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Uncertainty-Accounting Environmental Policy and Management of Water Systems

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Environmental policies for water quality and ecosystem management do not commonly require explicit stochastic accounts of uncertainty and risk associated with the quantification and prediction of waterborne pollutant loads and abatement effects. In this study, we formulate and investigate a possible environmental policy that does require an explicit stochastic uncertainty account. We compare both the environmental and economic resource allocation performance of such an uncertainty-accounting environmental policy with that of deterministic, risk-prone and risk-averse environmental policies under a range of different hypothetical, yet still possible, scenarios. The comparison indicates that a stochastic uncertainty-accounting policy may perform better than deterministic policies over a range of different scenarios. Even in the absence of reliable site-specific data, reported literature values appear to be useful for such a stochastic account of uncertainty.

Introduction

Increasing water pollution and deterioration of aquatic ecosystems due to various anthropogenic pollutants may be one of the most serious current environmental problems (*1*−*6*). Environmental policies for mitigating and protecting water systems from such problems often seek to find a balance between maintaining or achieving certain environmental standards and minimizing possible impairments to economically and socially beneficial uses of the water environment. Objective investigation of where this balance may lie or efficient achievement of already decided standards requires quantification of expected minimum costs for environmental management and protection (*4*−*10*). The environmental standards may, for instance, be expressed as maximum concentration levels (MCLs), maximum pollutant loads (MPLs) (*5*), minimum aquatic ecosystem status (*4*), or minimum pollutant load reductions (e.g., HELCOM (*11*) for nitrogen loads to the Baltic Sea).

However, many studies have pointed out inherent uncertainties of complex environmental systems that make it difficult or impossible to reliably predict minimum costs and optimal allocation of pollution abatement measures (*5*, *12*−*20*). Policy analysts may also resist such optimization-type solutions, because in their experience, effective solutions may rather emerge from negotiation and compromise. For various reasons, it may, therefore, be necessary to investigate the performance of different environmental policies and management practices for a range of possible scenarios rather than to focus investigations on how to reach environmental targets optimally for some assumed single scenario (*20*).

For water quality and ecosystem management, the main uncertainties stem from difficulties to realistically quanitify the actual effects of upstream pollutant reductions on downstream pollutant loads and concentrations. Furthermore, it may be highly uncertain what reductions in pollutant load and concentration levels are necessary and sufficient for maintaining or achieving good water quality and ecosystem status in downstream water environments. Such difficulties are, for example, evident in widely varying policies for targeted reductions of nitrogen loads to the Baltic Sea, where environmental targets for this problem vary between different agreements and legislations. Internationally, policymakers agreed on a 50% reduction of *total* nitrogen load from the whole Baltic Sea drainage basin to the sea (*11*), while a 30% reduction of only *anthropogenic* nitrogen loads to the Baltic Sea is required in Swedish national environmental goals (*21*). Finally, only a 10% reduction of *anthropogenic* nitrogen load from the large and most heavily populated Swedish Norrström basin is regionally targeted (*22*). In general, a combination of both economic feasibility considerations and uncertainties about the natural system behavior may govern the definition of environmental policies. In the particular Baltic Sea example, it is clear that natural system uncertainty has been and still is an important reason for the range of different environmental targets (*23*).

Choice of a low target for pollutant reduction in the face of uncertainty may reflect a *risk-prone* environmental policy. Such a policy may assume that a small pollutant reduction may be sufficient to achieve desired water quality and ecosystem improvements without wasting limited resources on unnecessary extra measures. Choice of a high target for pollutant reduction under uncertainty may instead reflect a *risk-averse* environmental policy. Here, achievement of necessary environmental improvements may be a priority without worrying too much about economic efficiency. Both of these policies, however, disregard the possibility to account explicitly for risk and uncertainty in environmental management. Scientific development has produced methods for such explicit risk and uncertainty accounting (at least to some degree) in terms of models for pollutant load and concentration statistics that go beyond common expected value analyses (*16*−*18*, *24*−*37*).

In this paper, we formulate and investigate a possible environmental policy, in the following referred to as a *stochastic uncertainty-accounting* policy, that includes an explicit stochastic account of risk and uncertainty. We compare both the environmental and economic resource allocation performance of this policy with that of alternative *deterministic risk-prone* and *risk-averse* environmental policies under a range of different possible scenarios.

General Problem Description

We consider a general pollutant spreading and recipient loading situation in the total drainage basin of a given water recipient which may, for instance, be a river, a lake, or a coastal zone. The total drainage basin includes $i = 1, ..., I$ different pollutant emissions $E_{i,m}$, (denoted by $E'_{i,m}$ in the pre-abatement state) from various point and diffuse sources within possible different catchment areas $m = 1, ..., M$ within the whole basin. For each source i, there may be $j = 1, ...,$

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TABLE 1. General Cost Minimization Model for Achieving Maximum Pollutant Load $L_{\max}$ for Either Assumed Deterministic or Stochastic Recipient Pollution Loads L[a]

general cost minimization model

$$\mathrm{Min}_{X_m^{ij}\tilde{X}_m^k} \sum_m \left[\sum_{ij} C_m^{ij}(X_m^{ij}) + \sum_k C_m^k(\tilde{X}_m^k) \right] \tag{1}$$

subject to environmentally necessary maximum pollutant load $L_{\max}$ for

deterministic constraint

$$L = \sum_m L_{R,m} = \sum_m \left[\sum_i (\alpha_m^i (E'_{i,m} - \sum_j X_m^{ij})) - \sum_k \tilde{X}_m^k \right] \leq L_{\max} \tag{2}$$

stochastic constraint

$$\mathrm{prob}\left(L = \sum_m L_{R,m} = \sum_m \left[\sum_i (\alpha_m^i (E'_{i,m} - \sum_j X_m^{ij})) - \sum_k \tilde{X}_m^k \right] \leq L_{\max} \right) \geq \beta \tag{3}$$

stochastic constraint expressed as equivalent to the deterministic one for normal pollutant load probability distribution

$$\sum_m E[L_{R,m}] + K^\beta \sqrt{\mathrm{Var}\left[\sum_m (L_{R,m}) \right]} \leq L_{\max} \tag{4}$$

[a] Abatement costs in catchment m consist of costs $C_m^{ij}(X_m^{ij})$for pollutant emission abatement by measure j at emission sources i and $C_m^k(\tilde{X}_m^k)$ for pollutant load reduction at the recipient by downstream-located pollution load reduction measures k. The catchment m load $L_{R,m} = \sum_i(\alpha_m^i(E'_{i,m} - \sum_j X_m^{ij})) - \sum_k \tilde{X}_m^k$ to the recipient depends on pre-abatement source emissions $E'_{i,m}$ and source emission reductions X_m^{ij}, pollutant delivery coefficients α_m^i and direct pollutant load reductions $\tilde{X}_m^k$ by downstream load abatement measures. The general target is to reduce total pollutant load to $L = \sum_m L_{R,m} \leq L_{\max}$ at minimum total abatement cost. Considering that L may be stochastic, the desired probability of abatement success is quantified by β, with $E[L_{R,m}]$ and $\mathrm{Var}[L_{R,m}]$ being the expected value and variance and K being the confidence interval parameter of assumed normal distribution $L_{R,m}$.

J different possible measures to reduce pollutant emission X_m^{ij} at the source. In addition, k downstream abatement measures, such as constructed wetlands, may yield a direct reduction in pollutant load $\tilde{X}_m^k$ at the outlet of each catchment area m into the water recipient.

For every reduction in source emission X_m^{ij}, the resulting pollutant load $L_{k,m} = \sum_i(\alpha_m^i(E'_{i,m} - \sum_j X_m^{ij}))$ at the catchment outlet, or into a downstream abatement measure k near the outlet, is determined by the natural pollutant attenuation that may occur along the pollutant transport pathway from the source to the recipient. The attenuation effect is quantified by a pollutant mass delivery fraction α_m^i (also referred to as a delivery coefficient) for each source i within each catchment area m. The total pollutant load after all reductions in upstream source emissions and downstream loads in a catchment area m is then $L_{R,m} = \sum_i(\alpha_m^i(E'_{i,m} - \sum_j X_m^{ij})) - \sum_k \tilde{X}_m^k$. Furthermore, the total pollutant load into the recipient after all reductions in source emissions and downstream loads in all catchment areas is $L = \sum_m L_{R,m}$; in the pre-abatement state, the corresponding total recipient load is $L' = \sum_m L'_{R,m} = \sum_m \sum_i \alpha_m^i E'_{i,m}$.

To achieve desired environmental conditions in the recipient, the total pollutant load L must meet some maximum pollutant load constraint $L \leq L_{\max}$. The abatement costs for meeting this constraint by reducing pre-abatement L' to postabatement $L \leq L_{\max}$ by some (optimal) combination of reductions in pollutant source emissions and outlet loads, X_m^{ij} and $\tilde{X}_m^k$, are given by cost functions $C_m^{ij}(X_m^{ij})$ and $C_m^k(\tilde{X}_m^k)$, respectively.

All of the different explanatory parameters in the expressions for L' and L, i.e. all preabatement pollutant source emissions $E'_{i,m}$, delivery coefficients α_m^i, pollutant emission and load reductions X_m^{ij} and $\tilde{X}_m^k$, and subcatchment loads $L_{R,m}$, may be subject to random variation and quantification uncertainty. As a consequence, all these explanatory parameters, and thus also the resulting total pollutant loads L' and L, may be expressed as stochastic variables to account explicitly for a range of different pollutant source-transport-effect uncertainties that have been identified and quantified in the scientific literature (*16*−*18*, *24*−*39*).

Explicit consideration of the randomness and uncertainty of stochastic loads $L_{R,m}$ implies that they are characterized by statistical measures, such as expected value $E[L_{R,m}]$, variance $\mathrm{Var}[L_{R,m}]$, standard deviation $\mathrm{STD}[L_{R,m}] \equiv (\mathrm{Var}[L_{R,m}])^{1/2}$ and coefficient of variation $\mathrm{CV}[L_{R,m}] \equiv \mathrm{STD}[L_{R,m}]/E[L_{R,m}]$. Moreover, uncertainty about the environmentally necessary pollutant load constraint $L \leq L_{\max}$ implies that this constraint can only be expected to be achieved with a certain probability $\beta \leq 1$, which depends on the pollutant load statistics. Assuming instead that all of the explanatory parameters and the resulting total pollutant load L are deterministically known implies that their randomness and uncertainty are either neglected or somehow implicitly accounted for.

Cost-efficient abatement requires that the pollutant load constraint $L \leq L_{\max}$ is achieved at minimum cost. Table 1 summarizes some general cost-minimization equations that need to be solved for either deterministic or stochastic pollutant-load assumptions. These equations have been used and described in more detail by Baresel et al. (*40*, *41*) and by the ERMITE Consortium (*6*) for deterministic and by Gren et al. (*16*, *17*) for stochastic pollutant-load considerations. The summary in Table 1 emphasizes that the stochastic formulation of the pollutant load constraint $L \leq L_{\max}$ has an equivalent form to the corresponding deterministic constraint for a normal probability distribution of stochastic pollutant loads (*42*). A similar constraint expression has also been formulated for log-normally distributed pollutant loads (*17*), but we consider here only the normal distribution expression for simplicity and clarity.

In general, uncertainty about health, ecosystem, and general environmental effects of different pollutant load levels L makes it difficult to identify the exact maximum pollutant load allowed $L_{\max}$ to ensure the desired environmental improvements are achieved. The pollutant load target chosen by any environmental policy is subsequently denoted by $L^*_{\max}$. The * index emphasizes that $L^*_{\max}$ is the load target chosen, which may or may not equal the real $L_{\max}$ value required to achieve the environmental improvements ultimately targeted by the environmental policy.

Even for the same environmental improvement aims, different possible environmental policies can be considered for choosing the maximum allowed pollutant load $L^*_{\max}$ to achieve these aims:

(I) A *deterministic risk-prone* policy may, for instance, promote the choice of a relatively large maximum allowed pollutant load $L^*_{\max}$ and neglect uncertainty by considering

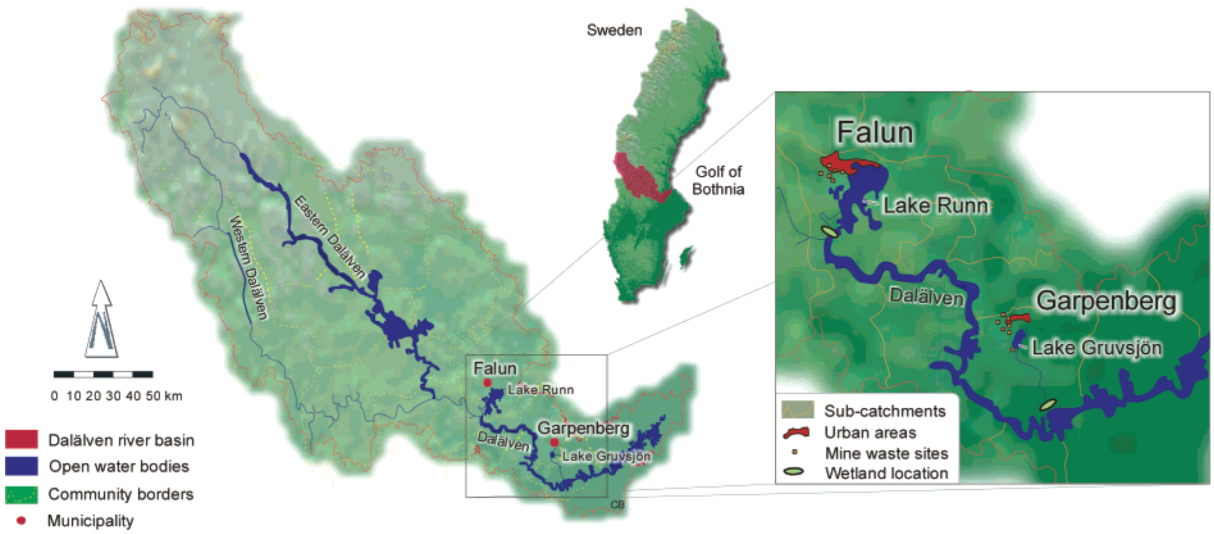

FIGURE 1. Location map of the Swedish Dalälven River basin and its main mining areas, Falun within the subcatchment of Lake Runn (∼3065 km²) and Garpenberg within the subcatchment of Lake Gruvsjön (∼126 km²).

a deterministic cost-minimization solution (Table 1) for efficient allocation of abatement measures to achieve the relatively relaxed environmental constraint $L \leq L^*_{max}$ chosen.

(II) A *deterministic risk-averse* policy may instead promote the choice of a relatively small maximum pollutant load L^*_{max} as a safety factor in an implicit uncertainty account, which still considers a deterministic cost-minimization solution (Table 1) for allocation of abatement measures to achieve the relatively strict constraint $L \leq L^*_{max}$ chosen.

(III) A *stochastic uncertainty-accounting* policy may promote the choice of a similarly large maximum pollutant load L^*_{max} as for policy I, but require also an explicit account of uncertainty in terms of a stochastic cost-minimization solution (Table 1) for abatement measure allocation to meet the relatively relaxed environmental constraint $L \leq L^*_{max}$ chosen with success probability β. This policy also necessitates an explicit quantification of uncertainty in terms of an estimated coefficient of variation of pollutant loads $CV[L_{R,m}]^*$, which may or may not equal the real population statistic of stochastic pollutant loads $CV[L_{R,m}] \equiv Var[L_{R,m}]/E[L_{R,m}]$.

To compare the performances of these different environmental policies, the following section exemplifies a case-specific policy and cost-minimization (Table 1) quantification.

Site-Specific Model and Policy Quantification Example

Case Study Area. The specific case study considers zinc loading to the Swedish River Dalälven (Figure 1) arising from mining activities within its total drainage basin. The Dalälven River basin (30 000 km²) is located in the Bergslagen mining region and extends from the Swedish border mountains in the west to the Gulf of Bothnia (Baltic Sea) in the east (Figure 1). The basin includes approximately 75% forestland, 6% water, 4% agricultural areas, and mountains, marshes, and urban areas in the remaining 15% of its total area. In the upper part of the catchment, where the river is divided into the Western and Eastern Dalälven, zinc concentrations are approximately at background levels (*43*). The lower industrialized part of the catchment includes two major areas of both historical and currently active mining importance, Falun and Garpenberg (Figure 1).

Model and Policy Quantification

Total zinc loading into the Dalälven River is considered to stem mainly from present and past mining wastes of non-ferrous ores in the Falun and Garpenberg subcatchments ($m = 1, 2$) (Figure 1). Table 2 lists reported zinc emission ($E'_{i,m}$) estimates for mine waste deposits and possible other diffuse sources in these subcatchments (*43–45*). Furthermore, it lists reported (*43, 45*) estimates of subcatchment average zinc delivery fractions (α_m) to the River Dalälven.

For site-specific quantification of the expressions in Table 1, we use reported estimates of source emissions $E'_{i,m}$ and delivery coefficients α_m, averaged over the subcatchments in Table 2 as fixed values in eq 2 and as mean values in eq 3 of Table 1. The source abatement measures considered to reduce pollutant emissions (X^{ij}_m) are soil- or water-covering of mine waste deposits. Downstream measures for reductions in pollutant loads ($\tilde{X}^k_m$) are possible wetland constructions just upstream of the subcatchment outlets into the River Dalälven. In general, we use the same model setup here as in previous deterministic cost-efficiency studies (using eqs 1 and 2) of zinc load abatement in the Dalälven River basin (*6, 40, 41*). This model setup includes fixed values in eq 2 and mean values in eq 3 to determine the effectiveness and costs

TABLE 2. Estimates of Zinc Loads into the River Dalälven for the Two Dominating Subcatchment Considered, Falun and Garpenberg, in Terms of Estimated Pre-abatement Emissions $E'_{i,m}$, Including Total Leakage from Mine Waste Sites, Estimated Leakage from Diffuse Sources and Loading from Upstream Areas. The Table Also Lists Reported Estimates of Subcatchment Average Zinc Delivery Fractions, α_m

	parameter	zinc
subcatchment Falun, $m = 1$		
leakage from mine waste deposits [kg yr^{-1}][a]	$E'_{i,1}$	289 600
diffuse metal leakage [kg yr^{-1}][a,b]	$E'_{i,1}$	31 250
delivery coefficient [%][c]	α_1	75
subcatchment Garpenberg, $m = 2$		
leakage from mine waste deposits [kg yr^{-1}][d]	$E'_{i,2}$	6031
diffuse metal leakage [kg yr^{-1}][e,b]	$E'_{i,2}$	587
delivery coefficient [%][c]	α_2	63

[a] From Hartlén and Lundgren (*45*), including the sites Kiesbränder, North Industry, Slag fills and heaps, Korsgården, Nya Sandmagasinet, Oxide paint materials, Gruvområdet, Gamla Berget and Galgbergsmagasinet (covered). [b] From Lindeström (*43*), including upstream sources, natural depositions, soil loads, water treatment plants. [c] From Hartlén and Lundgren (*45*), and Lindeström (*43*). [d] From Fällman and Qvarfort (*44*); Länsstyrelsen Dalarna Län (*46*), including the sites Herrgården, Järnvägsbanken, Odalfältet, Tappdammarna, Östra magasin, Västra Sandmagasin and Lilla Bredsjön. (e) From Hartlén and Lundgren (*45*).

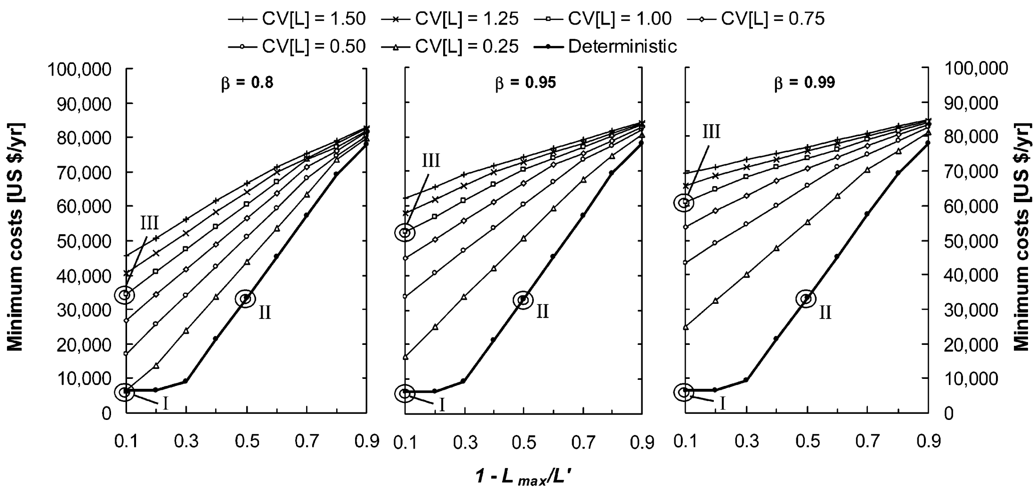

FIGURE 2. Minimum costs for cost-efficient solutions of the cost minimization models in Table 1 for deterministic (eqs 1 and 2) and stochastic (eqs 1 and 4) pollutant loads. These costs are functions of the reduction in zinc load necessary ($1 - L_{max}/L'$) for different variability values CV[L] for the zinc load and desired minimum probabilities β of reaching $1 - L_{max}/L'$ in the specific Dalälven case study. The minimum costs (see marked point values) are also identified for optimal abatement measure allocation under the three case-specific environmental policies: I, deterministic risk-prone; II, deterministic risk-averse; and III, stochastic uncertainty-accounting.

of abatement measures. (The Supporting Information provides a summary of the characteristics and cost assumptions of the abatement measures used, and references to previous papers using similar assumptions.)

As discussed above for the general case, in this specific case there are also several reasons to expect at least some unpredictable and uncertain variation of $E'_{i,m}$, α^i_m, X^{ij}_m, and $\tilde{X}^k_m$ around their reported average values. A previous study (*41*) discussed in detail the possible uncertainty of $E'_{i,m}$ data available for this case study. There are no further data available for accurate estimation of source-pathway-specific α^i_m values for zinc transport and attenuation. This is why only subcatchment average delivery coefficients α_m are used here, as reported in previous studies (Table 2), even though real source-pathway-specific α^i_m values will undoubtedly vary around average α_m values. In addition, such variability and randomness is also expected around the average values of pollutant reduction parameters X^{ij}_m and $\tilde{X}^k_m$. However, the aim of this paper is not to resolve the various variability, randomness, and uncertainty contributions of different parameters to the total pollutant load statistics, but to account for the possible latter statistics in stochastic eqs 3 and 4 of Table 1. For this purpose, we consider only the resulting pollutant load as the random variable and use the same CV[$L_{R,m}$] = CV[L], CV[$L_{R,m}$]* = CV[L]* and β values for both subcatchments in our example for simplicity. With the standard global optimization solver GAMS (General Algebraic Modeling System; ref *47*), we solve the stochastic cost-minimization problem (eqs 1, 3, and 4 in Table 1) and the deterministic cost-minimization problem that has also been investigated in previous studies (*6*, *40*, *41*).

Results in Figure 2 for different reduction levels in pollutant load $1 - L_{max}/L'$ show that the minimum costs increase with the increasing zinc load uncertainty CV[L] and the desired probability of abatement success β. For high target load reductions; however, the abatement costs converge because the remaining expected zinc load E[L] after abatement is then small, also implying a small remaining absolute uncertainty STD[L] for constant CV[L] = STD[L]/E[L].

Case-study specific application of the three different environmental policies may further be exemplified as follows: policy I, *deterministic risk-prone,* with a chosen minimum zinc load reduction of 10%, i.e. $1 - L^*_{max}/L' = 0.1$; policy II, *deterministic risk-averse,* with a chosen minimum zinc load reduction of 50%, i.e. $1 - L^*_{max}/L' = 0.5$; and policy III, *stochastic uncertainty-accounting,* with a chosen minimum zinc load reduction of 10%, i.e. $1 - L^*_{max}/L' = 0.1$ and probability of abatement success of $\beta = 0.99$ for assumed CV[L]* = 1. Any policy-related choice of L^*_{max} must assume that this maximum load value is sufficient (i.e., equals the real necessary maximum load L_{max}) to achieve the desired environmental improvements. Furthermore, any CV[L]* value assumed in policy III implies that the associated standard deviation STD[L] (here 100% of E[L] for CV[L]* = 1) is believed to realistically quantify the variation of the random pollutant load L around its expected value E[L]. Figure 2 identifies minimum abatement costs if these underlying assumptions were, in fact, true; these particular costs in our example are always higher for policy III than for policies I and II, but depend generally on the CV[L]* value assumed in policy III (Figure 2). In the following section, we investigate the resulting abatement and economic performance of these different policies in the general case, for which we do not really know if these underlying assumptions are true or not.

Comparative Environmental Policy Results

We follow a similar approach to that previously used by Popper et al. (*20*) for the present novel investigation of comparative policy performance under different hypothetical but possible scenarios for CV[L] and $1 - L_{max}/L'$. The CV[L] value in such a scenario determines the average magnitude of (deterministically) unpredictable and thus (apparently) random fluctuation of pollutant load L around its expected value E[L]. Such random L components arise from both temporal variation (e.g., random interannual variability around mean annual conditions if L is the annual load, refs *33*, *34*) and spatial variation (i.e., pollutant load $L_{R,m}$ variability around mean $L_{R,m}$ for different heterogeneous subcatchment areas m (*24*, *26*). Common measurement and monitoring gaps and errors imply that estimates of both E[L] and CV[L] are uncertain (*17*, *38*, *39*, *48*). In particular, CV[L], which is a measure of relative L variability around E[L], requires larger measurement and monitoring samples than E[L] for realistic quantification and is, therefore, more difficult and uncertain to quantify than E[L]. Similar statistical moment uncertainty applies to all underlying explanatory parameters of L: the source emissions $E'_{i,m}$, the delivery coefficients α^i_m, and the emission and load reductions X^{ij}_m and $\tilde{X}^k_m$ that may be achieved by abatement.

Furthermore, the load constraint parameter L_{max} in each scenario determines the relative pollutant load reduction 1

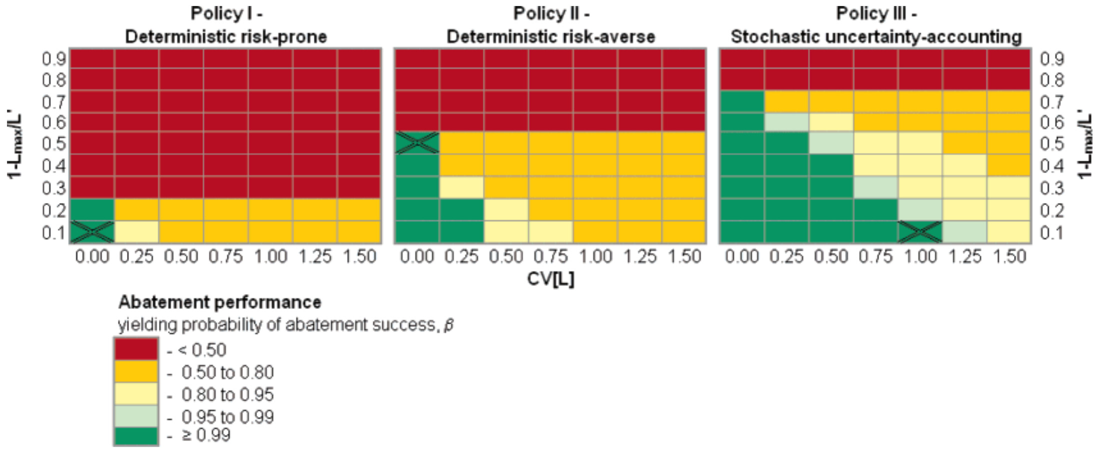

FIGURE 3. Abatement performance of policies I−III in reducing zinc loads in terms of the probability β of meeting the environmentally necessary reduction in zinc load $1 - L_{max}/L'$, for different possible realizations of $1 - L_{max}/L'$ and of the zinc load variability measure CV[L]. Policy-specific reductions in zinc load $1 - L_{max}/L' = 1 - L^*_{max}/L'$ and load variability CV[L] = CV[L^*] are marked with a cross in the relevant parameter combination cell for each policy.

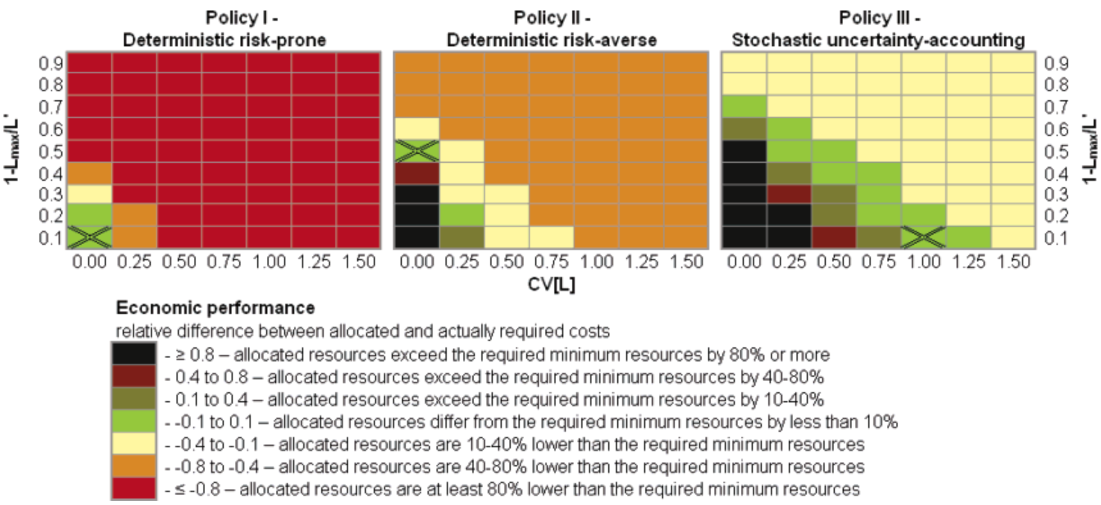

FIGURE 4. Economic performance of policies I−III in reducing zinc loads for the desired probability of success $\beta = 0.99$ for different possible realizations of the environmentally necessary zinc load reduction $1 - L_{max}/L'$ and of the zinc load variability measure CV[L]. The economic performance is measured as the relative difference between policy-specific minimum costs of abatement (see singular policy-related optimal points for $\beta = 0.99$ in Figure 2) and the minimum abatement costs for meeting $1 - L_{max}/L'$ in the actual scenario with given CV[L] value (obtained from the optimal solution functions for $\beta = 0.99$ in Figure 2). Policy-specific reductions in zinc load $1 - L_{max}/L' = 1 - L^*_{max}/L'$ and load variability CV[L] = CV[L^*] are marked with a cross in the relevant parameter combination cell for each policy.

$- L_{max}/L'$ necessary to achieve the desired environmental improvements in that scenario. Both L_{max} itself and the associated load reduction necessary $1 - L_{max}/L'$ are uncertain owing to a lack of full understanding of environmental, ecosystem, and toxicological functions and effects (*30*, *49*). For similar reasons as discussed above for L, the pre-abatement load L' in $1 - L_{max}/L'$ varies around its expected value $E[L']$, which implies that both the $E[L']$ value and the magnitude of its variability may affect the real load reduction necessary $1 - L_{max}/L'$. As a general consequence of the variability and uncertainty of L', L and L_{max}, actual realizations of CV[L] and $1 - L_{max}/L'$ may differ from the CV[L]* and L^*_{max} assumptions of the different policies.

In our example, environmental policies I−III will only perform optimally (i.e., the policy-related optimal points indicated in Figure 2 will only be truly optimal) in achieving the environmentally necessary reduction in zinc load $1 - L_{max}/L'$ at minimum cost if indeed the policy-assumed $1 - L^*_{max}/L' = 1 - L_{max}/L'$ and CV[L]* = CV[L]. For most real scenarios, however, $1 - L^*_{max}/L' \neq 1 - L_{max}/L'$ and CV[L]* ≠ CV[L] should be expected owing to the estimation uncertainty of these quantities. Under such uncertainty, the different environmental policies may perform more or less well and efficiently in achieving the reduction of zinc load necessary $1 - L_{max}/L'$ and in allocating the correct economic resources for this reduction. Figure 3 illustrates the performance calculated for the three policies considered in terms of resulting probability of abatement success, β, for different possible realizations of $1 - L_{max}/L'$ and CV[L].

The results in Figure 3 show that the management option of policy I, deterministic risk-prone, fails to result in even a 50% probability of abatement success in most scenarios investigated. Policy II, deterministic risk-averse, performs better, with at least a 50% probability of abatement success in slightly more than half of the scenarios investigated. Policy III, stochastic uncertainty-accounting yields the best management results with at least 80% probability of abatement success in more than half (60%) of the scenarios investigated. It further achieves its own chosen load reduction target ($1 - L^*_{max}/L' = 0.1$) with at least 95% probability for all but the highest pollutant load uncertainty investigated (CV[L] = 1.5). Policies I and II only achieve a similar probability of compliance with the policy-specific load reduction targets ($1 - L^*_{max}/L' = 0.1$ and 0.5, respectively) for zero pollutant load variability and uncertainty (CV[L] ≤ 0.25). Furthermore, policy II complies with the much lower chosen load reduction target of the other two policies, $1 - L^*_{max}/L' = 0.1$, only under conditions of small load variability and uncertainty (CV[L] ≤ 0.25).

Figure 4 illustrates the economic resource allocation performance of the three different policies. It shows the relative difference between the costs of optimal abatement measure allocation under policy-assumed conditions (see the singular policy-related optimal points in Figure 2) and those under actual scenario conditions (see optimal solutions for various conditions in Figure 2). The results in Figure 4 imply that economic resource allocation under policy I, deterministic risk-prone, would be insufficient for almost all scenarios investigated. Economic resources allocated under policy II, deterministic risk-averse, would be wasteful to optimal for some, but mostly insufficient for the majority of scenarios investigated. Policy III, stochastic uncertainty-accounting, would perform best economically for the investigated scenario range (which for the load reduction $1 - L_{max}/L'$ is the total possible range) among the policies investigated. This economic performance ensures a good or near-optimal allocation of economic resources for zinc load abatement in most scenarios investigated.

It is, of course, possible to reduce the uncertainty, and thereby facilitate better performance of deterministic policies I and II, by gathering additional information at some additional cost, so that any policy-specific CV[L]* and $1 - L^*_{max}/L'$ values are closer estimates of the real CV[L] and $1 - L_{max}/L'$ values (*50*, *51*). The present methodology and types of results may then indicate whether an additional reduction in uncertainty is economically justified. A decrease in uncertainty may, for instance, decrease pollutant load variability to the range 0 ≤ CV[L] ≤ 0.25. If the load reduction necessary is also small, say $0 \le 1 - L_{max}/L' \le 0.2$, the results in Figures 3 and 4 indicate that all policies investigated would perform well with regard to the probability of achieving such a low load reduction. The stochastic uncertainty-accounting policy III, however, would in this particular case be too costly, because both the load uncertainty and the load reduction necessary are small. Both the deterministic risk-prone policy I and risk-averse policy II in this case would show better economic performance than policy III. The additional cost of such a decrease in uncertainty, however, would only be economically justified if it were lower than the cost reduction (and possible other additional benefits) associated with choosing policy I or II instead of policy III. This possible cost reduction is explicitly quantified by the singular policy cost points in Figure 2 (panel for desired probability of abatement success $\beta = 0.99$, i.e., the same as for the results in Figures 3 and 4). This comparative approach can be used for different possible policy quantifications and expected decreases in uncertainty.

With regard to the possible generality of the present results, it may be instructive to consider the possibility of choosing a much higher target for load reduction, say $1 - L^*_{max}/L' = 0.9$, as an extra safety factor in the deterministic risk-averse policy II. With such a choice, policy II would, of course, perform better in terms of probability of abatement success over the whole range of scenarios investigated. With regard to economic performance, however, such a policy choice would be far too costly (>80% greater than the minimum cost requirements) for many scenarios (see Figure 2 for the cost increase with increasing load reduction target).

In general, our results indicate that a policy involving explicit stochastic accounting for uncertain pollutant loads may provide good combined environmental and economic performance over a wide range of different scenarios. The prerequisite for good performance of a stochastic uncertainty-accounting policy is of course that there is some significant uncertainty about pollutant loads; otherwise, there is no meaning or reason at all for using an uncertainty-accounting policy.

Other studies in the literature have suggested that the uncertainty of waterborne pollutant loads may often be large (e.g., *52, 53*), and the scientific basis for supporting a stochastic account of pollutant load uncertainties has developed considerably over the last 15 years (*16*−*18*, *24*−*37*). The very large input data requirements for such an uncertainty account, however, have so far appeared prohibitive for useful practical application (*5*, *38*, *39*).

The value range investigated here for the pollutant load coefficient of variation (CV between 0 and 150%) is consistent with reported CV values around peak loads and concentrations in calculated pollutant-breakthrough statistics (*27*−*29*, *31*−*34*). The present results are novel in that they indicate that, in the absence of reliable site-specific data, such literature ranges may be useful for capturing the essential uncertainty of pollutant load quantifications and predictions. Explicit uncertainty accounting, therefore, appears not only to be academically interesting, but also practically feasible for identifying environmental policies for water systems with good environmental and economical performance over a range of different scenarios.

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Supporting Information Available

A summary of abatement measure costs and characteristics, and references to related papers for more detailed descriptions. This material is available free of charge via the Internet at http://pubs.acs.org.

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