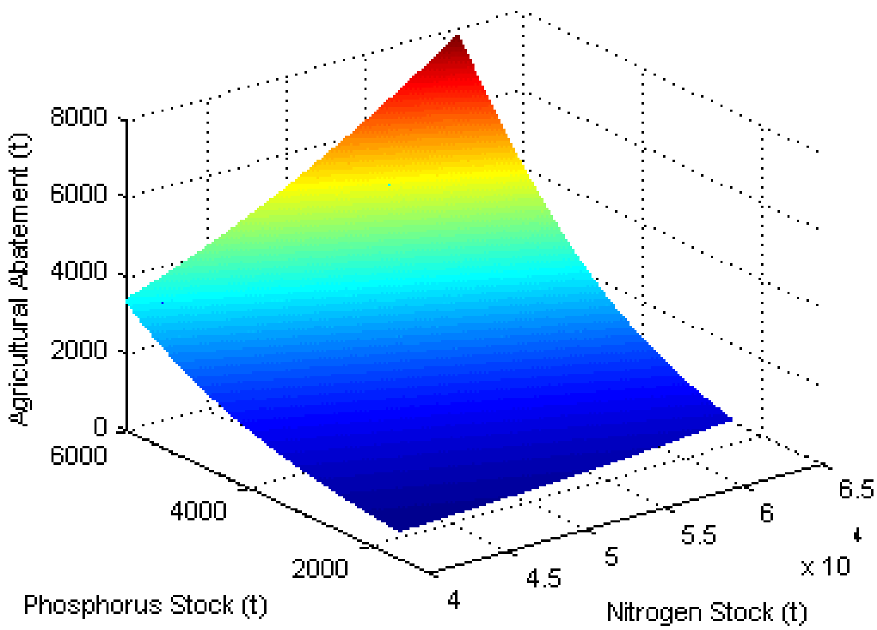

Figure 1. Optimal agricultural abatement policy when wastewater treatment is not possible.

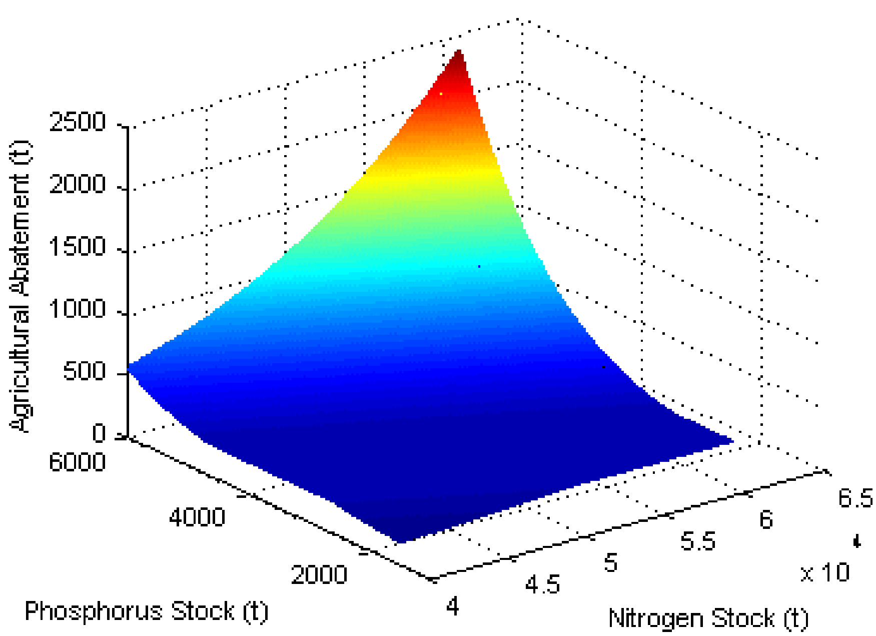

Figure 2. Optimal agriculural abatement policy when wastewater treatment is possible.

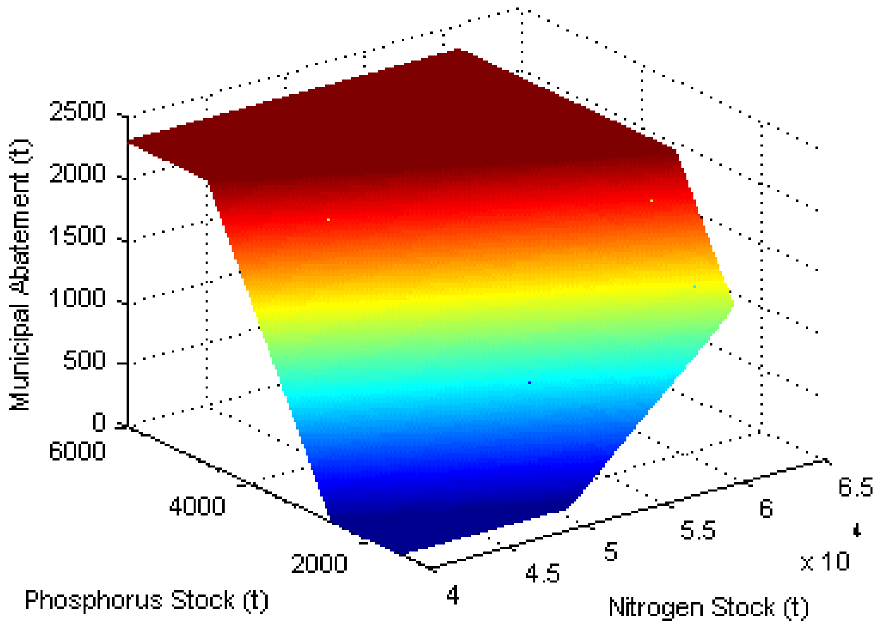

Figure 3. Optimal wastewater treatment policy.

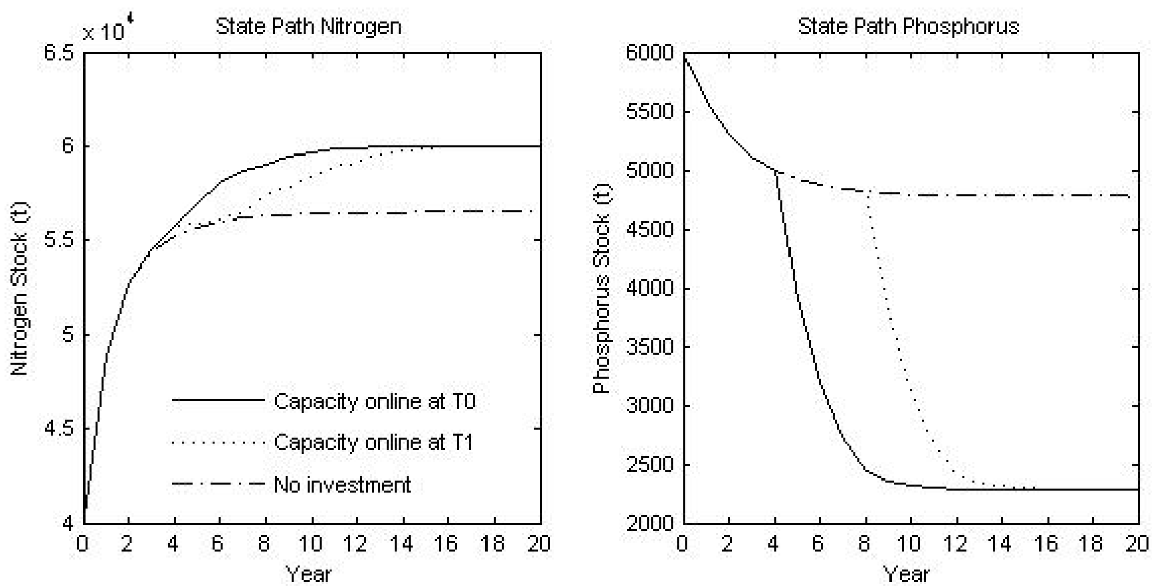

Figure 4. Optimal state paths for three possible cases: (i) wastewater treatment facilities are online at time $T_0 = 4$ following investment at time $t = 0$, (ii) facilities are online at time $T_1 = 8$ following investment at time $t = 0$, (iii) investment is not undertaken and hence only agricultural loads are abated.

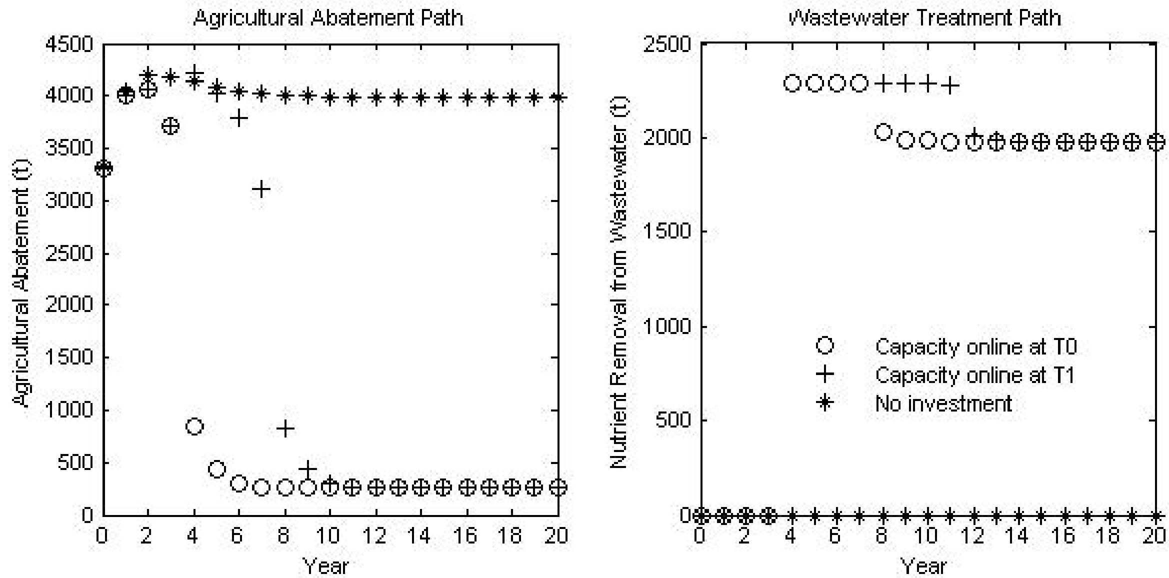

Figure 5. Optimal policy paths for three possible cases: (i) wastewater treatment facilities are online at time $T_0 = 4$, (ii) facilities are online at time $T_1 = 8$, (iii) investment is not undertaken.

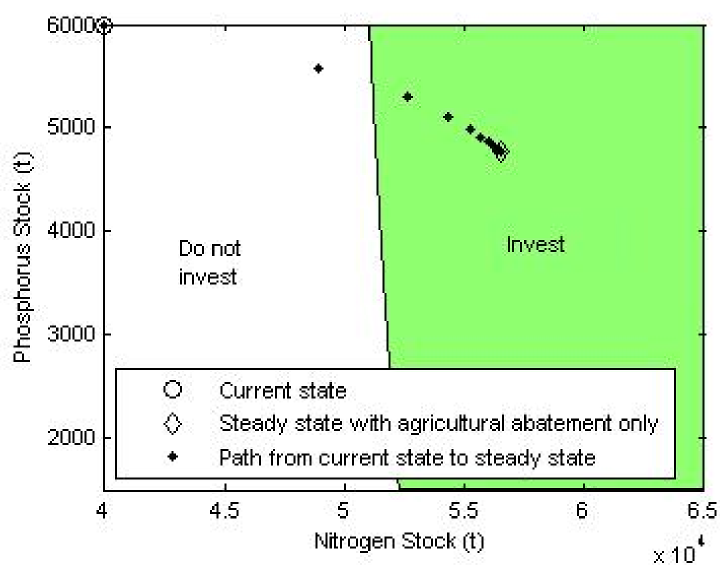

Figure 6. Illustration of a case where investment is undertaken after an initial phase of agricultural abatement only. At the initial state, refraining from investment is optimal. However, along the optimal path with agricultural abatement only, a state is reached where the value of investing exceeds that of agricultural abatement only (the shaded region in the figure), and investment becomes optimal.

Table 1. Parameters used in the simulation

Nutrient stock dynamics	
α_N	0.50
α_P	0.65
L_{AN}^*	7764 t
L_{AP}^*	522 t
q_A	0.0039
L_{WN}^*	4756 t
L_{WP}^*	640 t
q_W	0.45
L_{EN}^*	19714 t (exogenous nitrogenload)
L_{EP}^*	524 t (exogenous phosphorus load)
P_0	6000 t
N_0	40000 t
NE_0	83200 t
Costs of agricultural abatement, $C_A\left(R_A^t\right) = c_A \cdot \left(R_{AN}^t\right)^2$	
c_A	1.68 euro/t^2
Operational costs of wastewater treatment, $C_W(R_{NW}) = c_w R_{NW}$	
c_w	4460 euro/t
Investment costs and construction time	
K	330 million euro
$\sum p_i X_i$	87 million euro
p	0.5
T_0	4 years
T_1	8 years
Damage	$D(NE) = a_d + e^{b_d/(NE-c_d)}$
a_d	-170190
b_d	-1536900
c_d	179200

Table 2. Steady state nutrient stocks and abatement levels under the optimal policy

(i) No investment - agricultural abatement only		
Variable	Description	Value
R_{NA}	Reduction in agricultural nitrogen load	3980 t
R_{PA}	Reduction in agricultural phosphorus load	16 t
R_{NW}	Reduction in municipal nitrogen load	-
R_{PW}	Reduction in municipal phosphorus load	-
N	Nitrogen stock	56500 t
P	Phosphorus stock	4776 t
E	Nitrogen equivalents	90900 t
$C(R_{NA})$	Cost of agricultural nutrient abatement	27 million euro
$C(R_{NW})$	Cost of municipal nutrient abatement	-
$D(E)$	Damage	36 million euro
(ii) Investment is undertaken - agricultural abatement and wastewater treatment		
Variable	Description	Value
R_{NA}	Reduction in agricultural nitrogen load	254 t
R_{PA}	Reduction in agricultural phosphorus load	1 t
R_{NW}	Reduction in municipal nitrogen load	1976 t
R_{PW}	Reduction in municipal phosphorus load	890 t
N	Nitrogen stock	60 000 t
P	Phosphorus stock	2278 t
E	Nitrogen equivalents	76 400 t
$C(R_{NA})$	Cost of agricultural nutrient abatement	107600 euro
$C(R_{NW})$	Cost of municipal nutrient abatement	9 million euro
$D(E)$	Damages	2.9 million euro

Table 3. Results of the sensitivity analysis

	Δ	Steady State Variable Values (t) R_A	R_W	N	P	Timing of Investment
10% decrease in L^*_A, L^*_W, and L^*_E	1	253	1095	55323	2927	No investment
	0	1484	-	55051	4321	
10% decrease in α_P	1	284	1634	60631	2291	No investment
	0	2456	-	59555	4042	
10% increase in c_W	1	278	1920	60069	2348	Invest immediately
	0	3982	-	56502	4776	
10% decrease in c_A	1	281	1971	59962	2283	No investment
	0	4132	-	56201	4774	
50% decrease in WTP	1	253	1585	60790	2779	Invest immediately
	0	2859	-	58749	4788	
Probability of delay $(1-p) = 0.8$	1	253	1976	60008	2277	Invest immediately
	0	3982	-	56502	4776	
Maximum construction time T_1 is 20 years	1	253	1976	60008	2277	Invest immediately
	0	3982	-	56502	4776	
100% increase in investment cost K	1	253	1976	60008	2277	Invest after a lag
	0	3982	-	56502	4776	

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