Decision support systems (DSS) for Water management

Generic guideline

Target audience: Scientists, Water managers, Water companies & authorities

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1 INTRODUCTION

Water managers have to make decisions on the implementation of measures to improve the status of the aquatic ecosystem. The available information on innovative rehabilitation technologies, and more specifically groundwater remediation technologies, is complex and therefore difficult to incorporate in programmes of measures. Challenges related to the implementation of groundwater remediation technologies at the river basin or groundwater body scale are (1) the upscaling from field to catchment scale, (1) the interaction between groundwater and surface water, (3) the time delay between action and effect due to the attenuation processes, and (4) the assessment of the effects of multiple measures within one catchment. Often technologies act on just a specific set of chemicals whereas multiple chemicals end up in the groundwater or surface water and may cause adverse effects to ecology. Ecological effects of mixtures of chemicals arriving at different time periods in the catchment are difficult to assess.

The policy context for decision support systems in water management in Europe is largely defined by the European Water Framework Directive (WFD). This Directive, adopted in 2000, sets ambitious objectives to meet good status of all waters by 2015. To ensure that this goal will be met, member states must publish river basin management plans for each river basin district detailing the status we are in now, will be in in the future if we do nothing (BAU) and how this status will evolve towards 2015, 2021 and 2027 if we implement specific combinations of measures. The WFD also explicitly mentions the concept of a water body as the preferred scale, which can be quite detailed, depending on the Member State and the specific river basin. Member States need to report the amount of water bodies reaching good status now and in the future. The WFD requires that these plans include cost-effective programmes of measures. How to identify which measures are cost-effective is also an important target for a DSS. Developing web applications instead of desktop models also adds some additional complexities, especially if end users need to perform model simulations. Models need to be sufficiently simplified, have short calculation times and easy to operate to allow end users to perform so called “on the fly” calculations.

This guideline describes a generic framework to set up DSS for the evaluation of rehabilitation technologies. The document was composed in the frame of the FP7 project AQUAREHAB (GA 226565), and comprises outcomes and lessons learned during this project.

DISCLAIMER: Although the information described in this document is believed to be reliable and accurate, the guideline does not offer warranties of any kind.

2 GENERAL BACKGROUND

2.1 INTRODUCTION

Decision support systems (DSS) are computer-based information systems including knowledge based systems that support decision making activities (Wikipedia, 2013). DSS in the water management sector usually consist of simulation models, and/or of techniques and methods for decision analysis, recently extended to include the support to participatory processes. Therefore, a DSS typically integrates multi-source geographically referenced data and data management systems, a variety of models and elaboration procedures within a customized user interface. Emphasis is given to hydrologic models accompanied by environmental assessment and/or socio-economic evaluation. The models include both those aimed at reconstructing and simulating the physical reality, and those constructed to manage divergent objectives and to find a compromise among the expectations of different actors in a participatory process (Giupponi et al. 2007). DSS are developed to assist decision makers to address semi-structured - or ill-defined - tasks in a specific decision domain. They provide support of a formal type by allowing decision makers to access and use data and appropriate analytic models (El-Najdawi and Stylianou 1993). The terms 'semi-structured' and 'appropriate' in this definition refer to the fact that Decision Support Systems are typically applied to find answers for problems that, due to their specific nature and complexity lack an unambiguous solution method. Typically three essential components can be distinguished (Calewaert et al., 2007): end users, knowledge and technology. DSS gained attraction in the planning and environmental management community in the early 1990s when many initiatives aimed at developing DSS began in a variety of domains, including integrated coastal zone management (Westmacott, 2001; Engelen et al; 2003; De Kok et al., 2009), river basin management (Oxley et al., 2004; Van Delden et al, 2007) and urban and regional planning (Geertman and Stillwell, 2003). Only few applications survived beyond their development stage and provided support for real decision making and formulation of planning strategies. This moderated the belief in their feasibility and applicability from 2005 onwards. Failure is often attributed to a lack of transparency, inflexibility and their focus on technical capabilities rather than on the real problems (Uran and Janssen, 2003; Vonk et al., 2005; Geertman, 2006). Substantial stakeholder involvement, a combination of qualitative and quantitative modeling techniques which are tuned to the different phases of policy analysis (problem definition, inventory of solutions, analysis of the system, and evaluation of management options), a proper incorporation of local knowledge, and flexibility are essential prerequisites for these management tools to be effective and survive the projects' lifetime. The lessons learnt have also shown the importance of end-user involvement right from the start of the development process (Engelen et al., 2003; McIntosh et al., 2007, Van Esch et al., 2009). Among knowledge-related challenges of DSS, Van Kouwen et al. (2008) name the need for better handling of incomplete and uncertain knowledge, a better representation and visualization of uncertainty, and the fact that DSS should support policy-optimization in addition to evaluation of different options. A growing importance of the complexity of the problems addressed as well as the need for interdisciplinary cooperation is also reflected by the type of applications reported in dedicated journals such as Environmental Modelling & Software, Water Resources Management, Integrated Assessment Journal, Environmental Modelling & Assessment and Decision Support Systems.

Decision Support Systems are potentially also of use in integrated water management and more specifically for the implementation of the European Water Framework Directive (WFD). This Directive, adopted in 2000, sets ambitious objectives to meet good status of all waters by 2015. To ensure that this goal will be met, member states must publish river basin management plans for each river basin district. The WFD requires that these plans include cost-effective programmes of measures. Yet, the assessment of the cost-effectiveness of emission reduction measures has been one of the bottlenecks in designing the RBMP's (Cools et al., 2010). Despite the simplicity of the concept of cost-effectiveness (e.g. explained in Brouwer and De Blois, 2008), the availability of European Guidance documents (WATECO, 2002 and Interwies et al., 2004) and numerous publications on cost-effectiveness analysis for surface water quality improvements (e.g. Schleich et al., 1997, Lise and Van Der Veeren, 2002, Arabi et al., 2006, Fröschl et al., 2008), the development of a cost-effective Programme of Measures for the RBMPs has not been straightforward. An important reason for this is the requirement for multi-scale and multi-disciplinary inputs from environmental scientists (effectiveness), economists (costs), engineers (technical details of measures) and river basin managers (targets and policy priorities). It becomes evident that this is a challenging task which needs support from appropriate information systems and modelling tools that are able to cope with the complexity of the water system and planning process (Hattermann and Kundzewicz, 2010). However, despite their availability, up until now modelling tools have only been used to a limited extent in many river basins for the development of the programme of measures. Challenges are to visualise and make available complex model results to the enduser by providing collaborative management tools.

2.2 DSS FOR WATER QUALITY REHABILITATION TECHNOLOGIES

This guideline focuses on the requirements, setup and use of a DSS to decide on the implementation of various water rehabilitation technologies and mitigation measures (including groundwater remediation, waste water treatment, and reduction of diffuse pollution) aiming at water quality improvement at the river basin scale. An important condition for the implementation is the compliance with existing regulatory frameworks, definitions, and associated water quality and ecological criteria. At the EU level, this is driven by the Water Framework Directive with its Groundwater daughter directive. Related directives are the Nitrate Directive, Pesticide regulation Directive, Priority Pollutants Directive, Urban Waste Water Directive, Drinking Water directive and the REACH regulation. For an overview of relevant regulatory frameworks, see Annex 2 to this guideline. In this context, water managers have to make decisions on the implementation of measures to improve the actual status of the aquatic ecosystem. The available information on rehabilitation technologies, and more specifically groundwater technologies or management of diffuse pollution, is complex and difficult to incorporate in programmes of measures. Specific technical challenges related to the implementation of groundwater remediation technologies at the river basin scale are (1) scaling up the performance of the technology from the field scale to the catchment scale, (2) incorporating the specific interaction between groundwater and surface water quality, (3) accounting for the time delay between the rehabilitation action and its effect on water quality downstream due to the pollutant attenuation processes along the pathway from source to receptor, and (4) the assessment of the effects of multiple measures within one catchment. Furthermore, often groundwater remediation technologies act on just a specific set of chemicals whereas multiple chemicals may end up in the groundwater or surface water and may cause adverse effects to the ecosystem. Ecological effects of mixtures of chemicals arriving at different time periods in the catchment are difficult to assess.

The aforementioned challenges also pose additional requirements to the integration of cost effectiveness analysis and water modelling tools:

- Multi-pollutant and multi-location optimization: measures often have multiple impacts (for example nutrients and pesticides) on multiple locations (upstream-downstream effects). Some measures might not be cost-effective on single parameters but can be cost-effective when reaching good water status as a whole.
- Time-scale and dynamic impacts: the way measures influence water status differs. Measures aimed at reducing point sources will have an immediate impact whereas measures aimed at reducing diffuse sources (for example agricultural losses, groundwater pollution) will take years and decades to be at full impact. Seasonal

differences and the way targets are set (yearly vs. summer averages, 90-percentile vs. average values) also influence the cost-effectiveness.

- Groundwater-surface water interaction: most published river basin management focus on surface water status. The impact of measures on groundwater status and wider long-term consequences on surface water status due to interactions are rarely mentioned.

3 DSS IMPLEMENTATION

Figure 1 shows the steps needed to set up a DSS. Starting from regulatory and end user requirements, the objectives and scope of the DSS are defined. From there on, the components of the DSS can be defined: fate models to be used, databases and visualisation tools. Scenarios that are composed of rehabilitation measures, need to be defined among stakeholders before the implementation of the software.

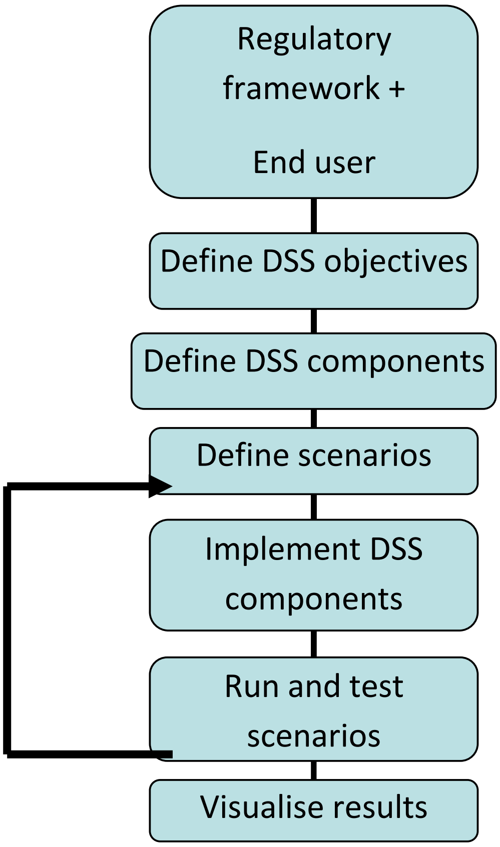


Figure 1: Overview of steps in setting up a web-based DSS for evaluating rehabilitation measures

Based on the objectives and the type of scenarios, it can also be defined which decision criteria are important to incorporate in the DSS and which model components need to be incorporated. A loop is initiated in which the components are programmed in the software tool and the scenarios are tested and evaluated by experts and stakeholders. Whenever the tool fails to reproduce reasonable results or display scenarios in a user friendly way, the loop is initiated until the test runs are satisfying and the results can be visualised. The user interface is an important tool for interaction with end users and screenshots of DSS input and output can be used to communicate on end user requirements in an early stage of the development process. In this section, the generic development process of a DSS will be elaborated and illustrated with examples from the Aquarehab project.

3.1 POLICY AND END-USER REQUIREMENTS

3.1.1 Policy context

The policy context for decision support systems in water management in Europe is largely defined by the European Water Framework Directive (WFD). This Directive, adopted in 2000, sets ambitious objectives to meet good status of all waters by 2015. To ensure that this goal will be met, member states must publish river basin management plans for each river basin district detailing the status we are in now, will be in in the future if we do nothing (BAU) and how this status will evolve towards 2015, 2021 and 2027 if we implement specific combinations of measures. The WFD also explicitly mentions the concept of a water body as the preferred scale, which can be quite detailed, depending on the Member State and the specific river basin. Member States need to report the amount of water bodies reaching good status now and in the future. The WFD requires that these plans include cost-effective programmes of measures. How to identify which measures are cost-effective is also an important target for a DSS. A last important aspect in the WFD is the concept of exemptions. These exemptions can be leading to an extension of the deadline by two times six years or the achievement of less stringent objectives. Possible motives for exemptions are natural conditions (it may take time for the conditions necessary to support good ecological status to be restored), technical feasibility (no technical solution is available, it takes longer to fix the problem than there is time available or there is no information on the cause of the problem) and disproportionate costs (CIS, 2008). Decision support systems can help to identify where exemptions are required and how they can be motivated.

The European Groundwater Directive (2006/118/EC) states that, where necessary, Member States should assess possible threats for human health and the environment of plumes resulting from point sources. In addition, Member States are establishing national inventories of contaminated sites in accordance with the proposal for the Soil Framework Directive (COM (2006) 232). During the Aquarehab project, a policy gap was detected between the site management and the regional management of groundwater resources. Therefore, the project examined how data from national inventories can be coupled to readily available data in Geographical Information Systems (GIS) to provide valuable information for risk assessment and trend analysis at a larger management scale.

3.1.2 End user requirements

At the start of developing a DSS, end user requirements need to be defined (Broekx et al., 2012). More specifically in water management, expert groups, responsible for setting up programs of measures for specific water aspects, and river basin managers, responsible for setting up management plans on local and regional scales, need to be consulted.

We here report the main results of an end user requirements analysis performed in the Flemish Region of Belgium in 2009 for decision support systems in integrated water management (Broekx et al., 2012).

A first user requirement is to provide information in a structured way in order to contribute to decision making. This includes a representation of the state of the water system, the pressures coming from different economic sectors and the potential impact of measures. Data on measures need to be transparent, detailed, include uncertainty margins and include the source of information. Boundary conditions for applying certain measures are also considered as important information.

The user requirement analysis also focused on the methodology of the economic analysis since this will largely affect the outcome of the DSS. Whenever possible, a cost effectiveness analysis needs to be included. If no quantitative data exist, qualitative information was also considered useful. Marginal cost curves were considered an informative instrument to get a better view on cost effectiveness analysis in general. Extensive, multi-objective optimization algorithms are less desired by potential end users. Reasons for this are twofold. On the one hand, optimal solutions do not exist in many cases, as not enough technical reduction potential exists to realize all targets. Consequently, multi-objective optimization problems cannot be solved or only be solved by reducing targets to the maximum potential, which in the end leads to a selection of all measures and to relatively little insight in the cost effectiveness of individual measures. On the other hand, a cost effectiveness analysis has difficulties in dealing with qualitative information as public acceptance and implementation complexity. End users see more added value in scenario development on a trial & error basis, as the amount of potential measures is not very large (< 100), especially on a local scale. The ability to easily compose and exchange scenarios across different water aspects was considered very interesting.

Besides a cost effectiveness analysis, also a disproportionate cost analysis was considered important as a possible motive for exemptions on reaching the good water status. Though widely discussed and explicitly mentioned in the European Water Framework Directive, no widely accepted methodologies exist on how to determine whether costs are disproportionate. Potential solutions here are an indicator approach, combining both affordability assessments, benchmarking indicators and a cost benefit analysis (Broekx et al., 2012).

Actualisation of data is another big challenge. Status and pressures of water systems evolve. Measures are implemented continuously. This means that on frequent points in time (yearly) data need to be updated. Also, end users need to be able to put in more accurate information of local circumstances where available.

Making data and models publicly available through web applications is also of added value for data quality. This makes it indeed possible for a larger amount of people to check data quality and possibly improve it. This does not only improve the trustworthiness of models but also can lead to drastic improvements and efficiency gains compared to typical desktop modelling.

However, building web applications requires additional time investments. To be sure that these investments have an added value, it needs to be clear that a sufficiently large amount of potential users exist and can be identified. Also, defining end user requirements is an essential first step. Developing web applications instead of desktop models also adds some additional complexities, especially if end users need to perform model simulations. Models need to be sufficiently simplified, have short calculation times and easy to operate to allow end users to perform so called “on the fly” calculations. Limitations on calculation time and model complexity are much more extensive compared to desktop modelling. E.g. the use of optimization algorithms in web based applications is difficult. If models are too complex or require too high calculation times, possible solutions are to limit applications to consultation of pre-calculated model results or to offer the possibility to schedule model simulations, where end users request simulations and receive results 1 day or 1 week later. This is certainly required if spatially explicit calculations are required for larger areas.

The potential end use for tools which assess groundwater pollution on larger scales, will probably be less related to soil management. Typically, authorities working on soil pollution are focused on individual site management, and less on a larger scale. Decision support can be provided for soil pollution to be tackled on larger scales (large plumes, mega-sites), but still at this scale more site specific modelling will be required to provide decision support. More potential end users can be expected in water management.

3.1.3 Potential scope

The DSS ultimately brings together information on pollution pathways and transport, water status (ecological and chemical), rehabilitation measures and their costs, and aggregates this information at the relevant management scale. Right at the start, it is important to delineate the scope of the DSS, agreed among stakeholders in the participatory process. The following elements are important in this context:

- What is the spatial scale for the DSS ?
- How is water status evaluated? Which water aspects do we take into account? Which type of pollution is considered in the DSS ?
- Which rehabilitation measures are considered ?
- How are economic aspects of rehabilitation measures taken into account ?

3.1.3.1 Spatial scales

Groundwater body: WFD unit of groundwater at for which status is monitored and for which effects of (a program of) measures are evaluated (see example Figure 2).

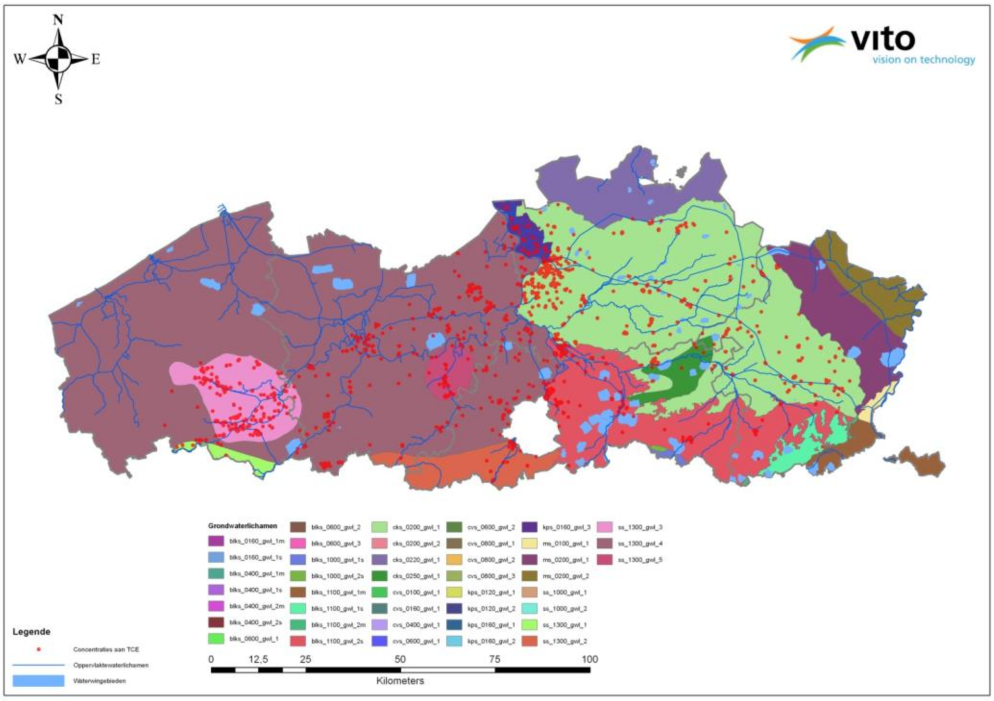


Figure 2: Example of the scale of a groundwater body (coloured entities) in comparison with TCE (tri-chloro-ethene) contaminated sites in Flanders (red dots).

Surface water body: WFD unit of surface water at which effects of (a program of) measures are evaluated (example see Figure 3).

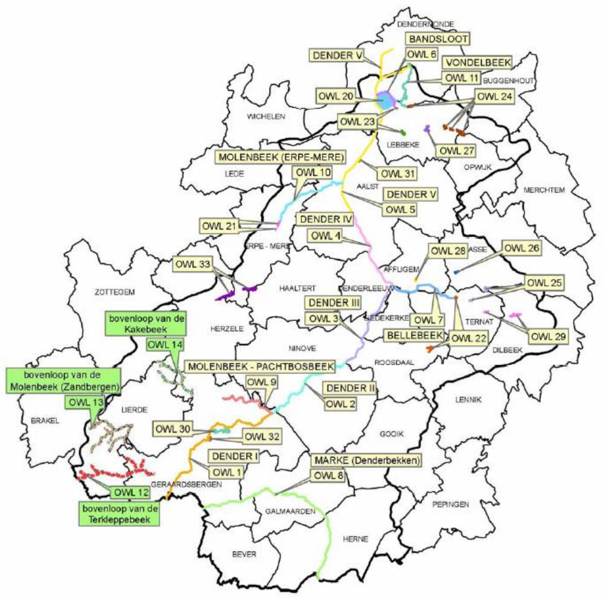


Figure 3: Example of a surface water body (coloured lines) for the subbasin of the Dender in Flanders

Sub-Basin/River basin: management unit (groundwater and surface water) for which a set of measures are ranked (prioritized) in a DSS (10^2-10^4 km^2)

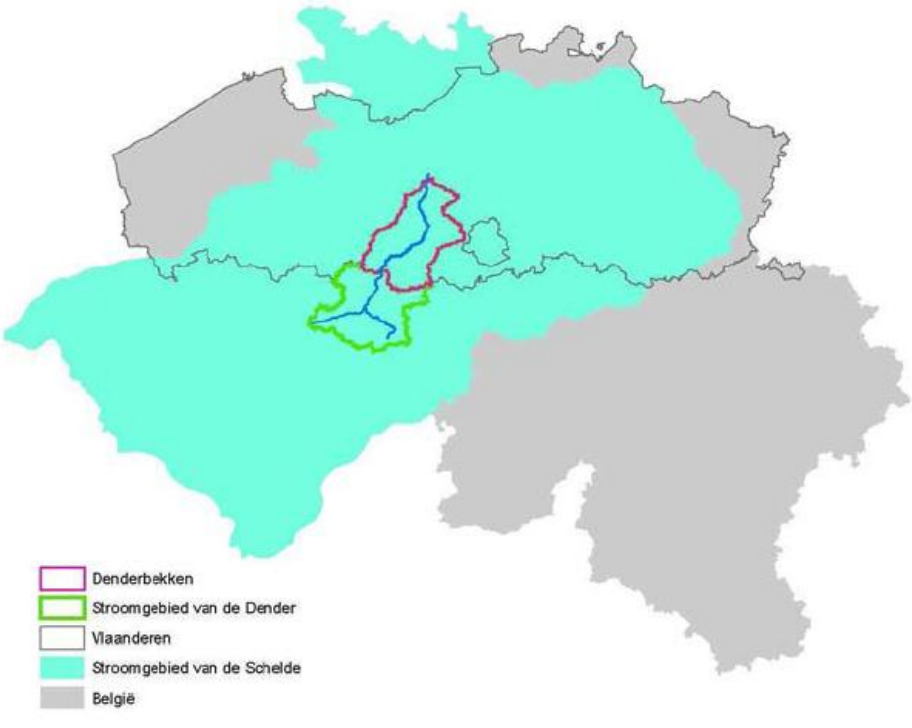


Figure 4: Example of a river basin and sub basin for the Scheldt river basin in Belgium-France-the Netherlands

Local or site scale: project management unit that is used to evaluate the performance of rehabilitation technologies for single or multiple plumes at a contaminated site based on performance tests and models at the field/site scale (1-10 km^2) [1]

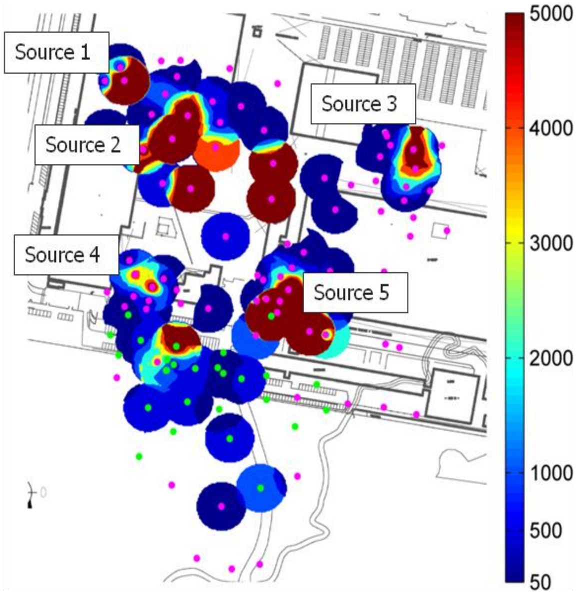


Figure 5: Example of a site scale pollution

Local pollution: contaminant plumes resulting from direct discharge in groundwater and contaminated land or direct discharge in surface water. Various pollutants and often a mixture of chemicals is present.

Diffuse pollution: pollution originating from diffuse sources. Diffuse pollution is characterized by the absence of a clearly defined source. Typical pollutants are pesticides, nitrate, phosphate, heavy metals.

[1] The groundwater directive also mentions: "plumes resulting from point sources and contaminated land, Member States shall carry out additional trend assessments for identified pollutants in order to verify that
plumes from contaminated sites do not expand, do not deteriorate the chemical status of the body or group of bodies of groundwater, and do not present a risk for human health and the environment. The results of these assessments shall be summarized in the river basin management plans …"

Example: Policy context, spatial scale and DSS development

In the context of Aquarehab, numerous discussions among experts and with stakeholders (e.g., Barcelona conference, 2012) highlighted the gap between water management at the river basin scale and groundwater remediation which is dealt with at the local scale. "In the discussions it was felt not necessary to integrate site remediation with river basin management. The latter is driven by the WFD and local scale problems of contaminated sites do not generally affect the large scale status of water bodies. The WFD is about focused in the long term, large scale management of water bodies, and is not about local scale pollution incidents or risks to health. Although the Groundwater Directive mentions groundwater plumes, there is no EU Soils Directive and so each country has adopted its own approach and legislation. This is different for diffuse pollution. The management of diffuse pollution is integrated across groundwater and surface water, both through the WFD and the Nitrate Directive, and more recently the Sustainable Use Directive for pesticides. There is convergence in policy across the EU, with risk assessment and management becoming the underlying philosophy." Therefore, to evaluate the impact of (multiple) measures related to diffuse pollution (nitrate, pesticides, …) it was decided to develop the REACHER DSS where the evaluation of (multiple) measures is done at the scale of a subbasin. To account for impact of technologies on groundwater due to local pollution or a cluster of locally polluted sites, it was decided to develop a REACHER-LOCAL DSS. REACHER Local is a prototype decision support tool for a regional-scale assessment of all known groundwater polluted sites in a region (e.g. Flanders, Belgium). Users are able to explore the status of polluted sites across a region, how this status evolves in time with/without remediation, which potential impacts can be expected for different sites, which societal cost we bear due to the environmental damage or the benefits that can be achieved by reducing pollution levels, and which technologies can be implemented for different sites and score best on costs, effectiveness (speed).

Four modules in the tool are distinguished:

1. *Status: mapping the groundwater quality and the expected evolution of the pollutant concentration in the groundwater, with and without measures*
2. *Impact: predicting whether the modelled groundwater quality will have an impact for different impact categories (drinking water, surface water, indoor air pollution)*
3. *Technology: predicting which technologies are the most suitable to tackle specific pollutants in specific sites*
4. *Damage: expected economic benefits we realize by reducing pollution below treshholds.*

3.1.3.2 Water aspects

A good water status can be very broad including aspects as water quantity (droughts, floods), water quality (surface and groundwater), sediments, hydromorphology and ecological quality.
For surface water quality, status is determined by ecological status and chemical status.

- *Ecological status*: refers to the combined status of biological, physical chemical and hydromorphological quality elements

- *Chemical status*: refers to the status, amongst others the EU priority substances and other EU level dangerous substances
- *Priority substances*: a set of the WFD priority substances and dangerous substances. Lists of pollutants at EU level are defined by the WFD (priority substances), the groundwater daughter directive or the nitrates directive (see Annex 2 and 3). Member states can establish their own monitoring list of dangerous pollutants for surface water (extensive list of pollutants) or groundwater (usually metals, pesticides, nitrate).
  A list of relevant pollutants for the DSS can be further established based on: occurrence on the WFD list for priority pollutants, registration deadlines, occurrence in river systems of interest in the EU, ecological relevance, physical chemical properties, and available databases from literature or other EU projects (Modelkey, SOCOPSE, SCOREPP, FOOTPRINT).
- *Environmental quality standards*: thresholds derived at EU level for priority substances and other EU level dangerous substances used to evaluate chemical water status (see Annex 4 regulatory limits for selected substances).
  For many substances environmental quality criteria or standards are still under development or lacking. Specific ecotoxicological approaches can be used to derive the criteria. Several approaches were previously suggested to predict or model impacts of toxic chemicals on aquatic biota (Solomon, 2008) but all of them have their own advantages as well as limitations. The approach used most generally is based on comparisons of water concentration of certain chemical with its environmental quality standard (EQS). Concentrations above the EQS indicate risk. The major limitation of this approach is often the lack of solid EQS values for most of the contaminants. Derivation of the EQS values must be based on complex risk assessment, which is very complicated, costly and time-consuming process as currently experienced e.g. during implementation of REACH in the EU or previous activities such as EU Combined Monitoring-Based and Modeling-Based Priority Setting (COMMPS). Uncertainty factors (UFs) must be applied during the EQS derivation, and this issue brings a lot of scientific discussion. EQS for individual chemicals also provides rather qualitative information (risk: yes/no), and it does not take into account any mixture effects and/or interactions among chemicals.
  Species sensitivity distribution (SSD) is another suggested approach. For a specific compound, its impacts on biological community is characterized by a collection of all ecotoxicity values available across species (EC50 values or NOECs for algae, plants, various invertebrates, fish, amphibia etc.). Parameters derived from the statistical distribution of the toxicity values (i.e. effective concentrations) can then be compared with actually measured or modelled environmental concentrations. This comparison then predicts "what fraction of the community is likely to be affected" by certain concentration (potentially affected fraction, PAF). PAFs obtained for different chemicals may be combined to derive msPAF (multi-substance PAF), which represents the impact of a mixture. The msPAF value (0 - 1) is then used as a parameter used in classification of risks at specific localities / time periods (higher msPAF values indicate higher risks of toxicants). Application of the chemical mixture concept in SSD is based on the assumption of independent action of chemicals. In some cases, this might not be necessarily true as compounds actually present in the environmental sample may share their mode of action (e.g. several different herbicides affecting one specific process in the photosynthesis). To fully address this problem, independent action (i.e. response addition model) should be combined with the concentration addition model (Posthuma et al. 2002). However, application of this approach is complicated by the fact that a single SSD model has to be constructed for each individual chemical, and this SSD is based on a combination of different

organisms and taxa. Combining algae, plants, invertebrates, fish etc., inherently assume different toxicity mechanisms in the overall mixture toxicity, which justifies application of the response addition model. Nevertheless, our recent study demonstrated successful application of the combined SSD mixture model in the assessment of risks (Jesenská et al. 2013).

Selection of substances and ecological criteria for the REACHER-DSS

In Aquarehab, a selection of substances was made based on their occurrence in the pilot river basins (Scheldt and Odense), available regulatory limits, and chemical properties (mainly solubility expressed by K_{ow}). An example of a list of contaminants is shown in Annex 4. The pollutant list was further reduced to the substances for which criteria were established and which could be taken up in the REACHER DSS (green color = relevant, orange = moderately relevant, red = non-relevant to project objectives). This resulted in the following parameters:

- *Nitrogen/nitrate*
- *Pesticides: Isoproturon, Simazine, Terbutylazine, MCPA, Bentazon, Glyphosate, AMPA, Mecoprop. The selection of the pesticides was done based on their occurrence in EU rivers, their presence on the market, their inclusion in the WFD priority substance list, their physical chemical properties (moderately sorbing), and the existence of physical and chemical data from other EU projects.*
- *Chlorinated aliphatics: trichloro-ethylene. This substance is considered the model substance for the CAHs. It was chosen because of the presence on the WFD priority substance list and information collected in the SCORE-PP project. For the development of the REACHER-local DSS 1,1,1-trichloroethane, 1,1-dichloroethane, Perchloroethene, Trichloroethene, Dichloroethene and Vinylchloride were selected*
- *BTEX: toluene and benzene. These substances are representative for the BTEXs. They were chosen because of their relevance for groundwater pollution and their inclusion in the WFD list (benzene). Ethylbenzene and xylene were selected in addition for the REACHER local DSS.*
- *Nonylphenol, DEHP: these substances were chosen for a dedicated study related to the Zenne river case where they occur and have a strong ecological relevance as evidenced in the Modelkey project. They are also included in the WFD priority substances list.*

The REACHER DSS considers chemical status based on EQS or SSD-based ecotoxicological criteria when no EQS are available. It was decided to use only EQS/ecotoxicological criteria based on the fact that a straightforward assessment can be made between a measure acting on a specific substance (flux reduction) and an "effect" (by comparing to an EQS concentration, or ecotoxicological threshold). The assessment of ecological effects is not evident since ecological status depends on multiple factors among which hydromorphology and general water quality. The REACHER DSS allows for visualization of the ecological status for punctual ecological assessments made in the pilot river basins.

3.1.3.3 Selection of rehabilitation/mitigation measures

Before selecting the measures to include in the decision support system it is important to distinguish between environmental measures and policy instruments. With an environmental measure, we mean techniques or actions that are undertaken with the explicit aim of preventing or addressing undesirable effects of human intervention on the environment (e.g. wastewater treatment). The assessment of policy instruments (e.g. regulation, subsidies, covenants, levies,

etc.) that the government can use to encourage the public and companies to take certain environmental measures is more complex (the response of the public/companies to policy instruments also needs to be modelled) and decision support systems are mostly not equipped to do this. Also, it is important to not double count the effects. A policy instrument might result in an environmental measure but we cannot take this effect into account both for the policy instrument and the environmental measure when we perform a holistic assessment.
For environmental measures we distinguish between:

- *Conventional measures*: the measures defined in river basin management plans (RBMPs)
- *Innovative measures*: non-conventional measures and innovative rehabilitation technologies for which less information exists on both costs and effects.

3.1.3.4 Economic analysis

Typically a distinction is made between 2 forms of economic appraisal techniques:

- *Cost-effectiveness analysis (CEA)*: economic analysis used to evaluate a set of measures in the DSS. In the context of water management, the purpose of a cost-effectiveness analysis is to find out how predetermined targets, for example pollutant loads in a water body, river basin or estuary, can be achieved at minimal costs (Lise and van der Veeren, 2002). It can be used as an appraisal technique for assessing and ranking the relative performance of different measures or combination of measures on the basis of their costs and their effectiveness.
- *Cost-benefit analysis (CBA)*: Cost-benefit analysis (CBA) is an applied economic tool often used to guide economic agents in resource allocation or investment project decisions or policy alternatives. It is a technique that is used to estimate and sum up (in present value terms) the future flows of benefits and costs of society's resource allocation decisions or policy alternatives to establish the worthiness of undertaking the stipulated activity or alternative, and inform the decision maker about economic efficiency. CBA addresses the question of whether the objective (or action) is economically worthwhile and finding the socially efficient level of emissions: do the benefits exceed the costs (Balana et al., 2011)?
- *Multicriteria analysis (MCA)*: A MCA is a generic term for a number of methods that use multiple criteria for evaluating alternatives. These criteria are usually related to the objectives and points of attention of the policy makers and stakeholders. The aim of the MCA is to support the decision making by ranking the options according to the regulator's preferences.

Brouwer and De Blois, 2008 defined 8 practical steps in performing a cost effectiveness analysis. Steps are taken in sequence, but important feedbacks usually exist between steps when learning more about the problem, the source-effect pathway and possible solutions. These basic steps are:

- Step 1: Identify the environmental objective(s) involved (target situation)
- Step 2: Determine the extent to which the environmental objective(s) is (are) met
- Step 3: Identify sources of pollution, pressures and impacts now and in the future over the appropriate time horizon and geographical scale (baseline situation)
- Step 4: Identify measures to bridge the gap between the reference (baseline) and target situation (environmental objective(s))

- Step 5: Assess the effectiveness of these measures in reaching the environmental objective(s)
- Step 6: Assess the direct (and if relevant indirect) costs of these measures
- Step 7: Rank measures in terms of increasing unit costs (cost effect ratios)
- Step 8: Determine the least cost way to reach the environmental objective(s) based on the ranking of measures

Steps 1 to 3 are typically dealt by environmental models and are performed by setting up fate models. The focus of the economic analysis starts at step 4 and the identification of measures.

Economic assessment
To prioritize and screen the measures, an economic cost-effectiveness analysis (CEA) is conducted in the REACHER DSS. Extensive, multi-objective optimization algorithms as envisioned at the start of the Aquarehab project, were at this stage less desired by potential end users (see 3.1.2), but maybe developed in future.

In the REACHER local a MCA was used. Regulator preferences are expressed through the weighting of the following criteria:

- *Cost: Refers to the price of the implementation of the technology, including all the services from the writing of the specifications to the project completion report. It includes the fixed costs and the variable costs like maintenance costs.*
- *Planning: This criterion evaluates the duration of the treatment and whether the implementation of the technology is compatible with short-term real estate projects.*
- *Availability: Refers to the current level of applicability of the technology. Is the technology commercially available? Only at research stage? etc.*
- *Efficiency : For each targeted substance, it provides the abatement rate (ratio: substance concentration after the technology implementation / substance concentration before).*
- *Co-effects: This criterion integrates two dimensions :*
- *The qualitative environmental footprint of the implementation of the technology, through the carbon footprint due to energy consumption, the potential damages to biodiversity, noise considerations…*
- *The possibility to reduce other contaminations. Five categories of contaminants are taken into account: pesticides, HAPs, heavy metals, explosives, and hydrocarbons.*
- *For the MCA purpose, each combination of technology and substance had to be assessed according to those five criteria. A normalized evaluation was done on a 0 to 10 scale, 10 being the best score.*

3.2 DSS OBJECTIVES

Based on the regulatory and end user requirements, the objectives of the DSS can be delineated. Decisions with regard to scale, selection of water quality parameters and thresholds, and method of economic analysis, determine the architecture of the DSS and the software used to build the tool. Every tool that relates measures to impacts also needs models that convert changes in fluxes due to mitigation or remediation technology into quantitative metrics that can be compared with regulatory thresholds. Also measures are taken in a spatial context, so the tool should be able to accommodate for spatial information. This poses additional requirements to the software architecture.

REACHER DSS objectives

"The REACHER tool should enable water managers to decide on the implementation of various water rehabilitation technologies and mitigation measures (including groundwater remediation, waste water treatment, and reduction of diffuse pollution) aiming at water quality improvement at the river basin scale."

Features of the DSS based on policy and enduser needs are:

- *Web-based tool with user interaction and visualisation*
- *Evaluation at river basin scale*
- *Multiple measures are taken into account in scenarios*
- *Routing at river basin scale: models need to calculate sufficiently fast*
- *Chemical status is considered: regulatory criteria or ecotoxicological limits are used to evaluate the measures*

Furthermore, specific technical challenges as outlined before need to be tackled:

- *Upstream-downstream effects are taken into account: fate models are needed*
- *Conversion from complex models to simple calculus enabling web application*
- *Effects of groundwater on surface water need to be accounted for, and time delays between application of the measure and the effect need to be assessed.*

Ecotoxicological criteria need to be established for mixtures of chemicals

"Reacher Local" is a prototype decision support tool supporting a rapid assessment for all known groundwater polluted sites in a region (e.g. Flanders, Belgium). The objective is to get a broad overview of the scale of the pollution problem on a surface water body or municipality scale, how this evolves in time, what the potential impacts are of this pollution and with which potential measures these problems can be solved.

Users are able to explore:

- the status on polluted sites across a region;
- how this status evolves in time with/without remediation;
- which potential impacts can be expected for different sites;
- which technologies can be implemented for different sites and score best on costs, effectiveness (speed), accuracy;
- the societal cost due to the environmental damage and the benefits that can be achieved by reducing pollution levels.

3.3 DSS Components and general architecture

A DSS architecture basically consists of three main components:

- Models
- Databases
- User interface

Models are used to process the data from the databases to outputs for end users.
Databases can contain:

- data from literature, inventories, expert knowledge
- data from lab tests
- output data from models

The user interface allows for:

- User entries: choice of management options for a certain geographical unit, choice of evaluation criteria, …
- User output: visualization of effects (indicator showing status) on maps, cost information, tables, graphs, probabilities…

For every component, choices in software need to be made. Free and Open Source Software (FOSS) can be used for the implementation. Table 1 shows examples of FOSS for these three components.

Table 1: Modules and example tools for implementation of a DSS

DSS Module	Example tools
Model	Bayesian Belief Network (GeNIe)
Database	PostgreSQL, PostGIS
Web-based Visualization	Apache, Mapserver, Openlayers, Extjs, WebKit

For REACHER we use a light-weight Bayesian Belief Network to perform the modelling as the complicated process-based fate models need too much calculation time for a web application. A Bayesian Belief Network (BBN) can represent the model results in a comprehensive and probabilistic way without having the high computing demand for many process-based models. It is a suitable management tool in constructing the decision support model. An example of a BNN is the GeNIe modelling environment in combination with the SMILE graphic user interface (developed by the Decision Systems Laboratory of the University of Pittsburgh). One advantage of the BBN is that the outputs of the model are only the relationships, represented as conditional probability tables (CPT). These tables can be easily transformed into spatial databases so that the user interface can connect to the model through spatial database. The graphic structure is another characteristic of the BBN and it has the advantage of representing the real world topology. WebDSS user interface is designed to communicate with the database following data transform standards.

Two kinds of layers have to be visualised: static and dynamic. The first ones are in charge of communication with database values, and rendering features is its main responsibility. Dynamic ones have the computational engine, which assists in generating the new layers according to the input parameters. Here, the computational engine consists of the BBN core functions and the dynamic layers type is the vector layer.

Interaction design is another important element in user interface and it influences the utility of the system directly. Therefore, the principle of the interaction style is designed based on a survey of the users' habits. Clickable items with popups are chosen following this principle and the gesture interaction is also taken into concern. The process flowchart is shown below (Figure 6):

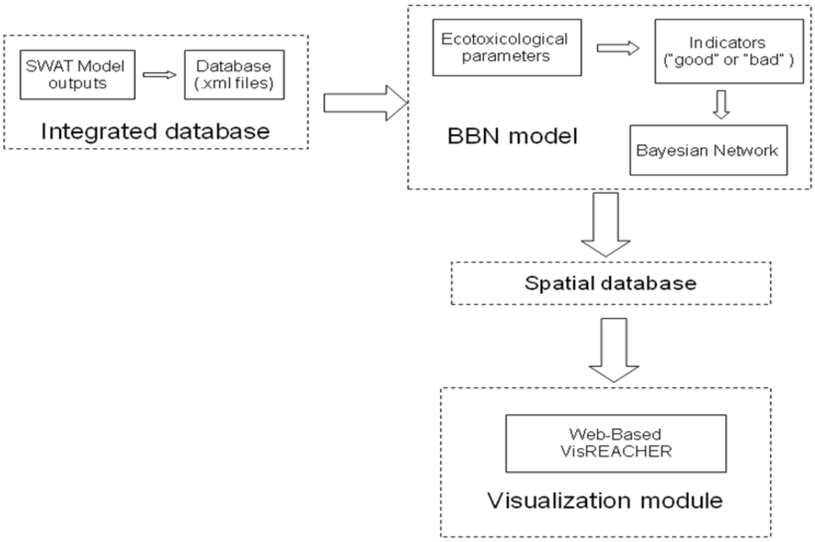


Figure 6: General layout of the DSS for evaluation of rehabilitation technologies in a geographical framework

3.3.1 Models

3.3.1.1 Hydrological fate models

The core of the DSS is formed by models that link the rehabilitation measures to a water status. Typically, this can be accomplished by running chemical fate models at the river basin scale. Fate models simulate the behavior and movement of chemicals at the watershed scale in the soil-groundwater-surface water continuum. Well-known examples of fate models at the river basin scale are SWAT, MIKE-SHE, MODHMS and HYDROGEOSPHERE. Whereas SWAT uses simplified hydrological process description and management-oriented, the other models are explicitly modeling water flow and chemical transport in loosely coupled or fully coupled soil, groundwater and surface water compartments. Data requirements and expert knowledge for the latter are considerably high and their application is mainly on well characterized and relatively small catchments.

Watershed fate models behind REACHER DSS

In the Aquarehab project, SWAT was used and adapted to accommodate for artificial drainage and wetlands. SWAT was applied to the Odense river catchment. A new conceptual modeling platform programmed in PCRASTER – Python, SECOMSA, was developed to account for multiple sources (point sources, diffuse sources) and multiple mitigation measures in the catchment (wetlands, riparian zones, connection households to waste water treatment, cattle reduction) and to model the impact of the measures on water quality at the outlet of the subbasin. The model was calibrated and evaluated for the Scheldt river basin.

The fate model engine behind REACHER local is the COnceptual Model For a Regional Assessment of Contaminated Sites: COMFRACS+.

COMFRACS+ was developed for groundwater contaminated with mineral oil and chlorinated hydrocarbons in Flanders, Belgium. It makes use of the extensive database of the Public Waste Agency of Flanders (OVAM) that is coupled to the Python/PCRaster numerical framework via a PostgreSQL/PostGIS database. The monitoring data of the surveyed sites were averaged in a regional grid with a spatial resolution of 25 m. Plume development was approximated assuming steady state groundwater flow with advective mass transport in the direction of the steepest gradient, including retardation and first order degradation.

Runs with river basin scale fate models in online DSS applications are not feasible and therefore the model results need to be translated to rules expressing the relationship between measure and status. Either multiple scenarios with fate models are run beforehand and the results are stored in a database, or a Bayesian belief network[2] is trained by multiple runs with the complex model. A BBN needs rules (cause-effect relationships) and links+ nodes (a river network). From there, the network can calculate a status without the need of running a complex model. The cause effect relationships need to be established by models that are run off-line. BBN are suitable for rivers with upstream –downstream interactions along the river network, but become increasingly complex for land- or groundwater related problems. The basis of the BBN model is the dataset of model simulations. The dataset is first transformed a standard format that can be easily manipulated by programming tools like MATLAB. A number of MATLAB programs are developed then to classify the dataset into discrete categories (in comparison with original continuous values). The MATLAB programs then communicate with the BBN software to build and complete the BBN that is ready to be used by other components of the DSS. MATLAB scripts can also be used to derive annual statistics from the simulations, e.g., maxima, minima and annual average, depending on the needs for the DSS.

[2] (Wikipedia) A Bayesian network, belief network or directed acyclic graphical model is a probabilistic graphical model that represents a set of random variables and their conditional dependencies via a directed acyclic graph (DAG). For example, a Bayesian network could represent the probabilistic relationships between diseases and symptoms. Given symptoms, the network can be used to compute the probabilities of the presence of various diseases. Formally, Bayesian networks are directed acyclic graphs whose nodes represent random variables in the Bayesian sense: they may be observable quantities, latent variables, unknown parameters or hypotheses. Edges represent conditional dependencies; nodes which are not connected represent variables which are conditionally independent of each other. Each node is associated with a probability function that takes as input a particular set of values for the node's parent variables and gives the probability of the variable represented by the node.

Real values are insufficient to reflect the common probabilistic behaviour of the modelled system, thus a classification step is introduced to group them into several ***statements***. In this study we only use two classes - “good” and “bad”, as nine-year annual data is not capable to support more statements. To determine “good” or “bad”, mean values are firstly compared, and annual maximum values will only be considered only if the mean values are “bad”. It is not suggested to build the classification in the original database since how to classify statements are demand based and could be plenty of options, also the model is adaptive to different classifications that no more re-programming is needed if implementing other splits.

3.3.1.2 Economic assessment

The aim of the economic assessment is to identify the most cost effective course of action. This can be performed by simulating the impact of different management scenarios on selected substances (emissions or concentrations) in a river basin and to put them into balance with the total cost of these scenarios.

The basic equation in cost-effectiveness analysis can be formulated as an optimization problem:

$Minimize \sum c_i \alpha_i$ with the following constraint:

$\sum \alpha_i \geq RT$

in addition to the usual non-negativity constraints, where c_i are costs for reducing pressures coming from source *i*, which are a function of *ai*, reduction of pressure by that source. The total of reductions must be not less than the total required reduction target *RT*. Similar formulations can be found in (Hanley et al., 1992; Schleich and White, 1997; van der Veeren and Lorenz, 2002; Fröschl et al., 2008; Lescot et al., 2013).

This formulation implies the ranking of measures by average cost per unit effectiveness. The calculation is also often represented as the calculation of cost-effectiveness ratios R, which are defined as:

R = AEC/Effectiveness

where *AEC* is the Annual Equivalent Cost (euros/year). ‘*Effectiveness*’ can be defined as the quantitative change in the pressure on the resource or the improvement of the state of the environment. (Berbel et al., 2011)

The results of a CEA can be represented in abatement cost functions. Abatement costs are the costs of reducing the quantity of pollution being emitted into the environment or of improving the environmental quality (Field and Field, 2009). Abatement cost functions are a graphical representation of abatement costs. Marginal abatement costs show the added costs of achieving a one-unit decrease in pollution or increase in environmental quality. Marginal cost curves rise from left to right, depicting rising marginal costs of reducing emissions further and further (option b in figure 2). The starting point on the curve is the existing emission or quality level E_0. Reducing the emissions to the requested target requires measures with a marginal cost lower than Ct to reach this target at the lowest cost achievable. Points on the curve can thus be understood as minimum costs of achieving certain levels of emission reductions. Marginal cost curves can also be depicted in function of emissions instead of emission reductions (option a in figure 2). In this case, curves rise from right to left depicting increasing marginal costs to realize lower levels of emissions.

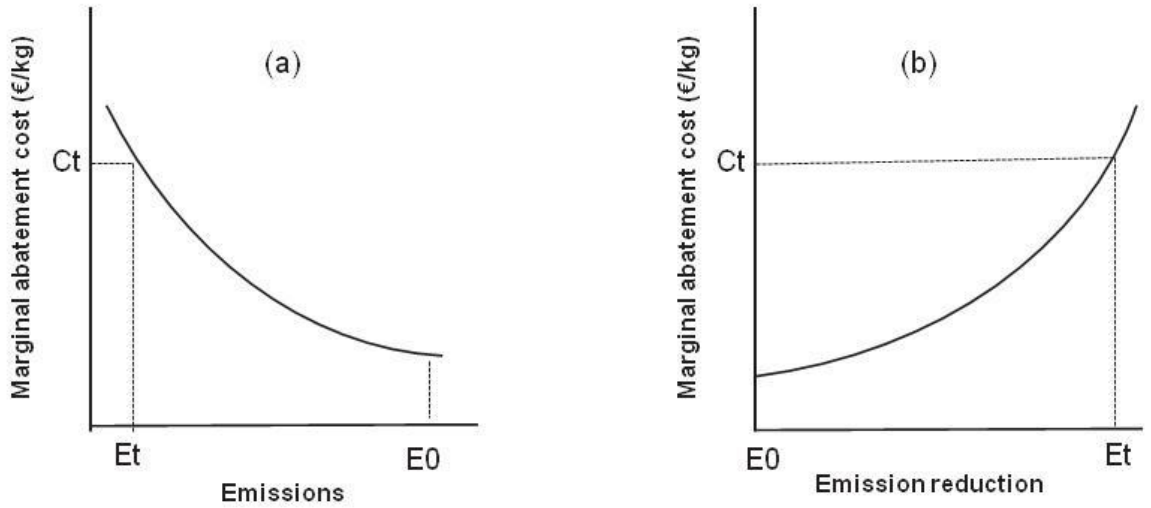


Figure: Marginal abatement cost function in function of emissions (a) or emission reductions (b).

Data requirements for measures can be summarized as:

- Costs: investment and annual operational costs
- Lifespan: Amount of years for which we can expect the measure to be operational.
- Effectiveness: Impact of measure on reducing emission loads or reaching a certain environmental quality standard. This depends on the effect (Eff, % removal we can expect), the implementation rate (IR, to what extent are we implementing a measure), and the application potential (App, where can we feasibly apply the measure):

$$Load\ reduction = IR \times Eff \times App \times Load$$

Data on measures and costs in REACHER DSS

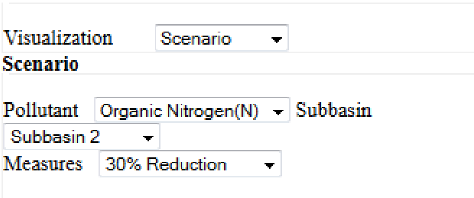
Nitrogen is one of the substances for which water quality targets (at the scale of a river basin) are difficult to reach. The substance is also emitted by a multitude of sources and for international river basins, in different member states. Conventional measures, included in the river basin management plans, typically include wastewater treatment for households (collective or individual) and reducing agricultural emissions by reducing livestock, fertilizer use or performing manure processing. Wetlands and buffer strips are also included in the different management plans and are considered effective in reducing run off losses. Information on costs and effectiveness of these measures is included in REACHER.

3.3.2 User interaction and visualisation

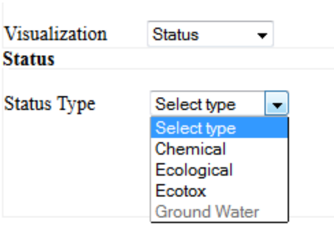
3.3.2.1 User interaction

The DSS can visualise the results of status and scenarios in both (geo)graphical form as well as in tables. Maps and tables need to be linked in both ways, enabling the end-user to select records in the table that highlight in the map and vice versa. Users need to be able to consult information on the current status of the water system (and subcatchments), the pressures that have an impact on this status and the measures that are able to reduce the pressures.

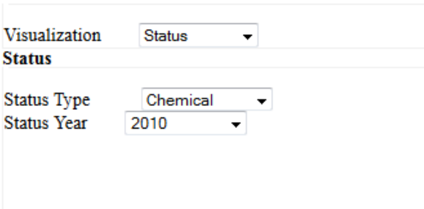
Selection of substances: A selection of a specific pollutant can be made (screenshot REACHER).



Selection of status: A selection between chemical, ecological and ecotoxicological status can be made (screenshot REACHER).



Selection of monitoring period: The user can select a period for which the water status is given (screenshot REACHER).



Selection of measures: Besides consulting information, users have also the possibility to compose scenarios (a selection of measures) for which calculations on total costs and effects can be performed. The impact of a scenario on the status is estimated with the hydrological module. The cost effectiveness analysis for each measure included in a scenario is based on a calculation of cost-effect ratios (cost divided by effect).

Implementation of measures in REACHER DSS

In the REACHER DSS, the user selects a scenario for visualisation. User selects a combination of measures. The model calculates concentrations and compares this with the base value. Additional columns will be added to display results of the economic analysis. For each measure in each subbasin, a database with total annual costs, total annual load reduction (kg/year) 2015-2021-2027. Depending on the selection of measures and the selected year, the sum of all costs and load reductions for all selected measures per subbasin is made. Results are shown in additional columns in the tabular view. One column displays all costs per subbasin, one column displays all load reductions per subbasin. It should be remarked that in the scenario runs all measures were applied homogenously over the entire catchment. A column displaying cost-effect ratios (costs divided by load reductions) can be added.

3.3.2.2 Visualisation of surface water status (ecological and chemical status)

The classification schemes in the DSS follow the WFD methodology (Figure 7) for assessment of the overall status of a surface water body. Specific descriptions of the ecological status assessment can be summarized as follows. Ecological status is a result of “the worst case scenario” bringing together the below scores of the three evaluated biological quality elements (BQEs) – benthic diatoms, macrophytes, benthic invertebrates (three sub-parameters – general degradation/saprobity/stream morphology) for each site and year.

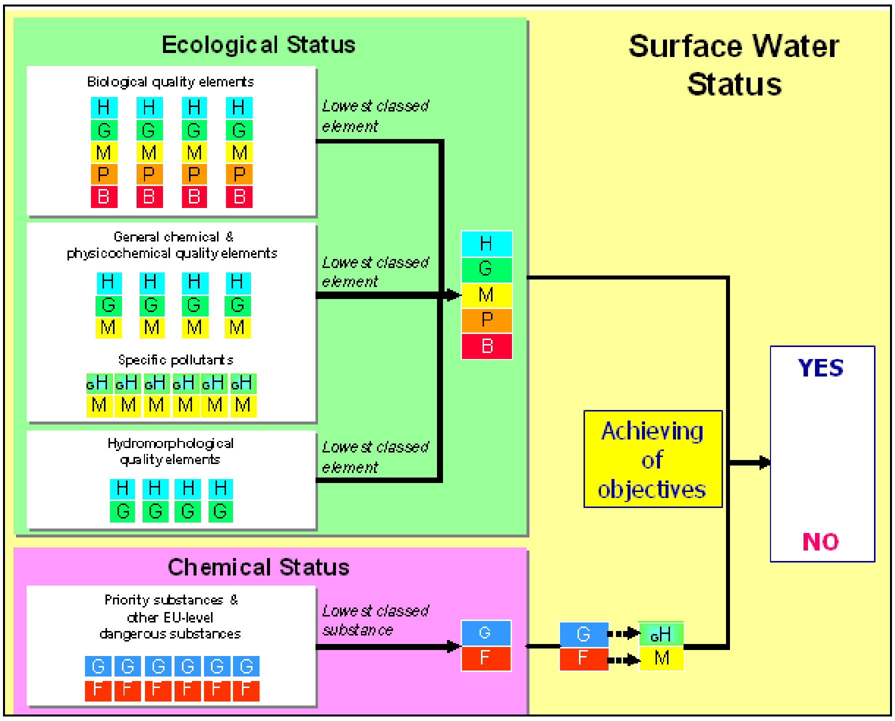


Figure 7: Basic scheme of ecological and chemical status assessment including quality elements. Ecological status is classified according to 5 categories: H – High; G – Good; M – Moderate; P – Poor and B – Bad.

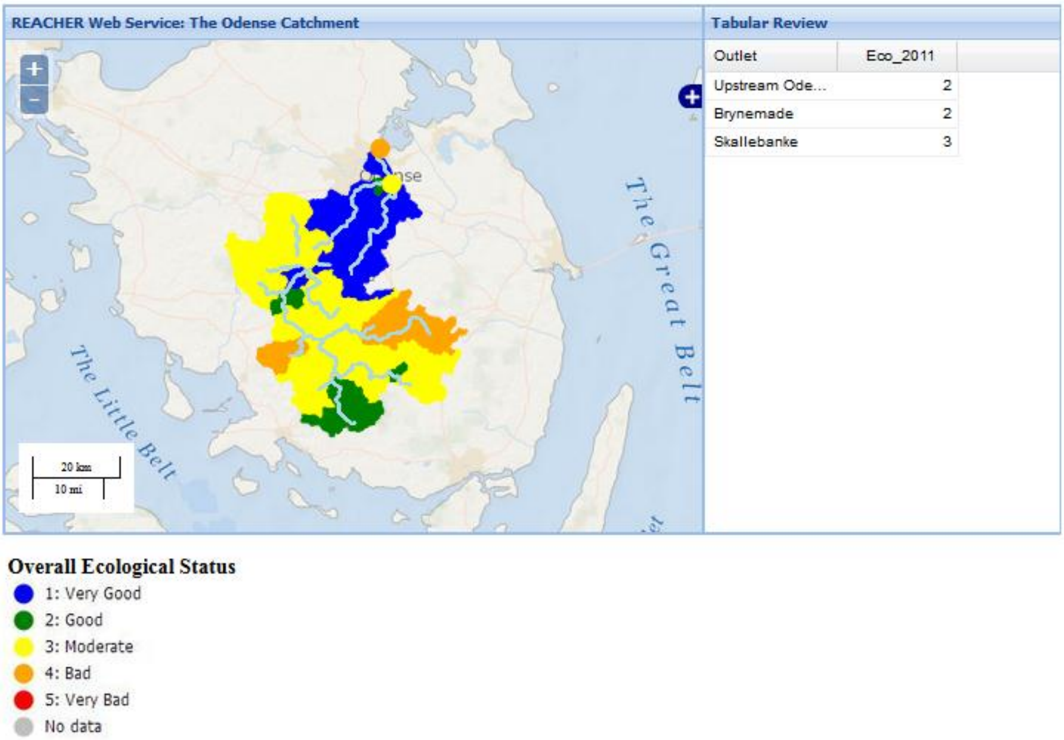


Figure 8: REACHER visualisation of the ecological status in the Odense catchment in 2011 using the WFD legend.

Surface water status in REACHER-DSS
In research projects, the requirements of the WFD are not accomplished since sampling sites may not be representative for a river body, and frequency of sampling is lower than required by the WFD. Therefore classification colours can only be assigned only to a certain location (displayed as a dot or circle on a map) and not to a water body (river stretch). Ecological and chemical status data are stored in a database. They are categorized into categories, (e.g. H - High; G - Good; M - Moderate; P - Poor; B- Bad) depending on the policy context. The ecological and chemical status are visualised using coloured dots at the location of the sampling sites. Colour coding, for example, follows the WFD methodology for assessment of the overall status of a water body.
Visualisation of ecological status is done at two levels:

1. *Overall score*
2. *Scores of the three evaluated biological quality elements (BQEs)*

Further levels (3 and 4) of information (metrics behind indicators and raw data) can be included as links to reports of ecological status.

3.3.2.3 Visualisation of ecotoxicological data

Ecotoxicological criteria are derived for specific chemicals based on the SSD approach resulting in PAF, ms-PAF or weighted mean PAFs or msPAFs (for dynamic data, i.e. chemical concentrations collected or modelled within a specified period of time).
If single chemical concentrations are measured for a certain river basin (site, locality), concentrations can be translated into the PAF value. The PAF value predicts the hazard that

specific fraction of a community (0-1) is likely to be affected by the chemical concentration (e.g. PAF of 0.05 indicates that it is likely that 5% of the community is affected; higher PAF value indicates higher hazard).

If multiple chemical concentrations are measured for a certain location, concentrations can be translated into msPAF value. The ms-PAF value predicts the hazard that specific fraction of a community (0-1) is likely to be affected by the mixture of chemical concentrations (e.g. ms-PAF of 0.05 indicates that it is likely that 5% of the community is affected; higher PAF value indicates higher hazard).

Table 2: Legend for the hazard classification according to PAF or msPAF as implemented in REACHER DSS.

Hazard category	Limits PAF [%]	Color
1 (no / lowest hazard)	<1	Blue
2	1 - 2.5	Green
3	2.5 - 5	Yellow
4	5 - 15	Orange
5 (highest hazard)	> 15	Red

If the dynamics of chemical concentrations are known through either high-frequent monitoring or modelling, the concentrations can be translated into weighted mean PAF values (or weighted mean msPAF).

Table 3: Legend for the hazard classification according to weighted mean ms-PAF as implemented in REACHER DSS

Hazard category	Limits weighted mean PAF or msPAF [%]	Color
0 (no hazard)	< 1E-02	
1 (lowest hazard)	1E-02 - 1	Blue
2 (low hazard)	1 – 2,5	Green
3	2,5 - 5	Yellow
4	5 - 15	Orange
5 (highest hazard)	> 15	Red

The information of the ecotoxicological database is used to define categories ("flag marks") for PAF values (potentially affected fractions derived with SSD model. The amount of categories depends on the user requirements. For example, a value of 1 indicates the lowest hazard and a value of 5 indicates the highest hazard. The limits for the categories need to be derived from the distribution of actual concentrations of the chemical. Table 2 and Table 3 show an example for the colour coding.

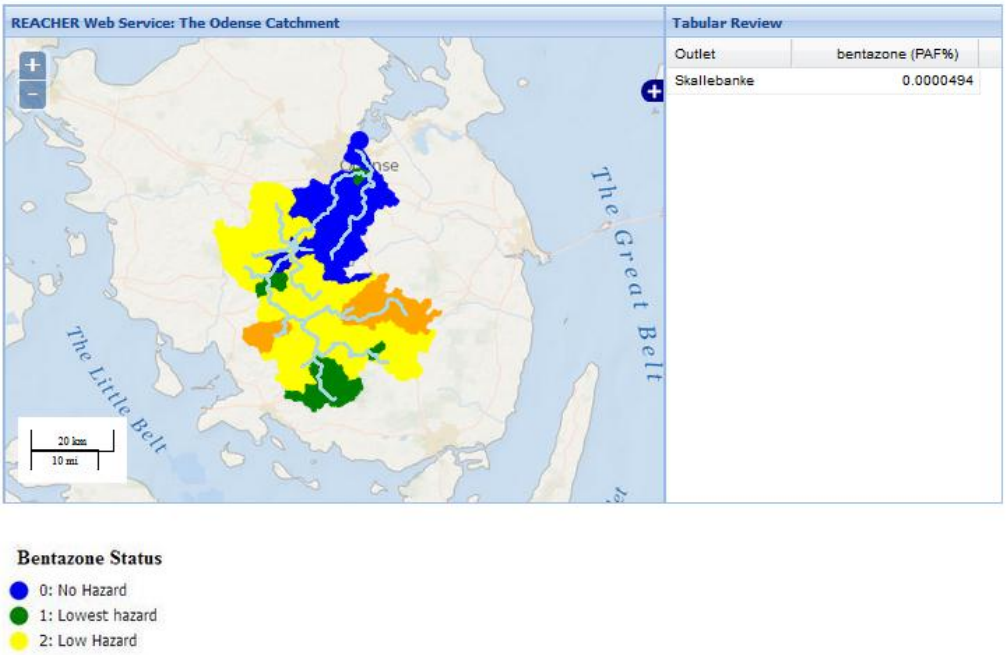


Figure 9: REACHER visualisation of ecotoxicological status based on the msPAF approach developed in the Aquarehab project to account for ecotoxicological effects from chemicals and mixtures of chemicals. PAF refers to Potentially Affected Fraction (%)

3.3.2.4 Visualisation of nutrient concentrations

Nutrient concentrations are visualised by assigning a colour to the entire subbasin. The colours are based on the policy context adopted by the Member state.

3.3.2.5 Tabular results

The results of the ecological, chemical and ecotoxicological status as well as the nutrient concentrations from the fate models also need to be presented in a table for more quantitative interpretation. The table shows the values for the status. For the results of the fate models, the table shows the number of the subbasin, the concentration as a result of the chosen scenario and the concentration of the default model run without implementing the scenario. The table and the geographical view are linked. In this way a selection can be made in the map that will be highlighted in the table and vice versa.

3.3.3 Databases and data requirements

The DSS needs information from geographical information systems for use in the fate models and for use in the output visualisation. In order to give the end user the context of the catchment, vector layers of water bodies are visualised in the WebDSS. For example Open Streetmaps data can be used. In addition, other topographical information, such as roads and relief, can be used.

Table 4: DSS Input requirements

Geographical information	
Fate models	-digital elevation map
	-soil map
	-landuse map
	-river network map
	-geological map
	-waste water treatment zones map
Visualisation	-subbasin delineation
	-basin delineation
	-groundwater bodies
	-surface water bodies
	-background Open Streetmap
	-administrative boundaries
Data	
Fate models	-concentration data
	-flow data
	-climatological data
	-pollutant emission data
	-aquifer data
	-soil data
Ecological assessment	-biological data (species)
	-hydromorphological data
	-physical chemical data
	-ecotoxicological data
	-environmental quality standards
Economic assessments	-costs of selected rehabilitation measures
	-lifespan of selection rehabilitation measures
	-implementation rate, effectiveness, applicability

4 DSS MAINTENANCE

Decision Support Systems developed in projects run the risk of being discontinued after the project. In order to guarantee the afterlife of a DSS the following issues should be addressed:

- Support to users, including updates of user manuals when new versions of the tools come out;
- Updates of the DSS when new model results come available or bugs have been corrected;
- Hosting of the tool should be sustainable.

These funding for resolving these issues need to be addressed in a business plan. The aim of the business plan is to secure funding for the project afterlife. This can be achieved when clients of the DSS have needs for improvement and are willing to invest in the improvements. Some business models are:

- Sustainability of the DSS through selling licenses that cover the cost of support
- Providing the DSS as open source tool in order to increase the outreach and attract potential business partners to continue the development of the tool

5 GENERAL CONCLUSIONS

Challenges related to the implementation of groundwater remediation technologies at the river basin or groundwater body scale are the upscaling from field to catchment scale, the interaction between groundwater and surface water, the time delay between action and effect due to the attenuation processes, and the assessment of the effects of multiple measures within one catchment.

In this guideline a general process flow was given to integrate the different aspects listed above in a decision support tool. The guideline was further illustrated with steps taken in the Aquarehab project leading to the development of a prototype DSS. The REACHER DSS prototype is developed as a generic software tool to assess the effects of rehabilitation measures on water quality. It contains fate model calculation functionalities, cost effectiveness economic analysis, and visualization options for status visualization in a geographical context. Depending on specific user requirements, the tool can be further developed (user specific classification criteria, spatial detail, measures and scenarios of measures, economic analysis) in a participatory framework.

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8 ANNEX 1: GLOSSARY

Aquarehab: Acronym of the 7FP Collaborative Project, Large-scale integrating project entitled "Development of rehabilitation technologies and approaches for multipressured degraded waters and the integration of their impact on river basin management", grant agreement no. 226565

REACHER: Aquarehab decision support tool for rehabilitation measures related to pollution of groundwater and surface water bodies

Decision support tool (DSS) (in the context of Aquarehab): tool that enables water managers to select appropriate rehabilitation measures for groundwater and surface water to comply with environmental quality standards for surface water and groundwater bodies

Surface water (WFD): means inland waters, except groundwater; transitional waters and coastal waters, except in respect of chemical status for which it shall also include territorial waters.

Groundwater (WFD): means all water which is below the surface of the ground in the saturation zone and in direct contact with the ground or subsoil.

Artificial water body (WFD): means a body of surface water created by human activity.

Heavily modified water body (WFD): means a body of surface water which as a result of physical alterations by human activity is substantially changed in character, as designated by the Member State in accordance with the provisions of Annex II.

Body of surface water (WFD): means a discrete and significant element of surface water such as a lake, a reservoir, a stream, river or canal, part of a stream, river or canal, a transitional water or a stretch of coastal water.

Aquifer (WFD): means a subsurface layer or layers of rock or other geological strata of sufficient porosity and permeability to allow either a significant flow of groundwater or the abstraction of significant quantities of groundwater.

Body of groundwater (WFD): means a distinct volume of groundwater within an aquifer or aquifers.

River basin (WFD): means the area of land from which all surface run-off flows through a sequence of streams, rivers and, possibly, lakes into the sea at a single river mouth, estuary or delta.

Sub-basin (WFD): means the area of land from which all surface run-off flows through a series of streams, rivers and, possibly, lakes to a particular point in a water course (normally a lake or a river confluence).

River basin district (WFD): means the area of land and sea, made up of one or more neighbouring river basins together with their associated groundwaters and coastal waters, which is identified under Article 3(1) as the main unit for management of river basins.

Surface water status (WFD): is the general expression of the status of a **body of surface water**, determined by the poorer of its ecological status and its chemical status.

Good surface water status (WFD): means the status achieved by a surface water body when both its **ecological** status and its **chemical** status are at least "good".

Groundwater status (WFD): is the general expression of the status of a **body of groundwater**, determined by the poorer of its quantitative status and its chemical status.

Good groundwater status (WFD): means the status achieved by a groundwater body when both its **quantitative** status and its **chemical** status are at least "good".

Ecological status (WFD): is an expression of the quality of the structure and functioning of aquatic ecosystems associated with **surface waters**, classified in accordance with Annex V.

Good ecological status (WFD): is the status of a body of **surface water**, so classified in accordance with Annex V.

Good ecological potential (WFD): is the status of a heavily modified or an artificial body of water, so classified in accordance with the relevant provisions of Annex V.
Good surface water chemical status (WFD): means the chemical status required to meet the environmental objectives for **surface waters** established in Article 4(1)(a), that is the chemical status achieved by a body of surface water in which concentrations of pollutants do not exceed the environmental quality standards established in Annex IX and under Article 16(7), and under other relevant Community legislation setting environmental quality standards at Community level.
Good groundwater chemical status (WFD): is the chemical status of a **body of groundwater**
Hazardous substances (WFD): means substances or groups of substances that are toxic, persistent and liable to bio-accumulate, and other substances or groups of substances which give rise to an equivalent level of concern.
Priority substances (WFD): means substances identified in accordance with Article 16(2) and listed in Annex X. Among these substances there are **priority hazardous substances**. Priority hazardous substances (PHS): will be subject to cessation or phasing out of discharges, emissions and losses within an appropriate timetable that shall not exceed 20 years
Pollutant (WFD): means any substance liable to cause pollution, in particular those listed in Annex VIII.
Pollution (WFD): means the **direct** or **indirect** introduction, as a result of human activity, of substances or heat into the air, water or land which may be harmful to human health or the quality of aquatic ecosystems or terrestrial ecosystems directly depending on aquatic ecosystems, which result in damage to material property, or which impair or interfere with amenities and other legitimate uses of the environment.
Point source pollution: pollution that originates from a well identified source at a single location
Diffuse pollution: pollution for which the origin is not well defined or a group of point sources in an area
Environmental quality standard (WFD): means the concentration of a particular pollutant or group of pollutants in water, sediment or biota which should not be exceeded in order to protect human health and the environment. environmental quality standards for sediment and biota could be used instead of those for water
Emission limit values (WFD): means the mass, expressed in terms of certain specific parameters, concentration and/or level of an emission, which may not be exceeded during any one or more periods of time. Emission limit values may also be laid down for certain groups, families or categories of substances, in particular for those identified under Article 16.
The emission limit values for substances shall normally apply at the point where the emissions leave the installation, dilution being disregarded when determining them. With regard to indirect releases into water, the effect of a waste-water treatment plant may be taken into account when determining the emission limit values of the installations involved, provided that an equivalent level is guaranteed for protection of the environment as a whole and provided that this does not lead to higher levels of pollution in the environment.
Mixing zones (PS): zones in a surface water body adjacent to discharge point where concentrations of the priority substances may exceed the EQS
Rehabilitation measures: refer to the technologies investigated in Aquarehab, i.e., wetlands, smart carriers, groundwater-surface water interaction zones, multibarriers, and injectable materials
Fate models: numerical models that simulate the evolution of chemical concentrations in groundwater and surface water as a function of time and space, from site scale to basin scale

9 ANNEX 2: RELEVANT EU REGULATION

http://ec.europa.eu/environment/water/index_en.htm
http://ec.europa.eu/food/plant/protection/index_en.htm

Regulation	Relevance to Aquarehab
Nitrates directive: Council Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources	Nitrate, diffuse pollution, wetlands
Pesticide regulation (COUNCIL DIRECTIVE of 15 July 1991 concerning the placing of plant protection products on the market (91/414/EEC) Directive 2009/128/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for Community action to achieve the sustainable use of pesticides	Regulation: pesticide risk assessment according to FOCUS guidelines (leaching to groundwater + runoff to surface water) Sustainable use: diffuse pollution, use of wetlands and buffer strips to mitigate surface water pollution
Water framework directive: Directive 2000/60/EC of the European Parliament and of the Council of 23 October 2000 establishing a framework for Community action in the field of water policy	General framework for reaching good status, time frames, status criteria and definition of water bodies
Groundwater daughter directive: directive 2006/118/EC of the European Parliament and of the Council of 12 December 2006 on the protection of groundwater against pollution and deterioration	groundwater quality standards; pollution trends measures to prevent or limit inputs of pollutants into groundwater, compliance with good chemical status criteria (based on EU standards of nitrates and pesticides and on threshold values established by Member States).
Directive on Priority Substances (Directive 2008/105/EC): good chemical status is reached for a water body when compliance with all environmental quality standards for the priority substances and other pollutants listed in Annex I of this directive is achieved	Priority pollutants, environmental quality standards
REACH regulation: Regulation (Ec) No 1907/2006 Of The European Parliament And Of The Council of 18 December 2006 concerning the Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH)	Specific compounds risk assessment, ecological criteria, fate, chemical properties
Urban waste water directive: Council Directive 91/271/EEC of 21 May 1991 concerning urban waste-water treatment	Point source pollution; discharges in surface water
Drinking water directive: Council Directive 98/83/EC of 3 November 1998 on the quality of water intended for human consumption	Drinking water abstraction from groundwater and surface water, nitrate, pesticides, chlorinated aliphatic hydrocarbons

10 ANNEX 3: AQUAREHAB SUBSTANCES AND EU STANDARDS[3]

Aquarehab Substance	Category	EU Regulation directive	EQS groundwater (µg/L)	EQS surface water (µg/L)
Nitrate		Groundwater direct. Nitrate directive	50	 50 (EU standard) 25 (EU guideline)
Isoproturon	PS	Priority substances Drinking water* Groundwater dir.*	 0.1 0.1	0.3 (AA) 1 (MAC) 0.1
Simazine	PS	Priority substances Drinking water Groundwater dir.	 0.1 0.1	1 (AA) 4 (MAC) 0.1
Terbutylazine		Drinking water Groundwater dir.	0.1 0.1	0.1
MCPA		Drinking water Groundwater dir.	0.1 0.1	0.1
Bentazon	PSR	Priority substances Drinking water Groundwater dir.	 0.1 0.1	 0.1
Glyphosate	PSR	Priority substances Drinking water Groundwater dir.	 0.1 0.1	 0.1
AMPA	PSR	Priority substances Drinking water Groundwater dir.	 0.1 0.1	 0.1
Mecoprop	PSR	Priority substances Drinking water Groundwater dir.	 0.1 0.1	 0.1
Trichloroethylene (TRI)	PS	Priority substances	10**	10 (AA)
Benzene	PS	Priority substances	10**	10 (AA-IW) 8 (AA-OW) 50 (MAC-IW) 50 (MAC-OW)
Toluene				
Nonylphenol	PHZ	Priority substances		0.3 (AA) 2 (MAC)
DEHP	PS	Priority substances		1.3 (AA)

PS: priority substance; PHZ: priority hazardous substance; PSR: priority substance under review; AA: annual average; MAC: maximum allowable concentration; IW: inland waters; OW: other waters;
*EQS for sum of pesticides = 0.5 µg/L
**EQS for Flanders

[3] Member states can derive EQS for other substances. Flanders e.g. derived EQS for groundwater and surface water for much more substances, and for different types of surface waters and seasons (e.g. total N).

11 ANNEX 4: EXAMPLE SUBSTANCE LIST

Aquarehab ID	Aquarehab substance group	WFD priority substances ID	Substance	Registration	Scheldt river*	Odense river	Other EU catchments*****	Ecological relevance	Model key**	Socopse***	Score-PP****	logKoc******	Footprint
1	Nitrate		Nitrate		x	x							
2	Pesticides	1	Alachlor	excluded Annex 1					x			1.63-2.28	moderately mobile
		3	Atrazine	excluded Annex 1 (10/09/2005, some formulations until 31/12/2007)	x				x	x	x	1.95-2.71	moderately mobile
		8	Chlorfenvinphos	excluded Annex 1 (31/12/2003)					x			2.83	slightly mobile
		9	Chlorpyrifos						x		x	3.44-4.49	non-mobile
		13	Diuron	excluded Annex 1 (31/12/2008)	x				x		x	2.67-3.22	slightly mobile
		14	Endosulfan	excluded Annex 1 (31/05/2004)	x						x	4.06	non-mobile
		16	Hexachlorobenzene	excluded Annex 1						x	x	4.70	non-mobile
		17	Hexachlorobutadiene								x		
		18	Hexachlorocyclohexane								x		
		19	Isoproturon	31/12/2012	x	x	x		x	x		1.56-2.38	moderately mobile
		26	Pentachlorobenzene								x		
		27	Pentachlorophenol	excluded Annex 1								1.48	mobile
		29	Simazine	excluded Annex 1 (1/03/2008)	x	x						2.14	moderately mobile
		30	Tributyltin							x	x		
		31	Trichlorobenzenes						x				
		33	Trifluralin	excluded Annex 1							x	3.81-4.13	non-mobile
		9b	DDT	excluded Annex 1					x		x	5.18	non-mobile

Aquarehab ID	Aquarehab substance group	WFD priority substances ID	Substance	Registration	Scheldt river*	Odense river	Other EU catchments*****	Ecological relevance	Model key**	Socopse***	Score-PP****	logKoc*****	Footprint
		9a	Cyclodiene pesticides	excluded Annex 1					x		x	2.60-4.69	non-mobile
		18	Lindane	excluded Annex 1 (20/12/2000)	x							3.04	slightly mobile
			Linuron	31/12/2013	x							2.61-2.67	slightly mobile
			Terbutylazine	excluded Annex 1	x	x	x		x			2.18-2.52	moderately mobile
			Chloridazon	31/12/2010	x							2.3	moderately mobile
			MCPA	18/02/2012	x	x	x					1.87	mobile
			Diazinon	excluded Annex 1 (5/12/2008)	x							2.62-2.88	slightly mobile
		ANNEX III	Bentazon	31/07/2011	x	x	x					1.71	mobile
			Carbendazim	8/12/2011	x							2.30-2.39	moderately mobile
		ANNEX III	Glyphosate	30/06/2012	x	x	x					2.95-4.78	non-mobile
		ANNEX III	AMPA	metabolite	x	x	x					3.90	non-mobile
			Metolachlor	excluded Annex 1	x				x			2.08-2.49	moderately mobile
			2-hydroxy-atrazine	metabolite	x								
			Desethylterbutylazine	metabolite					x			1.64-2.09	mobile
			Terbutryne	excluded Annex 1 (31/12/2003)					x			3.30	slightly mobile
			Hexazinone	excluded Annex 1					x			1.73	mobile
			Promethryn	excluded Annex 1					x			2.60	moderately mobile
			Chlorotoluron	22/05/2011	x				x			2.03-2.58	moderately mobile
			Desisopropyl atrazine	metabolite					x			2.15	moderately mobile
			Fluroxypyr	30/11/2010			x					1.71-1.91	mobile
			Methazachlor	decision postponed			x	x				1.73-2.34	moderately mobile

Aquarehab ID	Aquarehab substance group	WFD priority substances ID	Substance	Registration	Scheldt river*	Odense river	Other EU catchments*****	Ecological relevance	Model key**	Socopse***	Score-PP****	logKoc******	Footprint
			Diflufenican				x	x				3.21-3.87	slightly mobile
			Clopyralid				x					0.70	very mobile
		ANNEX III	Mecoprop	31/05/2014			x					1.30-1.63	mobile
			Metribuzin	9/03/2010			x	x				0.50-1.91	mobile
			Sulfosulforon	30/06/2011			x	x				0.72-1.94	mobile
			Dimethoaat	30/09/2009	x							1.21-1.74	mobile
			2,6-dichlorbenzamide			x							
		ANNEX III	**Dicofol**										
3	Chlorinated Aliphatic Hydrocarbons	10	1,2-dichloroethane										
		11	dichloromethane										
		29a	PER (tetrachloroethylene)										
		29b	TRI (trichloroethylene)								x		
4	Metals (mixed pollution)	6	cadmium and its compounds		x					x	x		
		20	lead and its compounds		x						x		
		21	mercury and its compounds							x	x		
		23	nickel and its compounds								x		
			antimony		x								
			cobalt		x								
			cupper		x								

Aquarehab ID	Aquarehab substance group	WFD priority substances ID	Substance	Registration	Scheldt river*	Odense river	Other EU catchments*****	Ecological relevance	Model key**	Socopse***	Score-PP****	logKoc******	Footprint
			zinc		x								
5	Other (mixed pollution)	4	Benzene		x						x		
			Toluene		x								
		15	Fluoranthene						x				
		24	Nonylphenol		x				x	x	x		
			Dibenz(ah) anthracene						x				
		2	anthracene							x	x		
		5	polybrominated diphenylethers							x	x		
		28	PAHs							x	x (BaP)		
		11	Methylenchloride								x		
		7	C10-13 chloroalkanes								x		
		12	DEHP		x						x		
		ANNEX III	**EDTA**										
		ANNEX III	**Bisphenol A**										
			tetra-ethyl lead								x		

* Flemish reduction programme priority substances, 2005. Flemish monitoring programme pesticides. 2007. Compounds as being identified as most important from a list of about 170 substances

** Identified in Modelkey project impacting ecology (fish, invertebrates, algae)

***Data collected in Socopse project

****Data collected in SCORE-PP project

*****Swedish monitoring programme 2002-2007. Results presented at the Conference of Pesticide behaviour in soils, water and air. York, Sept 2009.

******Footprint database

ANNEX III Substances subject to review for possible identification as priority substances or priority hazardous substances