



Analysis

Optimal Phosphorus Abatement Redefined: Insights From Coupled Element Cycles

Antti Iho ^{a,*}, Lassi Ahlvik ^{a,b}, Petri Ekholm ^c, Jouni Lehtoranta ^c, Pirkko Kortelainen ^c^a Natural Resources Institute Finland (Luke), Latokartanonkaari 9, FI-00790 Helsinki, Finland^b Department of Economics, NHH - Norwegian School of Economics, Helleveien 30, 5045 Bergen, Norway^c Finnish Environment Institute SYKE, P.O. Box 140, FI-00251 Helsinki, Finland.

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ABSTRACT

To successfully combat eutrophication caused by agricultural P loads, we need to understand how various forms of P respond to mitigation measures and thus how they contribute to algal growth. Failure to balance mitigation measures targeting dissolved inorganic P (DIP) and P in eroded soil (PP) may lead to economically inefficient measures at best, and to aggravated eutrophication at worst. We model dynamically optimal eutrophication management in a P-limited and SO₄-containing water body by taking into account the O₂ available and the coupling between the C, Fe, S and P cycles. We show that optimal management would put more weight on mitigating DIP than PP, and that the emphasis on DIP should be particularly strong in eutrophic water bodies. To foster influential and cost-efficient policies, we urge defining water body-specific multipliers to commensurate the main P forms into eutrophying phosphorus, much as greenhouse gases are converted to their CO₂ equivalents.

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1. Introduction

Despite efforts to reduce nutrient loading, lakes and coastal waters impacted by agriculture continue to suffer from eutrophication (Diaz and Rosenberg, 2008; Conley et al., 2011; Daloğlu et al., 2012; Michalak et al., 2013). As pressing a need as it is, management of eutrophication must be based on cost-efficiency, which means achieving the highest possible water quality with the resources available. A necessary condition for achieving cost-efficiency is that the effects on the trophic status of the last (i.e. marginal) actions to abate all controllable sources be equal in cost; that is, water quality cannot be improved with the available resources by reallocating abatement efforts from one source to any other. Economically efficient management of eutrophication thus requires that we be able to evaluate how different substances originating from controllable human activities affect the process.

In fresh water ecosystems particularly, eutrophication management has targeted phosphorus (P), as it is considered to be the major driver (Correll, 1998; Schindler et al., 2008; Schindler, 2012; Lewis and Wurtsbaugh, 2008). However, reducing P fluxes to aquatic systems is complicated by the fact that P is present in several forms differing as to where they originate, how they are best mitigated and what biological impact they have. Of the two major forms, dissolved inorganic P (DIP) is fully available to algae, whereas P bound by eroded soil

(particulate P, PP) contributes only moderately to bioavailable P in the productive water layers (DePinto et al., 1981; Ekholm and Krogerus, 2003).

Yet, PP may be transformed into DIP long after particles have entered bottom sediments (Lehtoranta et al., 2015). Let us call the contribution of PP to DIP before settling into sediments its short-term bioavailability, and its potential release from sediments its long-term bioavailability. It is often argued that mitigation of PP is beneficial because even though its short-term bioavailability is low, PP stored in sediments may someday be released as DIP, contributing to its long-term bioavailability. While the uncertainty surrounding PP's long-term bioavailability might be an argument for weighting PP and DIP equally, it would be even more judicious to aim for as precise a weighting of DIP and PP as possible: they both contribute to the same problem, that is, eutrophication. What makes proper weighting of DIP and PP extremely important, is that *i*) agriculture is the main source of P to surface waters (Carpenter et al., 1998; Sharpley et al., 2009) *ii*) all main conservation measures in agriculture exhibit strong interactions between PP and DP abatement, with measures targeting PP tending to elevate DIP loading (Dodd and Sharpley, 2016).

Optimal abatement of P from agriculture needs to be defined in terms of eutrophying P, an approach which assigns weights to PP and DIP according to their short- and long-run contributions to algae growth. To this end, we postulate a dynamic bio-economic model that takes into account both the short- and long-term bioavailability of PP. While the phosphorus cycle clearly includes many other forms of P than PP and DIP, we constrain the precision of our model to that

* Corresponding author.

E-mail address: antti.iho@luke.fi (A. Iho).

attainable in agri-environmental management practices. Moreover, our understanding on the effects of various measures on PP and DIP is relatively good (Dodd and Sharpley, 2016), whereas we do not have comprehensive information on how conservation practices affect other phosphorus fractions. On balance, conservation measures can and should be designed and evaluated based on their effect on eutrophying P, calculated as a properly weighted sum of DIP and PP. This should be the next step in improving the precision and efficiency of P abatement.

The long-term bioavailability of sediment phosphorus has been implicitly included in studies such as Tahvonen and Salo (1996), Mäler et al. (2003) and Polasky et al. (2011), which analyze optimal eutrophication management under potential regime shifts. In the case of P, a regime shift involves a change in sedimentary redox conditions resulting in a state in which the ecosystem sustains high concentrations of P, as well as primary production, even if the P load is substantially reduced (Mäler et al., 2003; Carpenter, 2005). However, sediment processes as such are not sensitive to the availability of DIP and PP. Given that P itself is not a redox-sensitive substance, the cycling of P must be coupled to redox-sensitive elements, mainly C, Fe and S in sediments, for a regime shift to occur (Ekholm and Lehtoranta, 2012).

Targeting eutrophying P, rather than total P, in abatement measures would increase economic efficiency of the measures. This paper identifies and analyzes the principles for weighting DIP and PP in efficient P management. The model and application described in what follows highlight the importance of synchronizing the modeling of both DIP and PP at their sources and sinks. We focus on a P-limited and SO₄-rich water body, consider DIP and PP as fractions of external P load and explicitly model the reactive Fe in eroded soils. These choices allow us to maintain a crisp focus in our analysis. The results of coupling the dynamics of P in the system to the cycles of redox-sensitive C, Fe and

S suggest that optimal eutrophication management is drastically different from that pursued to date: although DIP and PP both promote eutrophication, society should put substantially more emphasis on DIP abatement.

2. Materials and Methods

2.1. The Model

Our schematic model includes three interlinked state variables: (i) DIP, (ii) potentially mobile P in sediments (M) and (iii) reactive Fe in sediments (F) (Fig. 1). The changes in the variables from one period to another are driven by their current state, associated inputs and the within-system exchange between sediment and water, this last depending on the availability of O₂.

The stock of DIP in the water body develops according to Eq. (1). It is increased by DIP load (p_t), the DIP desorbed from PP before sedimentation ($\eta_1 e_t$) and the benthic release of P that reaches the euphotic zone (q_t). A share of DIP (γ) is taken up biologically by algae and removed from the water by sedimentation. During each period, a fraction (δ_p) of the DIP is exported from the system with the outflow.

The stock M (Eq. (2)) increases with the settling of organic matter containing P (γp_t) and the settling of PP ($\eta_2 e_t$). Each period, a fraction (δ_M) of M is permanently retained. M also decreases with the release of benthic P (q_t). The stock of reactive Fe in sediments (F) increases with the supply of Fe oxides in eroded soil ($\eta_3 e_t$) and decreases with the formation of solid Fe sulfides. The amount of Fe sulfides is denoted by s_t and the proportion of Fe in it by parameter ξ_4 . F is buried permanently at the rate of (δ_F).

Benthic P release, jointly driven by all three state variables, is a focal mechanism in our model. The sedimentation of algae and debris

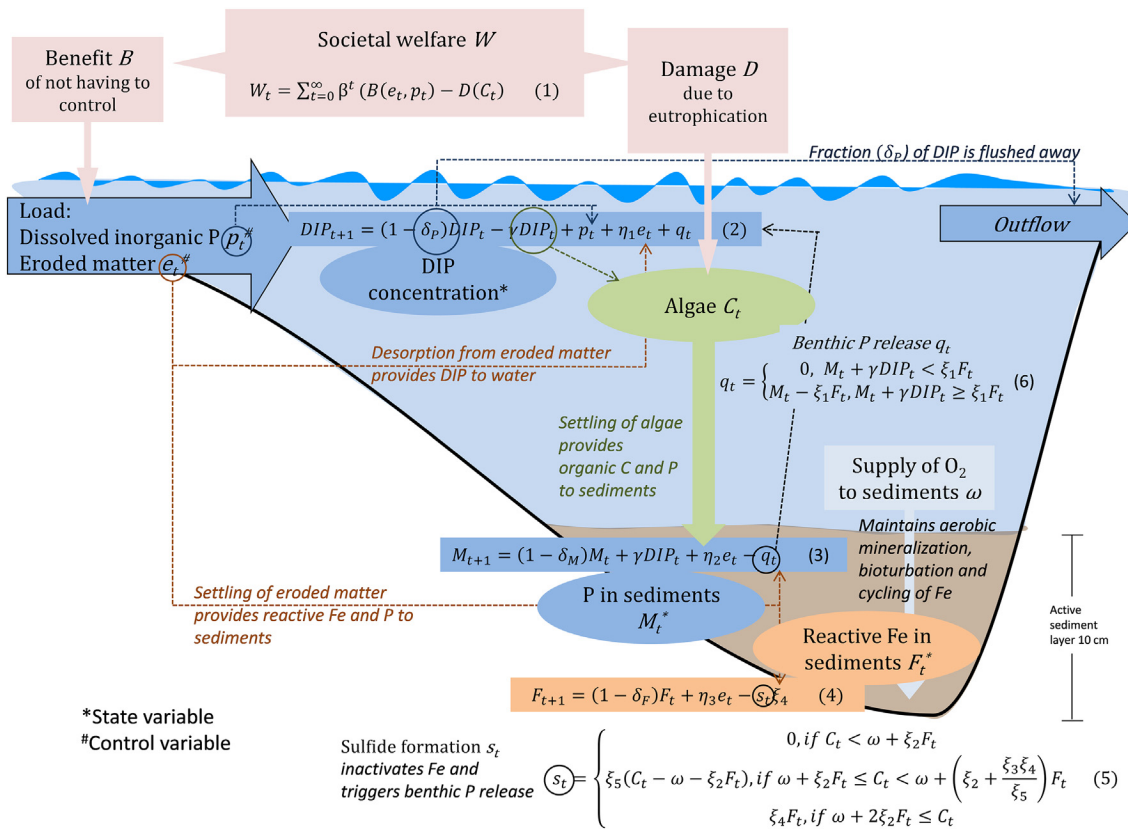


Fig. 1. Schematic model of the system. The arrows depict in- and outflows, ellipses the state variables and algae, and boxes at the top the economic variables: benefits from averted abatement costs and damage from eutrophication. The model works as follows: i) each period, DIP and erosion loading enter the system, increasing some or all of the stocks (2), (3) and (4) ii) DIP stock prompts algae growth iii) decaying algae consumes TEAs and provides organic P to sediments iv) P and Fe in sediments, TEA demand and supply of oxygen jointly determine the benthic release of P as given in Eqs. (3)–(6). Benthic release adds to DIP stock.

transports organic C to the bottom sediments, where it is oxidized by terminal electron acceptors (TEAs). The mineralization starts with O₂ and proceeds through alternative TEAs: nitrate (NO₃), manganese (Mn) and iron (Fe) oxides, and SO₄. We focus on the reduction of Fe and SO₄ because of their importance in the cycling of P. In the model, microbial Fe reduction and oxic conditions maintain the coupled cycling of Fe and P and thus sustain a low release of P. Specifically, microbial Fe(III) oxide reduction forms dissolved Fe(II), which is capable of forming a sufficient amount of P-binding Fe(III) oxides when in an oxic environment.

By contrast, sulfate reduction prevents the redox cycling of Fe, in a process that first generates sulfides (H₂S, HS⁻) that then build solid iron sulfides (FeS or FeS₂). This allows dissolved P released from Fe(III) oxides to be freely transported to the productive water layers. Whether Fe or SO₄ reduction dominates depends on the amount of labile organic C available, as well as on the availability of Fe oxides, SO₄ (Roden and Edmonds, 1997) and O₂ in the surface sediment.

The supply of O₂, which maintains the Fe-bound P in sediments, is driven by the physical transport of oxic water and the diffusion rate of O₂ at the sediment-water interface. The transport and diffusion rates are affected by temperature, wind, potential thermo- and haloclines, the presence of any ice cover, and morphological factors such as the shape of bottom areas and depth of water. Mineralization through SO₄ generates sulfides in proportion to the availability of O₂ and reactive Fe, as depicted in Eq. (5) in Fig. 1.

Eq. (5) calculates the amount of sulfides generated for each combination of C_t, F_t and ω. This function is illustrated graphically in Fig. 2. Oxic conditions prevail in our model when the supply of O₂ (ω) exceeds the amount of O₂ needed to decompose the labile organic C present in the sediment surface. In anoxic state, only O₂ acts as a TEA and no sulfides are generated in the surface layer. If the amount of organic C increases but remains below ω + ξ₂F_t, O₂ and Fe will be used as TEAs, but no sulfides will form yet. This state is depicted by region (i) in Fig. 2: there is either enough O₂ or reactive Fe to prevent the sulfate reduction and the consequent formation of iron sulfides. Parameter ξ₂ depicts the proportion of Fe needed to decompose a mass unit of C. If the level of organic C increases above ω + ξ₂F_t, SO₄ will start acting as a TEA and sulfides will form on the sediment surface; this is region (ii) in Fig. 2. The quantity of sulfides is given by a linear function (ξ₅/ξ₃)(C_t - ω - ξ₂F_t), where ξ₃ denotes the amount of SO₄ needed to reduce organic C, ξ₅ signifies the amount of FeS formed per unit of SO₄ and the equation in parentheses represents the amount of organic matter left to be oxidized after O₂ and Fe have been consumed as TEAs. If the amount of organic matter exceeds ω + (ξ₂ + $\frac{\xi_3 \xi_4}{\xi_5}$)F_t, the sulfides reach their maximum level, ξ₄F_t, where ξ₄ denotes the amount of Fe in one unit of FeS. This leads to the state depicted by region (iii) in Fig. 2, where most of the reactive Fe is in the form of Fe sulfides and there are no Fe(III) oxides available to capture P. The ensuing increase in eutrophy promotes settling of

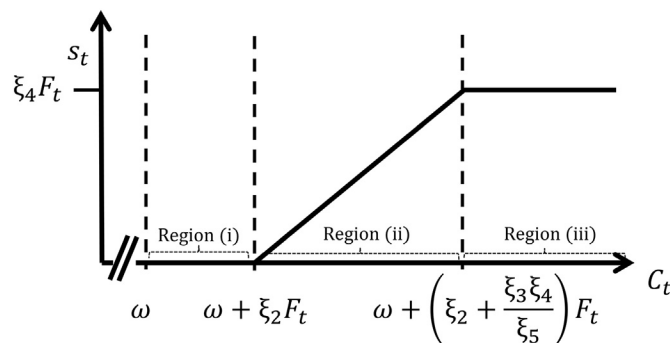


Fig. 2. Formation of sulfides (s_t) as a function of decaying organic matter (C_t) for a given oxygen supply (ω) and reactive iron stock (F_t).

organic matter and results in a reduction of SO₄ and extensive release of DIP without concomitant release of Fe into the water.

The second step in the model is to express the amount of benthic DIP released as a function of the formation of sulfides and stocks F_t and M_t (Eq. (6) in Fig. 1). Parameter ξ₁ is the ability of reactive Fe to capture P. Benthic P release is zero if there is enough reactive Fe to bind the P. If there is not enough Fe, a proportion of the sediment P is released into upper, productive water layers, where it adds to the DIP stock.

The economic problem is to choose values for the control variables – erosion (e) and DIP loading (p) – such that they maximize the stream of discounted social welfare. The welfare is the sum of averted DIP and erosion abatement minus the damage from eutrophication. The profits and damage from subsequent periods are weighted using a time-invariant discount factor (β). The model is solved using dynamic programming. All equations are presented in more detail in the Supplementary material.

3. Results

We parameterized the model based on data for Lake Pyhäjärvi in southwestern Finland. The catchment area of the lake is 477 km² (20% fields, 52% forests on mineral soils, 14% peatlands, and 2.5% built-up land). According to the criteria of the Water Framework Directive, Pyhäjärvi (155 km²) is a large clear-water lake and has good ecological status. Its mean depth is 5.5 m (max. 26 m), total water volume 0.85 km³ and the mean SO₄ concentration about 10 mg/l.

We calibrated the model with the use of stoichiometric constants and the model parameters obtained from the literature (η_1 , η_2 , and η_3). We use data on erosion and DIP inflow and outflow to the lake, as well as annual estimates of nutrients removed by fisheries. The outflow of DIP (δ_p) is calibrated based on the data on DIP concentration and outflow. For the calibration of δ_b and δ_f we assume that the depth of the active sediment layer is 10 cm.

Despite the circumscribed application area, results such as the optimal weight to be given to DIP and PP abatement are generally applicable, as is illustrated in the Supplementary material (“Sensitivity Analysis”). The parameters and other numerical values used in the model are given in Table 1.

3.1. Model Dynamics

We first illustrate the basic dynamic properties of the model by choosing alternative values for permanent DIP loading, keeping erosion fixed and not considering the economic optimality of the system. The three panels in Fig. 3 present the state variables at all possible steady states. Any point on the lines represents a feasible equilibrium, where the control and the state variables remain permanently unchanged. The figure thus does not reflect the duration or particular paths of, for instance, a possible regime shift from a mesotrophic to eutrophic state.

The horizontal axes of the panels denote DIP loading and the vertical axes the long-term levels of the state variables associated with these. For tractability, we have fixed the erosion loading at its current level (29 g per m² per year; see the Supplementary material (“P dynamics in Lake Pyhäjärvi”). Starting from the left, the vertical axes of the panels in Fig. 3 depict the DIP stock, sediment P and sediment reactive Fe. If the DIP loading is between the two vertical dotted lines at thresholds T_1 and T_2 , there are two possible steady states, a mesotrophic (solid line) and a eutrophic (dashed line). In the mesotrophic state, there is a large stock of reactive Fe in the sediments, no SO₄ reduction and no benthic P release (in reality, SO₄ reduction will always be found at certain sediment depths). Sufficient availability of Fe enables sediments to retain more P as permanent DIP loading increases (solid, steeply rising line in middle panel) while the stock of DIP remains low (solid line in left panel). The system will shift to the eutrophic state if DIP loading increases above threshold T_2 . In that state, SO₄ reduction has started, most of the stock of reactive Fe is bound in FeS and the ability of sediment to retain P

Table 1

Numerical values used in the application. The calibration of the lake- and basin-specific parameters in this table is described in detail in the Supplementary material.

Stoichiometric constants	Symbol	Mass ratio	Molar ratio
P captured by Fe	ξ_1	0.28	2 Fe per 1 P
Fe as TEA for mineralizing C	ξ_2	0.054	4 FEIII per 1 C
SO ₄ as TEA for mineralizing C	ξ_3	0.25	1 SO ₄ per 2 C
Fe in FeS	ξ_4	1.6	1 Fe per 1 FeS
SO ₄ in FeS	ξ_5	0.92	1 SO ₄ per 1 FeS
C/P in phytoplankton	R	41	106 C per 1 P
Model parameters	Value	Source	
DIP desorbed from eroded soils	η_1	0.16 mg/g	Lehtoranta et al. (2015)
PP in eroded soils	η_2	0.41 mg/g	Lehtoranta et al. (2015)
Fe in eroded soils	η_3	9.7 mg/g	Ekholm and Lehtoranta (2012)
Outflow of DIP	δ_P	0.20	Lake-specific (Supp. material) ^a
Permanent burial of P	δ_B	0.10	Lake-specific (Supp. material)
Permanent burial of Fe	δ_F	0.10	Lake-specific (Supp. material)
Supply of oxygen to sediment	ω	180	Lake-specific (Supp. material)
Share of DIP used by algae	γ	0.63	Lake-specific (Supp. material)
Discount rate	r	3.50%	Boardman et al. (2006)
DIP abatement cost parameter	Linear	142	Lake-specific (Supp. material) ^b
	Quadratic	387	Lake-specific (Supp. material)
PP abatement cost parameters	Linear	-11	Lake-specific (Supp. material)
	Quadratic	77	Lake-specific (Supp. material)
Damage function parameters	Linear	36	Lake-specific (Supp. material) ^c
	Quadratic	15	Lake-specific (Supp. material)

^a For parameters δ_P , δ_B , δ_F , ω and γ see Supplementary material, P dynamics in Lake Pyhäjärvi.

^b For cost parameters, see Supplementary material, Abatement Costs.

^c For damage parameters, see Supplementary material, Damage from Eutrophication.

has diminished substantially. The system will revert to the mesotrophic state if the DIP load permanently falls below threshold T_1 . That is, for loads between T_1 and T_2 both steady states are possible, depending on the initial stock.

3.2. Optimal Management

Economic optimization combines the system dynamics with costs of erosion and DIP control, time preferences and valuation for environmental quality. Dynamically optimal eutrophication management comprises one or several optimal steady states, as well as optimal paths leading to them. Our numerical application yields one optimal steady state: mesotrophic. One of the main drivers of the optimal solution is society's willingness to pay for water quality. Our basic parameterization (Table 1) assumes a fixed valuation of water quality. However, it

is plausible to assume that society's willingness to pay for water quality changes over time and for different areas. Simply having fewer people enjoying the lake's water would, other things being equal, decrease the total valuation of the water quality; with a lower willingness to pay, the model might well indicate that a eutrophic state is socially optimal.

To highlight the differences between the two states, we simultaneously present our baseline scenario (mesotrophic state) and the optimal solution obtained with damage from impaired water quality set at only 25% of the baseline values (eutrophic state). Fig. 4 presents the values of the state and control variables, as well as their marginal abatement costs, at the two optima.

The DIP stock of the lake is 1.7 tons in the optimal mesotrophic, and 8.7 tons in the optimal eutrophic steady state. The annual PP loading at both states is some 5.8 tons, being slightly higher at the eutrophic. This is reflected in the marginal abatement costs, which are slightly lower at the eutrophic state. For DIP loading, the differences are somewhat larger. Optimal loading at the mesotrophic state is 4.3 tons and 4.0 tons at the eutrophic. The associated marginal abatement costs are 76.0 and 274.6 € per kg.

The DIP stock in the lake is significantly lower in the mesotrophic state. The sum of sediment P and DIP stock, however, is higher than at a eutrophic state. Sediments of mesotrophic systems are able to bind substantially more P than those of eutrophic systems.

What makes it optimal to have higher DIP abatement at the eutrophic state? As was shown in Fig. 3, the marginal effect of a unit increase in DIP loading on the DIP stock and hence on algal biomass growth is substantially stronger at the eutrophic state than at the mesotrophic state. Furthermore, the damage function is strongly convex; that is, marginal damage increases rapidly with eutrophication. The implication here is that society ends up incurring higher marginal abatement costs at the eutrophic steady state.

Why is optimal PP abatement lower in the eutrophic state? In the mesotrophic state, there is an abundance of reactive Fe in sediments and additional Fe in eroded matter is not particularly important in retaining P. It stays in sediments, with a given share of it being permanently retained every year. In the eutrophic state, on the other hand, almost all P is released from the sediments and any additional Fe plays an important role in inhibiting benthic P release.

Thus, while PP in eroded matter contributes more to the DIP stock in the eutrophic state, iron in eroded matter has a more important role in inhibiting benthic P release. The joint effect is that erosion control, that is, PP abatement, need not be as intensive at the eutrophic state as it is at the mesotrophic state.

The results of the model suggest that the management of P-induced eutrophication should place more weight on DIP loading than on PP loading regardless of the state of the water body. For waters in a eutrophic state, however, the relative emphasis on DIP abatement should

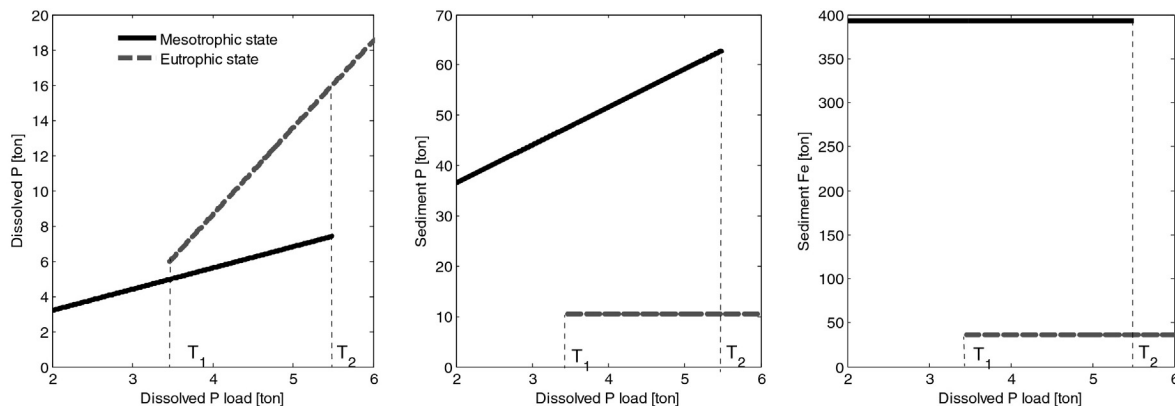


Fig. 3. Steady states of the three state variables for various levels of DIP loading. Solid lines represent the mesotrophic and dashed lines the eutrophic steady states.

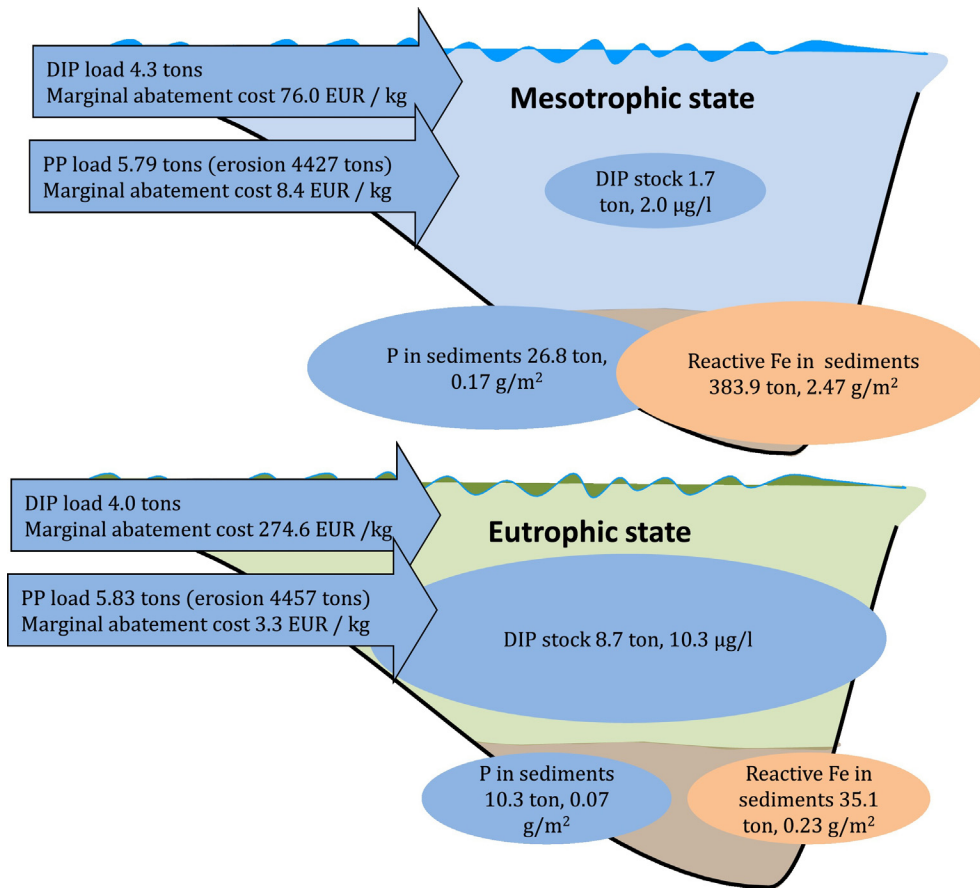


Fig. 4. Optimal DIP and erosion loadings and associated state variables for eutrophic and mesotrophic states.

be even higher. Society is willing to accept marginal costs for DIP abatement that are nearly one hundred times higher than those for PP abatement. The sensitivity analysis in the Supplementary material (Fig. S2) shows how the relative weights of PP and DIP change i) as the Fe content of eroded soils varies and ii) as the oxygen supply to

the system varies for an area where two optimal steady states exist simultaneously.

Fig. 5 illustrates the transition paths for optimal DIP and PP loading and for the associated state variables. To make the transition optimal, we start from the optimal solution associated with a willingness to

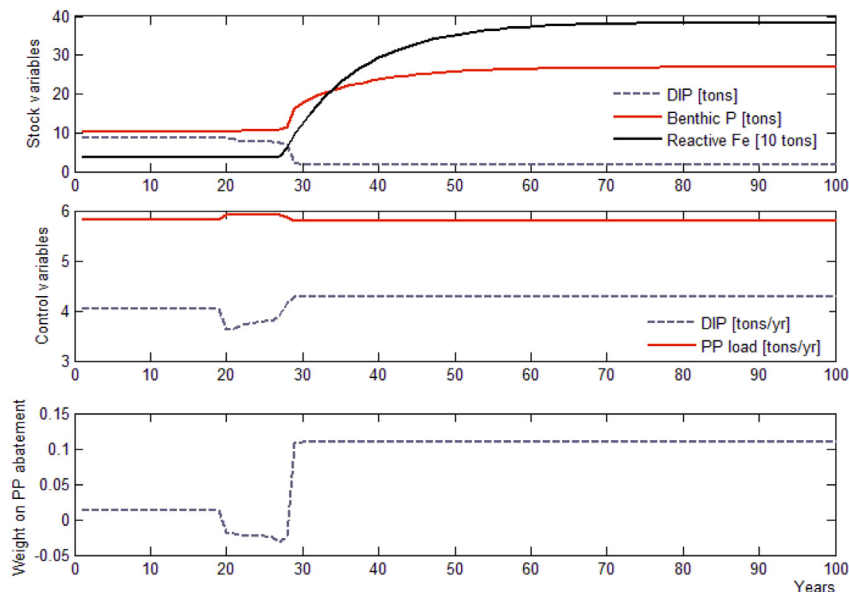


Fig. 5. State and control variables under optimal return from the eutrophic to mesotrophic state due to increase in societal valuation of water quality. (For interpretation of the references to color in this figure, the reader is referred to the web version of this article.)

pay that is 25% of the baseline level, and at time $t = 20$ (years) we allow the benefit to increase abruptly to the baseline level.

An increase in the social benefits associated with higher water quality prompts immediate changes in the optimal abatement of DIP and PP (middle panel in Fig. 5). These eventually bring the system back to the mesotrophic state, but the duration of the transition differs for each state variable (top panel in Fig. 5). The optimal relative weights that we assign to DIP and PP abatement differ for the two states and along the path from one state to another (lower two panels in Fig. 5).

The P-binding capacity of the sediments is restored through the slow accumulation of reactive sediment Fe (black line in the top panel), with a consequent increase in the amount of P in the sediments (red line in the top panel). The DIP stock will decline to a permanently lower level in about ten years, resulting in a lower phytoplankton biomass and hence improved water quality.

During the transition period, DIP abatement is intensified by 14% from the optimal eutrophic level. After the mesotrophic equilibrium is reached, the DIP load is allowed to attain a higher level than initially, at the eutrophic steady state.

The role of reactive iron is important in the recovery process, and it is therefore optimal to allow erosion to increase to its initial level; that is, all erosion control is temporarily ceased. In the transition process, the beneficial role of iron offsets the harmful effect of PP. While the system is returning to the mesotrophic steady state, PP abatement gradually increases to the level it was at in that state. At the mesotrophic state, the negative impact of eroded matter as a source of PP eclipses its favorable function as the source of reactive Fe. Oxic near-bottom water allows the coupled cycling of Fe and P and supports benthic animals in sediments. In the sensitivity analysis, we quantify the effect of iron content of soil on optimal policies. The optimal ratios remain almost unchanged up to a concentration half our default value. Below that, the relative importance of PP abatement starts approaching that of DIP abatement. However, even without any iron in eroded matter, marginal abatement costs for DIP should be about three times higher than those for PP.

The bottom panel in Fig. 5 presents the weights of optimal PP and DIP marginal abatement costs at the two steady states and along the transition path. The vertical axis presents the ratio of efficient unit abatement costs for DIP and PP. A value of 0.1, for instance, indicates that we should allow marginal abatement costs for DIP that are ten times higher than for PP; that is, the lower the ratio, the stronger the emphasis on DIP reduction.

Our model indicates that it would take less than ten years for the DIP stock to reach its new steady state but that it would take reactive iron, which inhibits benthic P release, several decades to reach the new equilibrium during the transition to the mesotrophic state (Fig. 5). In our deterministic framework, only the time lag between abatement and DIP stock is important, but obviously the system is less resilient during the first years after its return to the mesotrophic state.

4. Discussion and Policy Implications

Our approach combines the recent theory of coupled biogeochemical cycles (Burgin et al., 2011) and P speciation with a dynamic optimization model in order to analyze the relative importance of PP and DIP. Our study points out the need to understand the role of reactive iron in mitigating P-mediated eutrophication. The key results of our analysis are the following: i) society should put more weight on DIP than PP abatement; that is, it should accept markedly higher marginal abatement costs for DIP than for PP; and ii) if mesotrophic water ecosystems shift to a eutrophic state, the emphasis on DIP abatement should become even more pronounced. Given that resources are scarce, agri-environmental policies targeting P-driven eutrophication should thus focus more intensely on mitigating DIP than PP loading. We put forward this recommendation in light of our focus, the role of PP in P-mediated eutrophication control. We acknowledge that erosion control generates

economic benefits other than PP abatement at the farm level as well as environmental benefits for society at large.

Our results have implications for both policy making and future research. The most widely applied measures for abatement of P originating in agriculture are various forms of erosion control (e.g. the Conservation Reserve Program of the United States, Agri-Environmental Programmes in the EU). Research shows that while measures such as vegetative covers, no-till technologies or buffer zones do reduce PP loading, they increase or fail to affect DIP loading. To justify such measures from the viewpoint of eutrophication control, the resulting PP abatement should be very high if it is to offset the increase in DIP loading.

Mitigating DIP loading is time-consuming and costly. The main driver of DIP loading is the gradual accumulation of P in agricultural soils, the primary source at present being manure from animal agriculture (Innes, 2000; Kaplan et al., 2004). We should actively seek out economic and technological solutions that promote the use of manure in larger crop production areas, and thereby bring an end to the spatial accumulation of soil P. This would mean stricter P-index thresholds for following P-standards in the application of manure, limits which could not be circumvented by adopting management practices that reduce erosion loading. In the U.S., for instance, mandatory P standards that would become effective after certain P indices can be relaxed or avoided altogether by applying best management practices that target PP load. That is, DIP abatement measures can be offset by PP abatement measures at an (implicit) one-to-one ratio. See, for example, the manure management regulations for concentrated animal feeding operations (CAFO) in Pennsylvania (Pennsylvania, 2011).

5. Conclusions

We address long term efficiency of managing P mediated eutrophication of surface waters. Our model illustrated the economic consequences of hysteresis driven by benthic release of P: we may end paying more for P abatement and yet obtain inferior water quality compared to a state where healthy sediments retain substantial amounts of P, allowing us to enjoy higher water quality with less abatement effort. We demonstrate that if we are to curtail anthropogenic P loading efficiently, it will be crucial to harmonize the ecological models describing eutrophication of the target waters and the models and policies on P management in source areas. There are models which estimate the effects of agri-environmental measures on both PP and DIP loading, such as SWAT (Gassman et al., 2007); INCA and INCA-P (Jackson-Blake et al., 2015, 2016); and APEX, an extension of EPIC, [<http://epicapex.tamu.edu/epic/>]. Water quality models typically use even more detailed distinctions between different compounds containing P. The most compelling need in designing P abatement policies in agriculture, however, is that we acknowledge the well-established trade-offs between PP and DIP load indicated by experimental data and catchment model exercises, as well as the dissimilar effects of these two forms of P on eutrophication in the receiving waters. Increasing the precision of water quality and watershed models alone will not help: we also have to start adding precision to models of farmers' policy responses. When designing agri-environmental programs, we have to take into account to what extent the incentives prompt DIP mitigation and PP mitigation. The total effect on P-mediated eutrophication must be defined using water body-specific weights that commensurate DIP and PP into units of Eutrophying P. This has been standard procedure in designing climate policies, where outcomes of abatement efforts are evaluated as CO₂ equivalents. Our research highlights the necessity of taking similar steps in P-mediated eutrophication management.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <http://dx.doi.org/10.1016/j.ecolecon.2017.02.023>.

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