Water Economics and Policy, Vol. 3, No. 2 (2017) 1650031 (27 pages)

DOI: 10.1142/S2382624X16500314

The Role of Fisheries in Optimal Eutrophication Management

Antti Iho*,¶, Heini Ahtiainen*, Janne Artell*, Outi Heikinheimo*, Pirkko Kauppila†, Anna-Kaisa Kosenius‡, Marita Laukkanen§, Marko Lindroos‡, Soile Oinonen†, Kimmo Ollikka§, Katja Parkkila‡, Yulia Pavlova*, Heikki Peltonen†, Eija Pouta* and Laura Uusitalo†

*Natural Resources Institute Finland (Luke)
Viikinkaari 4, 00790 Helsinki, Finland

†Finnish Environment Institute (SYKE), Mechelininkatu 34A, Helsinki, Finland

‡University of Helsinki, Yliopistonkatu 4, 00100 Helsinki, Finland

§VATT Institute for Economic Research
Arkadiankatu 7, 00100 Helsinki, Finland
¶antti.iho@luke.fi

Received 28 September 2015
Revised 27 November 2015
Accepted 21 July 2016
Published 6 October 2016

We analyze dynamically optimal eutrophication management using two controls, targeted fishing and reduction of external nutrient loads. Fishing removes nutrients from the water ecosystem, and the size of the fish stock also influences eutrophication through food web effects and other mechanisms. We show that fisheries have a role to play in cost-efficient water quality management in combination with external load reductions. Our numerical application considers phosphorus driven eutrophication, agricultural phosphorus abatement and fisheries targeted on cyprinids on a coastal bay in the Baltic Sea. The socially and privately optimal intensity of fishing efforts, phosphorus abatement and the resulting water quality are influenced by damages, revenues and costs. Furthermore, we show that the link between cyprinid fish stock and water quality, and the form of the fishing industry — sole owner or open access — have joint dynamics that lead to very different outcomes. A weak link between cyprinid stock and water quality is associated with socially optimal stock close to its maximum sustainable yield. This maximizes phosphorus removal. With a strong link, socially optimal stock and phosphorus removal are low. Coincidentally, open-access fishing sometimes yields socially desirable outcome automatically — a market failure in industry structure may counteract eutrophication.

Keywords: Dynamic optimization; eutrophication; fisheries; phosphorus; agriculture.

1. Introduction

In freshwater ecosystems and coastal marine areas, eutrophication is a widespread phenomenon that adversely affects the ecosystem services which these areas can provide. Eutrophication is mainly caused by excessive anthropogenic nutrient release (Smith 2003; Diaz and Rosenberg 2008). Highly eutrophic lakes, rivers or marine areas are characterized by high nutrient concentrations, high phytoplankton biomass — often including nuisance blooms of phytoplankton — and oxygen deficiency in their deeper waters (Horne and Goldman 1994). These effects cause cascading changes in the structure and function of the ecosystem in its entirety.

Fishing also alters aquatic ecosystems directly by removing fish and indirectly by inducing changes in the systems' food-webs. Typically, fisheries first deplete the stocks of large predatory fish, whereupon decreased predation pressure enables increases in secondary consumers, such as zooplanktivorous species, and increases feeding on herbivorous zooplankton (Pauly *et al.* 1998; Christensen 2014). In eutrophic European freshwater lakes high biomasses of cyprinid fish recirculate nutrients back to the water column through excretion and bioturbation (i.e., reworking the bottom sediment). Anthropogenic nutrient loading and changes in the fish community thus jointly influence ecosystems' primary production, that is, phytoplankton growth.

Nutrient loading has a direct effect on the nutrient concentration of a waterbody, while fish and fisheries influence it indirectly. At least three major mechanisms of indirect influence have been identified: effects on food webs through changes in zooplankton communities; recycling of nutrients back to the water column; and permanent removal of nutrients in the harvested fish biomass. Socially optimal eutrophication management should consider not only nutrient loading from land but simultaneously also the impacts of fisheries and fish stocks on the ecosystem.

The basic efficiency condition equates marginal costs of control at all sources of pollution, evaluated at the receptor point (Baumol and Oates 1988). We argue that due to their effect on water quality, fisheries should be included in the framework of efficient eutrophication management. Traditionally, the economics of fisheries has focused on over-fishing and over-capitalization triggered by open access to fish stocks. The literature has promoted efficiency by recommending restrictions on overall fishing effort in the form of transferable or fixed individual quotas, taxes, TACs, capital constraints, etc. (see, e.g., Clark 2010; Costello *et al.* 2008; Nielsen *et al.* 2012). In the present context, socially optimal fisheries management may well call for more intensive fishing. This is particular the case if dense populations of zooplankton-feeding or sediment-reworking fish species — low in market value — maintain or even accelerate eutrophication. In other words, environmental considerations point to a reversal of the "normal" fisheries problem.

There are studies linking nutrient loading and production capacity of commercial fisheries. One of the earliest considers fertilizing lakes to increase productivity (Hasler and Einsele 1948). A study more closely related to ours is by Knowler *et al.* (2001) who link fish stock dynamics in open-access fisheries to ambient nutrient pollution and to a regime shift in the water ecosystem. Their analysis highlights the fisheries' benefits from pollution abatement. Our analysis considers fisheries as part of eutrophication management.

Economic literature on eutrophication management has focused on controlling external nutrient loads to surface waters. This orientation seemingly ignores the fact that, by and large, the society controls the focus and intensity of fisheries. In actual water protection, large-scale fisheries have often been used for ecosystem restoration (Sass *et al.* 2006; Nürnberg *et al.* 2012). There have also been attempts to include existing recreational fisheries in water quality management. Yet, the literature contains no studies that combine fisheries of different scales in dynamic water quality models that allow for control of external nutrient loading. Eutrophication is a stock pollution problem, as defined by Keeler *et al.* (1972). To our knowledge, analyses of optimal eutrophication management have thus far not considered the option of influencing the level of the pollution stock directly (even though this practical option is sometimes discussed, see e.g., Mäler *et al.* 2003). Effectively, fisheries provide the regulator with an option to influence the stock, in addition to controlling external sources. We contribute to literature by incorporating both management options in our optimal fisheries–eutrophication management framework.

Our main goal is to examine whether, and how intensively, the society should utilize targeted fisheries in eutrophication management; and how such activities should be combined with external nutrient control. To answer these questions, we develop a dynamic optimization model for eutrophication management through the reduction of external loads and through targeted fishing. We apply the model to cyprinid fisheries in the coastal bay of Mynälahti, South-West Finland, and to agricultural phosphorus abatement at its watershed. To keep the logic of the interplay of fisheries and external abatement as clear as possible, we confine the model to a single nutrient: phosphorus.[1]

[1] In most fresh water ecosystems phosphorus is the critical nutrient for algae growth (Schindler 2012). Including nitrogen to our model would add a new dimension but would not change the basic outcomes qualitatively.

Our results show that fisheries have a role in water quality management — in combination with external load reductions. The results quantify the importance of the yet poorly understood food web effects on optimal policies. If the cyprinid fish stock has limited or non-existent food web effects, the socially optimal policy is to have the fish stock close to its maximum biological sustainable yield. Doing so maximizes nutrient removal with the catch. As the negative effect of the fish stock on water quality increases, it becomes socially optimal to reduce the stocks to ever-lower levels, even though this increases the costs and decreases the annual phosphorus removal with the harvest. Due to the trade-off in phosphorus removal and the fish stock effect, the structure of fisheries industry affects the social welfare associated with the unregulated case. Open access tends to drive stocks to lower levels which in our case may be socially desirable. Over-fishing may thus be beneficial for eutrophication management. The result is akin to a monopoly being more cautious in extracting non-renewable resources: market failure in industry structure counteracts another.

The rest of the paper is organized as follows. The next section describes the model which is then parametrized in the third section. The results are presented in the fourth section and the final section discusses the policy implications and avenues for further research.

2. Model

Assume a waterbody that is a source of recreational benefits and profits from single-species fisheries. The former depends on water quality, which is influenced by the stocks of (total) phosphorus and cyprinid fish in the waterbody. The phosphorus stock is driven by external loading from agriculture, which can be abated at a cost, and by the phosphorus removed with the harvested fish biomass.

We consider three alternative decision makers: (1) a profit-maximizing individual fisher operating in an open-access fishery, (2) a profit-maximizing fisher operating as the sole owner of the fish stock and (3) a social planner who takes into account profits from fisheries and the non-market values of water quality. All decision makers are price takers.

2.1. *Socially optimal fisheries and phosphorus abatement*

We begin from the social planner's dynamic optimization problem. A social planner maximizes the discounted stream of profits from fisheries (π_f) and profits from agriculture (π_a) minus the environmental damages affecting recreation (D):

$$\max_{z_t, S_t} \sum_{t=0}^{\infty} \beta^t [\pi_f(X_t, S_t) + \pi_a(z_t) - D(Q_t, X_t)], \tag{1}$$

s.t.

$$Q_{t+1} = (1-\rho)Q_t + \bar{q} - z_t - \xi(X_t - S_t), \tag{2}$$

$$X_{t+1} \equiv F(S_t) = S_t + rS_t\left(1 - \frac{S_t}{K}\right). \tag{3}$$

The choice variables are stock escapement (S_t) and external phosphorus abatement (z_t). Stock escapement refers to the biomass of fish surviving into the season following the harvest while X_t refers to biomass before the harvest. β is the discount factor. In each period, the fraction ρ of the phosphorus stock (Q_t) decays via water outflow and sedimentation [Eq. (2)].[2] External phosphorus loading $(\bar{q} - z_t)$ adds to the next period's stock, whereas fisheries remove the quantity $\xi(X_t - S_t)$ from the stock, ξ denoting the phosphorus concentration of the catch. The growth of the fish stock is a function of the escapement S_t [Eq. (3)]. Intrinsic growth rate is denoted by r and K is the carrying capacity of the fish stock.[3]

The profits from fisheries are modeled following Laukkanen (2003):

$$\pi_f = p(X_t - S_t) - \gamma \ln\left(\frac{X_t}{S_t}\right), \tag{4}$$

where p is the market price of fish, X_t the fish stock in the beginning of the season and S_t the escapement; their difference represents the harvest. The marginal cost of catching one more fish is $/x$, where x is the size of the fish stock at any given time. The total harvesting cost of the period is thus $\gamma \ln(X_t/S_t)$.[4]

The profits from agriculture (with zero abatement) are normalized to zero. Effectively, this transforms the planner's problem into one of minimizing abatement costs:

$$\pi_a = -\frac{1}{2}\alpha z_t^2, \tag{5}$$

where the marginal abatement cost is αz.

[2]The model treats phosphorus sedimentation in the same way as riverine outflow. It thus does not take into account the possibility of sediments losing their capability of binding phosphorus after the phosphorus stock has reached some threshold level. See, e.g., (Carpenter 2005; Mäler *et al*. 2003) for implications of non-convexities in controlling external phosphorus loads.

[3]Birth rate and mortality are implicit in the growth function which captures the net growth rate of the stock. With $S_t = K$ the net growth rate is zero; i.e., the birth rate and natural mortality are equal. For simplicity, the fish growth is not affected by the phosphorus stock. In the sensitivity analysis, we will vary the intrinsic growth rate (r) and the carrying capacity (K) to see how they would affect the optimal solutions.

[4]The effort required to harvest one unit is $1/x$. The integral over fishing costs during the annual harvest is $\gamma \int_S^X \frac{1}{x}dx = \gamma(\ln(X) - \ln(S)) = \gamma \ln \frac{X}{S}$.

The environmental damage depends linearly on water quality:

$$D = \sigma\left[\kappa + \eta \ln(Q_t^c) - \omega\left(1 - \frac{X_t}{K}\right)\right], \tag{6}$$

where σ is the constant marginal damage.[5] The expression in square brackets yields a scalar valued usability index used in Finland until 2008. The lower the value, the higher the water quality. The index is a sum of two terms. The first one, $(\kappa + \eta \ln(Q_t^c))$, captures the effect of phosphorus stock and the second one, $\left(\omega\left(1 - \frac{X_t}{K}\right)\right)$, the effect of the fish stock on water quality. Steady state harvesting and the associated fish stock thus enter the damage function in two ways: (i) Harvesting lowers the phosphorus stock (Q), as expressed in Eq. (2) and (ii) Lowering the fish stock (X) from its carrying capacity level (K) may influence water quality. Henceforth, we will call the latter as the cyprinid effect. It is the effect the existing fish stock has on water quality via recycling of sediment nutrients and via changes in food web.

The cyprinid effect depends on the deviation of the actual stock size from its potential maximum.

The effect is scaled by a parameter ω. If the fish stock is at its carrying capacity, the cyprinid effect is zero regardless of the value of ω and the water quality is determined solely by the phosphorus stock. Also, if $\omega = 0$, the fish stock as such does not affect eutrophication, regardless of the stock size. With positive values of ω, the effect is determined by the ratio of the fish stock and the carrying capacity. The lower the stock, the stronger the positive effect on water quality. The theoretical maximum effect — equal to ω — is achieved when $X_t = 0$, that is, when the entire fish stock is removed.

Because of its complexity and limited understanding, the cyprinid effect is modeled in a very general way. The only assumption made in the theoretical model is that ω obtains weakly positive values, i.e., $\omega \geq 0$, indicating neutral to detrimental effect on the water quality. While a larger stock of some fish species, particularly predatory fish, has been theorized to have positive effects of water quality (Søndergaard *et al.* 1997; Skov and Nilsson 2007), the relationship between cyprinid fish and water quality has repeatedly been proven to be negative (e.g., Horppila and Kairesalo 1990; Breukelaar *et al.* 1994; Hansson *et al.* 1998; Horppila *et al.* 1998). A numerical application in Section 3.1 will present all key results for a wide range of values of ω.

[5]Equation 6 conditions damage to phosphorus concentration (Q_t^c), which is directly obtainable from the stock of phosphorus: $Q_t^c = \frac{Q_t}{Vol}$, where Vol is the volume of the waterbody. We will use the terms phosphorus stock and concentration interchangeably.

Assuming positive abatement and harvesting choices, their optimality conditions are (the complete set of optimality conditions and their derivation is presented in the Appendix):

$$\alpha z = \frac{\beta D_Q}{1-\beta(1-\rho)}, \tag{7}$$

$$p - \frac{\gamma}{S} = \beta F'(S)\left[\left(p - \frac{\gamma}{X}\right) - D_X\right] - \frac{\beta\xi(1-\beta F'(S))}{1-\beta(1-\rho)} D_Q, \tag{8}$$

where $D_Q = \frac{\sigma\eta}{Q^c}$ is marginal damage of phosphorus in the waterbody and $D_X = \frac{\sigma\omega}{K}$ marginal damage of the fish stock. Condition 7 states that at the optimal steady state — with positive abatement and fishing effort — the marginal abatement cost (αz) equals the shadow value of the phosphorus stock, discounted back one period. This condition is a classic result in stock pollution problems.

Equation (8) captures the optimality conditions of harvesting when fish stock's water quality effects are acknowledged. It indicates that the steady-state escapement, which is determined by harvesting, must equalize the marginal profits $(p - \frac{\gamma}{S})$ with a set of different marginal effects from an extra unit of escapement.

The first marginal effect on the right-hand side $(\beta F'(S)(p - \frac{\gamma}{X}))$ is the discounted value of increased escapement. The marginal increase in escapement, $(F'(S))$, is multiplied by the marginal revenue of an extra unit, discounted back one period, $\beta(p - \frac{\gamma}{X})$.[6] If there were no damages associated with either the fish stock or phosphorus stock (i.e., $D_X = D_Q = 0$), this would be the only term on the right-hand side of Eq. (8) and the optimality conditions would coincide with the classic conditions obtained in the case of sole ownership.

The second marginal effect, $(-\beta F'(S)D_X)$, depends on the marginal damage (D_X) the fish stock causes to water quality. Above, we justified that for cyprinid fish $\omega \geq 0$ and therefore $D_X \geq 0$. The term would thus be (weakly) negative, decreasing the optimal escapement (increasing harvesting efforts) and reducing the steady-state fish stock.

The third term, $(\frac{\beta\xi(1-\beta F'(S))}{1-\beta(1-\rho)} D_Q)$, captures the effect of marginal escapement on phosphorus removal and thereby on environmental damage. The marginal damage of the phosphorus stock is positive $(D_Q > 0)$. The sign of the third term thus depends on the magnitude of $\beta F'(S)$. Depending on escapement (S), a marginal change in escapement level may either increase or decrease the following period's steady-state harvest, that is, the amount of phosphorus removed. Harvesting

[6]Note that the current-period marginal profit increases in escapement: the higher the stock when we stop harvesting (i.e., the higher the escapement), the smaller the term $\frac{\gamma}{S}$. The value of the left-hand side of the equation thus asymptotically approaches $p - \frac{\gamma}{X}$ from below.

(and thus phosphorus removal) is higher, the closer the fisheries operates to the biological maximum sustainable yield (MSY).

2.2. *Private optima*

Private fisheries do not take into account the eutrophication caused by the phosphorus stock or the effects that fisheries and fish stocks have on water quality. We consider two forms of private fisheries: open access and sole ownership. The latter has a perfect control of all individual (or one representative) fishing units it comprises. Therefore, it has a perfect control over the development of the fish stock. Its maximization problem becomes

$$\max_{S_t} \sum_{t=0}^{\infty} \beta^t \left[p(X_t - S_t) - \gamma \ln\left(\frac{X_t}{S_t}\right)\right]$$

s.t.

$$X_{t+1} = F(S_t).$$

The privately optimal steady state escapement becomes:

$$p - \frac{\gamma}{S} = \beta F'(S)\left(p - \frac{\gamma}{F(S)}\right). \tag{9}$$

On the other hand, a private fisher operating in an open-access fishery does not take the stock development into account as it is beyond the control of any individual. At the optimum, fishing efforts are set at the level where the unit cost of effort is equal to price: $\frac{\gamma}{X} = p$. Assuming identical fishing units, the open-access escapement is defined by:

$$p = \frac{\gamma}{S}. \tag{10}$$

The first order conditions illustrate the presence of two externalities in the model: Private decision makers do not take damage due to eutrophication into account in their optimization problems and open-access fisheries do not take into account the development of the fish stock. In the model, all three types of decision makers will thus generate different outcomes regarding profits, environmental damage and social welfare.

3. Numerical Application and Optimal Policies

Coastal waters are particularly vulnerable to the effects of fishing and other anthropogenic perturbations (Botsford *et al.* 1997). Accordingly, we have chosen to apply our model to a relatively small and shallow bay, Mynälahti, in the

Archipelago Sea, South-West Finland. The size of the bay is 8,060 hectares, mean depth about 3.5 m and water volume 282 million cubic meters. The turnover of water is fairly rapid as the total discharge of the three rivers flowing into the bay is about 177 million cubic meters. The long-term average (during July–September) phosphorus concentration has been 36.3 μg/L.[7] The bulk of external phosphorus loading originates from non-point sources, mainly agriculture. The field acreage of the basin is about 17,700 hectares. Commercial fishery in the bay is intensive.

The parametrization is illustrative. Many of the parameters are at least partly based on expert evaluations — primarily due to a lack of data.[8] The main purposes of the numerical application are to provide a quantitative feel for the model and to pinpoint the parameters most critical for our results. The application thus helps steer future research efforts. Detailed presentation of each parameter is given in the Appendix.

Table 1 collects and presents the parameter values used in the application.

Table 2 presents the annual harvesting levels, associated escapement, external phosphorus abatement and phosphorus concentration at the steady state for our parametrization, using a cyprinid effect parameter $\omega = 1$. The second column from the left presents the social optimum, the third column the values associated with the sole owner's optimum and the column on the right the open-access fisheries optimum.

Table 2 quantifies the theoretical results discussed earlier. Our model encompasses two market failures: the tragedy of commons from open-access fisheries and the phosphorus externality that reduces the recreational value of the surface water. The social planner takes both externalities into account, the sole owner internalizes the open-access problem but ignores the phosphorus externality; and the open-access fisheries fail to take into account either of the market failures. Annual escapement is lowest at the social planner's optimum (at 309.8 tons); the associated harvesting is 79.9 tons. Note that the open-access harvesting levels are higher than this, 87.2 tons. That is, the open-access fisheries remove more phosphorus from the waterbody than the socially optimal level indicates. The difference is that the social planner also takes into account the negative effect of the fish stock itself and therefore keeps the fish stock permanently at a very low level. The social planner's optimal abatement is 0.47 tons and the associated phosphorus concentration of the waterbody is 30.0 μg/L.

[7]The values are based on unpublished data of the Finnish Environment Institute.

[8]We cannot, for instance, observe the fish stock sizes or the food web effect on water quality. The phosphorus dynamics of the bay are modeled in the simplest possible way; we consider a single fish species that combines two cyprinid fish, bream and roach.

Table 1. Fixed Parameter Values in Simulations

Parameter/Function		Value	Unit
i	Interest rate	0.03	—
β	Discount factor $\beta = \frac{1}{1+i}$	0.97	—
Transition of the phosphorus stock: $Q_{t+1} = (1-\rho)Q_t + \bar{q} - z_t - \xi(X_t - S_t)$			
ρ	Decay rate of the phosphorus stock	0.89	—
$\bar{q}$	Baseline loading	8.6	ton
ξ	Phosphorus concentration in cyprinid fish	0.75	per cent
Transition of the fish population: $X_{t+1} \equiv F(S_t) = S_t + rS_t(1 - \frac{S_t}{K})$			
r	Intrinsic growth rate	0.325	—
K	Carrying capacity of the fish stock	1,500	ton
Fishery profits: $\pi_f = p(X_t - S_t) - \gamma \ln(\frac{X_t}{S_t})$			
p	Market price for fish	400	EUR/ton
γ	Cost parameter for harvesting	460,000	EUR
Profits from agriculture: $\pi_a = \bar{\pi}_a - \frac{1}{2}\alpha z_t^2$			
$\bar{\pi}_a$	Normalized profits with zero abatement	0	EUR
α	Cost parameter for phosphorus abatement	113,600	EUR/ton
Damage function: $D = \sigma\left[\kappa + \eta \ln(Q_t^c) - \omega\left(1 - \frac{X_t}{K}\right)\right]$			
σ	Value of the marginal environmental damage	317,400	EUR
κ	Intercept of the phosphorus water quality function	−2.5	—
η	Slope of the phosphorus water quality function	1.3	—
ω	Cyprinid fish stock's direct effect on water quality	1	—

The sole owner harvests markedly less, the annual harvest being 58.1 tons (the associated escapement is 1,292.6 tons). The phosphorus concentration is 32.5 μg/L. Industry profits are EUR 3,014.

The open-access fisheries operate more intensively than the sole owner, with an annual harvest of 87.2 tons and escapement of 1,150 tons. This has two effects on

Table 2. Socially and Privately Optimal Steady State Phosphorus Abatement and Concentration, and Fisheries and Fish Stock

	Social Optimum	Sole Owner	Open Access
Annual Harvest $(X - S)$	79.9	58.1	87.2
Escapement (S)	309.8	1,292.6	1,150
Phosphorus Abatement (z)	0.47	0	0
Phosphorus Stock (Q)	8.47	9.17	8.93
Phosphorus Concentration (Q^c)	30.0	32.5	31.7
Profits $(\pi_f + \pi_a)$	$-85,937$	3,014	1,259
Damage (D)	375,234	611,723	576,513
Welfare (W)	$-461,170$	$-608,709$	$-575,254$

water quality. First, as harvesting is higher, phosphorus removal is higher too. Second, lower cyprinid fish stock decreases the detrimental cyprinid effect on water quality. The effects can be seen in the phosphorus concentration and the environmental damage, both of which are lower under optimum for open-access fisheries than that for the sole owner. The values for phosphorus concentration and environmental damage are higher than under the social optimum because there is no abatement under either of the two other optima. Open-access profits are lower than sole owner's but are not zero even though the marginal profits are: with a fixed price, each unit harvested before the marginal unit yields positive profits.

Overall, the structure of the fishing industry — whether based on open access or sole ownership — may influence in opposite directions in terms of the two market failures. The sole owner corrects the traditional over-fishing problem of the open-access fisheries. On the other hand, in our base case open-access fisheries are more favorable for the environment. The results could be reversed depending on whether open-access fisheries result in lower harvesting (and escapement) levels. This is typically the case in commercial fisheries around the world (see, e.g., Costello *et al.* 2008). If the harvesting levels were lower, the total effects on water quality would depend on the relative weights of the nutrient removal effect and the effect that the stock itself has on water quality. That is, it is not a general result that open-access fisheries are better for water quality. It is, however, a general result that open-access and sole-ownership fisheries differ in the optimal harvesting and escapement levels they generate, and that they therefore have differing effects on water quality.

3.1. *Numerical comparative statics*

The base case results provide a snap-shot of our model. Additional insights can be obtained by varying the important (and uncertain) parameters. Table 3 presents the

Table 3. Numerical Comparative Analysis — Social Optima for $\omega < \tilde{\omega}$ and $\omega > \tilde{\omega}$ where $\tilde{\omega} = 0.375$

	ω		ξ		p		K		r	
	$\omega < \tilde{\omega}$	$\omega > \tilde{\omega}$	$\omega < \tilde{\omega}$	$\omega > \tilde{\omega}$	$\omega < \tilde{\omega}$	$\omega > \tilde{\omega}$	$\omega < \tilde{\omega}$	$\omega > \tilde{\omega}$	$\omega < \tilde{\omega}$	$\omega > \tilde{\omega}$
Harvesting $(X - S)$	+	−	+	+	+	+	+	+	+	+
Escapement (S)	−	−	−	+	−	+	+	+	+	+
Fishing Effort (E)	+	+	+	−	+	−	+	−	+	+
Abatement (z)	+	−	+	+	+	+	+	+	+	+
Phosphorus Stock (Q)	−	+	−	−	−	−	−	−	−	−
Welfare (W)	+	+	+	+	+	+	+	+	+	−

directions of changes of social optima as the values of key parameters are increased. We give the values for ω greater and smaller than $\tilde{\omega} \approx 0.375$. This is the level of ω associated with a social optimum at the biological MSY of the fish stock. With some parameters, the changes in social optima are different on the two sides of the MSY.

Increases in the carrying capacity and growth rate have rather obvious effects on socially optimal choices. Both increase optimal harvesting and escapement, as well as abatement. The latter is caused by the concave form of our environmental damage function: improving water quality increases the optimal marginal abatement cost. Naturally, these combined decrease the optimal phosphorus stock. Higher carrying capacity increases (decreases) fishing efforts if the initial escapement level is initially above (below) the MSY level. The result is numeric and does not have a clear intuition: For $\omega > \tilde{\omega}$, the effect of increased harvesting levels are offset by the increase in the escapement level. For initial values above the MSY, the effort level increases even though both escapement and harvesting increase.

Let us then look in closer detail at some of the most interesting variations that increasing ω, ξ and p prompt in either the social optima or in optima of all three cases: social, open access and the sole owner. We first explore how the cyprinid effect parameter ω affects socially optimal fisheries and external phosphorus control.[9] Both panels of Fig. 1 depict ω on the horizontal axes. A value of zero means that the stock has no direct effect on water quality, a value of 2 means that it has a substantial effect [see Eq. (6)]. The vertical dashed line marks the level of ω for which the socially optimal steady state harvesting coincides with the biological MSY.

The left panel in Fig. 1 depicts harvesting and escapement levels in the socially optimal steady state (in tons per year). If the fish stock does not influence the water quality directly (i.e., $\omega = 0$), the socially optimal annual harvesting is somewhat over 100 tons and the escapement is about 1,000 tons. As expected, the optimal escapement decreases steadily as w increases. The annual harvesting first increases and then starts decreasing as the escapement falls closer to MSY and eventually below it. The lower the escapement, the more expensive it becomes to harvest a given amount of fish. Moreover, for landings below the MSY, phosphorus removal starts decreasing with ω. The social planner is thus willing to incur both higher fishery costs and lower phosphorus removal to keep the stock — that is itself the more harmful, the higher ω is — on a lower level permanently. If only phosphorus removal matters for the environment $(\omega = 0)$, the social planner considers only net revenues from fisheries and phosphorus removal.

[9]Both private decision makers' optima are insensitive toward variations in water quality, hence the focus on the social planner's optima only.

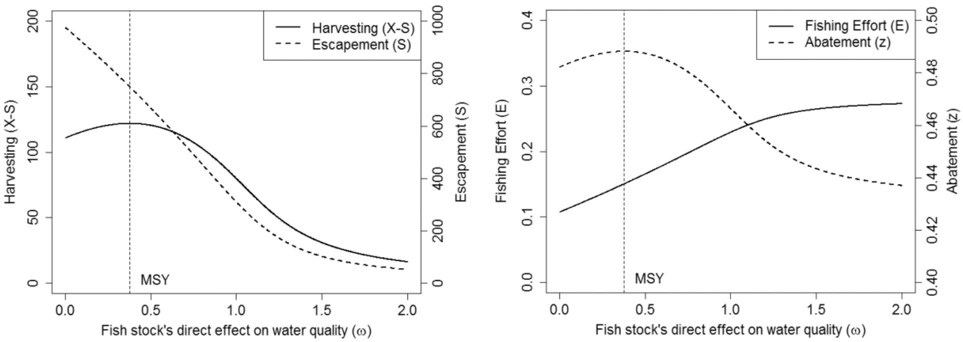

Figure 1. Socially Optimal Steady States for Alternative Direct Water Quality Effects of the Cyprinid Fish Stock (ω)

The right panel in Fig. 1 presents how the variation in ω affects optimal fishing efforts and external phosphorus abatement. The steadily rising dashed line denotes the fishing efforts, and the dotted curve rising before and descending after MSY the optimal abatement. If $\omega = 0$ and there is thus no direct water quality effect, the abatement is about 0.48 tons (out of the initial 8.6 tons). The left panel shows that a higher ω is associated with a lower steady state escapement. As the effort required to catch fish is inversely linked to escapement, the optimal effort increases with ω.

The property of the first increasing and then decreasing optimal abatement reflects the fact that the link between the general usability classification and phosphorus concentration is concave, and that the damage is linear in water quality. This makes the damage functions concave in phosphorus concentration. Therefore, the marginal damage decreases as the phosphorus concentration increases.[10] If both links were linear, the optimal abatement would not vary with ω. If the damage function were convex, the optimal abatement would first decrease, then increase.

Let us then look at the effect of phosphorus concentration in fish tissue on socially optimal escapement and harvesting (private optima are insensitive toward variations in phosphorus concentration). Because the effects depend on whether we are initially below or above the biological MSY, we consider two alternative cyprinid effects: $\omega = 0.2$ and $\omega = 1$.

[10]The economics literature typically assumes convex damage functions, although there are also examples of damage function specification that are concave at least for a certain range of the variables (see, e.g., Lichtenberg and Zilberman 1986; Kahn 1998). In our model, the concavity stems from the links between the general usability classification and phosphorus concentration. The main results of our analysis would not be qualitatively changed if we were to use a linear or convex damage specification.

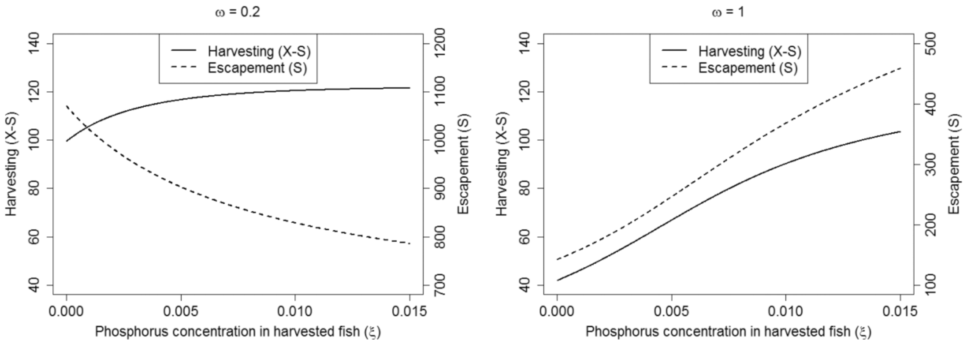

Figure 2. Socially Optimal Harvesting and Escapement for Alternative Phosphorus Concentration of Harvested Fish ξ for $\omega = 0.2$ and for $\omega = 1$

Figure 2 presents the socially optimal escapement and harvesting levels. As the phosphorus concentration in fish tissue increases, each harvested unit of fish improves water quality more. Other things being equal, the optimal harvesting level increases. If the escapement is initially above the MSY level (Fig. 2, left panel), higher harvesting is associated with lower escapement levels. If the escapement level is initially below the MSY (Fig. 2, right panel), both harvesting and escapement increase as the phosphorus concentration in fish tissue increases. The effect illustrates the trade-off between the two effects that fisheries have on water quality: the phosphorus removal effect gets stronger with increasing ξ while the cyprinid effect remains the same. Therefore, the social planner allows the escapement to change to increase the harvests and phosphorus removal.

Finally, we vary the long-term price of fish. We examine how price changes affect privately and socially optimal escapement (Fig. 3) and fishing efforts (Fig. 4). Social optima are evaluated at $\omega = 0.2$ and $\omega = 1$.

The dotted curve in Fig. 3 denotes the optimal steady-state escapement for the sole owner, the dashed curve the open-access optima and the solid curves the social optima for $\omega = 0.2$ and $\omega = 1$. The biological MSY is marked with a horizontal, dashed line. The price at which the open-access steady-state escapement coincides with the MSY is about 620 EUR/ton.

Price variation illustrates how the traditional fisheries externality related to open access starts to manifest itself as the market price increases. Uncoordinated fishing efforts increase rapidly with price and lower the associated escapement levels. A fishery that operates on the basis of sole ownership never allows the escapement to fall below the MSY, thus avoiding over-capitalization of the industry and over-exploitation of the fish stock.

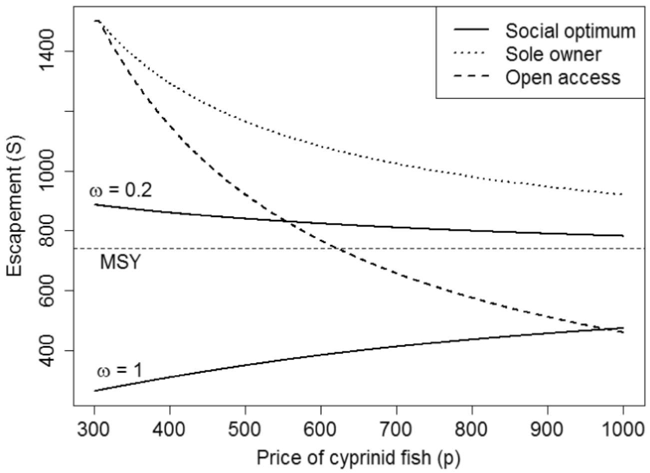

Figure 3. Optimal Steady State Escapement for Alternative Fish Prices, for $\omega = 0.2$ and $\omega = 1$

The increasing price may have opposite effects on privately and socially optimal fisheries. As the price increases, the increased profits from fisheries start weighing more in the social planner's optimal solution: a higher price increases the socially optimal escapement if it is initially below the MSY (the case with $\omega = 1$). Both sole owner and open-access optima start investing increasing effort in fisheries as the increasing price makes it profitable to harvest even from lower stocks (see also Fig. 4). We see that for a price of about 1,000 EUR/ton the open-access escapement would coincide with the social optimum associated with $\omega = 1$. At that price — by coincidence — one externality has partly removed the other. Even

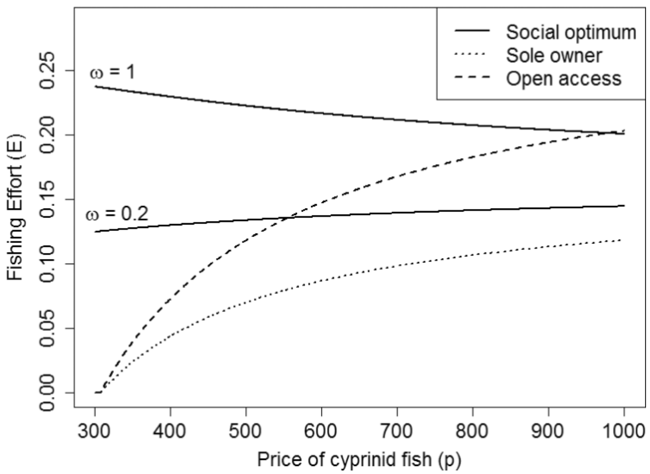

Figure 4. Optimal Steady State Fishing Effort for Alternative Prices, for $\omega = 0.2$ and for $\omega = 1$

though the open-access escapement is identical to the social optimum, private optima do not contain any abatement of external phosphorus loading from agriculture.

Figure 4 illustrates how the optimal fishing efforts develop as the price increases. The solid curves denote the socially optimal effort levels associated with the two values of ω, the dotted curve the level under sole owner access and the dashed curve the levels of the open-access fisheries.

The graph is a good illustration of the two sources of welfare which the social planner has to consider in the objective function [Eq. (1)]. The price increase makes it socially profitable to allow the escapement to rise (if initially below MSY) in order to increase the annual harvesting. There is a trade-off whereby the impairment in water quality from the higher escapement and increased sales revenues from fisheries is partly offset by the increased phosphorus removal due to higher harvesting levels. If initially below (above) MSY, the price increase results in social planner putting less (more) effort into fisheries because harvesting occurs under higher (lower) escapement levels, as shown in Fig. 3. Both private decision makers increase their effort as the price increases.

Both Figs. 3 and 4 highlight the differences between open-access and sole ownership for any given price level. These differences are well-established: Open-access leads to higher overall effort levels and lower stock levels (see, e.g., Bjørndal *et al.* 2004).

The numerical comparative statics highlight the interplay of two externalities, the social planner's efforts to curtail them and the differences in the extent to which they are generated by private fishers' optimal solutions depending on the structure of the fishery. Fisheries operating under open-access and sole ownership obtain different total catches and sustain different levels of fish stocks, thereby differentiating their contributions to water quality.

4. Discussion

In recent decades, biomanipulation, often in the form of selective fishing, has been undertaken in many temperate lakes, but the results have indicated variable success. In general, successful restoration of lakes with biomanipulation seems to require simultaneous abatement of external loading. Biomanipulation has two major effects on a water ecosystem: it removes nutrients with the fish tissue and it changes the stock(s) of the targeted fish. The stock changes affect water quality via the food web. The ecological impacts of nutrient abatement (known as bottom-up control) and consumer species (top-down control) have been widely analyzed and disputed. It has been suggested that both forms of control are effective in both seas and freshwater lakes.

In this paper, we have developed a stylized dynamic optimization model for water quality management that combines external abatement and fisheries to maximize social welfare. We consider three alternative cases, with decision makers considering only their own profits, profits from the fishery as a whole, or both water quality and the fishery as a whole. There are thus two market failures incorporated in our model: (potential) over-fishing resulting from open access and eutrophication.

We applied our model to a relatively closed coastal bay, Mynälahti, South-West Finland. We showed that optimal policies consist of both abatement of agricultural loads and fisheries targeting cyprinids. Two drivers proved extremely important. First, the effect that the fish stock has on water quality influences the socially optimal fisheries substantially. The stronger this effect is, the less emphasis the optimal eutrophication management puts on phosphorus removal in fish tissue and the more emphasis it puts on keeping the stock permanently at a low level. Second, the price of the fish affects the optimal allocation of efforts between fisheries and external phosphorus abatement. Furthermore, the price changes affect the social welfare differently depending on the structure of the fisheries. With suitable prices, open access may generate escapement levels that are below the biologically MSY — and these might be the socially desired escapement levels for fish species with negative water quality effects.

Our analysis shows that fisheries should be a part of efficient water quality management under a wide variety of conditions. In addition to traditional environmental policies, our results might have implications on regulatory systems such as nutrient trading schemes. It would be interesting to analyze if fisheries of some non-commercial species could be made part of some existing scheme, such as the Chesapeake Bay nutrient trading program. This would require establishing the ecosystem effects of the species in question.[11] In the Baltic Sea, an EU-funded project analyzes possibilities to develop and utilize some form of nutrient trading.[12] The development project includes a pilot that allocates funds for harvesting cyprinid species. Could cyprinid fish landings be used as nutrient credit offsets in future trading platforms?

[11] The Chesapeake Bay Foundation has already allocated funds to restore oyster population. Oysters have direct ecosystem benefits, and nutrients are removed from the system as they are being harvested. See http://www.cbf.org/how-we-save-the-bay/through-restoration for restoration efforts and https://www.epa.gov/chesapeake-bay-tmdl/trading-and-offsets-chesapeake-bay-watershed for the trading program.

[12] http://nutritradebaltic.eu/

Our research also opens up interesting avenues for further research. The intensity of optimal fishing efforts depends crucially on the effect fish stocks have on water quality via the food web. Changes in the fish stock change the amount and structure of the grazing pressure on the zooplankton and benthic fauna, the bioturbation that the fish cause to sediments, and the excretion rate per biomass unit. All these factors affect nutrient release from sediments, the bioavailability of nutrients to phytoplankton, and the biomass of zooplankton, all of which may restrict the biomass and affect the composition of phytoplankton. These processes may take place simultaneously and have mutually synergistic or antagonistic effects, which makes the net effect difficult to evaluate. It would be an important extension of the model to explicitly account for these effects.

The model could be modified to cover uncertainty. In our specification the cyprinid fish stock may have a direct effect on the level of algae in the water. One might ask: How do the probability distributions change with cyprinid fish stocks? Which one is more resilient: a waterbody with higher nutrient concentrations, a lower cyprinid fish stock and higher amount of zooplankton; or one with a lower nutrient concentration, a higher cyprinid fish stock and thus a lower amount of zooplankton?

We used a single dimensional and linear damage function which was based on estimates of the values of hypothetical improvements in water quality. Future studies can show whether, as we have assumed here, the marginal benefit of water quality improvement can represent the marginal damage from quality deterioration. In addition, the results from future valuation studies may help to allow for a more responsive, non-linear damage function.

The model could also be extended to multiple age or size-classes to account for the differences in the effects fish individuals of different sizes have on water quality. As seen from the theoretical model the fish stock dynamics play a central role in our model. Harvesting cyprinid fish removes the largest fish in the population. With two age groups, young and old, focusing on the old could temporarily increase the population of young (due to decreased competition for resources). It has been suggested that small cyprinid fish have a more harmful effect on ecosystems than large ones. They are more effective in eating zooplankton and may also contribute more strongly to the release of phosphates from the sediments. Hence, we would need two parameters with $\omega_{\text{small}} > \omega_{\text{large}}$. However, the costs of harvesting smaller fish are higher, that is, $\gamma_{\text{small}} > \gamma_{\text{large}}$. How would this change the optimal policy? Obviously, this depends on the costs of fisheries.

Finally, an analysis of economic instruments would be useful. How should existing fisheries be incentivized to harvest a given species that has no, or only little, market value? Optimal eutrophication management requires taking into account the removal of phosphorus as well as the effect the fish stock has on water quality. If the latter effect is negative and substantial, how can the society incentivize fishers to keep the stocks at low levels permanently even though the same harvesting levels would be less costly to obtain with higher stock levels? The fact that fish landings are observable but fish stocks are not makes this an interesting example of regulation under asymmetric information.

Acknowledgment

The authors thank Lassi Ahivik and participants at the 21st Conference of European Association of Environmental and Resource Economists for insightful comments. The research was funded by the Academy of Finland, grant no. 263337.

Appendix A

A.1. *Social optimum*

For the social planner, the shadow prices of the phosphorus stock and the fish stock are denoted by $\lambda_Q(X,Q)$ and $\lambda_X(X,Q)$. Also, let μ_S and μ_z be Kuhn–Tucker multipliers associated with the constraints on the control variables S and z, respectively. The (S, z) and (μ_S, μ_z) have to satisfy the complementary slackness conditions

$$S_t \le X_t; \quad \mu_{S,t} \le 0; \quad S_t < X_t \Rightarrow \mu_{S,t} = 0,$$

$$z_t \ge 0; \quad \mu_{z,t} \ge 0; \quad z_t > 0 \Rightarrow \mu_{z,t} = 0.$$

The complementary slackness conditions stipulate that at the steady state optimum, the escapement cannot exceed the stock size ($S_t \le X_t$), the current and future reward from an increase in fish stock must be less than equal to zero ($\mu_{S,t} \le 0$) and that if the fishing effort is positive ($S_t < X_t$), the optimal escapement is found where their marginal effect on current and future rewards is zero. Likewise, the external abatement must be non-negative; and if it is positive, its optimal level is found where its marginal effect on welfare (current and discounted future) is zero. The optimality conditions are:

$$-p + \frac{\gamma}{S} + \beta\lambda_Q\xi + \beta\lambda_X F'(S) = \mu_S,$$

$$-\alpha z - \beta\lambda_Q = \mu_z,$$

$$-D_Q + \beta(1-\rho)\lambda_Q = \lambda_Q,$$

$$p - \frac{\gamma}{X} - D_X - \beta\lambda_Q\xi + \max(\mu_S, 0) = \lambda_X,$$

where D_Q and D_X are the partial derivatives of the damage function.

Assuming a positive optimal abatement and escapement levels (and thus $\mu_z = \mu_S = 0$) and rearranging we get

$$\lambda_Q = \frac{-D_Q}{1-\beta(1-\rho)},$$

$$\lambda_X = p - \frac{\gamma}{X} - D_X + \beta\xi \frac{D_Q}{1-\beta(1-\rho)},$$

$$z = \frac{\beta}{\alpha}\frac{D_Q}{1-\beta(1-\rho)},$$

$$p - \frac{\gamma}{S} = \beta F'(S)\left[\left(p - \frac{\gamma}{X}\right) - D_X\right] - \frac{\beta\xi(1-\beta F'(S))}{1-\beta(1-\rho)} D_Q.$$

A.2. *Fish population growth*

Our approximations of fish population growth as well as of the costs of fishing (γ) are based on the data from a pilot project involving targeted fishery which ran from 2010 to 2011 in the Finnish Archipelago Sea and Gulf of Finland (Setälä *et al.* 2012). In particular, we use data from Mynälahti. We consider one representative cyprinid fish, combining data on bream and roach catches. The logistic growth function 3 is based on fish biomass without age classes. According to Setälä *et al.* (2012), the estimated biomass of cyprinid fish was approximately 1,284 tons in Mynälahti in the spring 2011 and we use this figure as an estimate for the stock. We assume that the cyprinid fish stock was in the steady state before the project. The average annual harvest between 1980 and 2009 was 63 tons (see Table A.1).[13] During the pilot project, the harvest of cyprinid fish in Mynälahti was 126 tons in 2010 and 129 tons in 2011.

There were 103 fishers participating in the pilot project. The escapement levels were approximately 1,270 tons, 1,207 tons and 1,155 tons in 2009, 2010 and 2011,

Table A.1. Stocks and Harvests of Cyprinid Fish in Mynälahti (tons). Growth Function Parameters: $K = 1{,}500$, $r = 0.325$

Year (t)	Stock (X_t)	Harvest $(X_t - S_t)$	Escapement (S_t)
1980–2009	1,333	63	1,270
2010	1,333	126	1,207
2011	1,284	129	1,155

[13]Database of the Finnish Game and Fisheries Research Institute.

respectively. There are infinitely many pairs of parameters K and r in Eq. (3) that would generate such escapement levels. To determine reasonable values, we have to fix either K or r after which the other value can be determined from the data. Based on the expert assessment, we set $K = 1{,}500$ which fixes the growth rate at $r = 0.325$.

A.3. *Costs of fisheries*

The costs of targeted fisheries (γ) comprise variable costs such as the costs of labor, fuel, operation and maintenance. We apply the condition of open-access fishery where fishers harvest until marginal revenue equals marginal costs: $p = \frac{\gamma}{S}$.

The price of cyprinid fish has been around 300 EUR/ton (i.e., 0.3 EUR/kg), but during the pilot project (2010–2011) the fishers were paid 400 EUR/ton. Before 2010 the catch of cyprinid fish was mainly bycatch of other fisheries. Because the pilot started midway through the 2010 harvesting season, our numerical simulations use approximations for 2011. Thus, $p = 400\,\text{EUR/ton}$ and $\gamma = pS|2011 \approx 460{,}000$.

A.4. *Phosphorus concentrations, food web and water quality*

We have long-term data on phosphorus loading, phosphorus concentrations in the bay and annual fisheries landings.[14] We calibrate the unknown parameter ρ and the baseline loading $\bar{q}$ in Eq. (2) using these data. The data leave room for interpretation: for instance, the magnitude of phosphorus transfer between the bay and the open sea as well as from and to bottom sediments is uncertain. The values of these two parameters influence how quickly and strongly the water quality responds to changes in external loading and fisheries.

The average riverine phosphorus loading to Mynälahti was about 15.1 tons between 1975 and 2012.[15] Phosphorus has been removed from the bay through fish landings. The average amounts (and phosphorus concentrations of the associated fish species according to Mäkinen (2008)) during the last three decades have been the following: perch, 179 tons (phosphorus concentration 1.1%); pike, 32 tons (0.66%); pike-perch, 53 tons (0.8%); burbot, 30 tons (0.43%); roach, 35 tons (0.8%) and bream, 28 tons (0.7%). There is also a significant herring catch in the bay (5,529 tons) but as it is a migratory fish, we make the assumption that only 15% of the phosphorus in the fish originates from the bay. The estimated total of phosphorus removal from the bay annually by fisheries is thus some 6.5 tons.

[14]Nutrient loads and concentrations in rivers and in the bay are based on data of the Finnish Environmental Center.

[15]Observations from eight years are missing during this period.

Using the phosphorus concentrations of the cyprinid fish in (Mäkinen 2008) yields the parameter value $\xi = 0.0075$ in Eq. (2).

The baseline phosphorus loading in tons is the net input, that is, the difference of the riverine loadings and phosphorus removed by the fisheries (other than the quantity in cyprinid fish which enters the equation separately). The parameter for baseline loading in Eq. (2) becomes $\bar{q} = 8.6$. The parameter ρ in (2) captures all other phenomena affecting the phosphorus balance, the most significant of which are inflow and outflow from the bay, sedimentation and benthic phosphorus release. Rather than capturing a single physical phenomenon within the water ecosystem, the parameter has a statistical interpretation. It is calibrated at $\rho = 0.89$, a level that holds the phosphorus concentration at its long-term average level with the baseline loading and fisheries. If, for instance, the total catch decreased by 10%, 20% or 30%, the long-run phosphorus concentration would end up being 39 μg/L, 42 μg/L and 45 μg/L, respectively. That is, the existing fisheries do play a role in controlling the water quality.

A.5. *Costs of abatement*

Mynälahti receives mainly non-point loading from the rivers Puttaanjoki, Laajoki and Mynäjoki. The field acreage of the basin is about 17.7 thousand hectares (about 20% of the total area of the basin). We assume that 70% of the phosphorus loading originates from agriculture (ELY Southwest Finland 2011). Based on this and the loading data, the total phosphorus loading *entering the bay*, that originates from agriculture is 10.6 tons. The average effective phosphorus loading is thus about 0.7 kg/ha, which is slightly lower than the usual estimate for agriculture. For the cost-efficiency analysis we must use the loading values at the receptor point, thus acknowledging the natural retention caused by the forests, wetlands and lakes in the basin. We assume that the loading intensity per hectare varies according to the soil test phosphorus distribution as quantified in (Iho *et al.* 2014). Hence, the 10% of the fields that pollute the least have an annual effective phosphorus loading of 0.23 kg/ha, and the 10% that pollute the most, 1.2 kg/ha.

It is not straightforward to determine the costs of measures to mitigate phosphorus loading from agriculture. Erosion control measures such as no-till reduce the loading of particulate phosphorus but simultaneously increase the loading of dissolved phosphorus (McDowell and McGregor 1984; Puustinen *et al.* 2005). Accumulated soil phosphorus is a reliable proxy for dissolved phosphorus loads, but estimating the costs of soil phosphorus depletion is a complex undertaking (see e.g., Iho and Laukkanen 2012b). One of the few measures with readily identifiable costs and straightforward results is spreading gypsum and we have opted to use it as our representative abatement method. We assume that half of the phosphorus load is in dissolved and half in particulate form. On this basis we calculate that the load reduction that can be achieved with gypsum treatment is 43% (Ekholm *et al.* 2012). The annualized per hectare cost is EUR 73 (Iho and Laukkanen 2012a). This yields the profit function for agriculture, that is, the abatement cost function:

$$\pi_a = -\frac{1}{2}\alpha z^2 = -56{,}800z^2,$$

where $z \geq 0$ is abatement in tons and hence $\alpha = 113{,}600$. The marginal abatement cost after the first ton of phosphorus is about 114 EUR/kg, followed by 227 EUR/kg, 340 EUR/kg, 454 EUR/kg, 568 EUR/kg, etc. for subsequent tons reduced.

A.6. *Value of environmental damage*

The parameter for the value of environmental damages captures the fact that clean waters are appreciated by society. As water quality is not traded in the markets, the monetary value of environmental damage (or benefit) from changes in water quality has to be estimated using non-market valuation methods. We use the phosphorus stock of the water column converted to water clarity as an indicator of water quality. The damage function (6), which maps water quality damages in monetary terms builds on Finnish value estimates from two choice experiment studies (Kosenius 2010; Ahtiainen *et al.* 2014). Table A.2 describes the interpretation of water quality based on these studies in relation to the general usability classification presented in (SYKE 2005).

Table A.2. Water Quality Levels in the Two Valuation Studies and the Usability Index for Sea Waters. WTP Values in Year 2013 Euros

Water Usability Classification	Total $P\left(\frac{\mu g}{L}\right)$	Water Clarity (m)	Perceived Water Clarity at One Meter Depth (Kosenius 2010)	Water Clarity (Ahtiainen *et al.* 2014)
Excellent	< 12	> 2.5		The bottom is visible from over 2 m in depth (WTP EUR 137.6)
Good	$<= 20$	$<= 2.5$	Clearly visible (WTP EUR 49.0)	The bottom is visible from one to 2 m in depth (WTP EUR 83.8)
Satisfactory	$<= 40$	< 1	Hardly visible (WTP EUR 24.5)	The bottom is visible from less than 1 m in depth (base case)
Passable	$<= 80$		Not at all visible (base case)	
Poor	> 80			

The two valuation studies estimate marginal values for changes in water quality rather than the total value of the waterbody at a certain water quality level. As the damage function needs to be anchored to an absolute scale, we assume that damage in monetary terms at excellent water clarity is zero on the five-step quality scale. The value estimates presented in (Kosenius 2010) and (Ahtiainen *et al.* 2014) are then inverted to represent damages, that is, good water clarity indicates less damage than satisfactory water clarity. When value estimates overlap we have assumed an average marginal change in willingness to pay (WTP) across the two studies. Based on these values, we then make a linear approximation of the damage function from known points to worse quality levels, that is, passable and poor quality, which represent very low usability of the waterbody for any purpose. Thus, we assume that the marginal benefit of water quality improvement can be inverted to represent the marginal damage from deterioration in quality and that the values for damage follow a linear slope. Studies have found that water quality may exhibit diminishing marginal value for improvements (e.g., Artell 2014). Our linearity assumption on the marginal values thus leads to conservative value estimates for improvements in poor quality areas.

We establish a simple correspondence between the usability classification, total phosphorus and the stock of cyprinid fish. Water quality is expressed as a scalar value, with 0 representing excellent quality and 4 (or above) poor. This scale corresponds to the general usability classification used in Finland until 2008, before the implementation of the Water Framework Directive.[16]

There is a clear logarithmic correspondence between total phosphorus concentration (Q) in brackish water and the usability classification.[17] The estimated parameters for function 6 are $\kappa = -2.5$ and $\eta = 1.3$.[18] For the base case, we assume that $\omega = 1$ and that the direct influence attainable with fish stock control is equal to one. That is, with maximum stock control, the water quality can be improved from poor to passable, from passable to satisfactory, and so on.

The water quality in Mynälahti is classified as passable, the second poorest in the five-step classification. In addition to classification, the total value of damage depends on the number of people affected by shortcomings in water quality. Here, we make the assumption that preferences used in calculating the original, national value estimates also represent the target area population. The marginal damage per person becomes approximately EUR 46 (in year 2013 value). It is based on the

[16]The questionnaires on which the valuation estimates are based used the general usability classification.

[17]The value, phosphorus concentration (class mean values) pairs are (6,0); (16,1); (30,2); (60,3) and (130,4).

[18]$R^2 = 0.99$.

marginal values for water clarity changes estimated in (Ahtiainen *et al.* 2014) and (Kosenius 2010).

According to the Finnish national outdoor recreation demand and supply inventory data, 90% of all water related recreational trips were made within a range of 70 km and 49% within 10 km of home (Pouta and Sievänen 2001). Using GIS techniques with Statistics Finland's 2010 Grid Database, we estimate that there are more than half a million people permanently residing within the range of 70 km of the midpoint of the bay. However, there is a wide range of substitutes for water-based recreational activities in this region. Using the entire population would grossly overestimate the value of water quality improvements in Mynälahti alone. For our base case, we use the distance of 10 km, which given the large size of the bay itself, is a robust lower limit for the size of the affected population. The estimate yields 6,900 permanent inhabitants. We calculate the results for alternative distances in our sensitivity analysis. Hence, the marginal damage parameter in 6 becomes $\sigma = 317{,}400$.

References

Ahtiainen, H, J Artell and E Pouta (2014). Using individual-specific status quo alternative in choice experiments: Heterogeneous preferences for water quality. '*5th World Congress of Environmental and Resource Economists*'.

Artell, J (2014). Lots of value? A spatial hedonic approach to water quality valuation. *Journal of Environmental Planning and Management*, 57(1), 862–882.

Baumol, WJ and WE Oates (1988). *The Theory of Environmental Policy*. Cambridge University Press.

Bjørndal, T, DV Gordon, V Kaitala and M Lindroos (2004). International management strategies for a straddling fish stock: A bio-economic simulation model of the Norwegian spring-spawning herring fishery. *Environmental and Resource Economics*, 29(4), 435–457.

Botsford, LW, JC Castilla and CH Peterson (1997). The management of fisheries and marine ecosystems. *Science*, 277(5325), 509–515.

Breukelaar, AW, EH Lammens, JGK Breteler and I Tatrai (1994). Effects of benthivorous bream (Abramis brama) and carp (Cyprinus carpio) on sediment resuspension and concentrations of nutrients and chlorophyll a. *Freshwater Biology*, 32(1), 113–121.

Carpenter, SR (2005). Eutrophication of aquatic ecosystems: Bistability and soil phosphorus. *Proceedings of the National Academy of Sciences USA*, 102(29), 10002–10005.

Christensen, V (2014). A century of fish biomass decline in the ocean. *Marine Ecology Progress Series*, 512, 155–166.

Clark, CW (2010). *Mathematical Bioeconomics: The Mathematics of Conservation*, Vol. 91. John Wiley & Sons.

Costello, C, S Gaines and J Lynham (2008). Can catch shares prevent fisheries collapse? *Science*, 321, 1678–1681.

Diaz, RJ and R Rosenberg (2008). Spreading dead zones and consequences for marine ecosystems. *Science*, 321, 926–929.

Ekholm, P, P Valkama, E Jaakkola, M Kiirikki, K Lahti and L Pietola (2012). Gypsum amendment of soils reduces phosphorus losses in an agricultural catchment. *Agricultural and Food Science*, 21, 279–291.

ELY Southwest Finland (2011). State of the Archipelago Sea and sea areas around the Åland islands and nutrient loading to them. Publications 62011, Centre for Economic Development, Transport and the Environment for Southwest Finland, Turku.

Hansson, L-A, H Annadotter, E Bergman, SF Hamrin, E Jeppesen, T Kairesalo, E Luokkanen, P-Å Nilsson, M Søndergaard and J Strand (1998). Biomanipulation as an application of food-chain theory: Constraints, synthesis, and recommendations for temperate lakes. *Ecosystems*, 1(1), 558–574.

Hasler, AD and WG Einsele (1948). Fertilization for increasing productivity of natural inland waters. *Transactions of the Thirteenth North American Wildlife Conference*, 13, 527–555.

Horne, AJ and C Goldman (1994). *Limnology*. McGraw-Hill, Inc.

Horppila, J and T Kairesalo (1990). A fading recovery: The role of roach (Rutilus rutilus L.) in maintaining high phytoplankton productivity and biomass in Lake Vesijärvi, Southern Finland. *Hydrobiologia*, 200/201, 153–165.

Horppila, J, H Peltonen, T Malinen, E Luokkanen and T Kairesalo (1998). Top-down or bottom-up effects by fish: Issues of concern in biomanipulation of lakes. *Restoration Ecology*, 6, 20–28.

Iho, A, J Lankoski, M Ollikainen, M Puustinen and J Lehtimäki (2014). Agrienvironmental auctions for phosphorus load reduction: Experiences from a Finnish pilot. *The Australian Journal of Agricultural and Resource Economics*, 58, 1–18.

Iho, A and M Laukkanen (2012a). Gypsum amendment as a means to reduce agricultural phosphorus loading: An economic appraisal. *Agricultural and Food Science*, 21, 307–324.

Iho, A and M Laukkanen (2012b). Precision phosphorus management and agricultural phosphorus loading. *Ecological Economics*, 77, 91–102.

Kahn, JR (1998). *The Economic Approach to Environmental and Natural Resources*. USA: The Dryden Press.

Keeler, E, M Spence and R Zeckhauser (1972). The optimal control of pollution. *Journal of Economic Theory*, 4(1), 19–34.

Knowler, D, EB Barbier and I Strand (2001). An open-access model of fisheries and nutrient enrichment in the black sea. *Marine Resource Economics*, 16(3), 195–217.

Kosenius, A-K (2010). Heterogeneous preferences for water quality attributes: The case of eutrophication of the Gulf of Finland, Baltic Sea. *Ecological Economics*, 69(3), 528–538.

Laukkanen, M (2003). Cooperative and non-cooperative harvesting in a stochastic sequential fishery. *Journal of Environmental Economics and Management*, 45(2), 454–473.

Lichtenberg, E and D Zilberman (1986). The econometrics of damage control: Why specification matters. *American Journal of Agricultural Economics*, 68(2), 261–273.

Mäkinen, T (2008). Voidaanko kalastuksella vähentää kalankasvatuksen ravinnekuormaa? (in Finnish) Selvityksiä 2/2008, Riista-ja kalatalouden tutkimuslaitos.

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

McDowell, L and K McGregor (1984). Plant nutrient losses in runoff from conservation tillage corn. *Soil and Tillage of Research*, 4, 79–91.

Nielsen, M, O Flaaten and S Waldo (2012). Management of and economic returns from selected fisheries in the Nordic countries. *Marine Resource Economics*, 27(1), 65–88.

Nürnberg, GK, M Tarvainen, A-M Ventelä and J Sarvala (2012). Internal phosphorus load estimation during biomanipulation in a large polymictic and mesotrophic lake. *Inland Waters*, 2, 147–162.

Pauly, D, V Christensen, J Dalsgaard, R Froese and F Torres (1998). Fishing down marine food webs. *Science*, 279, 860–863.

Pouta, E and T Sievänen (2001). Virkistyskysyntä Suomessa. In *Metsän eri käyttömuotojen arvottaminen ja yhteensovittaminen*, Vol. 800, J Kangas and E Kokko (eds.), pp. 53–61. Metsäntutkimuslaitoksen tiedonantoja (in Finnish).

Puustinen, M, J Koskiaho and K Peltonen (2005). Influence of cultivation methods on suspended solids and phosphorus concentrations in surface runoff on clayey sloped fields in Boreal climate. *Agriculture, Ecosystems and Environment*, 105, 565–579.

Sass, GG, JF Kitchell, SR Carpenter, TR Hrabik, AE Marburg and MG Turner (2006). Fish community and food web responses to a whole-lake removal of coarse woody habitat. *Fisheries*, 31, 321–330.

Schindler, D (2012). The dilemma of controlling cultural eutrophication of lakes. *Proceedings of the Royal Society B*, 279, 4322–4333.

Setälä, J, J Airaksinen, J Lilja and J Raitaniemi (2012). Pilottihanke vajaasti hyädynnetyn kalan käytän edistämiseksi (in Finnish). *RKTL:n Työraportteja*, 10.

Skov, C and PA Nilsson (2007). Evaluating stocking of YOY pike Esox lucius as a tool in the restoration of shallow lakes. *Freshwater Biology*, 52(9), 1834–1845.

Smith, VH (2003). Eutrophication of freshwater and coastal marine ecosystems — a global problem. *Environmental Science and Pollution Research*, 10, 126–139.

Søndergaard, M, E Jeppesen and S Berg (1997). Pike (Esox lucius L.) stocking as a biomanipulation tool 2. effects on lower trophic levels in Lake Lyng, Denmark. *Hydrobiologia*, 342, 319–325.

SYKE (2005). *Water Quality of Lakes, Rivers and Sea Areas in Finland in 2000–2003*. Brochure, Finnish Environment Institute.