Loga et al. Environmental Sciences Europe (2022) 34:42
https://doi.org/10.1186/s12302-022-00625-z

Environmental Sciences Europe

RESEARCH

Open Access

Bayesian decision tables for estimation of risk of water management decisions based on uncertain surface water status: a case study of a Polish catchment

Małgorzata Loga[1*], Mikołaj Piniewski[2] and Paweł Marcinkowski[2]

Abstract

Background: Uncertain results of the status assessment performed as required by the Water Framework Directive can be responsible for misclassification of a water body's status and may lead either to risk due to undertaking unnecessary remediation actions or risk of penalties for refraining from any action and not reaching environmental goals. Based on Bayesian decision theory, optimal decision tables are shown for two examples of water quality indicators, for a river catchment in central Poland. To overcome the problem of scarcity of publicly available monitoring data, the existing SWAT model for the studied catchment was used to generate nutrient concentration time series for the baseline conditions and under different scenarios. The status classes assessed based on annual mean concentrations of daily values for total phosphorus and total nitrogen were adopted as the 'true' status classes of the water bodies based on each indicator. SWAT simulation results enabled calculation of probability distributions of concentrations for the stochastic states of the water body, both for the period before and after the performance of corrective actions.

Results: Bayesian decision tables consisted of alternative management decisions including modernization of the existing wastewater treatment plants in the case of phosphorous and also of fallowing agricultural areas in the case of nitrogen. An example of a penalty calculation procedure is presented in the event that the subject of the case before the EU Court of Justice would be failure to achieve the environmental objectives by all water bodies belonging to the selected catchment.

Conclusions: Detailed discussion of this analysis indicates the potential benefits in terms of minimization of costs/losses that the proposed methodology may bring to the protection of surface waters.

The presented method of risk analysis for making decisions on remedial actions when uncertainty exists about the water status assessment, can be considered as a prototype of a general methodology prepared for implementation in water protection. Unfortunately paying fines instead of taking remediation measures might be optimal for uncertain status of water bodies.

Keywords: Water management, Bayesian decision theory, Uncertainty of water body status, SWAT model

*Correspondence: malgorzata.loga@pw.edu.pl

[1] Faculty of Building Services, Hydro and Environmental Engineering, Warsaw University of Technology, Nowowiejska 20, 00-653 Warsaw, Poland
Full list of author information is available at the end of the article

Background

Pursuant to the Water Framework Directive (WFD) [1], water management decisions regarding the protection of water resources should lead to corrective actions whenever the assessment of a given water body (wb) indicates bad status which can be either invoked by below good chemical status or/and moderate or worse ecological status. Remediation actions may be abandoned when the assessment of the status is good which is only in case of both good chemical and good or high ecological status. Assessing a status on the basis of measurement data burdened with random disturbances and obtained under changing environmental conditions requires treating the status itself as a random variable with all associated economic and environmental risks of such an approach.

Uncertain (biased) results of the status assessment can be responsible for misclassification of a wb thus creating problems in undertaking decisions and leading to undesired consequences, such as:

a) false-positive assessment (good status of a wb when its true status is below good). This can prevent implementation of corrective or remedial actions by water authorities;
b) false-negative assessment (below good status (bgs) of a wb when the true status is good). This can trigger a decision of implementing unnecessary, difficult and costly remedial measures.

The risk which appears in the title of this paper in case of false-positive assessment is understood as a financial penalty which would be imposed by the Court of Justice for not fulfilling the obligation of the member state country of achieving good status, which is imposed by the WFD throughout the whole EU. The risk, in the case of false-negative assessment, consists of costs of implementing corrective actions when in reality they are unnecessary (e.g. equipping sewage treatment plants with additional treatment stages or implementing new, more environmentally friendly, but expensive technology).

It is worth mentioning that currently, in 2021, some 6 years after the deadline of achieving good water status in the European Union area set by the WFD in 2000, all EU countries failed to reach this ambitious goal. Since no EU country has been brought before the Court of Justice for not achieving good water status, it is therefore difficult to predict, with any certainty, how these penalties could be calculated.

In order to check the risk connected with various water management decisions for the purpose of presenting an example of such a procedure, a' true' status of water body should be defined and known. When harmonizing national legislation with the WFD, in all EU member state countries, the class boundaries for physicochemical quality elements were determined and the principle of classification of wbs were set based on the annual mean value of the (usually 12) measurements. Such a rule of water status assessment may lead to a situation where a certain subset of measurement data in a given period consists of values smaller than the lower limit of the class indicated by the annual mean, and another subset comprises values greater than the upper limit of this class. Naturally, the mean value of all measurements, due to the compensation of values smaller and higher than this mean, falls within the class boundaries determined by this mean. Summing up, it can be stated that although the surface water monitoring system adopted in the EU works quite well, the above-described paradoxical situation indicates that the 'veracity of the wb status' determined in accordance with the WFD may have nothing to do with the physical condition of the water environment in the river.

There are a considerable number of papers focused on uncertainty indices of the biological quality elements which are most important in the ecological status assessment procedure. Some examples of these papers include: [2] for macrophytes; [3] or [4] for phytobenthos; and [5, 6] for phytoplankton in both flowing and standing waters. At the early stage of introducing the WFD to national monitoring systems, projects were oriented on intercalibration of methods and elaboration of consistent threshold values for classes [7–9]. Many papers tried to summarize the results of the WFD on the 10th,15th and 20th anniversary of its enforcement [10–14] and presenting results from the 2nd and 3rd water management cycles like [15] or [16] including ecological and chemical status assessments of water bodies in EU.

Definitely such estimates of status are uncertain and characterized by considerable probability of misclassification. The probability that wb is in particular status can be estimated on the basis of the existing monitoring measurements of the aquatic environment. It can be assumed that the distribution function g $(\overline{x})$ of the average value of the physiochemical element $\overline{x}$ is approximated by the Student's t-distribution function, where two parameters of this distribution—the expected value and the standard deviation are estimated from the monitoring data.The example of Poland shows how significant and common the problem of water status assessment uncertainty can be: 25% of water bodies were classified as good ecological status, but their assessment was characterized by very high (higher that 0.5) probability of misclassification [17]. For the chemical status it was found that assessments may be incorrect in the case of approximately 25% of river water bodies and 30% of lake water bodies categorized as good, and 20% of both types of water bodies classified as below good status [18].

To narrow the uncertainty ranges of water quality indicator values [19] and to decrease the probability of misclassification of water body status to assist the water managers in decision-making, some modification in the monitoring programmes especially in terms of frequency of sampling, should be introduced. In order to adjust the monitoring programme to meet the needs/demands of water managers, the acceptable level of probability of misclassification should be known. Despite seeking cooperation with water managers, to get to know the acceptable level of assessment of status uncertainty, no success has been achieved.

Although there are some hints on how to tackle the risk of misclassification within the prepared Common Implementation Strategy for The Water Framework Directive [1, 20] the authors are not aware of any study that attempted to quantitatively analyse the consequences of any uncertainties in status assessment for water management decision.

The goal of this study is to fill this gap and focus on estimating the risk associated with remedial actions in the context of the requirements of the WFD. It was decided to focus on Bayesian decision theory, as the result of it can be applied and become an assistance to water administrations, even if this assistance is not expected/welcome.

Based on the sampling, with patterns characteristic for operational monitoring set by the WFD, where status assessment uncertainty depends on the mean belonging to the interval of concentration values defining status class, optimal water management decisions for various alternatives concerning remediation measures have been worked out. Here, instead of historic data from the Polish State Environmental Monitoring Programme, virtual reality simulated by the SWAT model has been applied. Among the factors taken into account when preparing the decision tables were: costs of modernization of sewage purification facilities; decrease of areas used for agricultural purposes; and penalties for refraining from remediation actions. The resultant decision tables were prepared discretely for relatively narrow ranges of concentration of polluting substances.

There are many interesting examples reported in the literature where the Bayesian approach was used successfully for solving important problems for water environment protection. One of the examples involved cooperation by fisheries managers and wild life researchers, determining the optimal fish length limit for catching which was safe for the fish population stability and at the same time maximized the fishermens' satisfaction [21]. Another example of collaboration between water managers and fishery biologists was working out optimal flow regimes when operating a dozen dams to protect salmonid populations [22].

Similarly, a very interesting problem also solved with the assistance of the Bayesian approach was the design and optimal setting of a system of monitoring sensors in an urban drainage system for non-conservative contaminants [23].

Here, the adopted method is the Bayesian model of decision-making in the game with Nature [24]. In this paper, only ecological status assessment is taken into consideration.

The methodology of determination of risk for water management decisions has been exemplified based on two physiochemical quality elements, i.e. total phosphorus (TP) and total nitrogen (TN), as they are frequently the reason for bgs class.

Concerning the possibility of a decrease of phosphorus concentration in river water, building new WWTPs or modernizing the existing ones are the most efficient options. However, in terms of combating elevated concentration of nitrogen, there are also possible alternatives concerning agricultural practices. In rural areas with dispersed dwellings, it is not economically justified to build a sewage system connected with treatment plants [25]. In this case, measures aimed at solving the sewage problem are household sewage treatment plants or septic tanks since, when properly sealed, septic tanks pose no direct threat to waters. As the major source of nitrogen is agricultural areas due to either mineral fertilizers or manure, the potential remedial action may be reduction of fertilizer doses, introduction of buffer zones or reduction of the cultivated area by fallowed areas.

Precisely ascertaining the condition of a wb is difficult in a real water monitoring system, because only fundamental aquatic attributes such as DO, pH, soluble reactive phosphorus and NO_3 can be obtained via real time sensors [26], enabling their continuous measurement in time. For the purpose of this paper, to overcome this problem, time series of water quality elements were generated by the SWAT simulation model (Soil and Water Assessment Tool) [27]. Using the SWAT model also enables simulating the potential effects of hypothetical scenarios of various remediations measures. In this study, the true class was the one indicated by the annual mean concentration of 365 daily values generated by the SWAT model. In the adopted methodology, using the SWAT model enables calculation of probability distributions of the states $\mathbf{S_i}$ based on the frequency of occurrence of status classes in the time series of concentrations simulated both before and after the performance of corrective actions.

The quantification of the risk related to water management alternative decisions in uncertain conditions, based on a Bayesian probabilistic method of decision taking, are shown in this paper using the example of the Orla river catchment in Poland. The presented examples of risk analysis relate to a hypothetical situation in which the European Commission would indict Poland in the Court of Justice for failing to fulfil obligations of a Member State due to not achieving good ecological status / potential and good chemical status of waters in its territory. Despite the prospect of new technological solutions in wastewater treatment [28] and possible changes in mechanisms for enforcing compliance with environmental protection regulations, the adopted Bayesian approach will help water managers to take decisions optimal in the sense of economic risk. It needs to be emphasized that never before have the penalties been analysed among arguments concerning water management decisions.

Study area

In order to present risk of water management decisions, a typical agricultural area in central Poland with medium sized settlements was chosen. Major types of pressures on surface waters in such areas are natural and artificial fertilizers from arable land and waste waters. The most significant threat to the water status is eutrophication.

The river Orla (95.1 km long) is the right tributary of the Barycz River located in the Middle Odra water region of Poland (Fig. 1). Its catchment area is approximately 1600 km^2 and consists of eleven water bodies. The largest part of the catchment area is arable land with intensive agriculture and cattle breading (in total about 65% of the catchment area [29]).

The water quality of this river is strongly influenced by the input of waste water from two relatively big wastewater treatment plants (WWTPs) in Krotoszyn and Rawicz and the broad-based pollution from agricultural and breading areas. The results of the water monitoring carried out by the Voivodship Inspectorate of Environmental Protection (12 times per year) in the period 2004–2018 indicate that the quality class of the Orla river in terms of total nitrogen as well as in total phosphorus was below good or good with about 20% probability of misclassification [30].

Since the ecological status of the majority of Polish wbs is below good due to eutrophication caused by pressures from agricultural areas and municipal sewage, two physiochemical indicators, i.e. total nitrogen (TN) and total phosphorus (TP) were adopted and assumed as sufficient to represent the reaction of water status to both types of pressures and to the considered remedial measures.

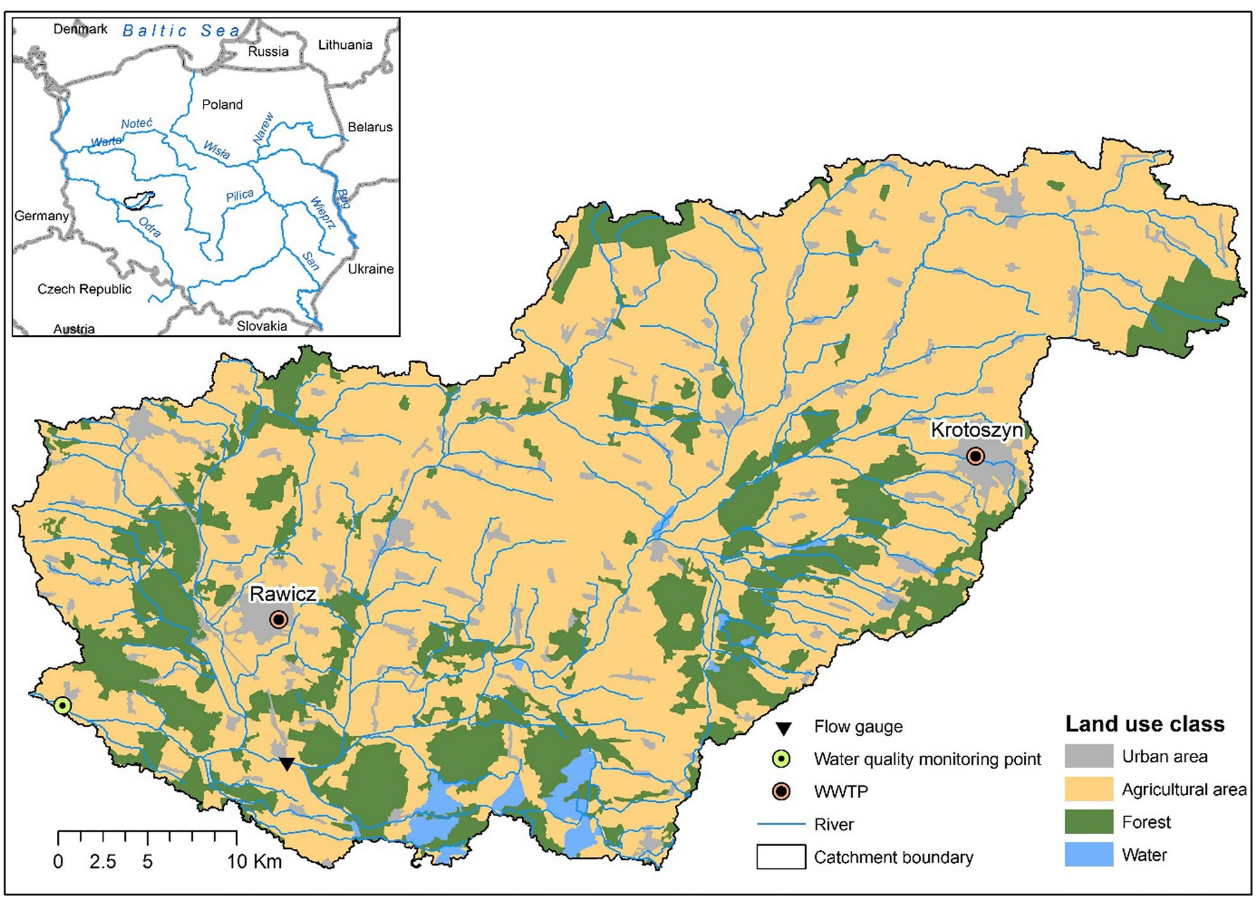

Fig. 1 A location and map of the Orla catchment

Water management decisions—programme of measures

During the period 2004–2018, sewage treatment plants in Rawicz and Krotoszyn discharged elevated concentrations of nitrogen and phosphorus compounds in treated wastewater. That is why the introduced programme of measures in the Orla catchment included the modernization of two WWTPs. In 2012, the Krotoszyn WWTP was modernized and the 2017 upgrading of Rawicz WWTP was completed. In this paper, however, modernization of both WWTPs are analysed as potential alternatives of management decisions based on previous characteristics of both wastewater effluents and water quality in the Orla river. Both WWTPs were based on activated sludge technology. Modernization of Krotoszyn WWTP consisted of upgrading the reactor, including the denitrification stage. In Rawicz WWTP a new, bigger reactor was added together with sand trap and secondary settling tank. The modernizations costs amounted to 2894 and 7764 k€, respectively, for Krotoszyn and Rawicz WWTPs.

Contrary to any decision concerning upgrading the existing WWTP or building a new one, which is taken at the level of local government administration and depends on getting funds (e.g. from EU), the decision about reduction of the cultivated area is an individual decision of a farmer and it is difficult to predict in the future. Therefore, the percentage of decrease in the cultivated area, regardless of its beneficial ecological effect, cannot be planned at the commune level. The incentive for a farmer to take such a decision is the possibility of getting subsidies for the fallowed area.

Several alternatives of the percentage of the fallowed harvest area were considered for the Orla river catchment. The decrease of the agriculturally used area by 1 to 25% was adopted as a potential management alternative. For costs of modernization of the WWTPs in Krotoszyn and Rawicz, only total costs were taken into consideration without considering operational costs. Since Bayesian decision tables give optimal, in a financial sense, decisions only when considering the long-term outlook, it was assumed that both costs of subsidies and potential penalties would be analysed for the 6-year period as presented in Table 1.

Legal mechanisms of enforcing environmental objectives set by the WFD—penalties for failure to achieve good status

The mechanism forcing the EU countries to achieve or maintain good status of water stems from the threat of penalties for failure to achieve the so-called environmental objectives defined by the WFD. The mechanism of imposing fines on a Member State that failed to comply with a judgment in an infringement case is provided for in the Maastricht Treaty (Article 228) [31].

The daily penalty payment for a Member State is determined by the formula:

$$\mathrm{Dp} = (\mathrm{Bfrap} \times \mathrm{Cs} \times \mathrm{Cd}) \times n, \quad (1)$$

where Dp = daily penalty payment; Bfrap = basic flat-rate penalty; Cs = coefficient for seriousness; Cd = coefficient for duration; and n = factor taking into account the capacity to pay of the Member State concerned.

Currently, the basic flat-rate is EUR 700, the duration factor can be a value from 1 to 3 (increasing by 0.10 per month from the date of the sentence) and the n-factor for Poland is 7.36 [32].

The method presented in this paper for calculating penalties for failure to achieve good status in part of the wbs, using the example of the Orla catchment area, is only illustrative. Estimation of penalties is attempted in order to include their values in the preparation of decision tables. For the purposes of the example, it is assumed that the duration of the infringement period started from 2015—that is, from the first time horizon specified in the

Table 1 Calculations of the penalty for not achieving environmental objectives in a WB and estimation of costs for subsidies for fallowing 8% of agricultural area of the Orla river catchment—both annual and 6-year penalty and costs

Penalties for not achieving environmental objectives in a WB						
Basic flat-rate penalty [€]	**n-factor**	**Coefficient for seriousness**	**Coefficient for duration**	**Daily penalty payment [€]**	**Annual penalty [k€]**	**6 year penalty [k€]**
700	7.36	1	1	51,152	1880.5	11,282.9
700	7.36	5	2	511,520	18,804.8	112,828.9
Costs of subsidies for fallowing 8% of agricultural area of the Orla river catchment						
Area [ha]	Percentage of agricultural areas %	Percentage of fallowing areas %	Area [ha]	Subsidy [€/ha]	Annual surcharge [k€]	Surcharge for 6 years [k€]
159,800	0.65	0.08	8309.6	73	606.6	3639.6

WFD, when all wbs in the EU should have achieved good status or good ecological potential.

The hypothetical annual and 6-year penalty is presented in Table 1, where the base flat rate Bfrap = € 700, the coefficient for seriousness factor Cs = 5, the coefficient for duration Cd = 2 and the n-factor for Poland of 7.36 were used for the calculations.

It may happen that failure to achieve good water status in some of the wbs in Poland causes the deterioration of the Baltic Sea status, which is a common good and for which there is a joint responsibility. This means that 'that a Member State that fails to fulfil its obligations violates the interests of other Member States' [33]. Then the coefficient of seriousness could be set at 10 (on a scale of 1 to 20), similar to the case Commission v. France case [33], and then the daily penalty would be € 511,520 (Table 1).

Methods

Since for both surveillance and operational monitoring, frequency of measurements of physiochemical quality elements is once a month, it is very difficult to estimate statistically sound and reliable probability distributions for TN and TP.

How to assess true status classes for both indicators could be also problematic. In order to overcome both difficulties, concentration time series for the SWAT-simulated models were applied.

SWAT model and simulated data

Soil and Water Assessment Tool (SWAT) is a continuous-time, process-based, semi-distributed hydrological model simulating the water flow, sediment, and nutrients on a catchment scale. The basic calculation unit—hydrologic response unit (HRU) is created by an overlay of land use, soil, and slope maps. Water balance and water quality components are calculated separately for each HRU, then aggregated at the sub-basin level and routed through the stream network to the main river outlet [34].

In this study, the existing SWAT model of the Barycz catchment was used [35]. While the full description of model setup, calibration and validation was presented in the latter study, here a brief overview is provided, important in the context of water quality aspects tackled in this study. Delineation of the catchment based on the 10-m resolution digital elevation model resulted in division of the catchment into 503 sub-basins. The land cover map was a combination of CORINE Land Cover (CLC) 2006 and post-processed Landsat 8 high-resolution images. The interpolated daily precipitation and air temperature (minimum and maximum) data (1951–2018) were acquired from a 5-km resolution gridded dataset [36]. Sensitivity analysis, calibration and validation were performed with the help of SWAT-Calibration Uncertainty Procedures (SWAT-CUP; [37]), using the Sequential Uncertainty Fitting Procedure Version 2 (SUFI-2 algorithm; [38]). A multi-site calibration was performed for discharge, total suspended sediment, nitrates, total nitrogen, phosphates and total phosphorus loads. The model was calibrated and validated. For SWAT calibration, monitoring data from the State Monitoring Program were used together with flow data collected by the State Meteorological Hydrological Institute.

The Kling–Gupta efficiency (KGE) was used as a goodness-of-fit measure [39] to calibrate and validate the SWAT model. KGE is a function of the correlation term (linear regression coefficient between the measured and simulated variable), the variability term (the ratio between simulated and measured standard deviation), and the bias term (the ratio between simulated and measured mean). Among the water quantity and quality calibration points, there was one flow gauge and one water quality monitoring point directly located near the outlet of the Orla river. The discharge KGE values for the Orla were 0.77 and 0.83 for calibration and validation, respectively. For water quality parameters, KGE values were also reported to be good: for total nitrogen (0.89—calibration and 0.87—validation) and total phosphorus (0.65—calibration and 0.81—validation) [35].

Historical data on the loads of polluting substances emitted by treatment plants into the river network in the Orla subcatchment in years 2004 -2018 were collected from operators, especially from the two WWTPs in Krotoszyn and Rawicz which were used as important point sources in the SWAT model. As the data concerning the quality of effluents from the WWTPs were in the form of monthly volumes and monthly mean concentrations, they were converted into daily values by imposing certain stochastic fluctuations of magnitude up to 30% for both flow and concentrations.

Time series of concentration of TP and TN in the Orla river at the cross section just before the inflow into the Barycz river are presented in Figs. 2 and 3, respectively. As the period after modernization of the Rawicz WWTP was relatively short, water management decisions were analysed only for two periods: before the modernization of the Krotoszyn WWTP in 2012 and the period after modernization until the beginning of 2017 when upgrading of the Rawicz WWTP started.

Bayesian approach

Bayesian decision theory is an important statistical method for quantification of the compromise between various decisions using probabilities and cost that accompany these decisions [40]. It allows for explicit

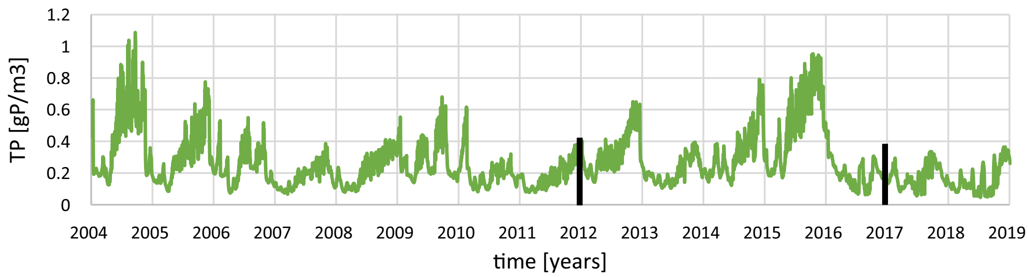

Fig. 2 Concentration of total phosphorus in Orla River in years 2004–2018—as simulated by SWAT. The times of modernization of Krotoszyn and Rawicz WWTPs are depicted with vertical lines

consideration of the cost of uncertainty and worth of data in the process of taking decisions. The Bayesian model of decision-making is based on the probability of a result when some prior information, here the probability of a stochastic state $\mathbf{S}_i$ of the water body, i.e. historical ecological status of a wb, as well as new evidence/observations—monitoring data, are taken into consideration.

Game theory [24] defines a 'game with Nature' as a game in which one of the partners—Nature—does not set specific goals for himself and his strategy does not take into account possible future actions of the opponent. It means that the stochastic states of the water environment ($\mathbf{S}_i$), i.e. Nature are characterized by a specific and unchanging probability distribution of occurrence until, as a result of human activity, sufficiently strong changes occur in the river or its catchment area. Bayesian decision theory allows for quantification of the compromise between various management decisions using probabilities and costs that accompany these decisions. Applying this theory, the problem of choosing the optimal decision from available remediation alternatives or paying fines as consequences of violating environmental objectives of water status can be solved. The result of this method in relation to corrective actions generates the so-called decision table assigned to a wb's catchment in which, for any possible value of selected indicator, the best alternative of corrective action is indicated.

The chosen water quality elements, i.e. TP and TN, representing the state of the aquatic environment (Nature) are characterized by a specific and invariable distribution of the probability of occurrence. In the adopted methodology, using the simulation model of the river dynamics, probability distributions of the state $\mathbf{S}_i$ (represented by TP and TN) can be estimated based on the frequency of occurrence of status classes in the series of values simulated in time before and after the implementation of remedial actions.

The nine-step algorithm presented in Fig. 4 served as a framework for elaborating decision tables according to Bayesian decision theory for both water quality elements, i.e. TP and TN.

Step 1: (description of steps of Bayesian algorithm presented in Fig. 4). There are three possible stochastic states $\mathbf{S}_i$, i.e. ecological status of wb ($\boldsymbol{w_i} = \mathbf{1,\ldots 3}$): high, good and below good.

Step 2: The annual mean values form the results of the simulated daily concentrations of TP and TN are assumed the 'true' states for the wbs, related separately

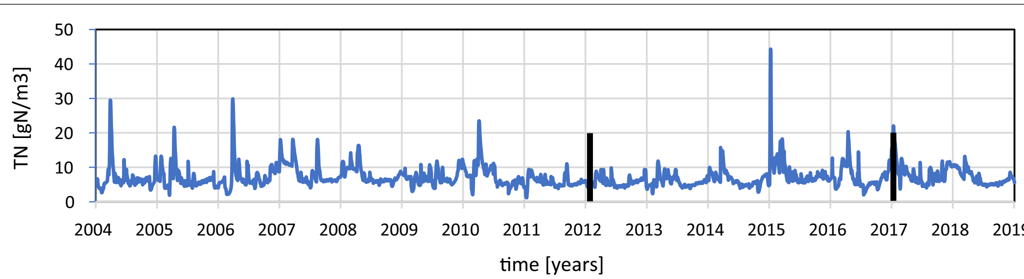

Fig. 3 Concentration of total nitrogen in years 2004–2018—as simulated by SWAT. The times of modernization of Krotoszyn and Rawicz WWTPs are depicted with vertical lines

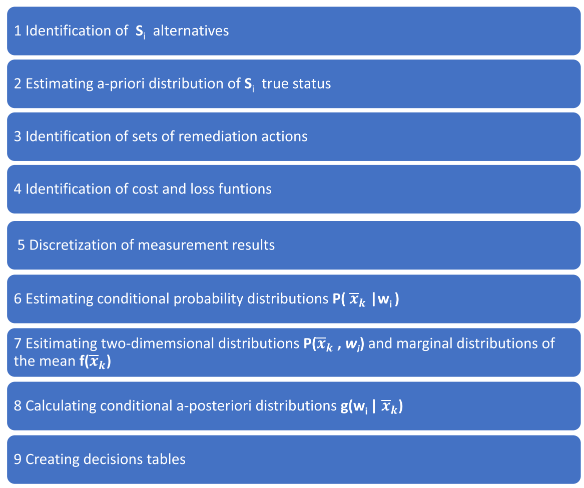

Fig. 4 Steps of the Bayesian algorithm of making decisions in uncertain conditions

for both status indicators. The a priori probability distribution of true states occurrence **g**($\boldsymbol{w}$) are/were estimated on the basis of SWAT's simulations of physicochemical elements in the river with a time step of 1 day for the 15-year period 2004–2018.

Step 3: Defining a set of alternative remediation actions $\mathbf{d_i}$, as modernization of WWTP or fallowing part of the area. A short discussion concerning remediation measures is presented in the section 'Water management decisions—programmes of measures'.

Step 4: Evaluations of costs **c** related to various remediation action and evaluation of potential penalties charged in case of bgs leads to formulation of 'pure costs/ losses function'.

$\mathbf{c = L(d_i,w)}$—deterministic cost/loss function defining the costs **c** borne by the decision-maker when making a decision $\mathbf{d_i}$ when the $\mathbf{S_i}$ is equal to $\mathbf{w_i}$.

Step 5: The whole range of concentrations, of water quality elements TP and TN observed as results of the SWAT model (virtual monitoring data), was split into separate intervals (bins). The middle value of concentration for each interval was assigned as the representative value for this interval. In accordance with the Classification Regulation [41], the intervals of the concentration values were assigned to one of the status classes: high, good and below good.

Step 6: Determination of the conditional probability distribution $\mathbf{P}(\overline{x}_k|\boldsymbol{w_i})$ that is estimation of the probability that in a particular $\mathbf{S_i}$ the mean value $\overline{x}_k$ of the indicator belongs to a certain range of concentration. The adopted method of determining conditional probability of the TP or TN concentration consisted of sets of 12 random selections out of 365 values of the concentration, simulated for each year. This procedure was chosen to be analogous to the monthly monitoring measurements carried out. From the drawn values of simulated concentrations, the annual average value ($\overline{\boldsymbol{x}}_{\boldsymbol{k}}$, $k = 1.2, \ldots$) was calculated and then used to determine the corresponding status class. Conditional probability distributions were based on 1000 sets of random selections. Every time the mean value of the twelve concentrations was assigned to one of three status classes, i.e. high, good or below good, split into narrow intervals within Step 5. The calculations were performed for the years selected to represent the appropriate $\mathbf{S_i}$.

Step 7: The two-dimensional probability distribution of random variables, i.e. status $\mathbf{S_i}$ and concentration of TN or TP, is calculated from the formula:

$$\mathbf{P}(\overline{x}_k, \boldsymbol{w_i}) = \mathbf{P}(\overline{x}_k|\boldsymbol{w_i}) * \mathrm{g}(\boldsymbol{w_i}), \quad (k = 1, \ldots; \ i = 1, 2, 3). \tag{2}$$

The marginal distribution of the mean $f(\overline{x}_k)$ is then determined by summing up the values of the probability distribution $\mathrm{P}(\overline{x}_k, w_i)$ over all $\mathbf{S_i}$.

Step 8: Determination of the conditional a posteriori distribution $\mathbf{g}(\boldsymbol{w_i}|\ \boldsymbol{x_k})$ using the Bayesian formula:

$$\boldsymbol{g}(\boldsymbol{w_i}|\overline{x}_k) = \frac{\boldsymbol{P}(\overline{x}_k|\boldsymbol{w_i})\boldsymbol{g}(\boldsymbol{w_i})}{\boldsymbol{f}(\overline{x}_k)} = \frac{\boldsymbol{P}(\overline{\boldsymbol{x}}_{\boldsymbol{k}}, \boldsymbol{w_i})}{\boldsymbol{f}(\overline{x}_k)}. \tag{3}$$

Step 9: Creating a decision table:

$$\begin{aligned} \overline{k}\left(\boldsymbol{d_j}; \overline{x}_k\right) &= L(\boldsymbol{d_j}, \boldsymbol{w}_1) \bullet g(\boldsymbol{w}_1|\overline{x}_k) \\ &\quad + L(\boldsymbol{d_j}, \boldsymbol{w}_2) \bullet g(\boldsymbol{w}_2|\overline{x}_k) \\ &\quad + L(\boldsymbol{d_j}, \boldsymbol{w}_3) \bullet g(\boldsymbol{w}_3|\overline{x}_k). \end{aligned} \tag{4}$$

The Bayesian decision table contains the results of estimating the average costs/losses that are incurred when the value of the annual mean of the indicator (based on measurements) is ($\overline{\boldsymbol{x}}_{\boldsymbol{k}}$, $k = 1.2, \ldots$) and the decision-maker makes one of two or more decisions $\boldsymbol{d_j}$, ($j = 1, 2, \ldots$). For the mean value, average costs are calculated from the matrix of pure costs L (d_j, w) separately for each of the possible decisions $\mathrm{d_j}$, according to formula (3). Thus, for a fixed $\overline{\boldsymbol{x}}_{\boldsymbol{k}}$ the optimal decision is the one for which the average costs are the lowest. For each of the possible average values of the indicator values measured during the year $\overline{\boldsymbol{x}}_{\boldsymbol{k}}$, ($k = 1, 2, \ldots$), the decision table contains the values of average losses corresponding to decisions $\boldsymbol{d_j}$ ($j = 1, 2 \ldots$).

Each 'virtual' measurement of a physicochemical element performed at a certain moment in time can be determined with certainty that it is performed under the conditions of occurrence of a certain $\mathbf{S_i}$. This last observation has important consequences in risk analysis and decision-making under conditions of uncertainty of measurements and $\mathbf{S_i}$.

It should be stressed that, in order to estimate a priori probabilities of states $\mathbf{S_i}$, the monitoring historical time series could have been used instead of simulated concentrations by SWAT. However, this would increase the uncertainty of the final decision tables since, based on monitoring data of TN, the calculated probabilities of misclassification of the status class were high—up to 0.8 in 2004 and 0.5 in 2002, 2005, 2006, 2007 and 2013 [30].

Analysis of water management decisions based on concentration of total phosphorus (TP)

On the basis of SWAT simulations of TP concentration presented in Fig. 2, for the period before 2012, a priori probabilities of each natural state were estimated as: high 0.25, good 0.375 and also 0.375 for below good status.

For the years 2004–2012, two variants of alternative decisions were analysed for the Orla River concerning decrease of pollution by phosphorus. Variant A) consists of altogether four alternative decisions presented in Table 3. Regarding the application of remediation measures in the form of modernizing only one WWTP (either Krotoszyn—decision d2 or Rawicz—d3) there was NO penalty included in the 'function of clean costs/losses' when bgs was assessed. Such an approach was justified on the basis that, although it may be insufficient, still some remediation action was undertaken. In case of the 'no action' scenario (code d1) the penalty value (BIG penalty) was based on the highest values of coefficients (WW = 2, WT = 5).

Variant B) consists of the same set of alternatives as variant A but is more restrictive: when only one WWTP is modernized (either Krotoszyn or Rawicz) there will be a penalty in bgs assessed but with the lowest values of both coefficients WW = 1, WT = 1). When both WWTPs are extended but the good status is not reached, no penalty will be imposed.

For the period after the modernization of Krotoszyn WWTP, a priori probabilities of each natural state were estimated as: high 0.2, good 0.2 and also 0.6 for below good status.

Two variants of alternatives sets of decisions were tested for the period after 2012. In both of them, the function of pure cost/losses for 'no action' included a penalty for bgs. In variant C1, failure to reach good status after implementing the modernization of Rawicz WWTP did not incur a penalty, as it would be difficult to introduce any other remediation measure. However, in variant C2 (modification of C1) in the same situation, the decision to modernize the WWTP did incur a penalty in the case of bgs .

The last variant attempted (Variant D), included additionally to variant C2 an alternative decision (d2) consisting of modernization of several small WWTPs located in this catchment (in small communities). The total cost of this would be similar to modernizing Krotoszyn WWTP. The last alternative decision (d4) consists of investments in those small WWTPs together with modernization of Rawicz WWTP. In a bgs situation, a penalty 10 times higher the 'standard' was included in the d1 costs function and smaller penalty d2 and d3 cost functions.

Analysis of water management decisions based on concentrations of total nitrogen (TN)

Similarly to the analysis presented in the previous section, analyses related to nitrogen were conducted separately for the period before modernization of Krotoszyn WWTP, i.e. before 2012 and after that year. The water quality in the river in the monitored cross section did not improve after 2012 [42]. There was also no significant decrease in modelled nitrogen concentration in this period (Fig. 3). In the calculations of conditional probability (6th step in Fig. 4), updated modelled concentrations of TN were included but there was no need to change values of a priori probabilities of $\mathbf{S_i}$ and for both periods a priori probabilities of high and good status were estimated as very low and equal to 0.05. When analysing histograms representing conditional probability distributions (6th step), it was observed that they were considerably different for both periods, so specific and different distributions for both periods were applied.

Concerning the alternative decisions focused on decrease of TN concentration, basic two variants were tested. Within each variant, various surfaces being fallowed were considered. Alternative decisions d2 and d4 (Table 4) assumed that, when modernization of Krotoszyn or Rawicz WWTP was envisaged, there would not be any penalties in below good status. For the decision d3 (Table 4), when protective measures were reduced only to purely accidental decisions of farmers to fallow fraction of their arable lands, the function of clean costs/losses included the penalty for the bgs case. This can be understood as an attempt to enforce a more permanent solution to the problem.

Results

The Bayesian approach to decision-making and resultant decision tables gives the optimal decision but only in the long-term perspective. That is why the analysis and calculations presented here were performed for a period of one River Management Plan.

The whole set of alternative decisions focused on decrease of TP concentration for the period before 2012,

Table 2 Alternative water management decisions for the Orla watershed for the period before 2012 (variants A and B) and for periods after 2012 (variants C1, C2 and D)

Alternative decisions before 2012		Alternative decisions after 2012			
Variant A	**Variant B**	**Variant C1**	**Variant C2**	**Variant D**	**Decision code**
No action (penalty in bgs)	No action (BIG penalty in bgs)	No action (penalty in bgs)	No action (penalty in bgs)	No action (BIG penalty in bgs)	d1
Modernization of the Krotoszyn WWTP (NO penalty in bgs)	Modernization of the Krotoszyn WWTP (penalty in bgs)			Modernization of several small WWTPs (penalty in bgs)	**d2**
Modernization of the Rawicz WWTP (NO penalty in bgs)	Modernization of the Rawicz WWTP (penalty in bgs)	Modernization of the Rawicz WWTP (NO penalty in bgs)	Modernization of the Rawicz WWTP (penalty in bgs)	Modernization of the Rawicz WWTP (penalty in bgs)	**d3**
Modernization of the Krotoszyn and Rawicz WWTPs (NO penalty in bgs)	Modernization of the Krotoszyn and Rawicz WWTPs (NO penalty in bgs)			Modernization of several small WWTPs and Rawicz WWTP (NO penalty in bgs)	**d4**

Table 3 Bayesian decision table for variants of alternatives presented in Table 3 based on total phosphorus concentrations

Middle value of the interval of TP concentration [mg P/dm^3]	Before year 2012 Optimal decision for variants		After year 2012 Optimal decision for variants		
	variant A	variant B	variant C1	variant C2	variant D
0.19	d1	d1	d1	d1	d1
0.21	d1	d1	d1	d1	d1
0.23	d1	d1	d1	d1	d1
0.25	d1	**d2**	**d3**	d1	**d4**
0.29	d1	**d4**	**d3**	**d3**	**d4**
0.31	d1	**d4**	**d3**	**d3**	**d4**
0.33	d1	**d4**	**d3**	**d3**	**d4**
0.35	d1	**d4**	**d3**	**d3**	**d4**
0.37	**d2**	**d4**	**d3**	**d3**	**d4**
0.39	**d2**	**d4**	**d3**	**d3**	**d4**
0.41	**d2**	**d4**	**d3**	**d3**	**d4**
0.43	**d2**	**d4**	**d3**	**d3**	**d4**
0.45	**d2**	**d4**	**d3**	**d3**	**d2**

Values of TP within the high status are marked in blue, in good status in green and in bgs in yellow (in line with the WFD colour code)

i.e. Variants A and B is presented on the left side of Table 2.

For the period after the modernization of Krotoszyn WWTP and when applying characteristics of sewage in the SWAT model as reported by the WWTP operator, it was necessary to change the probabilities of the natural states. The probabilities of high and bgs were increased slightly at the expense of the probability of good status. Two variants of alternatives sets of decisions were tested. The alternatives for period after 2012 are presented on the right side of Table 2.

Decision tables for all the above variants of analysed management alternatives are presented in Table 3. Colours in the left column for TP concentration are in line with the colour code expected by the WFD, i.e. blue for high status, green for good and yellow for below good status. Threshold values applied for classes were adopted as for abiotic-type waterbodies—a lowland sandy loam river [36].

Depending on the particular set of alternative decisions, different actions become optimal for different ranges for TP concentrations. Since the function of pure

Table 4 Alternative decisions on corrective measures within part of variant I (before 2012) and variant II (after 2012)

variant I (before 2012)	Decision code	variant II (after 2012)	Decision code
No action (penalty in *bgs*)	d1	No action (penalty in bgs)	d1
Modernization of Krotoszyn WWTP (no penalty in *bgs*)	**d2**	Modernization of Rawicz WWTP (no penalty in *bgs*)	**d2**
Fallowing (penalties in *bgs*)	**d3**	Fallowing (penalties in *bgs*)	**d3**
Modernization of Krotoszyn WWTP and fallowing (no penalty in *bgs*)	**d4**	Modernization of Rawicz WWTP and fallowing (no penalty in *bgs*)	**d4**

costs/ losses does include penalty in case of bgs if the probability of such status is very low, the optimal decision may be not the one precautionary for the environment but the most financially reasoned. This is exemplified by variant A where 'no action' decision d1 is the optimal one, which means paying penalties in bgs as long as TP < 0.36 mg P/dm^3. For variants B, C1 and D d1 is the optimal alternative only for TP < 0.24 mg P/dm^3, i.e. the middle of the interval for good status. Concerning decision d2, in variant B it appears optimal only for one interval, i.e. for 0.24 mg P/dm^3 < TP < 0.27 mg P/dm^3, whereas in variant A, its optimality is shown when TP ≥ 0.36 mg P/dm^3.

There is only a minor difference between C1 and C2 subvariants as optimality of decision d3, i.e. modernization of the Rawicz WWTP burdened by a penalty in bgs becomes the optimal one for one interval earlier (in comparison to C2), i.e. for TP > 0.24 mg P/dm^3.

Slight differences in results appeared depending on the year for which simulated concentrations were used for conditional probability estimation (step 6 Fig. 4). Taking as an example Variant A, decision d2 could be optimal either for slightly lower or higher concentrations of phosphorus.

As the costs of modernization of both WWTP are crucial within set of alternative decisions, the resultant decision tables were tested against differences/fluctuations in these costs values. Assuming a 20% decrease of either or both WWTP modernization costs or a 20% increase of either or both investments in WWTP, the resultant decision tables stayed unchanged.

All the calculations presented here were performed for a period of six years of one River Management Plan. They were performed repeatedly based on simulated concentrations for different years, in order to give an overview of many possible real situations. As it could have been expected, the resultant decision tables were not invariable for the distributions of concentrations sampled from simulated time series for different years, especially for those assessed as representing below good status based on annual mean value. It was decided that the comparison of alternative decision was performed eventually for the same, the most representative among all, set of a'priori distributions, i.e. for high, good and below good status.

Concerning the alternative decisions focused on decrease of TN concentration, the suggested basic two variants are presented in Table 4. Within each variant, various surfaces being fallowed were considered.

Decision tables for both periods are presented in Table 5. It can be seen that, in the period before 2012, the 'no action' d1 decision is the optimal one as long as the mean value of TN < 4.6 mg/dm^3 (so the mean value belongs to the interval 4.2–4.6 mg/dm^3 with the middle value 4.4 mg/dm^3) and does not depend on the percentage of area fallowed. For TN > 4.6 mg/dm^3, decision d2 of a small area being fallowed is the best one. Since the decision to fallow a larger area is linked with increasing costs of subsidies, and for the bgs case is burdened by penalties, that is why the decision d3 on modernization of Krotoszyn WWTP becomes more preferred together with the envisaged increased TN concentration. In variant I, it has been assumed that the d3 alternative is not linked to any penalties even in the bgs case. For quite an extreme value of 25% of area fallowed, decision d3 is no longer optimal for any concentration of TN.

For the period after 2012, the 'no action' decision—d1 is the optimal one for much higher concentrations of TN than before. It means that taking the risk of fines pays off even in bgs as long as the mean value TN ≤ 5.2 mg/dm^3 (Table 5). For higher concentrations of TN, the decision to fallow 1–2% of the area, despite the necessity of penalty payments (d3), appears to be the best one.

Table 5 Bayesian decision table for variant I—before 2012 and variant II—period after 2012*

Middle value of interval of TN concentration [mg N/dm^3]	variant I -before 2012 Optimal decisions for depending on the fallowed area [%]					variant II - period after 2012 Optimal decisions for depending on the fallowed area [%]				
	1	2	5-15	20	25	1-2	3	4-6	7-15	20-25
2	d1	d1	d1	d1	d1	d1	d1	d1	d1	d1
2.4	d1	d1	d1	d1	d1	d1	d1	d1	d1	d1
2.8	d1	d1	d1	d1	d1	d1	d1	d1	d1	d1
3.2	d1	d1	d1	d1	d1	d1	d1	d1	d1	d1
3.6	d1	d1	d1	d1	d1	d1	d1	d1	d1	d1
4	d1	d1	d1	d1	d1	d1	d1	d1	d1	d1
4.4	d1	d1	d1	d1	d1	d1	d1	d1	d1	d1
4.8	**d3**	**d3**	**d3**	**d3**	**d2**	d1	d1	d1	d1	d1
5.2	**d3**	**d3**	**d3**	**d2**	**d2**	d1	d1	d1	d1	d1
5.6	**d3**	**d3**	**d3**	**d2**	**d2**	**d3**	**d3**	**d3**	**d3**	**d2**
6	**d3**	**d3**	**d2**	**d2**	**d2**	**d3**	**d3**	**d3**	**d2**	**d2**
6.4	**d3**	**d2**	**d2**	**d2**	**d2**	**d3**	**d3**	**d2**	**d2**	**d2**
6.8	**d3**	**d2**	**d2**	**d2**	**d2**	**d3**	**d2**	**d2**	**d2**	**d2**

Colours of left column and are in line with colour code for classes of ecological status acc. WFD. Blue—high, green—good, yellow—bgs. Threshold values for classes for total nitrogen are applied as for abiotic-type waterbodies, a lowland sandy loam river

Similarly to the decision table for the first period, together with the increase of the area fallowed and the increase of costs related to subsidies, decision on modernization of a WWTP—this time in Rawicz (d2)—becomes the optimal option. Since taking the decision of fallowing for so narrow a range of concentration as 5.4–5.8 mgN/dm^3, as in the case of 7–15% of area fallowed, seem inappropriate (it would be very difficult to forecast such a narrow range of concentration with high certainty), the decision d3 on investing in modernization of WWTP in Rawicz should be taken as soon as TN approaches 5.4 mgN/dm^3.

According to the resultant decision tables presented in Table 4 for successive 1% increases, the fallowed area can be understood also as a sensitivity analysis of the optimal variants.

No analyses have been performed for fallowed areas > 25% as it can be concluded that this alternative is not optimal for any TN concentration expected.

Discussion

Many papers describing applications of Bayesian decision analysis can be clustered in three categories: decision trees [43], influence diagrams [44–46] and belief network [47–50]. Yet, only a limited amount of real case examples in hydrology and water management have been based on Bayesian decision theory.

The Bayesian decision-making algorithm applied in this paper is fundamentally similar to the one presented by Davis [50] for the problem of levee construction. In comparison to Davis, who developed the stochastic properties of the values of the state variables as continuous probability density functions, in this paper steps 5–7 as shown in Fig. 4 were based on simplified discrete calculations. Due to the limited possibility of estimating costs associated with getting more information concerning water quality, i.e. through more intensive monitoring programme instead of simulations, the assessment of the resultant decision based on an improved but more expensive a priori distribution of the Nature status, was not performed in this study. In a later publication by Davis [51], observations were made that the possibility of getting additional information, i.e. more frequent hydrological measurements, allowed for improved design and levee construction which tended to reduce the risk of flooding. They concluded that, despite the problems that were intertwined, in practice they were separated. Despite 50 years passing since that publication, their observations are still valid, as the monitoring programmes in terms of both their scope as well as frequency of measurements do not result from the water management needs but rather from financial limitations of the monitoring system which is run in isolation from the needs of management practitioners.

Another application of Bayesian decision theory in water management was presented by Grosser [52]. For the purpose of finding optimal frequency of groundwater sampling, they used various continuous probability distribution functions and discussed their applicability. When comparing his approach to the one applied in this paper there was no need to fit any sample likelihood function as it resulted from direct random selection (as if sampling) from simulated SWAT results.

Although the simulated concentrations of physicochemical elements can be called 'virtual reality' and randomly selected values of these concentrations as 'virtual measurements', the adopted method of analysis refers to the real object and real measurements. The use of a calibrated model (SWAT) makes it possible to analyse the true status of a wb in relation to the considered physicochemical element without incurring the costs of building and maintaining a network of stations that would measure indicators in the river continuously. Also, the duration and frequency of individual natural states can be easily estimated as long as the simulated sequences of concentration values are sufficiently long. Dealing in such a controlled environment it is important to remember that any river model, even one calibrated very carefully, is only a simplified description of flowing reality and its results are biased by uncertainty. In general, numerous sources of uncertainties can be listed in water quality modelling applications. Specifically, they might be associated with different model components, including their structure, input data quality and model parametrization [53, 54]. Among the input data, precipitation aggravated due to non-stationary conditions was reported to be the biggest source of uncertainty [55]. To reduce the level of uncertainty and increase the accuracy of model outputs in this study, a spatially interpolated climate dataset was used, because of an improvement in the goodness-of-fit measures for discharge simulation pointed out in a recent study in Poland [56]. In order to evaluate potential uncertainties in hydrological model performance due to model structure and parameter values, several uncertainty analysis methods have been developed. The SUFI-2 algorithm used in the study of Marcinkowski [57] accounts for uncertainties in both model structure and parameters [55].

One of limitations of the Bayesian approach is the necessity of including the subjective choice of the prior distribution in the analysis. Even the distribution is based on some previous research so the selection of the research is subjective. However, if analysis concerning optimal decisions of remediation actions are performed based on data for a particular river catchment but from various periods of time and hence can be averaged, to some extent, more robust and suitable decisions can be reached.

In this study we did not focus on the feedback mechanism between the monitoring programme, especially the appropriate frequency of data collection which could assist water managers in issuing lower-risk decisions at the expense of additional monitoring cost. Unfortunately such a mechanism is not provided for under the Water Framework Directive. The agency responsible for the State Monitoring Programme is expected to perform monitoring, followed by issuing a status assessment report which is then handed over to the water administration responsible for water management decision.

The presented example assumes application of a penalty in the event that the subject of the case before the Court of Justice would fail to achieve the environmental objectives for the Orla river catchment only, while the remaining waters in Poland would meet the criteria of good status or good potential. The full picture of Polish river water quality reveals that about 75% of river water bodies and reservoirs are in a bad status [15]. The imposition of a penalty for Poland could proceed in a manner analogous to that used in the case of the European Commission v. the Kingdom of Spain [58] for failure to implement the Bathing Directive [59]. In that case, higher values of the infringement severity and time coefficients were adopted so the total penalty amounted to about € 1.3 million per year; but, due to the fact that only about 20% of bathing waters did not meet the requirements of the Bathing Directive, the total penalty was decreased proportionally.

Applying similar reasoning in the hypothetical case of the European Commission against Poland, the annual penalty for 75% of waters not meeting the environmental target would then be around € 12 million per year. Penalties set in this way could not be an adequate instrument to ensure the implementation of corrective measures such as modernization of WWTP which are a thousand times more expensive than the calculated penalties for an individual wb or even for a catchment. For now, the European Union uses more incentives than penalties in relation to many EU countries, including Poland, through a system of subsidies for the construction of sewage treatment plants, distributed by the National Fund for Environmental Protection and Water Management.

In the examples analysed in this article, the ecological status assessment was estimated using values of only one indicator—total nitrogen or total phosphorus. The idea of including more than one water quality element when considering optimal decisions on remediation actions is very tempting.

However, in general, the catchment area may experience various pressures. For example, in one sub catchment, significant municipal or industrial pressures may be observed while, in another, pressures resulting from agricultural activities may dominate. Then, the assessment of the ecological status would require indicators representing both nutrients and indicators representing organic pollution. In such a case, possibly a sequence of decision tables should be prepared based on decisions for all important pressures, starting with the indicator most sensitive to corrective actions for the entire analysed area.

Conclusions

This paper has presented a risk analysis for making decisions on remedial actions when uncertainty exists about the water status assessment. It can be considered as a prototype of a general methodology prepared for implementation in water protection. Here, simulated data for the Orla river catchment as an example have been used but**,** in wider application, a priori probability distribution can be derived based on time series of historical monitoring data. To the best of our knowledge, the legal mechanisms including penalties for failure to achieve good status have not been analysed among arguments concerning water management decisions to date.

The presented problem and suggested method of overcoming it leads to a more general, but as yet theoretical and unanswered, question:

- how to design future water monitoring programmes to support decision-making and decrease the risk,
- what level of uncertainty in the assessment of the status of a water body can be accepted by the water administration?

The water monitoring programme is understood in this question not so much as the list of the monitored indicators, but above all, the decision related to the risk of incurring high costs. This, in turn, influences the decisions on the spatial density of measurement points and the frequency of measurements of water quality indicators. The above problem can also be formulated in an equivalent way in the form of the question. This question is:

- 'what is the acceptable risk of incurring costs related to the implementation of unnecessary remedial measures or paying any penalties due to failure in achieving good water status as a consequence of poorly chosen corrective measures?'

Abbreviations

DO: Dissolved oxygen; Bgs: Below good status; EU: European Union; HRU: Hydrologic response unit; SWAT: Soil and Water Assessment Tool; TN: Total nitrogen; TP: Total phosphorus; WFD: Water Framework Directive; Wb: Water body; WWTP: Waste water treatments plant.

Acknowledgements

Thanks are due to Sian Howard who proofread the manuscript.

Author contributions

ML created the whole concept of the study and wrote the first draft, MP consulted creation of modelling setup for SWAT simulation and assisted in writing the text, PM performed simulations in SWAT. All authors reviewed the manuscript before submission. All authors read and approved the final manuscript.

Funding

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

Availability of data and materials

All monitoring data used in this study are available on request via www.gios.gov.pl. The datasets produced as simulated time series and analysed during the current study are available from the corresponding author on reasonable request.

Declarations

Ethics approval and consent to participate

Not applicable.

Consent for publication

Not applicable.

Competing interests

The authors declare that they have no competing interests.

Author details

[1]Faculty of Building Services, Hydro and Environmental Engineering, Warsaw University of Technology, Nowowiejska 20, 00-653 Warsaw, Poland. [2]Department of Hydrology, Meteorology and Water Management, Warsaw University of Life Sciences, Nowoursynowska 166 St. 02-787, Warsaw, Poland.

Received: 30 December 2021 Accepted: 18 April 2022
Published online: 13 May 2022

References

1. https://ec.europa.eu/environment/water/water-framework/index_en.html.
2. Staniszewski R, Szoszkiewicz K, Zbierska J, Lesny J, Jusik S, Clarke RT (2006) Assessment of sources of uncertainty in macrophyte surveys and the consequences for river classification. Hydrobiologia 566(1):235–246
3. Kelly M, Bennion H, Burgess A, Ellis J, Juggins S, Guthrie R et al (2009) Uncertainty in ecological status assessments of lakes and rivers using diatoms. Hydrobiologia 633(1):5–15
4. Thackeray SJ, Noges P, Dunbar MJ, Dudley BJ, Skjelbred B, Morabito G et al (2013) Quantifying uncertainties in biologically-based water quality assessment: a pan-European analysis of lake phytoplankton community metrics. Ecol Ind 29:34–47
5. Pasztaleniec A, Hutorowicz A (2012) Phytoplankton metrics response to the increasing phosphorus and nitrogen gradient in shallow lakes. J Elem 17(2):289–303
6. Hutorowicz A, Pasztaleniec A (2021) Uncertainty in phytoplankton-based lake ecological status classification: Implications of sampling frequency and metric simplification. Ecol Ind 127:107754

7. Clarke RT, Lorenz A, Sandin L, Schmidt-Kloiber A, Strackbein J, Kneebone NT, Haase P. (2006). Effects of sampling and sub-sampling variation using the STAR-AQEM sampling protocol on the precision of macroinvertebrate metrics. In The ecological status of European Rivers: evaluation and intercalibration of assessment methods. Springer, Dordrecht, pp. 441–459.
8. Clarke RT (2013) Estimating confidence of European WFD ecological status class and WISER Bioassessment Uncertainty Guidance Software (WISERBUGS). Hydrobiologia 704(1):39–56
9. CIS (2018) Best Practice for establishing nutrient concentrations to support good ecological status. Guidance document WG ECOSTAT.
10. Hering D, Borja A, Carstensen J, Carvalho L, Elliott M, Feld CK et al (2010) The European Water Framework Directive at the age of 10: a critical review of the achievements with recommendations for the future. Sci Total Environ 408(19):4007–4019
11. Birk S, Bonne W, Borja A, Brucet S, Courrat A, Poikane S et al (2012) Three hundred ways to assess Europe's surface waters: an almost complete overview of biological methods to implement the Water Framework Directive. Ecol Ind 18:31–41
12. Poikane S, Kelly M, Cantonati M (2016) Benthic algal assessment of ecological status in European lakes and rivers: challenges and opportunities. Sci Total Environ 568:603–613
13. Munné A, Ginebreda A, Prat N (2015) Water status assessment in the Catalan River basin district: experience gathered after 15 years with the Water Framework Directive (WFD). In Experiences from surface water quality monitoring, pp 1–35, Springer, Cham
14. Zacharias I, Liakou P, Biliani I (2020) A Review of the status of surface European waters twenty years after WFD introduction. Environ Process. https://doi.org/10.1007/s40710-020-00458-z
15. Kristensen P, Whalley C, Zal FNN, Christiansen T (2018) European waters assessment of status and pressures 2018. EEA Report (7/2018)
16. Maia R (2017) The WFD implementation in the European member states. Water Resour Manag 31(10):3043–3060
17. Loga M, Wierzchołowska-Dziedzic A, Martyszunis A (2018) The problem of water body status misclassification—a hierarchical approach. Environ Monit Assess 190(5):1–16
18. Loga M, Przeździecki K (2021) Uncertainty of chemical status in surface waters. Sci Rep 11(1):1–12
19. Loga M (2016) Estimating confidence and precision—basic measures of uncertainty in the assessment of the state of surface waters. Ochr Srodowiska 38(1) (**in Polish**)
20. CIS (2005) Overall Approach to the Classification of Ecological Status and Ecological Potential. Common Implementation Strategy for The Water Framework Directive (2000/60/Ec). Guidance Document No 13
21. Peterson JT, Evans JW (2003) Quantitative decision analysis for sport fisheries management. Fisheries 28(1):10–21
22. Peterson JT, Pease JE, Whitman L, White J, Stratton-Garvin L, Rounds S, Wallick R (2022) Integrated tools for identifying optimal flow regimes and evaluating alternative minimum flows for recovering at-risk salmonids in a highly managed system. River Res Appl 38:293–308
23. Sambito M, Freni G (2021) Strategies for improving optimal positioning of quality sensors in urban drainage systems for non-conservative contaminants. Water 13(7):934
24. Luce RD, Raiffa H (1989) Games and decisions: introduction and critical survey. Courier Corporation.
25. Cramer M, Koegst T, Traenckner J (2018) Multi-criterial evaluation of P-removal optimization in rural wastewater treatment plants for a subcatchment of the Baltic Sea. Ambio 47(1):93–102
26. Rode M, Wade AJ, Cohen MJ, Hensley RT, Bowes MJ, Kirchner JW et al (2016) Sensors in the stream: the high-frequency wave of the present. Environ Sci Technol. https://doi.org/10.1021/acs.est.6b02155
27. Arnold JG, Srinivasan R, Muttiah RS, Williams JR (1998) Large area hydrologic modeling and assessment part I: model development 1. JAWRA J Am Water Resour Assoc 34(1):73–89
28. Armah EK, Chetty M, Adedeji JA, Kukwa DT, Mutsvene B, Shabangu KP, Bakare BF (2021) Emerging trends in wastewater treatment technologies: the current perspective. Promis Tech Wastewater Treat Water Qual Assess 1:71
29. https://wroclaw.rzgw.gov.pl/files_mce/Region%20wodny/Planowanie/RDW/Barycz%20Widawa/zadanie_ii_cz_tekstowa.pdf
30. Loga M (2019) Uncertainty in the assessment of the ecological status of surface waters. Scientific Works of the Warsaw University of Technology. Environ Eng 80:4–183 (**in Polish**)
31. SEC (2005) 1658 COMMUNICATION FROM THE COMMISSION application of Article 228 of the EC Treaty https://ec.europa.eu/atwork/applying-eu-law/docs/sec_2005_1658_en.pdf
32. Brussels, 18.9.2018 C (2018) 5851 final communication from the commission Updating of data used to calculate lump sum and penalty payments to be proposed by the Commission to the Court of Justice in infringement proceedings https://ec.europa.eu/atwork/applying-eu-law/docs/c_2018_5851_en.pdf
33. JUDGMENT OF 12. 7. 2005—CASE C-304/02 JUDGMENT OF THE COURT (Grand Chamber) 12 July 2005 * Maastricht Treaty
34. Neitsch SL, Arnold JG, Kiniry JR, Williams JR (2011) Soil and water assessment tool theoretical documentation version 2009. Texas Water Resources Institute
35. Marcinkowski P, Piniewski M, Kardel I, Srinivasan R, Okruszko T (2016) Challenges in modelling of water quantity and quality in two contrasting meso-scale catchments in Poland. J Water Land Dev 31:97–111
36. Berezowski T, Szcześniak M, Kardel I, Michałowski R, Okruszko T, Mezghani A, Piniewski M (2016) CPLFD-GDPT5: High-resolution gridded daily precipitation and temperature data set for two largest Polish river basins. Earth Syst Sci Data 8:127–139
37. Abbaspour K (2008) SWAT-CUP2: SWAT calibration and uncertainty programs—a user manual
38. Abbaspour K, Johnson CA, van Genuchten MT (2004) Estimating uncertain flow and transport parameters using a sequential uncertainty fitting procedure. Vadose Zone J 3:1340–1352
39. Gupta HV, Kling H, Yilmaz KK, Martinez GF (2009) Decomposition of the mean squared error and nse performance criteria: implications for improving hydrological modelling. J Hydrol 377:80–91
40. Berger JO (2013) Statistical decision theory and Bayesian analysis. Springer Science & Business Media, Cham
41. Laws J (2019) item. 2149. The Act onthe classification of ecological status, ecological potential and chemical status as well as the method of classifying the status of surface water bodies **(in Polish)**
42. https://www.gios.gov.pl/pl/stan-srodowiska/monitoring-wod
43. Varis O, Kuikka S (1999) Learning Bayesian decision analysis by doing: lessons from environmental and natural resources management. Ecol Model 119(2–3):177–195
44. Varis O (1997) Bayesian decision analysis for environmental and resource management. Environ Model Softw 12(2–3):177–185
45. Kuikka S, Hildén M, Gislason H, Hansson S, Sparholt H, Varis O (1999) Modeling environmentally driven uncertainties in Baltic cod (Gadus morhua) management by Bayesian influence diagrams. Can J Fish Aquat Sci 56(4):629–641
46. Hardy T, Wu W (2021) Impact of different restoration methods on coastal wetland loss in Louisiana: Bayesian analysis. Environ Monit Assess 193:1. https://doi.org/10.1007/s10661-020-08746-9
47. Sušnik J, Molina JL, Vamvakeridou-Lyroudia LS, Savić DA, Kapelan Z (2013) Comparative analysis of system dynamics and object-oriented bayesian networks modelling for water systems management. Water Resour Manag 27(3):819–841
48. Santillán D, Mediero L, Garrote L (2014) Modelling uncertainty of flood quantile estimations at ungauged sites by Bayesian networks. J Hydroinf 16(4):822–838
49. Pagano A, Pluchinotta I, Giordano R, Petrangeli AB, Fratino U, Vurro M (2018) Dealing with uncertainty in decision-making for drinking water supply systems exposed to extreme events. Water Resour Manag 32(6):2131–2145
50. Davis DR, Kisiel CC, Duckstein L (1972) Bayesian decision theory applied to design in hydrology. Water Resour Res 8(1):33–41
51. Davis DR, Duckstein L, Fogel MM (1976) Uncertainty in the return period of maximum hydrologic events: a Bayesian approach. J Hydrol 31(1–2):81–95
52. Grosser PW, Goodman AS (1985) Determination of groundwater sampling frequencies through Bayesian decision theory. Civ Eng Syst 2(4):186–194
53. Abbaspour KC, Rouholahnejad E, Vaghefi S, Srinivasan R, Yang H, Kløve B (2015) A continental-scale hydrology and water quality model for Europe: calibration and uncertainty of a high-resolution large-scale SWAT model. J Hydrol 524:733–752

54. Beven KJ (2006) A manifesto for the equifinality thesis. J Hydrol 320:18–36
55. Guzman JA, Shirmohammadi A, Sadeghi AM, Wang X, Chu ML, Jha MK, Parajuli PB, Harmel RD, Khare YP, Hernandez JE (2015) Uncertainty considerations in calibration and validation of hydrologic and water quality models. Trans ASABE 58(6):1745–1762
56. Szcześniak M, Piniewski M (2015) Improvement of hydrological simulations by applying daily precipitation interpolation schemes in meso-scale catchments. Water 7(2):747–779
57. Marcinkowski P, Piniewski M, Kardel I, Szcześniak M, Benestad R, Srinivasan R et al (2017) Effect of climate change on hydrology, sediment and nutrient losses in two lowland catchments in Poland. Water 9(3):156
58. https://curia.europa.eu/juris/liste.jsf?language=en&num=c-278/01
59. https://www.eea.europa.eu/policy-documents/quality-of-bathing-water-76

Publisher's Note

Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.