Decision Support for the Selection of Measures according to the Requirements of the EU Water Framework Directive

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Abstract One major scientific challenge posed by the EU Water Framework Directive (WFD) is the design of a decision support process that meets the Directive's requirement to achieve "good status" for all water bodies using a cost-effective combination of measures. This paper presents BAS*IN*FORM, a new decision methodology for selecting cost-effective management measures, developed in close co-operation with the water authorities and tested in the 5,154 km² mesoscale river Weisse Elster in central Germany. BAS*IN*FORM comprises (i) a procedure for framing the specific problems in the water bodies, including quantification of the need for action, (ii) modelling tools for quantifying the impacts of management measures, and (iii) a method for selecting cost-effective combinations of measures. One innovative feature of BAS*IN*FORM is that it structures the complex decision problems appropriately for practical use and provides an easy-to-use framework for integrating scientific and practical knowledge. A trial run applying BAS*IN*FORM to the Weisse Elster catchment revealed that good surface water status with respect to nutrient levels cannot be achieved if only the "standard" actions of current water management are taken to reduce point sources (sewage treatment) and diffuse agricultural sources. It also became clear that the nutrient-reduction measures available will generate considerable costs. The application of BAS*IN*FORM in this case study demonstrated its practical applicability in the WFD implementation process. Beyond the case study described here BAS*IN*FORM is currently being used for practical implementation of the WFD in the German Federal State of Thuringia.

Keywords EU water framework directive · Cost effectiveness · Decision support system · Decision making · Point and non-point pollution · BASINFORM · Meta-CANDY · WASP 5

1 Introduction: Challenges of the EU Water Framework Directive and Objectives of This Paper

An important institutional novelty introduced by the EU Water Framework Directive (WFD) was to hold the member states responsible for guaranteeing a certain status of

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waters (Petersen et al. 2009). Particularly in Article 4.1 the Directive demands that–in principle–all groundwater as well as all surface and coastal waters should achieve an ambitious environmental objective by 2015, namely "good status". The exact definition of criteria for "good status" is left to each member state of the EU. However, the Directive follows the idea of a "natural state" of water bodies as a reference point for defining good status. It also sets forth that criteria have to encompass physico-chemical and ecological features.

Results from the River Basin District Analysis conducted in 2004 in accordance with WFD Article 5 revealed an urgent need for action. It was very likely that "good status" would not be achieved in many parts of Europe unless adequate measures were taken: 40% of surface water bodies are "at risk" of failing to achieve good status and for another 30% the data was not sufficient to make a prognosis regarding water status (European Commission 2007: 28). WFD Articles 13 and 11 obliged the member states to establish a management plan for each river basin district by December 2009, in which they had to set out a programme of measures to be taken in order to achieve the Directive's environmental objectives and explain how the cost-effectiveness of these measures is taken into account. The latter is a particularly complex and laborious task, because a methodology for selecting a cost-effective combination of measures needs to provide both appropriate standardisations and sufficient flexibility to incorporate specific expert knowledge and regional specificities.

In academia as well as in professional consultancy, a number of decision support systems (DSS) for river basin management have been proposed to comply with WFD, but these have rarely been used by the competent authorities (de Kok et al. 2009). While some important conceptual work has been done (e.g. WATECO 2003; Interwies et al. 2004; Klauer et al. 2008) many of the available DSS concentrate on the specific technical planning aspects of measures. They are designed, for instance, to improve river morphology (e.g. Kofalk et al. 2005; Möltgen et al. 2005; Sewilam et al. 2007; Beechie et al. 2008), to manage water quantities (e.g. Giupponi et al. 2004; Giupponi 2007; Gastelum et al. 2009; Maia and Silva 2009; Khadra and Lamaddalena 2010), to reduce nutrient emissions (e.g. Schumann et al. 2005) or to moderate flood risks (e.g. Evers 2005; Winterscheid and Hübner 2006); for an overview see e.g. Matthies et al. (2007).

Many of the DSS are, however, less suitable for integrating multi-dimensional objectives, such as several quality criteria and cost-effectiveness, for structuring adequate decision procedures, and for considering different types of knowledge, e.g. scientific, expert and practical knowledge. Reviewing existing DSS supporting the WFD, Todini et al. (2006: 15) accordingly remark: "Most commercially available packages include extremely sophisticated components, but are deficient in the integration of interrelations between the different social, environmental, economic and technological dimensions of water resource planning." Although economic concepts in general play an important role in water management (Ward 2007), the issue of cost-effectiveness has so far been widely neglected in decision support concerning WFD. Moreover, the challenge of covering the various aspects necessary for holistic river basin management as required by the WFD in an integrated manner (including upstream-downstream problems, groundwater-surface water interaction, etc.) has not been addressed sufficiently.

This situation is mirrored in practical river basin management: In Germany, studies of the existing drafts of river basin management plans for the ten German river basin districts show that the methodological basis for selecting measures is generally not well documented, and that there are considerable differences between the plans themselves and the ways they are implemented (Bathe 2010). In particular, it remains unclear how cost-

effectiveness has been taken into account by the authorities when selecting measures, and how it can and should be taken into account in future management plans.

Against this background, the objective of this paper is twofold: It presents a new River BAS*IN IN*FORMation and Management System (BAS*IN*FORM) for selecting management measures in river basins that accounts for multi-dimensional management goals and explicitly addresses the issue of cost-effectiveness. Secondly, the paper describes how this tool was applied and tested in the 5,154 km² mesoscale catchment of the river Weisse Elster in central Germany in close co-operation with the relevant water authority (Ministry of the Environment for the German federal state of Thuringia) (Klauer et al. 2008).[1] The focus of this case study is on reducing nutrient pollution.

In contrast to many formal DSS, BAS*IN*FORM is a comprehensive and open decision methodology that encompasses decision procedures as well as elements of formal DSS, such as modelling tools. It consists of

(i)   a procedure to frame the specific problems for a water body at hand, including proposals for quantifying the need for action (i.e. the discrepancy between the status quo and the good water status to be achieved),
(ii) modelling tools for quantifying the impacts of management measures with a special focus on diffuse nutrient pollution (e.g. Meta-CANDY, WASP 5), and
(iii) a method for selecting cost-effective combinations of measures.

The innovative features of BAS*IN*FORM are that

– it structures the decision-making process for establishing a programme of measures by providing individual work packages as well as the necessary evaluation methods. Thereby it facilitates parallel decision-making in different administrative units.
– BASINFORM can handle multi-dimensional objectives, such as uncertainty concerning the impact analysis, acceptance of measures or impacts on other fields (nature conservation, flood protection etc.). For the final selection of measures in a water body, BAS*IN*FORM explicitly addresses cost-effectiveness.
– The decision methodology of BAS*IN*FORM is generic, i.e. it can consider different knowledge sources, e.g. scientific knowledge from modelling, expert knowledge, and practitioners' knowledge. Due to its open architecture the decision methodology is not restricted to the modelling tools proposed for the impact analyses.

BAS*IN*FORM can therefore be directly applied by water authorities. As an open decision-making framework it can be easily modified to meet the needs of different German federal states[2] as well as other EU member states.

The paper is organised as follows: The basic structure and methodological cornerstones of the decision methodology BAS*IN*FORM are described in the next section. Section 3 reports on the application of BAS*IN*FORM in the Weisse Elster river basin. It includes a brief description of the catchment, existing water management problems, their conceptualisation using BAS*IN*-FORM, and an introduction of the modelling tools for the impact assessment. In Section 4 the results of the modelling, the cost estimates and the selection of measures are presented. The paper closes with a discussion and summary of the benefits, transferability and application of BAS*IN*FORM in Section 5.

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[1] BAS*IN*FORM was also applied to the Emsbach water body (320 km², German federal state of Hessen, Richter et al. 2009a, b) and used for practical implementation of the WFD in Thuringia (Section 5).
[2] In fact, BAS*IN*FORM is currently (2010) being applied by the German federal state of Thuringia. The state of Brandenburg is currently considering BAS*IN*FORM for improving the hydro-morphology of river water bodies (Klauer and Bathe 2010).

2 BAS*IN*FORM—A New Decision Tool for Selecting Cost-Effective Management Measures

BAS*IN*FORM supports competent authorities in establishing a programme of measures by structuring the complex decision-making process into steps and integrating them into a workflow scheme (for an overview, see Fig. 2 below). In accordance with the WFD, BAS*IN*FORM identifies water bodies (or groups of hydrologically connected water bodies) as the basic spatial unit for water management. BAS*IN*FORM structures the planning process in three main phases:

Phase 1. Evaluation of the status quo and identification of the need for action
Phase 2. Identification of alternative measures and impact assessment
Phase 3. Selection and implementation of measures

2.1 Phase 1: Evaluation of the Status Quo and Identification of the Need for Action

2.1.1 Setting up Quantitative Development Targets

For surface water bodies, water status is composed of chemical and ecological status; for ground water bodies it consists of chemical and quantitative status (cf. WFD Article 2 and Annex V). The WFD defines the environmental objectives 'good *chemical* status' and 'good *quantitative* status' by reference to thresholds of certain quantitative parameters. The effects of measures on such parameters can be assessed comparatively easily. According to the WFD it is the responsibility of the EU member states to define the thresholds.[3] In contrast to good chemical and quantitative status, good *ecological* status of surface waters is much more difficult to underpin with quantitative parameters because it is closely dependent on the characteristics of the respective water body. Moreover, due to the complexity of the ecological system, the effects of measures on the ecological status of surface waters are very uncertain. Hence, they are difficult to model and to quantify. BAS*IN*FORM deals with this as follows:

Quantitative parameters for good chemical and quantitative status are used as management parameters with quantitative development targets for every water body involved. Defining such targets is necessary in order to quantify the impact of management measures, which is a requirement for assessing their effectiveness and cost-effectiveness. Where possible, threshold values defined by the "German Working Group on Water Issues of the Federal States and the Federal Government" (LAWA) are used as development targets. The discrepancy between status quo and targets essentially determines the *need for action*.

Striving towards supporting practical decision-making, BAS*IN*FORM also addresses the ecological status of surface waters in a way that is accessible to practitioners. Given the lack of comprehensive tools for modelling the ecological status, BAS*IN*FORM adopts quantitative development targets aiming at good ecological status that can–to the best of our knowledge–be considered as *prerequisites* for the development of satisfactory ecological conditions in surface waters. To take one significant example, rather than adopting the abundance of certain organisms as a direct development target for the

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[3] In Germany, it is the "German Working Group on Water Issues of the Federal States and the Federal Government" (Bund/Länder-Arbeitsgemeinschaft Wasser, LAWA) that establishes the thresholds for good water status. The relevant LAWA (2004) water quality classes (WQC) are discussed in Section 3.2.

ecological quality of a river, BAS*IN*FORM considers a combination of critical concentrations of specific nutrients and hydro-morphological parameters as development targets. This is one way to consider the limits of knowledge as well as modelling and forecasting uncertainties. The effects of measures aiming at ecological status can then be expressed in terms of their effects on these preventive management parameters.

BAS*IN*FORM expresses and considers critical concentrations of pollutants (e.g. nutrients or organic substances) as loads. This facilitates easy quantification of the effectiveness of a management measure as a reduced load of a certain pollutant, and the combined effect of several measures can easily be computed. In order to calculate the target values (i.e. the loads to be reduced) assumptions about discharge conditions (and their future development) have to be made in order to convert critical concentrations to critical loads.

Satisfactory hydro-morphology of rivers is another prominent factor for the achievement of good ecological status (Irmer and von Keitz 2002). Relevant quantitative parameters such as river depth and width variation, structure and substrate of the river bed, flow diversity as well as lateral and longitudinal banks can be handled by BAS*IN*FORM in an analogue way to pollutant loads. However, *comprehensive* implementation of good hydro-morphological conditions is in most cases not feasible or too costly and, on the other hand, it is not necessary for reaching a good ecological status (Völker and Borchardt 2007). Thus the spatial arrangement of zones of different morphological quality and, hence, of measures for improving hydro-morphology within the water body becomes important. For the German federal state of Brandenburg BAS*IN*FORM has been complemented by a module to support decision-making with respect to a suitable and cost-efficient spatial arrangement of hydro-morphological measures (Klauer and Bathe 2010) based on the ecological stepping stone concept (Völker and Borchardt 2007; DRL 2008).

2.1.2 Considering the Interaction of Water Bodies and Determining the Need for Action

When defining the target load for any particular water body, the upstream/downstream conditions need to be taken into account. As a first approach BAS*IN*FORM assumes that the quality standard is met at every point along the course of the river, hence concentration target values just meet the defined thresholds at every gauge (cf. Fig. 1). In particular this implies:

– *For upstream bodies*: The load target value should be set so that the development target (critical concentration) is exactly met.
– *For downstream bodies*: The water quality in downstream bodies depends (among others) on the water coming from the upstream bodies. In setting the target values, it is assumed that upstream water meets the development target. Hence, if this is not the case in the status quo, the effects of upstream management measures are anticipated.

The following additional aspects need to be considered in order to determine the need for action in a specific water body:

– *Impacts of deviating target values in upstream water bodies:* If pollutants are only slowly biodegradable (nutrients in particular), an excess or shortfall of a target in an upstream water body can (and should) be taken into account by adequate adjustment of the targeted load reduction.
– *Impacts of finalised groundwater measures:* Measures for improving groundwater quality in hydraulically connected groundwater bodies may also influence the quality of the surface water and need to be considered.
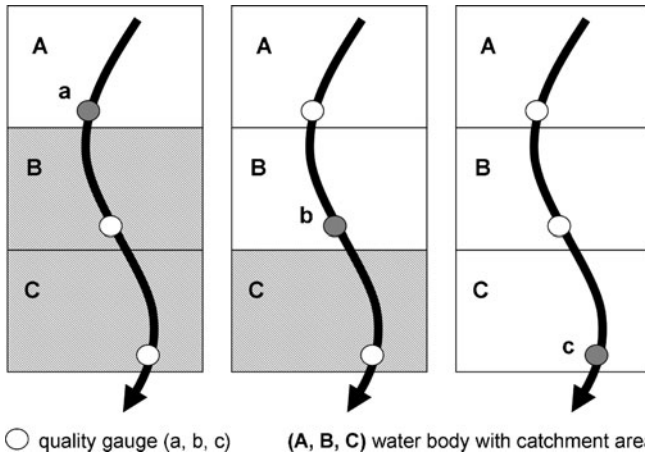
Fig. 1 Spatial definition of target values. The target values are determined successively starting in the upstream water body A in such a way that at each water quality gauge (**a, b, c**) the development target is just met

– *Baseline scenario:* The baseline scenario includes the effects of anticipated future developments that will most probably influence water pollution. Examples of such pressures include new sewage treatment plants or high-impact industries such as paper mills.

The way the need for action is calculated and, in particular, how these influences are taken into account is shown in Table 1.

2.2 Phase 2: Identification of Alternative Measures and Impact Assessment

A *measure* is an action taken by the competent authorities that either directly (in that it directly impacts the environment) or indirectly (in that it influences the behaviour of relevant third party actors) contributes towards improving water quality. For example a direct measure could be the renaturation of a river segment; an indirect measure would be a promotional or financial programme for organic farming. Here we assume that private agents such as farmers can be influenced primarily indirectly via financial or promotional instruments.

In many cases authorities already have extensive knowledge about possible measures for the improvement of the quality status of waters. BAS*IN*FORM therefore assumes that authorities pre-select potential measures from a catalogue. Criteria for this pre-selection of measures are their usefulness for a specific water body as well as their technical and political feasibility.

As the description of the measures in the catalogue is necessarily very general, they must be spatially and temporally specified as well as technically concretised with respect to the particular situation of the water body at hand. The effectiveness of the adapted measures, their costs and other impacts are then assessed by experts in greater detail. These estimates may be based on targeted measurements or on a modelling of the measures' impacts, on scientific results from expert reports or literature, on experts' personal experience, or a combination of these. BAS*IN*FORM uses some modelling tools for the impact assessment of measures–these are introduced in section 3. However, a key feature of BAS*IN*FORM,

Table 1 Examples for calculating the need for action for specific management parameters (hypothetical figures) in a given water body

Water body xy

Targets, status quo value				
Cause	Organic matter		Nitrogen	Phosphorus
Management parameter	BOD_5	NH_4-N	NO_3-N	P
Development target	4 mg/l	0.3 mg/l	6 mg/l	0.3 mg/l
Target–character	load	load	load	load
Target–unit	[t/year]	[t/year]	[t/year]	[t/year]
Target–value	50	8.0	50	4.2
Status quo value	500	25.5	200	9
Impacts of developments to be considered				
A_1: baseline scenario				
- expected investments	120	9.0	30	−0.3
- impacts of future developments	30	1.5	10	0
A_2: impacts of hydraulically connected groundwater and surface water bodies				
- Groundwater body 1	0	0	50	0
- …	….	…	…	…
- Groundwater body k	0	0	0	0
A_3: impacts of a deviation from the target value in the upstream water body				
- upstream water body 1	−10	−0.5	0	0
- …	…	…	…	…
- upstream water body m	0	0	0	0.2
Calculating the need for action				
Need for action = status-quo–A_1–A_2–A_3–target value	310	7.5	60	4.9

which is important for its practical applicability, is that any method of impact assessment can be integrated into its open decision framework.

The results of the impact analysis are presented in an impact matrix (see Table 2). In addition to information about effectiveness and costs, it may also contain additional information about the acceptance of management options, about further impacts (e.g. cross effects or impact on further management parameters) and about uncertainty. This additional information may be quantitative, e.g. in the case of modelling uncertainty, or qualitative.[4]

2.3 Phase 3: Selection of Measures Taking Cost-Effectiveness into Account

As the focus of BAS*IN*FORM is on practical decision-making, simplifications are introduced. To reduce complexity, BAS*IN*FORM establishes five modules in which the selection of measures is carried out separately. These modules are:

Module 1:   Nutrients and pesticides in groundwater

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[4] Information about degrees of uncertainty may in fact be very important for deciding about management options. Such information may stem from different sources: in the approach to impact analysis introduced in Section 3, quantitative information about modelling uncertainty is obtained. In more practical approaches quantitative information may not be available—in these cases uncertainties may be estimated by the experts involved.

Table 2 Example of an impact matrix (hypothetical entries). The cost-effectiveness columns are calculated by dividing cost by effectiveness

Impacts	Reduction (effectiveness)			Costs	Cost-effectiveness			Other impacts		
Individual measures	N_{tot} [t/a]	P_{tot} [t/a]	Organic matter (BOD_5) [t/a]	[T€/a]	N [€/t]	P [€/t]	Organic matter [€/t]	Acceptance	Uncertainty	Further impacts
Water body ##										
Sewage treatment 1	20	1.5	40	100	5,000	66,700	2,500	medium	low	none
Sewage treatment 2	15	1.6	0	70	4,700	43,800	–	good	low	none
Canal system 1	10	0.6	20	70	7,000	117,000	3,500	low	low	none
Canal system 2	20	1.4	40	60	3,000	42,900	1,500	good	low	none
Riparian buffer strips	5	0.8	0	40	8,000	50,000	–	low	high	positive
Soil conserving cultivation	20	2.2	0	120	6,000	54,500	–	good	medium	none
Total impact	40	2.9	80	160						
Management target	60	4.9	70							

Module 2: Other groundwater pollution
Module 3: Organic matter, nutrients and pesticides in surface water
Module 4: Structure and hydro-morphology of surface water
Module 5: Other surface water pollution

This division into modules is important to facilitate task sharing and parallel processing within the administration. They are chosen in such a way that the remaining interactions between them are reduced to a minimum. And some interaction is integrated within the BASINFORM procedure–for example measures targeting groundwater status are considered in phase one of Modules 3–5 if they are hydrologically relevant to the respective water body (see Fig. 2). Experience from practical implementation in Thuringia shows that coordination across the modules is improved if experts working on the different modules engage in a discussion process.

The most suitable process for finding the best combination of measures within a module is case-specific. If the measures impact on only a single target parameter and costs are the sole criterion, then selection is a problem of simple optimisation. If, however, measures have an impact on multiple targets or show cross-effects (as is regularly the case), and total costs are only one of several evaluation criteria, then the decision situation is more complex. For such complex cases, we suggest the following two-step approach for selection of measures:

1. Measures from the list of potential measures are grouped into different combinations of measures (using different algorithms), all of which are, as a bundle, capable of meeting the need for action. The cost-effectiveness of individual measures is an important criterion.
2. From this bundle, the "best" combination of measures is selected using total costs and further evaluation criteria, such as positive or negative cross-effects on agriculture, the economy, transportation and recreation, and their acceptance among those affected, etc.
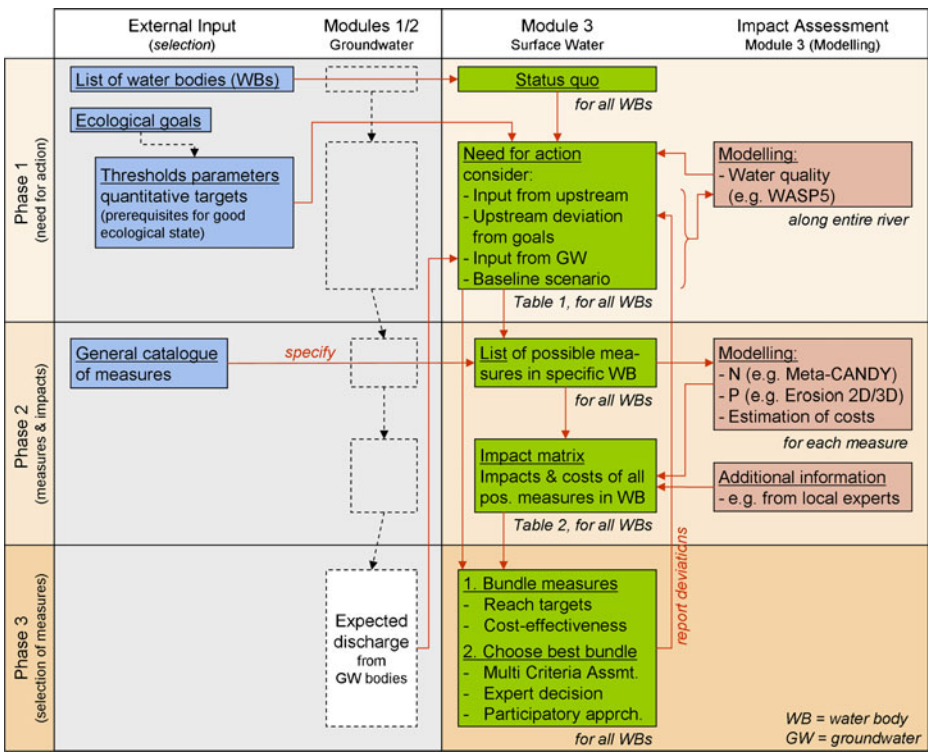
Fig. 2 Overview of BAS*IN*FORM decision methodology for module 3. Selected inputs (upstream water body and groundwater) are illustrated as well as the modelling approach for the impact assessment that is employed in the Weisse Elster case study (see Section 3)

Step 1: Different algorithms exist for combining individual measures. One reasonable algorithm based essentially on information about cost-effectiveness is as follows: The basic idea is a sequential addition of measures according to their cost-effectiveness (see impact matrix, Table 2) that proceeds until the needs for action have been accomplished for each management parameter. It is then necessary to check whether targets have been surpassed, and if so single measures can be removed. This step-by-step build-up of combinations is intended to encourage the experts involved to compare the advantages and disadvantages of individual measures. The problems attached to specific measures can be recognised through this process and the logic of the measures tested accordingly. This basic algorithm can be varied by considering further criteria besides cost-effectiveness when adding new measures. Different algorithms will generally lead to different combinations of measures. One of the decisive ideas of BAS*IN*FORM is that this process of creating combinations of measures should not rely primarily on formal (optimisation) algorithms. As problems and adequate measures are multi-dimensional in most cases, experts shall be encouraged to use their knowledge and intuition rather than rely on any 'automated' optimisation procedure. The outcome of this stage is a shortlist of combinations of measures, all of which meet the need for action.

Step 2: For each module three suitable combinations should be selected and, finally, the "best" overall combination identified. One important criterion in this stage of the

selection process is that of total costs. Further criteria such as political feasibility, economic impact, and degree of uncertainty of target achievement should also be included. This step can be supported by a multi-criteria analysis (e.g. a scoring approach) if the decision problem is complex (e.g. Vincke 1992; Roy 1996; Klauer et al. 2006) but alternative approaches may also be appropriate (see Klauer et al. 2007: 29).

Figure 2 provides an overview of the BAS*IN*FORM decision methodology using the example of Module 3 that was applied in the Weisse Elster case study.

3 Application of BAS*IN*FORM to the Weisse Elster River

The BAS*IN*FORM procedure for identifying the need for action and the "best" combinations of required measures has been developed, applied and tested on the problem of nutrient pollution in the Weisse Elster River in central Germany. Within this project, impact assessment has been carried out using several modelling approaches (Rode et al. 2008b)–these approaches and key results are introduced in the following two sections.

3.1 The Weisse Elster Catchment

3.1.1 Catchment Characteristics

The Weisse Elster river basin is a sub-catchment of the River Saale, which is the second largest tributary of the River Elbe. The catchment area covers 5,154 km² (gauging station Oberthau) and is mainly situated in the German federal states of Saxony, Thuringia and Saxony-Anhalt. The river originates in the Ore Mountains in the Czech Republic, is 250 km long, and has a mean discharge of 26 m³/s (Oberthau station). The river channel structure is very diverse, with near-natural stretches as well as concrete-lined segments.

Land use in the basin is dominated by agriculture (43% cropland, 16% pasture), especially in the lower part, and by forestry (21%), mainly in the upper part (see Fig. 3). The upper part of the basin is mountainous and dominated by steep slopes and narrow valleys, with hardly any floodplains and scarce groundwater resources. The geology is characterised by igneous and metamorphic rocks and consolidated tertiary rocks (sandstone). The lower part of the basin is situated in the lowlands and consists of mainly Pleistocene coverage. Precipitation varies between 500 mm in the northern part of the basin (lowlands) to 1,000 mm in the southern part (mountains). Annual runoff varies between approximately 50 mm and 600 mm.

The main field crops are grains and root crops in the northern part of the catchment and forage crops in the southern, more hilly parts of the basin. Conventional farming makes up 98%, organic farming 2% of the total cropland. Livestock in the area consists mainly of cattle. Settlements, industrial areas, and infrastructure account for 16% of land use, with Leipzig and Halle being the major cities located in the catchment. The area south-west of Leipzig is dominated by active and reclaimed open pit mines.

In Germany the federal states, rather than the federal government, are responsible for water management. The implementation of the WFD for the Weisse Elster River is coordinated by representatives from the environment ministries of the three federal states of Thuringia, Saxony and Saxony-Anhalt.
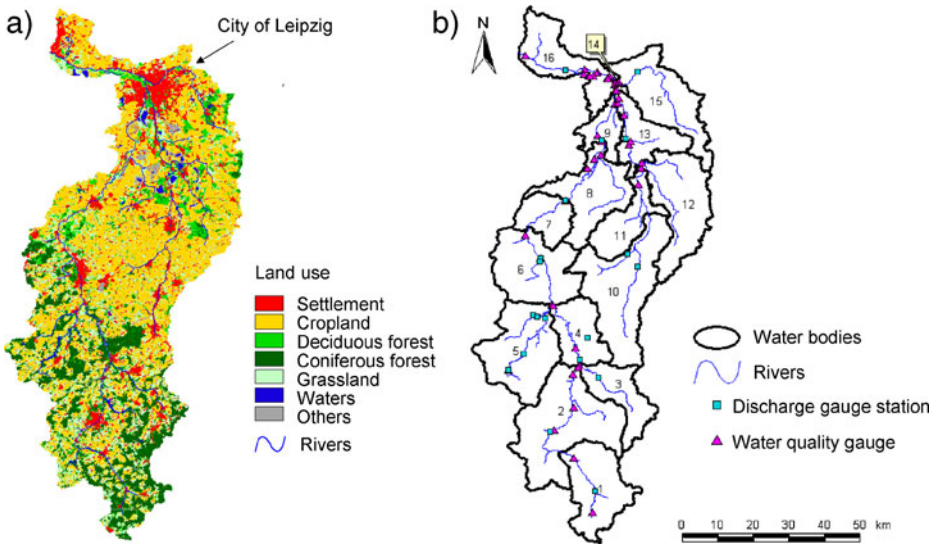
Fig. 3 a Land use in the Weisse Elster River basin. Classification on the basis of Landsat ETM Data of 1999 (Source: Klauer et al. 2008), **b** Water bodies and gauging stations of the Weisse Elster River Basin employed in the case study

3.1.2 Data Availability and Sources

Meteorological data were provided by the German national meteorological service. There are about 60 precipitation and 11 climate stations in and around the Weisse Elster basin. Daily data exist for most of the precipitation gauging stations, while six-hourly or hourly data were available from the climate stations. Time series data were collected from 1990 to 2003. Daily water level measurements were provided for about 20 gauging stations in the Weisse Elster catchment. Water level time series and discharge data for the years 1990 to 2002 were used. While there are about 20 water quality monitoring stations in the Weisse Elster catchment, measurements were taken only 1 to 2 times per month. A large number of physico-chemical properties were measured. Measured data on water extraction and discharge to the river are not available, so data are based on official permitted levels.

A digital elevation model was used at 50 m resolution. Land-use information was derived from Landsat imagery at a spatial resolution of 30 m for 1989 and 1999. A detailed biotope map derived from aerial photography was also used. Several soil maps exist in digital format with spatial resolutions of 1:1,000,000 and 1: 200,000.

3.2 Water Problems in the Weisse Elster River

3.2.1 Identification of the Need for Action

Most of the Weisse Elster River has been classified as moderately polluted, with some segments classified as critically polluted. Nutrients are the main problem, which is why nitrogen and phosphorus in the surface water have been selected for the case study as the main management parameters for ecological status to be treated with BAS*IN*FORM. In fact, this case study can be considered representative for a wide range of water

management problems in central Europe, because alongside problems of river morphology and poor passability for migrating aquatic organisms, high nutrient loads are the main reason why river water bodies in Germany and in central Europe are at risk of failing to achieve 'good ecological status' (Borchardt et al. 2005; European Commission 2007: 33).

Due to the fact that the German federal states are politically independent, different development targets have been set for nutrient contamination in surface waters: Saxony and Saxony-Anhalt take 3 mg N_{tot}/l and 0.15 mg P_{tot}/l as threshold concentrations, corresponding to LAWA[5] Water Quality Class 2 (WQC 2). Thuringia, by contrast, takes 6 mg N_{tot}/l, 0.3 mg P_{tot}/l as a development target, corresponding to LAWA water quality class 2–3 (WQC 2–3, see LAWA 2004). In our case study we investigate both thresholds, WQC 2 and WQC 2–3, as alternative scenarios.

Observed nutrient concentrations in the Weisse Elster and its major tributaries (given as 90 percentiles for the year 2001) range between 6.1 and 13.0 mg N_{tot}/l and 0.14 and 0.74 mg P_{tot}/l. Diffuse discharges, originating mainly from agriculture, have been estimated to contribute to the overall nutrient load by 84% (nitrogen) and 65% (phosphorus) (Behrendt et al. 1999). High NH_4 and PO_4 concentrations in lowland river reaches are caused by high sewage inputs from urban areas. The percentage of households connected to the public sewage system ranges from over 97% in the cities of Leipzig, Halle and Zwickau to slightly over 60% in rural areas. More than 64% of the sewage network is operated as combined sewers (domestic wastewater and storm water), resulting in a high risk of overflow at sewage plants during rainstorm events (Haase et al. 2008: 70).

Based on the general nutrient development targets, water body-specific target values and needs for action were determined–see Table 3. As an example consider the combined downstream water bodies 14/16 that cover the urban area of Leipzig (Fig. 3): The corresponding water quality gauging station "Ammendorf" is located near the outlet of the Weisse Elster River. The total annual nitrate-N load at Ammendorf was 4,273 t/a in 2001. A target concentration of 3 mgN/l for WQC 2 corresponds to a nitrate-N load of 1,550 t/a. The nitrate-N load produced *within* water bodies 14/16 is 361 t/a and the corresponding target value is 88 t/a. Based on the assumption that all upstream water bodies meet their respective target thresholds, it is determined as the load of nitrate-N originating from water bodies 14/16 which facilitates compliance with the target concentration. The difference between N load produced and target value indicates a need for a nitrate-N reduction (need for action) of 273 t/a within the water bodies. Table 3 lists the needs for action for all water bodies in the Weisse Elster Basin and for both water quality classes.

3.2.2 Pre-selection of Potential Management Measures

The following list of potential measures for mitigating the nutrient contamination in the Weisse Elster River was obtained from a detailed review of the literature. This list was discussed with experts from the authorities responsible for managing the Weisse Elster who considered it to be "complete" in the sense that it comprises all potentially effective and practicable measures for mitigating eutrophication (see Schiller et al. 2008: 96–98):

1. Construction of new and improvement of existing *sewage treatment plants*

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[5] LAWA stands for "German Working Group on Water Issues of the Federal States and the Federal Government".

Table 3 Needs for action for all 16 water bodies with respect to nitrate and phosphorus and for water quality classes 2 and 2–3, based on the status quo in 2001

Water body no.	Nitrate [t/a]			Phosphorus [t/a]		
	Need for action (WQC 2)	Need for action (WQC 2–3)	Maximum possible reduction*	Need for action (WQC 2)	Need for action (WQC 2–3)	Maximal possible reduction*
1	176	21.8	59.3	1.80	0.0	1.99
2	336	192	173	5.20	2.6	5.87
3	157	73.2	79.3	5.20	0.0	1.00
4	6.3	0.0	64.0	0.00	0.0	2.09
5	228	157	120	1.90	0.0	3.47
6	239	162	104	7.10	3.5	2.05
7	335	191	64.5	10.0	7.1	2.10
8	210	0.0	133	5.00	2.2	4.00
9	17	11.2	28.1	0.00	0.0	0.75
10	292	203	269	7.80	4.3	8.91
11	118	81.9	77.0	3.20	1.7	2.94
12	213	148	157	5.70	3.1	4.04
13	82	57.2	124	2.20	1.2	0.46
15	136	93.6	135	10.7	6.0	3.74
14 & 16	273	222	332	15.8	9.8	2.14

* "Maximum possible reduction" refers to the maximum possible load reduction by means of *all* measures shortlisted for the respective water body, irrespective of their costs and further selection criteria–see Section 4 for the calculation using the examples of water bodies 8 and 13

2. Increase in the share of organic and/or integrated farming as a proportion of total cropland from currently 98% conventional farming, 2% organic farming and 0% integrated farming to one of the following:
   a) *100% integrated farming,*
   b) *15% organic farming, 85% conventional farming,*
   c) *15% organic farming, 85% integrated farming,*
   d) *30% organic farming, 70% conventional farming,*
   e) *30% organic farming, 70% integrated farming.*

As discussed in Section 2.2, a "measure" in this sense could be a promotional programme targeting the given shares of organic/integrated farming in the catchment of the respective water body. The given proportions are a pragmatic compromise between achieving a significant impact and feasibility. Integrated farming is characterised by more efficient use of fertiliser at a lower level overall (Schiller et al. 2008). Organic farming is defined in EG regulation 2092/91 and differs from conventional farming principally in its abandonment of pesticides and synthetic fertilisers. Obviously, management measures 2a through 2e are mutually exclusive.

3. Conversion from conventional to *conservation farming* using erosion-minimising techniques on all arable land. Conservation farming is compatible with integrated farming and (at least partly) with organic farming.
4. *Restoration* of river segments with poor hydro-morphology.

With the exception of measures 2a through 2e, all management measures are compatible with each other.

3.3 Quantifying the Impacts: The Modelling Tools

The impacts of these potential management measures have been estimated using modelling tools. Three different modelling approaches were used to simulate nitrate leaching from the soil zone, soil erosion and diffuse phosphorus inputs, and in-stream nutrient retention in the main reach of the Weisse Elster River.

3.3.1 Modelling Nitrate Leaching: Meta-CANDY

Nitrogen loading of subsurface flows is simulated using a meta model (Schmidt et al. 2008; Hesser et al. 2008, 2010). This is a regression model using just a few important factors as explanatory variables that control nitrate leaching from agricultural land. The quantification of these factors is based on simulations of the more detailed agro-ecosystem model CANDY (Franko et al. 1995).

The meta model calculates nitrogen leaching from the soil zone for so-called *nitrate response units* (NRU). These NRUs are defined as areas which behave similarly in terms of nitrogen leaching due to their comparable soil, land use and climate conditions.[6] Within the relatively homogeneous NRUs, the simulated farming systems within CANDY were used to carry out a multiple regression analysis. The analysis expresses nitrate leaching from the soil zone (N_{leach}) in a given NRU as a dependent variable which is dependent on the independent variables (I_i) that are significantly correlated with N_{leach}.

$$N_{leach} = a_0 + \sum_{i=1}^{n} (a_i \cdot I_i) \qquad (1)$$

with a_0 = intercept and a_i = regression coefficients for each NRU.

Livestock and the share of cereals in crop rotation were identified as the most significant parameters (I_i). The spatial availability of these two parameters in common agricultural statistics is a condition for applying the model at larger scales within BAS*IN*FORM.

The regression (meta) model allows a large number of model runs and, therefore, is able to account for uncertain fertiliser application. For that the model simulates nitrogen leaching for every NRU in the catchment using a probability distribution function for fertiliser input. The meta model outputs average as well as normal distributions of N-leaching from the soil zone (Schmidt et al. 2008).

The parameters of the meta model were calculated for the Weisse Elster catchment using the CANDY model based on 125 virtual crop rotations for selected most important soil types. Meta model runs were then carried out for the three agricultural land-use scenarios conventional farming, ecological farming, and integrated farming and pasture. Nitrogen leaching from the soil zone is calculated for each land-use scenario on the basis of the meta model calculation for each nitrogen response unit.

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[6] The NRUs were aggregated from *hydrological response units* (HRUs, Flügel 1995), because not all HRU attributes (such as elevation, slope, exposition, geology) proved to be important for nitrogen leaching calculations with the meta model.

3.3.2 Modelling Soil Erosion and Phosphorus Input

Soil erosion and phosphorus input simulations are based on the method of the Erosion 2D/3D model combined with a Monte Carlo approach for disaggregating spatially uncertain input data. The basic idea behind the model is the assumption that the erosive impact of overland flow and droplets is proportional to the momentum fluxes exerted by the flow and the falling droplets respectively (Schmidt et al. 1999; Shbaita and Rode 2008). By analogy, soil erosion resistance is expressed in the form of a critical momentum flux. Sediment losses from agricultural land are calculated by taking into account deposition based on the sediment delivery ratio (according to Neufang et al. 1989) and the catchment size. The empirical enrichment ratio for phosphorus in eroded sediments (Neufang et al. 1989) is used to calculate particulate P loss. Soluble phosphorus losses are simulated using a constant P concentration of 2 mg/l in surface runoff and the calculated surface runoff of each grid (Rode and Frede 1997). Soil erosion is calculated on the basis of regionally specific high-resolution breakpoint storm rainfall data. The selected storm events represent mean annual climate conditions within the Weisse Elster catchment. The dependency of soil physical properties (such as hydraulic conductivity, content of organic matter, and bulk density) on soil cultivation data were defined according to Schmidt et al. (1996).

The P-transport model was used to simulate the baseline scenario as well as selected management measures. The new approach makes it possible to generate spatially high-resolution soil and P loss maps. The method is restricted to summer rainfall events; winter conditions with soil frost are not considered. For this reason, the calculated soil and P losses are conservative estimates.

3.3.3 Modelling In-Stream Nutrient Retention: WASP 5

In-stream matter transport was simulated using the water quality model WASP5 (Water Quality Analysis Simulation Program, Ambrose et al. 1993). WASP5 is a 1- to 3-dimensional numerical model and adopts a deterministic approach to describe the hydrodynamics and the turnover of nutrients and chemicals in the water column and in sediments. The WASP5 modelling system consists of three stand-alone computer programmes that can be run either in conjunction or separately: DYNHYD is a hydrodynamic model based on the Saint Venant equations. In this study a modified version of DYNHYD (Warwick 1999) was used which allows for consideration of weirs. EUTRO can be used to model oxygen depletion, eutrophication, and nutrient enrichment in the river. An extended version of EUTRO was applied in the study (Shanahan and Alam 2001), which uses nine model variables: biomass of phytoplankton, biomass of periphyton, dissolved oxygen, biochemical oxygen demand (BOD), ammonia nitrogen (NH_4), nitrate nitrogen (NO_3), organic nitrogen, phosphate (PO_4) and organic phosphorus. The complex system of these variables is described by several processes, such as growth and decay of the autographs, settling, reaeration, sediment oxygen demand, nitrification, denitrification and mineralisation. Finally, TOXI simulates sediment transport and the fate of toxic inorganic and organic chemicals (see also Wagenschein and Rode 2008).

The Weisse Elster River water quality model consists of 872 river cross sections, with some river sections of only 100 m length. This was necessary to account for the high variability of river morphology which has a considerable impact on denitrification processes and nutrient uptake by primary production. According to the temporal variability of in-stream nutrient turnover processes the suggested river water quality model runs on an hourly time step. Discharge and nutrient load input data were obtained from the water

authorities and other measurement activities. Point source data from sewage systems were used directly as inputs into the WASP5 model for the Weisse Elster River.[7] In-stream denitrification, which is the most important nitrogen removal process, varies considerably within the main course of the river, depending on river morphology, and tends to decrease with increasing flow length (see Fig. 4), due to the decreased ratio of river circumference to cross section. For the main course of the Weisse Elster (Strassberg-Großzschocher), the simulated mean annual retention was 19% of the total nitrogen load (Rode et al. 2008a).

Within BAS*IN*FORM, the water quality model was used to quantify the interaction along the river course with respect to nutrients, i.e. the influence of the headwater nutrient loads on subsequent water bodies. Also the impact of renaturation measures on nutrient depletion was simulated.

3.3.4 Calculation of Costs

When evaluating management measures it is not only the impacts on water status but also the costs that are important. In the case of waste water treatment both aspects are closely interrelated and are therefore addressed together. There are 196 sewage treatment plants of different sizes and technical standards in the Weisse Elster basin. During the initial screening stage so-called "priority plants" were identified. These are the plants in which development or reinforcement measures promise a substantial reduction of nutrient loads. The costs of reducing a certain load of N and P depend mainly on the size and on the level of nutrient elimination already achieved. Using cost data from the literature and from interviews with companies operating treatment plants, a calculation scheme was developed that assigns the potential nutrient reduction as well as the related costs to each plant (for details see Bräuer and Neubert 2008: 182–199).

The costs involved in changing farming systems, e.g. increasing the share of organic and/or integrative farming, were calculated in a two-step approach on the basis of data taken from official agricultural statistics at district level:

1. For each district the costs-per-hectare ratio for arable land production with (i) conventional farming, (ii) organic farming, and (iii) integrated farming are calculated on the basis of data on the standard contribution margin taken from agricultural statistics.
2. This cost information is used to determine the total cost of the different management measures 2a–2e (see Section 3.2.2) for the specific situation (e.g. the relation of arable land to grassland) of the water body at hand (see Bräuer and Neubert 2008: 176–182).

The costs of introducing conservation farming were calculated according to subsidy rates offered by the agricultural promotional programmes of the German federal states. These subsidy rates approximately reflect the factual costs; they amount to approximately 25 euros/(ha·a).

There are considerable uncertainties attached to estimating costs. While they may be the lowest in the case of technical measures with existing extensive experience, cost estimations are more uncertain when it comes down to measures involving large groups of actors, such as the modification of agricultural systems. The costs of the renaturation measures could not be estimated in the present case study–this would have required detailed

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[7] An uncertainty analysis based on the Monte Carlo approach (Reichert and Vanrolleghem 2001) was carried out for the calibrated model (using PEST, Doherty 2002). A detailed sensitivity analysis can be found in Wagenschein (2006). During the calibration process, PEST can be used to quantify the 95% confidence interval.
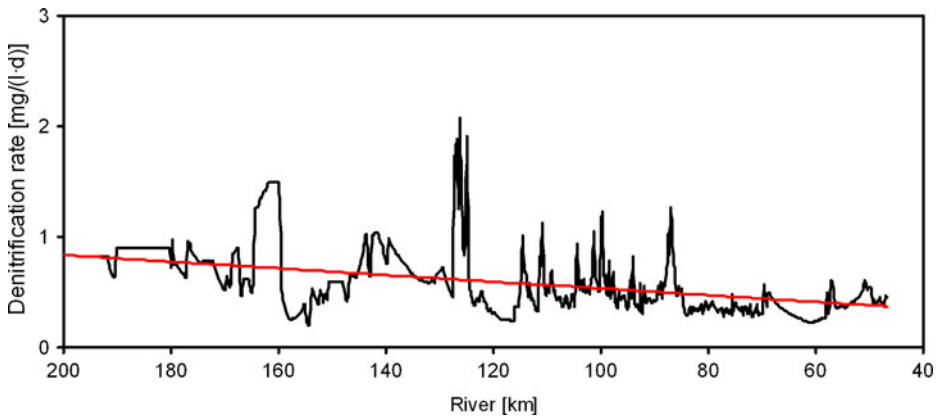
Fig. 4 Simulated denitrification rate along the main course of the Weisse Elster River in summertime conditions

planning. However, when applied to nutrient reduction they are expected to be very high compared to the alternatives.

4 Weisse Elster Case Study: Results

The methods and models described in the previous section were used to assess the effects of the pre-selected measures for mitigating nutrient contamination in all Weisse Elster water bodies. The results are summarised in Tables 4 and 5 taking water bodies 8 and 13 as examples.

Table 4 shows that in water body No. 8 maximum reduction[8] can be achieved if the measures numbered in bold (## 1, 2e, 3, 4) are implemented simultaneously: The sewage treatment plant Lucka is improved with respect to its N- and P-elimination capacities, the current share of organic farming is increased to 30% and the share of integrated farming is increased to 70% of total cropland, conservation farming is implemented area-wide, and renaturation measures are implemented. However, all these measures are not sufficient to achieve WQC 2 for nitrogen or for phosphorus because the need for action surpasses the maximum reduction. If the standard is lowered to WQC 2–3, the status quo already meets the reduction target with respect to nitrogen. In order to achieve WQC 2–3 for phosphorus, only the measure 'conservation farming' must be taken, which would cost approximately 480,000 €/a. In this example the number of alternatives and the room for manoeuvre is so small that costs and cost-effectiveness of the measures are irrelevant for the selection—the selection problem becomes rather trivial.

Table 5 shows the results for water body No. 13 where, by contrast, cost considerations were helpful in selecting the measures. To meet the need for action (WQC 2) concerning nitrate the sewage treatment plant Markkleeberg has to be improved (measure 1b) and integrated farming should be implemented on all cropland (2a). As an alternative to measure 2a the share of organic farming could be raised to 15% or 30% with the share of integrated farming at 85%/70% respectively (measures 2c or 2e). However, both of these

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[8] Maximum possible load reductions in this sense are given in Table 3 (Section 3.2) for all the water bodies in the Weisse Elster river.

Table 4 Impact matrix for water body no. 8: results of the impact assessment

No	Impacts	Reduction effectiveness		Costs	Cost-effectiveness		Other impacts	
	Individual measures	N_{tot} [t/a]	P_{tot} [t/a]	[€/a]	N [€/t]	P [€/t]	Acceptance	Uncertainty
	Water body no. 8							
1a	Seawage treatmt. Lucka (N-elim.)	0.66		3400	5150		good	medium
1b	Seawage treatmt. Lucka (P-elim.)		0.30	2300		7670	good	medium
2a	100% integrated farming	96		73300	760		good	high
2b	15% organic 85% conventl. farm.	25		332200	13290		medium	high
2c	15% organic 85% integrated farm.	110		437900	3980		medium	high
2d	30% organic 70% conventl. farm.	54		715500	13250		low	high
2e	30% organic 70% integrated farm.	124		802500	6470		low	high
3	Soil conservation		3.70	483200		130590	good	low
4	Renaturation	8.3					low	medium
	Maximum reduction	133	4.00	1,291000				
	Need for action (WQC 2)	210	5.00	1,291000				
	Need for action (WQC 2–3)	0	2.20	483000				

alternatives are more expensive in absolute terms and less cost-effective. If WQC 2–3 is taken as the quality standard, the first best choice remains the same.[9] With respect to phosphorus it is not possible to meet the targets for WQC 2 or for WQC 2–3.

Table 6 lists the number of failures of the different water bodies of the Weisse Elster River with respect to (i) nitrate and phosphate and (ii) to WQC 2 and 2–3. The overview reveals that the management measures so far considered by the experts are not sufficient to meet the objectives of the WFD. It should be kept in mind that the list includes all the measures considered by the competent authorities to be (i) potentially effective, (ii) technically practicable and to have (iii) at least a slight chance of political acceptance at the present time. So this analysis shows that there is an urgent need to develop innovative, more effective measures to reduce nutrient pollution. One way forward in this respect might be so-called "precision farming" using farming techniques (crop rotation, soil cultivation, fertilizer input etc.) specifically adapted to small-scale local natural conditions.

Table 7 shows the costs of the most cost-effective combinations of measures for achieving (i) maximum reduction, (ii) WQC 2 and (iii) WQC 2–3 for the different water bodies. It has been assumed that in case of failure to achieve the WQC, all effective measures are taken, irrespective of their costs and cost-effectiveness.

Table 7 gives an idea of the dimensions of the total costs of implementing the WFD with respect to nutrient reduction in the Weisse Elster Basin. A reduction in nutrient contamination to the level of water quality class 2 costs approx. 9 million euros every year. Even with this expenditure, the development target is missed in 13 water bodies. If water quality class 2–3 is envisaged, costs fall to 6 million euros per year and even then the lower development target would still not be achieved in 8 water bodies.

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[9] In this case measures 1b and 2a together lead to an overfulfillment of the quality target. One option might then be to introduce integrated farming only on a percentage of cropland. Whether this is possible depends on the chosen instrument for implementation, i.e. whether technical farming standards are modified or promotional programmes are introduced. This option was not investigated in the project. In a real decision context this would be a starting point for designing a specifically adapted measure.

Table 5 Impact matrix for water body no. 13: results of the impact assessment

No	Impacts	Reduction effectiveness		Costs	Cost-effectiveness		Other impacts	
	Individual measures	N_{tot} [t/a]	P_{tot} [t/a]	[€/a]	N [€/t]	P [€/t]	Acceptance	Uncertainty
Water body no. 13								
1a	Seawage treatmt. Espenhain	30.1		34500	1150		good	medium
1b	Seawage treatmt. Markkleeberg	41.9		10600	250		good	medium
2a	100% integrated farming	42		33200	790		good	high
2b	15% organic 85% conventl. farm.	10		112000	11200		medium	high
2c	15% organic 85% integrated farm.	47		155000	3300		medium	high
2d	30% organic 70% conventl. farm.	21		242000	11500		low	high
2e	30% organic 70% integrated farm.	52		277000	5320		low	high
3	Soil-conserving farming		0.46	237000		515000	good	low
	Maximal reduction	124	0.46	559000				
	Need for action (WQC 2)	82	2.20	281000				
	Need for action (WQC 2–3)	57	1.20	281000				

5 Discussion and Perspectives for BAS*IN*FORM

The BAS*IN*FORM decision-making methodology introduced here is specifically designed to meet the requirements of the EU Water Framework Directive. It considers the basic spatial units for practical water management, namely water bodies and their associated river reaches. In the case study the BAS*IN*FORM approach was applied to the reduction of nutrient loads, but the suggested methodology is generic and therefore not restricted to nutrient load problems. BAS*IN*FORM can also be used for other water management problems, e.g. the implementation of river restoration measures, where measures have to be prioritised according to specific river units. The range of applications is basically restricted by the availability of valid water quality and ecological models and cost calculation methods for predicting the costs and effects of management measures. The BAS*IN*FORM methodology is designed in such a way that different modelling tools can be applied for impact assessment.

In the Weisse Elster case study we used a meta model (Meta-CANDY) for calculating nutrient loading from agricultural land. As a statistical tool it can be applied to meso-scale catchments of up to several thousand square kilometres whereas the size of single NRUs depends on the input data available and can therefore be small, e.g. several hectares (Schmidt et al. 2008). The Meta-CANDY model calculates yearly nutrient losses from agricultural land. The WASP5 river water quality model offers high spatial and temporal resolution calculation for in-stream nutrient turnover (Section 3.3.3). The presented WASP5

Table 6 Overview of failures to achieve target values even if all potential measures are taken simultaneously in the different water bodies with respect to the parameters N and P as well as with respect to WQC 2 and WQC 2–3

Need for action	WQC 2		WQC 2–3	
Parameter	N	P	N	P
# of failures	11	9	5	5
# of simultaneous failures of N and P	13		8	

Table 7 Costs of the most cost-effective combination to reach (i) maximum reduction, (ii) WQC 2 and (iii) WQC 2–3

Costs for Water body no.	maximum reduction [1000 €/a]	WQC 2 [1000 €/a]	WQC 2–3 [1000 €/a]
1	371	369	24
2	710	699	700
3	260	260	123
4	589	34	0
5	460	460	23
6	280	280	280
7	714	714	714
8	1291	1291	483
9	321	22	22
10	902	856	839
11	404	404	404
12	1692	1692	822
13	559	281	281
15	1021	1022	609
14&16	812	433	422
total	10386	8773	5702

river water quality model is also applicable to reservoirs and estuaries and therefore offers a wide range of applications. In our case study we also used the WASP5 model for the calculation of river impoundments because the river under investigation is impacted by a considerable number of weirs. Our simulations showed that these weirs lower denitrification because of decreased flow velocities and associated reduced advective water exchange between the water column and the hyporheic zone (Wagenschein and Rode 2008).

The suggested Meta-CANDY model for calculating nitrogen leaching from agricultural land is capable of simulating potential losses to groundwater systems. It does not, however, consider transport and transformation processes in the aquifer. Therefore the modelling system proposed here allows only the calculation of potential pressures on groundwater quality. If this is assumed to be sufficient to define agricultural management measures to reduce non-point source pollution of groundwater quality the suggested BAS*IN*FORM methodology can also be used to specify cost effective measures to improve groundwater quality. If a more detailed consideration of groundwater transformation and transport processes is needed a more comprehensive modelling approach has to be used within the suggested decision support methodology (see e.g. Hesser et al. 2010).

Taking the example of nutrient contamination of the Weisse Elster River, Sections 3 and 4 demonstrate how BAS*IN*FORM can be used to structure decision-making, to quantify the impacts of management measures, and to establish a selection of management measures in a river catchment. Compared to real decision-making in the implementation of the WFD, this example is somewhat limited in scope due to its focus on one problem area, and on the other hand it is quite elaborate with respect to the impact assessment by means of specialised modelling.

Nevertheless, the case study demonstrated the practicability and usefulness of the general decision methodology and of important elements of BAS*IN*FORM, for example the identification of the needs for action within the discrete water bodies. The transferability of BAS*IN*FORM to tackle nutrient pollution and improve hydro-morphology in other

catchments has been positively tested in a pilot study by Richter et al. (2009a, b) on the Emsbach water body in the German federal state of Hesse. BAS*IN*FORM has also been applied by public decision makers: The competent authorities in the German federal state of Thuringia used BAS*IN*FORM as a conceptual basis for the selection of measures to tackle point and non-point nutrient pollution at the federal state scale. In particular they employed it for structuring their WFD implementation process between different administrative authorities.

A key feature of BAS*IN*FORM is its flexible architecture with regard to impact assessment, knowledge sources to be incorporated, and the final decision-making stage. This flexibility makes it a suitable tool that can be adjusted to the needs of a variety of authorities responsible for the implementation of the WFD. Since the focus here is on nutrient pollution, the wide-spread problems of degraded hydro-morphology have not been at the centre of this paper. As part of a consulting project for the German federal state of Brandenburg, BAS*IN*FORM has been extended to structure the selection process for measures to improve river hydro-morphology (Klauer and Bathe 2010). Additionally, attempts are currently underway to develop a GIS tool based on the conceptual ideas of BAS*IN*FORM. The modular structure of the decision support tool as well as its workflow scheme are being used to design the architecture of a geodatabase in which all basic data, including the intermediate and final results of the different planning, modelling and evaluation tasks, will be stored.

To sum up: The ongoing complex process of implementing the WFD calls for decision tools that support the different planning steps necessary for establishing the next management plan in 2015 in a coherent but flexible manner. BAS*IN*FORM seems to be a suitable tool for addressing this challenge.

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