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Decision support for the selection of measures according to the requirements of the EU Water Framework Directive

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Abstract

One of the major scientific challenges 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 using a cost-effective combination of measures. This paper presents BAS*IN*FORM, a new decision tool for selecting cost-effective management measures that has been developed in close co-operation with a number of water authorities and tested in the 5154 km² mesoscale 4th order river Weisse Elster in central Germany. BAS*IN*FORM comprises (i) a set procedure for framing the problem concerned, including proposals for quantifying the discrepancy between the status quo and the good water status to be achieved (i.e. the need for action); (ii) modelling tools quantifying the impacts of management measures (e.g. Meta-CANDY and WASP 5); and (iii) a method for selecting cost-effective combinations of measures. A trial run of BAS*IN*FORM in the Weisse Elster catchment revealed that (i) good surface water status with respect to nutrients can not be achieved if only the "standard" actions of current water management are taken to reduce point sources (sewage treatment) and diffuse agricultural sources and (ii) the available measures for nutrient reduction will generate considerable costs. The testing of BAS*IN*FORM in the case study proved its practical applicability in the WFD implementation process.

Key words

EU Water Framework Directive, cost effectiveness, decision support system, decision making, point and non-point pollution, BAS*IN*FORM, Meta-CANDY, WASP 5

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1 Challenges of the EU Water Framework Directive

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 waters (Petersen/Klauer/Manstetten 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”. Results from the river basin district analysis conducted in 2004 in accordance with WFD Article 5 revealed an urgent need for action. It was highly 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 % there was not sufficient data to make a prognosis regarding water status (European Commission 2007, 28). WFD Articles 13 and 11 oblige the member states to establish a management plan for each river basin district by December 2009, in which they 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 to be taken into account. The latter is a particularly complex and laborious task. A methodology for the selection of a cost-effective combination of measures needs to provide both appropriate standardisations and sufficient flexibility to incorporate expert knowledge and regional specificities.

In academia as well as in professional consultancy, many decision support systems for river basin management have been proposed to comply with WFD but have rarely been used by the competent authorities. While some important conceptual work has been done (e.g. WATECO 2003, Interwies et al. 2004, Klauer et al. 2008) most of the available DSS concentrate on the technical planning aspects of measures, for instance, to improve river morphology (e.g. Elbe River-DSS – Kofalk et al. 2005, DSS-WRRL – Bartussek 2008, FLUMAGIS – Möltgen 2005)[2], to manage water quantities (e.g. Mulino-DSS – Giupponi et al. 2004), to reduce nutrient emissions (e.g. Werra River – Schumann et al. 2005) or to moderate flood risks (e.g. nofdp IDSS – Winterscheid/Hübner 2006)[3]. Reviewing existing DSS supporting the WFD, Todini and co-authors (2006, 15) 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 dimension of water resource planning*”. In particular, the issue of cost-effectiveness has been widely neglected, and the challenge to cover 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, justification of exemptions, integration of separate evaluations, etc.) has not been addressed

[2] Bartussek (2008, Chap. 4) provides an overview over DSS for selecting measures particularly aiming at improving river morphology.

[3] Evers (2005) gives an overview over DSS for the management of floods in the North Sea-region.

sufficiently. In Germany, ongoing studies of the drafts of river basin management plans show that

(i) the methodological basis for selecting measures in the ten German river basin districts is generally not very well documented, and

(ii) there are considerable differences between the plans as well as in the way they are implemented.[4]

To sum up, our assessment is that it remains unclear to date how cost-effectiveness has been taken into account by the authorities when selecting measures, and how it should be taken into account in future management plans.

The objective of this paper is

(i) to present a new River BAS*IN* *IN*FORMation and Management System (BAS*IN*-FORM) for selecting cost-effective management measures, and

(ii) to describe how the tool was tested in the 5154 km² mesoscale 4th order river Weisse Elster in central Germany in close co-operation with the relevant water authority (Environmental Ministry of the German federal state of Thuringia) (Klauer et al. 2008).[5]

BAS*IN*FORM structures the decision-making process for establishing a programme of measures by providing a management scheme describing the individual work packages as well as the necessary evaluation methods in detail. It offers a framework that can be moulded and modified for different German federal states or EU Member States. BAS*IN*FORM consists of

(i) a procedure to frame the problem 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 quantifying the impacts of management measures (e.g. Meta-CANDY, WASP 5), and

(iii) a method for selecting cost-effective combinations of measures.

The basic structure and methodological cornerstones of the decision tool 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. After a description (including a problem description) of the catchment and of its conceptualisation through BAS*IN*FORM, the modelling tools for the impact assessment are introduced. In Section 4 the results of the modelling, the cost estimates and the selection of measures are presented and discussed. The paper closes with a summary of the benefits and transferability of BAS*IN*FORM and discusses briefly the need for software solutions.

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[4] Similar investigations are underway throughout Europe but have not yet been published.

[5] BAS*IN*FORM was also applied to and tested in the water body Emsbach (320 km^2, German federal state of Hesse, Richter et al. 2009).

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 decision-making steps and integrating them into a workflow scheme. In accordance with the WFD, BAS*IN*FORM identifies water bodies (or hydrologically connected groups of water bodies) as the basic spatial unit for water management. BAS*IN*FORM structures the planning in three 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

As a first step BAS*IN*FORM (i) defines so-called *management parameters* for evaluating the current water status and (ii) translates the environmental objective *good water status* into quantitative *development targets* for every water body involved. Defining such targets is necessary in order to quantify the impact of management measures with respect to these targets, which in turn is a requirement for assessing their cost-effectiveness. The discrepancy between status quo and targets essentially determines the *need for action*.

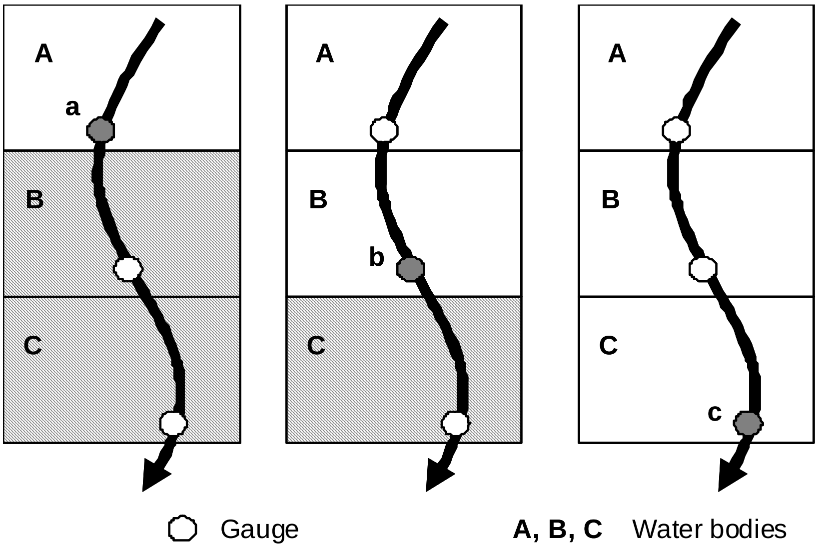
For surface water bodies, water status is composed of (i) chemical and (ii) ecological status; for ground water bodies it consists of (i) chemical and (ii) quantitative status (cf. WFD Article 2 and Annex V). The WFD defines the environmental objectives ‘good chemical status’ and ‘good quantitative status’ by reference to thresholds of certain quantitative parameters. The effects of measures on these parameters can be assessed comparatively easily. In contrast to this, the effects of measures on the ecological status of surface water are very difficult to quantify, mainly because of the complexity of the ecological system. As a support for quantifying effectiveness with respect to ecology as well, BAS*IN*FORM “translates” good ecological status into quantitative targets in such a way that (i) it creates – to best available knowledge – the conditions necessary for attaining good ecological status, and (ii) the effects of measures can be expressed in the same quantitative terms. To take one specific example, rather than measuring the ecological quality of a river by the abundance of certain organisms contained in it, critical concentrations of specific nutrients are defined instead.

In a further step, the critical concentrations of nutrients and other pollutants (e.g. organic substances) are expressed in loads. This is because the effectiveness of a management measure is most usefully quantified by establishing the reduced load of a certain pollutant rather than the reduced concentration. In order to calculate the target values (i.e. the loads to be reduced) the upstream/downstream conditions need to be taken into account by making the following assumptions (cf. Figure 1):

– *For upstream bodies*: The target value (load) should be set so that the development target (critical concentration) is met exactly.

– *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 development target (even if this is not the case in reality, i.e. the effects of upstream management measures are anticipated).

Figure 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.



The following additional aspects need to be considered in order to determine the need for action:

– *Impacts of deviating target values in upstream water bodies*: In the case of pollutants that are not quickly biodegradable (nutrients in particular), an excess or shortfall of a target in an upstream water body should be taken into account by an adequate adjustment of the targeted load reduction downstream.

– *Impacts of finalised groundwater measures*: If surface water bodies are hydraulically connected to groundwater bodies, measures for improving groundwater quality also may influence the quality of the surface water.

– *Baseline scenario*: The baseline scenario includes the effects of future developments that will most probably have an influence on 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.

Table 1: Examples for calculating the need for action for specific management parameters (hypothetical figures)

Water body xy					
Targets, status quo value					
Cause	Organic matter		Nitrogen	Phos-phorus	Morphology structure
Management parameter	BOD_5	NH_4-N	NO_3-N	P	Quality class
Development target	4 mg/l	0.3 mg/l	6 mg/l	0.3 mg/l	2
Target – character	load	load	load	load	class
Target – unit	[t/year]	[t/year]	[t/year]	[t/year]	-
Target – value	50	8.0	50	4.2	2
Status quo value	500	25.5	200	9	2-3
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	2-3 → 2

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 the relevant actors) contributes towards improving water quality. Authorities pre-select potential measures from a catalogue. Criteria for the pre-selection of measures should be their usefulness for a specific water body as well as their technical and political feasibility in principle. Because descriptions of the measures in the catalogue are very general, they must be spatially and temporally specified as well as technically concretised according to the particular situation of the water body. The result should be a list containing enough measures to meet all needs for action.

The effectiveness of the measures along with their cost 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. The results of the impact analysis are presented in an impact matrix (see Table 2).

Table 2: Impact matrix. The cost-effectiveness columns are calculated by dividing cost by effectiveness (hypothetical entries).

Impacts / Individual measures	Effectiveness			Costs	Cost-effectiveness			Other impacts		
	N [t/yr]	P [t/yr]	Organic matter (BOD_5) [t/yr]	costs [T€/yr]	N [€/t]	P [€/t]	Organic matter [€/t]	Acceptance	Uncertainty	Further impacts
Sewage treatment 1	20	1.5	40	100	5,000	66,700	2,500	Problematic	…	…
Sewage treatment 2	15	1.6	0	70	4,700	43,800	-	Okay	…	…
Canal system 1	10	0.6	20	70	7,000	116,700	3,500	Okay	…	…
Canal system 2	20	1.4	40	60	3,000	42,900	1,500	Okay	…	…
Riparian buffer strips	5	0.8	0	40	8,000	50,000	-	Problematic	…	…
Soil conservation	20	2.2	0	120	6,000	54,500	-	Okay	…	…
Total impact	**40**	**2.9**	**80**	**160**	**-**	**-**	**-**			
Management target	**60**	**4.9**	**70**							

2.3 Phase 3. Selection of measures taking cost-effectiveness into account

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

Module 2: Other groundwater pollution

Module 3: Organic matter, nutrients and pesticides in surface water

Module 4: Structure and hydromorphology of surface water

Module 5: Other surface water pollution

The right process for finding the best combination of measures is case-specific. If the measures impact on only a single target and costs are the sole criterion, then selection is more of a trivial problem of optimisation. If, however, (i) measures have an impact on multiple targets or show cross-effects (as is regularly the case), and (ii) total costs are only one evaluation criterion out of many, the decision situation is more complex. Therefore, we suggest the following two-step approach:

1. Measures from the list of potential measures are grouped into different combinations of measures, all of which could meet the need for action (using different algorithms). Cost-effectiveness is the dominant criterion.

2. The ‘best’ combination of measures is selected using total costs and further evaluation criteria, such as the positive or negative cross-effects of the measures on agriculture, the economy, transportation, recreation, their acceptance among those affected, etc.[6]

Step 1. Different algorithms exist for combining measures. One algorithm based essentially on information regarding cost-effectiveness is as follows:

The basic structure of this algorithm is a sequential addition of measures according to their cost-effectiveness (see impact matrix, Table 3) that proceeds until the needs for action for each management parameter have been accomplished. It is then necessary to check whether targets have been surpassed and 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 algorithm can be varied by considering other criteria besides cost-effectiveness when adding further measures.

Generally speaking, different algorithms will lead to different combinations of measures. The process of creating combinations of measures should not rely on formal (optimisation) algorithms only, but should be based also on the knowledge and intuition of experts. 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 the three best combinations should be selected and, finally, the “best” overall combination identified. One important criterion in this stage of the selection is that of total costs. Further criteria should also be included. This step can be supported by a multicriteria analysis (e.g. a scoring approach) if the decision problem is complex (e.g. Vincke 1992, Roy 1996, Klauer et al. 2006).

The procedure presented above for identifying the need for action and the “best” combinations of measures to meet them has been tested for the problem of nutrient pollution in the Weisse Elster River in central Germany.

[6] The exact characteristics of the evaluation criteria guiding the selection of measures are defined explicitly or implicitly by the staff of the authorities themselves. Formal multicriteria analysis tools, such as scoring or PROMETHEE, may be applied (see e.g. Roy 1985, Vincke 1992 for an overview) but alternative approaches may also be appropriate (see Klauer et al. 2007, 29).

3 Testing BAS*IN*FORM in the Weisse Elster River case study

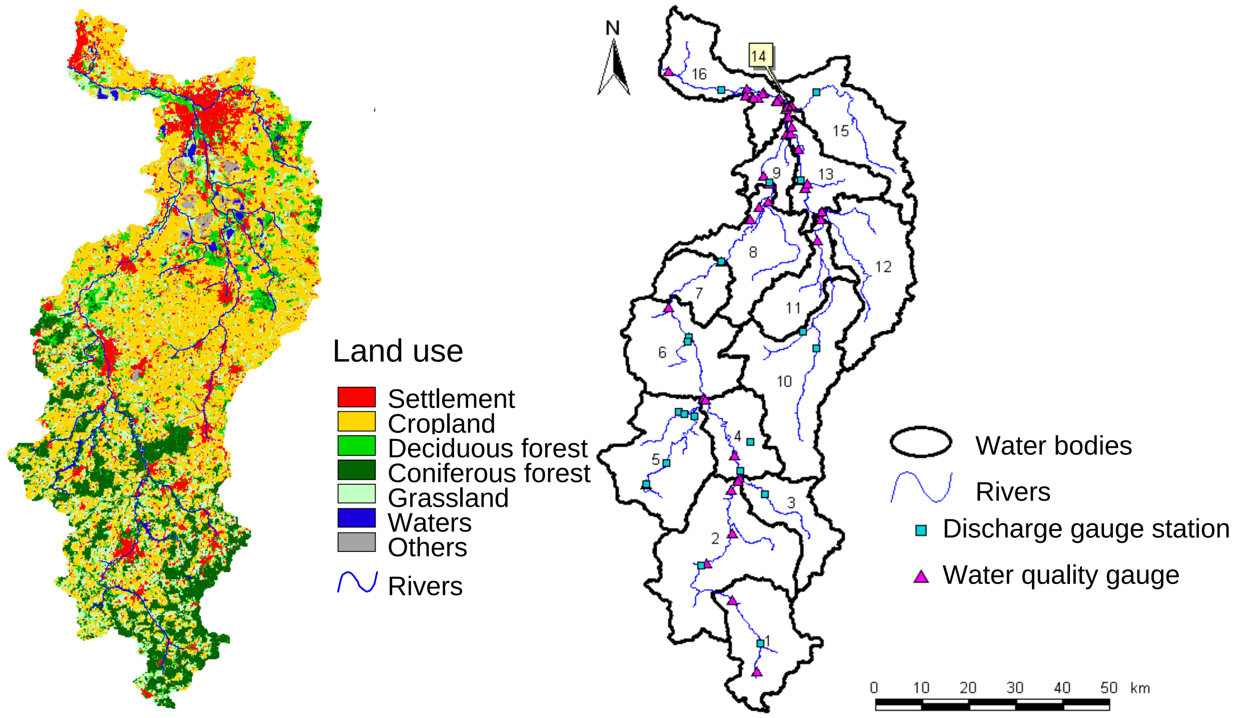
3.1 Catchment characteristics

The Weisse Elster river basin is a subcatchment of the Saale River, which is the second largest tributary of the Elbe River. The catchment area is 5154 km^2 (gauging station Oberthau) and is mainly situated in the German federal states of Saxony, Thuringia and Saxony-Anhalt, originating in the Ore Mountains in the Czech Republic. The river is 250 km long and has a mean discharge of 26 m^3/s (gauging station Oberthau). 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 Figure 2). The upper part of the basin is mountainous and is characterised 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 1000 mm in the southern part (mountains). Annual runoff varies between approximately 50 mm and 600 mm.

Figure 2: a) Land use in the Weisse Elster River basin. Classification on the basis of Landsat ETM Data of 1999 (Source: Klauer et al. 2008, 391)
b) Water bodies and gauging stations of the Weisse Elster River Basin[7]

[7] The shapes of the water bodies in the case study were already established before the water bodies were officially defined by the German states. For this reason, the official water bodies and the ones in the case study differ with respect to their number and shapes.



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. The proportion of conventional farming is 98 % and that of organic farming 2 % of 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 the major cities located in the catchment. The area south west of Leipzig is dominated by active and reclaimed open pit mines.

Meteorological data were made available from 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 made available 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. Data on water extraction and discharge to the river were mostly taken as permitted values.

A digital elevation model was available 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.

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.

3.2 Description of 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. The main problems are nutrients (N and P) with high ammonia concentrations in some river reaches. For this reason, nitrogen and phosphorus in the surface water have been selected for the case study as the main management parameters for ecological status. In fact, alongside problems of degraded river morphology and poor passability for migrating aquatic organisms, high nutrient loads are the main reason for river water bodies in Germany and in central Europe generally being at risk of failing to achieve good ecological status (Borchardt et al. 2005, European Commission 2007, 33). Thus this case study can be seen as representative for a wide range of water management problems in central Europe.

As a consequence of the political independence of the German federal states, different development targets have been set for nutrient contamination in surface waters: While Saxony and Saxony-Anhalt use 3 mg N_{tot}/l and 0.15 mg P_{tot}/l as threshold concentrations (corresponding to LAWA-Water Quality Class 2, abbreviated to WQC 2), Thuringia takes 6 mg N_{tot}/l , 0.3 mg P_{tot}/l as a development target (corresponding to LAWA-water quality class 2-3, abbreviated to WQC 2-3) (see LAWA 2004). In our case study we investigate both pairs of thresholds, WQC 2 as well as WQC 2-3, as scenarios.

Nutrient concentrations in the Weisse Elster River and its major tributaries (given as 90-percentiles for the year 2001) range between 6.1 and 13.0 mg N/l and 0.14 and 0.74 mg P/l. Diffuse discharges, originating mainly from agriculture, have been estimated to contribute to the overall nutrient load by 84 % (nitrogen) and 65 % (phosphorus) (Behrendt 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 97 % and more 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).

The nutrient development targets constitute the basis for determining water body-specific target values and the need for action, as shown in Table 3 (see also Section 2.1). The calculation is shown using downstream water bodies 14 & 16 and their water quality gauging station “Ammendorf” as an example (see Figure 2).[8] The station is located near the outlet of the Weisse Elster River. The total annual nitrate-N load at Ammendorf was 4273 t/a.[9] The target concentration of 3 mgN/l for WQC 2 corresponds to a nitrate-N load of 1550 t/a. The nitrate-N load produced in water body Nos. 14 & 16 is 361 t/a and the target value is 88 t/a (determined as the load of nitrate-N that originates from this water body only and meets the target concentration, on the assumption that all upstream water bodies meet their target thresholds). This indicates a need for a nitrate-N reduction (need for action) of 273 t/a within this water body. Table 3 lists the needs for action for all water bodies in the Weisse Elster Basin and for both pairs of water quality classes.

Table 3: Needs for action for all 16 water bodies with respect to nitrate and phosphorus and for water quality class 2 and 2-3 (in 2001).

Water Body	**Nitrate** [t/a]			**Phosphorus** [t/a]		
No.	**Quality class 2**	**Quality class 2-3**	**Maximal reduction**	**Quality class 2**	**Quality class 2-3**	**Maximal reduction**
1	176	21.8	59.3	1.80	0.0	1.99
2	336	192.1	173.0	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	157.1	120.0	1.90	0.0	3.47
6	239	162.0	103.7	7.10	3.5	2.05
7	335	191,3	64.5	10.00	7.1	2.10
8	210	0.0	124.0	5.00	2.2	4.00
9	17	11.2	27.9	0.00	0.0	0.75
10	292	202.9	268.8	7.80	4.3	8.91
11	118	81.9	77.0	3.20	1.7	2.94
12	213	147.8	157.2	5.70	3.1	4.04
13	82	57.2	124.0	2.20	1.2	0.46
15	136	93.6	135.4	10.70	6.0	3.74
14 & 16	273	222.5	332.0	15.80	9.8	2.14

3.2.2 Pre-selection of potential management measures

A detailed literature review led to the following list of potential measures for mitigating the nutrient contamination in the Weisse Elster River. The list has been discussed with experts from the authorities responsible for managing the Weisse Elster. According to these experts the list is “complete” in the sense that it comprises virtually all potentially effective and practicable measures for mitigating eutrophication (see Schiller et al. 2008, 96-98):

[8] Water Body No. 14 covers the area of Leipzig. In the case study water bodies 14 and 16 have been merged and considered as a single water body.

[9] This and the following figures relate to the year 2001.

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

   Integrated farming is characterised by more efficient use of fertiliser at an overall lower level (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 to 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 morphology.

With the exception of the exclusions mentioned, all management measures are compatible with each other. The impacts of these potential management measures have been estimated using various modelling tools.

3.3 Modelling tools for quantifying the impacts of management measures

Three different modelling approaches were used to simulate soil erosion and diffuse phosphorus inputs, nitrate leaching from the soil zone, and in-stream nutrient retention in the main reach of the Weisse Elster River.

3.3.1 Meta-CANDY

Nitrogen loading of subsurface flows is simulated using a meta model (Schmidt et al. 2008, Hesser et al. 2008, Hesser et al. 2010). This is a regression model using only a few important factors, which control nitrate leaching from agricultural land. The quantification of these factors is based on simulations of the agroecosystem model CANDY (Franko et al. 1995). This meta model calculates nitrogen leaching from the soil zone for so-called nitrate response units (NRU). NRUs are defined as areas which behave similarly in terms of nitrogen leaching due to their comparable soil, land use and climate conditions. These NRUs were aggregated from hydrological response units (HRUs, Flügel 1995), because not all HRU attributes (such as elevation, slope, exposition, geology) were used for nitrogen leaching calculations with the meta model.

Within the relatively homogeneous NRUs, the simulated farming systems were used to carry out a multiple regression analysis. The analysis looked at nitrate leaching from the soil zone (N_{leach}) as a dependent variable along with independent variables (I_i) that were significantly correlated with N_{leach}.

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

with a_0 = intercept and a_i = regression coefficient.

Livestock and the share of cereals in crop rotation were identified as the most significant parameters. The spatial availability of these parameters in common agricultural statistics is a further condition for applying the model at larger scales. The regression model (meta model) calculates average as well as normal distributions of N-leaching from the soil zone (Schmidt et al. 2008). The meta model allows a large number of model runs and is, therefore, able to account for uncertain fertiliser application. The model simulates nitrogen leaching for every NRU in the catchment using a probability distribution function for fertiliser input.

For the Weisse Elster catchment the parameters of the meta model were calculated using the CANDY model based on 125 virtual crop rotations for selected most important soil types. Meta model calculations were 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.

3.3.2 Soil erosion and P-input simulation

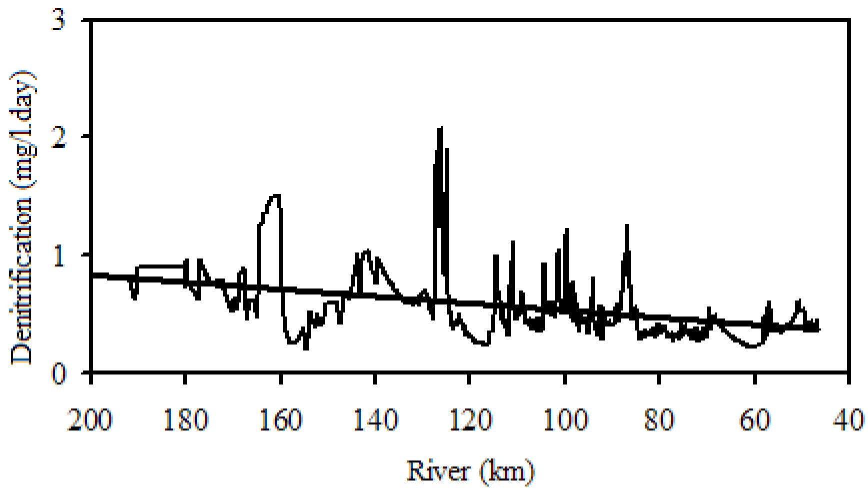
Soil erosion and phosphorus input simulations are based on the method of the Erosion 2D/3D model combined with a Monte Carlo approach for disaggregating spatially uncertain input data. The basic idea of the model is the assumption that the erosive impact of overland flow and droplets is proportional 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 devised by 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. Soil physical and crop rotation data were defined according to Schmidt et al. (1996). The P-transport model was used to simulate baseline 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, calculated soil and P losses are conservative estimates.

3.3.3 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; EUTRO can be used to model oxygen depletion, eutrophication, and nutrient enrichment in the river; and TOXI simulates sediment transport and the fate of toxic inorganic and organic chemicals. In this study a modified version of DYNHYD (Warwick 1999) was used which allows for consideration of weirs. An extended version of EUTRO was also applied (Shanahan and Alam 2001), which consists of nine model variables: biomass of phytoplankton (PHYT), biomass of periphyton (PERI), dissolved oxygen (DO), biochemical oxygen demand (BOD), ammonia nitrogen (NH_4), nitrate nitrogen (NO_3), organic nitrogen (ON), phosphate (PO_4) and organic phosphorus (OP). 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. Additional information on the latest model modification can be found in Wagenschein and Rode (2008).

The Weisse Elster River water quality model consists of 872 river cross sections. 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. An uncertainty analysis based on the Monte Carlo approach (Reichert and Vanrolleghem 2001) was carried out for the calibrated model (using PEST, Doherty 2004). A detailed sensitivity analysis can be found in Wagenschein (2006). During the calibration process, PEST can be used to quantify the 95 %-confidence interval. 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 Figure 3). For the main course (Strassberg-Großzschocher), simulated mean annual retention was 19 % of total nitrogen load (Rode et al. 2008).

Figure 3: Denitrification rate along the main course of the Weisse Elster River in summertime conditions.



3.3.4 Calculation of costs

The calculation of costs and the effects of the management measures for improving waste water treatment are closely interrelated and are therefore addressed together. In the Weisse Elster basin there are 196 sewage treatment plants of different sizes and technical standards. In an initial screening stage so-called priority plants were identified. Priority plants are those 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 (i) on the size and (ii) on the level of nutrient elimination already reached. Using cost data from the literature as well as from interviews with companies operating treatment plants, a calculation scheme was developed that attaches to each plant its potential nutrient reduction as well as the related costs (for details see Breuer and Neubert 2008, 182-199).

The costs involved in changing the system of farming, i.e. 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 through to 2e (see above) for the specific situation (e.g. arable land-grassland relation) of the various water bodies (see Breuer and Neubert 2008, 176-182).

The costs of introducing conservation farming are calculated according to subsidy rates offered by the agricultural programmes of the German federal states. These subsidy rates reflect (at least approximately) the factual costs and are assumed to amount to 25 euros/(ha·a).

The costs of the renaturation measures could not been estimated in the case study because the information required was not available. However, they are expected to be very high.

4 Results and discussion

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 for water bodies Nos. 8 and 13 are summarised by way of an example in Tables 4 and 5.

Table 4: Results of the impact assessment for water body No. 8

Water body No. 8					
Effects **Measures**	**N** [t/a]	**P** [t/a]	**Costs** [€/a]	**N cost-effective-ness** [€/t]	**P cost-effective-ness** [€/t]
Seawage treatment Lucka (N-elim.)	0.66		3,417	5,177	
Seawage treatment Lucka (P-elim.)		0.30	2,284		7,612
100% integrated farming	96.0		73,256	763	
15% organic 85% conventional f.	25.0		332,190	13,302	
15% organic 85% integrated farm.	110.0		437,897	3,982	
30% organic 70% integrated farm.	124.0		802,538	6,473	
30% organic 70% conventional f.	54.0		715,485	13,256	
Soil conservation		3.70	483,208		130,561
Renaturation	8.3				
Max. reduction / costs	132.9	4.00	1,291,446		
Need for action WQC 2 / costs	210.0	5.00	1,291,446		
Need for action WQC 2-3 / costs	0.0	2.20	483,208		

Table 4 shows that in water body No. 8 the maximal reduction can be achieved if the shaded measures are implemented simultaneously: The sewage treatment plant Lucka is improved with respect to (i) its N- and (ii) its P-elimination capacities, (iii) the current share of organic farming is increased to 30 % and the share of integrated farming is increased to 70 % of total cropland, (iv) conservation farming is implemented area-wide, and renaturation measures are implemented. However, all these measures are not sufficient to achieve WQC 2 for nitrogen as well as for P because the need for action surpasses the maximal reduction. If one lowers the standard to WQC 2-3, the status quo already reaches 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 costs 483,208 €/a.

In this example the number of alternatives and the room for manoeuvre is so small that the costs and cost-effectiveness of the measures are irrelevant for the selection of measures – the selection problem is rather trivial. Table 5 shows the results for water body No. 13 where, by contrast, cost considerations were helpful in selecting the measures.

Tables 5: Results of the impact assessment for water body No. 13

Water body No. 13					
Effects / **Measures**	**N-reduction** [t/a]	**P-reduction** [t/a]	**Costs** [€/a]	**N-cost-effective-ness** [€/t]	**P-cost-effective-ness** [€/t]
Seawage treatment Espenhain	30.1		34,500	1,146	
Seawage treatment Markkleeberg	41.9		10,648	254	
100% integrated farming	42.0		33,225	791	
15% organic 85% conventional f.	10.0		112,114	11,207	
15% organic 85% integrated farm.	47.0		155,017	3,298	
30% organic 70% integrated farm.	52.0		276,808	5,323	
30% organic 70% conventional f.	21.0		241,477	11,497	
Soil-conserving farming		0.46	236,828		513,726
Maximal reduction/costs	124.0	0.46	558,784		
Need for action (WQC 2)	82.0	2.20			
Need for action (WQC 2-3)	57.2	1.20			

To meet the need for action (WQC 2) for nitrate (i) the sewage treatment plant Markkleeberg should be improved and (ii) integrated farming should be implemented on all cropland. Alternatively the share of organic farming could have been raised to 15 % or 30 % and the share of integrated farming to 85 % or 70 % (dark grey rows). However, these alternatives are more expensive (in absolute terms) and less cost-effective. With respect to phosphorus the targets can be reached neither for WQC 2 nor 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 considered are not sufficient to meet the objectives of the WFD. It should be kept in mind that the list includes virtually 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. Our analysis shows that there is an urgent need to develop innovative, more effective measures to reduce nutrient pollution.

Table 6: Overview over failures to achieve the target values even if all potential measures are simultaneously taken 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 action	11	9	5	5
# of simultaneous failures of N and P	13		8	

Table 7 shows the costs of the most cost-effective combinations of measures for achieving (i) maximal 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: Costs of the most cost-effective combination to reach (i) maximal reduction, (ii) WQC 2 and (iii) WQC 2–3

Costs for Water body No.	maximal reduction [1000 €/a]	WQC 2 [1000 €/a]	WQC 2–3 [1000 €/a]
1	370.9	368.7	24.1
2	709.5	699.3	699.5
3	259.8	259.8	123.4
4	589.3	33.8	0.0
5	460.4	460.4	23.2
6	279.6	279.6	279.6
7	713.6	713.6	713.6
8	1.291.4	1.291.4	483.2
9	321.1	22,4	22.4
10	902.2	856.1	839.4
11	403.9	403.9	403.9
12	1.691.5	1.691.5	822.2
13	558.7	236.9	236.9
15	1.021.4	1.021.5	608.6
14&16	811.6	432.8	421.8
Sum	10.385.0	8.771.6	5.701.6

Uncertainties and assumptions notwithstanding, Table 7 gives an idea of the dimensions of the costs of implementing the WFD in the Weisse Elster Basin: a reduction in nutrient contamination to the level of quality class 2 costs approx. 9 million euros every year; even then, the development target is missed in 13 water bodies. If only quality class 2-3 is envisaged, the costs fall to 6 million euros per year and the development target is missed in only 8 water bodies.

5 Conclusion

The implementation of the WFD is a major task for the water authorities in all EU member states. Despite the draft of the first management plans having already been published, experts expect there to be a growing demand for decision tools that support the different planning steps necessary for establishing the next management plan in 2015 in a coherent manner. BAS*IN*FORM supports the competent authorities in establishing the programme of measures by structuring the different work and decision steps needed and integrating them into a workflow scheme. The Weisse Elster Basin case study proved the practicability of important elements of BAS*IN*FORM, in particular the identification of the need for action.

BAS*IN*FORM has already been used by the competent authorities in the German federal state Thuringia as a conceptual basis for the selection of measures to combat diffuse nutrient pollution. The German federal state of Brandenburg is currently investigating whether BAS*IN*FORM can be extended to structure the selection process for measures to improve the hydromorphology of river water bodies. Parallel work is currently ongoing on the development of a GIS tool building on the conceptual ideas of BAS*IN*FORM. The modular structure of the decision support tool as well as its work-flow scheme are being used to design an architecture supporting a geodatabase in which all basic data, including the intermediate and final results of the different planning, modelling and evaluation tasks, will be stored.

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